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Thematic assessment of land degradation and restoration

Chapters of the thematic assessment of land degradation and restoration

Note by the secretariat

1. In paragraph 2 of section IV of decision IPBES-3/1, the Plenary of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services approved the undertaking of a thematic assessment of land degradation and restoration in accordance with the procedures for the preparation of the Platform's deliverables set out in annex I to decision IPBES-3/3, based on the scoping report for the assessment set out in annex VIII to decision IPBES-3/1.

2. In response to the decision, a set of eight chapters (IPBES/6/INF/1) and a summary for policymakers (IPBES/6/3) were produced by an expert group in accordance with the procedures for the preparation of the Platform's deliverables for consideration by the Plenary at its sixth session.

3. In paragraph 1 of section V of decision IPBES-6/1, the Plenary approved the summary for policymakers of the thematic assessment of land degradation and restoration (IPBES/6/15/Add.5) and accepted the individual chapters of the assessment, on the understanding that the chapters would be revised following the sixth session as document IPBES/6/INF/1/Rev.1 to correct factual errors and to ensure consistency with the summary for policymakers as approved. The annex to the present note, which is presented without formal editing, sets out the final set of chapters of the thematic assessment of land degradation and restoration including their executive summaries.

4. A laid-out version of the final thematic assessment report on land degradation and restoration (including a foreword, statements from key partners, acknowledgements, a preface, the summary for policymakers, the revised chapters and annexes setting out a glossary and lists of acronyms, authors, review editors and expert reviewers) will be made available on the website of the Platform prior to the seventh session of the Plenary.

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Annex

Chapters of the thematic assessment report on land degradation and restoration of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services

Disclaimer on maps

The designations employed and the presentation of material on the maps used in this report do not imply the expression of any opinion whatsoever on the part of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystems Services concerning the legal status of any country, territory, city or area or of its authorities, or concerning the delimitation of its frontiers or boundaries. These maps have been prepared for the sole purpose of facilitating the assessment of the broad biogeographical areas represented therein.

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Preface[1]

1. Why is the Land Degradation and Restoration Assessment important, different and new?

Land degradation, as defined by the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES), refers to the many processes that drive the decline of biodiversity, ecosystem functions or ecosystem services. The assessment includes the degradation of all terrestrial ecosystems, along with the aquatic ecosystems within the land mass. While it has often been conceived as mainly a regional concern, land degradation is a problem of global dimensions and affects ecosystems on every continent and small island states: wet and dry; cold and warm; developed and developing. Land degradation has been recognized for over 100 years in Africa (Hubert, 1920), with concerns of Sahel desertification becoming prominent in the 1970s (Le Houerou, 1980). For instance, at its conception, the United Nations Convention to Combat Desertification (UNCCD) focused on countries experiencing serious drought or desertification, particularly in Africa (1996). The other two Rio Conventions, the 1994 United Nations Framework Convention on Climate Change (UNFCCC) and the 1992 Convention on Biological Diversity (CBD), indirectly address land degradation – in this case, from a more global perspective. The United Nations Sustainable Development (Rio+20) Agenda and the Sustainable Development Goals (SDGs), adopted in 2015 (UN, 2015), positioned land degradation as a global issue.

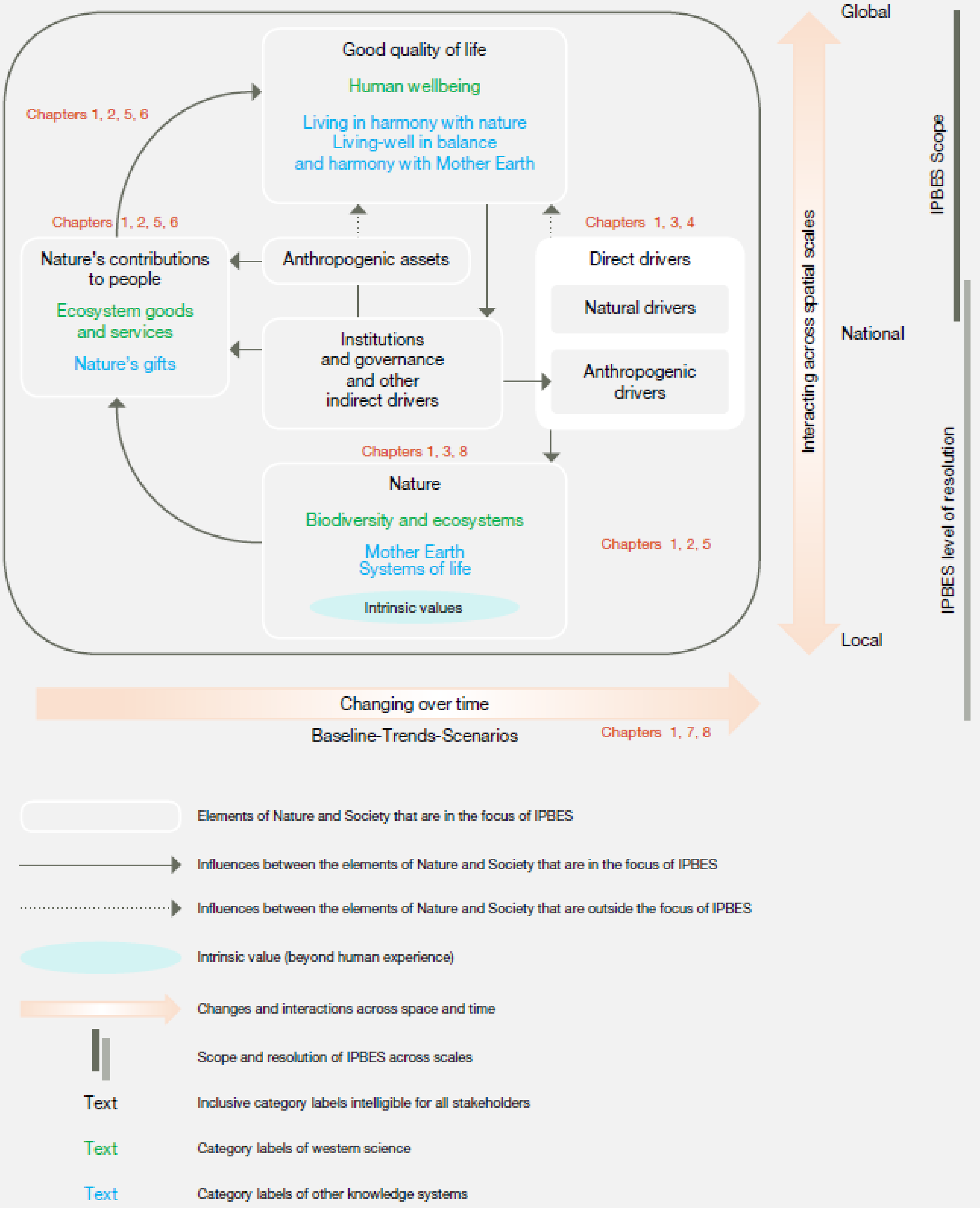
The Land Degradation and Restoration Assessment has been conducted following the IPBES Conceptual Framework (Díaz *et al.*, 2015a), which was adopted by the second session of IPBES Plenary (IPBES-2) in Antalya, Turkey (December 2013). This framework has evolved following evaluation of existing conceptual frameworks and an extensive review process (Figure 1). While past land degradation assessments have focused (often separately) on various aspects of the biophysical resource (e.g., soils, forest, rangelands and so on), this thematic assessment integrates both the biophysical and non-anthropocentric values of the land (including the species it supports) with the contributions the land makes to people (Pascual *et al.*, 2017). It further embraces human-created assets (including ecosystems transformed to serve human needs), institutions and governance structures. The evolving IPBES approach to how nature's contributions to people (NCP) are valued is more inclusive than previous studies (Pascual *et al.*, 2017), but is only partly implemented in this assessment as the concept was developed late in the cycle to be fully incorporated. The assessment draws on new findings and advances in our understanding and considers multiple governance, policy and stakeholder levels.

Up until now, no comprehensive scientific assessment on land degradation has been compiled at a global scale. Responding to the need of a solid scientific basis for implementing the land-related policy priorities identified by the United Nations within the SDG process and the related multilateral environmental agreements (UNFCCC, CBD and UNCCD), the Plenary of IPBES approved, at its third session (IPBES-3) (January 2015), the scoping report for a thematic assessment on land degradation and restoration. This scoping document (Annex VIII to decision IPBES-3/1) presents an agreed-upon outline of the full report and the subject matter to be covered in each of the eight chapters (IPBES, 2015a). This scoping document has been strictly followed in the compilation of this Land Degradation and Restoration Assessment.

[1] Preface was drafted by Coordinating Lead Author of Chapter 1, Judith Fisher (Australia).

Figure 1 **The IPBES Conceptual Framework summarizes the system through which nature and people interact.**

The boxes relate to elements of people and nature, and the thin arrows to the relationships between them. The broad arrows represent recognition that the system has spatial variation, multiple scales, and dynamics over time. The connections of the eight chapters of this thematic assessment to the Conceptual Framework are indicated in red. Source: Díaz *et al.* (2015).



The IPBES Conceptual Framework (Figure 1) summarizes the components of the system comprised of people and nature, and the relationships between them. It provides common terminology for use across IPBES assessments. Integrative but explicit, conceptual frameworks are particularly useful tools in fields requiring interdisciplinary collaboration. They help make sense of complexity by clarifying and focusing thinking about relationships, and supporting communication across disciplines and knowledge systems as well as between knowledge and policy. Nature's contributions to people (NCP) includes all the contributions of nature, both positive and negative, to the quality of life of humans as individuals and societies (Díaz *et al.*, 2015a; IPBES, 2013).

The grey boxes and their connecting grey arrows denote the elements of nature and society that are the focus of IPBES. In each of the boxes, the headlines in black are inclusive categories that should be relevant to all stakeholders involved in IPBES and embrace the categories of science (in green) and comparable or similar categories according to others knowledge systems (in blue). Solid grey arrows denote influence between elements; the dotted grey arrows denote links that are acknowledged as important, but are not the focus of IPBES. Interactions between the elements change over time (horizontal broad orange arrow) and occur at various spatial scales (vertical broad orange arrow). The vertical lines on the right indicate that the scope of IPBES assessments will be at the supranational (from sub-regional to global) scale, but that they will build on properties and relationships often assessed at finer (national and subnational) scales. The line indicating level of resolution does not extend all the way up to the global level because, for the types of relationship explored by IPBES, the spatially heterogeneous nature of biodiversity is important. IPBES assessments will be most useful if they retain finer resolution. This figure is a simplified version of that adopted by the second session of the IPBES Plenary (IPBES, 2013). A more complete description of all elements and linkages in the Conceptual Framework, together with examples, are given in Diaz *et al.* (2015b).

2. Assessment structure

The Land Degradation and Restoration Assessment is comprised of eight chapters with their linkages to the Conceptual Framework shown in Figure 1. **Chapter 1** introduces the topic, establishes the geographic scope, and definitions. It also reviews approaches, and identifies success cases on how to avoid, reduce and reverse land degradation through land restoration and rehabilitation, while benefitting human well-being and quality of life and incorporating nature's contributions to people. **Chapter 2** explores concepts, perceptions and differing worldviews of land degradation and restoration. **Chapter 3** documents the causes of land degradation and factors favouring restoration. **Chapter 4** assesses the current state and trends of land degradation and restoration and associated changes in biodiversity and ecosystem functions. **Chapter 5** explores the changes in benefit flows to people resulting from land degradation and restoration, the consequences for people of incorporating differing values, including changes in ecosystem services and functions, human well-being and good quality of life, and embracing the many worldviews on human-nature relations. **Chapter 6** presents and discusses the actions, which can be taken to prevent or reverse land degradation including restoration. **Chapter 7** provides future projections of land degradation and restoration under several scenarios to better understand and synthesize a broad range of options, and to alert policymakers to future impacts of global changes. **Chapter 8** evaluates tools, competencies and actions to support evidence-based decision-making and policy-relevant guidance to reduce land degradation and promote restoration activities.

3. The IPBES approach

The key aspects of the IPBES approach are its transparent and participatory structure – with explicit consideration of diverse scientific disciplines, stakeholders, knowledge and evidence sources – and its inclusive approach to incorporating differing worldviews, including those of indigenous peoples and local communities (Pascual *et al.*, 2017).

Indigenous and local knowledge (ILK) systems are considered by IPBES to be dynamic bodies of social-ecological knowledge, practices and beliefs about the relationship of living beings, including humans, with one another and with their environment. Indigenous and local knowledge is highly diverse, produced in a collective manner and reproduced at the interface between the diversity of ecosystems and human cultural systems. It is continuously evolving through the interaction of experiences and different types of knowledge (written, oral, tacit, practical and scientific) among indigenous peoples and local communities. IPBES has developed guidance for the integration of indigenous and local knowledge into its assessments that respects not only the diversity and the value of indigenous peoples and local communities but also their rights to share the benefits of knowledge gained from the assessments (IPBES, 2017a). Participation is also dependant on available resources (McCormick, 2014). This assessment integrates indigenous and local knowledge through the involvement of individuals with knowledge and expertise in ILK; through interactions with indigenous knowledge holders, indigenous peoples and local communities, and indigenous organizations; and through engagement of indigenous peoples and local communities in the external review phases of the assessment. Broad questions with specific relevance to each chapter were circulated to indigenous peoples and local community knowledge holders, including established groups recognized by the UN Permanent Forum on Indigenous Issues (UNPFII), ICCA Consortium, Equator Initiative, CBD International Indigenous Forum on Biodiversity (IIFB), UN International Indigenous Forum on Climate Change (UNIFCC), Forest Peoples Programme and other known Indigenous Networks. Specific locations across the 8 chapters for inclusion of indigenous peoples and local communities content were established.

4. Audience and beneficiaries

The intended audiences of this assessment are policy and decision-makers whose work may affect or be affected by land productivity, biodiversity or nature’s contributions to people at all levels (local, national and global), as well as the United Nations entities and multilateral environmental agreements. The assessment is also relevant to the business and finance sectors in achieving positive impact (UNEFI, 2016). Most businesses depend on natural capital in their supply chains. Positive impact finance aims to deliver a positive contribution to the environment and society. Broader intended audiences include the scientific community, indigenous peoples and local communities, indigenous knowledge holders and experts, business and industry, practitioners, intergovernmental and non-governmental organizations, the media, communities of stakeholders and the public at large (IPBES, 2015a).

Another subset of beneficiaries are the people whose health and well-being depends on keeping land in its most productive state (including biodiversity of the land and its ecosystem services); those whose livelihoods depend on reducing degradation; and those who, through sustainable land management, avoid and reduce land degradation. This, arguably, includes every person on Earth, now and in the future, but especially people dependent on livelihoods from currently degraded lands.

5. Process summary

The Land Degradation and Restoration Assessment, unlike other past assessments, arose following a request from governments, several UN conventions and non-government stakeholders. Approval for the development of a scoping document occurred at the second session of the IPBES Plenary (IPBES-2) in Antalya, Turkey (9-14 December 2013). In decision IPBES-3/1 (Work programme for the period 2014–2018), section IV (Thematic assessments), the Plenary approved the undertaking of a thematic assessment of land degradation and restoration, in accordance with the procedures for the preparation of the

Platform's deliverables set out in Annex I to decision IPBES-3/3, based on the scoping report for the assessment set out in Annex VIII to decision IPBES-3/1. The Plenary, at its sixth session, will be invited to approve the summary for policy makers. The IPBES Plenary approved the summary for policymakers, and accepted the chapters of the Assessment Report, at its sixth session (IPBES-6) in March 2018 in Medellín, Colombia.

The IPBES approach includes analysing the latest scientific peer-reviewed literature and published knowledge in the public domain in order to assess the extent, causes and processes of land degradation and the resulting consequences for people and the land. It evaluates responses to restoration and rehabilitation of degraded lands and how future degradation can be avoided and reduced. The inclusion of diverse conceptualization of values as well as the indigenous and local knowledge makes the assessment more comprehensive than assessments conducted previously, such as the Land Degradation and Assessment in Drylands (LADA) (FAO, 2010).

Understanding values, how they are conceptualized and formed and how they change across contexts and scales, is critical to inform decision-making and policy design at local, national and global levels (IPBES, 2016a). The ways in which nature and its contributions to people are perceived and valued may be starkly different and even conflicting (IPBES, 2016a; Pascual *et al.*, 2017). Multiple values can be associated with multiple cultural and institutional contexts and may often be difficult to compare by the same measure. Therefore, IPBES recognizes that the word **value** is not necessarily always a monetary value and can refer to: a given worldview or cultural context; a preference someone has for a particular state of the world; the importance of something for itself or for others; or simply a measure (IPBES, 2016a; Pascual *et al.*, 2017).

An integrative approach to values allows the opportunity to bridge nature's contributions to people while considering different values and perspectives (Pascual *et al.*, 2017). It also allows for recognizing different perceptions of what constitutes a **good life** across social groups and cultures. Furthermore, it highlights the need to acknowledge the role of institutions and social norms that underpin human-nature relations (Pascual *et al.*, 2017).

6. Who conducted the assessment and what were their tasks?

A worldwide call for experts was made in 2015. A total of 86 experts, nominated by IPBES members and organisations, including two Co-Chairs, were selected by the Multidisciplinary Expert Panel (MEP). Selection was based on expertise, knowledge, credentials on specific issues, including a range of scientific, technical and socio-economic views, geographical representation, diversity of knowledge systems and gender balance. Each chapter was guided by Coordinating Lead Authors (CLAs), who shared responsibility to coordinate the writing process and chapter content. The chapters themselves could solicit assistance on specific issues by appointing contributing authors, who do not follow the same process of nomination as the rest of the author team and who are acknowledged in a separate line for their focused contribution on a specific topic. The Multidisciplinary Expert Panel also selected two Review Editors (REs) for each of the eight chapters, whose task it was to oversee the fair and thorough application of the review process. At the beginning of each chapter, all chapter experts (Coordinating Lead Authors, Lead Authors, Fellows, Review Editors, Fellows and Contributing Authors) are listed in an alphabetical order.

7. Confidence levels of key findings

Each key finding in the Executive Summaries of each chapter as well as in the summary for policymakers (SPM), is accompanied by a confidence statement, which refers to the amount of evidence that is available and the degree of agreement by knowledgeable sources (Figure 2). Low confidence describes a situation of incomplete knowledge – when an outcome cannot be fully explained or reliably predicted, whereas high confidence conveys extensive knowledge and the ability to explain an outcome or predict a future outcome with much greater certainty. Low confidence indicates the need for further research (IPBES, 2017b).

Figure 2 The four-box model for the qualitative communication of confidence.

Confidence increases towards the top-right corner as suggested by the increasing strength of shading. Source: IPBES (2016c).

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Chapter 1

Benefits to people from avoiding land degradation and restoring degraded land

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Executive Summary

This report presents the first comprehensive global assessment of land degradation incorporating restoration and rehabilitation responses to avoid, reduce and reverse land degradation (*well established*). The assessment is guided by the IPBES Conceptual Framework, draws on evidence from previous reviews on aspects of land degradation and aims to transform human understandings and behaviour to avoid, reduce and reverse land degradation. The assessment is a structured, evidence-based, multi-authored, expert-reviewed process by which knowledge from diverse scientific disciplines, stakeholder groups, evidence sources, including indigenous and local knowledge systems, differing values and worldviews is evaluated, summarized and presented to guide decisions {1.1}.

It is a challenge to bring together diverse understandings of land degradation as they respond to varied contexts, some of which are more closely related to decision-making (*well established*). The third session of the IPBES Plenary (IPBES, 2015) established definitions and geographic scope for this assessment whereby **degraded land** is defined as a state of land which results from the persistent decline or loss in biodiversity ecosystem functions or services that cannot fully recover unaided within decadal time scales. **Land degradation** refers to the many processes that drive the decline or loss in biodiversity, ecosystem functions or services, and includes the degradation of freshwater and coastal ecosystems that are closely interconnected with terrestrial ecosystems. **Restoration** is defined as any intentional activity that initiates or accelerates the recovery of an ecosystem from a degraded state. **Rehabilitation** is defined as restoration activities that may fall short of fully restoring the biotic community to its pre-degradation state {1.1}. The geographic coverage encompasses all terrestrial regions and biomes of the world, excluding Antarctica, and encompasses the full range of human-altered systems, including but not limited to drylands, agricultural and agroforestry systems, savannahs and forests and associated aquatic systems. Here, land includes all the non-ocean and non-permanently ice-covered regions of the Earth, the freshwater bodies that drain them, and is defined as the terrestrial bio-productive system that comprises soil, vegetation, other biota and the ecological and hydrological processes that operate within the system {1.1}.

Actions that incorporate full and effective participation of indigenous peoples and local communities, including their knowledge in decision-making and in applying traditional systems of land use and resource management, have in many cases demonstrated solutions to avoid and reduce land degradation, recover degraded ecosystems while providing multiple benefits for the well-being of the society (*well established*). The inclusion of indigenous and local knowledge is a distinctive feature of the IPBES assessments. The Land Degradation and Restoration Assessment has incorporated a participatory mechanism and provided opportunities for indigenous knowledge holders, indigenous peoples, indigenous peoples' recognized groups and local communities to contribute to the assessment.

An operational framework, incorporating an integrated socio-ecological landscape approach, has been developed by this chapter. This framework can provide guidance on the interacting criteria most likely to deliver solutions to avoid, reduce and reverse land degradation, incorporating restoration and rehabilitation (*established but incomplete*). It supports policy, governance, economic, financial legal and regulatory decisions at the global to local scales {1.3, Figure 1.2}. This tool interlinks multidimensional processes, aimed at establishing effective socio-ecological governance, incorporating nature's contributions to people, diverse values and the demands of the biophysical environment, considering and incorporating approaches to deal with rapid change and guide co-ordinated solutions.

Rehabilitation of degraded lands has been successfully achieved in many places (*well established*). Successful cases of restoration or rehabilitation of formerly degraded land are presented in this chapter. These cases were selected from different systems, degradation types, parts of the world and with differing socio-ecological interactions {1.4} and the evaluation of their success to stated objective is laid out against the operation framework developed by this chapter {1.3.1}.

1.1 Introduction to the land degradation and restoration assessment

Land degradation is a global issue, costing the world an estimated 10-17% of the global Gross Domestic Product annually (ELD Initiative, 2015). Human well-being costs, associated with land degradation, are not only monetary in nature, but include negative outcomes for health, social cohesion and impacts on local management practices (see also Chapter 5). Food systems operating in the 21st century have developed as major innovations over a significant period; however, the impacts of many of these systems on the degradation of land provide significant threats to people's long term health and prosperity (IPES-Food, 2016)**.** One and a half billion people inhabit and depend on degraded land (UNCCD, 2015b). According to the ELD Initiative, the estimated global economic services loss due to land degradation is up to $10.6 trillion per year (ELD Initiative, 2015). On the basis of the estimates of annual soil erosion by Pimentel *et al.* (1995), a minimal estimate of the economic impact of land degradation is $40 billion annually (FAO, 2010), with large but unknown additional costs for human well-being.

The geographical scope of this assessment encompasses all the terrestrial regions and biomes of the world, excluding only the continent of Antarctica. This encompasses the full range of human-altered systems, including but not limited to drylands, agricultural and agroforestry systems, savannahs and forests and associated aquatic systems. This includes wetland and aquifer systems that are embedded in the land mass, to the landward side of coastal ecosystems and including saline systems. The state of wetlands is inextricably linked to actions in the drier parts of the landscape which drain into them. This scope includes the wetlands as defined within the Ramsar Convention on Wetlands, including areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, and including areas of marine water, the depth of which at low tides does not exceed six meters (Ramsar, 1994).

The definition for **land** for this assessment was that adopted by the UNCCD: **land** means the terrestrial bio-productive system that comprises soil, vegetation, other biota, and the ecological and hydrological processes that operate within the system.

This definition of land matches the IPBES adopted definition of land degradation (see below), which is essentially ecosystem-based and includes the decline or loss of biodiversity, which is considered an integral part of land as a terrestrial ecosystem.

At its third session, the IPBES Plenary (IPBES-3) approved definitions for degraded land, land degradation, restoration and rehabilitation (IPBES, 2015). The expert team was not empowered to change these definitions or adopt other definitions. The process of the assessment revealed both strengths and limitations in the definitions, which are discussed below:

Degraded land is defined as land in a state that results from the persistent decline or loss of biodiversity, ecosystem functions and services that cannot fully recover unaided within decadal time scales.

Land degradation refers to the many processes that drive the decline or loss in biodiversity, ecosystem functions or services, and includes the degradation of all terrestrial ecosystems including associated aquatic ecosystems that are impacted by land degradation.

This is a broader definition than the one adopted by the UNCCD in Article 1 of the Convention text (UNCCD, 1994), whereby land degradation was defined as "reduction or loss, in arid, semi-arid and dry sub-humid areas, of the biological or economic productivity and complexity of rainfed cropland, irrigated cropland, or range, pasture, forest and woodlands resulting from land uses or from a process or combination of processes, including processes arising from human activities and habitation patterns including soil erosion, deterioration in physical, chemical, biological or economic properties of soil and long term loss of vegetation."

The IPBES-adopted definition of land degradation fully includes the narrower definition adopted in 1994 by the UNCCD Convention and is the basis for this Land Degradation and Restoration Assessment. Hence, this assessment is fully compatible with the scope and mandate of the UNCCD and intends to contribute to the actions implemented within that multilateral environmental agreement in reversing land degradation in affected countries.

Note that degradation *sensu* IPBES is restricted to anthropogenic processes. A full discussion of the different perceptions and worldviews related to land degradation is available in Chapter 2. The assessment also recognizes that land degradation, including its drivers and processes, can vary in severity within regions and countries as much as between them.

Restoration is defined as any intentional activity that initiates or accelerates the recovery of an ecosystem from a degraded state. This definition covers all forms and intensities of the degradation state and is in this sense inclusive of the definition adopted by the Society for Ecological Restoration (SER) (Mcdonald *et al.*, 2016).

Rehabilitation is used to refer to restoration activities that may fall short of fully restoring the biotic community to its pre-degradation state, including natural regeneration and emergent ecosystems.

The origin of the degraded land definition adopted by the IPBES plenary can be traced to the desertification synthesis of the Millennium Ecosystem Assessment (MA, 2005), which proposed that degradation could be unambiguously defined as a persistent reduction in ecosystem services. The word **persistent** is intended to exclude short-term fluctuations, such as between summer and winter or from a short run of dry years (MA, 2005). It also implies that the recovery processes are slow, even if the driver of the decline has been alleviated. This idea is consistent with the UNCCD definition of desertification, which is defined as land degradation in arid, semi-arid and sub-humid lands, where degradation is, among other things, a long-term loss of vegetation (UNCCD, 1994). It is important not to confuse areas of inherently low biodiversity, ecological function, or ecosystem service with degraded areas. They may be low in productivity or biodiversity for a range of entirely natural reasons, including among others, because they are climatically too dry or too cold to support much life, have thin soils, or are naturally saline.

Subsequent to the adoption of the land degradation definition by the third session of IPBES Plenary, the fifth session of IPBES Plenary (IPBES, 2017) replaced ecosystem services by nature's contributions to people (NCP). The new terminology includes all the contributions of nature, both positive and negative, to the quality of life of humans as individuals and societies. In this assessment we use both phrases – ecosystem services and nature's contribution to people – since much of the literature we assess uses the older terminology, as does our scoping document and definitions. Where appropriate and where it causes no ambiguity, we use the new terminology of nature's contribution to people.

The various parts of nature's contributions to people are conceptually similar to provisioning, cultural and regulating ecosystem services, but exclude supporting services (which are now considered ecosystem functions) and include natural harms such as floods. The nature's contribution to people terminology also avoids the perceived association of ecosystem services with economic valuation. The evolving IPBES approach to how nature's contributions to people are valued is more inclusive than previous studies (Pascual *et al.*, 2017). Ecosystem services (and nature's contribution to people) are linked to living organisms, but neither are synonymous with biodiversity in its widely-accepted sense of the variety of nature (CBD, 1992). For these reasons, loss of biodiversity and ecosystem functions were both made explicit in the IPBES definition of land degradation. However, doing so can result in ambiguities in quantifying and mapping land degradation or restoration. When ecosystem services, ecosystem functions and biodiversity all decline and fail to recover within ten or more years, it is clear that degradation has occurred. What can be concluded if one or more declines, but the others do not, or perhaps even increase? This situation occurs frequently. For instance, when land cover or land use is changed in order to promote the production of a particular ecosystem service (for example, food from agricultural systems, or timber from plantation forestry), biodiversity almost always declines, and other non-prioritized ecosystem services may also decline (MEA, 2005). However, much human well-being rests on such deliberate and socially-sanctioned conversions and land uses, and it would be perverse to automatically regard them as degradation. On the other hand, conversion to land uses focusing on a restricted set of ecosystem services – and the ongoing management actions used to maximize the yield of those services within the new land use – is a major cause of loss of biodiversity worldwide (MA, 2005; Sala *et al.*, 2000; Wood *et al.*, 2000) and the decline of ecosystem services such as climate regulation and the supply of clean water (Allan *et al.*, 2015; Oliver *et al.*, 2015).

In order to navigate the internal contradictions which, arise from the definition presented to it, this assessment makes a distinction between land transformation and land degradation. **Land transformation** – including the reverse transformation resulting from the abandonment or rewilding of formerly cultivated, settled or domestically grazed lands – has impacts on biodiversity, ecosystem functions and ecosystem services, some of which lead to either an increase or decrease in particular factors. The latter can therefore be considered a form of degradation. Since land transformations are by definition very apparent, they can usually be unambiguously identified and mapped. Therefore, transformation is often expressed in terms of the area affected: for instance, the number of square kilometres deforested, or the percentage of wetlands restored. Implicitly, targets such as the Aichi Target 15 of the Strategic Plan for Biodiversity 2011-2020 (CBD, 2010) and the UNCCD Land Degradation Neutrality Target (Orr *et al.*, 2017) rest on the assumption that such changes can be expressed in area terms.

Within a land use or cover, persistent changes in ecosystem services, function and biodiversity can also occur. These changes are often slower, continuous and thus difficult to detect, but nevertheless constitute land degradation as defined. They may apply over very large areas to varying degrees and cumulatively have large consequences. Defining the affected area also requires a determination of the degree of change (severity) considered to constitute degradation. Therefore, a more meaningful indicator of impact is the integral of severity over the area, and perhaps over time as well (duration), since long-lasting effects are more important than ephemeral effects. Past failures to effectively quantify severity and duration have hampered the ability of this assessment and previous studies to quantify this perhaps most important form of land degradation (i.e., the deterioration of the functioning of composition of an ecosystem without registering a change of area).

The final element in the land degradation definition is how to meaningfully combine a number of simultaneous changes of different magnitudes and even directions, into a single indicator. The ecosystem services literature uses the notion of bundles, which are groups of services that co-vary positively, to help reduce the dimensions which need to be considered (Raudsepp-Hearne *et al.*, 2010), but this approach does not solve the fundamental problem of incommensurability. Relationships exist between restoration and ecosystem services (Aronson *et al.*, 2016). Natural Capital Accounts (Robinson, *et al.*, 2014) show some promise in being able to combine ecosystem service changes of different types, extents, severities and durations into a single framework; in which case, it would be possible to say unambiguously whether the natural asset had on aggregate increased or decreased. To date it has not been possible to satisfactorily include all aspects and values of biodiversity in this framework. Furthermore, some perspectives reject any attempt to do so on the grounds that it may be unethical (Robinson *at al*., 2014).

As a result of the issues raised above, it is currently not possible to operationalize a land degradation definition alike the one provided to this assessment, which includes both ecosystem services and biodiversity. The compromise implemented in this assessment is to treat biodiversity and loss of ecosystem services separately where necessary, and to quantify land transformation separately from land degradation without transformation, within a land use.

Definitions of degradation and restoration also require a measurement of change over time if they are to be detected and quantified. Box 1.1 outlines this discussion briefly (for more detail, see Chapter 2, Section 2.3.1 and Chapter 4, Sections 4.1.3, 4.1.4, 4.4.2 and 4.4.3).

Box 1.1 Targets and baselines

Degradation and restoration are relative terms: “degraded relative to what?” and “restored towards what?” Thus, a reference state is required to detect and assess both the magnitude of degradation and the progress of restoration. Since degradation and restoration refer to change over time, information is needed at two or more times. There is no perfect reference state for all purposes, but allowing free selection of the reference is likely to reduce comparability and increase the risk of deliberate bias. In practice, the nature of a specific data set often dictates the choice of reference state.

The term **baseline** is defined as a reference state in the past up to the present, and is in principle verifiable by observation.

It should not be confused with a **target** which may exist now or, more commonly, is set in the future, and whereby its achievement can only be verified at that time in the future. A target is a political choice, weighing societal, economic and ecological factors, and it can vary case by case and be revised over time.

1. Targets

A target is a desired state. It is typically used for purposes of restoration, though it can be applied to measure degradation as well. The target is perhaps the most important of the reference states for policy purposes, since it represents the future, and thus a state whose achievement can be influenced by policy. It is based on a deliberate, societally-informed choice and is therefore context-dependent. The target may be updated over time, as societal preferences or circumstances change, or as knowledge accumulates, will generally vary from place to place. For example, the aim of restricting global mean temperature rise within 2°C of the pre-industrial mean is a target. An ecosystem target can be considered from the perspective of biodiversity (e.g., protect 17% of the original area of each ecosystem), or it can be considered from the perspective of ecosystem services (e.g., achieve a prescribed sustained flow of clean water). Targets can range from being pragmatic – based on modest investments and readily available technology (such as to slow the rate of species loss) – to aspirational, an ideal outcome with little practical chance of being reached. In the former case, outcome-based metrics are usually set, whereas in the latter case effort-based metrics are more relevant.

2. Baselines

There are two qualitatively different types of baselines which have been used for the measurement of human-caused ecosystem degradation and restoration. The first refers to the distant past, a “natural” state before human modification. The second is a "historical" state that refers to much more contemporary states, for which we have increasingly precise data.

2.1 Natural baselines

Establishing a natural reference state for an ecosystem is challenging, since most ecosystems have been influenced to some degree by humans for a very long time. Two approaches have been used:

2.1.1 Pre-modern natural baseline

This can be thought of as the ecosystem condition within the Holocene, but before the Anthropocene – in other words, sometime between 10 000 and 100 years ago. This seems to be an obvious baseline from which to assess degradation and recovery since it is before the onset of the profound modifications brought about by the rapid increases in the human population, consumption and waste production in the modern era, at which point a distinct discontinuity appears in the degree and type of disturbance. The pre-modern natural baseline has the advantage of not being easily manipulated. Several examples show it to be implementable in appropriately-selected cases, though not without challenges. Practically, it is rare to find data from so far in the past that includes all the variables needed to compare with current ecosystem condition. Proxies are commonly used, such as paleo-ecological data, which is sparse, expensive to collect and requires great expertise to interpret. Another strategy is "space-for-time" substitution, where a currently existing ecosystem in another place (for instance, a protected area) is taken to represent the pre-modern past of the human-altered ecosystem under consideration. But the climate and other biophysical environmental conditions may have changed in the intervening time, or may be subtly different at the reference location, and it is difficult to disentangle the effect of anthropogenic degradation from natural environmental change. In some cases, the ecosystem structure, composition and function which we desire to retain or achieve is inextricably a product of human actions, and in these cases, considering the ecosystem without human influence makes no sense.

***2.1.2 Counterfactual natural baseline*.**

Perhaps a more operational approach for establishing a natural state baseline is to use the current time, but apply counterfactual thinking, which can be characterized by the phrase "what might have been in the absence of human influences". Counterfactual natural baselines avoid some of the challenges of pre-modern observation-based baselines, but they require a high level of expertise, sometimes using explicit process knowledge that constitutes a "model" of what would have happened in the absence of human effects. Some implementable examples exist: for instance, enough is known about the ecosystem dynamics of carbon to be able to state with good confidence what the soil carbon content at a site would have been under a natural cover.

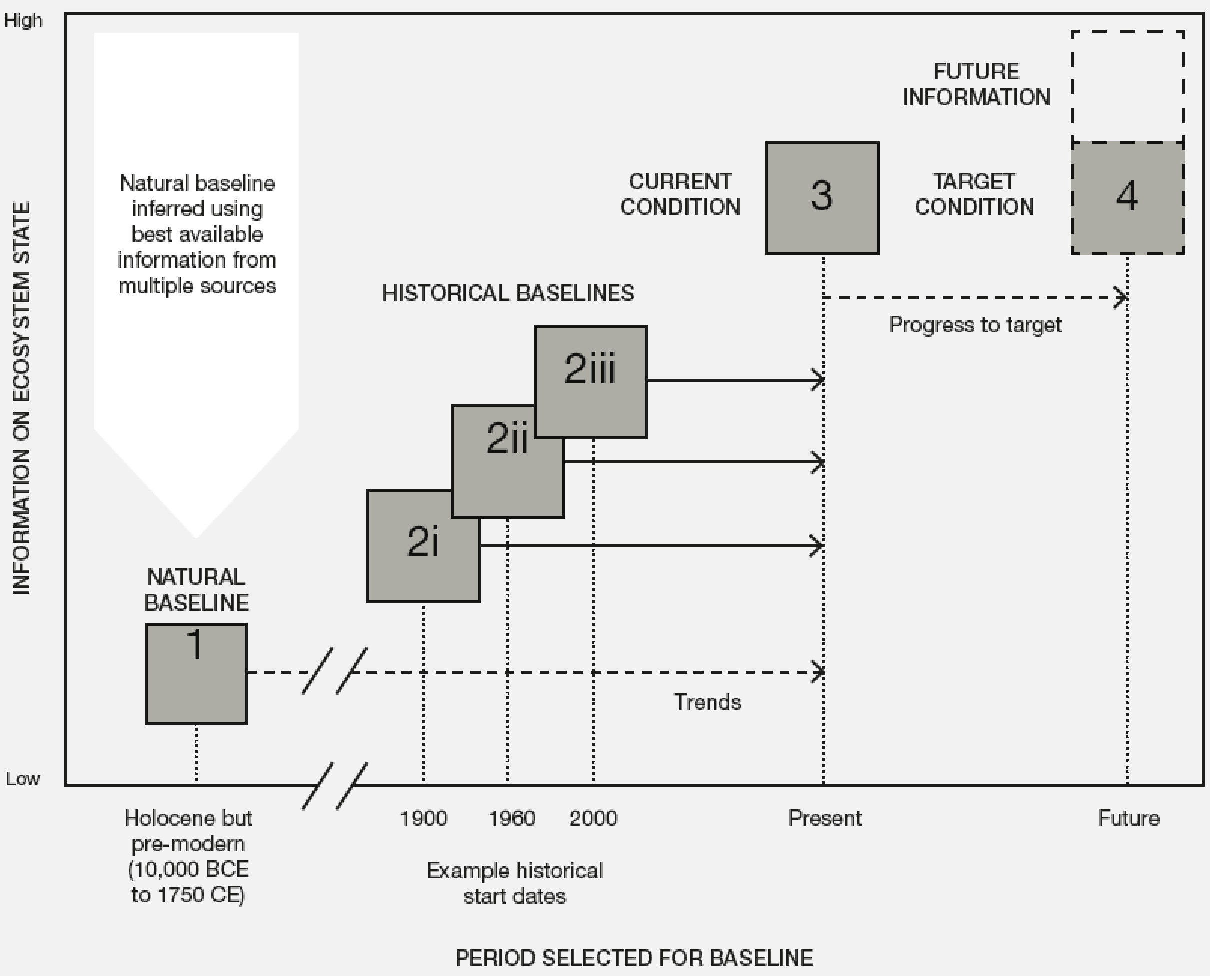
2.2 Historical baselines

Historical baselines use direct observations of the ecosystem state, and therefore seldom extend before about 1950; but they include our most reliable datasets, such as long-term datasets and ecological experiments, and are therefore an invaluable resource. Quantitative trend analysis sets no explicit baseline, but unavoidably uses the start of the record. Unlike natural baselines, it is accepted that historical baselines may have undergone some human-induced change prior to their establishment, and therefore provide underestimates of the totality of degradation or restoration. Particularly in the case of non-linear change (for instance, degradation which levels off at a limit), a recent historical baseline underestimates the total degradation, relative to those where it occurred before the baseline was established. The closer to the present baselines are established, the more data are available, but the less they represent the totality of degradation. The advantage of earlier references is that they allow better detection of slow changes, particularly against noisy short-term variation. Various historical baselines have been used in the land degradation and restoration domain. Their differing and sometimes arbitrary starting dates make comparisons difficult and are open to self-serving manipulation. When we are interested in the impacts of policy or management changes, a recent baseline can be used - for instance the date at which an agreement came into force.

For further discussion regarding baselines and targets, and citations of the underlying literature, see Chapter 2, Section 2.2.1.1, Figure 2.4 and Figure 2.5 and Chapter 4, Sections 4.1.3, 4.1.4, 4.4.2 and 4.4.3.

Fig. 1 1 **Schematic diagram of various types of baselines (reference conditions) which can be used to identify degradation and restoration, and as a starting condition from which to measure trends.**

1. Pre-modern Natural baseline - the information is inferred from the current state, historical data, paleo-ecological proxies and expert opinion. Since the actual date of this state is rarely known, the derived trend is indicated by a dashed line;
2. i, ii, iii. Historical baselines - data gathered in the recorded past (e.g. 1900, 1960, 2000);
3. Current state - used to measure past trends and to provide a reference for future monitoring;
4. Target – the state chosen as an objective for restoration.



1.2 When is the avoidance or reversal of land degradation successful?

1.2.1 An operational framework

The scope of this chapter is to provide examples of success cases which demonstrate the benefits to human well-being and quality of life achieved by avoiding, reducing and reversing land degradation through restoration and rehabilitation. The objective in highlighting cases is to show how land management and restoration measures can help improve livelihoods, reduce poverty and strengthen long-term sustainability of land use in different situations.

To determine the approach to the selection of cases, scientific and other literature was systematically assessed (see Section 1.2.1.1). More specifically, this literature search was done to identify, summarize and evaluate key recurring factors and criteria which are most likely to contribute to such success and to assist in determining the success cases to be highlighted in Chapter 1. The outcome of this systematic review lends itself to the development of an operational framework (Figures 1.2 and 1.3), which incorporates the landscape socio-ecological approach. This framework was subsequently used to guide the choice of cases and the quantitative assessment of their success (see Sections 1.3.1 and 1.4). The Operational Framework may also assist with project development, implementation and assessment.

1.2.1.1 Methodology to identify key criteria

A systematic seven-step methodology was developed to identify the key criteria most likely to deliver outcomes which will benefit human well-being and quality of life through the avoidance, reduction and reversal of land degradation, incorporating successful restoration and rehabilitation of degraded lands. This seven-step approach integrated the main elements of the IPBES Conceptual Framework (i.e., nature, anthropogenic assets, nature's contributions to people, drivers of change and good quality of life) (Figure 1 in Preface based on Díaz *et al.* (2015)), the IPBES approach to the valuation of nature's contributions to people (Pascual *et al.*, 2017), and the evolving IPBES approach to the inclusion of indigenous and local knowledge. The approach drew on information and insights from all other chapters. This seven-step methodology is described below:

Step 1: Search terms were established using the main elements of the IPBES Conceptual Framework and the valuation of nature's contributions to people, incorporating causes and consequences of land degradation. Search terms elements were also drawn from the Sustainable Development Goals, the Aichi Biodiversity Targets, and the UNCCD Convention. The authors incorporated differing knowledge systems and worldviews (including indigenous and local knowledge), the elements of quality of life and human well-being, the quality of life of individuals, communities, societies, nations and humanity, and successful solutions including restoration and rehabilitation (Chapter 1). Key elements from other chapters were reviewed and incorporated, including different perceptions (Chapter 2), direct and indirect drivers (Chapter 3), status and trends of biodiversity and ecosystem services (Chapter 4), scale and trade-offs (Chapters 4 and 5), changes in ecosystem functions, human well-being and good quality of life (Chapter 5), responses to land degradation and restoration (Chapter 6), trade-offs between social, economic and environmental objectives (chapters 4, 5, 6 and 7) and decision-support approaches (chapter 8).

Step 2: Using the aforementioned terms, a systematic literature search was conducted, incorporating the cycle of events from causes through to solutions, drawing on relevant articles, books, regional and national assessments, reports by governments, United Nations bodies, national and international non-government organisations and indigenous peoples and local community knowledge sources. A total of 260 references were accessed during this search.

Step 3: The content of the 260 references were subjected to a systematic review process to identify key recurring and common terms associated with the causes of land degradation, its impacts on human well-being and quality of life, restoration, rehabilitation, successful outcomes and solutions. This review of literature revealed 106 key terms.

Step 4: The 106 key terms were grouped by similarity, reflecting on the initial search criteria. This resulted in 15 key headings, based on the frequency in which the term occurred. The information from the literature search was gathered into a table listing the pertinent references and divided by: (i) perspective; (ii) initial search criteria; (iii) the key term to which it is related; (iv) implementation outcomes; and (v) other factors.

Step 5: The information in the summary table (Step 4) was further analysed to reveal three overarching and overlapping criteria. The three overarching **criteria** emerging from this systematic iterative process were: (1) guiding instruments: (2) nature’s contributions to people; and (3) biophysical conditions. In addition, three overarching **principles** emerged. These were: (1) communication; (2) coordination; and (3) participatory processes.

Step 6: All information in steps 1 through 5 was grouped within each of the relevant three key overarching criteria. This resulted in a number of sub-categories within each criterion, including those which overlapped with the three criteria, demonstrating the importance of interconnections between criteria for successful outcomes. An internal review of the initial outcomes occurred across all chapters in the assessment. Inputs from two external reviews enhanced the outcomes presented in Chapter 1 (Figures 1.2 and 1.3).

Step 7: Figure 1.2 represents the outcomes of the iterative systematic review process, summarising an operating approach which may guide actions. Section 1.3 expands on Figure 1.2 and provides information on the subcategory elements, their interlinkages and interconnections and their usefulness in potentially identifying and achieving future successful outcomes. A further literature search based on the developed Figure 1.2 was conducted. Additional 250 references supporting the outcomes of the systematic review process (total of 510 references) have been utilised to substantiate the information presented in Figure 1.2 and 1.3.

This systematic review process is summarised into an operational framework (Figure 1.2 and Section 1.3) which may guide coordinated approaches to achieve successful outcomes (Chapter 1) to avoid, reduce and reverse land degradation (Chapter 6) while benefiting human well-being and quality of life (Chapters 1, 2, 5), incorporating different perceptions and worldviews (Chapter 2) and understandings of the biophysical environment (Chapters 3, 4), including decision processes and tools (Chapters 7, 8). This review has demonstrated the importance of including information and insights from all chapters of the assessment, the IPBES Conceptual Framework and approach to values and nature’s contributions to people, to identify an approach which may guide actions to achieve and measure the success of outcomes. The evaluation methodology (Figure 1.2 and Section 1.3.1), provides a quantitative approach to identify which criteria, and their sub-elements, have been achieved successfully and the elements for which improvements can be made.

1.2.1.2 Key aspects of the operational framework

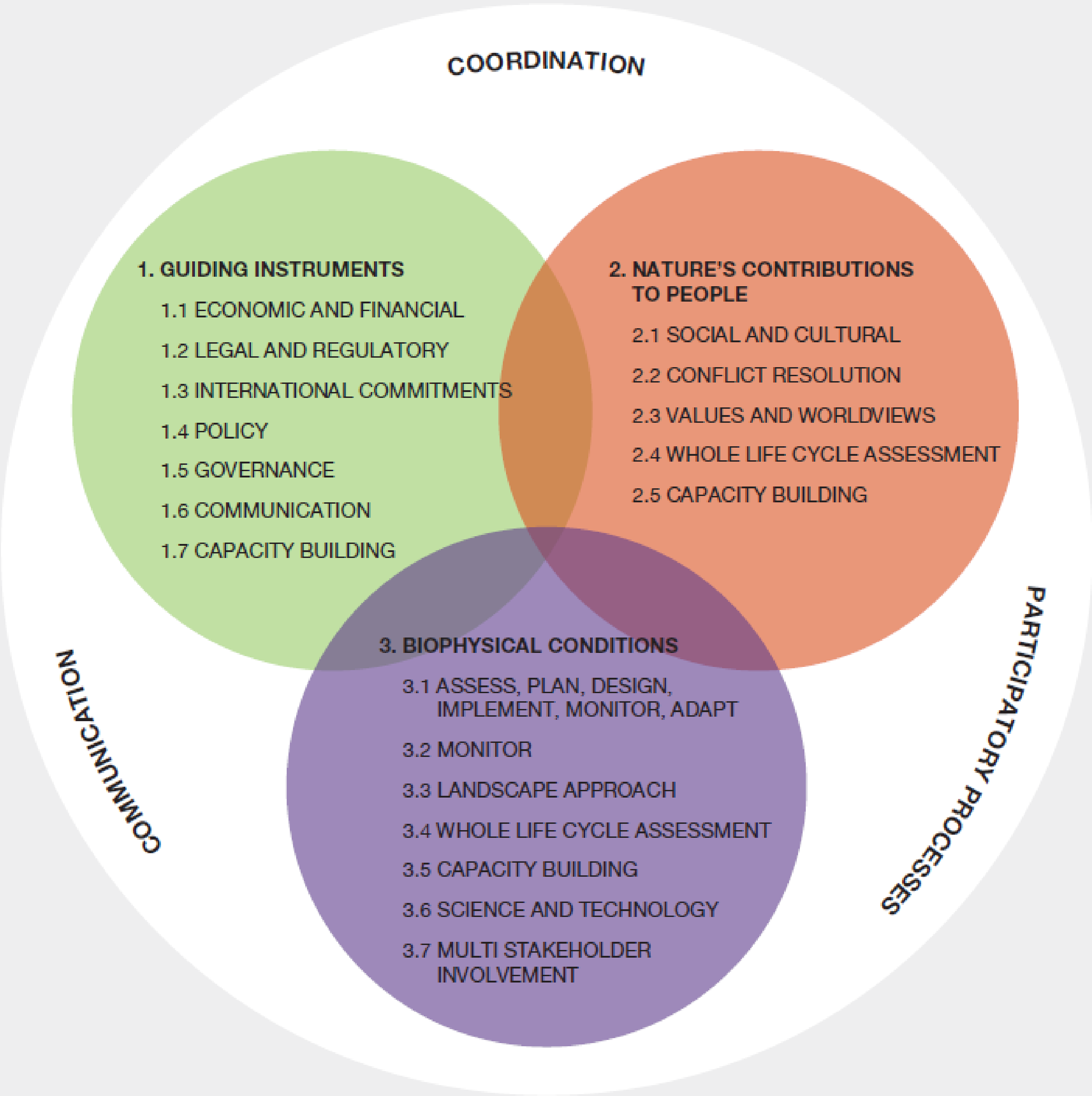
Key aspects of the operational framework are the socio-cultural relations between people and nature (Figure 1.2). This cultural context influences the perceptions and experiences of actions and what counts as success. Effective outcomes occur when actions are co-produced with people and nature and include the application of their knowledge and work. This guidance draws on insights from the seven subsequent chapters of the assessment, underpinned by a firm evidence base (Estrada-Carmona *et al.*, 2014). When all factors are implemented in a coordinated, interacting manner and communicated to all levels of society, outcomes are most likely to lead to positive solutions to avoid, reduce and reverse land degradation, benefitting human well-being, quality of life and nature (see Section 1.3).

Figure 1.2 provides direction for the selection of the eight success cases presented (see Section 1.4). To assess the outcomes of the success stories, our approach considers aspirations to benefit human well-being and quality of life while avoiding, reducing and reversing degradation processes utilising the restoration and/or rehabilitation of degraded land. The three key interacting criteria and associated elements have been used to frame, along with a quantitative evaluation (see Section 1.3.1), the outcomes of the success cases.

The three interacting criteria (i.e., guiding Instruments, nature's contributions to people and biophysical conditions) depend on active, multi-stakeholder involvement to ensure outcomes that: (i) incorporate human well-being, differing values and good quality of life; (ii) are technically and legally feasible, while being environmentally and socially acceptable; (iii) incorporate knowledge and capacity-building, establishing an enabling environment which is well understood, communicated and supported by all stakeholders; and (iv) incorporate economic and financial mechanisms compatible with all three interacting criteria (Figure 1.2). The operational framework utilizes the ecosystem approach at a landscape scale – that is, a socio-ecological ecosystem, delivering multiple functions, including multiple stakeholders with differing values. The **landscape-scale approach** incorporates the socio-ecological system, including natural and human-modified ecosystems, influenced by ecological, historical, economic, and socio-cultural processes. The landscape includes an array of stakeholders small enough to be manageable but large enough to deliver multiple functions for stakeholders with differing interests (Denier *et al.*, 2015; Scherr *et al.*, 2017).

Figure 1 2 Operational framework for guiding decisions and actions to establish and identify success in avoiding, reducing and reversing land degradation for the benefits of human well-being and good quality of life, while restoring and rehabilitating degraded land.

This approach is underpinned by coordination, communication and participatory processes; and the main IPBES elements (**Figure 1**) of nature, anthropogenic assets, nature's contributions to people, drivers of change, good quality of life and diverse values.



1.3 Understanding the operational framework

Coordination, communication and participatory processes are key influences of the three overarching criteria. They are underpinned by participatory planning and different knowledge systems (Brancalion, 2015; Guilfoyle, 2004; Hill *et al.*, 2013; Laestadius *et al.*, 2015). Together they may create evidence-based, enabling instruments and environments to avoid land degradation and deliver effective restoration and rehabilitation (ELD Initiative, 2015; Joly *et al.*, 2010).

Evaluating success

Several elements support each of the three overarching criteria (Table 1.1). None of these elements, across and within the three criteria are sufficient individually to establish or identify success. Positive solutions rarely, if ever, operate in isolation from all other factors. Our literature review (see Section 1.2.1.1) has demonstrated that interactions, alignments, implementation and measurements across the three criteria can be critical for success. A quantitative method is presented which can evaluate effectiveness of individual success stories (Table 1.1, Box 1.2), and may also provide an approach to measure effectiveness of new projects into the future. The scoring is conducted against and within each of the three criteria (Table 1.1, Figure 1.2), using scoring values as outlined in Box 1.2. All factors (Table 1.1) are given a scoring value between -1 to +5 (Box 1.2). These quantitative measurements can be used prior to restoration and rehabilitation actions, during implementation, at the end of implementation and can also assist project adaptation.

Table 1 1 Factors linked to the 3 overarching criteria of the operational framework (Figure 1.2) to score against to evaluate success, using scoring values -1 to 5* (Box 1.2).

* Scoring values

1. GUIDING INSTRUMENTS	2. NATURE'S CONTRIBUTIONS TO PEOPLE	3. BIOPHYSICAL CONDITIONS
1.1 Economic and finance	**2.1 Social and cultural instruments**	**3.1 Assess, Plan, Design, Implement, Monitor, Adapt** Land degradation state
1.2 Legal and regulatory 1.2.1 Formal recognition property rights, land tenure	**2.2 Conflict resolution** 2.2.1 Food and biodiversity 2.2.2 Livelihoods 2.2.3 International/national interests	**3.2 Monitor**
1.3 International Commitments	**2.3 Values and worldviews** 2.3.1 Non-monetary valuation 2.3.2 Human well-being, quality of life 2.3.3 Indigenous people & local communities	**3.3 Landscape approach** 3.3.1 Biodiversity, food, water, soils, carbon, climate
1.4 Policy Instruments 1.4.1 Formal recognition Property rights, land tenure	**2.4 Whole of life cycle assessment**	**3.4 Whole of life cycle assessment**
1.5 Governance 1.5.1 Active multiple stakeholder engagement	**2.5 Capacity-building**	**3.5 Capacity-building**
1.6 Communication		**3.6 Science and technology**
1.7 Capacity-building		**3.7 Stakeholder involvement**

Box 1 2 **Methodology to evaluate success of solution-based projects designed to improve human well-being and quality of life by avoiding and reducing land degradation and restoring and rehabilitating degraded lands.**

- -1 Negative
- 1 Limited
- 2 Slight
- 3 Slight to moderate
- 4 Moderate
- 5 Good

1. Guiding instruments	11 factors score each factor (-1 to + 5) max value 55 = **total 1**
2. Nature's contributions to people	11 factors score each factor (-1 to + 5) max value 55 = **total 2**
3. Biophysical Conditions	9 factors score each factor (-1 to + 5) max value 45 = **total 3**

Success value % = (total 1 + total 2+ total 3) / (55+55+45) * 100

* Scoring values

1.3.1 Guiding instruments

The guiding instruments (Figure 1.2, Points 1.1-1.7) are the core instruments which, if effectively developed, integrated and aligned, can provide opportunities for a positive impact for people and the land. Good governance structures (1.5) incorporating differing values, worldviews and indigenous and local knowledge can stimulate successful strategies which may reduce negative impacts of conflicting interests. Communication and capacity-building potentially can align all players.

1.3.1.1 Effective and implemented economic and financial instruments (Figure 1.2, point 1.1)

Successful restoration is underpinned by a strong business case, which incorporates ecological, social and economic benefits (FAO, 2015; IUCN & WRI, 2014). Successful restoration also needs to be supported by a decision-making framework aiming for net social and economic benefits, and implemented within strong legal, governance and institutional contexts (Laestadius *et al.*, 2015; Wortley *et al.*, 2013). The correct mix of policy incentives, excluding perverse incentives, can lead to the establishment of new incentives to lower or remove economic barriers (Global Landscapes Forum, 2015b), and encourage the adoption of more sustainable management practices (ELD Initiative, 2015). Subsidies which stimulate low profit agriculture, and negative landscape impacts, such as the European Union's Less Favoured Areas subsidies, predicates a support scheme (Salvati & Carlucci, 2014) with perverse incentives, hence this subsidy is being reviewed by the European Union. Policies and schemes for the payment of ecosystem services, which provide incentives for investment in land improvement and reward sustainable land use, have been employed as economic instruments in some parts of the world (Nkonya *et al.*, 2016). Successful application is relative to the country and its legislation. However, a singular focus on economic value, such as the payment of ecosystem services, provides limited opportunity to incorporate a pluralistic approach which embraces a diversity of non-monetary values, and limits opportunities for transformative integrated practices (Pascual *et al.*, 2017). Economic incentives for one ecosystem function or service can lead to unbalanced outcomes and negative impacts on communities, including indigenous peoples and local communities – particularly women, who disproportionally depend on non-monetary values. Private markets often fail to assign a price to many ecosystem services that adequately reflects their benefits to society as a whole (Kroeger & Casey, 2007). The Kisoro District in Uganda provides an example where fragmented landscapes and lack of collaboration, between upstream and downstream communities in the Chuho springs watershed, has resulted in upstream land degradation due to intensive agricultural practices and a lowered water supply to downstream users. The potential for a payment for ecosystem services scheme to benefit both communities was found to be very limited (Sengalama & Quillérou, 2016).

Effective examples incorporating financial instruments

Landscape partnerships, including businesses, have the potential to be effective for reducing land degradation, while benefitting and contributing to local communities, businesses, landscapes, food and nature. The Business for Sustainable Landscapes project, created by the Landscapes for People, Food and Nature Initiative, (partnered by EcoAgriculture, IUCN's SUSTAIN-Africa Programme, SAI Platform and the Sustainable Food Lab) catalysed input from 40 companies and organizations, to advance landscape partnerships - resulting in an Action Agenda to strengthen business participation and contributions. The Action Agenda aims to improve the quality of business engagement and scale up landscape partnerships for sustainable development including food, nature, business, local communities and landscapes (Scherr *et al.*, 2017).

Australia's Indigenous Land Corporation's National Indigenous Land Strategy is linked to Australia's Indigenous Economic Development Strategy and enables the Indigenous Land Corporation to meet their legislated function to assist indigenous people to acquire and manage land to achieve economic, environmental, social or cultural benefits (Indigenous Land Corporation & Australian Government, 2012; Indigenous Land Corporation, 2013).

A local Kenyan organization, Kijabe Environment Volunteers in the Kikuyu escarpment landscape has mobilized communities across their landscapes. These landscapes are rich in wild biodiversity, have strong cultural heritage and important areas of agricultural production. A landscape perspective was adopted to sustainably manage natural resources and balance the multiple functions of the landscape, enabling local communities to define and pursue their goals related to agricultural development and profitability while conserving the area's critical natural capital (Buck *et al.*, 2014).

Countries experiencing salt-induced land degradation have recognised the cost-effectiveness of investing in land remediation, incorporated into a broader strategy for food security. Including remediation in national action plans can identify and remove barriers to the adoption of sustainable land management, including perverse subsidies (Qadir *et al.*, 2014).

1.3.1.2 Effective and implemented legal and regulatory instruments (Figure 1.2, point 1.2)

Legal and regulatory instruments that guide countries' and states' policies for land restoration and rehabilitation, including extraction of natural resources, establish legal and regulatory frameworks to improve restoration outcomes and success. Such legal instruments are only as good as their implementation, particularly in controlling compliance and implementing potential prosecutions. Latin American countries have developed regulatory frameworks and supportive instruments aimed at guiding restoration. However, exclusion of stakeholder groups, limited institutional and organizational capacity to operationalize large-scale restoration and particularities of the high socio-ecological heterogeneity in legal and regulatory instruments have limited their effectiveness (Meli *et al.*, 2017). For example, the Secretariat for the Environment of the State of São Paulo, Brazil, drives planning and assesses achievement of legally-established goals and compulsory restoration targets. These are however only biophysical and exclude impacts on people, particularly indigenous peoples and local communities (Chaves *et al.*, 2015).

The Western Australian State legal and regulatory instruments (Western Australian Department of Mines Industry and Regulation, 1978), linked to Australian government legislation (Commonwealth of Australia, 2016), direct the formulation of policy and guidance statements around the extraction of natural resources, including rehabilitation and restoration completion criteria, definitions, measurement of success and timeframes, and are auditable (EPA, 2006). South Africa requires mining companies to rehabilitate land after open cast mining, which is costly. Estimating the farming revenue of land prior to and after open-cast mining can establish what the value of land use will be after mining, and can shift scenarios toward a win-win situation for all land users (McNeill & Quillerou, 2016).

Legal policies based on environmental compensation, without restoration recovery conditions, have failed in mangrove recovery projects in Mexico (Zaldivar-Jimenez *et al.*, 2010). To compensate for wetland losses through the implementation of the Clean Water Act in the United States, performance standards for wetland creation and restoration have been established (National Research Council 2001a, 2001b).

Formal recognition of property rights and land tenure (Figure 1.2, points 1.2 and 2.2)

Land tenure is the legal status and ownership of land, often with a mixture of formal and informal tenure systems and a mosaic of property rights, individual and collective. Effective rule of law – including property rights allocation and women's land tenure rights (Silverman, 2015; Plurality in Public Policy, 2014) – provides certainty, reduces conflict and land degradation. Case studies from 10 countries (Chile, Ethiopia, Iran, Panama, Paraguay, Russia, Samoa, Solomon Islands, South Africa and Uganda) established that legislation recognizing community land, conserved areas and traditional knowledge further enhanced project success (Global Forest Coalition, 2015).

Solid evidence exists that strong customary tenure and clear, uncontested land rights have a positive impact on good stewardship of landscapes and are critical to the success of large projects such as REDD+, community forest programs and integrated landscape management. Strong correlations exist between weak, poorly defined rights and insecure tenure, deforestation and landscape-level degradation (Global Landscapes Forum, 2015b). A lack of formal registration of customary property rights may not benefit the local and poorer populations, potentially causing unrest and marginalization of local communities (ELD Initiative, 2015). Difficulties occur where modernization has diluted such "law", and in colonial disputed lands where differing views exist on land tenure regimes (see Case Study 8).

Restoration and rehabilitation of degraded land can benefit by working with the knowledge of indigenous and local knowledge holders to aid restoration approaches, who have been on the land for generations, and have relevant intergenerational observational knowledge, as articulated in the Indigenous and Tribal Peoples Convention, 1989 (No 169) (ILO, 1991).

Indigenous law has key connections to sustainable land management. Adult traditional owners of the Girringun in northern Australia (and other indigenous traditional owners across the country) hold formal legal, cultural and spiritual obligations to care for ancestral lands and waters – based on a worldview and customary planning system with spiritual, social and physical connections between land and people, in addition to their responsibilities under customary law (Guilfoyle & Mitchell, 2015). Negative changes in ecosystem components, directly affect the mental health and spiritual well-being of these indigenous communities, including the quality of food and plant resources (Fisher, 2013; Robinson *et al.*, 2016).

1.3.1.3 Implementation of international commitments (Figure 1.2, point 1.3)

International commitments and targets can only be effectively implemented if there is local action and support. The following commitments all have provisions relevant to land degradation and restoration with obligations entered into by signatory countries: Sustainable Development Goals 2, 13 and 15; the land degradation neutrality (LDN) of the United Nations Convention to Combat Desertification (UNCCD); the United Nations Framework Convention on Climate Change (UNFCCC); The Ramsar Convention through the 4th Strategic Plan 2016-2024 (Ramsar, 2015) and the Convention on Biological Diversity (CBD) Aichi Target 15 of the Strategic Plan for Biodiversity 2011-2020 (Paustian *et al.*, 2016, Montanarella & Lobos, 2015). Land and soils are considered across the three Rio Conventions (UNFCCC, CBD and UNCCD), and while some advances have been made in the past two decades, land and soil degradation persist. This calls for a more integrated approach for the implementation across the Conventions. Opportunities exist to strengthen linkages between the Rio Conventions (UNFCCC, CBD and UNCCD) and the Sustainable Development Goals (SDGs), utilizing soil-based greenhouse gas mitigation policies (Paustian *et al.*, 2016), consolidating associations with the UNFCCC and the 171 countries who have become signatories to the

Box 1.3 Sendai Framework complementarities to the operational framework of this chapter

Elements of the Sendai Framework for Disaster Risk Reduction (2015-2030) which are complementary to the ecosystem and landscape approach proposed within the operational framework include:

> 28 (d) To promote transboundary cooperation to enable policy and planning for the implementation of ecosystem-based approaches with regard to shared resources, such as within river basins and along coastlines, to build resilience and reduce disaster risk, including epidemic and displacement risk;
>
> 30 (f) To promote the mainstreaming of disaster risk assessments into land-use policy development and implementation, including urban planning, land degradation assessments … the use of guidelines and follow-up tools informed by anticipated demographic and environmental changes;
>
> 30 (g) To promote the mainstreaming of disaster risk assessment, mapping and management into rural development planning and management of, *inter alia*, mountains, rivers, coastal flood plain areas, drylands, wetlands and all other areas prone to droughts and flooding,… and at the same time preserving ecosystem functions that help to reduce risks (UNISDR 2015);
>
> 30 (n) To strengthen the sustainable use and management of ecosystems and implement integrated environmental and natural resource management approaches that incorporate disaster risk reduction.

Paris Agreement (April, 2016). Similarly, soils and land play a key role to achieve the post-2015 development agenda and can be found across the Sustainable Development Goals (Montanarella & Lobos, 2015).

SDG 15 (Targets 15.1-15.9) is relevant to this assessment and pertinent to the operational framework of success. Coordination and incorporation of all elements as outlined in the operational framework (Figure 1.2) may assist governments in choosing an appropriate suite of strategies to reach net positive impacts and the mitigation hierarchy (BBOP & UNEP 2010), the Bonn Challenge (Chazdon *et al.*, 2015) and the Latin American Initiative of 20x20 – a country-led initiative to restore 20 million hectares of degraded land in Latin America and the Caribbean by 2020, which is guided by the World Resources Institute and strongly influenced by the political agenda (Vergara *et al.*, 2016). The Sendai Framework for Disaster Risk

Reduction 2015-2030, adopted in 2015, is relevant to this assessment as it recognizes the benefits in reducing risk to the degradation of ecosystem services, and prioritizes a number of related actions (including at a landscape-scale) on ecosystem-based approaches to disaster risk reduction. A number of elements within the Disaster Risk Reduction Framework are well aligned with and complement the approaches of this chapter's operational framework (Figure 1.2) (Box 1.3).

1.3.1.4 Enabling policy instruments (Figure 1.2, point 1.4)

Enabling circumstances include coordination and communication across all **success factors** and provide strategic and coordinated efforts to strengthen them. Implementation of the following enabling instruments provide opportunities to achieve successful land degradation and restoration outcomes.

Successful policy instruments prioritize incentives and practices which increase restoration outcomes: removing disincentives; incorporating secure land and natural resource tenure; aligning with policies to avoid land degradation; and encouraging effective institutional coordination while incorporating good governance (ELD Initiative, 2015; Laestadius *et al.*, 2015). They also incorporate ecosystem services, economic, social and ecological benefits, enhance livelihoods and address political, cultural and economic concerns (Chazdon *et al.*, 2015). When integrated with national policy and international commitments, their effectiveness increases (Natural Resource Management Ministerial Council Government of Australia, 2010; COAG Standing Council on Environment and Water, 2012).

Formal recognition of property rights and land tenure through policy

Land tenure is the legal status and ownership of land, often with a mixture of formal and informal tenure systems and mosaic of property rights, individual and collective. A study of 21 indigenous and mestizo communities in four landscape mosaics in the Peruvian and Ecuadoran Amazon, demonstrates that social relationships, and not only legal formalities, play a powerful role in tenure security (Global Landscapes Forum, 2015a; Cronkleton & Larson, 2015). In many cases, the type of land tenure – such as private ownership, community-based, government protected areas – has created conflicts and been associated with degradation. Weak or poorly defined rights and insecure tenure are strongly associated with land degradation, while uncontested land rights and strong customary tenure have provided good landscape stewardship (Global Landscapes Forum, 2015a; ELD Initiative, 2015), strengthening dialogues which entrench free, prior and informed consent (Global Forest Coalition, 2015; Guilfoyle *et al.*, 2009)**.** FAO members, nearly all countries of the world, have adopted Voluntary Guidelines to improve governance of land tenure, fisheries and forests to achieve food security (FAO, 2012).

The SDG Indicators (United Nations Economic and Social Council, 2016) include specific indicators which address land tenure. Specific example includes SDG Indicator 1.4.2: proportion of total adult population with secure tenure rights to land, with legally recognized documentation and who perceive their rights to land as secure, by sex and by type of tenure. Prohibiting formal registration of customary property rights and land tenure can lead to governments and international investors excluding local and poorer populations in restoration and rehabilitation projects, causing or exacerbating social unrest and marginalization (ELD Initiative, 2015; Plurality in Public Policy, 2014). Acknowledgment of distinct indigenous rights, including women's tenure rights (Silverman, 2015) and collaborative approaches combining different knowledge and "ways of knowing", offers the potential for successful co-generated outcomes (Araujo *et al.*, 2015; Feit *et al.*, 2013; Robinson *et al.*, 2016), including two-way knowledge techniques (Ens *et al.*, 2012; Kok & van Delden, 2009).

1.3.1.5 Good governance structures (Figure 1.2, point 1.5)

Governance, defined by the World Governance Indicators framework, is the traditions and institutions by which authority in a country is exercised (Kaufmann, 2011). This includes: (i) the process by which governments are selected, monitored and replaced; (ii) the capacity of the government to effectively formulate and implement sound policies; (iii) political commitment at the highest level; (iv) the role of coordination mechanisms that cross sectors, scales and administrative boundaries; (v) demonstrated value of mechanisms for science-policy dialogue with stakeholders; and (vi) the respect of citizens and the state for the institutions that govern economic and social interactions among them (Edelman *et al*., 2014).

Ecosystem governance integrates social and ecological components into ecosystem co-management, incorporating democracy and accountability (Vasseur *et al.*, 2017). In so doing goals, priorities, decision-making and management of the environment are determined by society, incorporating indigenous, local and practitioner knowledge to achieve successful outcomes (IUCN & State Forestry Ministry China, 2015).

Good governance affords sustainable management of environmental, economic and social resources. Multi-stakeholder involvement ensures transparency and accommodates multiple stakeholders' needs and concerns, establishing a cooperative mechanism for improving responses to avoid and reduce degradation and restore degraded lands (IUCN & State Forestry Ministry China, 2015).

Integral to good governance structures is the provision of access to information that: supports an informed dialogue; recognizes and includes multi-stakeholder engagement incorporating indigenous and local knowledge bases; and recognizes the value of diverse knowledge and opportunities for innovation, including intergenerational conservation and farming knowledge, incorporating western scientific knowledge (Fisher, 2012; FAO, 2012; Iniesta-Arandia *et al.*, 2015; Hill *et al.*, 2012; Murcia *et al.*, 2015; Robinson *et al.*, 2016)**.** Successful governance incorporates and respects indigenous and local knowledge (IUCN & State Forestry Ministry China, 2015).

An assessment of 21 case studies identified the importance of robust governance incorporating the integration of indigenous knowledge through four types of engagement: (i) indigenous-governed collaborations; (ii) indigenous-driven co-governance; (iii) agency-driven co-governance; and (iv) agency governance. The most successful outcomes have been shown to be derived from type (i) indigenous governance and type (ii) indigenous-driven co governance (Hill *et al.*, 2012; Robinson *et al.*, 2016)**.**

Active multiple stakeholder involvement and governance

A place-based approach may lead to effective economic, environmental and social outcomes. Success may result from involvement between communities, indigenous and local knowledge, business, national institutions, government officials and international institutions to achieve equal and full representation (ELD Initiative, 2015; Global Forest Coalition, 2015; Guilfoyle *et al.*, 2009; Latawiec *et al.*, 2015; Pinto *et al.*, 2014; Robinson *et al.*, 2016). Organizations in the finance sector are key partners for multi-stakeholder collaborations to avoid and reduce land degradation and restore landscapes (Van Leenders & Bor, 2016). Business and finance institutions are becoming increasingly aware of their dependency on a healthy natural environment, and understand that if their impacts are neutral, nature may sustain or regenerate itself. Degradation of the health of the ecosystems on which business depend is linked to vulnerability in business performance (Scherr *et al.*, 2017).

1.3.1.6 Communication and coordination (Figure 1.2, point 1.6)

Good communication begets good coordination. (Gottschalk-Druschke & Hychka 2015; Meli *et al.*, 2017; Robinson *et al.*, 2016; Thomas *et al.*, 2014; Schultz *et al.*, 2016). Therefore, unless all stakeholders – including legislators, policymakers, decision makers, scientists, managers, indigenous peoples and local communities, restoration innovators and others – are aware of the decisions and how they influence actions, approaches in different sectors may fail. Good communication includes horizontal frameworks as well as innovative and varying communication techniques.

1.3.1.7 Capacity-building (Figure 1.2, point 1.7)

A key factor in successful avoidance and reduction of land degradation and informed restoration is capacity-building. As we move forward with new ways of caring for the Earth and its people, it is important that everyone understands, is trained in and has capacity for implementing new and varied approaches. Capacity-building across the guiding principles is important for all elements and at all levels of understanding. Its effectiveness will be enhanced when innovative communication approaches are utilized (Calle *et al.*, 2013; Forest Peoples Programme, 2016; Ramsar, 2015; Rodrigues *et al.*, 2011; Scherr *et al.*, 2017; United Nations Economic and Social Council, 2016).

1.3.2 Solutions and nature's contributions to people

1.3.2.1 Incorporation of social and cultural instruments (Figure 1.2, point 2.1)

The IPBES Land Degradation and Restoration Assessment provides the first opportunity to catalyse the intangible assets of cultural ecosystem services by assessing and incorporating these indicators, which are strongly correlated with well-being and directly associated with land use (Hernández-Morcillo *et al.*, 2013), and pivotal to achieve effective solutions. The success and effectiveness of restoration actions may be significantly enhanced by the inclusion of traditional knowledge and local communities who live in and understand their local habitats, and are also motivated to restore them (Hallet *et al*., 2015). Perceptions and differing worldviews strongly influence understandings of success within and across the landscape and are incorporated into the assessment of success (Latawiec & Agol 2016; Nkonya *et al.*, 2016).

Cultures and the values established by people's relationships with their local environments, over time, result in the transfer of knowledge between generations – which end up playing an important role in maintaining resilient landscapes (Chazdon, 2008; Guilfoyle *et al.*, 2009; Guilfoyle, 2004; Kohler *et al.*, 2015; Kok *et al.*, 2017; Walsh *et al.*, 2015; Zheng *et al.*, 2015). Removing cultural, social, environment, legal and technical barriers improves the management of degraded land (ELD Initiative, 2015).

Across many landscapes and over time, traditional and local knowledge has decayed, whether due to immigration, emigration, marginalization or colonialism (see case study 8). For such communities to contribute positively, capacity-building mechanisms designed to restore social, cultural and local knowledge are required, such as two-way knowledge systems (Ens *et al.*, 2010; Ens, 2012).

The inclusion of social and cultural traditional practices into restoration and rehabilitation may enhance the success of projects and provide opportunities to include the key dynamics of the traditional approach into management policies (Ens *et al.*, 2015; Finlayson *et al.*, 2012; Ens *et al.*, 2010; Fisher, 2013; Fisher *et al.*, 2014; Hill *et al.*, 2013; Iniesta-Arandia *et al.*, 2015; Zheng *et al.*, 2015). Evidence from 15 countries and a wide range of traditional communities working on landscape-scale projects has identified bottom-up, place-based, participatory approaches incorporating cultural, social and differing worldviews to be highly successful in consensus decision-making (Brancalion *et al.*, 2015; Guilfoyle & Mitchell, 2015; Global Forest Coalition, 2015; Guilfoyle *et al.*, 2011; Hernández-Morcillo *et al.*, 2013; Pinto *et al.*, 2014; Robinson *et al.*, 2016).

1.3.2.2 Incorporation of approaches and strategies to resolve conflicting interests (Figure 1.2, point 2.2)

Successful mitigation and land restoration cases will be those that acknowledge that conflicts may exist, identify potential conflicts and develop a strategy to deal with known and potential conflicts (Sayer *et al.*, 2013; Scherr & Willemen, 2014).

Potential areas of conflict

Conflicting interests have the potential to impact all success factors. Conflict may occur in differing arenas and subsequently influence the degradation of land, with resultant negative impacts on people. Some examples are the extraction of natural resources (ICMM, 2013), offset proposals creating conflict between businesses, local communities and livelihood impacts (FAO, 2015), between food production, biodiversity conservation and poverty reduction (Ciccarese *et al.*, 2012), land claims and tenure (International Council on Mining and Metals, 2015; Hill *et al.*, 2013) and long term sustainability of land (IUCN & State Forestry Ministry China, 2015).

Corruption can directly impact the success or failure of excellent government policies and procedures developed for environmental and social-cultural protection. When high-level corruption occurs between, for example, government officials, large foreign enterprises, police and military, it can be difficult to stop land degradation and rehabilitate areas unless corruption can be addressed and eliminated.

Conflicts may arise among diverse values, thus integrated valuation may recognize values of multiple stakeholders, their worldviews regarding land and its values, and provide opportunities for more successful decision-making (Pascual *et al.*, 2017; Fontaine *et al.*, 2014). A coordinated landscape approach (as proposed by the operational framework) may provide opportunities to overcome such conflicts.

Food security competing with biodiversity conservation

Competition for land between, for example, agriculture and biodiversity, commercial operations and biodiversity, forest conversion, general land-use change and restoration, may result in poorly managed large-scale restoration projects. The potential outcomes being: inequality between landowners; displacement of marginalized community members; indirect land-use change; and associated social problems (Locatelli *et al.*, 2015; Latawiec *et al.*, 2015).

It is possible to maintain and increase agricultural productivity, while at the same time protecting natural resources at a national scale (Isbell *et al.*, 2015; Latawiec *et al.*, 2015; Seppelt *et al.*, 2016). To minimize agricultural impacts on biodiversity, Seppelt *et al.* (2016) proposed a framework to manage trade-offs between agriculture production and biodiversity conservation, namely land sharing and sparing. The most economically-desirable option needs to be compatible with existing economic mechanisms, while being technically, legally, environmentally and socially acceptable and feasible. This approach requires pre-conditions, an integrated suite of policies to ensure sustainable improvements in agriculture productivity, biodiversity outcomes and restoration resulting in long-term environmental and social benefits through an integrated landscape approach (Latawiec *et al.*, 2015; Seppelt *et al.*, 2016). Success would not include an “ecosystem service debt” by removing biodiverse areas for other outcomes, such as agriculture production (Isbell *et al.*, 2015).

A **whole of landscape ecosystem** approach provides possible solutions where food security and biodiversity concerns may be in conflict (Sengalama & Quillérou, 2016). Diversifying agricultural landscapes from large-scale industrial farming – such as intensive crop monocultures and industrial-scale feedlots, which can generate negative outcomes including widespread degradation of land, water and ecosystems, biodiversity losses, micro-nutrient deficiencies and livelihood stresses for farmers – has the potential to reduce land degradation, while incorporating the diversity of values of those engaged with food production. Diversified agroecological landscapes incorporate diverse farming practices which replace or greatly reduce chemical inputs, optimize biodiversity and stimulate interactions between different species. These approaches may provide a basis for secure farm livelihoods by including comprehensive strategies to build long-term soil fertility, keep carbon in the ground and sustain yields over time (IPES-Food, 2016).

Loss of livelihoods

Environmental policy designed to reduce land degradation, using livelihood change, should ensure that outcomes do not go against local interests. Successful solutions to avoid land degradation include biophysical processes and social issues, locally and broadly across the landscape and the spectrum of players. If not considered, outcomes that support more powerful actors who take control of resources while depriving villagers of their control over resources, may occur (Lestrelin & Giordano, 2007).

Substitution of natural capital with human-made capital

The replacement of resilient, self-repairing ecosystems with technological substitutes often does not provide all natural ecosystem services, and can require large engineering and maintenance costs (Moberg & Rönnbäck, 2003; UNEP-FI, 2012). Technological approaches, including environmental engineering, can often lose control and power over evolutionary functions and do not conserve natural capital (Sarrazin & Lecomte, 2016). Ecological constraints and the limiting growth factors of a site need to be considered – for example in China, learning from nature has proved to be more successful than utilizing artificial solutions alone (Grainger *et al.*, 2015; Wang, 2013). Nature-based solutions provide opportunities to sustainably manage and restore natural or modified ecosystems. Nature-based solutions, either on their own or in concert with technological and engineering solutions, aim to address societal challenges while incorporating human well-being and biodiversity benefits (Cohen-Shacham *et al.*, 2016).

Conflict between international and national interests

Clarity over acceptable trade-offs and effective strategies to deal with conflicting interests and competing objectives requires management in an all-encompassing manner to identify and prioritize impact avoidance and minimization actions, which determine whether to effectively use or avoid offsetting (Gibbons *et al.*, 2017). Drivers of degradation are not always found where local solutions are designed. Therefore, an understanding of trade policies and transboundary issues is important to establish and implement successful actions to reduce impacts of degradation activities associated with trade at the local scale (IUCN, 2016).

1.3.2.3 Values and worldviews (Figure 1.2, point 2.3)

Understanding the plurality of worldviews and diversity of values enhances coordination across the three overarching criteria and underlying factors of the operational framework. This applies particularly to situations of conflict wherein an understanding of the plurality of world views and diversity of values can provide opportunities to work towards developing effective solutions (Pascual *et al.*, 2017).

Values, human well-being and a good quality of life

The understanding of well-being and what constitutes a good quality of life is dependent on a complex mixture of values, cultures, traditions and interrelationships (Latawiec & Agol, 2016), including the point of view of those who analyse values. Some social upliftment programmes, poverty reduction schemes and agricultural policies designed to enhance human well-being may compromise the environment, human well-being and good quality of life, as was the case in Boteti, Botswana. In this case, formal land-use and management institutions have negatively influenced environmental change, through overstocking, land clearance and wildlife protection in conflict with traditional uses. These actions have led to the shrinking of Boteti's commons. Mulale's research recommends community-based natural resource strategies to secure livelihoods and conserve the commons (Mulale *et al.*, 2014). In order to achieve this outcome, it is also important for policymakers to avoid working in silos.

Effective incorporation of analyses to assess non-monetary, whole of life cycle valuation of a restoration project

Transdisciplinary approaches to valuation analyses of restoration projects incorporating nature's contributions to people may better inform decision-making and lead to greater success (Baker *et al.*, 2013; Pascual *et al.*, 2017).

The use of economics, alone, to assess projects aimed at rehabilitating and restoring degraded lands, may result in unanticipated project outcomes, potentially leading to conflict with local communities. Cultural factors can have a powerful and long-lasting effect on how individuals, communities and nations relate and respond to local implementations. Many local communities place a high value on non-monetary benefits, which are reflected in regionally-relevant social and cultural values (Easterlin *et al.*, 2010).

To avoid conflict, the development of projects would be better informed using a whole of life cycle assessment, incorporating public and private funds and including an impact measure of project outcomes (Van Leenders & Bor, 2016). A whole of life cycle assessment takes social and cultural values (i.e., non-monetary benefits) into account and includes fair participation of various stakeholder groups (Sutherland *et al.*, 2014). An impact measure could provide insights into potential negative outcomes on biodiversity and people, including values, health and well-being (Pascual *et al.*, 2017).

As countries, such as those in Latin America (Murcia *et al.*, 2015), move to reach ambitious large-scale restoration targets (Vergara *et al.*, 2016), a whole of life cycle assessment has the potential to provide an evidence base on which to operate and measure success (Murcia *et al.*, 2015). Such analyses provide opportunities to identify and remove potential barriers prior to the establishment of projects leading to greater opportunities for successful implementation (ELD Initiative, 2015).

1.3.2.4 Capacity-building (Figure 1.2, points 1.6, 2.4, and 3.5)

Successful integration of values, worldviews and nature's contributions to people within social and cultural instruments, conflict resolution, human well-being, quality of life and interactions with diverse communities may be achieved through capacity-building by fostering learning and leadership skills, and through integrated cross-sectoral approaches and communication (Cohen-Shacham *et al.*, 2016).

1.3.3 Biophysical conditions

In this section, we focus on the opportunities to enhance biophysical outcomes. Initial assessment of social and biophysical causes of land degradation provide evidence to set long-term restoration targets including comprehensive monitoring programmes to measure outcomes and adapt actions if required (Zaldivar-Jimenez *et al.*, 2010; Convertino *et al.*, 2013). Achieving successful changes to the biophysical condition is dependent on effective and well-designed biophysical and social measurements (Acuña *et al.*, 2013) . These include pre-condition and ongoing assessments in planning, design, monitoring, implementation, management and adaptation actions (see also Chapter 8, Section 8.2.3) to provide an evidence-based understanding of the outcomes of landscape change, while gaining an understanding of requirements to adapt management actions (Jackson *et al.*, 2010; Sayer *et al.*, 2013; Stanturf *et al.*, 2015; Weinstein *et al.*, 1996).

Restoration project design needs to consider potential impacts from biophysical conditions which may hinder its success – for example, through potential damage to a restoration site from hurricanes, winds, water currents, erosion and sediment. Lack of consideration may lead to projects doomed to failure (Zaldivar-Jimenez *et al.*, 2010).

1.3.3.1 Accurate assessment of ecological and biophysical conditions (Figure 1.3, point 3.1)

Successful restoration projects incorporate the establishment of firm goals (Matthews & Endress, 2008; Melo *et al.*, 2013; Ryder & Miller, 2005), include wide ranging measurements of processes and indicators (Wortley *et al.*, 2013) that are the result of inclusive and extensive consultations with scientists, policymakers, managers, stakeholders and local knowledge holders (Brancalion *et al.*, 2013; Latawiec & Agol, 2016). Successful outcomes may benefit from an assessment of ecological conditions prior to project implementation, assessing the state of land degradation (Weinstein *et al.*, 1996; Westwood *et al.*, 2014).

1.3.3.2 Monitoring (Figure 3.1, point 3.2)

Monitoring is a key procedure to measure and understand restoration success for the implementation of numerous international agreements (Murcia *et al*., 2015) such as Aichi Target 15 of the Strategic Plan for Biodiversity 2011-2020 (CBD, 2010), CBD’s Decision XI/16 (CBD 2012), the Bonn Challenge (IUCN & WRI, 2014), the New York Declaration (Murcia *et al.*, 2015) and the WRI Initiative 20x20 (IUCN & WRI, 2014). These country commitments require significant human and financial resources, for which accountability is key to understanding if actions reduce and reverse degradation and provide climate change adaptation benefits (Murcia *et al.*, 2015). Concerns exist in Latin America and other regions where, in response to countries commitments, large-scale restoration projects are being implemented with limited understanding of how to measure and guarantee success (Sansevero & Garbin, 2015; Aguilar *et al.*, 2015; Ehrenfeld 2000). An understanding of restoration responses can only be accurately determined with the incorporation of accurate evidenced-based monitoring prior to, throughout and post-restoration (Sondergaard *et al.*, 2007). Different restoration scales, ecosystem types require both their own approach and methodologies, and extensive knowledge of the dynamics, multifunctionality and interconnectedness across the landscape (Pinto *et al.*, 2014; Rodrigues *et al.*, 2011).

Similarly, understanding monitoring and design in successful agrobiodiversity projects requires an understanding of multiple socio-ecological options which improve the sustainability of the system, while improving livelihoods and providing benefits for future generations (Jackson *et al.*, 2010). The incorporation of effective landscape-scale systematic planning over time may benefit the implementation, management and success of restoration (Fisher, 2010; Grainger *et al.*, 2015; Wang, 2013; Palmer & Bernhardt, 2004; Turner II *et al.*, 2016; Pressey & Bottrill, 2008; Knight *et al.*, 2011; Knight *et al.*, 2006). There are examples where planning for conservation has been ineffective (Game *et al.*, 2013; Knight *et al.*, 2008).

To assess the ecological success of restoration projects, reliable measures of ecosystem health and function are beneficial (Jansson *et al.*, 2005; Martin *et al.*, 2005). The setting of long-term restoration targets can support and improve understanding of the cumulative impacts of climate change (FAO, 2015), which operate in concert with other degrading processes (see Chapters 3 and 4), including likely regional effects. Restoration provides opportunities to mitigate against cumulative impacts.

1.3.3.3 Landscape-scale ecological approach (Figure 1.2, point 3.3)

A landscape-scale approach considers degradation and restoration within the spatial context of the ecosystems and social systems which affect it or are affected by it – not only considering the immediate effects at the local site, but across the landscape including long-term timescales. An example of an active initiative using a landscape approach is the International Partnership for the Satoyama Initiative, which comprises 172 member organisations working to help maintain and rebuild more than 65 socio- ecological production landscapes and seascapes in at least 30 countries (Denier *et al.*, 2015; Forest Peoples Programme, 2016).

The Anthropocene is dominated by humans at all scales. Social and ecological actions in one location often influence responses some distance away (for further discussion on this see Chapter 2, Section 2.2.1.3). There is a need to mainstream a landscape and systems approach into land degradation and restoration policy and for effective monitoring over time. The landscape approach provides opportunities, for example, to incorporate existing protected areas into restoration beyond site-based activities (Bowman *et al.*, 2011; Díaz *et al.*, 2015; Grainger *et al.*, 2015; Haider *et al.*, 2016; Keenan *et al.*, 2015; Müller *et al.*, 2015; The Pew Charitable Trusts, 2014; Vellend *et al.*, 2013; Waters *et al.*, 2016).

Biodiversity, food, water, soils, carbon, climate

Accurate assessment of ecological and biophysical conditions, including reliable measures of ecosystem health and function, and landscape-scale ecological approaches (Doren *et al.*, 2009), are necessary to identify restoration success and changes in degradation in biodiversity, food, water, timber, soil, carbon, climate, wetland and urbanized landscapes (for detailed discussion of drivers and biophysical processes, see Chapters 3 and 4).

1.3.3.4 Whole of life cycle assessment (Figure 1.2, points 3.4 and 2.4)

To adequately assess the biophysical outcomes of restoration and rehabilitation programmes a whole of life cycle assessment, including biophysical, socio-ecological, financial, non-material values and fair inclusion of multiple stakeholders throughout the project, will accurately identify project results, particularly when assessed from project inception to completion (Robinson, *et al.*, 2014; Van Leenders & Bor, 2016).

1.3.3.5 Capacity-building (Figure 1.2, points 3.5, 4.5, and 1.7)

As governments work to achieve international commitments, capacity-building may assist delivery of successful outcomes in view of a potentially incremental increase of workforce in this field (Meli *et al.*, 2017; Rodrigues *et al.*, 2011; Vasseur *et al.*, 2017).

1.3.3.6 Incorporation of science and technology (Figure 1.3, point 3.6)

There are gaps and unevenness around the globe in the availability and understandings of scientific and technical knowledge to enhance restoration outcomes. In many regions, insufficient scientific and technical knowledge exists, while in other regions scientific and technical knowledge is very advanced (Grant & Koch, 2007). In situations where technological solutions are being considered to reduce degradation, the choice of technology can benefit by using interdisciplinary science to understand social, cultural and environmental effects. Any risks associated with the long-term outcomes of the introduction of new technologies will benefit from careful assessment (Similä *et al.*, 2014). Nature-based solutions provide opportunities to incorporate natural responses to reduce degradation alongside limited technological approaches (Cohen-Shacham *et al.*, 2016).

1.3.3.7 Multi-stakeholder involvement (Figure 1.2, points 1.5 and 3.7)

It is common agreement across all levels – including for implementing international commitments, effective restoration, indigenous and local communities, decision-making and policy formulation (to name a few) – that for successful outcomes to be achieved active multi-stakeholder inclusion and involvement is crucial (Van Leenders & Bor, 2016; United Nations Economic and Social Council, 2016; UN, 2012; United Nations Environment Finance Initiative, 2016; Murcia *et al.*, 2015).

1.4 A selection of success cases

These success stories represent a small number, selected from many others, with the objective to show how land management and restoration measures help improve livelihoods, reduce poverty and strengthen long-term sustainability of land use in different situations. Success cases are: results driven; have been established over a long period; provide evidence of positive ecological change, socio-economic improvements; lead, for instance, to greater food security, reduction in degradation, adaptation to change, improvement in human rights; and demonstrate long-lasting gains across the three interacting groups of the operating framework criteria (Figure 1.2). These cases show how land conservation and restoration measures have helped to deliver improvements in livelihoods, reduce poverty and strengthen long-term sustainability of land use and the extraction of natural resources.

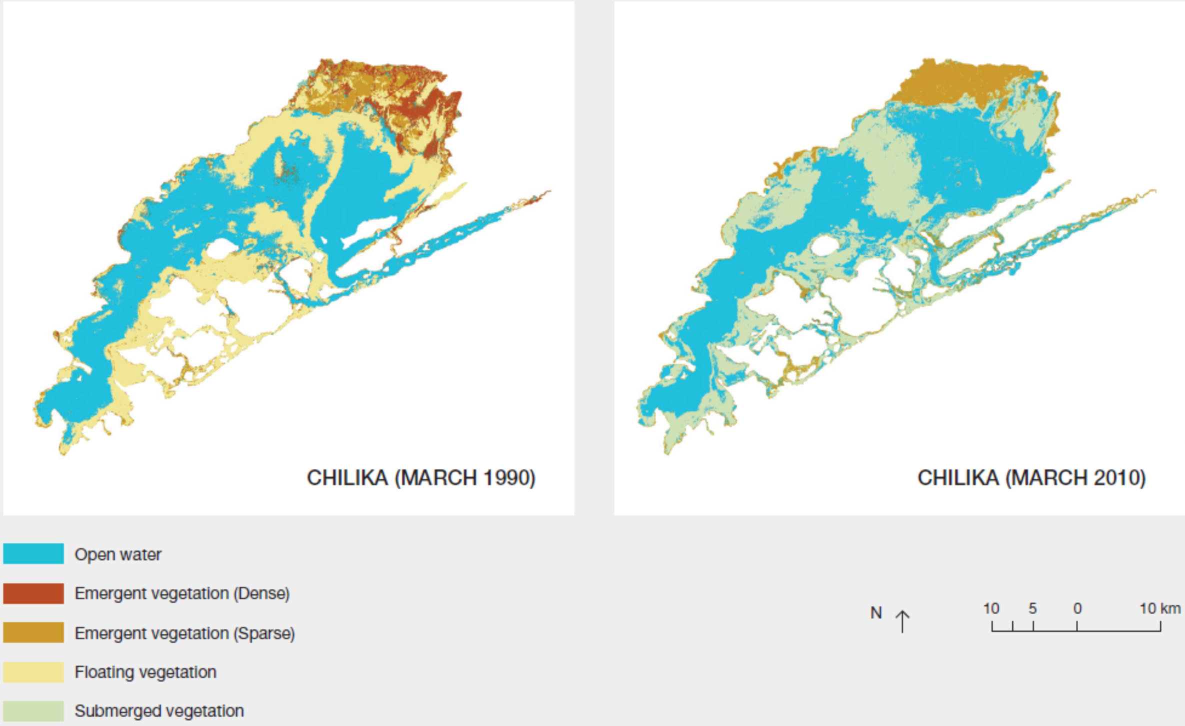
The eight success stories are deliberately selected from different regions of the world, in different landscapes and ecosystems impacted by different degradation processes. Comparisons of success evaluation scores across cases should be conducted with caution, due to these differences.

There are many other examples of successful avoidance of degradation and/or restoration of degraded land. Please see chapters 2 through 8 for further examples of successful cases.

1.4.1 Success Story 1: Lake Chilika, Odisha, India

Figure 1 3 Change in vegetation structure of Lake Chilika.

The image on the left shows the structure of Lake Chilika before hydrological restoration (March 1990) and the right panel shows the structure after restoration (March 2010). The dominant floating vegetation is *Eichornia* crassipes and the dominant emergent vegetation is Phragmites karka. Source: Pattnaik & Kumar (2016).



1.4.1.1 Context and degradation

Chilika, a brackishwater coastal lagoon on the east coast of India, in the state of Odisha, forms the base of livelihood security of more than 200,000 fishers and 400,000 farmers. The inundated area is 1,165 km^2, flanked by ephemeral floodplains of 400 km^2. Chilika is an assemblage of shallow to very shallow marine, brackish and freshwater ecosystems. Designated as a Wetland of International Importance in 1981, Chilika is famed as one of the largest congregation sites of migrating water birds in the Central Asian Flyaway, the habitat of globally vulnerable Irrawaddy Dolphin (*Orcaella brevirostris*) population and has contiguous seagrass bed in the adjacent ocean exceeding 10,000 ha.

Nature and nature's contributions to the people of Chilika are closely related to the maintenance of coastal and freshwater hydrological processes. The wetland went through a phase of reduced connectivity to the sea (1950-2000) owing to increasing sediment loads from upstream degrading catchments. As the lagoon evolved towards a freshwater environment, its fisheries rapidly declined (from an annual landing of 8600 metric tonnes in 1985/86 to 1702 metric tonnes in 1998/99), invasive freshwater aquatic plants choked the waterspread and the lagoon shrank in size. The introduction of shrimp culture in a predominantly capture fisheries setting led to the gradual breakdown of community management systems, loss of traditional fishing grounds and conflicts. Chilika was ultimately placed in the Ramsar Convention's Montreux Record in 1993 (sites having undergone adverse ecological character change).

1.4.1.2 Restoration

Responding to the immense social pressure to address wetland degradation, the Government of Odisha created the Chilika Development Authority (CDA) in 1991 as the nodal agency to undertake ecological restoration. The Authority was constituted as a multi-stakeholder institution, under the chairmanship of Chief Minister of the state. In 2000, a major hydrological intervention in the form of opening of a new mouth to the sea was undertaken based on modelling and stakeholder consultations. The intervention was complemented by basin-wide measures for treating degraded catchments, improving the well-being of fishers, communication and outreach on needs of integrated management and systematic ecosystem monitoring.

1.4.1.3 Outcomes for nature and nature's contributions to people

The response of the hydrological intervention and lake basin management has been rapid and sustained. After initial trophic bursts, the annual fish landing stabilised at nearly 13,000 metric tonnes per year. Annual censuses of Irrawaddy dolphins within Chilika reported an increase from 89 to 158 individuals between 2003 and 2015, an increase in habitat use, as well as improved breeding, dispersal and decline in mortality rates. The sea grass meadows expanded from 20 km^2 in 2000 to 80 km^2, and a significant decline in freshwater invasive species. In 2001, the site was de-listed from Montreaux Record and the intervention recognzed with the Ramsar Wetland Conservation Award and Evian Special Prize for "wetland conservation and management initiatives". Management continues under the framework of a basin-scale stakeholder-endorsed integrated management plan. Changing patterns of extreme events (as floods and cyclones) in the region, intensification of water use in the upstream reaches and rising sea-levels are major challenges which are currently being addressed through specific research (Pattnaik & Kumar, 2016).

1.4.1.4 Evaluation of success

Table 1 2 Scoring for all factors (Table 1.1) across the 3 overarching criteria (Figure 1.2) of the operational framework, both pre- and post-restoration, to evaluate the success of the project, scored against the coloured scoring values (-1 to 5) (Box 1.2).

1. Guiding instruments		Coord.*	1.1	1.2	1.2.1	1.3	1.4	1.4.1	1.5	1.5.1	1.6	1.7
2. Nature's contributions to people		2.1	2.2	2.2.1	2.2.2	2.2.3	2.3	2.3.1	2.3.2	2.3.3	2.4	2.5
3. Biophysical conditions		LD**	3.1	3.2	3.3	3.3.1	3.4	3.5	3.6	3.7		
Guiding instruments	Pre	-1	-1	-1	-1	-1	-1	-1	-1	-1	-1	-1
	Post	4	4	4	1	4	4	3	4	4	4	4
Nature's contributions to people	Pre	-1	-1	-1	-1	-1	-1	-1	-1	-1	-1	-1
	Post	4	4	4	4	5	3	3	4	4	4	4
Biophysical conditions	Pre	-1	-1	-1	-1	-1	-1	-1	-1	-1		
	Post	5	4	4	4	4	3	3	4	4		

LEGEND

1. Guiding Instruments	40/55
2. Nature's Contributions to People	43/55
3. Biophysical Conditions	35/45
1, 2, 3 combined	% Total 76%

-1	Negative
1	Limited
2	Slight
3	Slight to moderate
4	Moderate
5	Good

** Coord. = coordination*
*** LD = land degradation*

1.4.2 Success Story 2: Dune forest ecosystem rehabilitation after titanium mining

Figure 1 4 Dune restoration in Richards Bay, South Africa.

The left panel shows dune restoration weeks after mining and the right panel shows restoration after 25 years. Location: Richards Bay, South Africa (28.758 S 32.114 E to 28.705 S 32.404 E). Photo courtesy: R van Aarde.

1.4.2.1 Degradation process

The dune cordon on the north-east coast of South Africa is enriched with about 5% with the minerals ilmenite, rutile and zircon, which have been mined since about 1980 (van Aarde *et al.*, 1996). The undisturbed dunes are covered by species-rich forests and grasslands of the Maputaland centre of endemism (a "centre of endemism" is an area with an unusually high diversity of species not found elsewhere) (Wassenaar & Van Aarde, 2005) and known as a dune forest for being established on an old dune substrate. This is a fossil dune (along the coast from Richards Bay with titanium mines until

Mozambique). Further north of the mine, these littoral dunes are protected in a National Park. They provide inland protection against Indian Ocean storms, and are a source of many benefits to the local communities. Extracting the heavy metal particles involves complete removal of the plant cover and topsoil, forming a freshwater pond which is dredged to the entire depth of the deposit, up to 100 m. What is left behind is low-nutrient sand, devoid of vegetation and organic matter. Unrehabilitated, it would remain in this state for many decades while slow succession by primary dune colonizing plants occurred. During the non-vegetated time, it is a source of dust pollution, is severely compromised as a bulwark against beach erosion and produces little in the way of grazing, fuelwood, medicinal plants, edible organisms and/or tourist attractions.

1.4.2.2 Rehabilitation process

The topsoil is removed in 100m wide strips ahead of the mine and replaced within 2 months to cover the tailings behind the mine, after they have been reshaped into correctly oriented bi-parabolic dunes. Fast growing annual exotic grass (*Sorghum spp*), sunflowers, the nitrogen-fixing forb *Crotolaria* spp and the indigenous grass *Digitaria eriantha* are seeded into the 150mm thick topsoil layer, which already contains propagules of many indigenous species. The germinating cover is protected from sand-blasting with low plastic mesh windbreaks and the endemic dune pioneer tree *Vacheria* (*Acacia*) *kosiensis* is planted among the nursery cover, which is weeded to remove alien species. Once a stable cover has formed after a few years, a selection of other indigenous dune forest trees is planted as saplings (Richards, 2017).

1.4.2.3 Outcomes

Herbaceous cover is established within a year. A monodominant *Vacheria kosiensis* tree cover is complete within roughly 10 years and forest gaps begin to open after about 15 years. A three-layered forest structure (herbs, sub-canopy shrubs and canopy trees) is present by 25 years, but even by 32 years, only two-fifths of the original forest tree species are present (van Aarde *et al.*, 2012). During this period, the soil organisms, arthropods, birds and small mammals are all on a recovery trajectory which mimics that of natural dune succession (van Aarde *et al.*, 1996; Davis *et al.*, 2003; Ferreira & van Aarde, 2000; Kritzinger & van Aarde, 1998; van Aarde *et al.*, 1998; Wassenaar & van Aarde, 2001). Functions that are restored very early in the process include erosion control, storm protection, hydrological and visual rehabilitation. Grazing, fuelwood and other useful resources become available from around year 10. Biodiversity-friendly habitat structure consolidate after a couple of decades, but a full complement of pre-degradation species has not returned over a 40-year observation period (van Aarde *et al.*, 2012).

1.4.2.4 Evaluation of success

The mining company, the mine regulation authorities, the ecological research community and some local communities and environmental NGOs regard the process as a success (van Aarde *et al.*, 2012). On the other hand, other local communities and environmental NGOs have argued that the local communities have reaped few benefits and are intimidated by the propaganda power of the industry, which is a major local source of employment. (Richards Bay Minerals Dune Mining, 2017).

Table 1 3 Scoring for all factors (Table 1.1) across the 3 overarching criteria (Figure 1.2) of the operational framework, both pre- and post-restoration, to evaluate the success of the project, scored against the coloured scoring values (-1 to 5) (Box 1.2).

1. Guiding instruments		Coord.*	1.1	1.2	1.2.1	1.3	1.4	1.4.1	1.5	1.5.1	1.6	1.7
2. Nature's contributions to people		2.1	2.2	2.2.1	2.2.2	2.2.3	2.3	2.3.1	2.3.2	2.3.3	2.4	2.5
3. Biophysical conditions		LD**	3.1	3.2	3.3	3.3.1	3.4	3.5	3.6	3.7		
Guiding instruments	Pre	-1	4	1	-1	-1	1	-1	-1	-1	-1	-1
	Post	4	4	4	-1	3	4	-1	3	3	3	1
Nature's contributions to people	Pre	-1	-1	-1	-1	1	-1	-1	-1	-1	-1	-1
	Post	3	3	4	4	3	3	3	3	2	3	3
Biophysical conditions	Pre	-1	-1	-1	-1	-1	-1	-1	-1	-1		
	Post	4	4	4	4	4	3	2	4	2		

LEGEND

1. Guiding Instruments	27/55
2. Nature's Contributions to People	34/55
3. Biophysical Conditions	31/45
1, 2, 3 combined	% Total 61.50%

-1	Negative
1	Limited
2	Slight
3	Slight to moderate
4	Moderate
5	Good

* *Coord. = coordination*
** *LD = land degradation*

1.4.3 Success Story 3: indigenous land, culture and fire management in the tropical Kimberley Region, Australia

Figure 1 5 Location of the Kimberley fire management area and Indigenous Ranger Network who implement indigenous fire management to reduce land degradation from large uncontrolled wildfires. Source of the map: Kimberley Land Council.

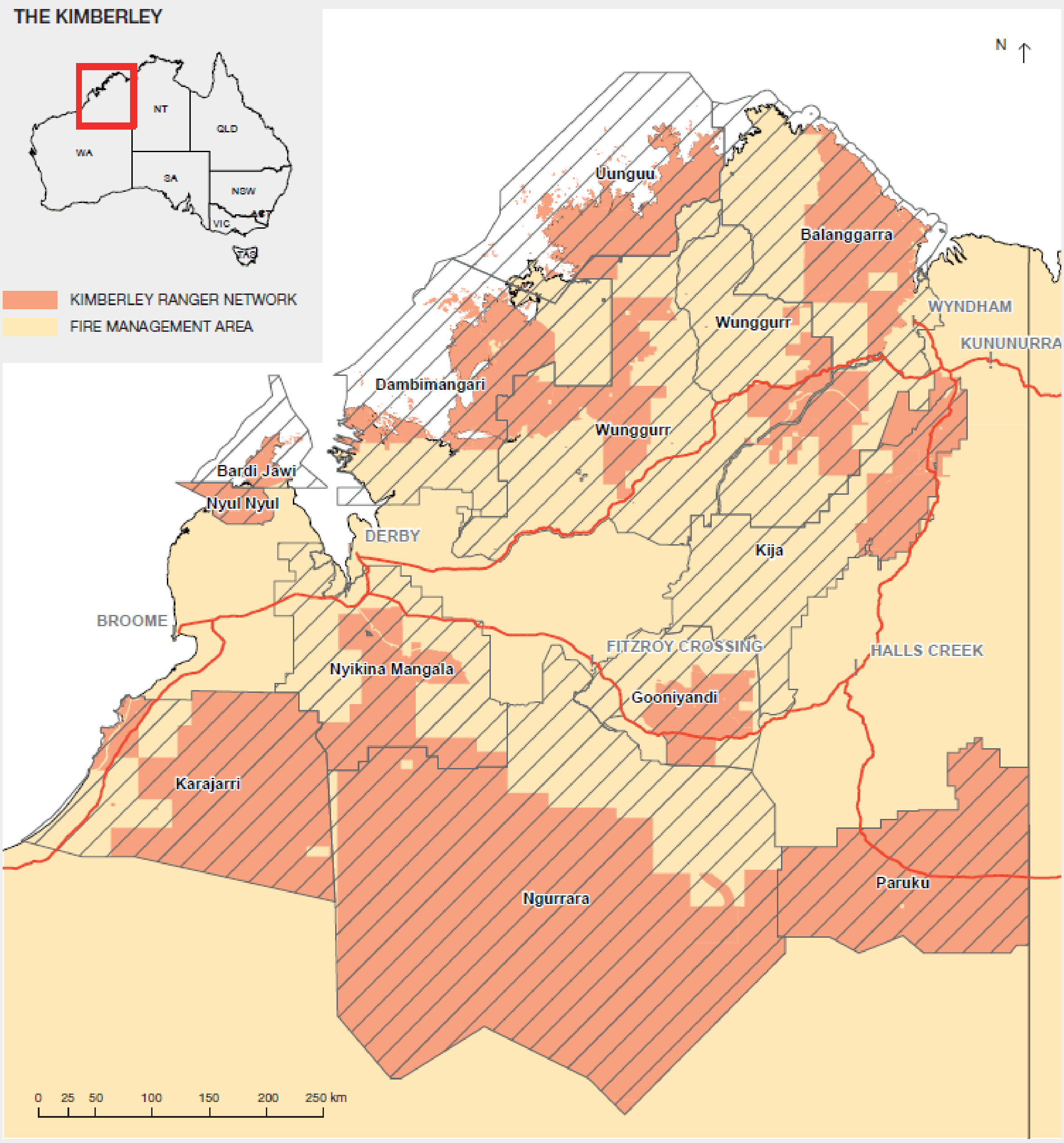


Figure 1 6 **Fire management in the tropical Kimberley Region, Australia.**

Left panel shows wildfire in the Kimberley region. Right panel shows the indigenous, early dry season mosaic burning, which reduces fire-induced land degradation. Photo credit: CSIRO Tropical Ecosystems Research Centre.

1.4.3.1 Context and degradation activity

Aboriginal people in the Kimberley Region of North Western Australia, covering 423,000 km ^{2} (Figure 1.6), have been managing their country for more than 40,000 years. They have a cultural, spiritual and social connection to country that adapts with time and space. Indigenous law, culture, language, knowledge, traditions, stories and people are embedded in the landscape, being interconnected and dependent on each other (Kimberley Land Council, 2016 b). With the onset of colonization and the removal of aboriginal people from traditional lands, during the 20th century, traditional burning practices were largely stopped (Vigilante, 2001). This led to the emergence of large, uncontrolled tropical wildfires, usually occurring late in the dry season, burning for long periods (Russell-Smith *et al.*, 2003) and damaging important ecosystems, habitats, culturally-significant sites, degrading the landscape and promoting the invasion of invasive species (Figure 1.7) (Fisher *et al.*, 2014; Russell-Smith *et al.*, 2003; Vigilante *et al.*, 2004). At the end of the dry season, the savannah grasslands across the region are extremely dry and burn out of control across large areas. Late dry season wildfires impact and degrade grazing pasture, cultural sites, biodiversity infrastructure and other assets (Russell-Smith *et al.*, 2003). Years of neglect and mismanagement, particularly of fire, and dispossession of traditional owners have created major environmental degradation problems for the savannah, pindan woodland and monsoon vine thicket plant communities and heavily impacted livestock grazing. The lower socio-economic circumstances of the aboriginal people also make it more difficult for them to adapt to and respond to the cumulative impacts of climate change (Kimberley Land Council 2016b, 2016a).

1.4.3.2 Rehabilitation actions

The Kimberley Land Council was formed in 1978 and works with aboriginal people to look after their country and gain control of their future. The Kimberley Land Council Land and Sea Management Unit began in 1998. This has enabled aboriginal people to create strong regional organisations, founded on aboriginal cultural values and governance structures. A network of 13 ranger groups, who look after land and sea across 378,704 km^2 of the Kimberley, now exists. They work to avoid and reduce degradation and restore degraded lands, achieving the cultural and environmental management outcomes that their elders and cultural advisors want to see happen on the ground (Kimberley Land Council, 2016b). Fire management, wildlife and biodiversity monitoring, and the passing on of traditional knowledge and cultural practices from old people to young people, are key priorities of the ranger groups (Kimberley Land Council 2016a). In the last 25 years, with the introduction of native title and the recognition that western fire prevention methods have not been working effectively, there has been a reinvigoration of traditional fire management in the Kimberley and across northern Australia (Legge *et al.*, 2011). In addition to improving degraded landscapes with traditional mosaic early dry season fires, aboriginal people achieved some economic independence using traditional fire management practices to develop carbon businesses (Walton *et al.*, 2014; Walsh, Russell-Smith, & Cowley, 2014) through the Indigenous Savanna Burning Carbon Projects (Figure 1.7) (Sigma Global, 2015). The North Kimberley Fire Abatement Project (Kimberley Land Council, 2016b) – working with indigenous traditional knowledge and modern scientific practices – reduces land degradation, builds cultural intergenerational knowledge transfer and is reducing the amount of greenhouse gas emissions released into the atmosphere from unmanaged and potentially dangerous wildfires (Dore *et al.*, 2014).

1.4.3.3 Outcomes

Indigenous people using traditional knowledge for fire management have reduced the greenhouse gases released into the atmosphere. For example, single wildfire events once burned up to half the 800,000 ha the Wunambal Gaamberaa project area. In the managed period, fires have been contained to within 10,000 ha in size (Moorcroft *et al.*, 2012) – avoiding emissions of 350,000 tonnes of carbon dioxide equivalent. In northern Australia, traditional fire management has proven to deliver as much as a 50% reduction in wildfires reduced emissions by 8 million tonnes, enriched biodiversity and generated more than $85 million for indigenous communities. North Kimberley native title groups generated 230,000 Kyoto Carbon Credit Units in two years. The sale of these credits provides an economic boost, delivering social and environmental outcomes through improved biodiversity and landscape health, reinvigorating social and cultural traditions, strengthening climate change adaptability, reversing socio-economic disadvantage and increasing employment opportunities (Heckbert *et al.*, 2012; Sigma Global, 2015; Dore *et al.*, 2014; Walton *et al.*, 2014). Uunguu Rangers have found major reductions in the negative impacts of uncontrolled wildfires since ramping up traditional burning methods four years ago. Through this project, traditional owners spend more time on country looking after important cultural sites and facilitating the sharing of traditional knowledge across generations, while caring for country and reducing degradation (Fitzsimons *et al.*, 2012). The Kimberley Land Council is working with the corporate sector to secure long–term benefits to increase the demand and value paid for the biodiversity, social and cultural benefits generated (Kimberley Land Council, 2016a)***.***

1.4.3.4 Evaluation of success

The change in fire management approaches has been considered a major success by land managers, indigenous communities and state and federal government departments. Positive outcomes have occurred for biodiversity, providing concurrently indigenous economic development and cultural traditional benefits, re-engaging aboriginal people with their traditional practices across generations.

Table 1.4 Scoring for all factors (Table 1.1) across the 3 overarching criteria (Figure 1.2) of the operational framework, both pre- and post-restoration, to evaluate the success of the project, scored against the coloured scoring values (-1 to 5) (Box 1.2).

1. Guiding instruments		Coord.*	1.1	1.2	1.2.1	1.3	1.4	1.4.1	1.5	1.5.1	1.6	1.7
2. Nature's contributions to people		2.1	2.2	2.2.1	2.2.2	2.2.3	2.3	2.3.1	2.3.2	2.3.3	2.4	2.5
3. Biophysical conditions		LD**	3.1	3.2	3.3	3.3.1	3.4	3.5	3.6	3.7		
Guiding instruments	Pre	-1	-1	-1	-1	-1	-1	-1	-1	-1	-1	-1
	Post	4	4	4	4	4	4	4	4	4	4	4
Nature's contributions to people	Pre	-1	-1	-1	-1	-1	-1	-1	-1	-1	-1	-1
	Post	4	3	4	4	4	4	3	3	4	3	4
Biophysical conditions	Pre	-1	-1	-1	-1	-1	-1	-1	-1	-1		
	Post	4	4	4	4	4	3	4	3	4		

LEGEND

1. Guiding Instruments	44/55
2. Nature's Contributions to People	40/55
3. Biophysical Conditions	34/45
1, 2, 3 combined	% Total 76%

-1	Negative
1	Limited
2	Slight
3	Slight to moderate
4	Moderate
5	Good

* *Coord. = coordination*
** *LD = land degradation*

1.4.4 Success Story 4: adoption of conservation tillage in Prairie Canada

Figure 1.7 Adoption of conservation tillage in Prairie Canada.

Left panel shows bare soil after being fallowed using tillage. Right panel shows soil protected by residue cover.
Photo: courtesy of Department of Soil Science, University of Saskatchewan.

1.4.4.1 Context and degradation

The former grasslands of western Canada were almost entirely converted to agricultural production during the 20th century, with an estimated 29 Mha of cropland in the region. For the first 75 years of the 20th century, the dominant soil management practice was a two-year crop-fallow system, with multiple tillage events in the fallow year leaving the soil completely bare (termed "tillage summer fallow"). Tillage summer fallow was used primarily as a water conservation measure, with soil moisture recharge during the fallow year contributing to higher yields in the crop year. The bare soil fallow and high tillage intensity led to losses of soil organic carbon estimated at approximately 25% compared to native soils and to high and continuing rates of erosion, especially wind erosion. Significant areas of the region were abandoned during the 1930s due to catastrophic wind erosion events. The high tillage intensity also led to significant tillage erosion on knolls and upper slope positions in agricultural fields, creating a patchwork of soil distribution in fields and hence high levels of within-field crop yield variability.

1.4.4.2 Description of rehabilitation actions

In the 1970s, progressive producers in the region began to adopt tillage and cropping practices that provided significantly more protection for the soil. First and most importantly, producers began to adopt conservation tillage (defined in the Canadian context as where at least 30% of the crop residue is left on the surface after seeding) and zero tillage, rather than the conventional tillage practices that left the soil bare. Second, producers reduced the frequency of fallow in the crop system. The reduction in fallow was coupled with the introduction of new crops to the region, principally canola (rape) and pulse crops such as lentils. Weed control, which had previously been accomplished with multiple tillage events each year, was instead accomplished with a broad spectrum of herbicides, especially glyphosate. Adoption of the new practices spread slowly until the 1990s, when improvements in seeding equipment, rising fuel costs and rising public concern about soil degradation combined to spur high rates of adoption. The area under conservation tillage in the region was less than 5% in 1981; by 2011, of the 29.6 Mha seeded, 16.7 Mha (56%) were in no-till and a further 7.2 M ha (24%) in conservation tillage. Only 1.4 M ha (5%) was in tillage summer fallow, down from 5.3 M ha in 1991. Throughout this period the main impetus for adoption came from the producers themselves, assisted by public sector research and extension from conservation organizations.

1.4.4.3 Outcomes

The widespread adoption of conservation tillage or no-till in Prairie Canada has led to major reductions in the risk of erosion from water, wind and tillage, and an increase in soil organic carbon levels. The erosion risk indicator calculated by Agriculture and Agri-Food Canada has steadily decreased: in 2011, 61% of cropland was in the very low risk category, whereas in 1981 only 29% was in this category. The shift to improved tillage has also led to small increases in soil organic carbon storage. A recent meta-analysis found increases in soil organic carbon in the Prairie region of approximately 3 Mg soil organic carbon ha^{-1} over the past 20 years. Although the per hectare amount is small (perhaps equal to 10 to 15% of the soil organic carbon lost due to initial cultivation), the overall contribution to Canada’s greenhouse gas budget is substantial - soils went from being a 1 Mt CO_2e source in 1981 to an 11.7 Mt CO_2e sink in 2006, driven largely by the shift in management practices in the Canadian Prairies. Concerns continue to be raised, however, about the continuing use of glyphosate to suppress weeds and its possible effects on soil biota and aquatic ecosystems (AAFC, 2013; Awada *et al.*, 2014; Clearwater *et al.,* 2016; Statistics Canada, 2015; Vandenbygaart *et al.*, 2003). A detailed account on the impact of glyphosate is available in Chapter 4 (see Section 4.2.4.2).

1.4.4.4 Evaluation of success

Table 1 5 Scoring for all factors (Table 1.1) across the 3 overarching criteria (Figure 1.2) of the operational framework, both pre- and post-restoration, to evaluate the success of the project, scored against the coloured scoring values (-1 to 5) (Box 1.2).

1. Guiding instruments		Coord.*	1.1	1.2	1.2.1	1.3	1.4	1.4.1	1.5	1.5.1	1.6	1.7
2. Nature's contributions to people		2.1	2.2	2.2.1	2.2.2	2.2.3	2.3	2.3.1	2.3.2	2.3.3	2.4	2.5
3. Biophysical conditions		LD**	3.1	3.2	3.3	3.3.1	3.4	3.5	3.6	3.7		
Guiding instruments	Pre	-1	-1	-1	-1	-1	-1	-1	-1	-1	-1	-1
	Post	5	4	3	1	3	3	1	4	4	5	5
Nature's contributions to people	Pre	-1	-1	-1	-1	-1	-1	-1	-1	-1	-1	-1
	Post	4	4	5	4	2	4	3	4	1	3	4
Biophysical conditions	Pre	-1	-1	-1	-1	-1	-1	-1	-1	-1		
	Post	5	4	4	4	5	4	4	4	4		

LEGEND

1. Guiding Instruments	38/55
2. Nature's Contributions to People	38/55
3. Biophysical Conditions	38/45
1, 2, 3 combined	% Total 73%

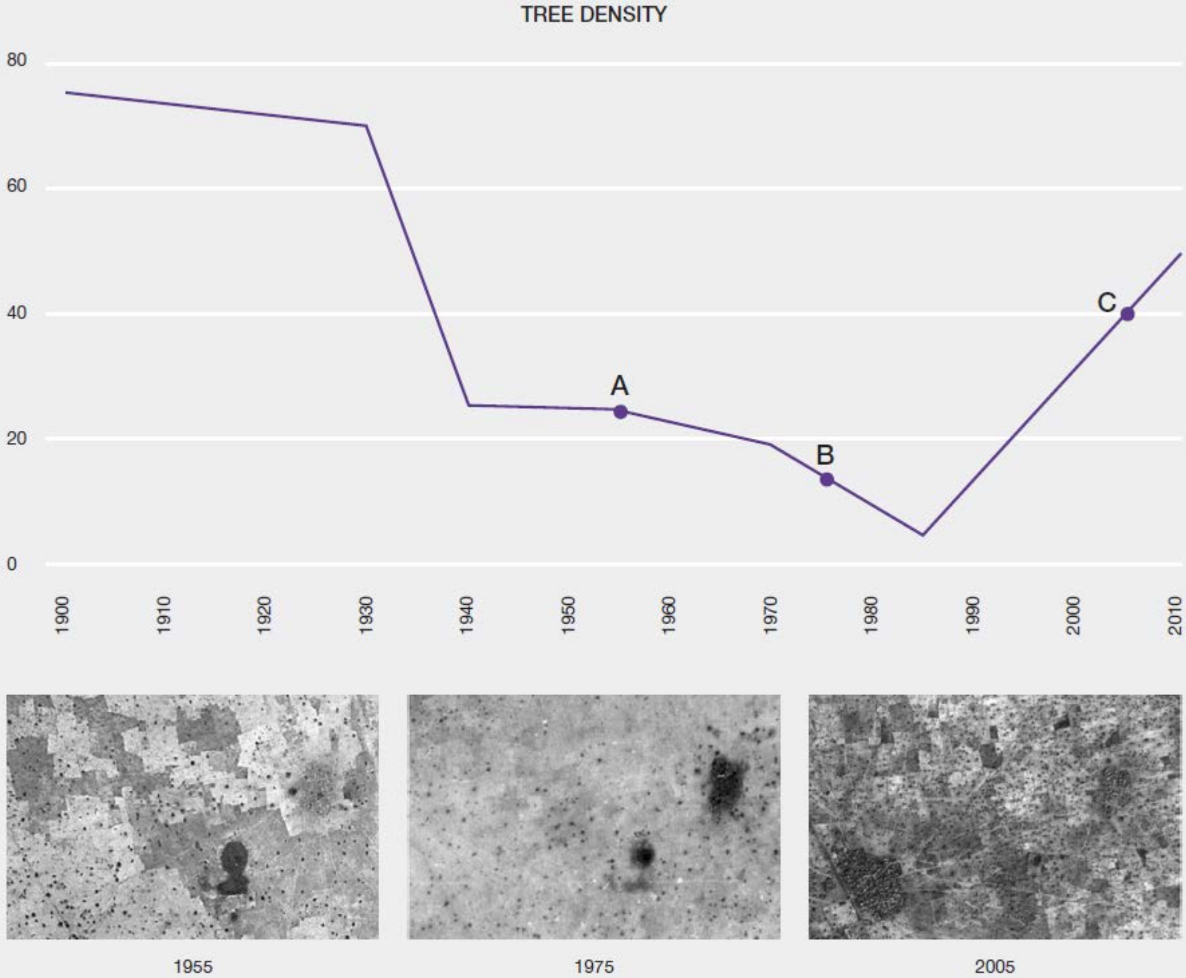
-1	Negative
1	Limited
2	Slight
3	Slight to moderate
4	Moderate
5	Good

* *Coord. = coordination*
** *LD = land degradation*

1.4.5 Success Story 5: regreening the Sahel through tree regeneration

Figure 1 8 Regreening the Sahel through tree regeneration.

The tree and field cover trends estimated as changes in tree density (unit = percent of precolonial tree cover): landscape dynamics in southwest Zinder. Scale is 1:5000. The 1975 aerial photos (1:50 000) were zoomed in to the specific terroir shown. Note: the emergence of a large village and severe shrinking of a wet area east of it may suggest that the 2005 image is of a different area, but all three images cover the identical geographical location. Remote sensing imagery courtesy of Dr. G. Tappan (Tappan & Cushing 2004, 2008).



1.4.5.1 Degradation process

The Sahel is a semi-arid region (200-700 mm annual rainfall) immediately south of the Sahara Desert, an approximately 500 km wide band stretching almost across Africa, with a total area of around 160 million ha and a population of 100 million people, mostly very poor. The annual rainfall, highly variable throughout the period of record, decreased abruptly and persistently by about a fifth between 1968 and 2005 and then apparently recovered (Mitchell, 1997; Ouedraogo *et al.*, 2014). Severe food insecurity, increased morbidity, loss of livestock and livelihoods was a region-wide phenomenon during the three-decade dry period (Franke & Chasin, 1980). The prolonged dry phase is now attributed to a temporary change in ocean circulation (Giannini *et al.*, 2003). At the time, it was thought that land degradation was either directly caused by overgrazing and tree cutting (Mainguet & Chemin, 1991; Le Houérou, 2002), or those activities had led to regional-scale desiccation (Xue & Shukla, 1988) – although some viewed the changes as mostly reflecting decadal rainfall variability (Nicholson, 2001).

The traditional farming system includes crops grown interspersed with selected and nurtured trees, in a rangeland matrix supporting cattle and goats. Clearing of the trees was advised by colonial and post-colonial extension services, since the trees were viewed as “weeds” competing with the crops and grass. Without the trees, however, soil exposed to sun and wind lost its capacity to absorb and retain water. Fertility declined and wind-blown sand covered the exposed crops. Crop plagues and pests increased over time, while the population of insects and birds that control them, deprived of their habitats, declined. Crop and livestock yields fell, increasing chronic hunger. Without fuelwood, people burned manure and crop residues for domestic cooking fuel, eliminating the main source of soil improvement (Reij et al., 2005; Herrmann & Tappan, 2013) .

1.4.5.2 Rehabilitation actions

The dry “mode” of regional climate apparently returned to “normal” mode, without human intervention. Yet, it remains an open question as to whether future reverse flips will occur and if they are and will be related to global climate changes (Giannini *et al.,* 2003). As a response to the degraded conditions, a project was set up in Niger to encourage farmers to regenerate natural trees from stumps. The new trees provided firewood, fruits, edible leaves and nuts, timber, medicines, fodder, dyes, soil protection and ameliorated the microclimate. Using the wood, provided for fire once again, freed-up crop residues and manure as soil amendments, improving their fertility, structure and reducing soil erosion, and leading to greater rainwater infiltration. Fewer pests and diseases were observed. The return of favourable conditions of both rainfall and soils led to higher crop yields and diversification of food sources and income - which in turn increased production resilience to extreme weather events. However, it remains disputed what fraction of this recovery was due to active rehabilitation efforts and how much was due to the return of the previous climate (Brandt *et al.*, 2015; Mbow *et al.*, 2015; Brandt *et al.*, 2017; Olsen *et al.*, 2015; Fensholt & Rasmussen, 2011), but all agree that active tree regeneration played a significant role (Behnke & Mortimore, 2015). Regulation also played an important role; previous attempts to plant windbreaks and woodlots of exotic trees in the region failed because trees were state property, thus farmers could not cut the trees planted on their land. Changes in the laws gave farmers ownership of the trees. Advantages derived from trees on the land stimulated more farmers to adopt this practice. The initial project spread to Burkina Faso, Mali and Senegal (Reij *et al*., 2005; Herrmann & Tappan, 2013).

1.4.5.3 Outcomes

The vegetation cover of the Sahel, as observed by satellites and measured by the Normalised Difference Vegetation Index (NDVI), has generally increased over the period 1987 to 2015 (Anyamba & Tucker, 2005; Anyamba *et al.*, 2014; Dardel *et al.*, 2014; Fensholt *et al.*, 2009; Horion *et al.*, 2014), but not everywhere (Rasmussen *et al.*, 2014). Much of this increase has been attributed to the return of higher rainfall and some is due to tree planting (Brandt *et al.*, 2015; Mbow *et al.*, 2015; Brandt *et al.*, 2017; Olsen *et al.*, 2015; Fensholt & Rasmussen, 2011). There is field- and satellite-based evidence for increases in tree and shrub cover (Brandt *et al.*, 2017; Horion *et al.*, 2014; Hänke *et al.*, 2016). More than 200 million trees of various species, generally indigenous and local, were established or planted since 1985 – restoring more than 5 million ha of land. Grain production increased by half a million tonnes per year and there was fodder for many more livestock. As a result, food security improved for more than 2.5 million people (Reij *et al.*, 2009). The capacity of the Sahelian landscape to deliver natural contributions to people is agreed by all to have increased over the past two decades, relative to the previous three decades.

1.4.5.4 Evaluation of success

Table 1 6 Scoring for all factors (Table 1.1) across the 3 overarching criteria (Figure 1.2) of the operational framework, both pre- and post-restoration, to evaluate the success of the project, scored against the coloured scoring values (-1 to 5) (Box 1.2).

1. Guiding instruments		Coord.*	1.1	1.2	1.2.1	1.3	1.4	1.4.1	1.5	1.5.1	1.6	1.7
2. Nature's contributions to people		2.1	2.2	2.2.1	2.2.2	2.2.3	2.3	2.3.1	2.3.2	2.3.3	2.4	2.5
3. Biophysical conditions		LD**	3.1	3.2	3.3	3.3.1	3.4	3.5	3.6	3.7		
Guiding instruments	Pre	-1	-1	-1	-1	-1	-1	-1	-1	-1	-1	-1
	Post	5	4	5	4	5	4	4	4	5	4	4
Nature's contributions to people	Pre	-1	-1	-1	-1	-1	-1	-1	-1	-1	-1	-1
	Post	4	4	5	5	4	4	4	5	3	4	4
Biophysical conditions	Pre	-1	-1	-1	-1	-1	-1	-1	-1	-1		
	Post	5	3	4	5	5	3	4	3	4		

LEGEND

1. Guiding Instruments	48/55
2. Nature's Contributions to People	46/55
3. Biophysical Conditions	36/45
1, 2, 3 combined	% Total 82%

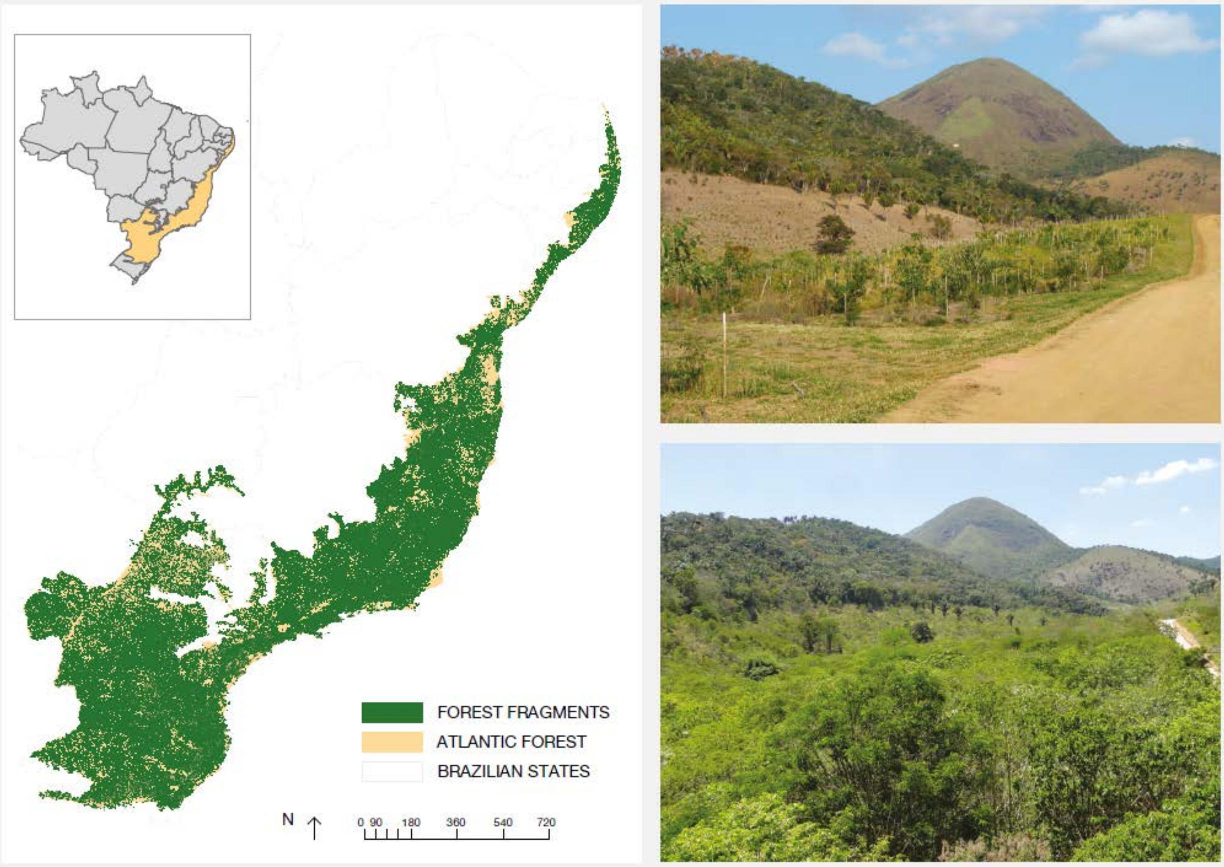
-1	Negative
1	Limited
2	Slight
3	Slight to moderate
4	Moderate
5	Good

* *Coord. = coordination*
** *LD = land degradation*

1.4.6 Success Story 6: the Brazilian Atlantic Forest

Figure 1 9 The Brazilian Atlantic Forest.

The map on the left shows the location of the Atlantic Forests and Atlantic Forests fragments.
The right panels show an example of a restoration site before restoration (top right) and after restoration (bottom right).
Map source: Instituto Internacional de Sustentabilidade. Photos: courtesy of Ricardo R. Rodrigues.



1.4.6.1 Context and degradation activity

The Atlantic Forests, with high species diversity and endemism, extend along the Atlantic coast of Brazil from Rio Grande do Norte, in the north, to Rio Grande do Sul, in the South, and inland as far as Paraguay and the Misiones province of Argentina. The Tupi people dominated the Brazilian Atlantic coast before the arrival of European settlers. After 500 years of land-use change, less than 12% of the original forest cover (1.2 million km^2) remains, mostly in isolated fragments and of which 90% is privately held. Forest clearing for coffee plantations and cattle ranching, and logging for hardwoods are the principles threats (Pinto *et al.*, 2014). Throughout the twentieth century, the Brazilian Government enacted a series of legal instruments to support sustainable forest use, including laws regulating the use of native forests (1965). Weak environmental governance, poor compliance and - from the 1980s onward social concern for the Atlantic Forest pressured governments to enforce laws more rigorously and support grew for the restoration of the Atlantic Forest (Rodrigues *et al.*, 2009). In 1988, the Brazilian Federal Constitution established that authorities should promote restoration of ecological processes with the aim to guarantee a healthy environment for Brazilian society (Pinto *et al.*, 2014). Public prosecution, from 2000 onwards, resulted in large-scale restoration projects – with more recent innovative legal instruments regulating forest restoration and incorporating socio-ecological benefits. Despite such instruments and social understanding of the need for restoration, the restoration process was disorganized, with poor dialogue between the multiple stakeholders and limited incentives for implementation. A disaggregated approach to forest landscape restoration led to inefficiencies which, in the end, did not lead to effective restoration at the landscape scale. The solution was to bring everyone together with the creation of the Atlantic Forest Restoration Pact.

1.4.6.2 Description of rehabilitation actions

In 2006, a group of NGOs and researchers developed a plan, including a diverse coalition of interests and agendas from all forest restoration actors, which resulted the 2009 Atlantic Forest Restoration Pact. The Pact is a multi-stakeholder coalition aiming to restore 1 million ha of the Atlantic Forest by 2020 and 15 million ha by 2050, doubling native cover to at least 30% of the original biome area (Aguilar *et al.*, 2015). The Pact aims to: promote biodiversity conservation; create jobs and provide income generating opportunities through the restoration supply chain; restore key ecosystem services for millions of people; and establish incentives for landowners to comply with the Forest Act. The joint effort of more than 270 members from the private sector, governments, NGOs and research organisations has changed how large-scale forest landscape restoration is practiced in the region. The development of a new web-based database allows continuous monitoring of progress towards the ambitious goal and allows project implementers to optimise the benefits from restoration. The Atlantic Forest Restoration Pact has produced thematic maps to guide restoration, economic models to lead forest rehabilitation projects, guides for restoration and monitoring and capacity-building programs (Brancalion *et al.*, 2013; Calmon *et al.*, 2011; Melo *et al.*, 2013; Pinto *et al.*, 2014; Rodrigues *et al.*, 2011).

1.4.6.3 Outcomes

The Atlantic Forest Restoration Pact aims to restore tens of thousands of hectares (as of late 2017). It is estimated that the potential for job creation is as high as 6 million new jobs (Melo *et al.*, 2013), mostly in rural communities, for full implementation. Maintaining the Pact’s governance mechanisms is fundamental to its success. Several challenges need to be overcome, such as representation from all four major sectors. Moreover, the uneven geographical distribution of its members will need to be addressed in the future. Achieving success is dependent on the engagement and commitment of all its members towards a common vision, goals and objectives. The Atlantic Forest Restoration Pact is incorporating people and human well-being into restoration planning and action, and working to reverse the Atlantic Forests’ reputation as a dwindling biodiversity hot spot, into a region of hope for the future. To reduce the negative impacts of climate change on society and their livelihoods, the Pact is involving society in the protection and restoration of nature to improve peoples’ standards of living (McKenna & Hemphill, 2010; Rodrigues *et al.*, 2011; Scarano & Ceotto, 2015).

1.4.6.4 Evaluation of success

Despite innovative legal instruments, problems occurred in implementing effective restoration of the Atlantic Forest due to weak environmental governance, poor compliance and limited connections between multiple stakeholders. The establishment of the Atlantic Forest Pact (2009) has played a key role in working to overcome these conflicts by fostering collaborations. A consistent monitoring approach has been developed, capacity-building and guidelines established, with the AFRP having more than 40,000 ha of restoration projects registered. At this stage, it is too early to understand the long-term ecological and social effectiveness of these projects and, to date, there does not appear to be much engagement with or involvement from indigenous peoples. For these reasons, a low value was given for biophysical conditions and a medium value for nature’s contributions to people.

Table 1 7 Scoring for all factors (Table 1.1) across the 3 overarching criteria (Figure 1.2) of the operational framework, both pre- and post-restoration, to evaluate the success of the project, scored against the coloured scoring values (-1 to 5) (Box 1.2).

1. Guiding instruments		Coord.*	1.1	1.2	1.2.1	1.3	1.4	1.4.1	1.5	1.5.1	1.6	1.7
2. Nature's contributions to people		2.1	2.2	2.2.1	2.2.2	2.2.3	2.3	2.3.1	2.3.2	2.3.3	2.4	2.5
3. Biophysical conditions		LD**	3.1	3.2	3.3	3.3.1	3.4	3.5	3.6	3.7		
Guiding instruments	Pre	-1	1	3	-1	1	1	-1	-1	-1	-1	-1
	Post	5	4	5	1	4	4	1	5	5	4	4
Nature's contributions to people	Pre	-1	-1	-1	-1	-1	-1	-1	-1	-1	-1	-1
	Post	4	4	3	3	4	4	3	3	1	2	4
Biophysical conditions	Pre	-1	-1	-1	-1	-1	-1	-1	-1	-1		
	Post	1	4	4	4	3	3	4	5	4		

LEGEND

1. Guiding Instruments	42/55
2. Nature's Contributions to People	35/55
3. Biophysical Conditions	36/45
1, 2, 3 combined	% Total 69%

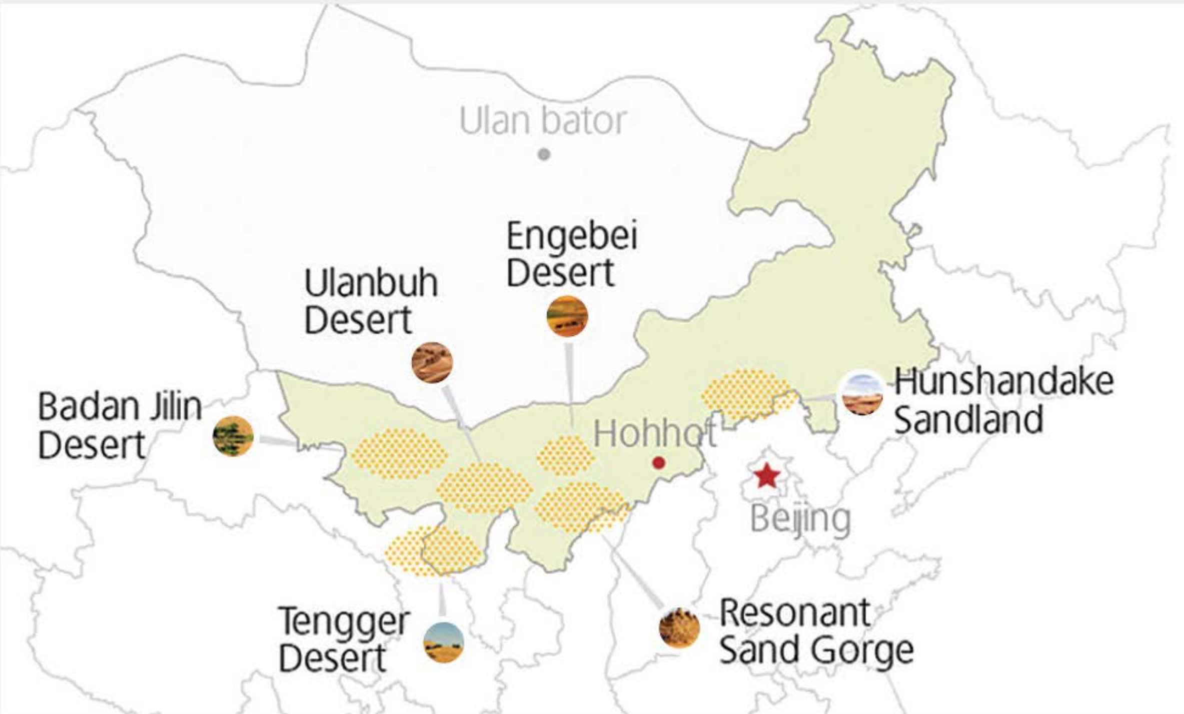
-1	Negative
1	Limited
2	Slight
3	Slight to moderate
4	Moderate
5	Good

** Coord. = coordination*
*** LD = land degradation*

1.4.7 Success Story 7: Hunshandake Sandland Inner Mongolia - sustainable management of marginal drylands (SUMAMAD)

Figure 1 10 Sustainable management of marginal drylands (SUMAMAD) in Hunshandake Sandland Inner Mongolia.

The map below shows the location of Hunshandake Sandland Inner Mongolia: 41°56'-44°22' N, 112°22'-117°57' E, 1100-1300 m a.s.l.



1.4.7.1 Context and degradation

China's rangelands are the second largest in the world. Hunshandake Sandland (41°56′-44°22′ N, 112°22′-117°57′ E, 1100-1300 m a.s.l.) is located within the Xilin Gol Plateau close to the Xilin Gol Biosphere Reserve, in a semi-arid grassland ecosystem - with habitats of sparse elm forests, lowlands, hills and wetlands. It is 450 km long, 50~300 km wide and has an area of 53,000 km^2. Monthly temperatures range from -18.3 °C in January to +18.5 °C in July and most of the annual precipitation (250 to 400 mm) falls during summer. Hunshandake has a population of 128,000 people, 40% of whom are Mongolian (Thomas *et al.*, 2014). Virtually all (92%) of the local population's income is derived from stockbreeding, including cattle, goats, sheep, horses and camels. Towards the end of the twentieth century, these animal numbers increased rapidly, reaching 108,0000 animals. The large number of medium to-large mammals is the main reason for the serious degradation of the Hunshandake Sandland. Serious land degradation has limited the ability of the land to carry enough animals to sustain the livelihoods of local families (Liu *et al.*, 2013; Jiang, 2009).

1.4.7.2 Restoration

The sustainable management of Marginal Drylands established a comprehensive, multi-partner/stakeholder project, which included government, local farmers, scientists/experts and businesses, (Thomas *et al.*, 2014).

This project adopted an alternate strategy to that usually employed in grassland restoration, artificially increasing primary production. This alternative replaced the major grassland consumers with less destructive animals (i.e., chickens). The natural grasslands were used for chicken farming, reducing overgrazing ruminant pressure, establishing a different source of income for the local community. However, it is important that these practices are designed in such a way that they have minimal impact on traditional nomadic cultures (Su *et al.*, 2017).

The community's work intensity has been reduced. Chicken farming requires 4 months of activity, while the traditional practices of intensive rearing of lambs and calves requires 12 months of continuous activity. Grasslands have a variety of trees, shrubs, forbs and grasses with fruits, leaves and insects - forming the natural diet for free-range chickens. The above-ground plant biomass was similar between the chicken faming and the control situations. Pecking and scratching caused less soil disturbance and compaction than in the case of large and middle-sized mammals. More water was found in soils manured by chickens, sustaining non-degrading grassland soils. As a restoration pathway, chicken farming also enhances local people's income. The economic benefit of chicken farming, raised organically, was approximately six times higher (per hectare) than grazing sheep. This restoration approach has been applied across 10 800 km^2 of the of the Hunshandake sandland and sequesters more carbon than the degraded ecosystem (Su *et al.*, 2017; Liu *et al.*, 2007, 2013). Satellite images were used to calculate land-use patterns for different land coverages (e.g., meadow, steppe, spare elm tree, desert and crop farm) throughout the restoration process (Schaaf, 2011).

Further research is being conducted to establish the impacts on grassland ecosystems of selective feeding of chickens. Future use of this restoration approach would limit the number of medium and large livestock and ensure traditional nomadic practices, however not prohibit livestock grazing, to ensure traditional nomadic practices are enduring (Liu *et al.*, 2013). The deep-rooted attachments of the local herdsmen to livestock grazing, suggest that the most effective approach is an integrative land-use approach, where herders systematically use their rangelands incorporating both practices (Li, 2011; Papanastasis *et al.*, 2015; Li & Huntsinger, 2011; Papanastasis *et al.*, 2015).

1.4.7.3 Outcomes

Thanks in part to the uptake of policy recommendations and good restoration outcomes on degraded grasslands, there has been a three-fold increase in above- ground plant biomass in chicken farmed land compared to land with medium to large animals. The sustainable management of Marginal Drylands project has received large financial investments from the Chinese government and other partners. Potential has also been identified for carbon payments. Together with the traditional deep-rooted livestock grazing of the local herdsmen, organic chicken farming is a viable integrated and comprehensive landscape-farming method. Farmers have received a six-fold increase in economic return, for less intensive time commitments. Raising free-range chickens increased the communities' income by 54%, compared with sheep grazing. The reduction in livestock grazing has resulted in an increase in biomass of groundcover, reducing soil erosion, and land degradation.

1.4.7.4 Evaluation of success

Table 1 8 **Scoring for all factors (Table 1.1) across the 3 overarching criteria (Figure 1.2) of the operational framework, both pre- and post-restoration, to evaluate the success of the project, scored against the coloured scoring values (-1 to 5) (Box 1.2).**

1. Guiding instruments		Coord.*	1.1	1.2	1.2.1	1.3	1.4	1.4.1	1.5	1.5.1	1.6	1.7
2. Nature's contributions to people		2.1	2.2	2.2.1	2.2.2	2.2.3	2.3	2.3.1	2.3.2	2.3.3	2.4	2.5
3. Biophysical conditions		LD**	3.1	3.2	3.3	3.3.1	3.4	3.5	3.6	3.7		
Guiding instruments	Pre	2	1	1	-1	1	-1	1	-1	-1	-1	-1
	Post	4	5	1	-1	2	4	3	4	5	4	4
Nature's contributions to people	Pre	2	-1	2	2	1	-1	-1	-1	2	-1	-1
	Post	4	4	5	5	1	3	4	4	4	4	4
Biophysical conditions	Pre	-1	-1	-1	-1	-1	-1	-1	-1	-1		
	Post	4	4	3	4	4	3	4	4	4		

LEGEND

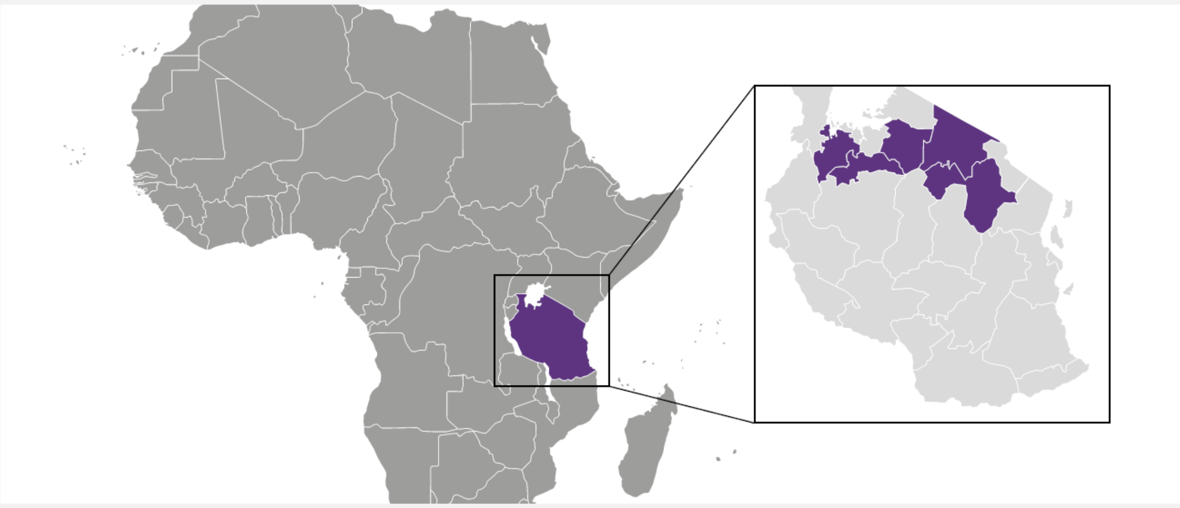
1. Guiding Instruments	34/55
2. Nature's Contributions to People	42/55
3. Biophysical Conditions	34/45
1, 2, 3 combined	% Total 71%

-1	Negative
1	Limited
2	Slight
3	Slight to moderate
4	Moderate
5	Good

* *Coord. = coordination*
** *LD = land degradation*

1.4.8 Success Story 8: Ujamma community resource team - northern Tanzania pastoralist and agro-pastoralist communities

Figure 1 11 Ujamaa Community Resource Team location. Source: Tanzania/UNDP Equator Initiative.



1.4.8.1 Context and degradation

Northern Tanzania has rich savannas, grasslands and montane landscapes, a diverse array of farmers, traditional pastoralists and hunter-gatherer communities. Longstanding competition over land and its resources exists amongst local communities. Over the last century, the loss of extensive areas of land to large-scale commercial farms or state protected areas has had negative impacts on indigenous communities. Legal and policy instruments often commandeered local resources, degrading landscapes and traditional livelihoods, and failing to recognise traditional systems of land use. The livelihoods of pastoralist, agro-pastoralist and hunter-gatherer communities, such as the Maasai, Barabaig, Akie, Sonjo and Hadzabe communities, are under threat from: the overexploitation of natural resources; political marginalization; limited resources; and access to knowledge. Marginalization has been further exacerbated by the geographical remoteness of many ethnic minority communities.

This has resulted in less productive agriculture, exacerbated by drought, loss of fertility and climate change. Moreover, the kinds of knowledge that hunter-gatherers possess about harvesting wild foods (plants, honey and so on) become more important to food security and nutritional well-being. While the policy environment enables local groups to formalise rights over lands and resources, the political economic environment can skew power relations in favour of non-local actors, such as commercial investors or national government bodies.

1.4.8.2 Restoration and rehabilitation processes

The Ujamma Community Resource Team was founded in 1998 and operates across the Yaeda valley, as well as in the Kiteto, Ngorongoro, Simanjiro, Longido and Hanang districts of northern Tanzania. The Ujamma Community Resource Teams' mandate is to work with indigenous groups in Northern Tanzania who depend on communal natural resources to support their livelihoods, towards rehabilitating and restoring northern Tanzania's degraded landscapes by including their customary rights and practices. Ujamma Community Resource Team works with Tanzania's village land legislation (Tanzanian Land Act of

1999) and assists communities to develop by-laws from this legislation and develop land-use plans for their customary lands, while focusing on improving their ecosystem management capacity.

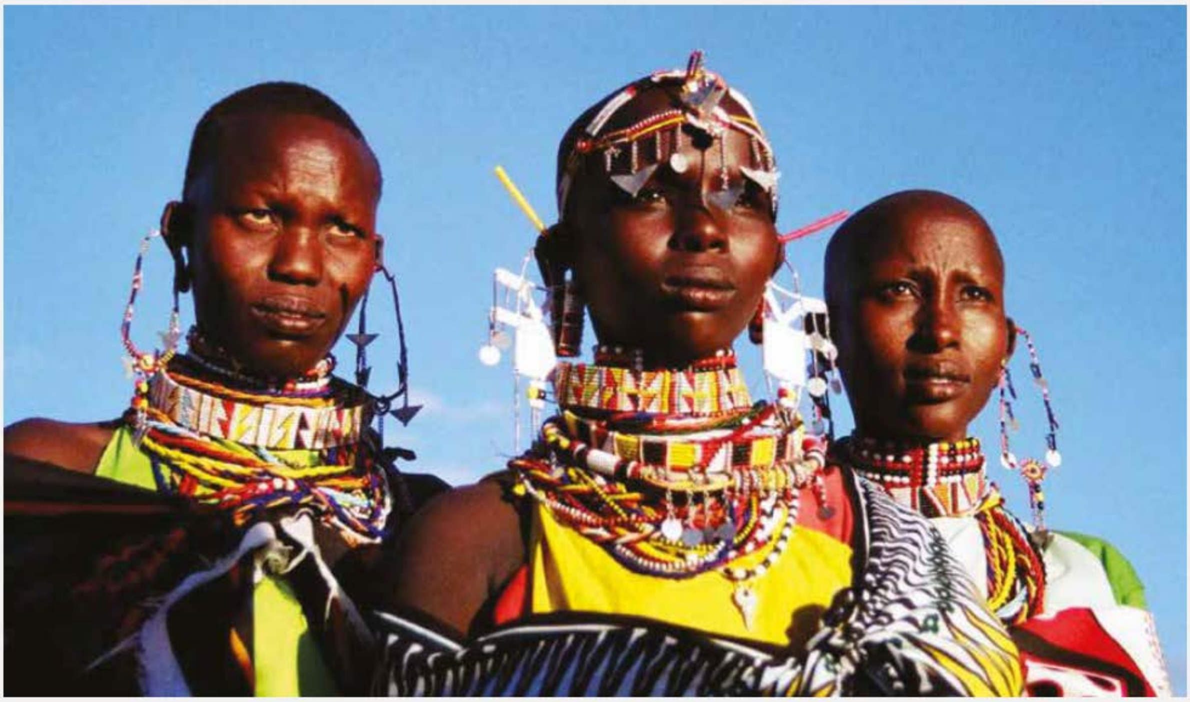
Figure 1 12 Ujamaa Community Resource Team. Photo credit: Tanzania/UNDP Equator Initiative.

They operate across four key foci: land use, natural resource management, community empowerment and advocacy. The goal is the restoration and rehabilitation of marginalized lands and communities to: secure land and resource rights; improve natural resource management capacities; develop management skills and tools; establish and manage community reserved areas using indigenous land management practices, while enhancing economic benefits. Capacity-building, conflict resolution and sustainable livelihood programmes underpin the work, enhancing the effectiveness of the rural communities as land and resource managers. Ujamma Community Resource Team has secured several landmark agreements, including the legal demarcation of the first village for hunter-gatherers in Tanzania - which has increased land access and security, improved gender rights and raised community confidence across marginalized indigenous communities, while reducing land degradation.

The Ujamma Community Resource Team assists with the development of land-use plans that ensure communities have secure property rights and resource access, and has assisted with surveying, mapping and demarcating community lands to ease inter-community conflicts and the process of formalizing tenure. To ensure good governance they assist committees within village councils to oversee resource plans and monitor resource use. This resource mapping has resulted in innovative partnerships between communities.

Ujamma has worked with four other Tanzanian groups to found the Mama Ardhi Alliance, which has played an instrumental role in successful efforts to ensure provisions enshrining women's rights to land ownership, were included in the new proposed Constitution 2014, or Katiba inayopendekezwa, passed by the Constituent Assembly in October 2014. Women's empowerment programmes are operated in conjunction with the Pastoral Women's Council of Tanzania: an NGO working with pastoralist groups in northern Tanzania to advance women's rights and the education of Maasai girls.

Figure 1 13 Ujamaa Community Resource Team. Photo credit: Tanzania/UNDP Equator Initiative.



1.4.8.3 Outcomes

These sustainable management practices have reduced conflict, achieved secure land tenure and provided improvements in the health and well-being of the land, wildlife and communities between 1998 and 2016. In 2008 the Ujamaa Community Resource Team was awarded the UNDP Equator Prize and, in 2016, Edward Loure, the Director for a decade, was the 2016 Goldman Environment Prize Winner for Africa (United Nations Development Programme, 2012; Siandei, 2016; Ujamaa Community Resource Team, 2015). The continued success of these partnerships has brought awareness, understanding and acceptance at all levels of society. One of the main socio-economic impacts has been the fostering of private sector partnerships that have enabled villages to earn income.

The ecological condition of this area has improved considerably over the past decade and can support hunter-gatherer livelihoods. It has also allowed the recovery of local wildlife populations, which faced pressures from competing livestock grazing, as well as hunting by farmers that had immigrated to the area. The recovery of natural resources (e.g., water sources, forested areas) has improved the food security of the local people and established clear rules for governing access to land and resources - in conjunction with local government authorities to demarcate, plan and legally formalize ownership of their land. Large numbers of people and communities have gained responsibility for the management of their land and livelihoods (Ujamaa Community Resource Team, 2010, 2011, 2015; Siandei, 2016; Nelson & Makko, 2005; UNDP, 2012; Katiba Initiative, 2012; Ardhi, 2013).

1.4.8.4 Evaluation of success

Table 1 9 Scoring for all factors (Table 1.1) across the 3 overarching criteria (Figure 1.2) of the operational framework, both pre- and post-restoration, to evaluate the success of the project, scored against the coloured scoring values (-1 to 5) (Box 1.2).

1. Guiding instruments		Coord.*	1.1	1.2	1.2.1	1.3	1.4	1.4.1	1.5	1.5.1	1.6	1.7
2. Nature's contributions to people		2.1	2.2	2.2.1	2.2.2	2.2.3	2.3	2.3.1	2.3.2	2.3.3	2.4	2.5
3. Biophysical conditions		LD**	3.1	3.2	3.3	3.3.1	3.4	3.5	3.6	3.7		
Guiding instruments	Pre	-1	-1	1	-1	-1	-1	-1	-1	-1	-1	-1
	Post	4	5	5	4	4	4	5	4	4	4	5
Nature's contributions to people	Pre	-1	-1	-1	-1	-1	-1	-1	-1	-1	-1	-1
	Post	4	5	4	4	4	4	4	4	5	4	4
Biophysical conditions	Pre	-1	-1	-1	-1	-1	-1	-1	-1	-1		
	Post	4	4	4	4	4	4	4	2	5		

LEGEND

1. Guiding Instruments	48/55
2. Nature's Contributions to People	46/55
3. Biophysical Conditions	35/45
1, 2, 3 combined	% Total 82%

-1	Negative
1	Limited
2	Slight
3	Slight to moderate
4	Moderate
5	Good

* *Coord. = coordination*
** *LD = land degradation*

1.5 Conclusion

This chapter has developed an operational framework, incorporating the socio-ecological landscape approach, which may provide guidance and direction on the planning and implementation of new projects with the aim to improve human well-being and quality of life, while avoiding and reducing the impacts of land degradation and restoring and rehabilitating degraded lands. This operational framework incorporates a whole of life cycle implementation and evaluation process with the active participation of multiple stakeholders, including indigenous peoples and local communities, and businesses in order to embrace both monetary and non-monetary valuations of natural resources. Eight existing long-term cases have been evaluated against the three overarching criteria and the underlying elements of the operational framework. This approach has proven to be useful in gaining a holistic understanding of the outcomes of the eight case projects and in identifying future directions.

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Chapter 2

Concepts and perceptions of land degradation and restoration

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Executive Summary

When dominant or mainstream perceptions and concepts have an undesired impact on nature and its contributions to people, promoting alternative perceptions and concepts may transform practices towards more desired impacts (*established but incomplete*). Individual perceptions of the surrounding world are organized into concepts that vary depending on the knowledge, norms, values and beliefs of the community to which an individual belongs (Figure 2.1). These perceptions and concepts influence the way a society builds its own reality and acts on it (*well established*) {2.1, 2.2.1.2}. The dominant worldviews of a given society or community can affect, positively or negatively, nature and nature's contributions to people {2.2.2, 2.2.3, 2.2.4}. To achieve Sustainable Development Goal 15.3 of a land degradation neutral world, a shift in worldviews in necessary: from one where land degradation is seen as collateral damage or an externality of desired development, to one where land degradation to achieve development is unacceptable {2.2.1.5, 2.3.3).

Sustainable development is based on three pillars: social, environmental and economic. In its implementation, however, economic growth is often considered as the overarching driver of social and environmental progress (*well established*). Land degradation is sometimes perceived as a result of underdevelopment, while the impacts of development on land degradation tend to be disregarded (e.g., public policies supporting export crops or huge infrastructures) {Box 2.4}. For example, in 2012, 26 out of 40 Agenda 21 targets were "far from being reached" and six were in recession {2.2.4}. Among the six were "fighting global climate change" and "changing consumption patterns" {2.2.4}. Development and economic activity can also cause negative externalities and degradation {2.2.1.5}. A successful example of creating disincentives for negative externalities is the "polluter pays principle" {2.2.1.5}. Widening the scope of this principle to make it more broadly applicable to land degradation might be considered.

People are often uninformed about the undesirable environmental impacts of goods and commodities (*well established*). Raising awareness on how individual consumption choices can have unintended consequences in distant locations is a necessity (*well established*) {2.2.1.3}. Marketing disinformation about environmental impacts is a rule, not an exception {2.2.3.3, 2.3.2, 2.3.1.3, 2.3.1.4}. Trade competition externalizes social-environmental impacts to lower the prices {2.2.1.5, 2.2.3}. Internalizing the environmental costs of staple, clothes and other goods would raise public awareness, create a strong demand for low-impact products and promote more equity between people in developed and developing countries {2.2.1.5, 2.2.2.3}. Farmers and agribusiness corporations have a major role to play in inventing products and practices reflecting people's expectation for low footprint agriculture (2.2.3).

When land degradation affects cultural diversity and its associated biodiversity, not only are unique social-ecological systems threatened, but society also risks losing the local cultural knowledge that can inspire more sustainable practices (*well established*). The pervasive absorption or loss of traditional knowledge and management systems, which have proven sustainable over decades or centuries, affects cultural, biological, agricultural diversity and ecosystem services {2.2.2.1}. Land and water degradation in or around traditional territories is mainly caused by external population pressure and development programmes such as dams or monoculture {2.2.2.3, 2.3.1.1}. The precarious situation of many indigenous and local people, and their knowledge systems, is an environmental as well as a social issue. Indigenous and local practices and values are embedded in worldviews and can provide alternatives to mainstream practices. For example, indigenous and local value that link the "good life" or "Buen Vivir" {2.2.2.1} to a fulfilling social life in a non-degraded environment point to more sustainable pathways through new worldviews, such as the expansion of traditional and/or agroecological practices along with new conscious consumption patterns. These have already been adopted by growing segments of civil society around the world and could be further promoted {2.3.1.2, 2.3.2.1}.

High and rising population numbers in many parts of the world pose profound challenges for environmental sustainability in both developed and developing countries (*well established*). While human demography is predominantly seen as a matter of poverty and underdevelopment to be dealt with by increasing food production, it is nonetheless a crucial but tabooed environmental issue (*unresolved*). Successful closing of the transnational development gap and eradication of the difference in per capita consumption highlights the importance of the population size. Thus, the focus on reducing consumption might be extended to embrace an inclusive demographic policy. In 1972, the declaration of Stockholm acknowledged the environmental problems caused by overpopulation and stated that countries should control their demography without affecting basic human rights. Soon after Stockholm, however, the population problem was deemed a social and educational problem, and was addressed as an underdevelopment issue. Measures to curb population growth are available and can deliver significant and lasting environmental and social benefits. These include improved access to education, family planning and gender equality (*well established*), and improved access to social welfare to support ageing populations (*established but incomplete*). The role of subsidies that may be further stimulating population growth in more developed nations should also come under scrutiny as one of the measures to curb population growth {2.2.4.2, 2.3.1.4}.

The short-term financial costs of restoration are easy to quantify and may seem high, while the short-, medium-, and long-term effects of restoration on nature's contributions to people are less easy to perceive and value (*well established*). The benefits of avoiding and reversing land degradation are undeniable and go beyond monetary valuation (*well established*). Raising awareness of the multiple benefits of both avoiding land degradation and restoring ecosystems might justify raising the resources to achieve restoration and land degradation neutrality targets. Moreover, a more holistic approach to nature's contributions to people could embrace and meet the expectations of a part of the civil society with knowledge systems that place social-ecological harmony above other considerations. While economic valuation of ecosystem services is common, many of the nature's contributions to people have no market prices {2.2.1.3, 2.2.1.5} and are therefore undervalued, if valued at all. This practice diminishes not only the economic, but also the multiple non-monetary and intrinsic values associated with nature and nature's contributions to people, be it spiritual, cultural or ethical {2.2.2.1, 2.3.1.2}. In addition, the concrete benefits of restoration might take longer to be achieved, while the costs of restoration are rather immediate {2.2.1.3, 2.3.1}. Costs and benefits of degrading or restoring can be defined in monetary terms {2.2.1.5}, but the question is multidimensional and includes the imperative to maintain biological and cultural diversity {2.2.2.1}. Benefits will be underestimated when the concept of "good quality of life" is limited to purchasing power (*well established*) {2.2.4.3, 2.3.2, 2.3.2.2}. These benefits would be easier to perceive if the dominant systems of value focused on the good quality of life with individuals having a fulfilling social life in a non-degraded environment {2.2.2.1, 2.3}.

The international community has recognized that a collapse of ecosystem functions would not be restrained by sovereign national borders. However, decisions to address urgent environmental problems are still guided by the incremental and discretionary jurisprudence of international conventions (*well established*). Since the 1970s, international environmental law has been constantly developed and enriched to account for both the progress of science and environmental degradation. Nonetheless, global ecological deterioration, including climate change, is continuing (*well established)*. Creating a proactive, new ground for international negotiation could be a first step to facilitate reversing land degradation, from which new jurisprudence could arise. This would include overcoming the old “environment versus development” dilemma and foster cooperation policies motivated by a common interest {2.2.4.1}. “Ecological solidarity” is a promising legal principle, which could renew the perception of the links between humans and their environment {2.2.4.3}. This principle embraces three dimensions: it recognizes the planetary interconnectedness of ecosystems and ecological process {2.2.1.3}; it may foster intergovernmental negotiations based on global and mutual solidarity; and it has a fundamental moral meaning emphasizing the common fate of humankind and all living beings {2.3.1.2}. If human progress was understood through these dimensions, efforts to prevent land degradation and to restore degraded land might be facilitated.

A global consensus on the definition and baseline for land degradation does not exist (*well established*), precluding sound scientific assessment of the extent and severity of global degradation, as well as the possibility of measuring success towards quantitative restoration targets such as Aichi Biodiversity Target 15 reinforced in Sustainable Development Goal 15 (*established but incomplete*). Quantifying land degradation and its reversal through restoration requires assessment of the geographic extent and severity of damage at the current and restored state of the ecosystem, against a baseline (*well established*) {2.2.1.1}. Lack of consensus over baselines has led to debates over what constitutes degradation and subsequently to inconsistent estimates of the extent and severity of land degradation {2.2.1.2} (Figure 2.5, Figure 2.7, Figure 2.8). This, in its turn, resulted in differing interpretations of the consequences of degradation for human well-being. To overcome this challenge, a shared global baseline could be adopted (*well established*) and a good candidate would be the natural state of ecosystems, deviation from which would be degradation {2.2.1.1} (Figure 2.5) (*established but incomplete*). Adopting natural state of ecosystems as the baseline against which to measure the extent and severity of degradation ensures a comparable assessment of land degradation in general, and a fair assessment of success in meeting the Aichi Biodiversity Targets across countries at different stages of economic and social development. Without this, more developed countries – that have transformed much of their environment centuries ago – are able, in practice, to assume much less ambitious restoration measures than less developed countries {2.2.1.1} (Figure 2.5}. For the aspiration to achieve land degradation neutrality by 2030, as agreed in SDG 15.3, the baseline for assessing success is different, namely the state of the ecosystems at 2030.

2.1 Introduction

Diverse perceptions, concepts and worldviews serve to shape one's affinity to the land. This affinity is generally shared by the society to which an individual belongs. Because societies are diverse, arriving at consensus about the state of land degradation and the need for restoration is never easy, especially when restoration does not create immediate economic profit. Summarizing the viewpoints of even a small range of stakeholders highlights the complexity of the perceptions and concepts that influence the practice of decision-making.

The purpose of Chapter 2 is to examine the concepts used by different stakeholders, assess how perceptions and concepts lead to degradation and suggest changes in policy that could help avoid degradation and facilitate restoration.

There are two ways to define concepts. The first is concepts as tools, to understand and organize the world. The second is concepts as social constructs, whose importance, validity and use vary across time and space. For instance, the concept of "race" was crucial in the nineteenth century to understand human variability, and led to scientific racism and colonization. Hence, the way a concept is understood and used can have a strong impact on social organization, geopolitics and environmental management.

This chapter, as other chapters in this assessment, was written by both natural and social scientists. Social sciences such as history, philosophy, legal or political science or anthropology do not obey the same regime of proof as natural sciences, such as ecology, biology or genetics. Many social facts and representations – including worldviews – cannot be quantified as "well established". Only a qualitative approach, then, can underline their importance and validity.

2.1.1 Organization of the chapter

Following the scoping document accepted by the Plenary of IPBES at its third session (IPBES-3) in January 2015, this chapter follows the structure as outlined in the scoping document (Annex VIII to decision IPBES-3/1) and consists of two main parts.

Section 2.2 is dedicated to perceptions, concepts and approaches to land degradation and restoration from different stakeholders' points of view. Cross-disciplinary concepts are explored throughout this section, such as the use of baseline as a tool to assess degradation and evaluate restoration success, and perceptions of these concepts by scientists, jurists, indigenous and local peoples, NGO managers, conventional farmers, agribusiness actors and decision-makers.

Section 2.3 explains why the impacts of land degradation on nature's contributions to people and human well-being are frequently difficult to perceive and how this can affect the decision-making process. We provide an overview of several obstacles to people's awareness, including "fuzzy concepts", but also underline people's collective reaction and eagerness to be involved in the development of environmental policing. We then examine how, in spite of these obstacles, awareness-raising may elicit public reactions, especially when policymakers' reaction appears to be too slow in the eyes of other stakeholders. The capacity of civil society (including NGOs) to propose alternative policies or practices is a powerful instrument to contribute to decisions at all political scales. It is also the main reason for being optimistic about our capacity, as citizens and human beings, to avoid and reverse environmental degradation.

2.1.2 What do we mean by perceptions, concepts, and worldviews?

In this section, we are not only dealing with facts, but also with cognitive (i.e., mental) processes that feed into worldviews, and specifically how these worldviews have affected and still affect current land degradation. Worldviews are reflected in practices and more generally in day-to-day attitudes and actions. Hence, a global effort to avoid or mitigate land degradation and to rehabilitate and restore degraded lands can be fostered by considering other worldviews and the related concepts and perceptions. We adopt a four-step explanation process to be as clear as possible in this chapter:

1) Presentation of definitions of reality, perceptions, concepts, worldviews and human well-being.
2) An illustration of cognitive processes as embedded in worldviews and reality (Figure 2.1).
3) A practical illustration of these cognitive processes, through a very simple example of divergence among actors' perceptions (Figure 2.2).
4) The IPBES Conceptual Framework and how this chapter is embedded into it (Figure 2.3).

2.1.2.1 Definitions for the purpose of this chapter

The cognitive processes synthesized in Figure 2.1 are based on Damasio (1994), Laplane (2005), Norman (1988), and Pinker (1999). For the purpose of this chapter, the "reality" we refer to is the current state of biodiversity and ecosystem functions independent of human knowledge and perceptions and ecosystem services ("nature" in IPBES conceptual framework, Figure 2.3). Hereafter we will use "nature" as synonymous with this reality. Dealing with perceptions and concepts means that the focus is on what is perceived by humans about nature and nature's contributions to people. This human-centred view has been adopted at the second session of IPBES Plenary (IPBES-2).

Perceptions are the first stage of the human cognitive process. We can see a global picture of the reality, but we perceive what we focus on. What we see results from a neurological processing of the stimuli in our environment, while our perceptions are not neutral as they pass through rational and emotional filters which assess and interpret the relevance of what we see. These filters are conditioned by individual experience, education and by collective worldviews (Dickman *et al.*, 2013).

Concepts are defined as the second stage of the cognitive process. Perceptions are selected, organized, classified and hierarchized into concepts. This process is influenced by collective filters which are human systems of values, norms and beliefs. Concepts do not come alone, but as integrated networks. This is the reason why there is often a mismatch between environmental risk assessments, scientific alerts and pre-existing categories and beliefs in public opinion (Fischhoff *et al.*, 1992; Wallner *et al.*, 2003).

Worldviews are defined by the connections between networks of concepts and systems of knowledge, values, norms and beliefs. Individual worldviews are moulded by the community the person belongs to, which also applies to the scientific community. This is what we mean by a collective filter. To give a very simplified example, a Catholic will assign to a cross a symbolic dimension while a Siberian shaman will perceive it as a mere geometrical form. Practices are embedded in worldviews and are intrinsically part of them (e.g., through rituals, institutional regimes, social organization, but also in environmental policies, in development choices, etc.).

Human well-being (see Glossary) will be here considered in its relation with ecosystem services (Agarwala *et al.*, 2014). Land degradation and restoration have a direct and indirect influence on the quality of life and on human well-being. Once acknowledged, these impacts may modify perceptions, reorder concepts, change worldviews and thus foster new policies and practices.

Perceptions can be used as instruments to reorient policies by creating new concepts about land degradation and restoration and how they affect human well-being. Can we change priorities or increase awareness so that perceptions correspond to reality and evolve accordingly? The goal is to formulate different approaches to land degradation and restoration to minimize environmental impacts, which will have a more positive effect on human well-being for all members of society.

Figure 2 1 A conceptual illustration about how perceptions and concepts are articulated and how they interact with reality ("world" or "nature").

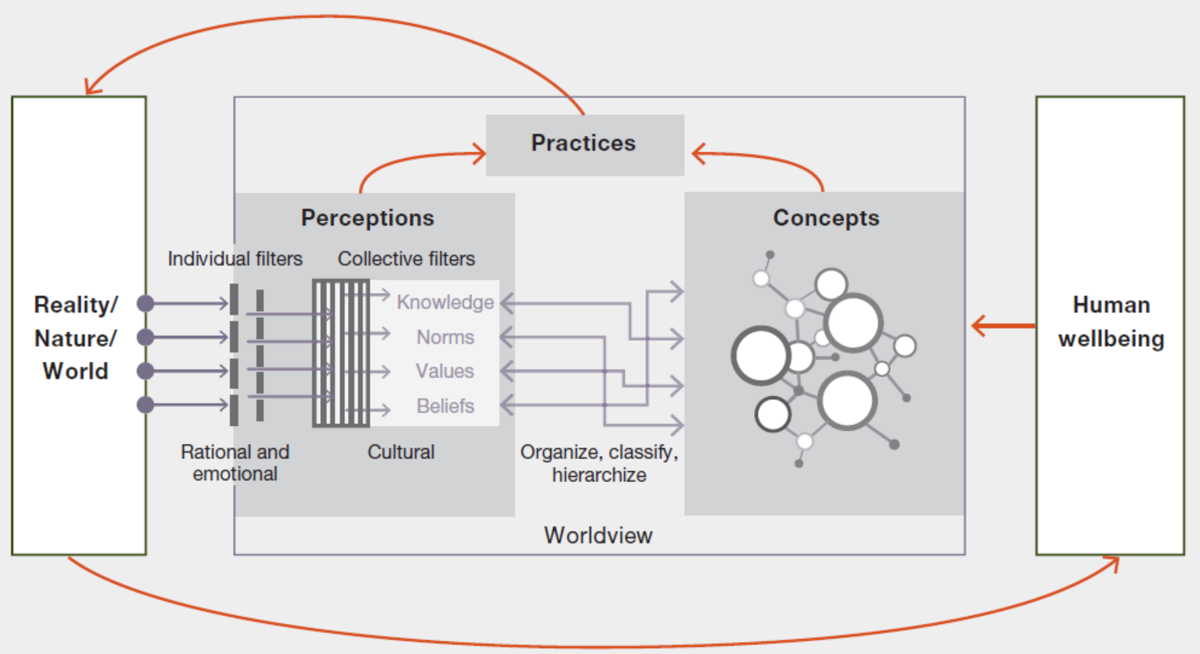


Figure 2 2 Practical illustration of how seeing the same reality leads to different perceptions embedded in different sets of concepts.

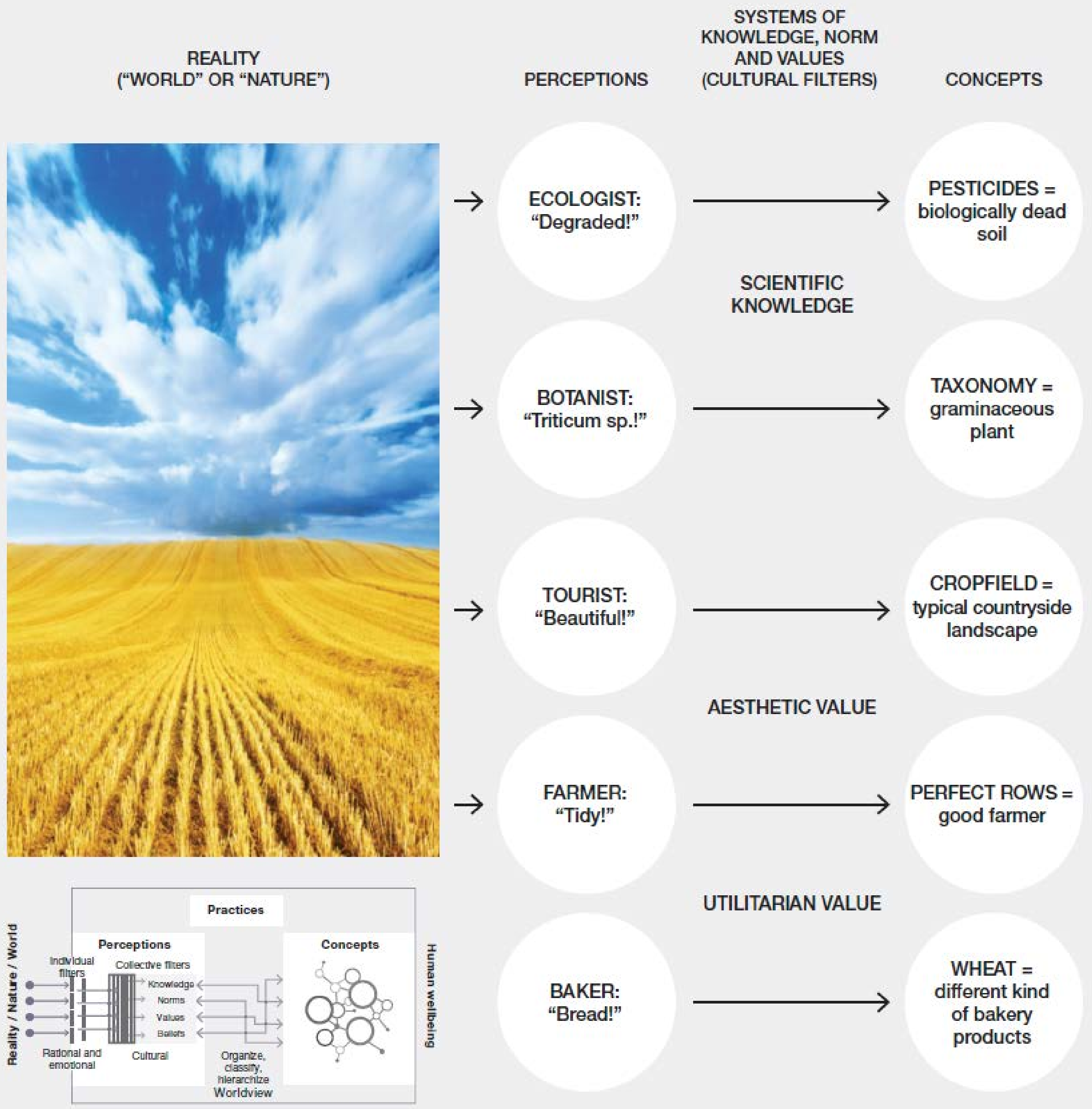
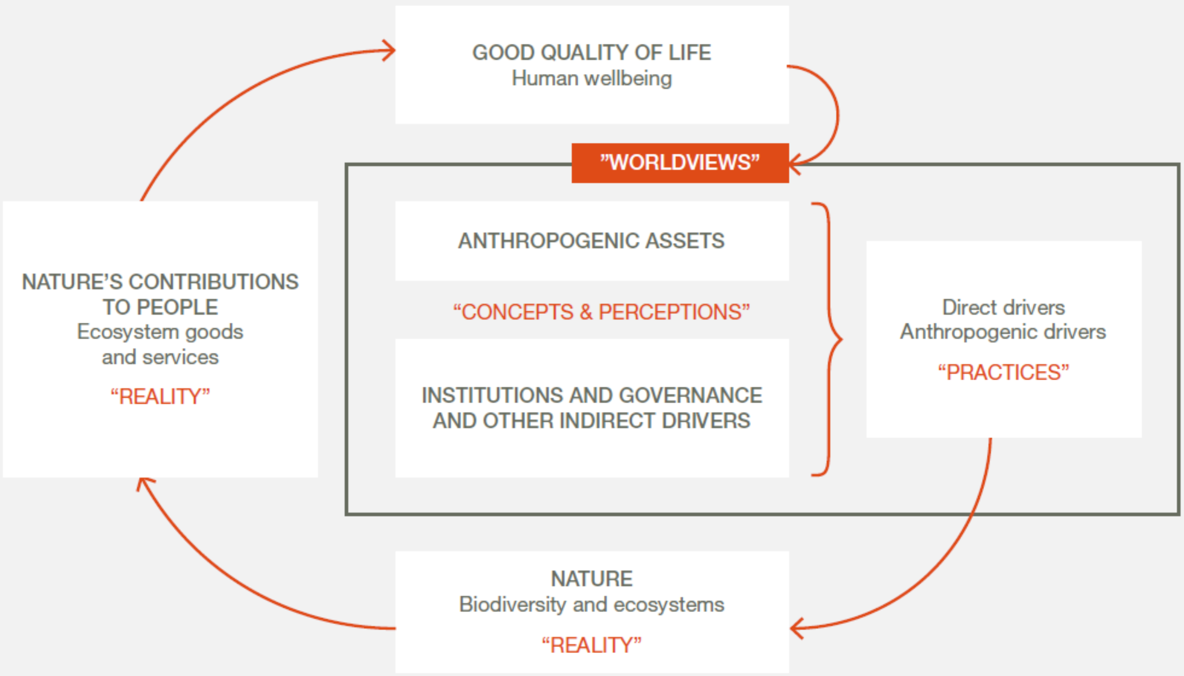


Figure 2 3 Chapter 2 (in red) as included in IPBES Conceptual Framework. Source: Modified from Díaz *et al.* (2015).



2.2 Perceiving and conceptualizing the reality of land degradation and opportunities for restoration

Vogt *et al.* (2011) identified several groups of actors that have different needs in terms of type and frequency of information related to land degradation and different capability for response: (i) the policymakers organized at different spatial scales (e.g., local, national, supra-national, global); (ii) land owners, users and managers (i.e., those interacting directly with the land and responding to the policies defined by the first group); (iii) the scientific community that both needs and produces information; (iv) the development community and NGOs, particularly in the case of desertification; (v) society at large, which relies on information for financial and public/political support; and (vi) the media, which translates and distributes the information to other groups. It is thus crucial to properly assess and understand the role and responsibilities of each of those different groups if deep changes in societal efforts – to avoid or mitigate land degradation and to rehabilitate and restore degraded lands – are to be successful (Vogt *et al.*, 2011).

This subchapter discusses the concepts and perceptions by grouping the six sets of actors above into four broader stakeholder groups: (i) scientists and jurists; (ii) indigenous groups and local populations; (iii) farmers and agribusiness companies; and (iv) decision makers, from national to international levels (civil society as a stakeholder and an actor will be considered in Section 2.3). In 2.2.1, we focus on the most important concepts developed by scientists to assess the status and responses of biodiversity and ecosystem functions and services to degradation and restoration processes. At the same time, Section 2.2.1 also attempts to show how the law and economics perceive and address these concepts by turning them into legal principles.

2.2.1 Ecological knowledge to assess degradation, facilitate restoration and inform legal and economical responses

The goal of the natural sciences is primarily to describe and understand the environment we live in and how people affect that environment, while the focus of humanities and social sciences is more on human societies, including their interactions with the environment (Sessions, 1987). The scientific approach, unlike others, is based on: observable, testable and measurable facts; evidence; transparency of the data and results; the peer-reviewed process; and is open to contradiction and further investigation, thanks to the accessibility of the data. In this section, we identify the most important concepts that natural scientists use to assess the status and responses of biodiversity and ecosystem functions and services. It should be noted that scientific concepts evolve with time, some of them appearing or disappearing according to the context and their practical value. For instance, “ecosystem services”, which appeared in the 1980s, is widely used today (Chaudhary *et al.*, 2015). Science is a dynamic process and perpetually creates conceptual tools adapted to new or newly discovered realities (Kuhn, 1962).

We also consider how law and economics perceive these scientific concepts and discuss the most important additional concepts that these disciplines recognize and use. This is important because law and economics, among other social sciences, have offered central support to the analysis and formulation of land-use policies and instruments. Regarding their purposes, they can be a driver of land degradation (see Chapter 3) and a response to enhance restoration measures (see Chapter 6). This section attempts to demonstrate a gap between ecological concepts and their legal translation, which may lead to the perception that the land is not degraded.

2.2.1.1 The significance of baselines in assessing degradation and restoration

For the assessment at hand, the definitions of degraded land and restoration were provided by the IPBES Plenary (IPBES, 2015) and are fully described in Chapter 1 (based on Annex VIII to decision IPBES-3/1). Here we recap the essential sections of the definitions to aid understanding of the below discussion. “Degraded land” is defined as the state of land which results from the persistent decline or loss in biodiversity and ecosystem functions and services that cannot fully recover unaided within decadal time scales. “Restoration” is defined as any intentional activity that initiates or accelerates the recovery of an ecosystem from a degraded state. “Rehabilitation” is used to refer to restoration activities that may fall short of fully restoring the biotic community to its pre-degradation state. Taken together these definitions mean that the concept of restoration refers to interventions whose intended outcome is full recovery of the ecosystem to its pre-degradation state, while rehabilitation has the intended outcome of partial recovery of the ecosystem. Inability to recover unaided is caused by: (i) crossing an ecological tipping point to a new state or regime, such that the ecosystem is unable to recover on its own within decadal time scales (see Chapter 4, Section 4.1.2); or (ii) business-as-usual land-use management that prevents an ecosystem from recovering unless aided by an alteration or cessation of the management.

Based on these definitions, any ecosystem that has experienced loss in biodiversity or ecosystem functions and services is considered degraded, provided it cannot fully recover unaided within decadal time scales. To understand if the “unaided” and “decadal” criteria can be met even from the perspective of biodiversity alone, a mechanistic understanding of succession and species community assembly processes is needed. There are only four mechanisms that can influence community composition as a result of community assembly processes: selection, drift, dispersal and speciation (Chase, 2010; Chase & Myers, 2011; Elo *et al.*, 2016; Gilbert & Lechowicz, 2004; Hubbell, 2001; Kahilainen *et al.*, 2014; Tuomisto *et al.*, 2003; Vellend, 2010). Unfortunately, assessing ecosystem degradation and recovery at the global scale, with a level of detail needed for the mechanistic understanding, is not feasible. Moreover, this only concerns biodiversity and community composition; the recovery of ecosystem functions or ecosystem services must be understood at the same level of detail (see also Skidmore & Pettorelli, 2015). Thus, degraded land might be better understood simply as land that has experienced a decline or loss of biodiversity and ecosystem functions and services – without a reference to the ability of the land to recover unaided (within decadal time scales). In this definition, the pre-degradation natural state can be understood as the state of land prior to the decline or loss of biodiversity or ecosystem functions and services. It is worth noting that regardless of the definition of degradation, one needs to be explicit regarding whether one is talking about degradation in terms of loss of biodiversity, loss of ecosystem function and/or loss of ecosystem services as there can be trade-offs amongst them (e.g. Bennett *et al.*, 2009; McShane *et al.*, 2011; Schröter *et al.*, 2014; Spake *et al.*, 2017).

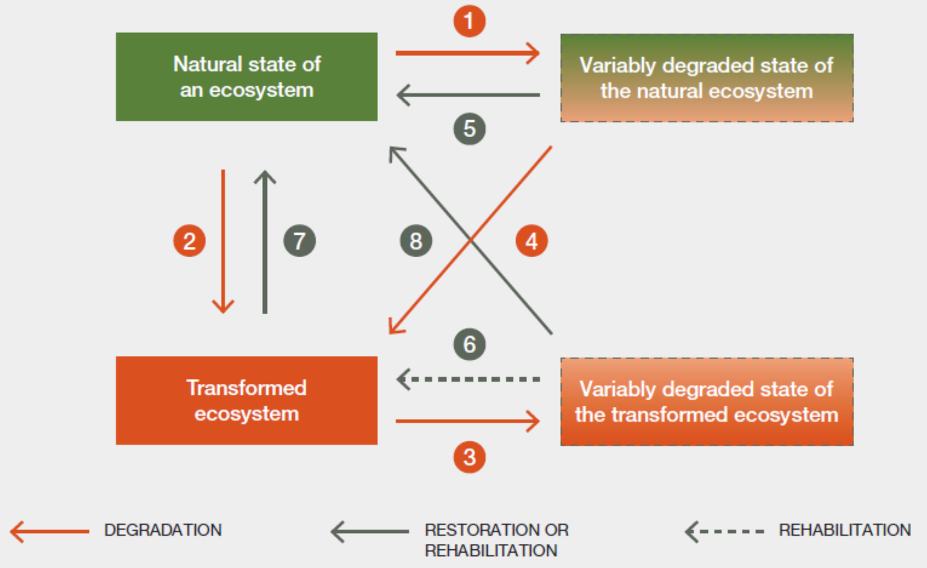
Since the IPBES Plenary, at its third session (IPBES, 2015), adopted the use of pre-degradation state in the definitions of restoration and rehabilitation, the above definition of the pre-degradation state is an important guiding principle. In general, to obtain a genuine estimate of the magnitude of damage or recovery, the choice of a reference frame or a baseline is of critical importance (Bull *et al.*, 2014; Kotiaho *et al.*, 2016a, 2016b; McDonald-Madden *et al.*, 2009; Prince, 2016; UNEP, 2003) (See also Chapter 4, Section 4.1.2).

While in practice it appears to be difficult to reach an agreement on a perfect pre-degradation reference state or a baseline against which the degree of damage should be compared, in theory, we can come close to one (Kotiaho *et al.*, 2016a). The question of "how much damage has humankind caused on ecosystems?" contains an inherent, natural baseline, which is the state in which there was no damage caused by humankind (i.e., the pre-degradation state). This question should not be confused with the question about whether humans are part of nature or not (Haila *et al.*, 1997; Hunter, 1996), as we are one species among others. Rather, it is about our desire to restore the ecosystems we have damaged, as has been firmly established in a number of international conventions. The selected reference state or baseline will always influence the assessment of the magnitude of damage (see also Section 2.2.1.2) and this becomes vitally important when we set quantitative targets for restoration – such as the Aichi Biodiversity Target 15 that aims to restore at least 15% of degraded ecosystems globally, by 2020 (CBD, 2011; Kotiaho, 2015; Kotiaho *et al.*, 2016a, 2016b; Kotiaho & Moilanen, 2015).

When considering the quantitative restoration target it is worth noting that degradation has at least two dimensions: the extent of area that has been degraded and the magnitude or severity of degradation (or loss of condition) within that area (Kotiaho *et al.*, 2015; Kotiaho & Moilanen, 2015; Nkonya *et al.*, 2016). In addition, currently well over 50% of natural terrestrial ecosystems have been transformed to other ecosystems (Ellis *et al.*, 2010; Hooke & Martín-Duque, 2012; Houghton, 1994; Vitousek *et al.*, 1997). Transformation of natural ecosystems causes loss of ecosystem area and is degradation from the perspective of the original natural ecosystem (Figure 2.4). The impact of degradation on biodiversity, ecosystem functions and nature's contributions to people are very different for ecosystems with little loss of condition compared with those where condition has severely declined or been transformed.

Figure 2 4 **Land degradation can occur either through a loss of biodiversity, ecosystem functions or services, without a change in land cover class or use (1), or by the transformation to a derived ecosystem type such as the conversion of natural cover to a crop field (2), delivering a different spectrum of benefits, but also typically involving loss of biodiversity and reduction of some ecosystem functions and services.**

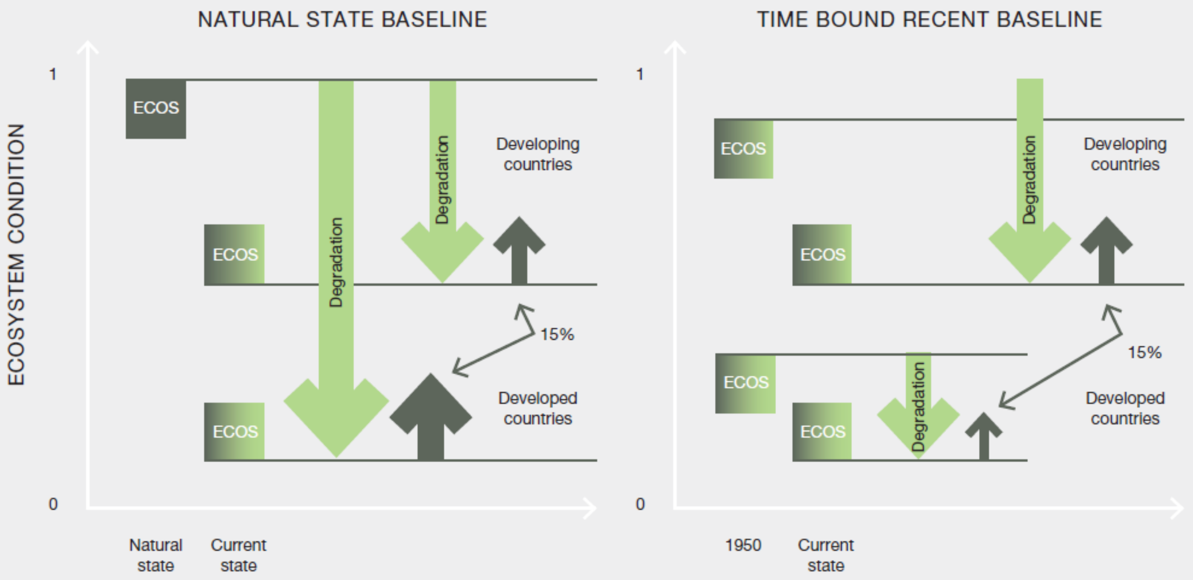
The transformed ecosystem can also be degraded with respect to the new societal expectations associated with that land use (3). Degraded natural ecosystems can also be transformed to another ecosystem (4), or restored towards their original natural state, either completely or partially ("rehabilitated") (5). Degraded transformed ecosystems can be rehabilitated towards a less degraded state, with respect to the expectation for a deliberately modified landscape (6). Both degraded and undegraded transformed lands can, under many circumstances, be restored or rehabilitated towards their original natural state (7 and 8). Success in achieving the aspirational goal of land degradation neutrality by 2030 in Sustainable Development Goal 15 may be measured based on whether biodiversity, ecosystem functions and services are stable or increasing in each of the focal ecosystems compared to their state in 2015.



For the purpose of assessing anthropogenic ecosystem degradation, an obvious reference is the natural state without any human modification. Establishing the natural state for an ecosystem is challenging and some of the approaches are described in Box 2.1. Despite the challenges, when the goal is to estimate global and regional magnitudes of degradation, like in the current IPBES work programme, global geographic variation in the timing of economic and social development, and ecosystem degradation, makes a strong case for the adoption of the natural state baseline as a reference. To illustrate the point, let us consider the state of ecosystems in some recent past as a baseline. If we assess degradation against a recent time-bound baseline (e.g., 1950 in Figure 2.5), developed countries will show low degradation since they degraded much of their land before 1950. On the other hand, developing countries will show high degradation since they started to transform their environment more recently. In this case, the 15% restoration target for developed countries will require less restoration than the same target for developing countries, and thus is not equitable. By contrast, the concept of natural state baseline is independent from variations in the time of development of countries, and therefore it will provide a fair baseline for comparisons among countries at different stages of socio-economic development. When using natural state baseline, absolute degradation is reported to be greater in the most developed countries and smaller in the least developed countries, and the 15% restoration target for developed countries fairly involves more actual restoration than the same target for developing countries (Figure 2.5). It is worth mentioning that to achieve land degradation neutrality by 2030 as aspired in SDG 15.3, the baseline for assessing success is different – namely, the state of the ecosystems at 2030.

Figure 2 5 **How the choice of a baseline influences the effort required to reach the Aichi Biodiversity Target 15 of restoring 15% of degraded ecosystems in developing and developed countries.**

Magnitude of ecosystem degradation is the difference between the current state and the baseline (green downward arrows). On the left, the current state of ecosystems is compared to the natural state baseline and the magnitude of degradation and thus restoration effort (grey upward arrows) required from the developed countries is greater compared to the developing countries. On the right, a recent 1950 time-bound baseline is used. Due to different timing of development, and thus degradation, the restoration effort required from developed countries is less compared to the developing countries.



Ecosystem services are not a biological phenomenon, but they are, by definition, the ecosystem attributes that humans value (MA, 2005b), and that trade-offs between them and biodiversity exist (McShane *et al.*, 2011; Schröter *et al.*, 2014; Spake *et al.*, 2017). Anthropogenic decrease or increase of the service may cause degradation of the ecosystem and therefore, while securing valuable ecosystem services, care must be taken to avoid levels of degradation which may compromise biodiversity, ecosystem functions or less valued ecosystem services (Bennett *et al.*, 2009).

Finally, the pre-degradation natural state baseline should not be confused with the goal or target of restoration or rehabilitation. A pre-degradation state baseline is necessary for assessing the magnitude of damage, and while the target should be directed towards the pre-degradation state baseline, the pre-degradation state itself need not be the target. In practice, the target will often be only partial rehabilitation towards the pre-degradation state (see also Kotiaho *et al.*, 2015, 2016a, 2016b).

It is worth noting however, that arguments have been put forward that interventions may aim at replacement of the natural state ecosystem with a different system (Bradshaw, 1984). Today replacements are called novel ecosystems (Hobbs *et al.*, 2006; Hobbs *et al.*, 2009, 2013). However, interventions that aim at replacement, or novel ecosystems, should not be regarded as restoration or rehabilitation *sensu* IPBES (IPBES, 2015). Instead, this debated concept (e.g. Hobbs *et al.*, 2014; Murcia *et al.*, 2014) should be referred to as maintaining, and sometimes fostering, of alterations which nevertheless have resulted in self-sustained ecosystems (Hobbs *et al.*, 2009; Perring *et al.*, 2013).

Box 2.1 Approaches to baselines and targets

This Box enlarges on Box 1.1 in Chapter 1, and further information can also be found in Chapter 4, Section 4.1.2. A reference or baseline is essential to detect and assess the magnitude and direction of degradation (Prince, 2016; UNEP, 2003). Thus, an unambiguous implementation of the concepts of land degradation and restoration requires asking “degraded relative to what?” and “restored towards what?” Furthermore, both degradation and restoration refer to change over time and establishing the magnitude of change requires information at two or more times, or by inference, between two or more places thought to be initially the same (see Section 2.2.1.4).

There is no perfect reference state or baseline for all purposes, but allowing free selection of a reference state increases the possibility of deliberate bias and arguments. Nevertheless, for the purpose of assessing anthropogenic ecosystem degradation, an obvious reference is the natural state without any human modification. Establishing natural state for an ecosystem is challenging but there are at least two approaches that can be used, **time bound** and **counterfactual** natural state. Other reference states that have been used include various time bound **historical** baselines. Finally, while a reference is necessary for assessing the magnitude of degradation, it should not be confused with a **target**. Targets are always a matter of political choice – weighing societal, economic and ecological interests – and will vary case by case (Kotiaho *et al.*, 2016a). For further discussion about baselines and targets see main text in Section 2.2.1.1.

1. Time bound natural state baseline. Natural state can be understood as the ecosystem condition before degradation by human activities – that could be some time in the Holocene, ≤10,000 yr BP. This seems to be an obvious baseline from which to assess degradation and recovery, since it is before any human modification, but it is riddled with practical and theoretical issues. Practically, it is rare to find data from such distant past that includes all the variables needed to draw a comparison with current ecosystem conditions (Broothaerts *et al.*, 2014; Hoffmann, Erkens *et al.*, 2009; Vanacker *et al.*, 2014). There are also at least two conceptual challenges with the time bound natural state baseline. First, the climate and other biophysical environmental conditions have changed in the intervening time (from the baseline to present day) and it is difficult to disentangle the effect of anthropogenic degradation from natural environmental change (Bennion *et al.*, 2011). The second challenge arises from the fact that some degree of disturbance by humans is part of the evolutionary history of many current organisms, and such potentially cascading ecological changes are challenging to identify or take into account (Jackson & Hobbs, 2009).

***2. Counterfactual natural state baseline*.** Another perhaps more operational approach for establishing the natural state baseline is the use of counterfactual thinking. In psychology, counterfactual thinking is a mental representation of alternatives to past events and it can be characterized by the phrase “what might have been” (Byrne, 2007; Epstude & Roese, 2008; Roese & Olson, 1997). Thinking about alternatives to our own pasts is central to human thinking and emotion (Epstude & Roese, 2008; Sanna *et al.*, 2003; Summerville & Roese, 2008; Wheeler & Miyake, 1992) and common across nations and cultures (Au, 1983; Gilovich *et al.*, 1985). Therefore, it may be a globally functional and understandable approach for establishing the natural state baseline for an assessment of the magnitude of degradation in a given ecosystem.

By asking what the environment would have looked like in the absence of the intervention or development, counterfactual thinking can be used and has been used in environmental impact scenario-modelling and in environmental impact evaluations for establishing references for the current state

(Caplow *et al.*, 2011; Davis *et al.*, 2011; Ferraro, 2009). Although the approach has been rare in the environmental literature (Ferraro, 2009), the number of cases where it has been successfully applied to questions relevant to land degradation and restoration is increasing (e.g., Andam *et al.*, 2008; Joppa & Pfaff, 2011; Kotiaho *et al.*, 2016b; Robinson *et al.*, 2014; Urama, 2005). For example, Andam *et al.* (2008) estimated the effectiveness of conservation areas of Costa Rica, in preventing deforestation, by finding an answer to the question: how much more forest would have been cleared if the protected areas had not been established? In another example, Kotiaho *et al.* (2015, 2016b) assessed the magnitude of degradation across all terrestrial ecosystems of Finland by comparing the current state of the ecosystems to the state that would have existed had humans not disturbed the ecosystems. In the latter case, the counterfactual state is the natural state and functioned as the natural state baseline for measuring anthropogenic ecosystem degradation. The counterfactual natural state baseline does not suffer from the natural change challenge, but the availability of data or expertise can still be an issue. In addition, a method known as space-for-time substitution (Johnson & Miyanishi, 2008; Pickett *et al.*, 1998) or process-based modelling (Bowker *et al.*, 2006) can provide a reference approximating the time independent natural state (see Section 2.2.1.4).

3. Time bound historical baselines. Unlike a natural state baseline, time bound historical baselines may have suffered some degradation and thus provide underestimates of actual degradation. On the other hand, when the more recent past is chosen as the historical baseline, more data is available. Various historical baselines are used for trend studies (e.g. Bakker *et al.*, 1996; Keith *et al.*, 2013), however, they often suffer from arbitrary starting dates which makes comparisons difficult.

More recent historical baselines are useful for detecting contemporary past and future trends in biodiversity, ecosystem functions and nature’s contributions to people – in particular, when we are interested in impacts of policy or management changes, such as the land degradation neutrality target of the Sustainable Development Goal 15, for which the baseline will be the state of the ecosystems in 2030. Assessing deviations from the natural state would function equally well for this purpose, but as stated above, an estimated “natural state” can be more laborious to establish.

A distinct discontinuity exists in the degree and type of disturbance around the onset of the modern era, about two-three centuries ago around 1750-1850. This “pre-modern Holocene”, before the “great acceleration” reference state, is not easily manipulated and many examples show it to be implementable, though not without its challenges (e.g. Bennion *et al.*, 2011; Jenkins *et al.*, 1990; Keith *et al.*, 2013; Naudts *et al.*, 2016). The same challenges as with the time bound natural state exist, but are generally not as problematic.

4. Target. A target is the desired state – in this case, for the purposes of restoration. A reference or baseline is needed to assess the magnitude of degradation and should ultimately be based on scientific research, while the target is based on a deliberate choice and is therefore context dependent. The target may change over time and will certainly vary from place to place. The target state need not be universal, unless so agreed. It is perhaps the most important of the states for policy purposes, since it represents the future and thus a state whose achievement can be influenced by policy.

A target state of an ecosystem can be derived from the perspective of biodiversity (as is most often the case in ecological restoration) or it can be considered from the perspective of nature’s contributions to people or ecosystem services. Nature’s contributions to people (or ecosystem services) are goods and services valued by human beings. They are a measure of human preference, which is similar to the “utilitarian” concept of the Millennium Ecosystem Assessment (MA, 2005a).

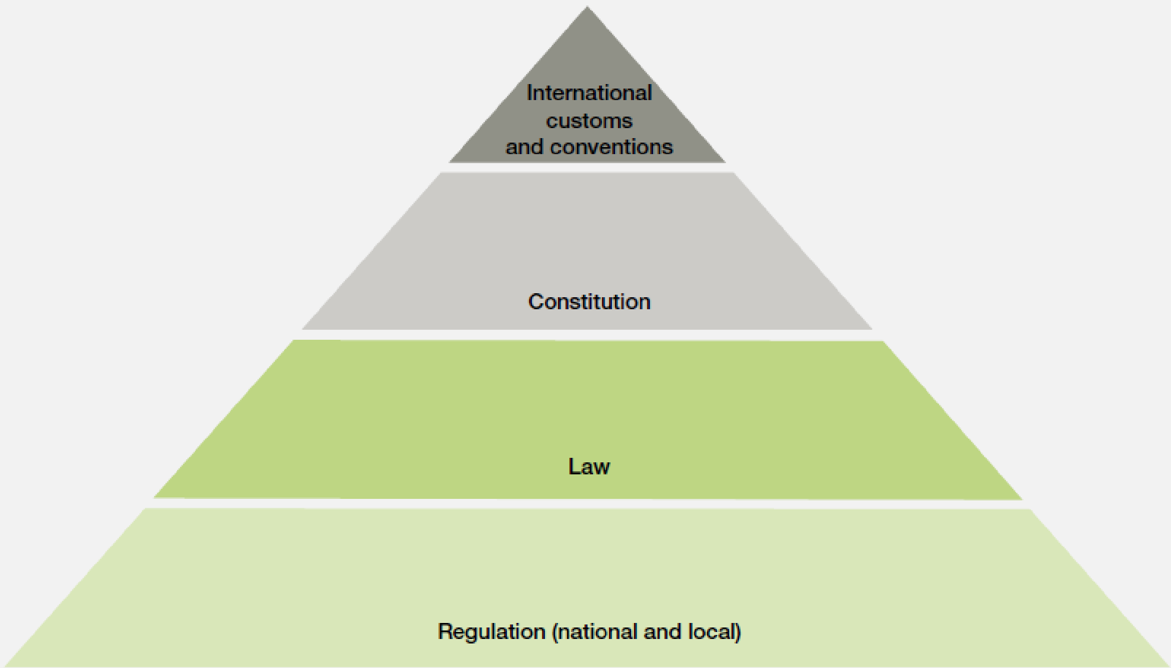
The concept of baseline in the law

The concept of baseline is central also to the law, as impacts and damages are estimated relative to a reference state. Judges need a baseline to quantify the compensation measures and the law usually provides a definition of the baseline. This baseline can either converge or diverge from its ecological definition, even though ecological concepts are more and more integrated into environmental law (Naim-Gesbert, 1999) and tend to guide restoration and rehabilitation measures.

For example, in the European environmental liability regime, the "baseline condition" is the condition of the land immediately prior to the observed degradation, based on the best information available (Directive on Environmental Liability, 2004). In law, the baseline condition is often simultaneously the target of restoration after damage, which makes it different from the assessment and restoration of land degradation discussed above.

According to Kelsen (1960), a "hierarchy of norms" (Figure 2.6) organizes the legal order. It is designed by order of importance. Considering states' organization, the value of international law varies, but generally, international public law constitutes the supreme legal order insofar as the Constitution is modified to adapt to new international treaties. Consequently, if a definition of a baseline condition was given by an international convention, it could be ratified and integrated in national legal orders by the state parties.

Figure 2 6 The hierarchy of norms in internal legal orders. Source: Kelsen (1960).



Another interesting tool dealing with the concept of baseline is the Environmental Impact Assessment (EIA). It describes a "process that produces a written statement to be used to guide decision making" (Sands & Peel, 2012) and is meant to determine the state of ecosystems before plans, programmes or projects. In this context, unlike Box 2.1, the baseline will be the target of rehabilitation measures once the activity stops. In this chapter, we do not mention the several functions of Environmental Impact

Assessment as a tool, but we question its ability to mitigate land degradation and facilitate restoration. Indeed, the written statements of Environmental Impact Assessment rely on the perception of their authors and on the control made by public authorities. Hence, the main question is "what is being assessed?". As many forms of land degradation are not perceived by the law as degradation *sensu stricto*, most of the impacts on land are not considered in these assessments. In other words, if the law does not perceive the land as degraded, there cannot be a legal obligation to restore (Boer & Hannam, 2004; Wyatt, 2008). Our point here is to demonstrate that a common understanding of land degradation in international environmental law, for national impacts and transboundary impacts, would guide the elaboration of Environmental Impact Assessment, acknowledging that it is also an international tool (e.g., Nordic Environmental Protection Convention of 1974), although many of the conventions that mention it are non-binding (e.g., Principle 17 of Rio Declaration of 1992) (Castillo & Bian, 2014). However, the definition of the concept of land degradation in an international convention would have to overcome a severe obstacle made by the International Court of Justice. In the Pulp Mills case (Argentina v. Uruguay, 2010) the Court stated that international law does not “specify the scope and content of an Environmental Impact Assessment and that it is for each state to determine in its domestic legislation or in the authorization process for the project, the specific content of the Environmental Impact Assessment required in each case” (Johnstone, 2014).

With regards to waste management, industrial activities or polluted sites, legal frameworks and regulations aim to remediate (see glossary) contaminated or impacted land to levels where introduced contaminants do not impact the future use of the land in question (Layard, 2004; Carella & Chiappini, 1995; Jahiel, 1998; Mu *et al.*, 2014; Seerden & Deketelaere, 2000). This perspective is generally considered unambitious on its own as the objective is not ecological restoration (Billet, 2014; Brandon, 2013; Lambert, 2014; Zhao & Zhang, 2013). Furthermore, operation of controls by sworn agents on the exploitation sites needs to be enforced (Bryant & Akers, 1999; Cho, 1999; Mu *et al.*, 2014). Belgian law is particularly interesting in this aspect, because Wallonia, the Flemish Region, and Brussel’s Region have separately adopted very detailed regulations that set standards of remediation. The remediation standards are the strictest for “green” forms of land use (e.g., nature and woodland) and the most tolerant for industrial uses of land (e.g., industrial area, area for waste disposal). However, for groundwater the law carries a harmonized remediation standard (see also Conference of the European Union Forum of Judges for the Environment, 2009).

Finally, the impacts which cannot be avoided or mitigated can, as a last resort, eventually be offset. The land degradation neutrality programme of the United Nations Convention to Combat Desertification (UNCCD) was set up to implement Sustainable Development Goal 15 (Target 15.3), namely to "protect, restore and promote sustainable use of terrestrial ecosystems, sustainably manage forests, combat desertification, and halt and reverse land degradation and halt biodiversity loss". More specifically, it states: “by 2030, combat desertification, restore degraded land and soil, including land affected by desertification, drought and floods, and strive to achieve a land degradation-neutral world”.

While the Sustainable Development Goal Target 15.3 is an international goal, the UNCCD’s programme currently supports land degradation neutrality at national levels. Land degradation neutrality needs territorial boundaries or to be led by the concept of ecological equivalence to be fully efficient. In fact, it is worth noting that under the Land Degradation Neutrality Target Setting Programme (LDN TSP), an overarching Conceptual Framework has been established and neutrality indicators were introduced by the UNCCD and its Global Mechanism for baseline and target-setting, using a combination of land cover type, net primary productivity level and soil organic carbon level. Neutrality is a new concept to the law and no frame has been developed yet. Hence, neutrality should only be considered sufficient when the impacts on a degraded land are compensated by the restoration of an equivalent and close land. We suggest taking into consideration the French policy on compensation measures – eviter, réduire, compenser (i.e., avoid, reduce or eventually compensate for it). It is, in other words, the mitigation hierarchy (for further discussion on mitigation hierarchy, see Chapter 6).

2.2.1.2 Outcomes of using various definitions or reference frames to assess degradation

The magnitude of degradation can be perceived differently by different actors and/or stakeholders. One reason for varying perceptions is the “shifting baseline syndrome”, which refers to changing human perceptions of an ecosystem over time (Pauly, 1995). Shifting baseline syndrome occurs when humans adjust their perception of the state of the environment unconsciously and whereby the abnormal easily becomes the new normal (Papworth *et al*., 2009). It is worth noting that while the use of local ecological knowledge for regional and global assessments (such as the ones produced by IPBES) are becoming more common (Danielsen *et al.*, 2003; Jones *et al.*, 2008; van der Hoeven *et al.*, 2004), the shifting baseline syndromes does entail that such data should be used with caution (Papworth *et al.*, 2009).

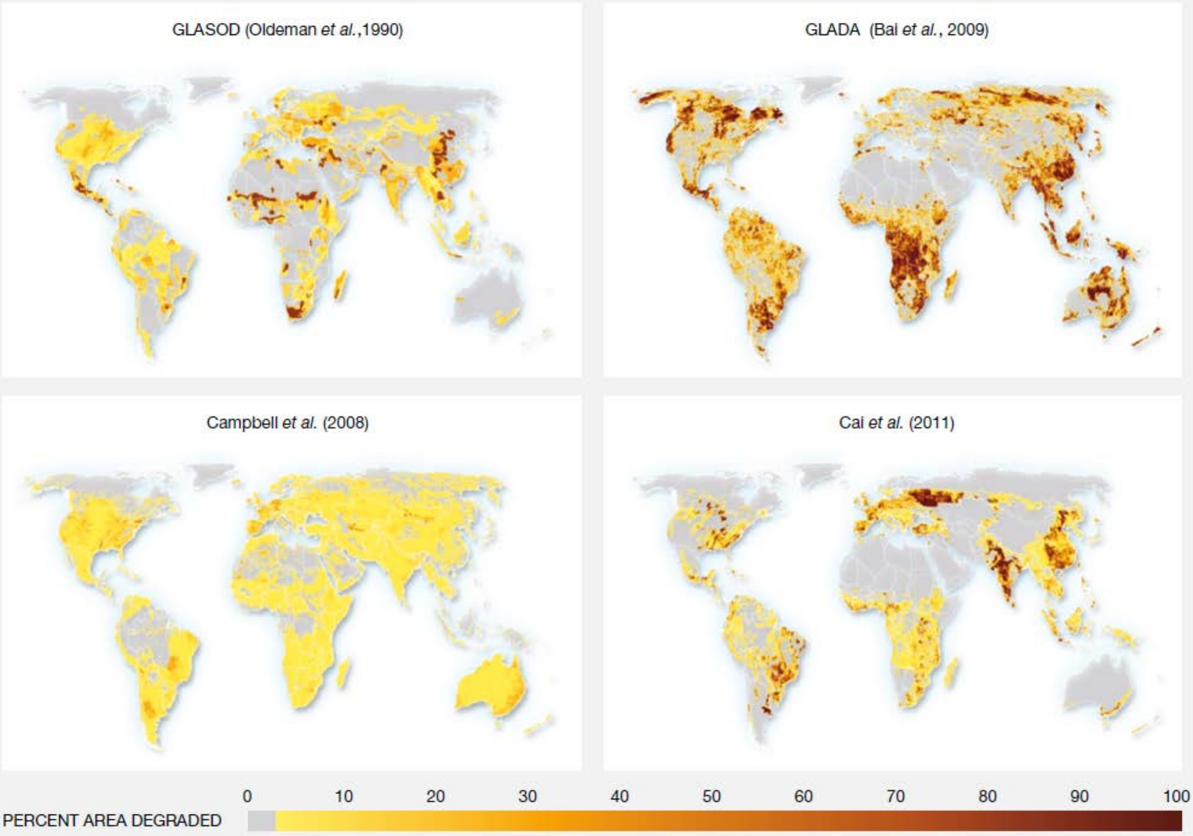
When assessing the current magnitude of degradation, there are concerns regarding the variability in definitions of concepts or principles which work towards deriving the pre-degradation reference frame (Hooke & Martín-Duque, 2012). Lack of consensus in the reference frame will cause the assessments of degradation and/or success in restoration to vary substantially (Gibbs & Salmon, 2015; Pereira *et al.*, 2014; Vogt *et al.*, 2011; van der Esch *et al.*, 2017). These estimates will often not agree with the one possible value of deviation from the natural state baseline for biodiversity and ecosystem functions. Furthermore, the lack of a common definition means that there will be different monitoring approaches, different indicators and different thresholds (e.g., Vogt *et al.*, 2011) which will considerably limit interoperability and integration across temporal and spatial scales for meaningful assessments. An additional source of variation between assessments can arise from the use of different methods. Gibbs and Salmon (2015) compared different approaches to assess degradation (Table 2.1), namely expert opinion (e.g., Oldeman *et al.*, 1991), satellite- derived primary productivity (e.g., Bai *et al.*, 2008b), biophysical models, and the identification of abandoned or marginal cropland (Cai *et al.*, 2011; Campbell *et al.*, 2008). They found that there was more agreement between maps showing areas with little to no degradation than for areas with more degradation. Disagreement between different approaches was noted by Gibbs and Salmon (2015) who calculated an estimate global extent of degradation ranging between 470 million ha and 6.14 billion ha (see Figure 2.7). The disagreement was stronger in Asia (Gibbs & Salmon, 2015).

Table 2.1. Benefits and limitations of major approaches used to map and quantify degraded lands (Gibbs & Salmon, 2015). Benefits and limitation refer to existing databases, not necessary the approaches as a whole, which could be improved to overcome limitations.

Approach	Benefits	Limitations
Expert opinion: Oldeman *et al.*, 1991 Dregne & Chou, 1992 Bot *et al.*, 2000	• Captures degradation in the past • Measures actual and potential degradation • Can consider both soil and vegetation degradation	• Not globally consistent • Subjective and qualitative • Actual and potential degradation sometimes combined • The state and process of degradation often combined

Satellite-derived net primary productivity: Bai *et al.*, 2008	• Globally consistent • Qualitative • Readily repeatable • Measures actual rather than potential changes	• Neglects soil degradation • Only captures the process of degradation occuring following 1980, rather than complete status of land • Can be confounded by other biophysical conditions
Biophysical models: Cai *et al.*, 2011	• Globally consistent • Quantitative	• Limited to current croplands • Does not include vegetation degradation • Measures potential, rather than actual degradation
Abandoned cropland: Field *et al.*, 2008 Campbell *et al.*, 2008	• Globally consistend • Quantitateive • Captures changes 1700 onward • Measures actual rather than potential changes	• Neglects land and soil degradation outside of abandonment • Includes lands not necessarily degraded

Figure 2.7 Maps of land areas (percent of cell area) affected by degradation; each panel represents one of the methods described, all shown with common legend and 20 km grid. Source: Gibbs & Salmon (2015).



This issue is further exemplified by looking at more approaches to assess degradation and the resulting estimates (Figure 2.7, Figure 2.8). In the early 1990s, focusing on the status of soils, the UNEP Global Assessment of Human-Induced Soil Degradation (GLASOD) identified areas where "human intervention [had resulted] in a decreased current and/or future capacity of the soil to support life", based on expert opinion (Oldeman *et al.*, 1991). Two categories of degradation processes were identified: displacement of soil material (water and wind erosion) and deterioration (physical or chemical). Note that in this assessment, soils that are “actively managed” in “relatively stable agricultural systems” were not considered as degraded. Human-induced soil degradation was found to affect 1.964 million hectares worldwide (i.e., 15% of the terrestrial land), mainly due to water erosion (Oldeman *et al.*, 1991). In particular, 2% of the soils were considered extremely or strongly degraded.

More recently, efforts to assess the degree of land degradation globally have expanded their definitions, allowing the use of different methods and approaches (Figure 2.7, Figure 2.8). For instance, the Global Assessment of Land Degradation and Improvement (GLADA) defined land degradation as “a long-term decline in ecosystem function and measured in terms of net primary productivity” (Bai *et al.*, 2008a). Technological improvement and the use of remote sensing also allowed for the use of the Normalized Difference Vegetation Index (NDVI) as a proxy to assess land degradation. However, the use of the index as a proxy for degradation, without considering land-use and land cover, has been criticized (Gibbs & Salmon, 2015; Vogt *et al.*, 2011). Biophysical models of agricultural productivity, combined with current land-use maps, are used to identify crops on land with marginal productivity, because these lands are prone to overutilization and subsequent degradation (Cai *et al.*, 2011; Gibbs & Salmon, 2015).

Wetlands are a further example of ecosystems for which a global assessment of degradation is particularly complex (see also Chapter 6, Section 6.3.2.5). Through rigorous assessment, Davidson (2014) recently confirmed the veracity of the longstanding estimate of wetland loss worldwide, namely 50% since the beginning of the 20th century. The first difficulty in devising a comprehensive estimate arises from a lack of knowledge on the distribution and extent of wetlands, with estimates ranging from 530 to 1280 Mha globally (Finlayson *et al.*, 1999; Lehner & Döll, 2004). Emerging technologies and better access to Earth observation products are promising advances to refine the global mapping of wetland (e.g. for peatlands see Dargie *et al.*, 2017; for global surface water see Pekel *et al.*, 2016). However, caution is advisable when defining a baseline for wetlands, because an increase in extent might be an artefact of technological improvement in measurement, rather than a result of conservation and restoration actions. Secondly, the assessment of wetland degradation is further complicated by the varying definitions of wetlands in use, in scientific publications and assessments. For instance, similar to the definition adopted for IPBES assessments, the Clean Water Act of the USA (EPA, 1990) considers wetlands to “generally include swamps, marshes, bogs and similar areas”. Yet, the Ramsar Convention on Wetlands expands this definition to sites that “incorporate riparian and coastal zones adjacent to the wetlands, and islands or bodies of marine water deeper than six metres at low tide lying within the wetlands” (Ramsar, 2013). In the Ecosystem Typology of the European Union, wetlands are represented by two categories: “inland wetlands” and “marine inlets and transitional waters” (EEA, 2015; Maes *et al.*, 2013). Analogous to the Living Planet Index , the Wetland Extent Trends index was recently proposed to overcome the incompleteness and heterogeneity of data on wetlands, and estimated a decline of 30% in the state of global wetlands between 1970 and 2008, particularly marked in Europe with a 50% decline (Dixon *et al.*, 2016). Using a current estimate of 900Mha of wetlands globally (Lehner & Döll, 2004), this loss in wetlands represents the degradation of 3% of the ice-free land surface since 1970 (Figure 2.8). While these estimates provide information on the area of wetland loss as a proxy for their degradation, they do not account for other forms of perturbation such as pollution and thus underestimate the magnitude of wetland degradation. For further discussion on wetlands and degradation of carbon stocks in wetlands, please refer to Chapter 4, Sections 4.2.3 and 4.2.5.

When looking at estimates of the global area under human pressures, considerably higher values for potential land degradation appear (Figure 2.8.). Between 35 and 47% of the terrestrial ice-free habitats have been converted to cropland, pastures and tree plantations (Hooke & Martín-Duque, 2012; Pereira *et al.*, 2012) and a further 7% to human infrastructure (Hooke & Martín-Duque, 2012). More than 75% of the global land area has been transformed by humans and can be placed within an “anthrome” – an anthropogenic biome (Ellis *et al.*, 2010). The Temporal Human Pressure Index – based on changes in stable nightlights, human population and cropland area – estimated that human pressure increased in 64% of the terrestrial area between 1990 and 2010 (Geldmann *et al.*, 2014). Though the link between human pressure and degradation is limited by the scarcity of global and spatially-explicit data, identifying those areas altered by human activities can be a first step towards assessing degradation and potential restoration (Geldmann *et al.*, 2014). This type of assessment is all the more relevant considering the livelihoods of the human populations relying on land as a resource. It was for instance estimated that 1.33 billion people lived on “degrading agricultural land” in 2000 (Barbier & Hochard, 2016), 95% of which were in developing countries. The number of people living on this degraded land increased by 13% by 2012. Similarly, Bai *et al.* (2008b) estimated that over 1.5 billion people (i.e., 24% of the world population at the time of their study) were affected by land degradation. This further suggests that even though some developing countries might experience economic growth, the proportion of their population living in degraded rural areas, particularly in remote areas, might not benefit from it (Barbier & Hochard, 2016).

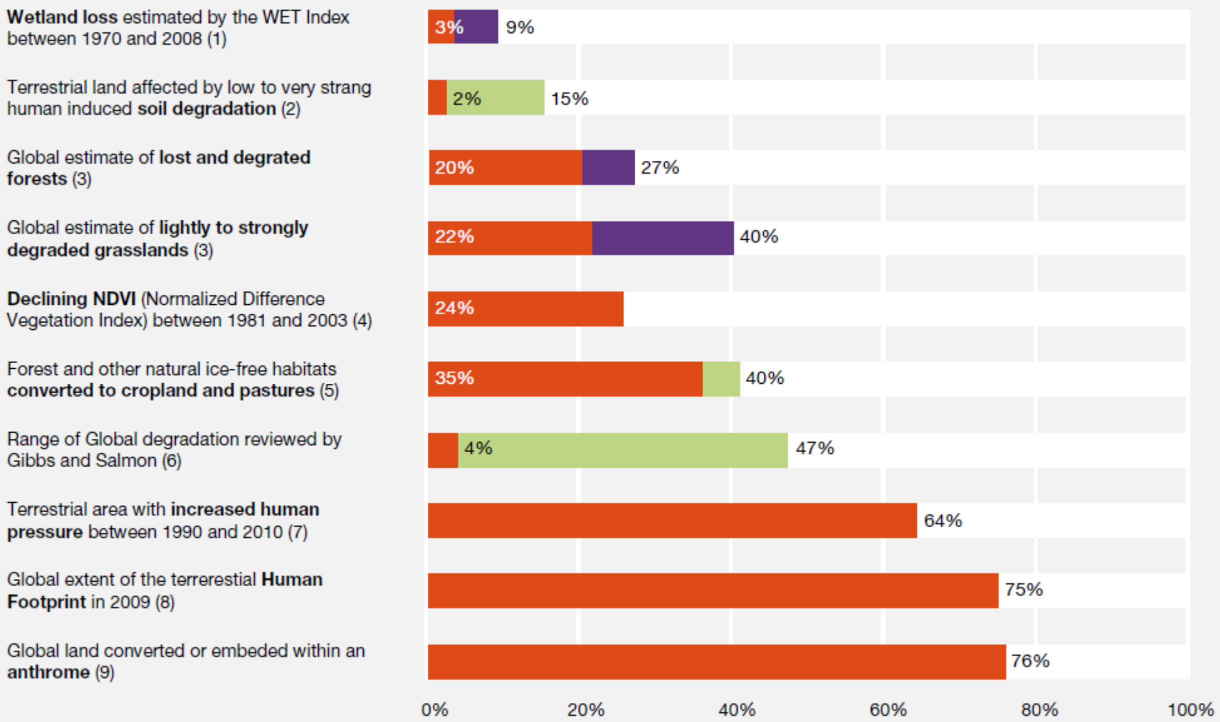
Estimates of land degradation can also show different results depending on the scale of the assessment (e.g., global versus national). By conducting a detailed assessment across all terrestrial ecosystem types in Finland, Kotiaho *et al.* (2015, 2016b) created a framework for assessing and reversing ecosystem degradation to support the national implementation of Aichi Biodiversity Target 15 and EU Biodiversity Strategy Target 2. Expert evaluations and all available data were utilized to construct pre-degradation natural state baselines for features important for biodiversity and for each ecosystem type, separately. In the assessment, “pre-degradation state for each feature” was defined as “the state of the feature in the ecosystems that would be existent in the absence of human intervention”. This corresponds to the counterfactual natural state baseline explained in Box 2.1. Degradation percentages were shown to be relatively greater than those of previous global assessments (Figure 2.8). The extent of degraded area across all terrestrial ecosystems was 84% of the area of Finland, while the overall average loss of ecosystem condition was 61%. A decade earlier and using a global assessment, only 8.2% of the terrestrial area of Finland were considered degraded (Bai *et al.*, 2008a) and nearly all of the country was considered part of the remaining global wilderness (Mittermeier *et al.*, 2003). This may suggest that many of the global-level assessments may not capture the true magnitude of damage that has been caused to biodiversity and ecosystem functions and services.

Assessing and mapping degradation can be a difficult task, even when the drivers of degradation are relatively well identified (see Chapter 3 for details discussion of drivers). This is illustrated by the ongoing European project RECARE (http://www.recare-project.eu), designed to develop a harmonized methodology to assess both the state of degradation of soil systems and its impact on functions and services. However, comprehensive knowledge on where, when and how known drivers affect the soil and methodologies for their assessments are often lacking (Stolte *et al.*, 2016). In some cases, the risk of, or susceptibility to, a given driver can be used as a proxy for the actual degree of degradation since they are easier to quantify and map.

Ultimately, the use of different models, input data and spatial and temporal resolutions can lead to heterogeneous assessments across countries, leading to an inability to capture the true nature of human-induced impacts on biodiversity and ecosystem functions and services. Regardless of the ecosystem, type of data or assessment methods used, uncertainty will be minimized with conformity to a singular consistent set of rules for deriving a baseline, evaluating the extent of degradation and assessing restoration success.

Figure 2 8 **Estimates of human pressure and degradation. Global estimates of the ice-free land surface affected by human pressure and/or assessed as degraded.**

Orange bars represent the percentage of terrestrial area affected by human pressure or degradation. Purple bars refer to the estimate of the proportion of the land surface covered by the ecosystem type (i.e., wetland, forests and grasslands). Green bars distinguish the upper from the lower estimates when both figures are provided in the study.
Sources: (1) Dixon *et al.* (2016); (2) Oldeman *et al.* (1991); (3) 3160 van Kolck *et al.* (2014); (4) Bai *et al.* (2008b); (5) Pereira *et al.* (2012); (6) Gibbs & Salmon (2015); (7) Geldmann *et al.* (2014); (8) Venter *et al.* (2016); (9) Ellis *et al.* (2010). [Adapted from Pereira *et al.*, 2014] Note that some of these estimates are dynamic and show an increase in degradation between two points in time (e.g., 4), while others are static and refer to the current percentage of a system being degraded (e.g., 3).
The estimate for wetland loss should be considered with caution, because we used an estimate of 900 Mha of wetlands globally (from Lehner & Döll, 2004) and applied a 30% increase backcasting to 1970 considering the Wetland Extent Trends index, from 1970 to 2008. The 900 Mha estimate is thus represented by the remaining 6% of ice-free land surface covered by wetlands in the figure.



2.2.1.3 Difficult concepts that may impact land degradation and restoration: time lags, regime shifts, long-distance connections and scarcity

A few additional concepts are relevant for assessing the state and responses of biodiversity and ecosystem functions and services, but may be difficult to perceive as such. These concepts include time lags, resilience, regime shifts, irreversibility, long distance connections and land as a scarce resource. Difficulty arises from the fact that these concepts are often invisible at the local scale and can occur over long periods. Ignoring these concepts may lead to erroneous conclusions about the state and responses of biodiversity and ecosystem functions and services.

Time lags

Often, there is a time lag (or time delay) between the start of a degrading activity and its effect on the environment. For example, the IPBES Plenary (IPBES, 2015) adopted a definition of degraded land that had at its base the observed loss of biodiversity, but it should ideally have also incorporated time lags. Generally, the death and/or extinction of species in any given location does not follow immediately after the anthropogenic environmental change. In the ecological literature this phenomenon is known as extinction debt, and the time delay is called relaxation time (Jackson & Sax, 2010; Kuussaari *et al.*, 2009; Tillman *et al.*, 1994).

After the environmental change, the threshold condition for survival of some species may no longer be met, but these species are still extant because of the time delay in their response to the environmental change. For instance, using data on bird populations in a fragmented forest in Kenya, Brooks *et al.* (1999) estimated that 50 years after the isolation of forest fragments of 1000 ha, only half of the expected extinctions had already occurred. Even though our current understanding of the extent and time scale of extinction debt is limited (Essl *et al.*, 2015; Kuussaari *et al.*, 2009), it is expected to be greatest where large-scale habitat destruction has occurred recently (Hanski & Ovaskainen, 2002). Recently, the extinction debt concept was extended to include ecosystem services (Isbell *et al.*, 2015). Incorporating time lags, such as extinction debts, can lessen the impact of degradation by buying more time to land managers and conservation planners to improve the ecosystem conditions (via restoration or sufficient rehabilitation) before the projected extinctions occur (Brooks *et al.*, 1999).

Time lags are also present, and may be considerable, in the recovery of ecosystems after restoration and rehabilitation. In particular, in cases where species have gone locally extinct and restoration or rehabilitation is undertaken, ecological successions and natural recolonizations are also likely to happen with time lags (Hanski, 2000). For instance, a wildlife comeback is currently being observed in Europe (Chapron *et al.*, 2014; Deinet *et al.*, 2013). This comeback is partly due to conservation actions and changes in legislations (Deinet *et al.*, 2013), but was also facilitated by the abandonment of remote and marginal agricultural areas. This land abandonment created an opportunity for restoration via ecological rewilding: the passive management of ecological succession with the goal of restoring natural ecosystem processes and reducing the human control of landscapes (Navarro & Pereira, 2012; Pereira & Navarro, 2015). The colonization of new suitable habitats may even be faster than the relaxation of the extinction debt if the change of the environment is slow enough (Svenning & Sandel, 2013).

Time lags presents a key question for environmental law as well, as it frames public actions. In many countries, public actions to repair a crime or a felony must be conducted within the time frame from one to thirty years. This rule is explained by the principle of legal certainty to protect citizens. However, when it comes to environmental law, these time frames are far from being widely adopted. Moreover, the statute of limitation that limits public actions commences after the event causing damage and not from the moment the damage is perceived. Therefore, if the damage appears or is perceived ten years or more after the damage was caused, the possibilities of a judicial action become void. The principle of legal certainty thus currently protects the polluters and does not account for ecological reality (Larson, 2005). Exceptions exist, such as in Alberta, Canada, where the law prescribes a 25-year liability for surface reclamation issues (topography, vegetation, soil texture, drainage and so on) and a lifetime liability for contamination associated with upstream oil and gas activities (Province of Alberta, 2016).

Resilience, regime shifts and irreversibility

The concept of resilience is common to both the natural and social sciences. In ecology, resilience refers to the ability of ecosystems to absorb disturbances while remaining in a stable state (Carpenter *et al.*, 2001; Holling, 1973; Kinzig *et al.*, 2006b; Scheffer *et al.*, 2015; Standish *et al.*, 2014a), while in social science, resilience is the capacity of human populations to adapt to new social-economic (development pressure, urbanization) or environmental contexts (climate change, deforestation, desertification). The main discrepancy between the definitions of resilience in the social and natural sciences is that social resilience can be defined as independent from the destruction or modification of the ecosystem, so long as human societies find subsistence alternatives (Adger, 2000).

Despite its growing popularity with policymakers and managers, some authors have recently pointed out the vagueness of the concept of resilience in ecology and its many definitions (Mumby *et al*., 2014; Myers-Smith *et al.*, 2012; Standish *et al.*, 2014a). Nonetheless, resilience is particularly relevant to degradation and restoration (see also Chapter 4, Section 4.1.2.1 for further discussion on the role of ecological resilience in degradation processes). Ecological resilience highlights the level of disturbance that an ecosystem can sustain and can guide restoration. For instance, if a system is resilient to disturbance, its recovery to a pre-disturbance state can be passive and may not require human intervention other than cessation (Mumby *et al.*, 2014; Standish *et al.*, 2014a). Recovery time – the time required by an ecosystem to return to pre-disturbance state (Myers-Smith *et al.*, 2012; Standish *et al.*, 2014a) – is essential to consider, as ignoring it could lead to a premature assessment of impacts and thus underestimation of the potential success of restoration interventions (Haapalehto *et al.*, 2017).

Continuous and long-term pressure on ecosystems can lead to a loss of resilience and cause them to shift to an alternative stable state, a phenomenon called a "regime shift" (Barnosky *et al.*, 2012; Folke *et al.*, 2004; Kinzig *et al.*, 2006; Scheffer *et al.*, 2001, 2015; Scheffer & Carpenter, 2003). Examples of regime shifts are soil salinization, the transition from forests to savannas, fisheries collapse and the mangrove transition (Folke *et al.*, 2004; Leadley *et al.*, 2014; Rocha *et al.*, 2015). Disturbance thresholds are used to estimate the level of disturbance that a system can sustain before moving to an alternate state (Standish *et al.*, 2014a). Regime shifts can be rapid or more gradual (Walker & Meyers, 2004), the latter being potentially harder to identify and assess (Scheffer & Carpenter, 2003). Furthermore, the fact that the shift can be either smooth or abrupt, as is the case when the system reaches a tipping-point (Folke *et al.*, 2004; Leadley *et al.*, 2014), will have an impact on how the transition is perceived by different stakeholders.

The direct and indirect drivers of regime shifts were recently classified in five broad categories which also match to some extent the different drivers of land degradation discussed in Chapter 3 of this assessment: (i) habitat modification; (ii) food production; (iii) nutrients and pollutants; (iv) resource extraction; and (v) spill-over effects such as the indirect effect of human activities on natural processes (Rocha *et al.*, 2015). Those drivers can also be placed into networks of interaction within and across those categories, which highlights the risk of “cascading regime shifts,” even more so when most of those drivers are linked to human activity (Kinzig *et al.*, 2006; Rocha *et al.*, 2015). Regime shifts can also be caused by the overexploitation or introduction of species (Leadley *et al.*, 2010). Invasive alien species have, for instance, changed biotic and abiotic conditions in island ecosystems (Burgiel, 2010) and caused shifts from submerged to floating plants in aquatic ecosystems (Nolzen *et al.*, 2017.). More generally, they can alter trophic cascades (Estes *et al.*, 2011) which can result in collapses in ecosystems (e.g., predator invasion in Downing *et al.*, 2012).

While the resilience of a system prevents it from crossing a threshold, the term “unhelpful resilience” was recently used to describe the fact that an ecosystem can be resilient in a degraded state, limiting the effectiveness of restoration (Standish *et al.*, 2014). Indeed, once in an alternative state, the process to reverse the system to its natural state might be too difficult or too costly (Folke *et al.*, 2004). Given our definition of degradation (see Section 2.2.1.1), a regime shift can often cause a system to remain degraded, even if the cause of the degradation is removed.

Many regime shifts are caused by climate change and other anthropogenic drivers, and have hence been extensively studied within socio-ecological systems. In those systems, the human impact is due to resource management – driven by local, regional and global socio-economic factors (e.g. Kinzig *et al.*, 2006) – while the state of the ecosystem will in turn impact the amount and quality of available resources. Regime shift can thus directly and indirectly affect the supply of ecosystem services and human well-being (Rocha *et al.*, 2015).

Thresholds in ecosystems are difficult and complex to observe and perceive, but can be assessed using observations of temporal data or experimentation (Mumby *et al.*, 2014; Scheffer *et al.*, 2015; Standish *et al.*, 2014; Laliberté *et al.*, 2010; Standish *et al.*, 2014). In addition, there are several databases and online resources to inform researchers and managers (e.g., http://www.resalliance.org/ ; http://www.regimeshifts.org/; and http://www.early-warning-signals.org) (Walker *et al.*, 2004; Rocha *et al.*, 2015).

Legal thresholds are the result of a social compromise defining what is acceptable and what is not. Hence, the change of status occurs when the degradation is no longer socially acceptable. Therefore, the legal perception of regime shifts is not in accordance with its ecological counterpart. Many judges lack environmental and ecological knowledge, which contributes to this effect and leads to the misunderstanding and subsequent discounting or dismissal of environmental impacts in legal proceedings (Lecuq & Maljean-Dubois, 2008). Nevertheless, creating specific environmental courts, like those created in India or Chile in 2012, might help remediate this shortcoming.

Timescales and the perception of land degradation and restoration

Humans and human activities have altered and/or degraded ecosystems since the late Pleistocene (Ellis *et al.*, 2013; Pereira *et al.*, 2012). In fact, relatively little of the Earth's land area can be considered natural or “wild” today (Mittermeier *et al.*, 2003; Sanderson *et al.*, 2002), while “intact landscapes” such as forest continue to decrease in extent (Potapov *et al.*, 2017). Yet, due to the timescale of such phenomena, even heavily-altered systems are not always perceived as degraded. For instance in Europe, some valued cultural landscapes – such as the Causses and Cevennes World Heritage site – or terraced farming are the products of intense and long-lasting alterations and use of ecosystems (Halada et al., 2011; Navarro & Pereira, 2012). Their perception as “natural” and their acceptance as the “normal state of nature” (Vera, 2010) constitute an example of the shifting baseline syndrome (see 2.2.1.2).

Progressive or gradual degradation processes that occur during one’s lifetime might also be difficult to perceive. Degradation, for example, due to overgrazing and non-sustainable agricultural practices (Leadley *et al.*, 2014; Scheffer *et al.*, 2001), can be a gradual process that can go unnoticed until a tipping-point or threshold is reached and the stakeholders start perceiving the intensity of degradation and its impact on their well-being (Folke *et al.*, 2004). This is also the case of the long-term degradation of the Amazonian forest which, in combination with climate change at the global scale, could lead to a sudden regime shift and a transition to a savannah-type ecosystem (Leadley *et al.*, 2014).

Other types of degradation that are easy to perceive are immediate catastrophic events. Those events are typically perceived and acknowledged by the public and demand concrete responses. A recent example is the breaking of the dam holding wastewater from Samarco mining Company that affected the Rio Doce in Minas Gerais, Brazil (see Box 5.8, Section 5.5.2) and was described by the Brazilian president as the “worst environmental disaster in the history of Brazil” (Escobar, 2015). The event was widely covered by the media internationally and triggered strong public outrage. The perception of emergency in the response to degradation is indeed a crucial point. A catastrophic event is more salient and might thus have more impact on policies and response (Jørgensen *et al.*, 2014). On the contrary, when degradation processes are slow, and their impact on human well-being are not immediately perceived or felt, the societies are less likely to stop the degradation process or initiate a restoration effort.

The slow recognition that desertification had to be internationally resolved is one such example. As pointed out by Corell (1999), the international community was mobilized several times on this topic before the United Nations Convention to Combat desertification (UNCCD) was signed in 1994. Severe environmental disasters had by then accelerated the process, such as the Sahelian drought (see Behnke & Mortimore (2015) for more on this discussion), and policymakers resorted to using a vocabulary of emergency (e.g., “disappearance of countries”) in order to accelerate actions. Still, it took fifteen years to sign UNCCD into force.

Likewise, the time for ecosystem recovery after restoration can vary greatly and should be systematically considered. Many ecosystems can recover assisted or in some cases, non-assisted, from disturbances but the time scale of such processes can span from decades to centuries (Jones & Schmitz, 2009; Kotiaho & Mönkkönen, 2017; Haapalehto, *et al.*, 2017). For instance, abandoned agricultural lands in Europe could take between several decades to over a century for ecological successions to occur and to naturally become forested (Verburg & Overmars, 2009). Active restoration must also be understood as a long-term process. We are only now starting to draw some conclusions from long-term and large-scale restoration programs, such as the restoration of the Mata Atlantica rainforest in Brazil (see Chapter 6, Box 6.4 and Section 6.3.1.2), one of the most endangered hotspots of biodiversity (Brancalion *et al.*, 2014; Melo *et al.*, 2013), or the Grain for Green program, a large-scale plan of restoration of set-aside land, initiated in 1999 in China to combat soil erosion and desertification (Cao *et al.*, 2009; Feng *et al.*, 2013).

By ignoring the potential time-lags between an action and the response of a system, a “short term” vision to assess the outcomes of conservation policies and restoration actions might also impact the capacity to observe and perceive successes (Tittensor *et al.*, 2014) or failures. Furthermore, the time-scale of restoration processes can become an issue when considering its mismatch with the duration of decision makers’ political mandates (Villalba, 2010), and during which tangible restoration results are often expected.

Global conservation targets are also typically time-bound. For example, Aichi Biodiversity Target 15 sets the target of restoring 15% of degraded land by 2020 (CBD, 2011). In contrast, having long-term perspectives could allow for the development of progressive approaches, where meeting the goals are reassessed through time, as the focal ecosystem is recovering (Chazdon, 2008). It was thus argued that restoration should be understood as an investment rather than a direct cost for society (de Groot *et al.*, 2013). It is important to allow the time needed to achieve restoration goals to avoid the premature perception of failure or non-achievability. Finally, it is important to recognize that human action targeted at specific species, ecosystems or ecosystem services – including through the degradation process or restoration and rehabilitation actions – can have an impact on the selective forces acting on biodiversity over long temporal scales (Sarrazin & Lecomte, 2016). Yet, those interactions are rarely accounted for. Hence, Sarrazin and Lecomte (2016) recently advocated for an "evocentric" (i.e., centred on evolution) approach to conservation, where strategies are developed to preserve both nature and future generations’ well-being, while considering processes acting at an evolutionary time-scale rather than opting for a "blind Anthropocene" in which any consideration for the conservation of the non-human is ignored (see also Kotiaho & Mönkkönen, 2017).

Long-distance impacts and their legal implications

There are often long-distance connections between land degradation and human well-being that are invisible to most stakeholders, but must be taken into account (see Chapter 5, Section 5.3.2.5). For example, consumption and pollution put major pressures on biodiversity and have shown worsening trends, both past and projected (Tittensor *et al.*, 2014). The global production and trading of goods to satisfy demand is also one of the main drivers of land degradation (Lambin & Meyfroidt, 2011a; Lenzen *et al.*, 2012). One clear example is the case of increasing meat consumption and soy production as drivers of deforestation (see Figure 2.9) (Marchand, 2009; Nepstad *et al.*, 2006). In particular, consumers in developed countries tend to have larger "biodiversity footprints" abroad than within their countries - contributing to significant negative impacts in developing countries (Lenzen *et al.*, 2012).

Figure 2 9 An illustration of how long-distance connections are obstacles to full awareness of consumer choices.

Increased demand for soy for animal feed, in Europe and Eastern Asia, encourages deforestation in South America, including the Cerrado savanna, Amazon forest and Pampa. Intensive pork breeding pollutes rivers and provokes the phenomenon of "green tides" on the seashores. Photo source: Creative Commons, licensed under CC BY-SA / Compiled by F. Kohler.

The consequences of local degradation processes can also have long-distance negative impacts on biodiversity and societies (Liu *et al.*, 2015). This is for instance the case with transboundary haze pollution in South East Asia – resulting from palm oil production and forest fires in Indonesia – which also raises the issue of perceived responsibility between countries (Forsyth, 2014). Furthermore, there are concerns that increasing EU demand for biofuels will increase indirect land-use change in countries where biofuels are produced (mostly in South America). In reaction, a directive on the promotion of the use of energy from renewable sources (European Commission, 2009) was adopted to provide a transnational legal framework for dealing with these issues (Farber, 2011). Failing to take into account these long-distance connections limits the ability of conventions and governments to design appropriate policies for mitigation, restoration and compensation. These considerations prompted the development of the "telecoupling framework" (i.e., socio-economic and environmental interactions over long distances), including assessments of its impact on land-use change globally (Liu *et al.*, 2013).

An additional long-distance connection of land-use change is caused by the transition of developed countries from net forest losses to net forest gains (Meyfroidt *et al.*, 2010), accompanied by urbanization and agricultural land abandonment. If and when the demand for agricultural and timber goods stagnates or increases, this transition might lead to the "outsourcing of degradation" (Meyfroidt & Lambin, 2011) – a process also known as land-use displacement. Similarly, there is a danger that strict conservation policies and the setting aside of land for conservation and/or restoration might become drivers of degradation elsewhere – a phenomenon known as "leakage of environmental impact" (Andam *et al.*, 2008; Armsworth *et al.*, 2006; Lambin & Meyfroidt, 2011b; Latawiec *et al.*, 2015; Lenzen *et al.*, 2012; Liu *et al.*, 2015). For instance, reforestation projects on productive land of the Mata Atlantica, in Brazil, could lead to the displacement of grazing pressures elsewhere (Latawiec *et al.*, 2015). Likewise, strong leakages were observed when Vietnam implemented a reforestation policy and increased its forest cover at the expense of neighbouring countries, where deforestation increased in order to satisfy the domestic demand in timber products (Meyfroidt & Lambin, 2009). Nonetheless, one positive form of long-distance connection occurs when the benefits of restoration are not only felt locally, at the spatial scale of the site being restored, but have downstream positive effects at a larger scale (de Groot *et al.*, 2013; Liu *et al.*, 2015).

Long-distance impacts caused by land degradation are hardly considered by national legal orders and even less by the international legal order. Thus, the legal concepts of land degradation and restoration are often constrained to local scales. This perception differs from the existing international legal order and its treaties and conventions for the protection of air and water quality, for example. Such a difference can be partially explained by the fact that land generally falls under state territory and national jurisdiction, despite its transnational characteristics. And despite the existence of general legal instruments, transboundary impacts caused by land degradation are often underestimated and not taken into account by the law (Convention on Environmental Impact Assessment in a Transboundary Context, 1991; European Commission, 2010; Gray, 2000; Johnstone, 2013). For example, select Member States have rejected the EU's proposal for a Soil Framework Directive – referring to the subsidiarity principle (Olazabal, 2007) and arguing that soil protection is a national matter and hence outside the scope of the EU.

Internationally, there is a lack of strong conceptual foundations for building effective international mechanisms. There are first and foremost conceptual and practical issues with the "sovereignty principle", because of the various hurdles it can create for an international organization or a country to investigate the state of land within national borders. Consequently, international conventions that focus on land have generally revolved around developing support approaches (Ramsar, 1971; Ninan, 2001; UNCCD, 1994) and are seldom legally binding (Friedrich, 2013; Revised European Soil Charter, 2003). Hence the current status of land prevents the development of alternative and legitimate (Bodansky, 1999) forms of ecological governance (Camanho, 2009; Angus, 2007; Woolley, 2015) based on the legal implementation of the concept of ecological solidarity, for example (Naim-Gesbert, 2014; Thompson *et al.*, 2011). Ecological solidarity (see Glossary) is a legal concept of French environmental law. It provides a step toward consolidating ecological and social interdependence in biodiversity policy. In the words of Thompson *et al.* (2011): "from ecology based on interactions to solidarity based on links between individuals united around a common goal and conscious of their common interests and their moral obligation and responsibility to help others, we define ecological solidarity as the reciprocal interdependence of living organisms amongst each other and with spatial and temporal variation in their physical environment". The idea is that in order to increase the efficiency of conservation measures, the surrounding landscape of the protected area must be integrated. In other words, ecological solidarity "could ensure the protection of the ecological and human dimensions of landscape functioning, where a multitude of (mostly undervalued) services are provided" (Thompson *et al.*, 2011) (see Section 2.2.3.3 for more detailed discussion about ecological solidarity).

Nonetheless, when countries share common concerns, the protection and sustainable management of land can become an international matter. The Alpine Convention (Dallinger, 1994), signed by the eight Alpine countries (Germany, Austria, France, Italy, Liechtenstein, Monaco, Slovenia and Switzerland) illustrates this idea. Its purpose is to create a common framework to manage and preserve the alpine environment. The convention is based on nine protocols and at least five of them are related to land issues: (i) mountain farming; (ii) mountain forest; (iii) spatial planning and sustainable development; (iv) conservation of nature and countryside; and (v) the most directly land-related soil conservation protocol of 1995. All alpine countries, except Switzerland, have ratified all of these protocols.

Although the whole mechanism of the Alpine Convention is facing governance and implementation issues, it nevertheless demonstrates that land (and more specifically soils) can be managed at a supranational level. Within this framework, parties have shared their knowledge to elaborate an appropriate text (Balsiger, 2007; Simon, 2011). For instance, the Soil Protocol conveys the definition of soil given by the European Soil Charter of the Council of Europe, by the European Commission and by the German Soil Protection Act (see also Chapter 6, Section 6.3.2.2). Moreover, this example illustrates that, as these alpine countries share a mountain area with specific threats and ecosystems, they have an accurate perception of the consequences caused by land degradation (Desrousseaux, 2014).

The progressive recognition of land as a scarce resource

Soil protection, in itself, is perceived as a national matter. Land and soil are two different legal objects and only specific threats or types of land are internationally preserved: the threat of desertification, high interest wetlands and natural and agricultural landscapes. Land, as a scarce resource (Lambin *et al.*, 2001; Lambin & Meyfroidt, 2011b), is largely unmanaged by international environmental law (Kiss & Shelton, 1991) except for the UNCCD.

International community, supported by soil specialists, have elaborated the concept of "soil security". It is described as an overarching concept of soil motivated by sustainable development and "concerned with the maintenance and improvement of the global soil resource to produce food, fibre and freshwater, contribute to energy and climate sustainability, and to maintain the biodiversity and the overall protection of the ecosystem. Security is used here for soil in the same sense that it is used widely for food and water" (Brauch & Spring, 2009; Keesstra *et al.*, 2016; Koch *et al.*, 2013). Traces of this concept are found in international working documents of the UNCCD. It refers to "existential threats for survival [of humankind] and requires extraordinary measures to face and cope with these concerns. Security concepts offer tools to analyse, interpret, and assess past actions and to request or legitimize present or future activities" (Brauch & Spring, 2009). As food or water are already considered security issues, the concept of soil security put soil issues at the same level of importance. For instance, while the right to water has been assigned a constitutional level of protection in most national legal orders (for the highest level possible, see Figure 2.6), such right has not been assigned for land (May *et al.*, 2015) – except where it concerns women or indigenous peoples in specific cases. Soil protection, therefore, needs to be developed at the international level (Boer & Hannam, 2004; Desrousseaux *et al.*, 2016). At this time, policymakers have access to non-binding instruments, such as the newly adopted Voluntary Guidelines for Sustainable Soil Management, which provides general technical and policy recommendations for soil preservation measures (FAO, 2017a).

Related to the concept of “soil” there is one further challenge for the law. Land and soil are frequently ambiguous in law, as they are not clearly separated or made distinguishable. On this matter, proposals have been made to adopt a Soil Protocol under the authority of the UNCCD (Boer & Hannam, 2015). Some institutions are aware of this situation and the European Commission, for instance, has expressively explained why soils should be differentiated from land. European Thematic Strategy for Soil Protection states that “while soil is the physical upper layer of what is usually referred to as ‘land’, the concept of ‘land’ is much wider and includes territorial and spatial dimensions. It is difficult to separate soil from its land context. However, this communication focuses on the need to protect the soil layer as such, due to its unique variety of functions vital to life” (2006).

At a national level, and due to their territorial specificities, some countries have an accurate perception of the scarcity of land and have thus built strong legal frameworks in order to prevent land degradation. For instance, Article 75 of the Federal Constitution of the Swiss Confederation, specifies that "the Confederation shall lay down principles on spatial planning. These principles are binding on the Cantons and serve to ensure the appropriate and economic use of the land and its properly ordered settlement" (1999). In other words, Switzerland has an accurate perception of the scarcity of its land and proactively attempts to limit its urbanization. Food safety is also one of its concerns. As a result, Switzerland is considered as one of the best performing countries of Europe to preserve land and associated food security (Dufourmantelle *et al.*, 2012; Karlaganis, 2001).

2.2.1.4 Approach to assess degradation and recovery of ecosystems

If assessment and monitoring of the negative effects (degradation) of management practices and development, or the positive effects of restoration and rehabilitation are to be done, they must be evidence-based (Block *et al.*, 2001). Measuring ecosystem degradation first requires determining a baseline, relative to which the current state of an ecosystem is compared. For the particular purpose of assessing anthropogenic ecosystem degradation, an obvious reference is the natural state without any human modification (see 2.2.1.1 and Box 2.1). Restoration success is in practical terms easier to assess and monitor than assessing degradation, because here the expected ecosystem changes are in the future and can be monitored. However, in order to do this rigorously and scientifically, there is a need for well-designed long-term monitoring programmes, following, for instance, the classical idea of the Before-After, Control-Impact design (Block *et al.*, 2001; Underwood, 1994) supplemented with replicates. First, one should establish replicated plots on independent ecosystems that are in a degraded state and on corresponding ecosystems that are in their pre-degradation state. The pre-degradation sites can be established by using the space-for-time substitution as a proxy (see below). The first inventory of the current state of all the plots should be conducted before any of the plots are restored. After the first inventory, half of the degraded plots should be restored and the other half left as controls. After the restoration measures have been completed there will be three different types of replicated plots: degraded plots, restored plots and plots in a pre-degradation state. The monitoring should be continued of all three of those plots. These replicated Before-After, Control-Impact designs allow the researcher to distinguish the true effects of restoration measures from natural succession and random changes in community composition, as well as other variables over time (see e.g. Elo *et al.*, 2016; Menberu *c*2017; Noreika *et al.*, 2016).

Space-for-time substitution, also known as a chronosequence (Blois *et al.*, 2013; Foster & Tilman, 2000; Haapalehto *et al.*, 2014; Johnson & Miyanishi, 2008), can be used to infer the magnitude of damage from a series of plots differing in terms of age since disturbance or restoration by humans. In this approach, pre-degradation state ecosystem plots that represent the same abiotic and biotic response attributes as the damaged target ecosystem (prior to degradation) are identified. Then, the attributes of the damaged and pre-degradation state plots are compared. This approach is commonly used in experimental ecology and in restoration ecology when assessing the success of restoration in reversing damage (e.g. Aide *et al.,* 2000; Kareksela *et al.*, 2015; White & Walker, 1997). In practice, some uncertainty exists regarding representativeness and the pre-degradation status of the chosen pre-degradation state ecosystem plots. In addition the assumption that all plots traced the same history in both abiotic and biotic attributes is unavoidable (Johnson *et al.*, 2008; Pickett, 1989).

2.2.1.5 Land-use change and externalities

There is no doubt that values play an important role in how societies treat nature, land and its ecosystem services, but there are also fundamental demographic and economic mechanisms leading to habitat loss and subsequent loss of biodiversity (Dasgupta, 2001; De Moor, 2008; Dietz, 2003; MEA, 2005b).

Biodiversity is something economists generally describe (in a largely anthropocentric approach) as displaying public-good characteristics. Public goods have non-excludable use by other potential users and are non-rivalrous in consumption (Kolstad, 2000). Ecosystem services are often rival non-excludable (common pool resource) or both non-rival non-excludable (public good). A market economy, based on private property and excludability, generates externalities (Kolstad, 2000; Pigou, 1920). Broadly speaking, the notion of an externality refers to a benefit or loss created by an individual's (or group of individuals') influence on production or consumptions possibilities for others, without any compensation or payment (Hanley *et al.*, 2007). Hence, externalities refer to economically important negative or positive impacts, not taken into account by markets.

Instruments to internalize negative externalities often revolve around attaching a cost (e.g., reflecting in the cost of commodities) to a negative impact (Kolstad, 2000; Pigou, 1920). Land-use changes can create biodiversity-related externalities by weakening life-supporting, regulating and cultural services, thereby inducing biodiversity loss. One way of addressing such negative environmental externalities is to develop policies for implementing compensation mechanisms (e.g., taxation). Examples of economic incentives to restrict negative externalities include taxes on emissions and pollutions, individual tradable quotas and quality standards. They directly target the rationale behind choices causing pollution and degradation, by internalizing the environmental cost into the price of a given good or service (e.g., industrial poultry or pork meat) under the "polluter-pays principle". Consequently prices of such products would rise, making abatement efforts and alternatives more economically appealing, thereby actively incentivising consumers to choose more environmentally-friendly products (Oosterhuis & ten Brink, 2014). Such an "ecotax" has been applied in Austria, Switzerland and Germany on heavy truck transportation and was quite effective in fostering local products or rail transportation (Sainteny, 2012). In some cases, removing "perverse subsidies" can be sufficient (Oosterhuis & ten Brink, 2014). Such subsidies are usually set up to support a given economic sector (e.g., agriculture), but in the process also contribute to increased negative externalities (e.g., nitrate pollution). By heavily subsidising agricultural production after World War II, the European Common Agricultural Policy is partially responsible for the overuse of fertilisers, leading to eutrophication since the 1970s (OECD, 2004). Instead of reducing such (perverse) subsidies for agricultural production, the EU decided to add new subsidies under a “second pillar” of the Common Agricultural Policy. These new subsidies pay for positive externalities of agriculture as well as reduction of negative externalities under the heading of “agri-environmental measures”.

Incentives and restrictions are generally based on environmental impact assessments and cost-benefit analyses of the direct environmental and economic impacts of particular practices. For decision makers, cost-benefit analysis provides a feedback mechanism which confronts the problem of market demand for commodities and the lack of accounting for externalities with the same tools, measuring rod and language (i.e., value and costs). As such, exercises of valuation can play an important role in calling attention to the value of biodiversity and to intangible ecosystem services (Brondizio *et al.,* 2010). In turn, multi-criteria assessments (Munda, 2008; Verburg *et al.*, 2014) and deliberative approaches (Habermas, 1984; Raymond *et al.*, 2014; Vatn, 2009) go beyond the exclusive focus of environmental impact assessments on ecological structure and processes to consider the context-specific and often conflicting values held by human communities on the issues at stake (Langemeyer *et al.*, 2016).

Ecosystems have relevance for human well-being beyond the satisfaction of individual preferences for tangible goods and services. These intangible values of nature belong to the cognitive and emotional realm of human beings, and, as such, are hard to quantify (Kumar & Kumar, 2008; Wegner & Pascual, 2011) (see also Chapter 5). These psycho-cultural benefits of nature (see Chapter 5, Section 5.4.6) are increasingly recognized (Chan *et al.,* 2012) and their neglect in policy appraisal and interventions can produce undesired consequences (e.g., Fankhauser *et al.*, 2014; West *et al.*, 2006). Along these lines, some researchers have questioned the use of cost-benefit analysis and valuation. A recent survey showed that the academic literature gives little attention to the issue and rarely reports cases where ecosystem services economic valuation has been put in actual use (i.e., ex-post examples) (Laurans *et al.*, 2013). Nevertheless, a survey of U.S. decision makers has shown that they highly value economic information along with history and context studies to inform their decision-making process (Avey & Desch, 2014).

As property rights on environmental resources (such as clean air, water, biodiversity) are not well defined, the rights of use often go to the spoiler, which may result in the negative externality of long-term depletion of natural resources and a decrease in returns for all (Ostrom, 2010; Poteete *et al.*, 2010). One alternative to pricing instruments is to improve the allocation of property rights. Collectively devised and accepted resource-use rules have proven most effective in managing common pool resources and can generate long-term benefits for the group as a whole (De Moor, 2008; Duraiappah *et al.*, 2012; Mongin, 2003; Ostrom, 1990). For instance, a recent study of community managed conservancies bordering the north of the Maasai Mara National Reserve indicates that pastoral livelihoods currently do not constitute a source of habitat degradation and livestock grazing intensity has no impact on prey species and carnivore populations. Instead, the major threat to the survival of endangered predatory species, like the lion, are retaliatory killings due to livestock depredation. Here, household-level cash incentives from community-managed wildlife tourism act as an effective strategy to reduce the frequency and/or severity of reaction to livestock depredation, and enable the recovery of lion populations (Blackburn *et al.*, 2016). Setting land aside or reducing livestock densities was not necessary.

In an ecological compensation market, developers degrading the environment demand offsets that are provided by landowners, who in turn may invest in restoration of large land areas and sell offsets from these habitat banks. The trades are verified by an administrator (Coggan *et al.*, 2013). If no net loss is requested, the trading rules must make the ecological value of the destroyed and restored sites equivalent (McKenney & Kiesecker, 2010). Buying and selling offsets creates prices that reflect the costs of habitat restoration and the developers' need for offsets (Doyle & Yates, 2010). The restoration costs determine the supply of offsets: the rarer the habitat in question, the more expensive the offset. In an ideal offset market the desired biodiversity outcome, such as no net loss of biodiversity, can be achieved and that the costs of offsetting might inhibit harm caused by any development project (Conway *et al.*, 2013; Wissel & Wätzold, 2010).

Ecological compensations are considered to work only for ordinary habitats, because areas with threatened species and rare habitats may be irreplaceable (Pilgrim *et al.*, 2013), are under strict regulation and probably should not be included in the market exchange (McGillivray, 2012). Monitoring and verification is an important part of ecological competition. It has been argued that no net loss can only be achieved if current regulations pertaining to the avoidance and minimization steps of the mitigation hierarchy continue to be stringently enforced (Dickie *et al.*, 2010) and possibly reinforced (Conway *et al.*, 2013). However, as offsets can be mandatory or voluntary, they can be partial, instead of fully compensating (Moilanen & Laitila, 2016). Unfortunately, too often these ecological compensation guidelines have been neglected (Briggs *et al.*, 2009; Coggan *et al.*, 2013).

Currently, efforts to render ecological compensation initiatives more effective are being explored under the land degradation neutrality component of Sustainable Development Goal 15 (Caspari *et al.*, 2015; Dooley *et al.*, 2015; Minelli *et al.*, 2016; Welton, 2015). Land degradation neutrality is defined as "a state whereby the amount and quality of land resources necessary to support ecosystem functions and services and enhance food security remain stable or increase within specified temporal and spatial scales and ecosystems" (UNCCD, 2015:4). Under this approach, the Science-Policy Interface of the UNCCD recommends that ecological compensation should be implemented by respecting the "mitigation hierarchy", as does IUCN (2016) and the Ramsar Convention through Resolution XI.9 (See Chapter 6, Section 6.2.1).

An important element to consider when predicting or assessing the effectiveness of economic incentive-based tools, is their interplay with the normative systems and motivations of targeted actors. The critics of ecological compensation are concerned that such schemes may create the false impression that any impact can be compensated for, whereas ecosystems' link to livelihood opportunities and psycho-cultural wellbeing (Brown *et al.*, 2013; Ryan *et al.*, 2010; Weimann *et al.*, 2015) are locally specific and therefore not fully replaceable (Escobar, 2008; Forest Peoples Programme, 2011; Quétier & Lavorel, 2011).

Nevertheless, common to many documents on ecological compensation is that, while they describe well the goals of ecological compensation or biodiversity offsetting including the mitigation hierarchy, they do not systematically cover the factors and decisions that effectively drive the outcome of offsetting. Recent work reviewed the concepts of offsetting and summarized the operational decisions that effectively determine how well ecological damage becomes compensated (Moilanen & Kotiaho, 2017, 2018). This document describes a framework allowing well-informed evaluation of biodiversity offsets. Factors treated in the document cover the three major axes of ecology, biodiversity, space and time as well as a host of additional factors, such as additionality, leakage, flexibility, connectivity, trading up, baseline trend assumptions and multipliers needed to account for various uncertainties. These should all be considered and addressed in the operationalization of any ecological compensation of biodiversity offsetting case.

2.2.2 Sense of place: indigenous and local peoples facing degradation and restoration

IPBES has, at its core, the integration of scientific, indigenous and local knowledge and practices so that degradation can be perceived and defined by different observers, and so that restoration can be achieved using both scientific and local expertise. Scientific knowledge tends to be specialized and deals with specific aspects of reality, while indigenous and local knowledge tend to be systemic (or holistic) (DeWalt, 1994; Lévi-Strauss, 1966; Pretty *et al.*, 2009; Roué & Nakashima, 2003). By systemic, we mean that indigenous and local knowledge and practices, in general*,* integrates both material and spiritual knowledge and practices (Nakashima *et al.*, 2012; Trosper & Parrotta, 2012).

Starting from the premise that indigenous and local knowledge and practices are integral to understanding the perceptions of land degradation and restoration, this subsection starts by reviewing the complexities of indigenous worldviews. This is followed by examples of indigenous and local classification systems related to soil degradation, showing how these different classifications may be useful for restoration projects. We then review obstacles, such as social inequities or discrimination, to the involvement of indigenous and local populations in conservation projects. We argue that the concept of "commons" is a useful tool for collective management, at the local scale (but also at international level, as explained in Section 2.2.3). Finally, we focus on NGOs and the dilemmas they can meet on the ground when trying to conciliate social and biodiversity conservation programmes.

There are two important challenges for "traditional" peoples. First, "being traditional" cannot be imposed on populations that might aspire to something else for themselves or their children (Kohler & Brondizio, 2017). "Being traditional" can be interpreted as being frozen in time, while in practice, being traditional means keeping a certain ethos, habitus (Bourdieu, 1977) or worldview even when adopting new practices and technologies. Many traditional populations are traditional exactly because they do not have access to full citizenship like basic public services. Keeping tradition alive should be a choice and not be imposed by conservation policies (Fukuyama, 2014), especially when access to benefit sharing is still to be enforced by national policies (Carrizosa, 2004; Stabinsky & Brush, 2007). The Nagoya Protocol paved the way by formalizing this access to benefit sharing (Bélair *et al.*, 2010).

Second, many public policies can sacrifice traditional practices to accelerate modernization (Roué & Molnár, 2016). Traditional populations are thus marginalized and forced to adapt to dominant market systems. Both challenges underscore the fact that traditional peoples need a legal forum to express their aspirations, while outsiders often view them as innate ecologists, supposed to compensate for environmental degradation brought on by development, or as obstacles to progress, requiring a quick assimilation (Chapin, 2004). In both cases, the interests of the environment and traditional peoples only partially coincide and environmental policies should not be limited to delegate environmental responsibilities to traditional peoples, because resolving environmental problems require a global rethinking of development trends.

For the purpose of this assessment, we will adopt the IPBES definition of indigenous and local people (which does not overlap exactly with the definition of the ILO, 1989), namely that indigenous and local people are those who rely on traditional cultural and subsistence practices and are at least partially dependent on local biodiversity and ecosystem services for their social reproduction (also see Glossary). Social reproduction here is understood as the phenomenon by which a society can perpetuate itself across time. For further discussion and definitions about indigenous and local knowledge and practices see Chapter 1.

Indigenous and local concepts and perceptions are embedded in worldviews deeply bonded to a specific territory, and some understanding of these worldviews is required to include them in this assessment. For example, concepts such as “taboo” (forbidden place, animal or action), “mana” (emanation of supernatural power) or “hau” (the spirit circulating through gifts) are seldom included in international assessments. The concept of "Mother Earth” used by IPBES, is specific to human groups (especially Andean), but was mentioned in the conceptual framework to signify the intimate relationship between human beings and their environment (Díaz *et al.*, 2015).

2.2.2.1 Nobody will survive the fall of the sky: spiritual knowledge against degradation

To understand the very specific link between indigenous and local peoples and their environment, we may have to rely, in many cases, on first-hand ethnography. Box 2.2 gives an example of the complexity of the interpretive system of Yanomami people of South America, an example intended to illustrate the difficulty of generalizing indigenous and local concepts. However, in general, the link between indigenous and local practices and the environment is neither “human-centric” nor “eco-centric”: human societies and the environment are perceived, not as separate entities, but as involved in a unique relationship (especially in totemic and animistic cosmologies - Descola, 2013). This relationship embraces also spiritual and symbolic values (Brondizio *et al.*, 2009; Díaz *et al.*, 2015).

For example, the concept of “mauri” among the Māori population of New Zealand is an expression of a balanced ecosystem and cosmic order (Harmsworth & Roskruge, 2014). A similar concept exists in Yanomami’s cosmology (see Box 2.2) and in many other indigenous groups. It expresses the transcendence of a spiritual/physical principle according to which degraded land and soils are spiritually damaged, affecting the connectedness between humans and nature. Such a spiritual relation between humans and land and soils was vivid in Europe before the Enlightenment period (Patzel, 2010). A slight modification in land cover or species distribution also affects social balance and culturally significant places. In present days, in New Zealand, researchers, including Māori, have used indigenous memory and knowledge (mātauranga Māori) – for example understandings of traditional Māori concepts such as taonga, mauri and kaitiakitanga – alongside science to develop an integrated inclusive approach to wetland classification, restoration and management (Harmsworth, 2002).

Box 2.2 Yanomami’s perception of gold mining in the Amazon

Yanomami's first contact with Brazilian pioneer fronts occurred in 1971 when the military regime decided to build a peripheral road in Northern Amazon. The situation got out of control in 1979 when the price of a gold ounce rose in the London Stock exchange, provoking a gold rush in Yanomami’s traditional territory. The pressure from thousands of gold miners on game and other resources reduced Yanomami population from 20,000 to 7000. Yanomami were subjected to new diseases and starvation due to the disappearance of bushmeat, the use of mercury, as well as to massacres, rapes and slavery.

Anthropologist Bruce Albert (1993) documented the words of shaman and spokesperson Davi Kopenawa’s about Yanomami’s perceptions of the land degradation provoked by the gold rush. Yanomami perceive gold mining as “forest eating” and gold miners as “supernatural peccaries” rummaging through the soil, threatening cosmological order (*urihiri*). In their worldview, Omamë, Yanomami’s creator of the universe, destroyed the first world he created by provoking the fall of the Sky, which became the new Earth surface. The ancient world was buried, including gold and other metals, along with malevolent spirits. Buried metals are conceived as pathogenic agents (*shawara wakëshi*), emanating a deadly smoke when extracted. That smoke affects and kills Yanomami. It affects also the “forest’s breath”, suffocating the trees and the living beings. Yanomamis now conceptualize all white men's activities through this lens and generalized the concept of *wakëshi* to embrace industrial pollution in a global perception of threatened sky and Earth. White men's greed is seen as a form of cannibalism, as it is contrary to Yanonami's worldview, according to which sociality is based on sharing food and goods. Thus, gold mining and wealth accumulation mean, not only ecological disaster, but also a perversion of human social order. Davi Kopenawa concludes: *"When we all have disappeared, when all our shamans will disappear, I think that the sky will fall again. […] The forest will be destroyed, the sky will darken. […] White people are smart, but they ignore the power of our shamans, and they are unable to hold the sky. […] Not only will the Yanomami die. White people will die also. Nobody will escape from this new fall of the sky."*

Based on Albert (1993).

As discussed above, indigenous and local knowledge and practices are not only about ecosystem management, but also about maintaining socio-ecological balance, often through spiritual principles (Box 2.2). As shown by Kalkanbekov and Samakov (2016), the rules of behaviour on sacred sites leads to preservation of biota located in these areas. Many peasant communities around the world, who are not legally recognized as indigenous, maintain this spiritual relation through ethical practices. Respecting this spirituality through the concept of sacred sites is a powerful tool for biocultural diversity conservation. The example of Uluṛu-Kata Tjuṯa (Ayers Rock-Mount Olga), in Australia – at first a National Park (1958) then part of UNESCO cultural heritage (1994) – is one of many (Whittaker, 1994). Some countries went even further by considering that the environment should be defended as such, thus acknowledging its spiritual, but also intrinsic value. Such is the case of the New Zealand Parliament that adopted an Act stipulating that Te Urewera was no longer a National Park, but a legal entity with "all the rights, powers, duties, and liabilities of a legal person" (Section 11(1) of the Te Urewara Act, New Zealand Legislation, 2014). This Act was based on the recognition of the spiritual bond of Te Urewera ecosystems and Landscapes and Ngāi Tūhoe people, who endorsed the role of "guardians" of its integrity. On 5 August 2014, another Act was approved, giving the status of legal entity to Whanganui River in New Zealand (Ruruku Whakatupua, 2014). Under this Act, the Māori community and the government will each appoint a member to represent the river's interests.

These inclusive policies should not be conceived as creating open-air museums, but as responding to the necessity of reconnecting nature and people via immaterial links (Dudley *et al.*, 2009; see also Chapter 5, Section 5.4.6). Many sacred sites were purposely considered as sacred precisely because of their ecological and/or aesthetic interest (e.g., the Meteora monasteries in Greece or Mount Saint-Michel in France). Spirituality diffuses in a day-to-day life by creating long-lasting ethical principles, for which the Yanomamis' forest is an example (Kopenawa, 2013). This *mana* (to use this generic indigenous concept for supernatural presence) challenges the limits between ecology, society and spirituality (Berkes, 2012). Sacred spaces that have spiritual significance create tangible opportunities for conservation of biodiversity and ecosystems (Bhagwat & Rutte, 2006), while preserving unique social-ecological systems, all of which are part of human cultural diversity. These considerations also raise the issue of the perception of restoration by indigenous and local populations in the case of sacred and symbolic sites. Although the ecological attributes of a degraded site can be, in theory, restored, one might question if the same can be said of its cultural value (Wild *et al.*, 2008).

This leads us to consider other ways of integrating indigenous and local concepts and perception not only in science, but also in industrial and post-industrial societies. An example of these alternative standards can be found in the Constitution of Ecuador (Constitution of Ecuador, 2008) and Bolivia (Constitution of Bolivia, 2009) which have integrated the concept of "Buen vivir" (or "Vivir bien") in order to recognize that individuals depend on nature (Acosta, 2008; Walsh, 2010). "Buen vivir" translates the Aymara concept of *Sumak Kawsay,* meaning "fulfilment". This ethics considers, for instance, that land is not only a means of production, but also a living territory with multiple, material and immaterial, dimensions (Borsatto & Carmo, 2013). Applied to nature, it leads to the restoration of land in accordance with a natural state baseline, a flourishing natural life. Applied to humans, it means that individuals should fulfil their lives through sociability, friendships and family ties, well-being, leisure, harmony with nature, and not just through work and material consumption. Amartya Sen (2001) proposes a similar concept, "capabilities", to describe the human potential to attain fulfilment. As a Constitutional principle, "Buen Vivir" refers to ancient and traditional Andean knowledge. Its concrete implementation in public policies, though, is still problematic (González & Vázquez, 2015; Gudynas, 2011; Villalba, 2013).

At an ideological level, "Buen vivir" entails an ethics that many rural social movements have adopted. This dimension of indigenous and local knowledge and practices transcends the limits of local projects: it constitutes a model of alternative connections between humans and their environment.

2.2.2.2 Withdrawing cash from the water bank: practical knowledge for restoration

Scientific assessments of land degradation and restoration are carried out using modern tools and technologies. However, it is important to recognise that the parameters by which indigenous and local people assess the indicators of land degradation and restoration are based on their traditional, long-term knowledge and have relevance to local resource management practices (Adams & Watson, 2003; Bollig & Schulte, 1999; Oba & Kotile, 2001; Talawar & Rhoades, 1998). The experiential and transgenerational knowledge of their surroundings, built on their close proximity and familiarity with their environment, is the key to the depth of indigenous and local perceptions of land degradation and restoration (Bennett, 2015) and their adaptive agrobiodiversity management (Jackson *et al.*, 2012). However, some of this knowledge may be subject to the shifting baseline syndrome discussed in 2.2.1.2. Nevertheless, studies have shown that, in many cases, indigenous and local people's soil classification systems are based on their in-depth knowledge of soils and often complements scientific assessments of soil properties aimed at determining the suitability of soils for agriculture (Adams & Watson, 2003; Cervantes-Gutierrez *et al.*, 2005; Critchley & Netshikovhela, 1998; Douangsavanh *et al.*, 2006; Peña-Venegas *et al.*, 2016; Pulido & Bocco, 2014).

Indigenous and local knowledge and practices about land management, and the causes and consequences of land degradation, can offer potential options for restoration. Thus, it is important to find "hybrid" solutions linking indigenous and local knowledge and scientific knowledge, as well as adopting interdisciplinary approaches to address these issues (Altieri, 2004; Andrade & Rhodes, 2012; DeWalt et al, 1999; DeWalt, 1994; Tengö et al, 2014). Today this complementarity is still problematic and different frameworks have been proposed for enabling successful collaboration between scientists and knowledge holders (Ens *et al.*, 2012; Trosper *et al.*, 2012).

The level of environmental knowledge of local and indigenous populations is today largely accepted and is unquestionable in its importance and relevance to conservation (Berkes & Davidson-Hunt, 2006; DeWalt, 1994; Tengö *et al.*, 2014). However, only recently has indigenous knowledge been welcomed and integrated into scientific knowledge in works on conservation issues (Reid *et al.*, 2009). This approach requires an equal partnership between scientists and local and indigenous peoples in every step of the research process. This integration is facilitated in *in-situ* conservation projects through a participatory approach (Borrini-Feyerabend *et al.*, 2000; Chambers, 1994), leading to community-based conservation programs (Berkes, 2004). The participation of local populations is not automatic, of course, and the efforts can be in vain because of the political context (McCormick, 2014). Nevertheless, there is reason to remain optimistic about this participatory process, as seen in Box 2.3., describing how a successful restoration project is perceived by local population in Abraha Atsbeha, a village of Northern Ethiopia.

Box 2.3 The case of Abraha Atsbeha: creating a "water bank" in Northern Ethiopia

Abraha Atsbeha is a village situated in Tigray, Northern Ethiopia, one of the driest parts of the country. By the end of the 1990s, after massive deforestation and overgrazing, the villagers relied almost exclusively on food aid. But, as Ato Gebremichael (main actor of the project and former chief of the village) put it: "for how long can you be a beggar for food?" In 1998, the Ethiopian Government, supported by GIZ (Deutsche Gesellschaft für Internationale Zusammenarbeit) and other donors, proposed that the villagers adopt a new management plan, consisting of fencing the cattle and restoring springs using traditional practices. Such a plan was successful thanks to a strong collective capacity to achieve common objectives, a capacity translated into the concept of "social capital" (Brondizio *et al.*, 2009; Putnam, 1995). Now, almost twenty years after the beginning of the program, the villagers can harvest vegetables and fruits three times a year and can sell their surplus at local markets. The experience spread across the regions of Tigray, Oromia and Amhara, and inspired the program Africa RISING (The Africa Research in Sustainable Intensification for the Next Generation), created in 2012.

Locals perceive the restored springs as a bank account and irrigation as withdrawing cash from the "water bank". Ato Gebremichael describes it as: "*Allowing regeneration of vegetation on the upper part of the watershed is like putting your money in the bank. The only difference is that we are withdrawing the cheque not from where we deposit it, the upper part of the catchment, but from another place, the lower part of the catchment*."

Perceiving restoration as a metaphor for financial investment, and harvesting as an investment return, is an interesting way of reversing the unidimensional monetary evaluation, by considering nature's contributions to people as the money itself.

Based on: Lamond (2012); Shiferaw *et al.* (2012).

See also: "Ethiopia: The highlands turn green" on GIZ official website: https://www.giz.de/en/mediacenter/18674.html

Many customary practices have a legal status within a tribe or even a state, if it recognizes customs as a source of law. Research in environmental law has demonstrated that many laws and decrees are based on customs, mostly regarding land management, fishing and hunting activities (Permingeat, 2009). Practical knowledge sometimes becomes a law regardless of its positive or negative impact on the environment. Nevertheless, this approach is fundamental to harnessing the solidarity between humans and their territory. Since the development of international environmental law, international and regional conventions have strived to preserve this knowledge. For instance, article VI of the African Convention on the Conservation of Nature and Natural Resources is dedicated to "land and soil" and calls for a sustainable management of land and its restoration. It explicitly mentions that local knowledge must be part of the management plans. In addition, article XVII of the Convention gives attention to the importance of respecting local farmers' rights and encourages their participation in decision-making processes (1968). However, the implementation of this Convention is still in process fifty years after it was signed (Ramutsindela, 2007) (see 2.2.3). Some countries specifically recognize indigenous rights, but international conventions are needed to protect traditional land tenure (e.g., Convention Concerning Indigenous and Tribal Peoples in Independent Countries, 1989) like the Voluntary Guidelines on Tenure

(FAO, 2012). Protecting access to land has now become an urgent matter in the face of 'land grabbing' – when a foreign country buys arable land for its own supply (Borras Jr. & Francott, 2010; Freiburg, 2014; Locher *et al.*, 2012) – and the preservation of traditional knowledge is recognised as a major, albeit still poorly functioning, lever (see Section 2.2.3).

Furthermore, the question of fair and equitable benefit sharing is still an open one (Tvedt, 2006). The Nagoya Protocol the Convention on Biological Diversity (Buck & Hamilton, 2011) is meant to clarify this legal and moral issue both for genetic resources and for traditional knowledge associated with genetic resources (Buck & Hamilton, 2011). An adapted payment for ecosystem services, similar to the framework of European Union Common Agricultural Policy, is another path that needs exploring, as suggested by Ivașcu and Rakosy (2016) and Babai (2016) for Romania.

2.2.2.3 Social inequities versus "the tragedy of the commons"

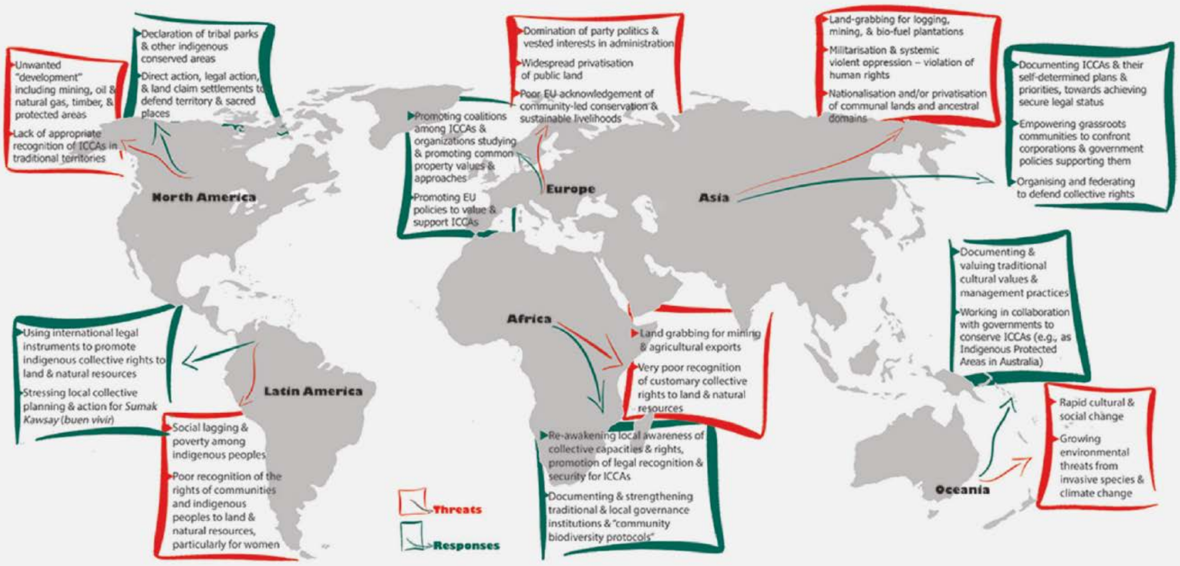
The precarious situation of many indigenous and local people and their knowledge systems cannot be addressed by local participation in conservation projects alone, when existing development models continue to put pressure on their resources and livelihoods (Brandon, 1998) (see also Box 2.4, Section 2.2.4.3). For instance, some traditional farmers and/or traditional herders' conflicts in Sub-Saharan Africa are due to the expansion of monocultures reducing the extent of traditional grazing territories, leading to competition between traditional herders and small farmers, and to land degradation due to overgrazing (Tschopp et al, 2010; Turner, 2004). Facing the problem of overgrazing and erosion, or the overexploitation of undomesticated plants or animals, governments tend to impose restrictions that are hardly respected, as vulnerable communities have few alternatives (Mekuria et al, 2011; Wezel & Haigis, 2002). Sometimes, coercive legislation about uninhabited protected areas deeply affects people's relationships with their environment, leading to retaliatory actions such as burning protected forests (Agrawal, 2005a, 2005b) and intensive wood-trafficking (Kohler, 2008), or the loss of knowledge about how to coexist with predators such as wolves or bears (Benhammou, 2009).

Poverty and land scarcity is a major obstacle that can undermine conservation programmes, especially when it comes to tropical forests (Songoro, 2014). Local people are sometimes compelled to degrade forests when they cannot alleviate poverty, and therefore log and transform forests into pastures and croplands (Durand & Lazos, 2008). To face an uncertain future, these populations migrate (Reuveny, 2007) (see also Chapter 5, Section 5.6.2.1) or strategically invest in their children's education by overexploiting the remaining resources. However, these local issues should be considered, not as singular cases, but in part as the result of strict national policies (see Chapter 5, Section 5.2.2.2). Social inequity and the lack of adapted public policies cause or exacerbate many of these harmful practices (Adams & Hutton, 2007; Brockington *et al.*, 2006; Brockington & Wilkie, 2015; Sanderson, 2005; West *et al.*, 2006), especially in case of "land grabbing" (Anderson, 2013; Martiniello, 2013) and land concentration for export crops (Guibert & Sili, 2011).

Many development projects occur in sparsely populated areas, which often coincide with traditional territories, such as hydroelectric dams (Rajagopal, 2014; World Commission on Dams, 2000). Pervasive deforestation in Africa (Kenrick & Lewis, 2001) and South-East Asia has led to the deterioration of "social ecosystems" in Indonesia (Anderson, 2013), Philippines (Eder, 1990; Zapico *et al.*, 2015) and many others (for an exhaustive list, see Survival International website: http://www.survivalinternational.org/). Indeed, negative environmental impacts can severely affect unique socioecological systems (i.e., human societies' reliance on the ecosystems they live in) and cultural diversity. In many instances, those most affected by these changes are also those most politically-marginalised (Kohler & Brondizio, 2017; Oyono, 2005). In such cases, especially, civil society can step in to stand for those segments of society that can hardly resolve these issues by themselves (Nonfodji, 2013). Figure 2.10 show some of these conflicts and the solutions adopted.

Figure 2 10 **Example of threats to and responses by indigenous peoples and local communities.**

Source: The ICCA Consortium, Indigenous Peoples' and Community Conserved Territories and Areas (ICCAs), eaflet, Cenesta, Teheran (2013). http://www.iccaconsortium.org/



Until recently, theories of human behaviour and common property contended that, left to its their own devices, individual pursuits and uses of common-pool resources inevitably lead to what was called by Hardin (Hardin, 1968) a "tragedy of the commons" (see also Chapter 5, Section 5.2.2.3). The underpinning rationale was as follows: under the shared management of common-pool resources, each individual engages in "free-riding" behaviour (Olson, 1965), whereby they hope to limit their own costs and maximize their own net benefits while benefitting from the conservation efforts of others. The predicted outcome is failure to cooperate and the unavoidable environmental degradation (Anderson & Hill, 1977; Demsetz, 1967; Hardin, 1968; North & Thomas, 1973).

In 1985, the National Research Council's Panel on Common Property Resource Management provided stimulus for an extensive number of case studies and meta-analyses on common property rights and collective action across the globe – an approach called Institutional Analysis for Development. These studies demonstrated that a "tragedy of the commons" was neither common nor inevitable (Berkes *et al.*, 1989; Bromley, 1991; Murphree, 1993; Ostrom, 1990). Throughout history there have been examples of socioecological systems in which the productivity of the land was low and human societies were unable to develop adequate collective institutions for internal regulation (e.g., the Polynesian Islands, the Easter Island) (Brander & Taylor, 1998; Caldararo, 2004). However, numerous case studies also demonstrated that self-organized collective institutions governed by stable communities that are buffered from outside forces have mostly sustained common-pool environmental goods and services successfully. Examples include collective rules for fisheries (e.g., Acheson, 2003; Davis, 1984), forests (e.g., Bray *et al.*, 2004; McKean, 1986) pastures (e.g., Campbell *et al.*, 2006; Netting, 1972), irrigation (e.g., Coward, 1977; Trawick, 2001), wild plants and animals (e.g., Dyson-Hudson & Smith, 1978; Eerkens, 1999) and production of landscapes (Bélair *et al.*, 2010). For a meta-analysis of the new commons and their implications for environmental management, see Duraiappah *et al.* (2014) and Lopez & Moran (2016).

Among the main concepts used to assess the efficiency of these systems are "human capital" and "social capital" (Brondizio *et al.*, 2009). Human capital represents all the knowledge, talents, skills, abilities, experience, intelligence, training, judgment and wisdom possessed individually and collectively by individuals in a population (Bourdieu, 1986). Social capital, as mentioned above, represents the capacity of a community (local or international) to gather and achieve common goals (Coleman, 1988; Putnam, 1995), sometimes by inventing new forms of governance, for example by empowering women (Banerjee & Duflo, 2012; Patel, 2012; Tripp, 2004).

Since the 1980s this new perspective on common property and collective action has given rise to community-based natural resources management policies and programmes that promote the collective ownership and management of common pool resources intended to deliver both conservation and community development outcomes (Ostrom, 2000; Poteete *et al.*, 2010; Roe *et al.*, 2009) (for a discussion of community-based natural resources management policies see Chapter 6). However, some critics observed that institutional analysis of development gave little space for ecological issues (Epstein *et al.*, 2013), including Ostrom herself (Ostrom & Cox, 2010). But lack of empowerment, land insecurity, resignation, poverty, social competition, lack of compensation, often inhibits a collective response if there is no international civil society support (Feldman & Geisler, 2012; Sanderson, 2005; Sanderson & Redford, 2004; Songoro, 2014) (see also Chapter 5, Section 5.2.3.3).

2.2.2.4 Facing human- wildlife conflict: NGOs' dilemma

Since Rio 1992, the strategies between environmental (e.g., WWF, TNC, Greenpeace) and human rights NGOs (e.g., Survival International, Brazilian Instituto Socioambiental) began to converge, with environmental NGOs becoming a major ally of indigenous and local populations in their struggle for civil and territorial rights. This convergence came from an initiative of indigenous and local people, as expressed by the final declaration of the conference Two Agendas on Amazon Development, held by the Coordinating Body for the Indigenous' Organisations of the Amazon Basin (2014: 81-93).

The main difference between major NGOs and governments is that the actions of the former are not limited by national borders, allowing them to have a global approach to problems that are often considered through the lens of sovereignty by governments. Major NGOs have the capacity to allocate funds where they are most needed. They can also cooperate with local groups to better target the desired objective, and thus, are major actors in channelling funds from developed to developing countries. This cooperation between international NGOs and local associations is crucial to avoid a standardized approach, disconnected from local realities. Instead, it can draw attention to the importance of listening to local populations as genuine stakeholders (Couix & Gonzalo-Turpin, 2015; Nastran, 2015), who must be given alternatives to meet their needs and social expectations (Sjögersten *et al.*, 2013).

This alliance between environmental and civil rights and/or humanitarian NGOs – and their commitment to local populations – can lead to positive results and achievements. Some well-thought and inclusive projects associate a broad range of stakeholders with diverging interests to promote common restoration projects – such as the restoration of the riverine forest of Xingu River, involving indigenous tribes, small farmers and soy producers (Arvor *et al.*, 2010; Campos-Filho et al, 2013; Schwartzman *et al.*, 2013) (see also Chapter 5, Box 5.5, Section 5.3.3.1 and Chapter 6, Box 6.5, Section 6.3.3.2).

However, these same alliances expose NGOs to a major dilemma provoked by land degradation – namely, the increased occurrence of "human-wildlife conflicts", involving moral, political and ecological choices. Human-wildlife conflicts become more frequent and acute because of the shrinking of wild habitats

(Dickman *et al.*, 2013), leading to extreme reactions such as culling (e.g., elephants) or poaching (mainly predators) (Distefano, 2005; Lamarque *et al.*, 2010; Löe & Röskaft, 2004; Woodroffe *et al.*, 2005). Emblematic apes (orangutans, chimpanzees and gorillas) are especially endangered by deforestation, leading them to feed on croplands. Furthermore, the increasing contacts between wild and domestic animals and human leads to the outbreak of zoonosis (Woodroffe *et al.*, 2005) such as aids, bird flu, bovine tuberculosis (which also affects baboons) (Sapolsky, 2002), swine fever, brucellosis, rabies or Ebola virus (see also Chapter 5, Box 5.7, Section 5.4.2). All of these diseases can mutate and affect humans as well as great apes, leading the latter to extinction (Ryan *et al.*, 2011). Human diseases (e.g., tuberculosis or yellow fever) can also affect great apes (Köndgen *et al.*, 2008; Wolf *et al.*, 2014) or New World monkeys (Crockett, 1998; Goenaga *et al.*, 2012; Mucci *et al.*, 2003). The Ebola outbreak in Gabon and Congo killed 5000 gorillas between 2002 and 2003 (Bermejo *et al.*, 2006). How can an NGO decide which species – endangered gorillas or humans – to deal with in the first place? An urgent situation should not prevent long-term programs, such as restoring deforested areas that create buffer zones to avoid future ethical dilemmas.

Much of the research on conservation conflict focuses on the adverse impacts that humans or wildlife have on one another (Conover, 2001), like the impact of predators on livestock (Marchini & Macdonald, 2012) or the impact of hunting on endangered species. A common response to these problems has been to scientifically quantify the impacts and then use legislative (e.g., bans and penalties), mitigation (e.g., financial compensation) and technical mechanisms (e.g., fencing livestock) to address them (Gutiérrez *et al.*, 2016). However, adverse interactions between humans and wildlife are frequently a manifestation of underlying clashes of interests and values between opposing human groups (Marchini, 2014). Beneath the observable actions and impacts lies a complex web of contrasting worldviews and deteriorating trust between those who want to preserve wildlife and those whose livelihood and well-being are affected by it (Redpath *et al.*, 2015). Moreover, conservation conflicts often serve as proxies for conflicts between people over other social and psychological issues, including: struggles over group identity or ways of life; recognition; socio-economic status; fear of loss of control; and anger over historical grievances (Madden & McQuinn, 2015).

Different groups may have different views of what a conservation conflict is about, or whether there is a conflict at all (Redpath *et al.*, 2015; Young *et al.*, 2016). The effects of conflict on health and well-being of local people have been acknowledged (Barua *et al.*, 2013) and a great variety of local approaches to conflict resolution exist (Reed & Del Ceno, 2015). There is often a reluctance on the part of NGOs and government actors, including their respective scientific advisors, to acknowledge local perceptions of conflict, which can lead to increased frustration and lack of cooperation (Hulme & Infield, 2001; Young *et al.*, 2016). In many situations a top-down approach might ultimately be counter-productive, since the frustrated party (generally the locals) may develop a sense of grievance and the conflict may re-emerge elsewhere or several years after (Redpath *et al.*, 2015; Redpath *et al.*, 2013). Another counterproductive approach is to forbid practices based on social-ecological balance (see for example totemic and animistic worldviews described in 2.2.2) in which humans and predators maintain social relations (sometimes conflictual) based on beliefs or history (e.g., tigers and Mishmi people on the Sino-India border in Aiyadurai (2016)).

Confronted with the difficulty of solving these situations, scholars and practitioners (officers and/or employees from both NGOs and government agencies) have started to address conservation conflicts through better integration of knowledge and concepts in the ecological sciences with those in the social sciences that regularly engage with the underpinnings of human conflicts, such as psychology, sociology and peace studies (White & Ward, 2011). A review of 52 environmental conflicts indicates that mutual engagement of the parties can contribute to the development of equitable and effective agreements and improved relationships (Emerson *et al.*, 2009).

As existing legislation may sometimes be perceived as discriminatory, especially if it derives from international agreements imposed on national policies (Kohler, 2008; Mermet & Benhammou, 2005), NGO practitioners are generally better accepted at the national scale (Heydon *et al.*, 2010). However, complexity and uncertainties characterize any conflict management process, whereby conflicts can re-emerge unexpectedly; a long-term adaptive management approach is therefore required (Milner-Gulland & Rowcliffe, 2007). But another problem arises from the fact that NGOs are often accountable to their donors, above and beyond local populations or governments. This is a key issue in understanding how human-wildlife conflicts remain frequently unsolved. There are situations, for instance, where a specific program can come to an end, along with the means allocated for its implementation, even if the situation is far from being stabilized (Desmarais, 2007; Kohler, 2008).

What is certain is that NGOs cannot address human-wildlife conflicts on their own. Their actions have to be supported by strong political decisions. Examples include: limiting demographic pressure (see Section 2.2.4.2); developing payment for ecosystem services; enforcing legislation against long-distance wildlife trafficking; and avoiding the conversion of protected areas for activities such as transportation infrastructure, mining activities, oil extraction, export crops, dams and so on (see also Chapter 5, Section 5.3.2.1). In addition, endowing local populations with the ability to manage their commons – with a strong commitment to conservation issues – is generally effective.

2.2.3 Farmers and agribusiness: the conservation paradox

According to Graeub *et al.* (2016) the broad term "family farming" can be divided into at least three groups with differing needs: "those that are well-endowed and well-integrated into markets ('Group A'); those with significant assets and favourable conditions but lacking critical elements (like sufficient credit or effective collective action) and who may not qualify for social safety nets ('Group B'); and land-poor farmers, who are primarily characterized by family subsistence and/or non-market activities and who require significant investment in social safety nets ('Group C')".

The current subsection will focus on Group A as the main, but not only, representative of developed and emerging countries. Because of the territorial extension of agriculture and livestock farming, farmers are considered major actors in land-use conservation and environmental policies (Mattison & Norris, 2005). In 2005, agriculture covered 40% of terrestrial land (Foley *et al.*, 2005) (see Figure 2.5). Agriculture is a major driver of land cover change (Gibbs *et al.*, 2010; Lambin & Meyfroidt, 2011b; Southgate, 1990; Tilman *et al.*, 2002) (see also Chapter 4, Section 4.3.3.2). Trade-offs exist between the necessity to feed over 7 billion human beings and to conserve natural resources.

A number of sociological studies have addressed the underlying attitudes behind farmers' practices (Ahnström *et al.*, 2009; Karali *et al.*, 2014; Kohler *et al.*, 2014; Sullivan *et al.*, 1996). These attitudes are not exclusively grounded in economic rationality, let alone the social reproduction of the production unit (understood here as the will to transmit the farm to next family generation). They are oriented by social context (Bieling & Plieninger, 2003; Burton, 2004), family history (Ahnström *et al.*, 2009), differing sensitivities regarding the environment (Siebert, Toogood, & Knierim, 2006), and economic opportunities (Karali *et al.*, 2014). Most of these case studies highlight a strong commitment to life "in open air" and a sentiment of proximity to nature. The longer a family has been settled in a region, the deeper the attachment to the land (Ahnström *et al.*, 2009) – also called "sense of belonging". These studies have shown that organic farmers are less likely to chiefly view land as a means to an end (i.e., producing food) (Sullivan *et al.*, 1996). However, in general, their privileged relationship with nature makes farmers averse to the idea that their activities are degrading land or should be supervised by national or local authorities (Léger *et al.*, 2006). Nevertheless, as shown in the following subsection, social expectations about the many dimensions of food production (including symbolic) can re-orient perception and practices to be more in line with a growing environmental concern (Michel-Guillou & Moser, 2006).

2.2.3.1 The consequences of the Green Revolution on farmers' perception

During the 1930s and after World War II, agriculture was considered a strategic issue for national food security. Nation-states became major actors in orienting and improving agricultural policies to achieve self-sufficiency. The Green Revolution – a major change in agricultural practice and technology, which occurred between the 1930s and the late 1960s – resulted in a change of perception toward the physical landscape of the land, which had been for centuries a family patrimony, endowed with meaning and memory (Juntti & Wilson, 2005). Feeding the world as a mission assigned to farmers was one among the new watchwords of the agricultural policies, with Farmer Unions' support and the involvement of agronomic engineers. Standardized and patented seeds prevailed as a rule (Boy, 2008). Many traditional landscapes were now perceived as obstacles to new farm machinery (Kohler *et al.*, 2014). Food became disconnected from local consumption to enter global markets.

Despite the visible negative environmental impacts (erosion, toxic runoff, biodiversity loss) and the threats to human health, anthropological investigations showed that farmers have often interpreted their farming practices as cooperation with nature, affecting the way they perceive the negative environmental impacts of their practices (Novotny & Olem, 1994; Silvasti, 2003). High yields, regular rows and absence of weeds have become the elements that define "a good farmer" in the eyes of a peer (Burton, 2004; McGuire *et al.*, 2013; Silvasti, 2003). This concept of "good farming" has become so important that, in some cases, croplands along roadsides (i.e., the visible plots) are treated with more herbicides than the other croplands (Burton, 2004).

This generation of farmers embraced the Green Revolution as a liberation from misery and "backwardness" (farmers' expression, associated with the old status of a "peasant"). The new worldview and professional pride in producing food and domesticating nature ("turning the land productive" – farmers' expression) has led them to prioritize utilitarian approach when adopting new practices (Ahnström *et al.*, 2009).

New environmental laws – such as the European Union Common Agricultural Policy's turn to incentivising eco-friendly practices – are frequently perceived as a burden (Burton *et al.*, 2008). This perception of environmental issues as being secondary has been reinforced by the fact that fuel, water and chemical inputs are often highly subsidized by governments or federations, thus sending contradictory messages to farmers (Bazin, 2003; Kirsch *et al.*, 2014).

Competition among farmers at a national and international scale was further encouraged by the agreement following the Uruguay round (WTO, 1995). It laid the basis for an open access market (Part III, Article 4), by discarding domestic support to agriculture (Part IV, Articles 6 and 7, and Part V, Articles 9 and 10) and limiting national adjustments through specific custom duties (Part V, Article 8 and Annex 5, Section A). The global agricultural market would now be overseen by a supranational Committee on Agriculture, a subsidiary of the World Trade Organization (Part XI, Articles 17 &18) (WTO, 1995).

Moreover, the agreement on intellectual property gave a major boost to biotechnologies, paving the way for corporations to be involved in the food production system (Lewontin, 1998; Desmarais, 2007). From then on, agriculture (which was until then a strategic national issue), became considered as a business like any other. In own words of the African Development Bank President: “agriculture is not a way of life. It is not a social sector or a development activity, despite what people may claim. Agriculture is a business. And the more we treat it as a business, as a way to create wealth, the more it will promote development and improve people’s lives” (Adesina, 2016). Confronted to the necessity of producing more produce at low prices, farmers became encouraged to invest in productivity, sometimes leading to a spiral of debt.

While farmers have long minimized the environmental impacts of their practices when compared with the necessity of producing food (Tucker & Napier, 2001), they are more and more inclined to adopt environmental concerns. Not only in high-income countries, but also in middle-income countries (Karali *et al.*, 2014; Paolisso & Maloney, 2000), a shift is induced by the changing rural population and more generally by the pressure of public opinion, which results in emphasis on health and consumption concerns over production. The gap between conventional farming practices and people’s awareness of the impact of the ‘productivist’ model on environment and food quality has been continuously increasing since the 1980s and the 1990s (Ward *et al.*, 1995). In other terms, the structuring concept of “good farmer” is now evolving to meet consumers’ expectations.

Although conservation agriculture (González-Sánchez *et al.,* 2017) can have some negative aspects (e.g., increased labour when herbicides are not used or lower yields in the years following conversion) (Brouder & Gomez-Macpherson, 2014; Giller *et al.*, 2009), an increasing number of farmers are opting for new practices to meet consumers’ willingness to pay for high-quality, low production footprint and locally-produced food, in developed as well as in emerging countries. In Sub-Saharan Africa and South Asia (Stevenson *et al.*, 2014), conversion to conservation agriculture is mostly meant to avoid land degradation and empower small farmers, when duly accompanied by private companies and investors (Jenkins *et al.*, 2004; Lambooy & Levashova, 2011), NGOs or government agencies. For higher income countries, provided they are correctly embedded in rural and/or urban social networks, farmers can escape from the spiral of debt and assume a more fulfilling social role (Knowler & Bradshaw, 2007; Padel, 2002; Strochlic & Sierra, 2007; Vogl *et al.*, 2015). Conversion to organic farming, adhesion to emerging social movements such as SlowFood (a grassroots movement in favour of locally and ecologically produced food) (for more details see http://www.slowfood.com/) or AMAP (French Association for the maintenance of a proximity agriculture, aiming at creating direct contact between producers and rural and/or urban consumers) (for more details see http://www.reseau-amap.org/), are potential pathways, as described in subsection 2.3.2.1.

Emerging concepts in agriculture, based on multifunctionality (Brouwer, 2004), are illustrative of this shift towards integrating environmental concerns in agricultural practices. The concept of “multifunctional agriculture” was adopted by the FAO (1999) and the EU Commission to foster an approach integrating landscape, biological connections and less environmentally-harmful practices. Traditional production practices that include these three aspects and contribute to the economy of the country already exist across Europe (e.g., olive gardens in Portugal, Greece, Italy and Spain) (Gu & Subramanian, 2014). Some developing countries also adopted this approach (Kriesemer *et al.*, 2016; Pham & Smith, 2013). Multifunctional agriculture is meant to integrate the economic, social and ecological aspects of land management. Two central concepts, those of land sparing and land sharing, have emerged and could be determinant (Hodgson *et al.*, 2010; Rey Benayas & Bullock, 2012).

Land sparing or "land separation" involves the agricultural intensification of existing land so that more land can be spared for wildlife conservation. It involves restoring or creating non-farmland habitat in agricultural landscapes at the expense of field-level agricultural production – for example, woodland, natural grassland, wetland and meadow on arable land. This approach does not necessarily imply high-yield farming of the non-restored, remaining agricultural land (Benayas & Bullock, 2012). See also 'Conservation agriculture' in Glossary.

High-yield farming requires less surface to produce the same quantity, or even more, assuming that modern technologies will continue to improve farming methods. Thus, arable land can be spared and restored to natural processes through fallows and afforestation. Land sparing is a trade-off between conventional methods, based on technological progress to overcome the limits imposed by the ecosystems, and the necessity to contain agricultural extension at the expense of natural processes (Adams & Mortimore, 1997; Bommarco *et al.*, 2013; Garnett *et al.*, 2013; Pender, 1998). Cultivation methods, in a context of land scarcity, could benefit from chemical and technological inputs (Brussaard *et al.*, 2010) – such as replacing bullocks and their manure by machinery (Gathorne-Hardy, 2016) – or could be used as an alternative for swidden fallow techniques in tropical contexts (Cardoso & Pinheiro, 2012; Ministério do Meio Ambiente, 2004). In other terms, intensification has the potential to simultaneously respond to farmers' demand for more productivity and competitiveness, while sparing land and preserving the environment (Barrett *et al.*, 2005; Foresight, 2011; The Royal Society, 2009; Rockström *et al.*, 2013; Roehrl, 2012; Smith *et al.*, 2010) (see also Chapter 7, Section 7.3.1). However, land sparing presents several limitations: it can spare ecological functions at the landscape level but not at the field level, and it tends to increase competition among farmers and make them even more dependent on off-farm resources (Benayas & Bullock, 2012).

Land sharing, on the other hand, is meant to restore ecological functions at the level of the field and to integrate agricultural production and natural processes. According to Benayas and Bullock (2012), five types of intervention follow the land sharing approach: "(i) adoption of biodiversity-based agricultural practices; (ii) learning from traditional practices; (iii) transformation of conventional agriculture into organic agriculture; (iv) transformation of 'simple' crops and pastures into agroforestry systems; and (v) restoring or creating specific elements to benefit wildlife and particular services without competition for agricultural land use." This approach enables crop production and wildlife conservation on the same land. There are several approaches to land sharing: organic farming, agroforestry, agroecology, biodynamic agriculture and permaculture – generally falling under the umbrella of "conservation agriculture" (see also Chapter 6, Sections 6.3.1.1 and 6.3.2.4).

Land sharing is a first step towards farming without agrochemicals, as it is meant to integrate natural processes into agricultural production. Examples include maintaining hedges and groves to fix the predators' guild and maintaining pollinators and using mixed crops to benefit from complementary processes (e.g., cereals and leguminous plants and/or fruit trees). The "Greening" shift of European Union's Common Agricultural Policy reform of 2013 is an innovation that makes the direct payments system more environment-friendly by subsidizing farmers who use farmland more sustainably and demonstrate care for natural resources (for more details, see https://ec.europa.eu/agriculture/direct-support/greening_en).

Both approaches have proven efficient for the restoration of degraded land and ecosystem services, but success has depended on the nature of landscape and varied from case to case (Barral *et al.*, 2015). What should be understood, however, is that from the biodiversity perspective, the best outcome may be the one where, at the landscape level, some areas are completely spared for biodiversity, some areas are shared with the emphasis on maintaining biodiversity and in some areas the production can be intensified (see e.g., Hanski, 2011; Kotiaho & Mönkkönen, 2017; Rybicki & Hanski, 2013) (see also Chapter 6, Section 6.3.1.2).

2.2.3.2 Agribusiness social and environmental policies: an asset for mitigation

According to the FAO, "agribusiness denotes the collective business activities that are performed from farm to table. It covers agricultural input suppliers, producers, agroprocessors, distributors, traders, exporters, retailers and consumers. Agro-industry refers to the establishment of linkages between enterprises and supply chains for developing, transforming and distributing specific inputs and products in the agriculture sector. Consequently, agro-industries are a subset of the agribusiness sector. Agribusiness and agro-industry both involve commercialization and value addition of agricultural and post-production enterprises, and the building of linkages among agricultural enterprises. The terms agribusiness and agro-industries are often associated with large-scale farming enterprises or enterprises involved in large-scale food production, processing, distribution and quality control of agricultural products" (FAO, 2013: 5-6).

A major change in agribusiness environmental policy was adopted after the Bhopal catastrophe (India, 1984) where an explosion in a pesticide plant belonging to a Union Carbide subsidiary officially killed 3828 (but Victims' Association count more than 20 000 collateral deaths). This catastrophe led the president of Union Carbide to declare at Davos, in 1991, that: "care for the planet has become a critical business issue – central to our jobs as senior managers" (Usunier & Lee, 2005:454).

Large corporations foster environmental consciousness by offering incentives to their suppliers. "For instance, responding to people's concerns about the destruction of rain forests and wetlands, multinational corporations such as Cargill and Unilever have invested in technology development and worked with farmers to develop sustainable practices in the cultivation of palm oil, soybeans, cacao and other agricultural commodities. This has resulted in techniques to improve crop yields and seed production" (Nidumolu *et al.*, 2009).

Many corporations respond to environmental concerns, especially when governments face stagnation of resources. On many occasions the private sector has been offered the opportunity to invest in market-based instrument and take a leading role in compensation, biodiversity offsets mechanisms (Jenkins *et al.*, 2012) and other schemes such as REDD+, ecotourism and/or sustainable forest and watershed management (Lambooy & Levashova, 2011). Lessening government involvement has led, in some areas, to the transfer of environmental management responsibilities to local or nongovernmental institutions, especially in Latin America (Liverman & Vilas, 2006).

The agribusiness sector responds mainly to social concerns by fostering programmes aiming at empowering small farmers to guarantee access to the global market. Five relevant concepts are highlighted in a report produced for the World Economic Forum, the New Vision for Agriculture's Partnership Model. This report underlined the necessity to provide solidarity and support to small farmers, especially in developing countries, through market-driven projects led by the private sector, rooted in viable business cases, integration of value chains that benefit all the stakeholders, and access to a globally connected market supported by an international network (World Economic Forum, 2016:3).

This initiative relies on multi-stakeholder conferences and workshops – associating farmers, rural outreach actors, policymakers and private sector leaders – and setting objectives for sustainable food production aligned with national objectives. For instance, at the May 2010 World Economic Forum on Africa, held in Tanzania, the multi-stakeholder taskforce was co-chaired by Tanzania's Minister of

Agriculture and Unilever's Executive Vice-President. In 2011, to achieve Mexico's agriculture goals, the Minister of Agriculture proposed a partnership to private sector leaders, among which were Nestlé and PepsiCo. In Indonesia, the partners included Monsanto, Cargill and Syngenta.

One of the main drivers of such a collaboration is the food security issue; according to which feeding 9 billion people by 2050 requires developing new technologies for improved productivity in a context of land and water scarcity (Godfray *et al.*, 2010; The Royal Society, 2009). In this context, large corporations play a major role by investing in research and development while bringing greater benefits to farmers and rural communities for social equity. To achieve these goals, agribusiness defends the idea of agriculture (including small farming) as a market-driven activity connected to global markets, by providing small farmers seeds, inputs and guaranteed purchase. Bringing benefits to small farmers thanks to technology and access to the market leads major corporations to implement local programmes based on soy, corn, palm oil - both for human and animal food. Box 2.4 (Section 2.2.4.3) gives the example of the Alliance for a Green Revolution in Africa, based on a public-private partnership. Developing and emerging countries are a promising market for GMO and agrochemicals, often presented by major corporations as "a technology for the poor" (Glover, 2010). Indeed, public opinion in developed countries (but not only) tends to be more and more reluctant to embrace biotechnologies and the use of agrochemicals, as shown by the "Monsanto Tribunal" held in The Hague on 15-16 October 2016 ("International Monsanto Tribunal," 2017) and a civil society initiative to promote the legal concept of "Ecocide" (or "crime against Nature"). This initiative was supported by 1200 organisations and signed by 90,000 petitioners (for further details, see http://en.monsantotribunal.org/signers-organisations).

2.2.3.3. Working towards transparency and ethical principles

The financial power of the research departments within agribusiness companies is quite enormous compared to public research funding in agronomy. The facts and data produced by researchers funded, directly or indirectly, by agribusiness companies (Simon, 2015) are in most cases legitimate, but generally focus on unidimensional evidence (e.g. restricted to nutrition facts without mentioning the environmental impacts and the risk of pesticide exposure in food) (e.g. Dangour *et al.*, 2009; Forman & Silverstein, 2012; Holzman, 2012). Moreover, by segmenting the studies, some companies do not disclose results about the "cocktail effect" of agrochemicals, nor do they conduct experiments based on public ordinary use, thus minimizing the level of exposure to pesticides and making it more arduous to identify more precisely the risks and impacts on human health (Damalas & Eleftherohorinos, 2011; Hernández *et al.*, 2013; Lee *et al.*, 2011).

Funding for public research generally glosses over areas where private research is perceived to be active. However, conflicts of interest have become an important theme in the scientific literature and community. A recent review of 672 scientific papers about genetically modified organisms (Guillemaud *et al.* 2016) showed that ties between researchers and the genetically modified crop industry were common, with 40% of the articles displaying conflicts of interest. The authors also found that the presence of conflict of interest was associated with a 50% higher frequency of outcomes favourable to the interests of the sponsoring company. Soon thereafter, another paper confirmed these conclusions (Krimsky *et al.*, 2017). For further discussion, see also Hicks (2017) and Wallack (2017).

Agribusiness specialized in chemical inputs and seeds also deploy a commercial strategy that considers farmers, not as primary producers, but as consumers (Diaz *et al.*, 2003). One of these strategies consists in offering packages of several products from the same brand, each tied to each other (UNCTAD, 2006), thus accentuating farmers' dependency on out-farm inputs and technical knowledge. It has been observed that this technical knowledge tends to disqualify local experiential knowledge, based on familiarity with soil and weather conditions (Desmarais, 2007; Marglin, 1996). Moreover, technical skills and understanding necessary for a proper use are extremely complex for farmers: studies conducted in Eastern Asia showed that farmers are not fully aware of the risks of using genetically modified seeds with high doses of pesticides on the development of secondary pests (for example on Bt cotton - see Ho *et al.* (2009) and Zhao *et al.* (2011)).

The dependency of farmers around the world is accentuated by the increasing concentration of the agricultural sector. According to the UN Conference on Trade and Development's report "Tracking the trend towards market concentration: the case of the agricultural input industry", less than ten major corporations (themselves results of mergers and acquisitions in the last 20 years) control more than half of the global seed market, with one corporation controlling 97% of the production of genetically modified seeds and three corporations controlling more than 50% of the global agrochemical industry (UNCTAD, 2006). The same report puts forward that the concentration of the sector sometimes leads to increased coordination and cooperation, such as contractual arrangements, alliances and collusive practices (UNCTAD, 2006). The report also states that the upstream production of seeds and agrochemicals is increasingly linked to the food processing industry: "it is also interesting to note vertical coordination upward and downward along the food chain, with the establishment of food chain clusters that combine agricultural inputs (agrochemicals, seeds and traits) with extensive handling, processing and marketing facilities" (UNCTAD, 2006).

These agrifood companies are generally reluctant to expose the ins and outs of the final products (Levin, 1999). A recent experiment with front-of-pack nutrition labels in France (Ducrot *et al.*, 2016) was met with strong opposition from major agrifood and distribution networks. Advertisements rarely mention actual facts: the information about production methods, socio-environmental impact, quality of ingredients, nutritional facts and types of additives is often incomplete or deficient. This tends to create a misperception of the origins and impacts of the food being consumed, thus hampering consumer awareness (e.g., the impact of meat consumption on climate change - Bailey *et al.,* 2014) (see also 2.2.1.3). Consumers in the lower economic classes are even less aware of the collateral effects of cheap and low quality food on weight, for instance (Cole et al., 2000; Guignon, 2017).

When it comes to land degradation, agrochemical and biotechnology industries are partly responsible (see also Chapter 4, Section 4.2.4.2), and yet their efforts in restoring degraded lands are very uneven. Moreover, greater complications come from the fact that degradation induced by the agrochemical industry or other market forces can apply to different levels of biodiversity: the level of landscape and field (ecosystem diversity), the level of specific biodiversity or genetic diversity.

Ecosystem diversity is strongly affected by open-field monocultures based on mechanization and heavy chemical inputs. Intensive monoculture reduces habitats, pollutes soils and rivers, and reduces soils' capacity to regenerate, due to the disappearance of its microbiota and microfauna (Beketov et al., 2013) (see also Chapter 4, Section 4.2.4.3). For instance, while using glyphosate in no-tillage agriculture is efficient against land degradation (see Section 1.3.4), the effects of this product on microbiota and aquatic ecosystems raises many concerns (Clearwater *et al.*, 2016). Regarding fertilizers, Reganold and Glover (2016) assert that soils in many regions across Sub-Saharan Africa are depleted to the extent that simply adding fertilizer will not improve soil health and may even make it worse (see also Box 2.4, Chapter 4, Section 4.2.9.5 and Chapter 5, Section 5.8.2.1).

Having a large share of the market gives corporations a potential leading role in reorienting practices and elaborating products less damaging to the environment and human health. Such a shift could be influenced by individual investors (by choosing ethical funds); by corporations (by negotiating between themselves a moral chart); or by governments, as suggested in 2012 UN Conference on Sustainable Development Declaration (point 47) (by creating a legal framework imposing transparency and fostering compensation, through restoration, and internalizing environmental costs in governmental taxes or in wholesale or retail prices). The liberalization of trade, in any of the cases, needs a high-level decision through international agreements.

Social and environmental concerns are now widely acknowledged by major corporations (WBCSD, 2008). However, remaining practices such as information retention – based on incomprehensive or loose legislations – tend to mislead consumers. This misinformation is further accentuated by the growing disconnect between food production, processing and consumption (Clapp, 2014; Henders & Ostwald, 2014).

2.2.4 Decision-making as a multifaceted (and endless) process

Decision makers at national or international levels have a major influence on the state of the planet, in matters of climate, degradation, overexploitation or sustainable use of natural resources.

This section begins with a summary of the concepts brought out in successive Earth Summits and the logic underlying international negotiations. Understanding what appears as political inertia (Brand & Görg, 2013) is central to shifting away from policies that aim to slow down degradation to implementing policies that seek to reverse it.

Another aspect explored in this section is the delay between scientific alerts and political decision (e.g. in Climate Change negotiations, 28 years - since 1988 - were necessary to take strongest but still non-coercive resolutions for its mitigation). The Alliance for a Green Revolution in Africa developed in Box 2.4 (Section 2.2.4.3) gives a strong starting point to explore the various trade-offs between international assessments and high-level recommendations on environmental issues and development priorities. In Section 2.3.1.1, we build on the ideas of development, and more specifically “sustainable development,” as the “fuzzy concepts”. A fuzzy concept contains more ideology than reality, generating multiple understandings, which can lead to damaging decisions.

2.2.4.1 From Stockholm to Rio + 20: the North-South tension

International negotiations on environment and climate have been shaped, since Stockholm 1972, by a North-South subjacent conflict and mutual distrust. This conflict is rooted in the first environmental report, *The Limits to Growth* (Meadows et al, 1972), published five to ten years after the independence of colonized countries, and in a period when a low oil price permitted accelerated growth in developing countries, such as “the Brazilian Miracle” (1968-1973). The conference was meant to raise global environmental concern and initiate a global eco-management strategy; and in practice, it catalysed an inflexion in environmental policies (White, 1982). It also introduced the idea of common but differentiated responsibilities.

The discussions in the Summit mostly revolved around development versus environment (Caldwell, 1972; Robinson, 1972; Rowland, 1973). The problems facing us today were already flagged in the preliminary debates and reports for Stockholm Conference (Hardin, 1968; Meadows *et al.*, 1972), and in the commentaries that followed its conclusion: demographic explosion, global climate change, collateral damages provoked by the Green Revolution (Joyner & Joyner, 1974).

Similar derivatives of the same discussion are ongoing almost half a century later and the problems policymakers have to solve today are still hampered by the same obstacles: difficulties in establishing effective supra-national environmental governance; a definition of sovereign rights that minimizes sovereign responsibilities (Caldwell, 1972; Coordinating Body for the Indigenous' Organisations of the Amazon Basin, 2014; Myers & Myers, 1982); and finally, guiding concepts based almost exclusively on economics (Robinson, 1972). In Stockholm, some developing countries strongly opposed environmental norms and taxes on the grounds that they could hamper socio-economic development (Robinson, 1972). For instance, José Augusto Araújo de Castro (1972), Brazilian Ambassador to the UN during the Summit, asserted that environmental issues concerned developed countries, while developing countries had no such problems. The necessities of achieving development was a priority to reduce poverty and reach Western standards of living (Castro, 1972) with a twofold ideological basis:

1. Environment was a matter of national priorities and developing countries' priority was development: "the implementation of any worldwide policy based on the realities of developed countries tends to perpetuate the existing gap in socioeconomic development […] and promote the freezing of the present international order. […] this permanent struggle between the two groups of countries persists in the present days and it is unlikely that it will cease in the near future" (Conca & Dabelko, 2015: 31).
2. Human beings stood above any environmental concern: "From the point of view of Man – and we have no other standpoint – Man […] is still more relevant than Nature" (cited in Conca & Dabelko, 2015:37). Hence, the idea that environmental concerns was a way for industrialized countries to impose restrictions on the development of other countries was deeply anchored (Head, 1977; Kennet, 1972; Kiss & Sicault, 1972).

By the time of the Rio Summit in 1992, developed countries had already accepted the idea of "common but differentiated responsibilities," according to which they should assume the financial burden of capacity building and technological transfer through the recently created Global Environment Facility (do Lago, 2009). The main achievement of this Summit, marked with optimism because of the end of the Cold War (Conca & Dabelko, 2010), were the United Nations Convention to Combat Desertification (UNCCD), the United Nations Framework Convention on Climate Change (UNFCCC) and the Convention on Biological Diversity (CBD).

Genetic diversity did not become the financial manna expected and the collective intellectual property of indigenous and local communities has not yet been clearly conceptualized (Görg & Brand, 2006) nor defined in law. It is only 24 years after Rio Conference that Brazil approved Law No. 13,123 on May 20, 2015 and Decree No. 8.772 on May 11, 2016, regarding this topic. The reluctance of corporations to invest in and pay for indigenous or local knowledge about biodiversity is partially due to the complexity of negotiating rights to access and to benefit-sharing (Rosendal, 2011).

Coming just after the events of 09/11 in the U.S., the Johannesburg Summit in 2002 demonstrated that terrorism could affect the perception of environmental urgencies, just as the oil crisis of 1973 spoilt the advances of Stockholm Summit. Being held in South Africa, the host country insisted on prioritizing poverty issues as a leverage for international aid (Seyfang, 2003), by linking biocultural diversity to the eradication of poverty (Conca & Dabelko, 2010; UNESCO, 2002) as a return to old assistance policy (do Lago, 2009). Other developing countries (G77) disagreed with this orientation (Visentini & da Silva, 2010).

According to many observers, the UN Conference on Sustainable Development, held in Rio in 2012, provided continuity to the Johannesburg Conference concerns about poverty. The first and second sections of the final declaration “The Future We Want” (UN, 2012) consist of 41 points, out of a total 283, none of which mention the word “environment” alone, but rather always preceded by the necessity of reducing poverty and improving social development, gender equality and children fulfilment (Point 2, 4, 6, 10, 11, 19, 30). Point 11 is illustrative of the multiple priorities of the Summit: "we reaffirm the need to achieve economic stability, sustained economic growth, promotion of social equity and protection of the environment, while enhancing gender equality, women's empowerment and equal opportunities for all, and the protection, survival and development of children to their full potential, including through education." The definition of “Green Economy”, a transversal concept widely used in the Declaration (point 26 and 58: b, g, h), insists on the necessary financial and technological support from developed to developing countries.

This last point strongly contrasts with the Stockholm principles, which asserted that sovereign rights came along with sovereign responsibilities. Another contrasting approach is about human demography: while Stockholm Declaration acknowledged the fact that demography was an environmental problem (see subsection 2.2.3.2 below), the Rio+20 declaration rejects all perspective of slowing down demographic growth, insisting on natality as a fundamental right (point 146), as well as universal access to assisted procreation (Point 145).

The focus on the human dimensions of sustainable development push us to think about different ways of conceptualizing socio-ecological relationship. As this chapter will further explore, we propose ecological solidarity (see Section 2.2.4.3 below) as an alternative paradigm. The next section revisits the demographic issue through the lens of environmental impact.

2.2.4.2 The taboo of demography as an environmental issue

Provided the average global fertility of humans declines to replacement level as projected, the human population will climb to 11.2 billion by 2100, from the current 7.5 billion. If fertility declines from what it is today, but remains half a child above the replacement level, human population will grow 120% and reach 16.6 billion by 2100 (United Nations, Department of Economic and Social Affairs, Population Division, 2015a, 2015b). This would lead, not only to an unsustainable demand in food and energy, but also to irreversibly transformed land through urban sprawl encroaching on croplands, thus threatening food security (Barbero-Sierra *et al.*, 2013; Doygun, 2009; Yeh & Li, 1999; Hasse & Lathrop, 2003; Jiang *et al.,* 2007; Johnson, 2001; Livanis *et al.*, 2006; Ministère de l’Environnement, 2017; Paül & Tonts, 2005; SAFER, 2013; Sheridan, 2007) (see also Chapter 4, Section 4.3.10)

How is human population size connected to degradation? For almost half a century, the growth of human populations has been blamed directly for environmental degradation (Diamond, 2005; Ehrlich, 1968; Hardin, 1968; Meyer & Turner, 1992; Robinson & Srinivasan, 1997). This led to years of discussion about the need to reduce population growth rates where they are high, often in developing countries. A UNEP report on the Economics of Land Degradation in Africa (ELD Initiative & UNEP, 2015), correlates land degradation and demographic growth: in 1962, each cultivated hectare supported 1.91 people; by 2009, one hectare supported 4.55 people (300% growth since 1962). Moreover, protected areas in poor countries tend to attract population for an easier access to natural resources, in the absence of better options (Joppa *et al.*, 2009; Struhsaker *et al.*, 2005; Wittemyer *et al.*, 2008), thus jeopardizing protection efforts (Liu *et al.*, 1999). Brashares *et al.* (2001) assert that where direct human influences put added pressure on species in remnant habitat patches, extinction rates are higher than those predicted by simple species and/or area models.

Many scholars objected to the focus on the number of people in developing countries. More attention is now given to how much each person consumes and how the Earth is used to support each person, especially in the context of growing meat consumption (Alexandratos & Bruinsma, 2012; Bailey *et al.,* 2014). If consumption per capita is important for degradation, then limiting consumption per person is also an appropriate goal (Ehrlich & Ehrlich, 2009; Ehrlich & Holdren, 2011). Both issues should be addressed in parallel, according to Ehrlich and Ehrlich (2009), along with the necessity of curbing economic growth by considering Earth's limits (Garcia, 2012; Meadows *et al.*, 1972). Both issues are equally complex as developing and emerging countries are striving to achieve Western standards of living and many developed countries are reluctant to change their way of life.

The declaration of Stockholm acknowledged the environmental problem caused by overpopulation in its 16th statement: countries should control their demography without affecting human basic rights. However, this matter was difficult to deal with, as the focus was mainly on developing countries' high natality rates. Once again, this approach was perceived as one more attempt from developed countries to interfere in developing countries' sovereign rights (Castro, 1972). The Stockholm Summit was followed by the World Conference on Population in Bucharest in 1974, where conflict led to the absence of a strong resolution (George, 1975). Soon after, the population problem was principally deemed a social and/or educational problem, excluding it from environmental discussions. A major step in this direction was the International Conference on Population and Development, held in Cairo in 1994. Its conclusion was that demography was a matter of education and empowerment of women, to be solved by international aid (Ashford, 2001; McIntosh & Finkle, 1995; Roseman & Reichenbach, 2010). "Since the use of family planning methods may prevent the prevalence of unplanned pregnancies, we call upon all national governments to reduce the need for abortion by providing universal access to family planning information and services" (UNPF, 1994, point 6). The Wall Chart developed by the Task Force on Basic Social Services for All (1997) focused on family planning, education, health care – addressing mainly the mother/child pairing and neglecting to address the connection between high birth rates, environmental degradation, migration flows and political instability.

Twenty years after Cairo, the International Conference on Population and Development (UNPF, 2014) published an assessment report on the Programme of Action adopted by the conferring parties. While the report acknowledged that a demographic transition occurred in many countries, it still highlighted that women's empowerment and gender equality were far from being achieved. A recent report by UNICEF (2014) dedicated to Africa, shows that the poorer the country and the social category, the less women have access to contraception – in Niger, for example, the number of women giving birth between 15 and 19 years old is 20,5%. According to the same report (2014:7): "in 2050, around 41% of the world's births, 40% of all under-fives, 37% of all children under 18 and 35% of all adolescents will be African – higher than previously projected." What is underlined is that family planning often fails to reach the most vulnerable fragments of the population and cannot fill the gap created by the lack of education combined with the lack of social inclusion. Hence, the question of human birth rate should be taken seriously – considering it both as a poverty issue and a high-priority environmental question (Crist *et al.*, 2017).

The main matter to discuss in developing countries is not only women's education or access to family planning, but the lack of retirement perspectives and, more specifically, the insecurity of people who fear to grow old without at least one child to support them. A solution, accordingly, could be to establish a universal retirement system, where pensions would be guaranteed even in case of political instability.

Agenda 21 (5.56) also mentions the link between birth rate and lack of access to education and family planning, but it is mentioned in the social and economic section, and old age issue is mentioned as a separate problem: "Proposals should be developed for local, national and international population/environment programmes in line with specific needs for achieving sustainability. Where appropriate, *institutional changes must be implemented so that old-age security does not entirely depend on input from family members*."

On the other hand, FAO report "The Future of Food and Agriculture," mentions that "social protection combined with pro-poor growth will help meet the challenge of ending hunger and addressing the triple burden of malnutrition through healthier diets" (FAO, 2017b: xii).

Demographic issue is even more of a delicate matter in those countries where having many children is an element of social prestige for men, especially, but not only, in polygamist countries (Fargues, 1994; Goldstone, 2010). Such a system of value cannot be changed by policies alone, but should be accompanied, where appropriate, by awareness-raising of the environmental impacts.

Demographic issues also apply to developed countries, especially where extensive welfare policies exist. Even after the demographic transition, the population does not diminish, partly because immigration from overpopulated or conflict-ridden countries compensates for the birth deficit (e.g., one million migrants and refugees were reported in Germany in 2016), and partly because family allowances are ideologically-anchored in pro-natalist policies going back to the time of the word wars, especially in France (Palier, 2005; Prost, 1984). The ghost of an unbalanced rate between retired and active workers also looms on these policies (Murray, 2008; Van De Kaa, 2006) (see Figure 2.11), leading to what Joseph Chamie, former director of the United Nations Population Division, called a "Ponzi scheme" (https://www.theglobalist.com/is-population-growth-a-ponzi-scheme/).

Figure 2 11 Pro-natalist campaign in Denmark.
Source: Spies Rejser (2014). https://www.spies.dk/do-it

Perhaps the key problem lies in the conception that birth limitation is invariably a violation of human rights. This perception is somewhat one-sided insofar as there is a distinction between controlling natality and not encouraging it. Family allowances are frequently proportionate to the number of children (Kalwij, 2010), hence discouraging natality would consist in limiting allowances to one or two children (Cochet, 2009). Not all birth limitation policies need to resemble the kind of totalitarian Malthusianism that is often assumed to accompany it, but rather can be stimulated through various socio-economic incentives and disincentives.

2.2.4.3 Towards new global concepts: ecological solidarity

For the purpose of this chapter, it is important to understand how a "common vision," as expressed in the Rio+20 Declaration can be based, forty years after Stockholm Summit, on reaffirming the necessity of economic growth to alleviate poverty, food production intensification thanks to agrochemicals and biotechnologies, liberalized global trade and other similar solutions. The Alliance for a Green Revolution in Africa programme, as discussed below (Box 2.4.) is an example of value-laden decision-making leading countries or economic federations to privilege one policy over others (i.e., a green revolution based on facilitated access to chemical inputs, mechanization, patented seeds and market-driven economy, as seen in Section 2.2.3.3).

Box 2.4 Diverging perceptions about the Alliance for a Green Revolution in Africa (AGRA) program

The Alliance for a Green Revolution in Africa launched in 2006, is mainly funded by the Rockefeller Foundation and the Bill and Melinda Gates Foundation. The current President of the African Development Bank declared, in 2016, that agriculture is a business and highlighted the importance of the Alliance for a Green Revolution in Africa for African food security (see http://www.afdb.org/en/news-and-events/article/agriculture-as-a-business-approaching-agriculture-as-an-investment-opportunity-15398/). The programme sets out to: encourage private investors in the agricultural sector; adopt hybrid varieties (e.g., maize and rice) tolerant to drought and pesticides; create local, African-owned seed companies that can multiply and distribute to retail shops locally; and adapt seeds and fertilizers to farmers, while training them in the use of these inputs. This view was expressed in a programmatic paper signed by two members of the Rockefeller Foundation and by the President of the African Bank of Development (Toenniessen *et al.*, 2008). The authors underlined that African farming systems were more diversified than in Southern Asia, where a Green Revolution occurred in the 1960s and 1970s, and led to a general improvement of farmers' condition and productivity (Pingali, 2012).

While the objective of an African Green Revolution is to ensure cereal self-sufficiency by 2050 (van Ittersum *et al.*, 2016) and integrate Sub-Saharan Africa into global markets as a competitive food producer, it is hard to find (ten years after the launch of the programme) openly positive assessments of the outcomes of this revolution. Most of the literature dealing with ex-post evaluation in several African countries (Ghana, Uganda, Tanzania and others) insist on the very context-specific successes or failures of this trend towards modernization and market-based policy (Dawson *et al.*, 2016; Moseley *et al.,* 2016; Moseley *et al.*, 2015). One of the inhibiting factors is the strongly anchored traditional seed exchange system, reluctant to adopt hybrid varieties (Louwaars & de Boef, 2012). Other authors underline the fact that AGRA should be accompanied by improvements in governance and democracy (Amanor, 2009; Markelova & Mwangi, 2010). A comparison between Asian and African Green Revolution shows that in the case of the former, the countries (especially India and Indonesia) were strongly supported and oriented by States, whereas Green Revolution in Africa relies more on markets for internal and external demand (Fischer, 2016). The same author asserts that African Green Revolution, contrarily to the Asian one, is not scale-neutral (i.e. of equal benefit to large-scale and small-scale farmers).

These structural problems – differing modes of production and social condition from one Sub-Saharan country to the other, along with generally poor environmental conditions – were acknowledged by the promotors of the project. Their anticipated response was that by increasing farmers' income thanks to a solid network of seed and fertilizer retailers and buyers, they would become economic actors in national and global markets while liberating workforce for industries (Toenniessen *et al.*, 2008), even in the absence of previous industrial revolution. Authors such as Frankema (2014) and Sheahan & Barrett (2017) are optimistic about the outcomes of today's improvements in technology, productivity and transportation, which could make an effective Green Revolution possible – able to improve farmers' condition along with the supply of a growing urban population.

On the other hand, the Alliance for a Green Revolution in Africa programme has been criticized by both scientists and international organizations. The same year Toenniessen *et al.* (2008) published their programmatic article, the International Assessment of Agricultural Knowledge, Science, and Technology for Development's Sub-Saharan Africa Summary for Decision Makers (Markwei et al, 2008) explicitly pointed at the danger of developing monocultures in Africa because of its social and environmental vulnerability, as did other researchers (Perfecto & Vandermeer, 2010; Scoones, 2009; Stigter, 2010). This assessment involved 400 researchers and dozens of national delegates (including those from Sub-Saharan Africa), who strongly recommended the adoption of agroecology as a sustainable practice.

The Alliance for a Green Revolution in Africa was also criticized by the special rapporteur on the Right to Food, in a statement submitted in 2009 to the Human Rights Council of the Office of the United Nations High Commissioner for Human Rights (Schutter, 2009). The conclusions of the Report on the Right to Food (Schutter, 2010) were identical. Finally, the International Panel of Experts on Sustainable Food Systems (IPES-Food, 2016) advocated for a paradigm shift from industrial agriculture to diversified agroecological systems (see also Chapter 6, Section 6.3.1.1). Many scholars also questioned such an orientation (Brown & Thomas, 1990; Holt-Giménez, 2008; Holt-Giménez & Altieri, 2006): small-scale farmers provide more than 70% of staple (FAO, 2014b) and are crucial for African food security (Altieri, 2009).

Indeed, a recent review showed that the cost of externalities provoked by pesticides is greater than the benefits of an increase in production (Bourguet & Guillemaud, 2016; Marcus & Simon, 2015; van Lexmond *et al.*, 2015). According to the Economics of Land Degradation report (ELD Initiative, 2015), the overuse and misuse of chemical fertilizer is a major cause of land degradation in Africa.

From a social point of view, some authors and institutions underline that the agroindustry leads to the displacement of rural populations to areas vulnerable to desertification and deforestation (Requier-Desjardins, 2008; Reuveny, 2007) – a situation worsened by climate change (FAO, 2008; IPCC, 2007) and by the absence of industrial jobs capable of receiving new workers. Land investment by multinational corporations can make the lives of small-scale farmers precarious because they are marginalized in the wider agricultural economy (Martiniello, 2013; Matondi *et al.*, 2011). It creates an underpaid rural class and also leads to rural exodus, increasing urban dwellers' economic insecurity, competition for subsistence and lack of options other than leaving agriculture all together (Bleibaum, 2010; Feintrenie *et al.*, 2014; Nonfodji, 2013; Richardson, 2010; Telenti, 2016) (see also Chapter 5, Box 5.4 and Section 5.3.2.5).

While the Alliance for a Green Revolution in Africa programme underlines that one of the major problems of African agriculture is crop losses, the FAO report on Food Wastage Footprint (FAO, 2013: 13) argues that the volume of food waste in agricultural production in Sub-Saharan Africa (35%) is equivalent to technologically-advanced European agriculture and less than Latin American agriculture. The main waste occurs in the phase of post-harvest handling and storage (35%), processing (12%) and distribution (12%). When the estimation is based on the number of calories, food loss in Sub-Saharan Africa goes up to 39%, while the main losses occur in the post-harvest handling phase (see Figure 2.12, Section 2.3.1.4). Food insecurity in Sub-Saharan Africa could (from these numbers) be considered a problem of conditioning and supply chain rather than one of production.

Almost inconceivably, for the first time in human history, geophysical, climatic and biological changes are outrunning the time of political decision-making and are reaching the point of no return, as recently confirmed by an opinion paper signed by more than 15000 scientists (Ripple *et al.*, 2017). Markets and economic competition still govern international relations, which in turn, often ignore the impacts of land degradation, overexploitation of natural assets and climate change on quality of life and human well-being (Chan *et al.*, 2012). Indeed, from Stockholm to Rio+20, and even UNFCCC COP21 on Climate Change, negotiators had a tendency to privilege a geopolitical outlook over an ecological one. One of the main reason is the aforementioned North-South tension and divide. Some of the principles or issues that could have been considered as efficient instruments to build a common ground for negotiation were not adopted because of this tension. While embargos or sanctions have been applied for ideological, ethical or security reasons, such embargos or sanctions are unheard of for environmental reasons (for further discussion on this see UN 2012, Point 58).

To explain these consensual positions, the concept of "hegemony" is worth exploring. This concept underlies yet another one, that of "common sense". Both of these were coined in the 1930s by Italian philosopher and dissident Antonio Gramsci. As Karriem (2009:317) put it: "for Gramsci (1971), ruling class hegemony is not based on force alone, but on a combination of coercion and consent. That is, a hegemonic class rules by incorporating some of the interests of subordinate classes. Intellectual or ideological leadership is not merely imposed; instead, subaltern classes consent to or are persuaded to accept dominant ideas as 'common sense'."

This "common sense" helps us to understand why, beyond geopolitical disputes, international negotiations tend to privilege the same responses, based on a common set of concepts, even if their efficiency is far from being constant or universal. New policy instruments could be used to facilitate international negotiations by fostering transnational and agreements. The concept of "ecological solidarity" (Naim-Gesbert, 2014; Thompson *et al.*, 2011) (see Glossary and Section 2.2.1.3, 2.2.4.3) would provide a useful framework for negotiating and implementing new and existing agreements (Pouzols *et al.*, 2014; Sarrazin & Lecomte, 2016).

"Ecological solidarity" is a French concept that needs further research. However, thanks to the revised law on National Parks of 2006 (Loi n°2006-436), this concept already exists in the French legal order. Some studies have been made to explore the possibilities to extend it as a fundamental principle in environmental law and as a powerful tool for policymakers. Originally, ecological solidarity serves to guide the definition of ecological territories around protected areas, but it could convey a more global message based on the commonly shared idea that humans are part of their environment. It has an ecological, social and moral dimension, which allows it to be placed among the ecocentric concepts (i.e., between biocentrism and anthropocentrism). As explained by Thompson *et al.* (2011): "from ecology based on interactions to solidarity based on links between individuals united around a common goal and conscious of their common interests and their moral obligation and responsibility to help others, we define ecological solidarity as the reciprocal interdependence of living organisms amongst each other and with spatial and temporal variation in their physical environment" (also quoted in Section 2.2.1.3). This concept has two main elements (one factual, the other normative): (i) the dynamics of ecological processes and biodiversity in space and time; and (ii) the recognition that human beings are an integral part of ecosystem function. This concept is worthy of attention from a legal point of view and for land restoration, because it relies on the paradigm of a collective duty of humans towards the environment.

The origins of the meaning of "solidarity" comes with the idea of debt. According to Bourgeois (1896), solidarism is based on the principle of the existence of a debt among generations. Hence, the principle of ecological solidarity in the legal order could integrate the idea that the current generation owes to the future ones, requiring legislators, judges, and other actors of the law to take into account the long-term consequences of their actions on nature and future generations. Meanwhile, as we will see in Section 2.3.2, in almost all countries in the world, concerned people acknowledge the difficulty for decision-makers to adopt global solutions. This is the reason why, new solutions emerge, many times inspired by traditional knowledge and practices, based on ecological consciousness, social concern and global citizenship.

2.3 Reality strikes back: impact of land degradation raises awareness and can modify perceptions

This section explores the main obstacles to the understanding of the reality of land degradation and the main reasons behind delays in decision-making. The section further explores how these delays can lead to informal social movements trying to adopt new practices and new forms of organization.

The first obstacle is that the temporal and spatial scales of land degradation sometimes make it difficult to perceive, as discussed in Section 2.2.1.3. As a result, inadequate understanding of land degradation and restoration – especially regarding timescales and long-distance connections – might cause policymakers and other stakeholders to create and support short-term and ultimately damaging policies.

The second obstacle is that concepts fundamental to land degradation or restoration are fuzzy (further discussion below in Section 2.3.1). This fuzziness can be worsened when private interests create uncertainty about the reality of environmental degradation, through lobbying or disinformation.

Finally, the incomplete understanding of land degradation and restoration may lead policymakers to perceive them exclusively from the perspective of food security. Indeed, global peace and political stability are threatened when basic needs of food and water are not adequately met due to land degradation (Barnosky *et al.*, 2012). Humans are thus posing a significant threat to themselves when they degrade the land. However, it is also important for policymakers to acknowledge that exclusive economic valuation of degradation and restoration may undervalue other dimensions important for a good quality of life (Wegner & Pascual, 2011a). The economic dimension is one among many dimensions of nature's contribution to people, which can be social, relational (Chan *et al.*, 2012), cultural (see Section 2.2.2), or intrinsic. This further emphasizes the importance and relevance of the multidimensional nature of human well-being (Jordan *et al.*, 2010).

In spite of these obstacles, information and awareness emerge and may elicit public reactions, especially when decision makers appear to be too cautious or risk averse (see Section 2.3.2.1). The capacity of civil society to organize and create alternatives can be a potent instrument to weigh on international decisions. However, many of these alternative solutions did not come to their full capacity as of yet.

2.3.1 Dealing with the multiple meanings of fuzzy concepts

This Chapter is about perceptions and how they gather into concepts. While many concepts intend to embrace the reality of human impacts on the environment, or to inform efficient tools for policy making, some can be misleading because they are 'fuzzy concepts' (Markusen, 1999). While they often facilitate consensual conclusions, this consensus is based on ambiguities and misunderstandings that can lead to future tensions. Examples are concepts like “sustainable development,” “human progress,” “precautionary principle” or “food security”. These concepts are vague and can be interpreted in a multitude of ways, hampering any coordinated collective action.

2.3.1.1 Sustainable development

In the words of UN World Commission on Environment and Development (WCED, 1987), sustainable development is “development that meets the needs of the present without compromising the ability of future generations to meet their own needs”. Today, however, sustainability is almost exclusively understood as having three dimensions: (i) economic development; (ii) social development; and (iii) environmental protection, as it was first captured by the United Nations in its Agenda for Development.

Sustainable development is perceived as a consensual issue, because nobody wants “unsustainable development.” This, however, does not mean that this concept is clearly defined, by default (Mebratu, 1998; Redclift, 2002; Robinson, 2004). What exactly does “sustainable” mean? Slowing down the rate of degradation? Maintaining accelerated developmental dynamics while considering environmental issues? In the forestry sector, for example, the concept of sustainability is frequently used to refer to securing a regular long-term supply of wood products from the forest ecosystems (Kuhlman & Farrington 2010; Kotiaho & Mönkkönen, 2017).

Moreover, as seen in Section 2.2.4.1 the concept of Green Growth adopted during Rio+20 clearly affirmed that economic growth was a priority to reduce poverty. Therefore, invoking sustainable development is the opposite of considering "the limits to growth": an unlimited development will affect sustainability in all cases. Development generates losing natural capital, dwindling natural resources, increasing social conflicts and growing inequalities (Le Billon, 2015). The Earth and its ecosystems have ecological limits beyond which the whole life-supporting system may lose its equilibrium (Schramski *et al.*, 2015).

2.3.1.2 Human progress versus ethics

While sustainable development is conceived as a mainly economic issue, “human progress” is seen as synonymous with “technological advance”. A human-centred perspective, placing humanity above all, has a tendency to oppose human progress to ecological issues, as expressed by Castro (1972). The problem with this humanistic vision of science and technology is that it does not include moral or ethical progress, which could compensate for this human self-centred (also called anthropocentric) vision of the planet (Rabhi, 2006, 2014). An alternative is “well-conceived humanism,” a concept advocated by a French anthropologist Claude Lévi-Strauss (1985), which would leave space for other species by not destroying the planet. Considering the interests of non-humans and allowing them to evolve and adapt would be an important step in a more inclusive human ethics and a first step to acknowledge nature’s intrinsic value (Burdon, 2011).

2.3.1.3 Precautionary principle versus “uncertainty principle”

The precautionary principle is a useful legal principle to enforce existing regulations when serious doubt exists. According to a common definition, the precautionary principle “enables rapid response in the face of a possible danger to human, animal or plant health, or to protect the environment” (Engle, 2008; EC, 2000). The precautionary principle is rooted in the idea that any decision that could affect the environment – and the services nature provides to humans – should be delayed until these impacts have been quantified. This applies mainly to new agrochemical molecules or genetically modified organisms that can have long-term consequences on the quality of soil and water, the trophic chains and/or pollinators (for past and current examples see the cases of DDT, chlordecone, neonicotinoids and even oceanic plastic particles).

The precautionary principle can be weakened, however, by over-emphasising "scientific uncertainty" and/or "lack of consensus" as a proof of internal contradictions (e.g., 97% of climatologists agree that climate change is anthropogenic, while the 3% who disagree are overrepresented in the media in the name of the "balanced" reporting). The invoked gaps in knowledge are often used as an argument to weaken the liability of industries when they cause damage (Mermet & Benhammou, 2005). This principle has been used by major companies or interest groups to discredit the scientific information against tobacco (Lee *et al.*, 2012), asbestos, junk food (Moodie *et al.*, 2013), neonicotinoids and, more recently, climate change. The uncertainty principle is efficient as it rests on the same elements as conspiracy theories: the best example is the "climategate" during Copenhagen COP 19 in 2009, when private e-mails were hacked and their meaning distorted.

Increasing knowledge through education is essential in solving environmental problems. However, it is important to keep in mind that while disinformation does not constitute knowledge, it nevertheless influences how people think about environmental issues. A good example comes from the International Panel on Climate Change (IPCC). The openness and massive IPCC scientific consensus about the causes of climate change struggles to counteract the large amount of attention the media gives to "sceptics", which yields significant influence on the social debate (Anderegg *et al.*, 2010; Antilla, 2005; Ehrlich & Ehrlich, 2009; Jacques *et al.*, 2008). Fuzzy concepts, disinformation and the "uncertainty principle," therefore, are dangerous as they can distort the urgency of situations and be used to avoid unpopular or costly decisions for the economy.

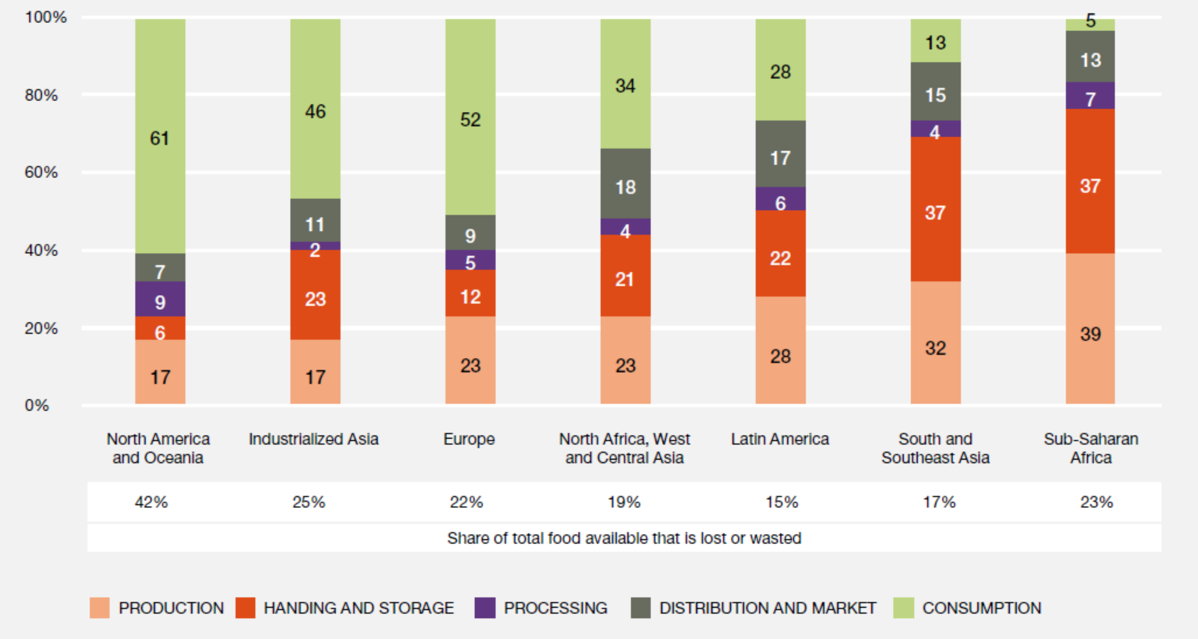
2.3.1.4 Feeding 9 billion people by 2050

Feeding 9 to 10 billion people by 2050 is a recurring theme in agriculture and international policies (see Chapter 5, Section 5.3 for more details). While the core meaning of food security is "sufficient food for all at all time" (Beddington *et al.*, 2011), the concept of food security is often boiled down to a need for producing more ("sufficient food"), missing the distributional aspects across people, space and time ("for all at all time") implicit in the food security concept.

Highly technologized and intensified agriculture is unquestionably part of the solution that needs to be further investigated, and can draw from techniques and technologies from biotechnology, engineering and nanotechnology (Beddington, 2010). Crop improvement, smarter use of water and fertilizers, new pesticides and their effective management to avoid resistance problems, and the introduction of novel non-chemical approaches to crop protection will certainly contribute towards achieving the needed increase in food production.

Feeding 9-10 billion people in 2050 while relying only on market-driven agriculture and progresses in new technologies and techniques (as seen in Section 2.2.3.3), goes against recent reports speaking in favour of a variety of approaches (Beddington *et al.*, 2011). Making more food available can be achieved by several complementary measures including reducing food waste (food purchased but thrown away) and food losses (from the crop field to the market). This vision is finding its way among international organizations such as the FAO, focusing on the urgency to reduce food waste and losses at a planetary scale (Koh & Lee, 2012; Parfitt *et al.*, 2010). Food waste is a major problem in developed countries (Hall *et al.*, 2009; Papargyropoulou *et al.*, 2014; Venkat, 2011; WRAP, 2009) (Figure 2.12). Hall *et al.* (2009) estimate food waste in the USA at 40%, with corresponding waste from associated production inputs such as energy and water. On the other hand, major problem in developing countries is not food waste, but food loss (Figure 2.14), mainly because of deficient distribution networks (Aulakh & Regmi, 2013; Kurwijila & Boki, 2003; Liu *et al.*, 2013). Even partial reductions in food losses and food waste have the potential to ease the pressure on needed increases. Information represented in Figure 2.12 can help public and private decision-makers target stages of the value chain where improvements could lead to the greatest reduction in food losses and waste.

Figure 2 12 Global food losses and food waste - extent, causes and prevention. WRI analysis based on FAO (2011). Source: From Lipinski *et al.* (2013).



Thus, among the fuzzy concepts, "food security" is certainly a powerful one, with ethical, moral and societal ramifications, especially when taken as a rationale for increasing production of food that will, in part, not even be consumed, while land and water are degraded to produce it. Food security is also frequently invoked by major actors of food production to justify agricultural productivity growth, sometimes to the detriment of organic farming or agroecology – which are said to be unable to deliver enough food to feed the world on their own and which are relegated as niche production systems for upper middle-class consumers. Such a position can be found in scientific papers, such as one by Connor (2013) where it is asserted that: "it is exactly because the world now faces an inescapable requirement to increase crop production by 70% on essentially current agricultural land to adequately feed an expected population of 9.2 billion by 2050 that low yielding systems [such as organic farming or agroecology] cannot contribute to the solution". Advocating that agricultural production should be increased by 70% to meet the challenge of feeding a human population growth overlooks the fact that highly technologized and intensified agriculture can have environmental and health impacts, including land degradation, loss of biodiversity, reduction of nutritional quality of food, and cannot be considered as the only solution to the food security problem (Horlings & Marsden, 2011).

In the meantime, many reports and papers support conservation agriculture (see Glossary) as a credible solution (Muller *et al.*, 2017). Organic farming, permaculture, biodynamic agriculture or agroecology defend local productions and human-scale farming, while having a positive environmental impact (Badgley *et al.*, 2007; González-Sánchez *et al.*, 2016; Halweil, 2006; Parrott & Marsden, 2002; Pretty &

Hine, 2001; Rundgren & Parrott, 2006). Recent studies tend to prove their potential not only in terms of productivity, but also in terms of farmers' income (e.g., in France - Dedieu *et al.*, 2017).

Today, environmental sustainability is commonly mentioned as a core component of successful business (Kareiva *et al.*, 2015), but the spread of disinformation is nevertheless still flourishing (Kareiva *et al.*, 2015; Lyon & Maxwell, 2011). Therefore, for the current assessment on land degradation and restoration, as well as for implementing measures to counteract degradation, it is important to recognize the threat of disinformation and find measures to overcome the disinformation through education and other appropriate measures.

2.3.2 Perception of policymakers' indecision and collective reactions

While conventional mainstream economics assumes that people act in their rational self-interest, recent studies from behavioural economics suggest that only 20-30 % of humans are purely selfish, while the remaining three quarters of people are egalitarians and composed of conditional co-operators (50%) and very pro-social individuals (20-30 %) (Meier, 2007). Members of these three quarters tend to evaluate self-interested individuals as evil individuals (Daly & Farley, 2011).

The emergence and empowerment of civil society is a major phenomenon since the 2000s (Schofer & Longhofer, 2011). This goes beyond being involved in an association or NGO. We call "civil society" the fraction of citizens who actively contribute to public debates (e.g., through demonstrations, new consumption patterns and life choices, blogging, petitions and so on). These concerned citizens realize that they could gain visibility and traction, not only by participating in demonstrations and social forums, but also through the internet (Ross, 2009). The example of Notre-Dame des Landes projected airport (Figure 2.13), for instance, led hundreds of militants to occupy the area for years, opposing to the destruction of wetlands.

Contrarily to usual political parties, these movements are leaderless (Fletcher, 2010; Sutherland *et al.*, 2013). They privilege new ways of life opposed to consumerism, such as veganism or less-meat initiatives, neoruralism (Méndez, 2012; Pandolfi, 2014), or the "degrowth" movement (Fournier, 2008; Schneider *et al.*, 2010). "Degrowth" or "downscaling" is a modern political concept, popularized and developed by French economist Serge Latouche (2009), which initiated a political, economic and social movement based on ecological economics (Georgescu-Roegen, 1971) and anti-consumerism. Such a proposal, being recent, obviously contains inconsistencies (Cosme *et al.*, 2017). Nevertheless it proposes a new economic strategy as a response to the limit to growth (Assadourian, 2012; Demaria *et al.*, 2013; Kallis *et al.*, 2012; Weiss & Cattaneo, 2017). Degrowth is also a theoretical frame applied to agriculture, invoking the necessity of downscaling and re-localizing production (Boillat *et al.*, 2012; Sekulova *et al.*, 2013). While these precepts are often discarded or marginalized, they are based on a simple fact: the energy input to produce food in an intensive system is often greater than the calories contained in finished food products (Amate & de Molina, 2013). In traditional systems of mixed cropping, such as Mexican *milpa* (corn, pumpkins and beans planted together), the net calories produced are greater than those produced by the same area under monoculture, as it does not require external energy input (Altieri *et al.*, 2012; Altieri & Toledo, 2011). Finally, a recent study exploring tens of scenarios point at the potential of conservation agriculture to feed the world, provided food waste and meat consumption are reduced (Muller *et al.*, 2017).

Figure 2 13 Zone à défendre (Area to protect) against the construction of Nantes' new airport, in Western France.

Mega infrastructure projects find strong opposition by civil society, not only through petition and protests but through actual occupation. Photo credit: Creative Commons licensed under CC BY-NC.

2.3.2.1 Towards alternative paradigms: downscaling production and consumption

Global warming and ecosystem collapse are two concerns that transcend social classes and interest groups. The example of food security, which is being treated throughout this assessment, transcends almost all socio-environmental issues, as the way food is produced and distributed will condition the future of humankind. Against the predominant way of thinking of food security (through technology, intensification and global competition), another paradigm has emerged since the 1990s – the "food sovereignty paradigm" – defined by transnational social movements as "the right of peoples to healthy and culturally appropriate food produced through ecologically sound and sustainable methods, and their right to define their own food and agriculture systems" (Forum for Food Sovereignty, 2007; Schiavoni, 2017). It received an important support from the United Nations Human Rights Council (Schutter, 2010), but also from the Food and Agriculture Organization (FAO, 2014a), and the International Panel of Experts on Sustainable Food Systems (IPES-Food, 2016). According to these reports, it would be necessary to reverse the productive trend adopted since World War II, maintain diversified systems of food production resilient to climate change, and try to shorten the distance from food to consumers, by revitalizing local food systems, particularly through agroecology (Altieri & Koohafkan, 2008) and agroforestry (see Chapter 5, Sections 5.3.3.1 and 5.5, and Chapter 6, Section 6.3.1.1).

"Agroecology" is a term used to describe the science of composition, function and structure of agroecosystems, the ideology of ecologically-friendly agriculture, the practices of farming that pay attention to conservation and the small-scale farmers' movements against industrialised modes of production in agriculture (Wezel & Haigis, 2002). Collectively, the science, ideology, practices and movements put forward an alternative worldview of how agriculture should be practised (Altieri & Toledo, 2011; Claeys, 2013; Rabhi, 2006; Schiavoni *et al.*, 2016). This alternative is primarily a reaction to the undesirable consequences of industrialised agriculture, including land degradation. In this context, a wide variety of terms have been used to describe these conservation agriculture alternatives: biodynamic, community based, ecoagriculture, ecological, environmentally sensitive, extensive, farm fresh, free range, low input, organic, permaculture, sustainable and wise use (Pretty, 2008). Until recently, these methods of sustainable agriculture were seen as alternatives rather than good practice principles in mainstream agriculture. Nevertheless, a recent trend in UN programs foster a generalization of sustainable and diversified practices (FAO, 2014b, 2017; IPES-Food, 2016). Further research is needed to understand its role in carbon sequestration (Govaerts *et al.*, 2009).

Alternative practices in agriculture also have an ideological dimension. They are now strongly supported by international small farmers organizations, such as La Via Campesina (created in 1993), including unions of developed as well as developing countries around an ideal of restoring traditional knowledge, gender equality and employment opportunities (also see Chapter 5, Section 5.2.3.2), virtuous environmental practices through agroecology (Perfecto & Vandermeer, 2010; Rabhi, 2006; Benayas & Bullock, 2012), and farmer empowerment (Altieri, 2009; Altieri & Toledo, 2011; Desmarais, 2010; Rosset & Torres, 2013). These movements try to create new community models, organized around the exchange of goods, food and services in moral (or social) economy (Edelman, 2005). La Via Campesina is an expression of collective and leaderless resistance; it associates indigenous and peasant movement, united in their claim for land and respect. Altieri and Toledo (2011) talk about a "new agrarian revolution" structured around agroecology. These new movements opt for a political resistance based on social practices, without directly confronting the neoliberal system. Williams (2008) defines this attitude as a "withdrawal from capitalism". The objective here is not the appropriation of the means of production, but the creation of a society with predominant values of solidarity, a non-materialistic approach to well-being based on sociability and respect for human and natural balance.

2.3.2.2 Creating active environmental subjects: the empowerment of civil society

At the global level, a new concept, "environmentality" (Agrawal, 2005a, 2005b) acknowledges the rise of "environmental subjects": people who no longer accept staying passive while the environment is threatened by global markets and unsustainable patterns of consumption (Fletcher, 2010).

Indeed, long supply chains (in kilometres or number of intermediaries) increase the profits of multinational corporations at the expense of producers, consumers and the environment (also see Chapter 5, Section 5.3.2.5). "Producing locally, consuming locally" is a new concept which is gaining influence in number of developed countries, including the USA, Canada, Germany, Italy, Spain and France (Deléage, 2011; Willer *et al.*, 2010) – although the contribution of food transportation on the carbon footprint remains relatively low compared to food production (Weber & Matthews, 2008), particularly for animal sources of proteins (Nijdam *et al.*, 2012). Raising ecological awareness is thus needed and could be achieved by making consumers aware of both their responsibility in environmental degradation and their power to solve the issue by adapting their behaviours (Peattie, 2010). In particular, the limitation of degradation, and accelerated restoration, can be addressed by either promoting sustainable practices by changing consumption behaviours, or a combination of approaches. Although progress was made in reducing the use of resources to produce goods, to date, the growing population has been increasing its consumption, thus limiting the positive impact of more efficient and sustainable production systems (Mont & Plepys, 2008).

Policymakers have a leading role in promoting new ideas and concepts about what would be our general interest, and enforcing them so that they become new realities (Fukuyama, 2014). This can be achieved through strong environmental policies. Some regulatory and economic instruments (e.g., taxes, products charges and standards) are meant to address both producers and consumers (Assadourian *et al.*, 2010; Mont & Plepys, 2008). Lenzen *et al.* (2012) argued that while international laws and regulations exist for the trade of endangered species, the same type of control could be applied on the trade of commodities whose production has a strong negative impact on biodiversity, including with policies targeting the consumers of products causing degradation. Wallner *et al.* (2003) show that ecological awareness might not change habits, but it does facilitate acceptance of more eco-friendly laws.

Promoting sustainable consumption is a major issue (UNEP, 2014). It requires revisiting some aspects of WTO agreements (see Section 2.2.3.1), especially when it comes to distorted competition. Several mechanism exist to promote sustainable or “green” consumption (Lebel & Lorek, 2008; Peattie, 2010). For instance, certifications and labels (e.g., FSC, Rainforest Alliance) aim to inform consumers, by raising ecological or environmental awareness and thus shifting purchasing behaviour towards products with reduced environmental impact (Lebel & Lorek, 2008). However, mechanisms for sustainable consumption appear most efficient when consumers are already sensitive to environmental issues (Rex & Baumann, 2007), otherwise the share of “green products” on the markets remains relatively low. Tukker *et al.* (2008) argued that sustainable production-consumption conflicts with the mainstream beliefs and paradigms about growth, markets and the institutions regulating them, and called for more evidence-based discussions.

This leads us to the major levers that policymakers could use: promoting new social norms, including through targeted taxes and an education, based on renewed ethical principles. People tend to adapt their behaviours to those perceived as common, normal, and/or morally and socially right (Goldstein *et al.*, 2008; Peattie, 2010; Schultz *et al.*, 2007). An education built upon ethical principles such as solidarity and cooperation would be a first step towards new perceptions. The current dominant model of social prestige is based ob raising the pattern of consumption to acquire expensive and/or rare products (e.g., expensive cars or clothes, ivory or rhino horn powder). An alternative model, based on a moral economy (Edelman, 2005), is emerging and growing with each year. This economy values social life, sobriety and solidarity, and is inspired by traditional populations and practices. Its aim is to consolidate social cohesion through community, mutual aid and production-consumption systems (Lebel & Lorek, 2008; Mont & Plepys, 2008; Tukker *et al.*, 2008). Education and awareness can contribute to transform passive citizens into environmental, proactive players, who feel concerned about their own impacts and responsibilities. Governments urgently need to take the lead in fostering an education system that values cooperation and solidarity, instead of competition and models based on high levels of consumption as a symbol of successful life.

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Chapter 3

Direct and indirect drivers of land degradation and restoration

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Executive Summary

Human activities now represent the most important force shaping the degradation of ecosystems in all of the world´s major biomes (*well established*). Long-established drivers of land degradation continue to increase across much of the world, including agricultural activities {3.3.1, 3.3.2}, driven by increasing demands for food and bioenergy. More recent global change drivers, such as climate change and atmospheric nitrogen deposition, further exacerbate impacts {3.4}. We are now in a qualitatively different and novel world, compared to only a few decades ago, and the combination of drivers creates significant challenges to restore degraded land and mitigate further degradation (*established but incomplete*). Few, if any, areas of the world are now free of some form of human influence (*well established*) and some systems are experiencing unprecedented challenges.

Changes in the extent and severity of both land degradation and restoration commonly result from multiple underlying social and economic factors – indirect drivers, many of which occur in places distant from where the impacts are felt (*well established*) {3.6.4}. Demand for food imports is increasing across much of the world. This high dependency on imported commodities means that a large share of the environmental impacts of consumption is felt in other parts of the world. The physical quantity of goods traded internationally only represents one third of the actual natural resources that were used to produce these traded goods. The sustainability of the commodity production systems that support global supply chains is thus substantially shaped by the sourcing and investment decisions of market actors who may have little direct connection to the production landscapes (*established but incomplete*). Moreover, the globalized nature of many commodity supply chains potentially elevates the relative importance of global-scale factors such as trade agreements, market prices and exchange rates, as well as distant linkages related to buyer and investment preferences, over national and regional governance arrangements and the agency of individual producers (*inconclusive*). Addressing this complexity to avoid and reverse land degradation therefore requires the building of effective multi-sector and multi-stakeholder partnerships that span national boundaries (*established but incomplete*) {3.6.6}.

Economic growth and per capita consumption, more than poverty, is one of the biggest threats to sustainable land management globally (*established but incomplete*) {3.6.3, 3.6.4}**.** Extreme poverty, combined with resource scarcity, can contribute to land degradation and unsustainable levels of natural resource use, but is rarely the major underlying cause *(well established)*. Many of the most marked changes in how land is used and managed come from individual and societal responses to economic opportunities, such as a shift in demand for a particular commodity or improved market access, moderated by institutional and political factors *(established but incomplete*). For example, clearance of native vegetation and land degradation across much of Latin America and Asia is linked to agricultural expansion and intensification at a commercial scale for export markets (*well established*). Reducing poverty, although a priority for sustainable development, is insufficient to mitigate land degradation if not accompanied by additional measures. Concurrently, rising per capita consumption levels can exacerbate degradation. Efforts to reverse degradation therefore require a combination of local and regional poverty-alleviation strategies, including the adoption of pro-poor food production systems, together with efforts to improve the enforcement of public regulations for sustainable land uses, and strengthening the accountability of global market actors in effectively supporting such strategies.

The highly interconnected and globalized nature of indirect drivers of land degradation and restoration means that the outcome of any global, regional or local intervention can be highly unpredictable, yet contextual generalizations are possible (*established but incomplete*) {3.6.2.3, 3.6.3}**.** The ways in which land is used in one part of the world can be highly sensitive to sudden, unexpected changes in economic and institutional factors elsewhere (*unresolved*). For example, changes in currency exchange rates, and cascading effects on the profitability of a given commodity, can markedly accelerate or decelerate the clearance of native vegetation for agriculture within a single year {3.6.2.3}. The sudden imposition of trade restrictions (e.g., due to disease controls), can have a similarly marked impact. However, with an improved understanding of the interactive effects amongst different drivers, it is possible to make predictions that are valid under a certain range of conditions. For example, agricultural intensification and agroforestry practices can help reduce the pressure on remaining areas of native vegetation under certain conditions (such as inelastic demand for staple crops), but unless such measures are coupled with increased enforcement of land-use policies they can result in a rebound effect that increases pressure on natural resources (*established but incomplete*) {3.6.3}.

Land degradation in any given place is rarely the consequence of a single anthropogenic driver, but is instead the result of a diverse and frequently mutually-reinforcing set of human activities and underlying drivers (*well established)* {3.4.5, 3.5, 3.6.2.1}**.** Typically, at least three types of indirect driver, such as economic, technological and institutional, underpin any direct driver of land degradation or restoration (*established but incomplete*). The complexity of drivers that commonly underpin land degradation highlights the fact that single factors, such as high rural population density, rarely provide an adequate underlying explanation on their own for observed impacts (*established but incomplete*) {3.6.3}. Land degradation is typically the result of multiple direct drivers, especially in instances of severe degradation (e.g., where land-use intensification drives increased species invasions and increases in fire frequency). This combination of drivers has resulted in large expanses of economically important grazing lands, including in North America, being transformed to fire-prone annual grass monocultures (*well established)* {3.3.7}. The multi-causality of land degradation requires commensurately holistic policy responses that operate across multiple scales and combine both regulatory and incentive based measures (*established but incomplete*).

Rapid expansion and inappropriate management of agricultural lands (including both grazing lands and croplands), especially in dryland ecosystems, is the most extensive land degradation driver globally (*well established*) {3.3.1, 3.3.2}**.** The expansion of grazing lands has largely stagnated globally with evidence for an approximate 1% decline in grazing land area over the past decade. Grazing pressure has been stable or only moderately increasing across the major land areas globally, although there are regional exceptions such as Southern Asia. Over half of grazing lands occur in dryland environments that are highly susceptible to land degradation (*established but incomplete*) {3.3.1.3}. More recently intensification and increasing industrialization of livestock production systems, especially in developed countries, has resulted in an increasing reliance on mixed crop-livestock production systems and industrialized "landless" systems. As a result, 35% of global crop production is now allocated to livestock feed. Globally, fertilizer and pesticide use is expected to double by 2050 {3.3.2.2}. Marked drops in nitrogen-use efficiency (change in yield per unit of fertilizer input) in many parts of the world, particularly the Asia Pacific region, often accompanied by continued excessive fertilizer application, underscore the critical importance of sustainable agricultural practices, including conservation agricultural techniques, to maintain yield improvements (*established but incomplete*) {3.3.2.3}.

Increases in consumption levels of many natural resources underpin increasing levels of degradation in many parts of the world (*well established*), with slow rates of adoption of sustainable production systems (*established but incomplete*) {3.6.2.2, 3.6.3.2, 3.6.4.2}**.** Projections to 2050 suggest that one billion ha of natural ecosystems could be converted to agriculture by that time. More than half of agricultural expansion in the last three decades has occurred in relatively intact tropical forests. Economic growth in the developing world is projected to double global consumption of forest and wood products by

2030, with demand likely to exceed production in many developing and emerging economies in Asia and Africa within the next decade. Traditional fuelwood and charcoal continue to represent a dominant share of total wood consumption in low income countries, up to 70%, especially in Sub-Saharan Africa (*well established*). Under current projections efforts to intensify wood production in plantation forests, together with increases in fuel-use efficiency and electrification are only likely to partly offset the pressure on native forests (*unresolved*). Adoption of more sustainable production systems continues to be slow, as seen, for example, by a slowdown in the expansion of the area of certified forests.

More than half of the terrestrial surface of the Earth has fire regimes outside the range of natural variability, with changes in fire frequency and intensity posing major challenges for land restoration (*established but incomplete*) {3.3.7}. The frequency of fires has increased in many areas – exacerbated by decreases in precipitation – including in many regions of humid and temperate forests that rarely experience large-scale fires naturally. Some changes in fire regimes, particularly in tropical forests, are sufficiently severe that recovery to pre-disturbance conditions may no longer be possible.

Increases in international trade, intensification of land use and urbanization have meant that few areas of the planet are free of invasive species (*established but incomplete*) {3.3.8}. Nearly one fifth of the Earth´s surface is at high risk of plant and animal invasion, including many biodiversity hotspots. Climate change, including increased nitrogen deposition and changes in CO_2, as well as increases in fire frequency with rising temperatures in many areas, are all likely to increase invasions {3.4}. Once established, the eradication of many invasive species is often very expensive, if not impossible, underscoring the need to develop proactive strategies to pre-empt invasions, including through inspections, research and education.

Activities related to industrialization, infrastructure development, urbanization, and many extractive industries result in complete transformation of ecosystems, accompanied by near or complete loss of biodiversity and ecosystem function and the services those ecosystems provide (*well established*) {3.3.6}**.** Infrastructure, industrial development and urbanization activities, often replace natural ecosystems with impervious or contaminated surfaces such as asphalt, concrete and rooftops, leading to the one of the most severe forms of land degradation in the form of soil sealing. Built-up areas, which are dominated by sealed soils, currently occupy nearly 0.6% of the global land surface. If population densities in cities remain stable, the extent of built-up areas in developed countries is expected to increase by 30% and triple in developing countries between 2000 and 2050. Under more extreme scenarios of increasing population density and economic development, the extent of built-up areas globally may increase to over 2% of the global land area over this same time period. New urban design and green technologies that incorporate features that promote sustainability and delivery of ecosystem services can play an important role in restoring some of the ecosystem functions and services of built environments.

The importance of climate change for land degradation is most prominent through its role in exacerbating the impacts of other human activities (*established but incomplete*) {3.4}**.** The exacerbating effect of climate change on the impact of degradation drivers, including land clearance and intensive farming techniques, can be felt both through chronic impacts and directional changes – like temperature changes, leading to shifts in species range sizes, as well as changes in average precipitation levels, atmospheric CO_2 and nitrogen deposition – and acute impacts through extreme weather events of flooding, drought, and other natural disasters (*well established*). Heavy rainfall events and storms as well as heat waves and droughts are predicted to increase in frequency over several parts of the globe, with cascading effects on the frequency, intensity, extent and timing of other drivers such as fires, pest and pathogen outbreaks, species invasions, soil erosion and landslides (*established but incomplete*).

The last decade has witnessed a rise in consumer-driven demand for sustainable land use and land management, as well as commitments to restore degraded land that is unprecedented in human history (*well established*) {3.6}**.** In the last decade hundreds of companies have made pledges to reduce their impacts on forests and on the rights of local communities, with many committing to eliminate deforestation from their supply chains entirely by 2020. In the same period, many governments and civil society groups have made ambitious commitments to restore hundreds of millions of hectares of degraded land. New players, such as the finance sector, who until recently have been completely detached from the mainstream sustainability agenda are also starting to make explicit commitments to avoiding environmental harm. The overall impact of these voluntary measures remains to be assessed but they offer a vital window of opportunity for reversing degradation trends and placing economies on a more sustainable footing – especially as large areas of marginal agricultural become increasingly abandoned with ongoing development (*unresolved*).

3.1 Introduction

Land degradation refers to the many processes that drive the decline or loss in biodiversity, ecosystem functions or services, and includes the degradation of all terrestrial ecosystems. This assessment is concerned with changes in both the extent of human activities and behaviours that drive land degradation, as well as the type or intensity of those activities. Specifically, we describe the different drivers of land degradation and the extent and severity of these drivers across biomes, as well as how these drivers link to declines in biodiversity and ecosystem functions and services (described fully in Chapters 4 and 5). Exploration of the interactions among drivers that further exacerbate the functioning of ecosystems, including the role of climate change as a threat multiplier of degradation drivers, are a key focus in this chapter. A thorough examination of land degradation drivers provides a critical first step toward an improved understanding of how we may successfully restore degraded lands and avoid further degradation in the future (see Chapters 6 and 8).

Drivers of land degradation include all external factors that can act either directly or indirectly to result in declines in nature (i.e., biodiversity), anthropogenic assets, nature's benefits to people and quality of life (Box 3.1, Table 3.1). **Direct drivers** have an unequivocal effect on the structure, function and composition of ecosystems. Direct **natural** drivers are those that are not the result of human activities (e.g., landslides, tectonic activity) and are beyond human control, while human-induced or **anthropogenic** direct drivers can most easily be conceptualized as the set of activities performed by humans that in turn give rise to degradation and restoration processes, such as land-use and land management activities (e.g., land clearance, introduction of invasive species, fire suppression).

Box 3.1 Definition of degradation drivers in the context of the IPBES Land Degradation and Restoration Assessment.

Drivers (of change): All the external factors that cause change in nature, anthropogenic assets, nature's benefits to people and a good quality of life. They include institutions and governance systems and other indirect drivers and direct drivers (both natural and anthropogenic).

Direct drivers: Drivers (both natural and anthropogenic) that affect nature directly (sometimes also called pressures).

Natural direct drivers: Direct drivers that are not the result of human activities and are beyond human control (e.g., natural climate and weather patterns, earthquakes, volcanic eruptions).

Anthropogenic direct drivers: Elements of direct drivers that are the result of human decisions and actions (e.g., land clearance, intensification of agriculture, harvesting of wild populations).

Indirect drivers: Drivers that operate by altering the level or rate of change of one or more direct drivers. They are the underlying causes of environmental change that are often external to the ecosystem in question (e.g., access rights to land, economic and agricultural policies, international agreements).

Indirect drivers, on the other hand, are factors that underpin direct drivers of change (see Box 3.1) and include key institutional and governance structures in addition to the social, economic and cultural context in which land degradation occurs. Indirect drivers operate more diffusely and are the forces that underlie and shape the extent, severity and combination of direct drivers that operate in a given place. Indirect drivers operate almost always in concert and across multiple scales and varying levels of proximity from the location in question, from the global (markets, commodity prices, consumption patterns), to the national and regional (demographic change, migration, domestic markets, national policies, governance, cultural and technological change) to the local (poverty, economic opportunities).

This chapter begins with an overview of the different direct drivers of land degradation, followed by a thorough examination of each driver. Although natural direct drivers may result in land degradation, the chapter focuses primarily on direct anthropogenic drivers and examines the extent and severity of the direct drivers of land degradation across biomes and geopolitical boundaries, reports on how these drivers have changed over the last century and describes how direct drivers may interact either singly or in concert to accelerate the rate of land degradation. This full examination of direct drivers of land degradation sets the stage to understand linked degradation processes and the resulting changes in biodiversity and ecosystem functioning addressed in Chapter 4.

Altering the nature, extent, and rate of change of direct degradation drivers in order to promote restoration of degraded lands occurs through indirect drivers. Indirect drivers, or human actions and decisions, provide the mechanism by which to avoid, reduce, and reverse land degradation. This includes indirect drivers that can be altered through interventions including changes to policies, governance and institutional structures, and markets, and connects to the discussion of policy responses in Chapters 6 and 8. Indirect drivers, specific to each direct driver, are assessed in the relevant section followed by an overarching assessment of the importance of indirect drivers at the end of the chapter.

3.2 Overview of drivers of land degradation and restoration

Land degradation and restoration processes are shaped by both natural drivers and anthropogenic direct drivers, which are in turn shaped by indirect drivers (Box 3.1, Table 3.1).

In some instances, land degradation can arise as a result of inherent natural processes such as earthquakes and volcanic eruptions, landslides, floods, hurricanes, typhoons and the periodic outbreaks of pests and pathogens. Such events are not necessarily the result of human-induced changes to ecosystems; they occur episodically with periodicities ranging from years to millennia, and can result in land degradation and the loss of biodiversity and ecosystem functions and services. However, it is important to recognize that in some cases, these same events can be exacerbated by anthropogenic activities, as in the case of landslides that result from road building or clear felling, or pest and pathogen outbreaks that arise following their introductions to new habitats by humans. The impacts of many natural drivers are also exacerbated by the effects of human-induced climate change (see Section 3.4).

The most widespread drivers of land degradation globally are those that are directly linked to human activities (Table 3.1). The spatial scales over which different direct anthropogenic drivers are manifested ranges from local (e.g., land conversion or localized mineral or sand extraction) to regional (e.g., invasive species) or global scales (e.g., climate change). The outcomes of different drivers for changes in biodiversity and ecosystem services are similarly varied with multiple interacting drivers often involved in shaping a particular outcome. A single driver can influence multiple degradation or restoration processes, while at the same time multiple interacting drivers can influence the same process (Table 3.1).

Indirect drivers are the ultimate underlying causes of land degradation. They arise from the way human societies function and organize themselves and interact with nature at different scales, and are typically external to the ecosystem in question (Díaz *et al.,* 2015) (Table 3.2). They are considered indirect drivers because they do not affect nature directly in most instances, but operate by altering the extent, severity or rate of change of one or more direct anthropogenic drivers (Díaz *et al.,* 2015).

Table 3.1 Anthropogenic direct drivers of land degradation and restoration.

Anthropogenic direct driver	Example subcategories of direct driver	Examples of linked degradation processes
Grazing land management	Change in extent of grazing lands, livestock type, stocking rates, rotation regimes, supplementary feeding, irrigation and water management, pasture improvement	Fragmentation of native vegetation, loss of biotic diversity, soil erosion, soil compaction, change in soil and nutrient content, salinization, change in runoff and infiltration regimes of water, nutrients and agrochemicals, invasive species, change in fire regimes, woody encroachment
Croplands and agroforestry management	Change in extent of croplands and agroforestry systems including drainage of wetlands, crop type, crop rotation and/or sequence, soil management, harvesting and fallow cycles, agricultural inputs, irrigation	Fragmentation of native vegetation, soil erosion, soil compaction, change in soil nutrient content, change in runoff and infiltration regimes of water, nutrients and agrochemicals, soil and water salinization, sedimentation, water contamination, species invasions, change in fire regime, atmospheric pollution and deposition
Forests and tree plantation management	Change in extent of managed and planted forests, harvesting intensity, rotation regimes, silvicultural techniques	Fragmentation of native vegetation, soil erosion, soil compaction, change in soil nutrient content, change in runoff and infiltration regimes of water, nutrients and agrochemicals, sedimentation, water contamination, change in species composition and invasions, changes in above-ground and below-ground biomass, changes in carbon stocks, fire regime change
Non-timber natural resource extraction	Fuelwood harvesting, hunting, harvesting of wild foods, fodder, medicinal and other products	Change in species abundance and composition, vegetation structure and above-ground biomass
Fire regime changes	Changes in frequency, intensity, season and timing of fire, including fire suppression	Change in species composition and above-ground biomass, soil erosion, species invasions, change in soil nutrient content, change in runoff and infiltration regimes of water, nutrients and agrochemicals
Introduction of invasive species	Not applicable	Change in species composition, vegetation structure and above-ground biomass, change in fire regime, spread of disease and pests
Extractive industry development	Mine type, extraction and refining techniques, pollutant	Soil pollution and compaction, water contamination, altered runoff regimes,

	discharge and spoil disposal, reclamation, spatial planning	change in groundwater reserves, atmospheric pollution and deposition
Infrastructure and industrial development and urbanization	Land clearance, dams and hydroelectric power plants, roads and railways, other infrastructure development, irrigation	Soil pollution and compaction, water contamination, altered runoff regimes, change in groundwater reserves, atmospheric pollution and deposition

Table 3.2 Indirect drivers of land degradation.

Indirect drivers	**Subcategories of indirect driver**
Demographic	Population growth rate; migration and population mobility (including to urban centers); density; age structure
Economic	Demand and consumption; poverty; commercialization and trade; urbanization; industrialization; labour markets; prices; finance
Science, knowledge and technology	Education; indigenous and local knowledge; taboos; research and development investments; access to technology; innovation; communication and outreach
Institutions and governance	Public policy (regulatory and incentive based); property rights; customary law; certification; international agreements and conventions (trade, environment and so on); competencies of formal institutions; informal institutions (social capital)
Cultural	Worldviews; values; religion; consumer behaviour; diet

3.3 Direct drivers of land degradation

This chapter assesses eight types of direct anthropogenic drivers: (i) management of grazing lands; (ii) management of croplands and agroforestry systems; (iii) management of forests and tree plantations; (iv) non-timber natural resource extraction; (v) alteration of fire regimes; (vi) extractive industry and energy development; (vii) infrastructure and other industrial developments; and (viii) introduction of invasive species (Table 3.1.). Each type of direct driver encompasses a range of specific human-related activities which, in turn, relate to a range of different land degradation and restoration processes (Table 3.1).

The impact of direct anthropogenic drivers is felt through their spatial extent, changes in extent over time, and the particular ways in which the driver is manifest in a given locality (e.g., through differences in management regime). In the following sections, each driver is assessed first with respect to the indirect drivers that are particular to that type of direct driver, followed by an assessment of the past, present and future extent and management systems related to that driver.

3.3.1 Grazing land management

Livestock production is critically important for human well-being across the globe, sustaining household incomes, providing food products such as meat, milk, eggs and animal oils, non-food material such as fibre and leather, and nutrients for crop production in manure. Livestock grazing on rangelands is the single largest human use of the natural environment and supports one-sixth of the global population (Reid *et al.,* 2008). Over 50% of agricultural land and 69% of drylands are used for livestock grazing (Asner *et al.,* 2004, Reid *et al.,* 2008).

We focus here on land degradation and the potential for restoration of grazing lands which includes grasslands and savannah ecosystems, grazed forests and woodlands, in addition to meadows and man-made pastures of temperate and tropical forest regions. Grazing land management encompasses all the strategies used by people to promote both high quality and quantity of forage for domesticated livestock. A broad range of domesticated livestock species and breeds are kept by humans with the most common being cattle, buffalo, sheep, pigs, chicken and goats.

Grazing lands in grassland and savannahs may be categorized into two broad types that differ in their grazing intensity and human population density. Extensive grazing lands, where grazing intensity and human population density is low, are most prevalent across arid and semi-arid regions of the globe. Extensive grazing lands are spatially extensive accounting for 91% of grazing lands globally (Reid *et al.,* 2008). In contrast, intensive grazing lands, which cover approximately 9% of global grazing lands, are those that are managed primarily for livestock production with few other uses of the land other than dispersed crops (Reid *et al.,* 2008). As grazing lands are becoming more fragmented by encroachment of cropland and urban areas – in addition to changes in land tenure policies that promote more sedentary lifestyle for pastoralists – income diversification and reliance on cash crops increase in importance in these areas (Hobbs *et al.,* 2008).

More recently, intensification and increasing industrialization of livestock production systems, especially in developed countries, has resulted in an increasing reliance on mixed crop-livestock production systems to industrialized "landless" systems. For further discussion of livestock production systems supported by cropland products such as grains and legumes see Section 3.3.2.

3.3.1.1 Indirect drivers of changes in the extent and management of grazing lands

Growing markets and marketing systems, linked with increasing human population and dietary changes related to increasing incomes are driving increasing demands for livestock products in both developed and developing countries (Steinfeld, 1998; Nkonya & Mirzabaev, 2016). Projected demand for livestock products is expected to double over the time period from 2000 through 2050. Increasing demand for animal products in turn has increased land demands to support livestock and intensified competition with other land uses, such as crop production for food and demands for non-food products to meet bioenergy needs. These increasing demands for land have driven agricultural extensification over the past half century resulting in land scarcity to support human needs in some regions of the world (Alexander *et al.,* 2015).

Human population growth is a key driver of the increasing demands for animal products globally. Since the early 1990s, human population has increased by 36% from 5.5 billion people in 1993 to 7.5 billion people today (UNPD, 2015). Population growth rates, however, are highly variable across the world, with lesser developed regions of the world exhibiting nearly five-fold higher population growth rates relative to more developed regions (UNPD, 2015). Following this, regions that are experiencing high population growth rates combined with more local and regional based livestock production systems are likely to experience increasing pressure on grazing lands.

Rising household incomes also strongly influence the demand for animal products. Over the period from 2001 through 2011, poverty levels decreased in 85 of 111 countries. In China, the most populous country, the share of middle income households rose from 3% in 2001 to 18% in 2011 (Pew Research Center, 2015). As incomes increase, per capita demand for meat and protein also increase. In the top five wealthiest countries, per capita demand for meat proteins was nearly eight-fold higher relative to the five poorest countries in 2009 (Tilman & Clark, 2014). It is important to note that income and meat consumption are not always related. For example, India has rising incomes but maintains the lowest meat consumption per capita. In this case, strong sociocultural norms drive diet and meat consumption even while incomes are on the rise (World Development Indicators, 2017). Other factors that further contribute to increasing demands for animal products are food waste and overeating with increasing incomes (Alexander *et al.*, 2016) (see also Chapter 2, Section 2.3.2).

Dietary changes toward a more meat-based diet have resulted in per capita increases in animal product consumption. Increases in meat consumption has been especially pronounced in developing countries of Asia with a 3% increase in meat consumption and 5% increase in dairy consumption per year from 2000 through 2010 (FAO, 2013). Growth in the poultry industry has been especially strong with a 3% growth each year over this same time period (FAO, 2013). In contrast, growth in cattle production declined from 2% per year in the 1960s to 1% over the decade from 2000-2010. Global increases in poultry production and declines in cattle production would result in lessening the pressure on grazing lands as poultry production relies primarily on feed from croplands and a more industrialized landless production system (FAO, 2006a).

Land tenure regimes, especially in developing countries, determine many aspects of how grazing lands are managed. Vast areas of grazing lands are still under some form of communal land management and there are many situations of communal grazing where social norms and traditional practices (Ostrom, 1990; Kioko *et al.,* 2012) or legislation, prevent overstocking and maintain the capacity of the land to provide high quality and quantity of forage. In cases where these rules and norms do not exist, or have broken down, the consequence is that less sustainable grazing land management practices may result in land degradation. For example, in Mongolia, a shift from transhumant and well-controlled communal grazing to privatization coupled with sedimentary grazing practices, increases in herd size, and fencing, resulted in significant overgrazing and rapid degradation of economically important grazing lands (Jiang *et al.*, 2006). In the socialist era in Mongolia, *negdel*, a formal institution created and maintained by the socialist government, controlled the movements of herders and provided social infrastructure, dug wells, and provided winter shelters. After the market economy was introduced, the negdel system was not privatized, and it soon collapsed along with the social services that it once provided (Sneath, 2003; Bedunah & Schmidt, 2004). Those social changes, in turn, resulted in declines in mobility and increasing livestock densities due to the rapid increase in the number of new herders, resulting in significant grassland degradation across the region.

Technological advances in livestock production systems have the potential to offset the increasing demand for animal products by increasing livestock production efficiency (Herrero *et al.,* 2013). With industrialization of livestock production systems, shifts to high quality feeds, such as grains and legumes result in higher livestock production efficiencies requiring less land to support an equivalent level of livestock biomass (Herrero *et al.,* 2013). Increasing livestock production efficiencies are also associated with high efficiency breeds, reducing disease and promoting healthier animals by evaluating nutritional needs. In addition, reducing animal waste through more efficient animal processing are part of the technological advances that are likely to reduce future demands for grazing lands. There are, however, negative consequences of intensifying livestock production systems such as declines in animal welfare, increases in land and water pollution, and risk of spread of livestock related infectious disease (see also Chapter 4, Section 4.3.2).

3.3.1.2 Grazing land management: past, present and future extent and management

As the demand for animal products has increased across the globe for the past half century, grazing pressure has also greatly increased. The amount of grazing land and animal numbers both determine the grazing pressure on grazing land ecosystems, thus trends in animal density (number head ha^{-1} of agricultural land) is a reasonably good indicator of grazing pressure. During 2000-2009, Southern Asia had the highest animal densities (which includes the categories of cattle, buffalo, goats, and sheep) and also experienced a 16% increase in ruminant density in the same time period. Although animal density is highly variable across regions, densities were mostly stable or decreasing across 80% of the sub-regions globally (Table 3.3).

For global livestock production to continually increase without changing animal density, there must be an expansion of grazing lands. Indeed, grazing land area increased linearly with human population and GDP between 1960 and the early 2000s (Tilman *et al.*, 2001). Since 2000, however, grazing land area has decreased by just over 1% across the globe (Table 3.3). Decreases in global grazing lands were driven by declines across Europe (2%), Asia (3 %), and Oceania (9%). During the time period from 2000 to 2016, grazing land area increased across the Americas (3%) and Africa (1%) (Figure 3.1).

Recent model predictions suggest that grazing lands will increase globally through 2030 (Alkemade *et al.,* 2013) but then decline from 2030 through 2050. The predicted decline in global grazing lands in 2030 through 2050 will likely be driven by a shift away from grazing in natural grasslands to more intensive grazing systems in which integrated crop-livestock systems and industrialized landless systems will rely more on cropland production and by-products from cereal and soybean production (Alkemade *et al.,* 2013). With further technological advances over the next several decades, reliance on natural grazing lands is expected to decline especially in regions such as Africa in which these advances will have the most impact on grazing practices (Alkemade *et al.,* 2013).

The nature of livestock production systems varies greatly by region and climate (Robinson *et al.,* 2011), with Africa still dominated by small farmer and extensive systems, South East Asia, India and China having mostly intensive systems, and Europe and North America dominated by large industrial systems, including for ruminants. The degree of national development, expressed as per capita GDP, appears to be a good predictor of countries moving from smallholder farmers to concentrated animal feeding operations (CAFO) systems for animal protein production (Gilbert *et al.,* 2015).

In arid and semi-arid regions where over half of grazing lands occur, climate and especially precipitation is an important factor in the vulnerability of grazing lands to degradation or desertification (Steinfeld *et al.,* 2010). Extensive areas of grazing lands in Africa and Asia occur in arid and semi-arid regions, and these regions may be especially vulnerable to degradation given the increasing animal density (especially of sheep and goats) across this region. Across the arid and semi-arid grazing lands of Eastern Africa, there are examples of indigenous practices that promote sustainable use of grazing lands. For example, pastoralism is the central livelihood for the Maasai and Samburu communities of Eastern Africa. The management, governance, and transmission of indigenous knowledge of natural resource management through rituals, ceremonies, folklore, and social networks is strongly focused on the long term management of livestock and grazing lands (Roué *et al.,* 2016)

In temperate and tropical regions, land transformation from forests to pastures has been extensive. Forest removal and conversion to pastures and croplands is key driver of deforestation, especially in tropical regions of the Americas. Livestock grazing in tropical and subtropical forests that were converted to pasture in Central and South America has increased continuously over the past decades (FAO, 2006b) and is now estimated to be greater than that of cropland in tropical Central and South America (Wassenaar *et al.,* 2007). For example, in Brazil 70–80% of deforestation was estimated to have resulted from the development of livestock systems (Tourrand *et al.,* 2004).

Table 3 3 **Global trends in the extent of permanent meadows and pastures (PMP) and livestock grazing density from 2000 through 2009.**

Data from FAO (2013). Grazing pressure from the dominant livestock types are highly variable across regions of the world. Trend in animal numbers over the time period from 2000 through 2009 have mostly been stable or decreasing across 80% of the regions. Southern Asia has seen the largest increases in animal density over this time period.

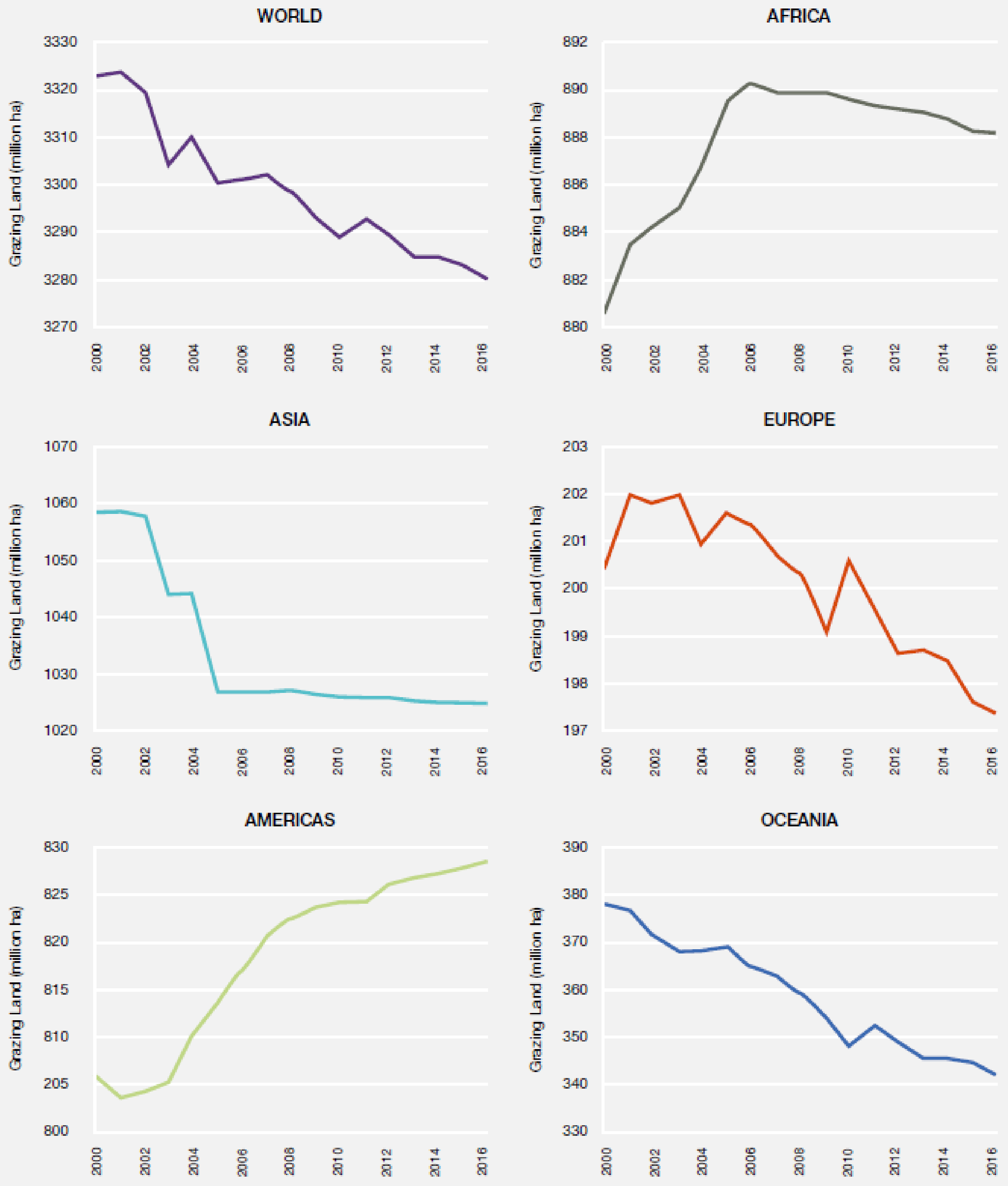
Region	% Area PMP	Cattle & Buffalo head ha^{-1}		Trend	Sheep & Goats head ha^{-1}		Trend
	2009	2000	2009	2000-2009	2000	2009	2000-2009
AFRICA	31	0.2	0.2	0.0	0.4	0.5	0.1
East Africa	39	0.3	0.4	0.1	0.4	0.5	0.1
Middle Africa	21	0.1	0.1	0.0	0.2	0.2	0.0
Northern Africa	26	0.2	0.2	0.0	0.6	0.7	0.1
Southern Africa	57	0.1	0.1	0.0	0.3	0.2	-0.1
Western Africa	32	0.2	0.2	0.0	0.6	0.7	0.1
AMERICAS	21	0.4	0.4	0.0	0.1	0.1	0.0
Latin America and Caribbean	27	0.5	0.6	0.1	0.2	0.2	0.0
North America	14	0.2	0.2	0.0	0.0	0.0	0.0
ASIA	35	0.4	0.4	0.0	0.5	0.6	0.1
Central Asia	64	0.0	0.1	0.1	0.1	0.2	0.1
East Asia	45	0.2	0.2	0.0	0.5	0.5	0.0
Southeast Asia	4	0.5	0.5	0.0	0.3	0.3	0.0
Southern Asia	12	1.2	1.4	0.2	1.2	1.5	0.3
Western Asia	48	0.1	0.1	0.0	0.4	0.4	0.0
EUROPE	8	0.3	0.3	0.0	0.3	0.3	0.0
Eastern Europe	6	0.2	0.1	0.1	0.1	0.1	0.0
Northern Europe	12	0.7	0.6	-0.1	1.3	1.0	-0.3
Southern Europe	20	0.3	0.3	0.0	0.9	0.8	-0.1
Western Europe	48	0.8	0.8	0.0	0.3	0.3	0.0
OCEANIA	44	0.1	0.1	0.0	0.3	0.3	0.0
Australia & NZ	47	0.1	0.1	0.0	0.3	0.3	0.0
Melanesia	1	0.4	0.3	-0.1	0.1	0.1	0.0
Micronesia	8	0.1	0.1	0.0	0.0	0.0	0.0
Polynesia	4	0.3	0.3	0.0	0.2	0.2	0.0

Pressure head ha^{-1}	Trend from 2000 to 2009
0 - 0.3	-0.3
0.4 - 0.7	-0.2
0.8 - 1.1	-0.1
1.2 - 1.5	0
	+0.1
	+0.2
	+0.3

A variety of grazing land management strategies exist to promote restoration of degraded grazing lands. Land degradation can be prevented and reversed through applying appropriate grazing management strategies, with strategies differing greatly in the ease and cost of implementation (see also Chapter 6, Section 6.3.1.3). Following on the example given above, in response to deteriorating grassland conditions in Mongolia during the late 1990s, community-based conservation initiatives were created to implement grassland conservation programs, develop alternative livelihoods for local communities and enhance communication between communities, national park administrators, and local government officials. After nearly a decade, both grassland productivity and median incomes increased (Leisher *et al.,* 2012). Although the absence or breakdown of social norms and traditional practices and the absence of land rights and land tenure policies may increase the risk of land degradation from poor livestock management practices, there is evidence that the creation of community based grazing land practices that target both grassland conservation and livelihoods may provide benefits to nature and humans (Leisher *et al.,* 2012).

Figure 3 1 **Global trends in grazing lands over the time period from 2000-2016.**

World grazing land area has declined globally but there are clear differences in these trends across regions. Grazing lands are defined are grasslands and savanna ecosystems, grazed forests and woodlands, in addition to meadows and man-made pastures of temperate and tropical forest regions as defined by UN FAO (http://faostat3.fao.org). Source: Data from Roser & Ritchie (2017).



3.3.2 Cropland and agroforestry management

Globally, some 1.58 billion hectares of natural ecosystems have been converted to croplands as of 2014 (www.fao.org/faostat). Croplands include both land under permanent crops and arable land which includes temporary crops, temporary meadows for mowing or pasture, gardens, and land temporarily fallow for less than five years (www.fao.org/faostat). Land-use systems where woody perennials (e.g. trees, shrubs, palms, bamboos) are produced in combination with crops, livestock, or both in some form of spatial arrangement or temporal sequence, include agrisilvicultural systems that combine crops and forestry, silvopastoral systems that combine forestry and livestock grazing, and agrisilvipastoral systems that combine crops, forestry, and livestock. Croplands are the second most extensive human activity on the planet, occupying more of Earth's land surface than all other activities except managed grazing (see Section 3.4.1). Fewer than 20 crops are planted on more than three quarters of global croplands (Foley *et al.*, 2011; Mueller *et al.*, 2012).

3.3.2.1 Indirect drivers of changes in the extent and management of croplands and agroforestry

Cropland intensification is driven by diverse combinations of demographic, economic, institutional and technological factors (Turner & Ali, 1996), with both the degree and types of intensification influencing land degradation and restoration (see Section 3.3).

Growing population and changing consumption patterns

Between 1963-2005, global cropland for food increased by some 270 M ha. Over that whole period, 26% of this expansion was attributed to dietary changes and 74% to population increase, but these two drivers are becoming equal over time with rising incomes (Kastner *et al.*, 2012). Rising household incomes are typically associated with an increased consumption of meat, dairy products, sugars and oils, and leisure crops such as coffee, tea and cocoa (Alexander *et al.*, 2015; Alexandratos & Bruinsma, 2012; Le Mouël *et al.*, 2016; Mora *et al.*, 2016). These commodities were responsible for the majority of net expansion in recent decades (Rueda & Lambin 2014). Indeed, half of the additional cropland production since 1963 is associated with the increased demand for animal feed (Alexander *et al.*, 2015; Kastner *et al.*, 2012). Other recent changes include emerging markets for diverse fruits, nuts and other products (Hecht, 2014). This demand may level off once an income threshold is passed, but evidence suggests that such thresholds are high for many products: an annual income threshold of around $36,375 per capita for stabilizing per capita demand of meat, for example, which is only currently achieved by 30 countries and is beyond the reach of most countries for several more decades (Cole & McCoskey, 2013).

Many studies have shown that modifying diets provides ample opportunities to meet societal demands without amplifying existing pressures on natural ecosystems (Bajželj *et al.*, 2014; Cassidy *et al.*, 2013; Herrero *et al.*, 2016; Le Mouël *et al.*, 2016; Mora *et al.*, 2016), but very few studies have empirically investigated the possible effectiveness of policy options to modify dietary choices (Meyfroidt, 2018). Two studies on carbon taxes applied to meat consumption showed a possible sparing of 4 Mha with a tax of 60 € / tCO_2eq (Wirsenius *et al.,* 2011) or a reduction of meat consumption by 5% for a tax of 80 $ / tCO_2eq (Revell, 2015).

Demand for livestock feed

Demand for livestock feed in concentrated animal feed operation (CAFO) production systems is increasing, particularly for pork and poultry. This is being driven by two factors: an increase in global population and a switch to higher animal protein diets with increased standards of living. Production efficiency within the CAFO systems will offset some of the additional land needed for inputs (Steinfeld *et al.*, 2006; Thornton & Herrero, 2010). Changes in socio-cultural values to lower meat diets may reduce the demand for production in this sector in the future (Thornton, 2010). Globalization of trade in fodder means that the impacts from CAFOs may be spatially displaced from the actual animal production, with animal production tending to concentrate close to markets (Steinfeld *et al.*, 2006).

Demand for bioenergy

Increased demand for bioethanol and biodiesel from crop products translated into a small area of land use in absolute terms (i.e., 81 Mha or 5% of global croplands in 2011, but with a rapid expansion of 4.4 Mha y^{-1} over 2001-2011) (Alexander *et al.,*, 2015), and a possibly high social and environmental impact, including due to their often-intensive nature. Biofuels have become a high priority issue in Brazil, the USA, the European Union as well as many other countries with aims of improving energy security and helping to mitigate CO_2 emissions (Birur, Hertel & Tyner 2008) (see also Chapter 2, Section 2.3.1.1). This policy emphasis initially took the shape of a set of supports for biofuel production, then with increasing concerns about the social and environmental impacts of biofuels, policies have been reoriented towards a more restrained role for biofuels. Bioenergy could become a severe driver of land degradation (see Chapter 7, Section 7.2.6). A review of 53 studies on impacts of bioenergy crops on biodiversity showed that these impacts are mostly negative, especially in tropical regions (Immerzeel *et al.,* 2014). Second-generation bioenergy crops tend to have less negative impacts, especially in temperate regions. Land-use change related to bioenergy crop expansion has been shown to result in habitat loss, alterations in species richness and abundance, and biological homogenization. Appropriate land-use planning can contribute to reduce the negative impacts of bioenergy crops (Joly *et al.,* 2015). Yet, mitigating climate change with bioenergy with carbon capture and storage (BECCS) would require more than 1.1 Gha of the most productive agricultural areas or the elimination of more than 50% of natural forests, thus entailing severe social or environmental trade-offs (Boysen *et al.,* 2016).

Technological advances

Between 1961-1990, most of the growth in agricultural output came through increased inputs of labour, capital, and material per unit area of agricultural land (Fuglie, 2012; Fuglie & Rada, 2015). From 1991-2010, however, rising total factor productivity (TFP) (i.e., technological and knowledge progresses) dominated the growth in agricultural outputs which is most evident between 2001-2010 where improved TFP accounted for more than three-quarters of the total growth in global agricultural outputs.

Cultural aspects

Multiple social-ecological systems that support sustainable use of land and biodiversity, preventing land degradation, are embedded within a specific cultural system of beliefs, values and practices (Bélair *et al.,* 2010). These land-use systems are not static but evolve over time, as shown by the emergence of various agroforestry systems through the long-term co-evolution of social-ecological systems that integrate cultural meanings and management practices as a way to reduce vulnerability to shocks, or improve resilience or sustainability of land systems (Hecht, 2014). Changing dynamics of urban-rural interactions may modify households' resources (capital, labour force, information), possibly leading to the spread of institutional or technical innovations or to the development of niche crops that fulfil emerging culturally-driven demands from urban areas such as açai berries (Hecht, 2014) or argan oil (de Waroux & Chiche,

2013) or increasing land scarcity triggering diverse forms of tree-based land use intensification (Lambin & Meyfroidt 2010; Meyfroidt & Lambin, 2011).

3.3.2.2 Croplands and agroforestry: past, present, and future extent and management

The rate of land conversion for croplands has accelerated markedly over the last three centuries (Ellis *et al.*, 2010), increasing linearly between 1960 and 1990 (Tilman *et al.*, 2001) but has subsequently slowed (Keenan *et al.*, 2015). An increasing proportion of these croplands has produced feed for livestock raised in concentrated animal feed operations (CAFOs), rather than producing food for direct human consumption. Worldwide, an estimated 33-39% of all crop production is used for animal feed and meat production (Manceron *et al.*, 2014; Paillard *et al.*, 2010; Steinfeld *et al.*, 2006). This accounts for 4.7 million km^2 of cropland (Steinfeld *et al.*, 2006).

This pattern of greater conversion rates of forest to pastures relative to cropland may be reversed in the coming decades as intensification of livestock production moves toward more landless types of production systems and becomes more reliant on cropland products. Predictions that livestock will be supported more by crops than natural grazing lands will shift land degradation impact of grazing more toward cropland and agricultural systems. Further deforestation will likely occur as croplands and pastures continue to expand to meet the growing demands of increases in both population and per capita food consumption. In Latin America, expansions of croplands and grazing lands into forest lands continued to increase over the time period from 2001-2013. By 2013, however, croplands expansion was outpacing that of grazing lands (Graesser *et al.,* 2015). Such extensification of agriculture is likely to provide diminishing returns of yields, given that croplands already occupy most of the lands best suited for agriculture (Foley *et al.*, 2011). For example, further expansion of croplands in the tropics would help provide food for growing and developing local populations, but for each unit of land cleared, the tropics lose nearly two times as much carbon and produce less than one-half the annual crop yield compared with temperate regions (West *et al.*, 2010). Recommended strategies for increasing food production while decreasing deforestation include closing yield gaps on underperforming lands (Mueller *et al.*, 2012), improving efficiency of agricultural input application, shifting diets (Tilman *et al.*, 2011; Tilman & Clark, 2014), and reducing food waste (Foley *et al.*, 2011).

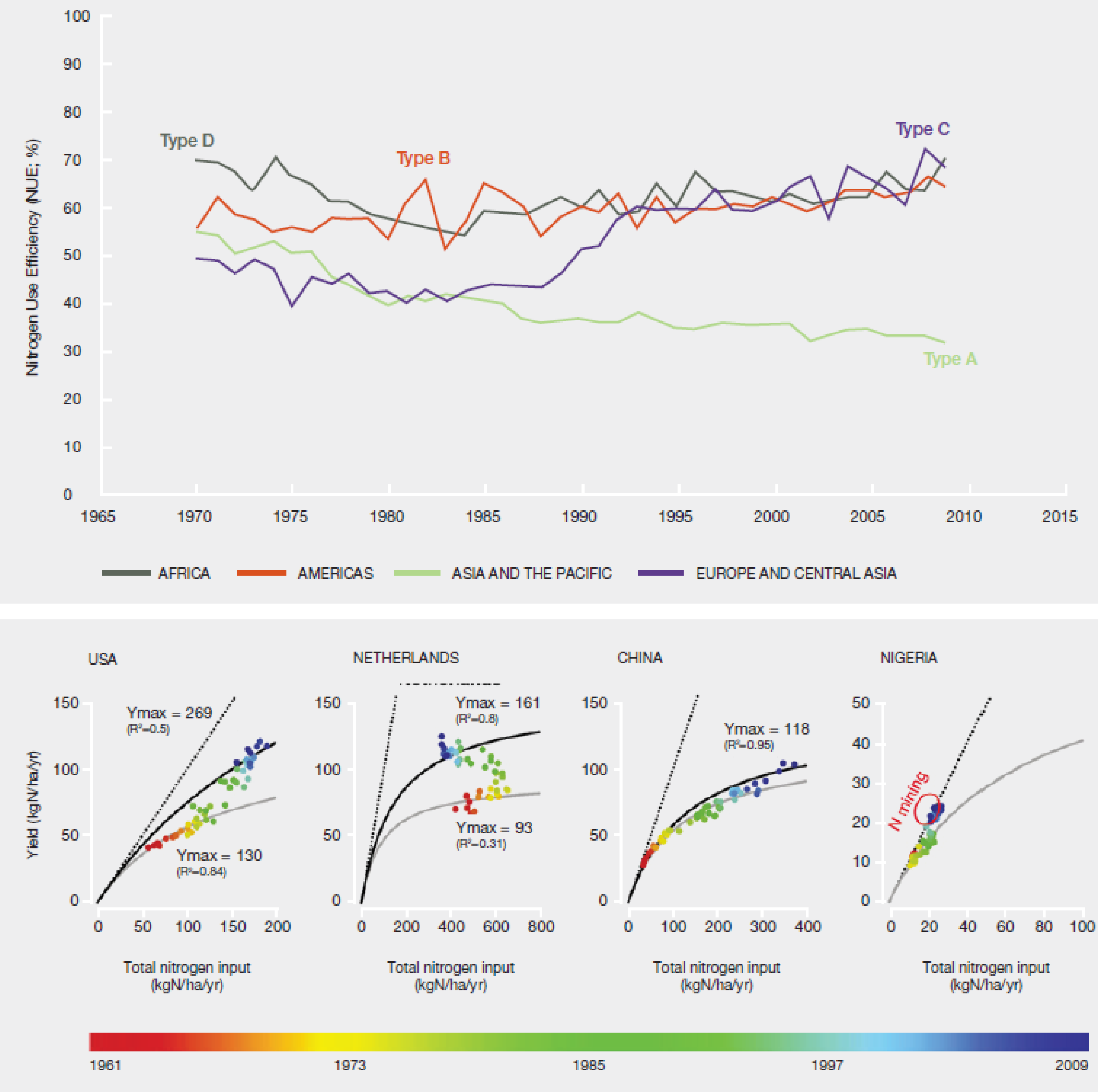
Considerable global intensification of croplands and agroforestry has also occurred over the last half century. Global fertilizer use and pesticide production increased linearly between 1960-2000 (Tilman *et al.*, 2001) including a seven-fold increase in global nitrogen-fertilizer use and a three-fold increase in phosphorus (Tilman *et al.*, 2001). According to the FAO, approximately 108.4 M tonnes of nitrogen fertilizer, 46.2 M tonnes of phosphate fertilizer, and 37.1 M tonnes of potash fertilizer were used in agriculture in 2013. Due to the potentially negative impacts of the widespread use of synthetic fertilizers – especially runoff of synthetic fertilizers into waterways such as lakes, rivers, and streams – there is renewed interest in employing traditional farming methods to reduce synthetic fertilizer use. One example of renewed interest in traditional farming approaches is in the use of rice-fish/rice-duck farming in Asian countries. Rice is critically important food source and is a staple crop for over half of the world's population (FAO, 2013). With a doubling of global rice production and increases in the use of synthetic fertilizers over the past half century, significant water pollution, eutrophication and land degradation accompanied agricultural intensification in rice producing regions (Zheng *et al.,* 2017). Rice-fish and rice-duck traditional rice farming in Asia uses fish, ducks, and other aquatic animals (both domestic and wild) to control pests and pathogens while also providing a source of organic fertilizers to the system resulting in a reduction of synthetic fertilizers and pesticides (Suh, 2014, Pernollet *et al.,* 2015; Zheng *et al.,* 2017). The key weaknesses in this traditional approach to rice farming is the labour intensiveness, scalability to large-scale agriculture, and the increased need for water of this more traditional approach (Suh 2014; Zheng *et al.,* 2017)

While some countries have managed to increase yields while reducing fertilizer inputs during this time, other countries have not seen similar improvements in nutrient-use efficiency (Lassaletta *et al*., 2014; Zhang *et al.*, 2015) (Figure 3.2). For example, many European countries (Figure 3.2, group c), initially increased yields by increasing nitrogen inputs, and later further increased yields while decreasing nitrogen inputs. In contrast, many countries in the Asia and the Pacific regions, have exhibited decreasing nitrogen use efficiency as fertilizer application rates have continued to increase but provided diminishing returns of yield (Figure 3.2, group a). Many African countries have seen little change in nitrogen-use efficiency over the same time period due to relatively small changes in fertilizer use and yields (Figure 3.2, group d). Many countries in the Americas, such as the USA, initially increased yields by substantially increasing fertilizer inputs, and later further increased yields by improving in other production factors such as by using irrigation (Figure 3.2, group b). Irrigation for croplands now consumes more than 70% of freshwater worldwide (FAO, 2016) – about 2,800 km^3 (Lambin & Meyfroidt, 2011) per year from groundwater, lakes and rivers. However, while irrigated crop production comprises only 20% of global cropping systems (Molden *et al.,* 2011), it accounts for 40% of global food production (Abdullah, 2006).

If temporal patterns of change in Nitrogen Use Efficiency (NUE) are analysed by regions of the world it is possible to see four types of trends (Figure 3.2b): (a) decreasing nitrogen use efficiency due to diminishing returns of crop yield from increases in fertilizer inputs – Asia and the Pacific, with a strong effect of China and India; (b) little change in nitrogen use efficiency due to first increasing yield by increasing nitrogen inputs and later further increasing yield by improving other production factors, such as irrigation – Americas; (c) increasing nitrogen use efficiency by first increasing yield by increasing nitrogen inputs and later further increasing yield while reducing nitrogen inputs – Europe and Central Asia, with changes strongly influenced by European countries; and (d) little change in nitrogen use efficiency due to little change in yield or nutrient inputs – Africa.

Figure 3 2 **Changes over the past half century in the efficiency with which crop yield are produced from nutrient inputs by major regions of the world.**

Types A - D represent four contrasting temporal trends for nitrogen use efficiency; see text for further explanation. Examples of trajectories followed by countries in the nitrogen output (yearly harvested yield of 178 primary crops and their nitrogen content) versus input (manure, synthetic fertilizer, symbiotic fixation and atmospheric deposition). Dashed lines represent the 1:1 line. The black and grey lines represent a single parameter hyperbolic curve fitted to the data, considering the first and the last decades of the time period separately. Source: Lassaletta *et al.* (2014).



In the future, further extensification and further intensification of croplands are both likely, given a projected increase in global food demand of 59-98% between 2005 and 2050 as populations and per capita consumption rates increase (Valin *et al.,* 2014). Forecasts based on past trends suggest that one billion ha of natural ecosystems could be converted to agriculture by 2050, accompanied by more than doubling of fertilizer and pesticide use (Tilman *et al.*, 2001). This agricultural extensification will likely provide diminishing returns of yields given that croplands already occupy most of the land best suited for agriculture (Foley *et al.*, 2011).

Extensive restoration of the ecosystems that have been converted to croplands and agroforestry systems is unlikely despite the importance of restoration being recognized globally through the Convention on Biological Diversity (CBD) as these areas are expected to continue to produce much needed commodities. Current objective of the Aichi Biodiversity Target 15 is to restore 15% of degraded ecosystems globally by 2020, while the 2011 Bonn Challenge of the International Union for Conservation of Nature (IUCN) aims to restore 150M ha by 2020 and 350M ha by 2030 (http://www.bonnchallenge.org). Positively, the Global Forest Assessment 2015 indicates that deforestation is slowing and afforestation is increasing with planted forests now progressively providing more goods and services previously derived from natural forests (Sloan & Sayer, 2015). However, considerable global disparity remains with forest gains being primarily located in richer countries and at higher latitudes, while ongoing decline is occurring in tropical regions (Sloan & Sayer, 2015). Debate on the merits of whether land for nature and for production should be segregated (land sparing) or integrated on the same land (land sharing, wildlife-friendly farming) continues but often fails to account for real world complexities (IPBES, 2016; Tscharntke *et al.*, 2012). Conventional intensification often overlooks associated disruptions to natural systems resulting in pollinator loss or increases in pest species impacting on production (IPBES, 2016; Tscharntke *et al.*, 2012).

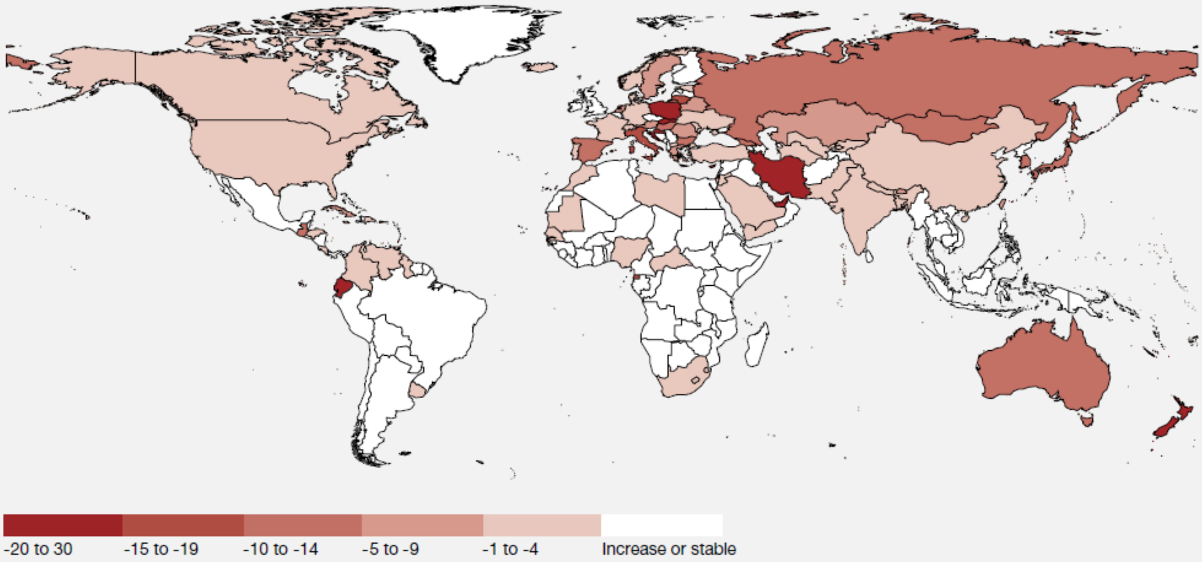
Current global food production is sufficient to feed the world but is inequitably distributed and unaffordable to many people, challenging the suggestions that ongoing agricultural intensification is necessary (Tscharntke *et al.*, 2012). Some opportunities to restore croplands include improving the condition of existing vegetation and species diversity in areas between fields where even small scale habitat restoration can improve biodiversity function (Kremen & M'Gonigle, 2015). Careful planning of restoration programs, especially those that are large, is necessary to avoid displacement of pre-existing land uses that results in the loss or damage to biodiversity elsewhere (e.g., Meyfroidt & Lambin, 2009) or damage to existing biodiversity and associated ecological processes (e.g., through over collection of plant propagules). There is a real imperative to make restoration economically viable, which can be especially effective if restoration activities are coupled with employment and income generation (CBD, 2014) and/or with demonstrable gains in biodiversity and ecosystem services.

The need for restoration is global with many initiatives underway including the Bonn Challenge mentioned previously, the New York Declaration on Forests (restore 350M ha by 2030, see UN, 2014), Initiative 20x20 (20M ha of degraded land in Latin America and the Caribbean into restoration by 2030; http://www.wri.org/our-work/project/initiative-20x20), AFR 100 (100 M ha of land in Africa into restoration by 2030; http://www.afr100.org/) and the UN Strategic Plan for Forests 2017-2030 to increase forest cover by 3% globally (UN, 2017). Most of the restoration action for these programs is primarily undertaken at local scales with success reliant on building partnerships between restoration practitioners, professionals and researchers including decision-makers, indigenous populations with local knowledge, corporate leaders and volunteers (Aronson & Alexander, 2013). Given the spatiotemporal and geo-political complexities of global restoration, four guiding principles have been suggested to generate best practice whereby restoration should: (i) increase ecological integrity; (ii) be sustainable in the long term; (iii) be informed by the past and the future; and (iv) provide benefits and engage society (Suding *et al.,* 2015).

In addition to land that is actively being restored, a considerable amount of land is now recovering after abandonment from human uses (Figure 3.3) (Poore, 2017). Land abandonment is a process "whereby human control over land (e.g., agriculture, forestry) is given up and the land is left to nature" (FAO, 2006; Munroe *et al.*, 2013). Cessation of agriculture is followed by an ecological succession and the recovery process may be gradual or abrupt and readily reversible or difficult-to-reverse (Munroe *et al.*, 2013).

Cropland abandonment has affected an estimated 1.47 million km² worldwide from the 1700s to 1992, and the rate has greatly increased since the 1950s (Ramankutty & Foley, 1999). Most of global areas with marginal agriculture have been affected (Cramer *et al.*, 2008). Some lands are abandoned only temporally (Meyfroidt *et al.*, 2016). Under most future scenarios abandonment is projected to continue, as has been shown by modelling for Europe (Stürck *et al.*, 2015).

Figure 3 3 Percentage change in farmland between 2000 and 2015, highlighting areas recently abandoned, particularly in Europe, Asia and Australia. Source: Recreation of a New Scientist graphic (2017) from Poore (2017).



In Eastern Europe and the former Soviet Union, land abandonment is mainly an outcome of the transition from planned to market-oriented economies (Smaliychuk *et al.*, 2016). In Eastern North America, agricultural abandonment facilitated large-scale forest regeneration in the early twentieth century, but that was partly offset by forest clearing in the Western United States (Ramankutty *et al.*, 2010). In Latin America, migration from marginal rural areas to urban centres has been associated with substantial abandonment (Aide *et al.*, 2013). In Africa, resource conservation efforts have resulted in land abandonment, whereas increased trade in forest products is influential in Asia (Munroe *et al.*, 2013).

Land abandonment typically occurs on remote, less productive land of lower agricultural profitability (Munroe *et al.*, 2013), but not exclusively (Hatna & Bakker, 2011). Globally, the most important drivers are socio-economic (Hobbs & Cramer, 2007); ecological drivers such as poor fertility and land mismanagement leading to soil erosion are secondary (Benayas *et al.*, 2007).

Few studies have quantified the extent, drivers, or outcomes of land abandonment globally (Queiroz *et al.*, 2014; Ramankutty & Foley, 1999; Benayas *et al.*, 2007) partly because of difficulties of measurement (Renwick *et al.*, 2013). A key challenge is to disentangle multiple drivers, pathways and outcomes and, in particular, to separate cyclical, reversible and permanent processes of abandonment (Munroe *et al.*, 2013).

3.3.3 Forests and tree plantation management

Forest management includes a wide range of human activities that shape the structure, species composition and function of both natural (unplanted) forests and tree plantations. The majority of forest landscapes in boreal, temperate and tropical realms have been drastically reduced in both area or structural biodiversity due to human influence (Hansen *et al.,* 2013; Malhi *et al.,* 2014; Moen *et al.,* 2014). Substitution by other land uses and overharvesting of timber for energy, metal industries and construction have been the main causes of human induced forest loss and degradation.

Although all logging activity impacts the structure and biodiversity of a forest (Siitonen *et al.,* 2000; Virkkala & Rajasärkkä, 2007; Nordén *et al.,* 2013), the extent of this impact depends on the logging intensity, including the number of trees removed per hectare, length of the rotation time, and site management practices (Ranius & Kindvall, 2004; Sverdrup-Thygeson *et al.,* 2014). The density of felled trees varies among regions and management regimes from as few as one tree every several hectares (e.g., mahogany, *Swietenia macrophylla* in South America) to more than 15/hectare in lowland dipterocarp forests of Southeast Asia (Fimbel *et al.,* 2001). The most typical harvesting regime in boreal forests of Fennoscandia, Baltic countries, Canada and Russia is clear-cutting. In Northern Europe, clear-cuts are followed either by artificial or natural regeneration including soil treatment and thinning. However, in recent years continuous cover management, based on the selected cutting of mature individual trees or small group of trees has been increasingly adopted as an alternative management regime (Kuuluvainen *et al.,* 2012; Tahvonen *et al.,* 2013).

In response to concerns about the future of forests and timber resources, and negative ecological and social impacts of unsustainable practices, the concept of sustainable forest management (SFM) emerged in the 1990s following the Brundtland Commission on Sustainable Development in 1987, the Earth Summit in Rio de Janeiro 1992, Ministerial Conference on the Protection of Forest in Europe 1993, and approaches for global forest certification standards provided first by the Forest Stewardship Council (FSC) (MacDicken *et al.,* 2015). Whilst the extent of their application remains limited, a wide range of Reduced Impact Logging (RIL) techniques have been developed that involve careful planning and controlled harvesting (e.g., pre-harvest planning and inventories, road planning, extraction techniques and post-harvest assessments) and are capable of greatly minimizing deleterious ecological impacts (Fimbel *et al.*, 2001; Putz *et al.*, 2008).

In more intensively managed forest landscapes, tree stands and planted forests can be subjected to a wide range of active management techniques, from the planting or sowing of genetically improved stock after site preparation, and thinning and fertilizing to maximize yield gains between harvesting rotations. Short-rotation monocultures, of genetically narrow populations, represent the most intensive forest managed regime, such as hybrids of *Eucalyptus spp*. in Brazil (Stape *et al.,* 2010; Couto *et al.,* 2011; Ledford, 2014), or hybrid poplar *Populus trichocarpa x P. deltoides* in Sweden (Christersson, 2006). The impact of intensively managed tree plantations on biological diversity depend mostly on the former land use they substitute. Replacement of high diversity valued native forests with tree plantations results in enormous loss of diversity. By contrast, reforestation of agricultural areas or degraded land by native tree species, especially in areas that retain fragments of native forest, can facilitate biodiversity recovery and enhance provision of ecosystem services (Brockerhoff *et al.,* 2008, 2013; Lamb, 2011). Introduction of alien tree species in tropical forest plantations in Africa has led to invasion of these species into native forests in case of many Australian *Acacia* and American *Prosopis* species (Mathews & Brand, 2004; Lotze *et al.,* 2006), while *Eucalyptus camaldulensis* has become a serious problem in southern Africa (Stanturf *et al.,* 2013). Since the quality of litter produced by introduced *Eucalyptus* is different from local litter, the decomposers are not adapted to it and nutrient release through mineralization is impaired (Pozo *et al.,* 1998).

Current efforts in forest landscape restoration (FLR), such as the Bonn Challenge and UN Strategic Plan for Forests, are not limited to planting trees. FLR is also about using land sustainably in regenerated forests, managed plantations, agroforestry and agricultural systems, as well as protecting wildlife reserves, creating ecological corridors and river or lakeside plantings to protect waterways. To be successful in delivering on its objectives, the global effort of Bonn Challenge – to bring 150 million hectares of the world´s deforested or degraded land into restoration by 2020, and 350 million hectares by 2030 (UN 2014) – must complement and not displace existing land uses on or adjacent to agricultural and pastoral land. Restoring large contiguous tracts of degraded or fragmented forest is a process of regaining the ecological functionality of the originally forested landscape (http://www.bonnchallenge.org/content/forest-landscape-restoration). Therefore, the replacement of grasslands or other naturally non-wooded biomes by planted forests results in widespread loss of biodiversity and other environmental impacts, including impacts on water security, and cannot be considered FLR (Abreu *et al.*, 2017a; Bond, 2016; Filoso *et al.*, 2010; Griffith *et al.*, 2017a; Jackson *et al.,* 2005; Parr *et al.*, 2014; Veldman *et al.*, 2015) (also see Section 3.5).

3.3.3.1 Indirect drivers of changes in the extent and management of forests and tree plantations

Two key societal demands for provisioning services act as fundamental indirect drivers of how forests are exploited: the demand for energy and for forest products as raw materials. In addition, the demand for other ecosystem services from forests (regulating, supporting, cultural) are playing an increasing role in shaping how forests are managed. Consumption of forest products typically increases with economic development, as evidenced by an approximately 50% increase in Europe in the second half of the twentieth century (Nabuurs *et al.,* 2007).

Fuelwood and wood charcoal represent a dominant share of total wood consumption in low income countries, up to 70% in Sub-Saharan Africa and Southern Asia (Bais *et al.,* 2015). By contrast, industrial roundwood products dominate wood removal in higher income regions (Bais *et al.,* 2015). A modern use of forest biomass for energy provision includes black liquor and bark used for energy in paper and pulp industry, as well as wood chips from saw mills and harvest residues exported from forests for combined heat and power production. Fossil fuel substitution targets in regions such as the European Union have contributed towards increased demand for fuelwood, with consumption of woodfuels in industrialized countries increasing by 82% from 1990 to 2010, much of which has come from imports (Bais *et al.,* 2015). The expanding use of forest harvest residues for energy production poses a risk to sustainable forest management. In Europe whole-tree harvesting, including branches, tree tops, and even stumps in addition to traditional stem harvest, has recently become more common in order to produce forest residue based renewable energy as a part of EU’s climate change mitigation policy (Berndes *et al.,* 2016). Such intensive harvesting regimes pose additional risks of site degradation due to nutrient depletion caused by intensified export of nutrient rich biomass (Inge *et al.*, 2011).

Demand for wood products in the Asia-Pacific region is projected to rise 80% by 2030 (FAO, 2010a). Economic growth in the developing world is projected to double global consumption of forest products by 2030 (WWF & IIASA, 2012). Demand for industrial forest products in Asia-Pacific and Africa may exceed forest production by 89 million m^3 by 2020, whereas Latin America may enjoy a modest surplus of 17 M m^3, albeit still accompanied by forest conversion to meet agricultural demand (Carle & Holmgren, 2008; Evans, 2009). Indeed, projections to 2030 suggest that 26% and 15% of current production forests in Latin America and Africa, respectively, are likely to be converted for agriculture (d’Annunzio *et al.,* 2015).

Scenario analyses point to a continuing increase in demand for forest products over the coming decades, with increases being almost entirely concentrated in developing and emerging economies, while demand is expected to stagnate in high-income countries (Meyfroidt & Lambin, 2011; Buongiorno *et al.,* 2012, Nilsson, 2015). Actual rates of change may depend on a series of factors, including the price of alternative energy sources and the dematerialization of information, and commensurate reductions in the demand for paper products.

Beyond the overall level of demand, a series of other indirect drivers influence how forests are exploited. A strong trend is the increasing expansion and importance of planted forests in supplying wood (Warman, 2014), underpinned by multiple drivers. First, cropland expansion, urbanization, environmental conservation and other land uses are increasing land-use competition, which favours more intensive forestry operations (Haberl *et al.*, 2014; Lambin & Meyfroidt, 2011). Second, the exhaustion of readily available timber in many natural forests is evidenced by reports of "peak timber" in tropical countries (Shearman *et al.,* 2012) and elsewhere (Warman, 2014). Third, technological changes, including genetic improvements, also play a strong role in increasing competitiveness of plantations (Payn *et al.,* 2015). Nevertheless, other drivers may hinder the expansion of intensive plantations, including the above-mentioned competition with other land uses (Haberl *et al.,* 2014), and conflicts with other local land users such as smallholders (Gerber, 2011).

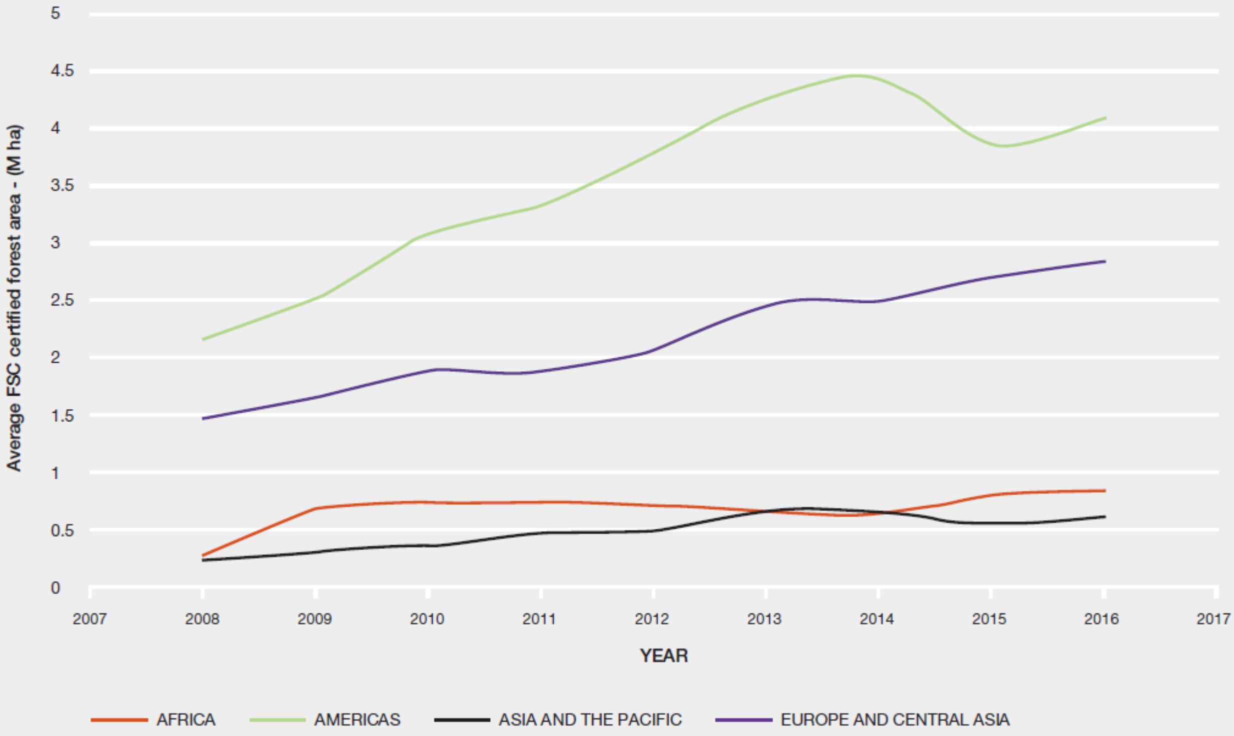
The effects of changes in forest management on the conservation or restoration of natural forests are complex. Natural forest restoration projects can displace other land uses, resulting indirectly in additional forest conversion elsewhere when poorly planned and not combined with improvements in agricultural productivity (Lataweic *et al.,* 2015). On the one hand, intensification of wood production on less land could reduce pressure on unmanaged forests, as seen for example in India (Foster & Rosenzweig, 2003; Heilmayr, 2014). On the other, at a local-level, plantations may also compete for space with natural forests (Zurita *et al.,* 2006; Heilmayr *et al.,* 2016), and the increasing profitability of plantations could incentivize a local rebound-effect reinforcing expansion of plantations over areas of natural forest. By reducing the market value of standing forests, plantations could also increase the vulnerability of natural forests to other uses such as agriculture or reduce the incentives for sustainable management of native forests (Jadin *et al.*, 2017; Pirard *et al.*, 2016; Sloan & Sayer, 2015).

Two main conditions need to hold for plantations to significantly reduce the pressure on natural forests: (i) logging is a major cause of deforestation due to forest degradation, and (ii) wood from plantations constitutes an effective substitute for wood from natural forests (Jadin *et al.*, 2016). Several studies have also shown that under appropriate conditions, the designation of natural forests for production purposes has significantly reduced rates of forest conversion, sometimes as effectively as protected areas (Curran *et al.,* 2004; Gaveau *et al.,* 2013; Bruggeman *et al.,* 2015).

The degree to which forest management can help deliver forest conservation objectives depends on the implementation of sustainable forest management (SFM) practices, including reduced impact logging and others (Putz & Romero, 2015; Putz, Zuidema, & Synnott, 2012; Sasaki & Putz, 2009). The spread of SFM is supported by various governance tools including certification systems (Auld & Gulbrandsen, 2008), programs of Payments of Environmental Service (PES), REDD+ projects and initiative and others (Freer-Smith & Carnus 2008; MacDicken *et al.,* 2015). The adoption of SFM is hindered by the increased costs of production compared to non-compliant operators, and the limited demand from major consumers. Whilst the area of FSC certified forests increased rapidly in Europe and Americas in the 2000s the expansion of certified forests in these regions has slowed in the last decade and has remained very low in Asia, the Pacific and Africa since the start (Figure 3.4).

Figure 3 4 Change in the average area of FSC certified forest area per region of the world between 2008 and 2016.

FSC certified forests expanded relatively rapidly in Europe and the Americas in the 2000s with slower growth in the last decade, whilst the area of certified forest has remained very low in Asia, the Pacific and Africa. Averages are taken across countries for each region, with 9-10 countries in Africa, depending on year of analysis, 19-20 in the Americas, 13-16 in Asia and the Pacific and 32 in Europe. Some countries were excluded as they have over 5 years of data gap between 2008-2016. Those countries are Cambodia, China, Fiji, Ghana, Greece, Guyana, Liechtenstein, Madagascar, Morocco, Pakistan, Philippines, Turkey, Zambia, and Zimbabwe. Source: Data from Forest Stewardship Council, modified by Task Group on Indicators and Knowledge and Data Technical Support Unit.



3.3.3.2 Forests and tree plantations: past, present and future

In 2010, forests covered about 31% of the world's total land area, about four billion hectares (FAO, 2010a). The estimated area of non-managed forest landscapes (forests not managed for timber extraction) by Potapov *et al.* (2008) was 13.1 million km^2 or 24% of the forested zone. The share of managed forests is highest in temperate broadleaf and mixed forests, while Canada, Russia, and Brazil contain 63.8% of the total intact forest landscape area (Potapov *et al.,* 2008). In Europe 4% of forest cover consists of protected natural forests, of which only 2% are strictly protected and can be considered totally undisturbed (Navarro & Pereira, 2012; Forest Europe, UNECE, & FAO, 2014). Most undisturbed forests are found in Northern and Central-East Europe (FAO, 2014). The forest area certified by FSC (Forest Stewardship Council) is over 195 million hectares, while PEFC (Programme for the Endorsement of Forest Certification) certified forest area is 300 million hectares.

According to FAO's most recent Global Forest Assessment (FAO, 2014), total global forest area declined by 3% between 1990 and 2015. When considering trends separately for natural and plantation forests, the area under natural forests declined by 6% (which is twice the total estimate) between 1990 and 2015, from 3961 M ha to 3721 M ha (Keenan *et al.*, 2015). This decline was offset to some extent by a 66% increase in the area of planted forest (including rubber plantations) from 168 M ha in 1990 to 278 M ha in 2015 (Keenan *et al.*, 2015).

Payn *et al.* (2015) used the 2015 dataset of the FAO Forest Resources Assessments (FRA) to estimate current status and trends in the area covered by planted forests globally. In 2015, altogether, 56% of planted forests were located in the temperate zone, 15% in boreal, 20% in tropical, and 9% in subtropical area. The top 20 countries accounted for approximately 85% of planted forest area globally, with the largest areas found in China (91.8 million ha), the USA (26.4 million ha), the Russian Federation (19.8 million ha) and Canada 15.8 million ha. In the tropics, 1.4 % of land is covered with plantations; most of this area is in Asia and Brazil (Boucher *et al.*, 2011). The Forest Resources Assessments data also show that from 2010 to 2015 there was a natural expansion of forest in abandoned agricultural lands of 2.2 M ha per year; whereas the establishment of plantations was at a rate of 3.1. M ha per year (FAO, 2014). However, most countries and regions showed a decrease in the expansion rate (annualized percent change) between 2010-2015 relative to earlier time periods, with the exception of East Africa where percentage change in planted forest area has consistently increased over time to peak at 2.65% in the 2010 - 2015 period.

Sweden and Finland are among the world leading exporters of wood based products: pulp, paper and sawn wood, holding the third and fourth position after Canada and USA (FAO, 2014). Although the total forest area of these countries has remained relatively stable, covering over 75% of total land area, the area of old-growth forests has decreased. In Finland, the area of forest stands older than 160 years has decreased by a quarter during the past 15 years (Kotiaho, 2017).

Combined industrial and fuelwood removals in the tropics increased by 35% (nearly 4 million m^3) over 1990 - 2015 or 1.4% per annum, whilst in other climatic domains there was either no change or a slight decline (Sloan & Sayer, 2015). Industrial and fuelwood removals over 1990 - 2015 increased most rapidly in lower-middle and lower income countries (Köhl *et al.,* 2015), where economic and demographic growth has been greatest.

In 2013, member nations of the International Tropical Timber Organization (ITTO) produced over 236 million m^3 of tropical, non-coniferous logs making a substantial contribution to the economies of these nations (ITTO, 2015). As a consequence, many of the world’s remaining tropical forests have been through at least one cycle of logging, with only 19 of 106 (18%) tropical nations reporting more unlogged primary forest than forest regenerating after logging or some other form of disturbance (FAO, 2010a). Logging intensities - stem removals per ha - have been particularly high across Southeast Asia, where forests are dominated by commercially valuable, high density dipterocarp tree species that enable logging intensities more than ten times higher than in Africa or the Americas. Between 1990 and 2009 some 80% of Malaysian Borneo was affected by high-intensity, multiple cycle logging or clearing operations (Bryan *et al.,* 2013). A recent study reported a nearly 45% loss of the total forest cover in Indonesia from 2000 to 2010 caused by four major industries: logging, wood fibre plantations, oil palm, and coal mining (Abood *et al.,* 2015). Logging and wood fibre plantations had the greatest relative impacts, 1.9 and 1.8 million hectares, respectively.

3.3.4 Non-timber natural resource extraction

For the purpose of this assessment, we consider non-timber resource products to encompass the wide range of natural resources other than timber extracted from forests and other ecosystems for human use, including charcoal, fuelwood, fodder, food and medicinal plants, roots, bark, cork, latex and resins, honey, bushmeat, building materials and fibres (see Beer & McDermott, 1989). Millions of people, mostly in developing countries and often the most impoverished and vulnerable members of communities, rely on non-timber resource products for their subsistence, to supplement diets and income, and to meet their medicinal needs (Arnold & Perez, 2001; Barata *et al.*, 2016; Endamana *et al.*, 2016; Fungo *et al.*, 2016; May-Tobin, 2011; Ticktin, 2004; van Andel *et al.*, 2015). Although extraction of non-timber resource products is often assumed to be ecologically less destructive than timber extraction and other land uses (Arnold & Perez, 2001; Fearnside, 1989; Forget & Jansen, 2007; Schwartzman *et al.*, 2000; Ticktin, 2004), unsustainable levels of extraction of these resources can drive degradation by impacting population sizes and long-term persistence of harvested species, influencing community structure and composition, disrupting plant-plant and plant-animal interactions, and by impacting nutrient and organic matter dynamics in ecosystems (Shankar *et al.,* 1998; Fa *et al.,* 2002; Milner-Gulland *et al.,* 2003; Ticktin, 2004, Ruwanza & Shackleton, 2017), with far reaching consequences for dependent human populations.

3.3.4.1 Indirect drivers of changes in the extent and management of non-timber natural resource extraction

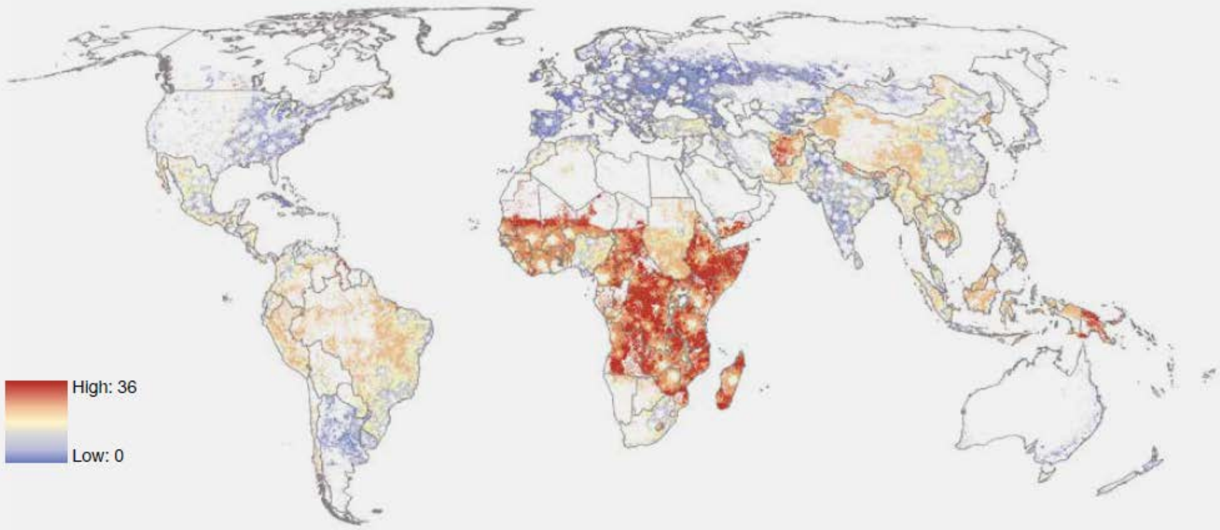
Multiple demographic, economic, cultural and institutional drivers including changes in human population sizes, human needs, cultural values, land tenure, urbanization and trade govern the extent and magnitude of non-timber resource extraction (Schippmann *et al.*, 2006; Shackleton & Pandey, 2014). The extraction of, and reliance on, non-timber resources including fuelwood and medicinal plants is typically highest in developing nations and in rural areas (Barata *et al.*, 2016; FAO, 2010a; May-Tobin, 2011). Although the majority of non-timber resource products harvested tend to be consumed or used by the people that collect them, a substantial amount is nevertheless traded in local, regional and global markets (Belcher *et al.,* 2005; Ingram & Schure 2010; Quiroz *et al.,* 2014; Towns *et al.,* 2014). Thus, both local and external drivers can determine the extent and nature of non-timber resource harvest.

With increasing urbanization, dependence on non-timber resources can decline as livelihoods, lifestyles, patterns of consumption and cultural values change, and alternatives become available. This, however, does not always translate to lowered resource extraction and land degradation in rural areas. For instance, despite widespread urbanization and reductions in the proportions of urban and rural households relying on fuelwood, the nationwide demand for fuelwood in India continued to increase between 1993 and 2005, largely as a result of population growth and increasing total number of households (DeFries & Pandey, 2010). Also, despite lowered reliance on fuelwood, charcoal still remains the predominant source fuel for many people living in cities, the production of which takes place in, and can drive resource extraction from, rural communities (May-Tobin, 2011). Studies have shown that forests around the Tanzanian capital of Dar es Salaam have been exploited for charcoal and timber in concentric waves that have expanded over time (Ahrends *et al.,* 2010) (also see Section 3.6.2.2). Urban centres are also responsible for driving unsustainable levels of exploitation of other non-timber resources including plant products and bush meat across the tropics (Brashares *et al.*, 2011; Ingram & Schure, 2010; Stoian, 2005; Towns *et al.*, 2014; van Vliet *et al.*, 2017).

Cultural traditions, wealth and food prices are also important drivers of non-timber resource use and harvest. For example, in urban centres in Vietnam, bushmeat is most commonly consumed by wealthy, high-status males, and is used as a medium to communicate prestige and obtain social leverage (Drury, 2011, Sandalj *et al.,* 2016). In Africa, bushmeat consumption tends to be greater in wealthier households near urban settlements, whereas the opposite holds true in more isolated settlements (Brashares *et al.,* 2011). In contrast to poor rural communities where bushmeat may often be the only source of protein available, bushmeat consumption is a luxury for the urban consumer, for which individuals are often willing to pay a premium (Bowen-Jones *et al.,* 2003). Cultural traditions can similarly drive the harvest and trade of plants for use in traditional medicine, where wild-gathered specimens are often considered qualitatively superior to cultivated ones, reinforcing harvest from wild populations. For example, wild harvested ginseng is 5-10 times more valuable than artificially propagated ones (Schippmann *et al.,* 2006). Trade and market forces are also important drivers of non-timber resource harvests. It is estimated that between 60 to 85% of the world's population uses or relies on traditional medicine (Figure 3.5), including substantial numbers in developed countries, driving a trade of 500,000 tons of medicinal and aromatic plants each year, much of which continues to be harvested from the wild (WHO, 2005; Barata *et al.,* 2016; Wolff *et al.,* 2017).

Figure 3 5 **Reliance of populations on local wild medicinal plants for their healthcare requirements, scored from 0 (very low reliance) to 36 (very high reliance).**

Africa is clearly distinguished as the continent with the highest level of dependence. Areas in white indicate no reliance because of access to health care or because of the absence of the beneficiary population. Source: Recreation of a New Scientist graphic (2017) from Wolff *et al.* (2017).



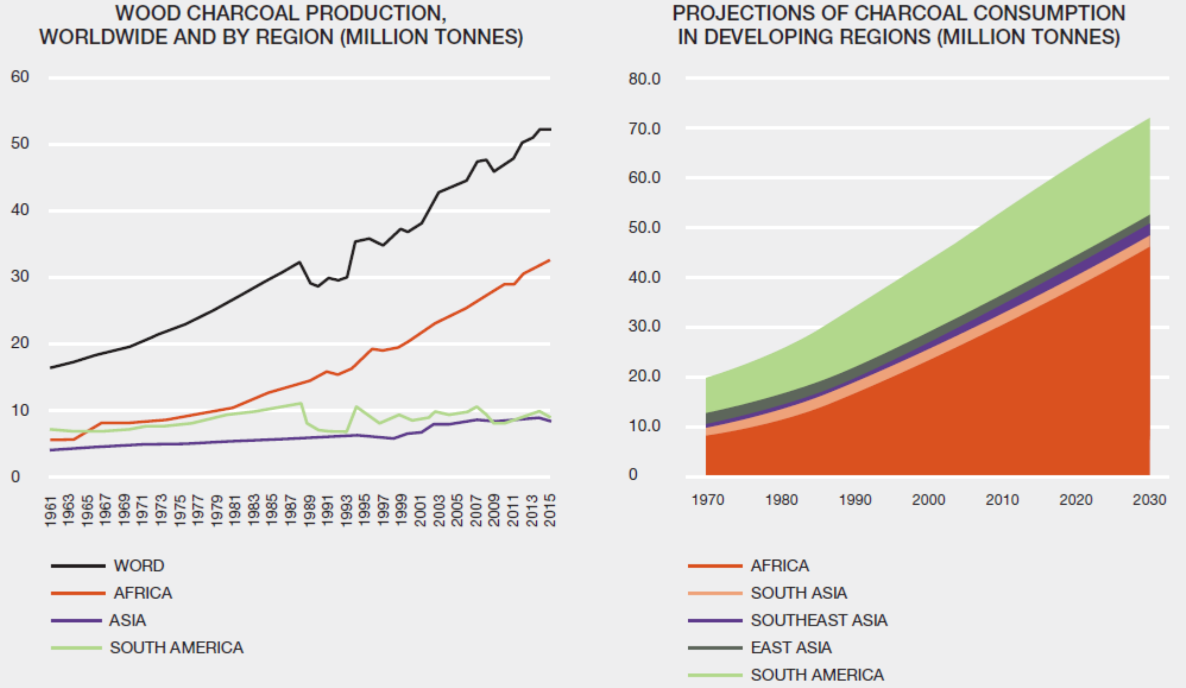
3.3.4.2 Non-timber natural resource extraction: past, present and future extent and management

Approximately a third of the world's population, about 2.4 billion people, depend on biomass such as fuelwood and charcoal for energy (FAO, 2017; IEA, 2006). Developing nations rely more heavily on wood fuels than developed countries, with over 50% of the population in developing countries reportedly using biomass for cooking (FAO, 2010c; IEA, 2006). Some of the highest dependence on biomass for energy is in regions of sub-Saharan Africa, where approximately 90% of rural populations rely on fuelwood and charcoal (IEA, 2006; May-Tobin, 2011). Wood energy provides over half of all energy supply in 29 countries, 22 of them in Africa (FAO, 2014).

Globally, around 3.7 billion cubic meters of wood was extracted from forests in 2015, of which nearly half (approximately 1.86 billion m^3) was used as fuel wood (FAO, 2016b, 2017). The Asia-Pacific region currently accounts for 40% of the fuel wood used globally, followed by Africa (35%) and Latin America and the Caribbean (14%) (FAO, 2014, 2017). Fuel wood production has remained relatively unchanged in the last two decades, although charcoal production continues to rise (Figure 3.6, left panel). Although numbers vary quite a bit between regions, it is estimated that about 17% of all wood used as fuel worldwide is converted to charcoal, over 60% of which is produced in Africa (FAO, 2017). Latin American countries generally use less fuelwood than African and Asian countries, but total and per-capita charcoal consumption is high in Latin America, second only to Africa (FAO, 2017). Brazil is the world's largest producer of charcoal, accounting for approximately 12% of global charcoal production (6.2 million tons in 2015 (FAO ,2017). However, unlike Africa, where the majority of charcoal is for household use, more than 90% of wood-based charcoal in Brazil is used in the industrial sector (FAO, 2017). In the coming years, charcoal use is expected to increase considerably in developing regions (Figure 3.6, right panel) while fuel wood use is expected to remain relatively unchanged (May-Tobin, 2011).

Figure 3 6 **Global and regional (Africa, Asia and South America) trends in wood charcoal production from 1961 to 2015 (left panel), and projected charcoal use until 2030 (right panel).**

Increased global production is dominated almost entirely by increases in Africa where charcoal remains the primary source of energy for many rural and urban households. Source left panel: FAO (2016a, 2017); right panel: May-Tobin (2011).



At local scales, unsustainable levels of fuel wood extraction can lead to widespread degradation of ecosystems. For example, it is estimated that if current levels of fuelwood consumption are maintained in the Lowveld savannahs of South Africa, outside the Kruger National Park, with some two-thirds of households relying exclusively on fuelwood, existing stocks of biomass in communal lands would be exhausted in little over a decade (Wessels *et al.,* 2013).

Besides fuelwood and charcoal, millions of people globally also rely on other non-timber products, both plant and animal derived, collectively worth about $90 billion each year (Pimentel *et al.,* 1997). Nearly 70,000 plant species are estimated to be used for medicinal and other purposes, of which between 4000 and 6000 species are traded commercially, and less than 1% cultivated (Schippmann *et al.,* 2006). The trade in 'bush meat', which comprises an important source of protein for millions of people worldwide, similarly amounts to billions of dollars annually (Brashares *et al.,* 2011). Available estimates of the magnitude of bush meat harvest range from 23,500 tonnes in the Malaysian state of Sarawak (Bennett, 2002; Milner-Gulland *et al.,* 2003), 0.15 million tonnes in Neotropical forests and 4.9 tonnes in Afrotropical forests (Fa *et al.,* 2002). For example, just within the 35,000 km^2 of the Cross-Sanaga rivers region of Nigeria and Cameroon alone, over 1.4 million amphibians, reptiles, birds and mammals, corresponding to over 12,000 tonnes of vertebrate biomass, are estimated to be traded in urban and rural markets annually (Fa *et al.,* 2006). Wildlife population collapses, as a result of unsustainable hunting, have already occurred in many Asian forests, and current extraction levels in several other regions are believed to be unsustainable (Fa *et al.,* 2002, 2003, 2005, 2006) with potential future negative consequences for both biodiversity and food security. Within the countries of the Congo basin, protein supply from bush meat currently ranges from about 30 g $person^{-1}$ day^{-1} (Democratic Republic of Congo) to about 180 g $person^{-1}$ day^{-1}(Gabon) (Fa *et al.,* 2003). If these extraction levels remain unchanged, it is likely to have significant negative consequences not just for wildlife populations but also for humans, with protein supply from bushmeat predicted to decline by over 80% in all the Congo basin countries by 2050 (Fa *et al.,* 2003).

3.3.5 Extractive industry and energy development

Extracting mineral and fossil fuel resources from the Earth's mantle is a significant driver of biodiversity decline and land degradation in countries and ecoregions throughout the world (Bridge, 2004; Townsend *et al.,* 2009; Butt *et al.,* 2013; Durán *et al.,* 2013; Murguía *et al.,* 2016), disturbing as much as 1% of Earth's ice-free surface (Hooke *et al.,* 2012b). In 2014 mining accounted for more than 60% of the GDP of 81 countries. Geographic databases compiled by the United States Geological Survey (USGS) (Matos *et al.*, 2015) show more than 17,000 different large-scale mining sites in 171 countries. Hundreds of different mineral commodities are mined for diverse uses including energy generation, construction, manufacturing and industry, fertilizers, electronics, and medicine. As negative impacts from extractive industries expand and more directly harm ecosystem services and biodiversity, countries have begun to regulate or incentivize restoration of abandoned mine lands as a way to recover some of the ecosystem services lost during the extractive process (Bradshaw, 1997; Cooke & Johnson, 2002; Bridge, 2004). Extractive industries can harmfully impact almost all ecosystem services from provisioning of clean water to biomass production on mine lands, along with causing significant declines in biodiversity. However, restoration efforts often fail to restore ecosystems to their prior state (Cooke & Johnson, 2002; Zipper *et al.,* 2011). Furthermore, many minerals, such as gold, are extracted through distributed artisanal mining operations that can account for 20% of total global production, making it especially difficult to track production and its impacts (Seccatore *et al.,* 2014). As globally traded commodities, the extraction of mineral resources is governed, in part, by global economic, social, and technological trends, which in turn determines the scale, type, and severity of ecosystem impacts.

Extractive industries can be usefully broken down into six distinct categories, with each category broadly linked to industry-wide extraction techniques, and therefore, to different drivers of land degradation and restoration approaches. These industries are: (1) ferrous minerals (iron, nickel, titanium and so on); (2) non-ferrous minerals (copper, gallium, aluminum); (3) liquid and gaseous fuels (oil and gas); (4) mineral fuels (coal and uranium); (5) industrial minerals (salt, concrete, gypsum, phosphate); and (6) precious metals (silver, gold, platinum). These industries are broadly associated with one of three extraction techniques that are highly variable in their disturbance and subsequent impacts on the land surface: (i) underground mining, where a shaft is dug into the earth and minerals are brought to the surface resulting in relatively small spatial surface impact directly from mining; (ii) surface and open pit mining with resource seams directly accessed from the surface with a much larger footprint of surface disturbance (coal, iron, copper, lithium, gravel, phosphates); and (iii) well extraction where a small platform is built to hold a well for extracting liquid and gaseous resources (gas and oil).

3.3.5.1 Indirect drivers of changes in the extent and management of extractive industry and energy development

Changes in extractive industries are tightly coupled with demand for specific commodities in the global economy, yet the impacts from global demand are felt differently in different regions of the world. For example, there has been an increase in global demand and production of rare earth elements used in high tech products like cellular phones, but as of 2015, almost all rare earth production comes from China (Du & Graedel, 2011) (Figure 3.7c). These elements are high cost per weight so their production can be distant from where they are ultimately used. In contrast, construction aggregates like sand and gravel used in concrete, asphalt and building materials are low value per weight so sourcing aggregates is often done more locally with the consequent environmental impacts more globally dispersed (Langer, 2009). This specific example represents a broad trend, where production of many relatively new, rarer mineral resources is highest in developing and emerging economies. While production volumes are accurately tracked for some mineral resources, in some countries, industry growth and impacts are harder to assess due to illegal and artisanal mining, weak institutional capacity to track resource production, and conflict or corruption.

Two major technological innovations have caused major shifts in how minerals are extracted. First, the steep decline in the cost of earth moving equipment (Hooke, 1999; Hooke *et al.*, 2012) and other factors has resulted in a shift from underground mining to surface mining, including open pit and mountaintop mining. Although these changes were beneficial for the mining industry from an economic and safety perspective allowing for deeper and broader surface mining for a broad range of resources (coal, iron, copper and copper), they cause more extensive land degradation (Lutz *et al.,* 2013) and the need for restoration over more landscapes with highly altered substrates (Ross *et al.,* 2016).

Second, a key innovation in the oil and gas sector has been the rapid deployment and development of hydraulic fracturing and horizontal drilling for oil and gas production. Hydraulic fracturing is a process by which oil and gas bound tightly in surrounding bedrock is broken apart with high pressure solvents, releasing tightly bound resources (Hubbert & Willis, 1972), while horizontal drilling allows a single well site to access oil and gas over a much larger footprint (Aucott & Melillo, 2013). These two processes make individual wells more productive than traditional oil and gas drilling and their rapid adaptation has dramatically altered the global oil and gas economies (Figure 3.7a).

3.3.5.2 Extractive industry and energy development: past, present and future extent and management

For all industries, except oil and gas, there has been a dramatic shift towards surface mining techniques in the past sixty years (Hartman & Mutmansky, 2002) due to a complex set of factors including cost, safety, and changing ore quality (Shahriar *et al.,* 2007). Declines in mineable ore quality drives the extensification of mining operations (Mudd, 2010; Prior *et al.,* 2012), degrading more land per unit resource yield (Prior *et al.,* 2012). For oil and gas, excluding tar sand oil extraction, which is more akin to surface mining (Bergerson *et al.,* 2012), all extraction is done through well drilling. These wells have small spatial footprints, but infrastructure (pipelines, roads, treatment plants) to move extracted oil and gas from well to industrial centres may be the primary drivers of land degradation in these cases (Durán *et al.,* 2013; Murguía *et al.,* 2016; Price *et al.,* 2016).

Many extractive industries are located near areas of high or intermediate biodiversity, though the reason for this is unclear (Durán *et al.,* 2013). Consequently, across many industries including coal (Lutz *et al.,* 2013), metals (Durán *et al.,* 2013; Murguía *et al.,* 2016), and oil and gas (Laurance *et al.*, 2009), total resource extraction by country and type can be used as a proxy for estimated ecosystem impact.

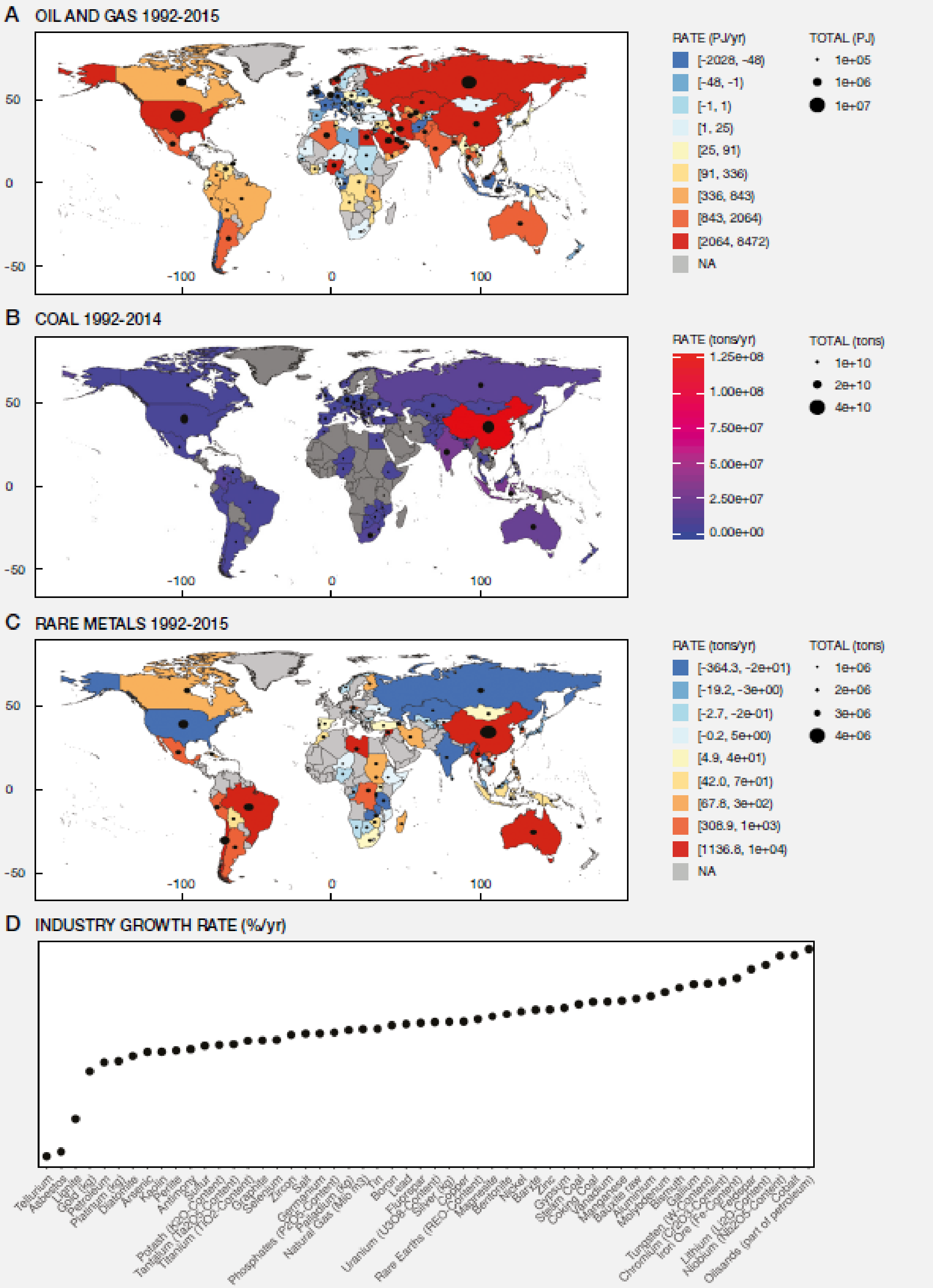
If the recent past of extractive industries can be characterized by the fossil fuel economy with historic growth in oil, coal, gas, steel, and concrete economies, the future of extractive industries may be characterized by the renewable energy economy. Renewable energy technologies and high-tech industries require specific minerals and resources, such as lithium, gallium, cobalt, nobium, and other metals. In the past 20 years, especially over the past decade, rates of production of these minerals have grown at least as fast and often faster than fossil fuel resources, even accounting for the recent glut of oil and gas from hydraulic fracturing (Figure 3.7d). With continued rapid growth in green energy economies there will be continued and sustained demand for these high-tech resources that will likely shape significant portions of the extractive industries in the future.

Across nearly all industries, except for oil and gas, there has been a strong movement towards production centres in advanced developing economies like China, Brazil, and parts of Oceania, especially for rare minerals and coal production (Figure 3.7 b & c). As many of these developing industries are associated with surface mining techniques, countries with these expanding extractive industries will likely bear substantial biodiversity and ecosystem service losses as these industries expand.

In all extractive industries, there has been substantial investment in efforts to restore landscapes following resource extraction. However, restoration – especially of surface mines – has proven difficult to recover or even approach pre-extraction ecosystem services (Zipper *et al.,* 2011; Rooney *et al.,* 2012; Ross *et al.,* 2016), while infrastructure (roads and pipelines) originally used for bringing goods to market often can be used as development corridors, permanently altering ecosystem structure (Barbosa *et al.*, 2010; Laurance, 1999). Historically, many of these impacts were felt primarily in developed countries that dominated the global production market; however, these impacts are broadening in advanced developed economies, making restoration a priority in these places.

Figure 3 7 **Maps show production growth rates for 1992-2014 for oil and gas, coal, and rare metals.**

Growth rates were calculated with a simple linear model of production for each country for each year. For almost all minerals production growth rates are highest in Asia, Africa, and Oceania. A notable exception is oil and gas where production continues to grow in the United States and Russia, partially due to increase production from the adoption of fracking technologies. The bottom panel shows global production growth rates for a diverse array of minerals. Data from World Mining Data 2016. Source: Reichl *et al.* (2016).



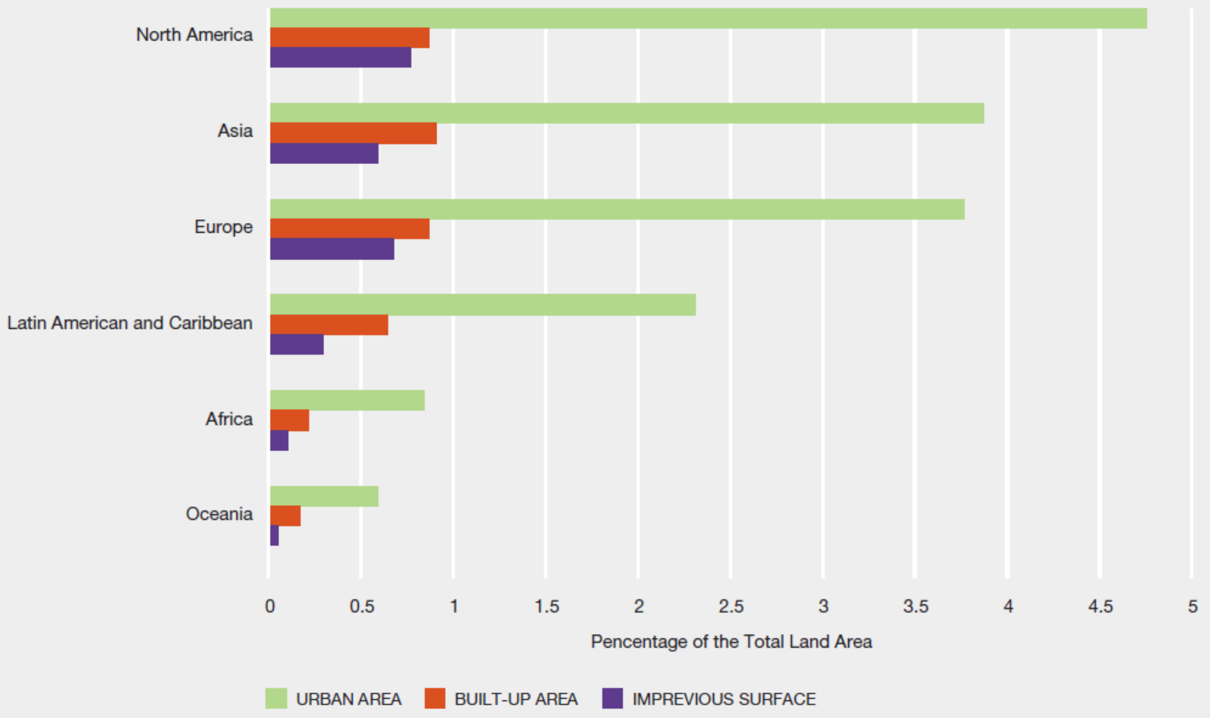
3.3.6 Infrastructure, industrial development and urbanization

As of 2008, more people now live in cities than in rural areas, a dramatic shift from 1900 when only 13% of people lived in urban areas (FAO Yearbook, 2015). Urbanization or the migration of large numbers of people from rural areas to urban centres is closely linked to infrastructure and industrial development. Investments in infrastructure and industrial development usually occur within or near such urbanized areas, and are associated with high activities of production or transformation. Infrastructure is associated with both industry and urban areas, and mainly includes structures necessary for prolonged human presence, such as roads, railways, powerlines, and pipelines.

Reports of the extent of global urban areas are highly variable with estimates ranging by an order of magnitude (less than 1 - 3 %) (Liu *et al.*, 2014; Potere *et al.*, 2009). One important factor in reporting urban land area is defining what constitutes an urban area. To address this issue, a hierarchical framework was developed to identify the spatial extent of urban land areas in addition to the extent in which these areas have been impacted by urban development (Liu *et al.*, 2014) (Figure 3.8). Within this framework, urban land areas are broadly defined by administrative boundaries. Nested within urban land areas are "built-up" areas, which are dominated by artificial surfaces. Large areas within built-up regions are dominated by impervious surfaces due to soil sealing by impermeable materials such as buildings and infrastructure in addition to artificial materials, such as asphalt and concrete, that seal soil surfaces (Figure 3.8) (Liu *et al.*, 2014). The development of impervious surfaces within urban areas is a direct and lasting consequence of most kinds of infrastructure and industrial development. Soil sealing represents one of the most severe forms of land degradation, with a near total loss of soil biological, hydrological, and biogeochemical functions (Burghardt, 2006; Prokop *et al.,* 2011).

Figure 3 8 **Comparison of world regions in terms of their percentage of urban land based on the hierarchical system of definitions.**

Across all regions impervious surfaces that are lacking any form of life make up 8 to 18% of total urban area administrative boundaries. These results suggest that there are significant opportunities for conservation and management of lands within urban areas. Source: Data from Lui *et al.* (2014).



Not only are infrastructure and industrial development important drivers of land, water and soil degradation, but indigenous lands and protected areas especially in tropical regions are becoming islands of biodiversity surrounded by multiple forms and drivers of land-use and land cover change. In the Amazon, for instance, in addition to mechanized agriculture and cattle ranching, infrastructure development (roads, ports, highways, hydroelectric dams) has been identified as a major threat to biodiversity conservation and protection of traditional livelihoods among indigenous and other local social groups (Killeen, 2007; Finer *et al.,* 2013, 2015; Barber *et al.,* 2014; Nobre *et al.,* 2016; Oldekop *et al.,* 2016). Roadways, while opening up avenues for people to sell forest goods, can lead to rising rates of deforestation, unsustainable off-take of high value forest goods and decreased reliance on forest goods by locals (Arnold *et al.*, 2011).

3.3.6.1 Indirect drivers of changes in the extent of infrastructure, industrial development, and urbanization

The main indirect drivers of changes of infrastructure and industrial development is human population growth, migration of people from rural to urban areas, and increases in urban population density.

The connection between human population size and infrastructure and industrial development is clear: the creation or expansion of built-up areas depends largely on demand for living areas, economic areas, and infrastructure to support the increasing population. With the current world population estimated at 7.3 billion people and projected to exceed 9.7 billion in 2050 (UNPD, 2015), demand for these areas will increase markedly in the next 30 years, leading to a concomitant increase in the extent of infrastructure and industrial development. Along with being affected by overall increases in population size, the extent of infrastructure and industrial development will also be affected by the on-going global shift from rural to urban living. In 1950, only 30% of the world's population lived in urban centres, while today that figure is around 54% and will likely rise to at least 66% by 2050 (UNPD, 2015). For example, with the continued rural to urban migration in China, an estimated 70% of the population – or 1 billion people – will reside in urban areas over the next 15 years (Taylor, 2015). The rural-to-urban migration has and will continue to increase demand for living spaces and other built-up areas in urban centres, which contain the vast majority of built-up areas globally. Such demand will be especially strong in less developed countries, where the majority of future rural to urban migrations are projected to take place.

Changes in population densities of large metropolitan areas may also drive changes in infrastructure and industrial development. In a sampling of 120 large cities, population densities declined by 2% per year between 1990 and 2000 and these declines were associated with low-density urban/sub-urban expansion of those areas (i.e., urban sprawl) (Angel *et al.*, 2010). It is not clear whether these declines arise from administrative policies or as a consequence of consumer preferences combined with higher incomes, cheaper travel, and the ability to work remotely (Ewing, 2008).

We present here the global perspective, but it is important to note that the relative importance of these and other indirect drivers differ greatly across regions. For example, Seto *et al.* (2011) found a strong association between urban expansion and gross domestic product (GDP) in China. In contrast to China, urban expansion in Africa was more closely associated with population growth. Therefore, while changes in infrastructure and industrial development will ultimately be driven by the presence and preferences of human populations, the factors affecting preferences and growth may not be universal.

3.3.6.2 Infrastructure, industrial development, and urbanization: past, present and future extent and management

Numerous large-scale datasets have been compiled to estimate the extent of built-up or artificial areas globally. For example, the Night Light Development Index (NLDI) (Elvidge *et al.,* 2012) uses night-time satellite imagery of artificial lights to estimate human presence. The World Bank's index (Angel *et al.*, 2005) on the other hand uses data on population sizes and other metrics from 3943 large cities (population greater than 100,000) to infer the global dynamics of urbanization. Eight global-scale datasets have been developed for the years 2000 and beyond, the majority of which contain medium resolution (less than or equal to 1 km^2) satellite data that have been converted to land cover data using trained statistical models. A recent review of these global maps (some of which differ by more than an order of magnitude in their estimates) found that the MODIS Urban Land Cover 500m (MOD500) (Schneider *et al.*, 2009) dataset had the best agreement with high resolution *Landsat* images of cities (Potere *et al.*, 2009). The MOD500 map is also one of the few maps that has been updated since 2000 and for which future projections have been made. Thus, we focus on the MOD500 dataset and its offshoots in assessing the past, present, and future of extent of infrastructure, industrial development and urban areas.

Using the MOD500 dataset (Angel *et al.*, 2005; Potere *et al.*, 2009), the most complete past record of the global extent of built-up areas is from the year 2000. In 2000, built-up areas were estimated to occupy just over 0.4 % of the total area of the globe (Angel *et al.*, 2011; Potere *et al.*, 2009) and nearly 4% of all arable land. These built-up areas were evenly divided between developed (50.7%) and developing countries (49.3%). Due to the difficulty of assembling global-scale datasets of land cover, a full version of the MOD500 urban/rural dataset beyond the year 2000 has not been developed. However, the dataset has been partially updated to include a random sampling of large cities (population greater than 100,000) circa 2010 (Angel *et al.,* 2005, 2016). Assuming no changes in population densities between 2000 to present, such estimates suggest that the current land occupied by built-up, largely impervious cover is 0.55% of the total area of the globe, an increase of approximately 17% in 10 years. Increases in Europe and Japan have been minimal (+2%), whereas built-up land has expanded considerably in North Africa (+27%), East Asia (+31%), and sub-Saharan Africa (+44%).

The sampling approach described above has also been used to project future changes in artificial land cover (Angel *et al.*, 2005, 2016). Based on predicted population size increases and assuming no changes in population densities, it is estimated that worldwide built-up areas in 2030 will occupy 0.73% of all land (+32% over 2010 levels) (Angel *et al.*, 2011). Less developed regions would account for nearly two-thirds of that area. However, if urban population densities decline, as they have in some older large cities (Angel *et al.*, 2010), built-up areas in 2030 could increase by 140% over 2010 levels. Projecting further forward, by 2050, global built-up areas may account for 0.88% of all land if population densities remain constant or as much as 2.4% if population densities decline. Other projections, using different data and modelling approaches, predict similar magnitude increases in global urban land cover between 2000 and 2030 (van Asselen & Verburg, 2013).

3.3.7 Fire regime change

Fire has been burning and shaping ecosystems for hundreds of millions of years, and is thus an intrinsic feature of most ecosystems (Pausas & Keeley, 2009) – occurring even in wetlands (Watts *et al.,* 2015) and below ground (Page *et al.*, 2002). Prior to human influence, fires occurred across most of the planet but the frequency of occurrence, severity, season, type and extent – collectively referred to as 'fire regimes' – differed between biomes and across regions (Bond & Keeley, 2005; Pausas & Ribeiro, 2013). However, increasing human presence across the world has significantly changed these fire regimes, causing them to deviate from 'natural' conditions.

Humans can alter natural fire activity by changing fuel availability and connectivity. This, in turn, can change fire severity and extent (e.g., through changes in land cover and land use), by altering ignition patterns or actively suppressing fires – with consequences for fire frequency and fire season – and indirectly by affecting climate and atmospheric CO_2 – with implications for all aspects of fire regimes (Archibald *et al.*, 2010; Bowman *et al.*, 2011; Le Page *et al.*, 2010; Midgley & Bond, 2015). In the contemporary world, more fires are ignited in tropical forests, savannahs and agricultural regions by humans than by natural sources, such as lightning (Andela *et al.,* 2017). Such human-driven changes in fire regimes can lead to biodiversity loss and degradation of ecosystem services through multiple pathways including changes in species composition, loss or, alternately, build-up of aboveground biomass (e.g., woody encroachment in grasslands and savannahs; also see Section 3.4), increased prevalence of alien species, soil erosion, and changes in runoff and infiltration regimes (also see Chapter 4, Sections 4.2.6.3 and 4.3.6). In addition, fires can cause substantial direct impacts on human populations due to the destruction of infrastructure, loss of human lives, and risk to human health (Bowman *et al.,* 2011; Doerr & Santín, 2016).

Fire is considered a driver of land degradation only in regions of the planet where anthropogenic changes to fire regimes have been substantial enough to significantly impact biodiversity and ecosystem functions and services. However, a major difficulty in assessing changes in fire regimes are the challenges associated with characterizing 'natural' background fire regimes that have changed across millennia (Bowman *et al.,* 2011).

3.3.7.1 Indirect drivers of changes in fire regimes

The main indirect drivers that can cause, and have caused, fire regimes to deviate from 'natural' conditions include changes in human population densities, human behaviour, socio-cultural factors, and policy and institutional drivers.

Population size is an important determinant of land cover and road network density, as well as the number of anthropogenic fire ignitions, which in turn determine ignition frequency, fuel continuity and patterns of fire spread (Bowman *et al.,* 2011, Bistinas *et al.,* 2013). In fact, human effects on fire regimes are substantial, often overriding climate effects (Archibald *et al.*, 2010). However, the links between population density and fire patterns are not straightforward. While increased human densities can serve to increase ignitions and thus fire frequency, it can also result in smaller burned areas in more populated regions due to more highly fragmented wild-land vegetation, early detection after ignition, early attack, and more effective suppression (Archibald *et al.*, 2009, 2010; Archibald, 2016; Lehmann *et al.*, 2014; Moreira *et al.*, 2010).

Fire regimes can also be influenced by socio-economic and management policies that promote changes in land use and thus fuel type and structure in the landscape. Examples include the impact of agricultural abandonment (e.g., in the Mediterranean regions of Europe) on fuel accumulation and fire risk, the effects of livestock grazing on fuel loads and thus fire intensity, clearance of forests for agriculture and urban development (Cochrane *et al.,* 2008), and the impact of tree species composition in planted forests on fire risk (Archibald *et al.*, 2009; Fernandes *et al.*, 2014; Moreira *et al.*, 2011; Pausas & Keeley, 2014). Alien invasive species have also been shown to cause notorious changes in fire regimes as a result of their greater flammability compared to native species in many instances (D'Antonio & Vitousek, 1992a; Hiremath & Sundaram, 2005; Foxcroft *et al.,* 2010).

Socio-cultural factors, along with traditional knowledge systems and practices, are also important drivers of fire regimes (See also Chapter 1, Section 1.4.3). These include cultural and traditional practices related to crop and grazing management, fuel harvesting, the use of fire to clear land for shifting cultivation, burning by forest-dependent communities to enhance fresh forage availability for livestock and to facilitate the collection of non-timber forest resources, fire ignition for other reasons including arson, and warfare (Hiremath & Sundaram, 2005; Taylor *et al.*, 2016).

Policy and institutional drivers of fire regimes include property rights, land-use planning approaches in wildland-urban interfaces as key areas where fire impacts can be significant, the use of prescribed fire as a management tool (Fernandes *et al.,* 2013), air quality regulations that prohibit cropland burning (Lin *et al.,* 2014), and fire management policies with an emphasis on suppression which can result in fuel accumulation and subsequent mega-fires (Bowman *et al.,* 2011; Durigan & Ratter, 2016), with substantial social and economic costs. For example, annual wildfire suppression costs exceed $1.7 billion on US federal lands and $1 billion in Canada, while total wildfire costs in Australia exceed $9 billion annually (Jolly *et al.,* 2015). Finally, technological drivers, particularly ones dealing with firefighting capabilities, also influence fire spread and severity patterns (Taylor *et al.*, 2016).

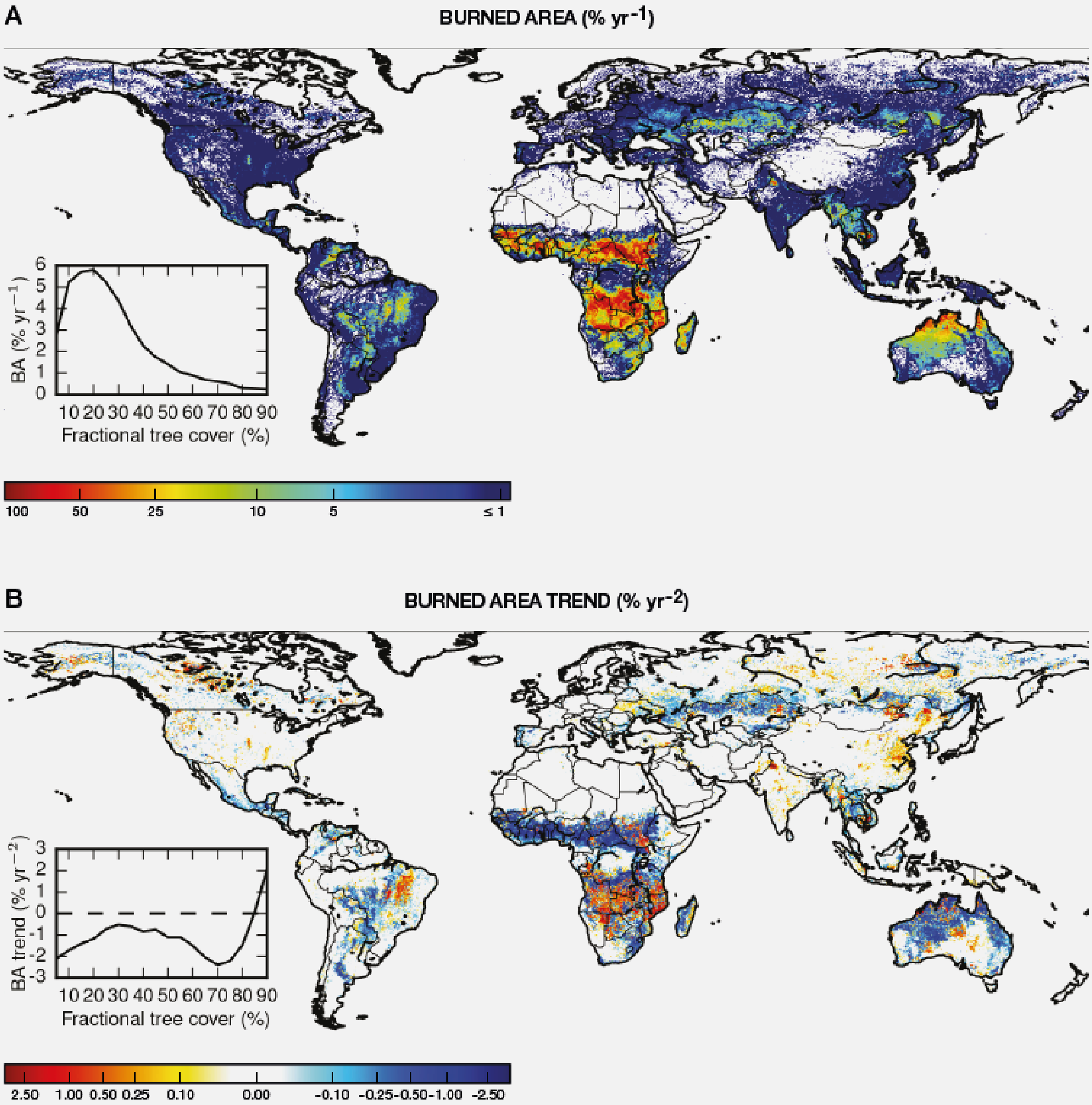
3.3.7.2 Fire regime changes: past, present, and future extent and management

Currently, nearly 350 million hectares of land are burned annually, a huge area that spans most of the terrestrial land surface except deserts, moist tropical forests and tundra (Giglio et al., 2013; Krawchuk *et al.*, 2009).

The time span of available datasets on global area burned annually remains too short to reliably identify long-term trends, and results from different modelling studies and methodological approaches are not always concordant (Arino, Casadio, & Serpe, 2012; Knorr, Arneth, & Jiang, 2016; Moritz *et al.*, 2012; Riaño *et al.*, 2007). However, recent data and model analyses suggest an overall reduction of fire incidence in the last century (Yang *et al.,* 2014; Andela *et al.,* 2017). Between 1998-2015, global burned area declined by nearly 25%, although trends differed between regions (Figure 3.9) (Andela *et al.,* 2017). Decreases were largely concentrated in areas with low to intermediate tree cover particularly the tropical savannahs of South America and Africa and the Asian steppes, while burned area in closed canopy forests showed an increasing trend (Figure 3.9b). Decreases in the numbers of fires contributed more to these declines than decreasing mean fire size (Andela *et al.,* 2017). Further, declines are attributable more to human activities that reduce burning than changes in climate-driven fire risk, which has potentially increased during this period (Jolly *et al.,* 2015; Andela *et al.,* 2017).

Figure 3 9 Map of A mean annual burned area, and B trends in burned area over time (1998-2005) as determined from the analysis of satellite observations.

Many areas of the tropical realm, including much of sub-Saharan Africa, Australia, and south-east Asia, as well as areas of temperate and boreal forests have seen marked increases in burned areas. Insets (line graphs) show the relationships between fractional tree cover and burned area (top panel) and trend in burned area (bottom panel). Source: Andela *et al.* (2017).



At present, it is estimated that approximately 53% of the global terrestrial area has fire regimes that are currently outside their range of natural variability but potentially restorable ('degraded' fire regimes), while 8% have fire regimes far outside their natural range of variability that may not be restorable ('very degraded' fire regimes) (Shlisky *et al.,* 2007).

Changed fire regimes represent a major potential driver of land degradation in tropical and subtropical moist broadleaf forests and boreal forests as these ecosystems are not adapted to fire. Tropical moist forests, in particular, rarely burn naturally because of the low probability of coincidence of ignition with dry climatic conditions necessary to propagate fire (Bowman *et al.,* 2011). However, these forests are productive ecosystems that support high fuel loads, and can become highly fire prone under exceptional drought conditions. Land clearance for agriculture, in particular, can have strong impacts on fire regimes in wet tropical forests, with fires set during droughts as part of the slash and burn process or clearance for broad-scale agriculture resulting in large burned areas. Large fires in Southeast Asia, in 1995 and 2015, which burned thousands of square kilometres of tropical forests (Chisholm *et al.,* 2015) provide reliable evidence of this risk. Although altered fire regimes can also be a major driver of degradation in boreal forests, only a small fraction of the area currently covered by boreal forests is currently believed to be degraded (Shlisky *et al.,* 2007).

Unlike tropical moist broadleaved forests and boreal forests, recurrent fires (albeit with variable fire return intervals) are an integral part of the natural fire regime of Mediterranean forests, woodlands and scrub, as well as temperate broadleaf and conifer forests. However, human-driven changes to fire regimes in these systems can nevertheless drive biodiversity loss and impair ecosystem service provisioning (Schroter, 2005; Moreira & Russo, 2007).

Tropical savannahs and grasslands are amongst the most fire-prone ecosystems in the world (Parr *et al.,* 2014; Archibald, 2016), and the impacts of human alterations of fire regimes in these ecosystems can be substantial. On the one hand, fire at high frequencies can reduce tree densities. On the other hand, fire suppression can promote encroachment by woody vegetation and invasive species with detrimental effects on both livestock and native biodiversity, lead to replacement by forest vegetation, and, when ignited, high intensity fires (Bond & Parr, 2010; Ratnam *et al.,* 2011; Parr *et al.,* 2014; Lehmann & Parr, 2016; Abreu *et al.,* 2017b). In wetter regions, fire suppression can also drive biome-switches from savannah to fire-sensitive closed canopy forests (Staver *et al.,* 2011; Parr *et al.,* 2014). Fire suppression is a major threat to savannahs on most continents, including the Brazilian Cerrado and the savannahs of Asia and northern Australia (Bond & Parr, 2010; Lehmann *et al.,* 2014; Parr *et al.,* 2014; Durigan & Ratter 2016; Ratnam *et al.,* 2016; Abreu *et al.,* 2017b).

Although global burned area extent has been declining, and may continue to decline in the coming decades as human activities reduce fire prevalence in several regions, climate change is nevertheless likely to lead to climatic conditions that increase fire risk in the future (Jolly *et al.,* 2015), but with important regional and biome-specific variations (also see Section 3.4.5). In low productivity regions increased droughts can reduce fuel loads and fire frequency, whereas in tropical forests increased prevalence of droughts under global warming (Pausas & Ribeiro, 2013) is likely to result in fires becoming frequent. Fire incidence in both tropical and boreal forests is strongly affected by weather conditions and anomalies such as El Niño (Le Page *et al.*, 2007), and several studies predict future increases in fire frequency due to climate change in these forests (Krawchuk & Moritz, 2011; Lavorel *et al.*, 2006; Moritz *et al.*, 2012), in some cases also extending this effect to tundra regions (Moritz *et al.*, 2012). In addition to biodiversity loss, the impacts of these forest fires on CO_2 release, air quality and human health are likely to be profound (Harrison *et al.,* 2009). Strong increases in fire risk in response to climate change is also predicted for temperate broadleaf and conifer forests, particularly in North America and central Europe, (Krawchuk *et al.*, 2009; Mouillot & Field, 2005). Similar trends are expected for Asian montane grasslands and shrublands (Krawchuk *et al.*, 2009). For tropical and subtropical grasslands and savannahs, which have witnessed some of the greatest increases in fire weather length in the last few decades, woody encroachment driven by changes in climate and grazing regimes, as well as land-use transformations, are also likely to have consequences for fire risk (Jolly *et al.*, 2015; Midgley & Bond, 2015; Stevens *et al.*, 2016).

Ultimately, reducing or mitigating the effects of fires as a degradation driver would imply taking action to change fire regimes towards natural pre-industrial times. However, this is not feasible or even desirable in several contexts, as other motivations for fire management may take priority (Freeman *et al.,* 2017). In fact, fire management has a variety of objectives, including protection of human communities and assets from fire, forest protection from fire, mitigation of carbon emissions, maintenance of grazing lands, restoration of disturbance-dependent habitats or species, and restoration of ecological processes (Bradstock *et al.,* 2012; Fernandes *et al.,* 2013), that result in different approaches to halt, reduce and mitigate undesirable impacts in terms of land degradation. The range of possibilities will always depend on the setting of management objectives for different geographic regions and contexts (Moreira *et al.,* 2012). For example, prescribed fire can be a powerful tool to restore fire-dependent habitats and species (Fernandes *et al.,* 2013). On the other hand, fire exclusion (tropical moist forests) and reducing fire frequency (boreal forests) may be a crucial aspect for the maintenance of other key ecosystems.

3.3.8 Invasive species

Invasive species are non-native (or alien) organisms (plants, animals, micro-organisms) that can establish and proliferate in ecosystems other than their native ones, often to the detriment of the invaded ecosystem (Mack *et al.*, 2000). Traits of invaders are generally some combination of high dispersal capacity, wide environmental tolerances, rapid reproduction, and fast growth (Mack *et al.,* 2000). Invasive species are not constrained to any particular region; rather, they represent a global threat to biodiversity and ecosystem services and they are often ranked as one of the main direct drivers of species extinction (Dukes & Mooney, 1999). The threat of invasive species has grown so large that only a few areas of the planet remain free of them (Figure 3.10) (Mack *et al.*, 2000).

3.3.8.1 Indirect drivers of invasion

The most important indirect driver of invasion is the global movement of goods and humans via international trade and transportation (Figure 3.10 A-I) (e.g., Early *et al.*, 2016; Pysek *et al.*, 2010; Seebens *et al.*, 2015). Accidental introductions occur mainly through the escape of pets, insects, pathogens, and plants that are being transported in luggage, goods, and/or packing material. Marine shipping ports are the main point of invasion, followed by airports (Fig 3.10 E-I) (Levine & D'Antonio, 2003; McCullough *et al.*, 2006). In addition, some of the world´s most damaging invasive species were originally introduced by humans as part of development programs. One such widely cited example is *Prosopis*, the mesquite tree, which is now naturalized or invasive in many arid and semi-arid areas of the world (Shackleton *et al.*, 2014).

Whereas ports provide entry for propagules, the points where they arrive have to then be suitable for establishment. Therefore, another important indirect driver of invasion is the vulnerability of a given site to establishment of the invader (Belnap *et al.,* 2016). For instance, Capinha *et al.* (2015) showed that while humans have dispersed non-native gastropods widely, thus breaking down global-scale biogeographic boundaries, the post-dispersal gastropod communities are strongly limited by climate, distance from the introduction point, and levels of trade (and thus opportunities for introduction), thereby reducing their overall impact at larger scales. However, site suitability may change through time, due to changes in direct drivers such as soil surface disturbance, altered fire regimes, or climate change (discussed below). In addition, the invasion of alien plants can alter soil microbiota, biogeochemical cycles and fire characteristics, indirectly facilitating further invasion (e.g., Belnap & Phillips, 2001). Invasive animals can alter animal and plant community structure and function, sometimes furthering the invasion of exotic plants and animals (e.g., Claxton *et al.*, 1998).

At evolutionary timescales, physical features such as oceans, mountains, and ice have functioned as effective barriers to the movement of organisms that may otherwise be able to survive and reproduce in new localities (Lowe *et al.,* 2000). Working over different timescales, a combination of changes in climatic and geological events, together with increased global trade, inter-connectedness of human populations, and economic activities, has served to break down past natural barriers, facilitating the spread of species beyond their original habitats to other parts of the world (McKinney & Lockwood, 1999; Lowe *et al.,* 2000).

3.3.8.2 Direct drivers of invasion

The most important direct drivers that allow exotic plants to become established and spread are climate change, abandonment of fertilized agricultural land, urbanization, and activities that disturb the soil surface or alter plant or animal community composition, such as herbivory (by livestock or wildlife), fire, energy and mineral development, recreation, hunting, and expanding human settlement (e.g., Dukes & Mooney, 1999; Barbosa *et al.*, 2010; Liu *et al.*, 2013).

The distribution of many plants and animals, including exotics, are limited by climatic regimes and a shift in a given climate envelope can easily lead to the extirpation or increase of a given species at a site (e.g., Allen & Bradley, 2016). Climate change simulations by Bellard *et al.* (2013) suggest that invasive aquatic and terrestrial invertebrate distributions are projected to substantially increase globally. Fertilized cropped lands, often barren between crops or when abandoned, can support invasives due to high nutrient levels favouring exotic over native plants (Brooks, 2003). Urbanization can be a major direct and indirect driver of invasion, as it is associated with both greater soil disturbance and introduction of propagules (Aronson *et al.,* 2016). Soil surface disturbance associated with all human-associated activities (e.g., energy and mineral extraction; recreation; agriculture, including grazing animals and cropping; and expanding human settlement, including associated infrastructure development such as dams, pipelines, and powerlines), can also increase invasion by altering native microbial, plant, animal community composition and biogeochemical cycles, including soil nutrient availability (e.g., Belnap *et al.*, 2001; Germino *et al.*, 2016).

Over-utilization by livestock commonly increases plant invasion in all types of ecosystems, from tropical rainforests to deserts, as native perennials are consumed and unable to recover from high grazing intensity (Pivello *et al.,* 1999; Martins *et al.,* 2004; Kato-Noguchi *et al.,* 2014). Declines in native perennial cover with overgrazing may then provide an opportunity for invasive plant establishment, especially of invasive annual grasses (D'Antonio & Vitousek, 1992b). Invasion of annual grasses is a serious issue across grazing lands globally leading to declines in both the quantity and quality of livestock forage. Across the Great Basin rangelands of North America, colonization of the invasive annual grass, *Bromus tectorum* (cheatgrass) has led to steep declines in native perennial grassland and shrublands (Knapp, 1996). Cheatgrass invasion also alters fire regimes in these ecosystems with the fine fuels promoting more frequent fires. Unlike the native perennial grasses and shrubs, cheatgrass is highly adapted to frequent fire, which in turn further increases cheatgrass cover. Large expanses of these economically important grazing lands in North America have now been transformed to annual grass monocultures.

Fire can increase the invasion of a site by reducing or eliminating native plants and increasing soil nutrients, as occurs in cheatgrass invasion in the western USA (Germino *et al.,* 2016b). In turn, the presence of exotic plants such as cheatgrass can increase the frequency, size, and intensity of wildfires by increasing the amount or continuity of fuels, thus directly accelerating the loss of non-fire adapted native plants, causing local plant extinctions, and/or increasing soil erosion that can then have further negative effects on native species (e.g., Busch & Smith, 1995; Brooks, 1999). If the invasive plants are annuals, they generally fail to germinate in the frequent dry years, leading to accelerated soil erosion which compromises soil health (Neff *et al.,* 2005), human health (Nguyen *et al.,* 2013), and water supplies if subsequent dust is deposited on snowpack, accelerating snowmelt and increasing water loss via evapotranspiration (Painter *et al.,* 2010). Changes in plant cover and composition also affect land surface albedo, which has implications for global carbon cycles (Poulter *et al.*, 2013) and the land-atmosphere exchange of CO_2, and other greenhouse gases (e.g., CH_4 and N_2O) (Brovkin *et al.*, 2013).

Many tree species used in commercial and agroforestry plantations tend to be non-indigenous (Richardson, 1998) and in many cases these species have become invasive. For example, this has been the case for many Australian *Acacia* species and American *Prosopis* species in Africa (Mathews & Brand, 2004). *Pinus* species, which along with *Eucalyptus* spp. and *Acacia* spp. is the most widely used non-indigenous genera for plantations (Richardson & Rejmanek, 2011; Richardson, 1998; Zobel *et al.*, 1987), is amongst the most problematic with at least 19 species considered invasive over large areas of the southern hemisphere (Richardson, 1998; Van Wilgen, 2015). While Eucalyptus species are not considered to be highly invasive in South America, *Eucalyptus camaldulensis* has become a serious invader in southern Africa (Stanturf *et al.,* 2013). Based on an extensive survey of 622 invasive alien woody plants globally, Richardson and Rejmánek (2011) estimated that after horticulture (62% of species), the most important reasons for invasive woody species introductions globally were forestry (13%), food (10%) and agroforestry (7%).

3.3.8.3 Trends in invasive species: past, current, and likely future extent

The current extent and causes of species invasion are best understood for vascular plants. It is currently estimated that over 13,000 plant species, or almost 4% of the global flora, has become naturalized in ecosystems other than their native ones, a majority of which have become invasive (van Kleunen *et al.,* 2015). As there are almost no data for almost 20% of the Earth (mostly in temperate Asia), the actual number is likely much larger. North America contains by far the largest number of non-native plants when all sources are considered, followed by Europe and Australia (van Kleunen *et al.,* 2015) and in a recent global-scale analysis, Early *et al.* (2016) found about 17% of the Earth's land area is currently at high risk of plant and animal invasion (Figure 3.10 - A), as well as approximately 16% of global biodiversity hotspots.

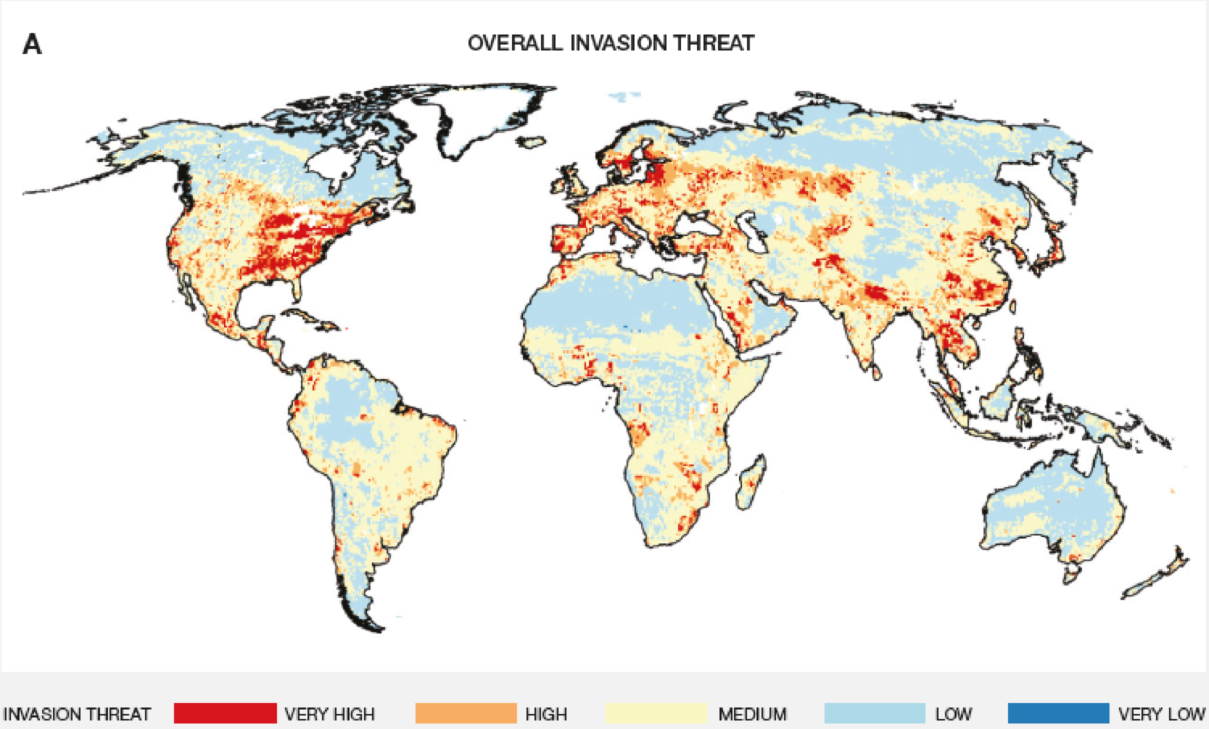
The extraordinarily large number of non-native plant species reported for North America may be due to more intensive sampling; however, studies have found a strong correlation with an index of the amount and historical duration of international trade and the number of invasive plant species, with the USA leading this index (Figure 3.10) (Early *et al.*, 2016; Seebens *et al.*, 2015). The largest number of non-natives from extra-continental sources is found in Australia (van Kleunen *et al.,* 2015). Whereas the long isolation of Australasian species may have resulted in many distinct species, these species may have been unable to take advantage of niches created by European settlement activities, thus leaving opportunities for non-native plants to become established, or may have been unable to resist the invaders. In contrast, despite a high rate of plant introduction to Europe, few plants have become established (van Kleunen *et al.,* 2015). Tropical regions show very low invasion rates, which may reflect fewer available niches or lower introduction rates (Rejmánek & Richardson, 1996). It has long been believed that the main trajectory of plant introductions was from the Old World to the New World. However, van Kleunen *et al.* (2015) show that continents in the Northern Hemisphere (especially temperate Asia, Europe, and North America) are most often a source of non-native plants for Southern Hemisphere continents.

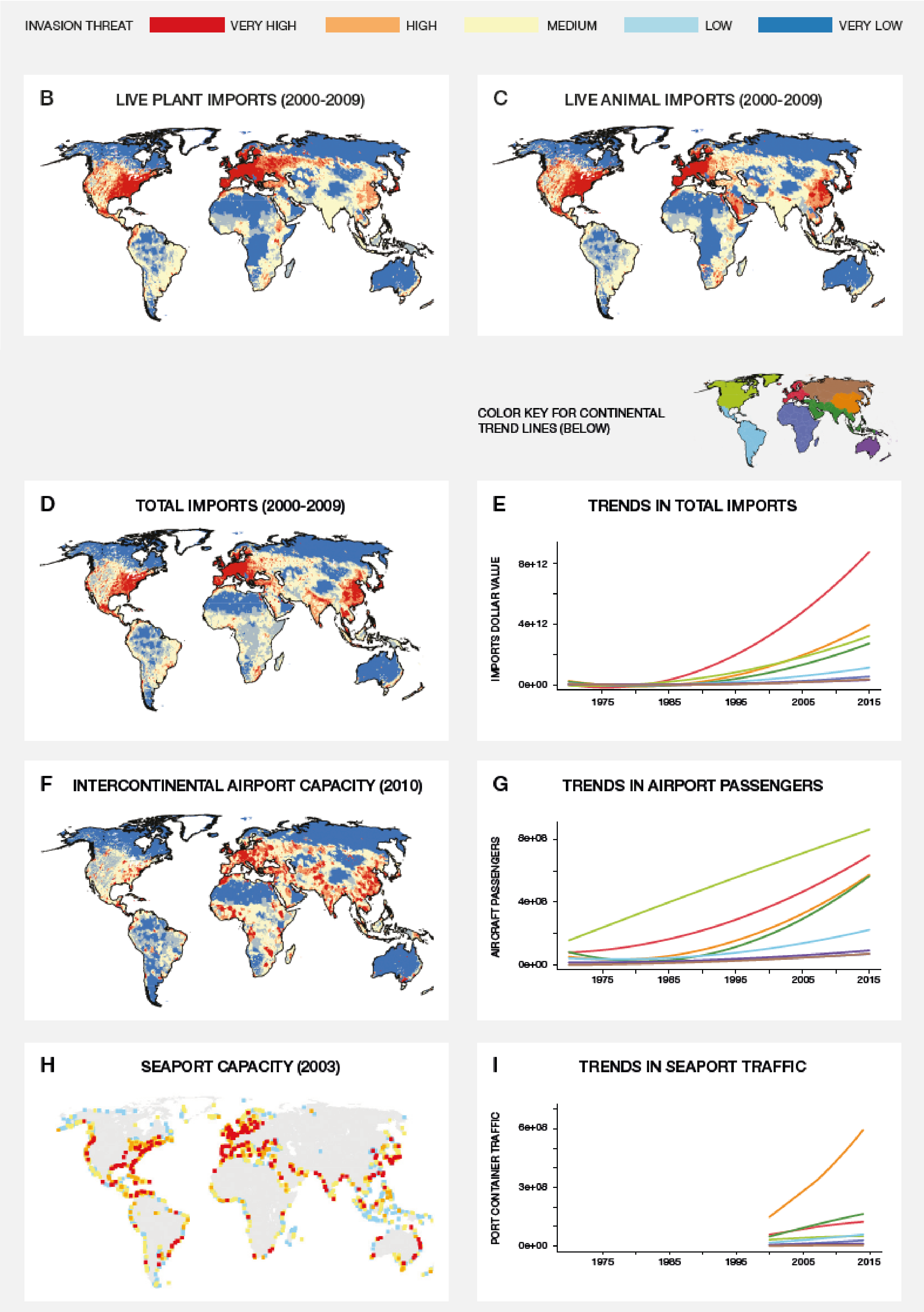
The analyses of Early *et al.* (2016) and Seebens *et al.* (2015) both indicate that invasions in high-income countries will continue to accelerate with the expected increases in trade (Figure 3.10 - G, H, I). These studies indicate invasion threat is especially high for temperate lands in the Northern Hemisphere (Figure

3.10 - A). For low-income countries, future invasion risk is expected to increase as tourism and outward migration increase air traffic, shifting the main introduction points from seaports to airports, and to increase with the growth of economic activities. Climate change will also affect invasion patterns, as some habitats will become less suitable for a given species while at the same time, may become more suitable for others (Blumenthal, 2005). Also, increased atmospheric nitrogen deposition and CO_2 will likely increase invasive plants in many regions (e.g., Belnap *et al.*, 2016). Increased rainfall variability, predicted to increase in many dryland regions, will likely favour invasive annuals over native perennial plants (Grime, 1979). Fire-return intervals are becoming shorter with increasing temperatures (e.g., Pausas & Keeley, 2014) and especially in non-fire adapted vegetation, often leads to invasion (Brooks, 1999), creating a positive feedback that results in more invasion as discussed above. Increases in human population will lead to more soil surface disturbance, as need for food, energy, mineral, and infrastructure increases, and an increase in global trade, and these factors are expected to increase the numbers of invasive plants and animals. Therefore, with the overall growth of human population and activity, it will be necessary to control the introduction and establishment of invasive species, as their eradication is generally much more difficult once they are established.

Figure 3 10 Overall invasion threat for the twenty-first century and patterns of establishment factors.

Trade is a major driver of invasion threat in higher income countries, with North America hosting by far the largest number of non-native plant species, followed by Europe and Australia. Map (A-D, F, H) colours indicate threat level: Very High (VH; red; 100-90%), High (H, 90-80%), Medium (M, 80-50%), Low L, 50-20%), Very Low (L, blue, 20-0%). Trend lines (E, G, I): colours correspond to the introduction trend colour key. Regions follow sub-continental boundaries determined by UNCTAD (United Nations Conference on Trade and Development). Sub-continents that had very similar values for introduction factors were grouped. Global introduction pressure (3A) is a composite analysis of projected biome shifts, increased agricultural intensity and fire frequency between 2000-2100; airport and seaport capacity and animal; plant, and total imports to illustrate likely invasion threat. Values were calculated using the highest value of the constituent factors within each grid cell. Live plant, animal, and plant plus animal imports (B, C, D) are shown in terms of mean annual US dollar value of live plants and animals and both combined not for human consumption imported by each country from 2000-2009. Data were extracted from the United Nations Commodity Trade Statistics database (Comtrade, 2010). Within each country it was assumed that alien species introductions would be greatest at the locations where human population density was highest, and so import value was distributed according to human population density within each country. Data before 1995 had too many missing values to be reliable. Change in total imports (E) shown as mean annual US dollar value of all goods imported by each country since 1970. Colours correspond to the introduction trend colour key. Intercontinental airport capacity (F) shown as estimated number of passengers arriving from intercontinental journeys at all airports located in cities with populations greater than 100,000 in 2010, where the airport is their final flight destination. Passengers are assumed to transport alien species to the population areas surrounding airports (Huang *et al.*, 2012). Change in airport capacity (G) shown as the total (international and domestic) air carrier traffic in a country in each year. Seaport capacity (H) shown as cargo traffic (port volume data, in metric tons) into each port listed in the World Port Index compiled in 2003 (Halpern *et al.*, 2008). Points where ports are located are enlarged for visibility. Change in seaport capacity (I) shown as container traffic into each port measured by Twenty foot Equivalent Units(World Bank, 2016). Data only available from 2000. All figures based on Early *et al.* (2016).




INVASION THREAT
VERY HIGH
HIGH
MEDIUM
LOW
VERY LOW
B
LIVE PLANT IMPORTS (2000-2009)
C
LIVE ANIMAL IMPORTS (2000-2009)
COLOR KEY FOR CONTINENTAL TREND LINES (BELOW)
D
TOTAL IMPORTS (2000-2009)
E
TRENDS IN TOTAL IMPORTS
IMPORTS DOLLAR VALUE
8e+12
4e+12
0e+00
1975
1985
1995
2005
2015
F
INTERCONTINENTAL AIRPORT CAPACITY (2010)
G
TRENDS IN AIRPORT PASSENGERS
AIRCRAFT PASSENGERS
8e+08
4e+08
0e+00
1975
1985
1995
2005
2015
H
SEAPORT CAPACITY (2003)
I
TRENDS IN SEAPORT TRAFFIC
PORT CONTAINER TRAFFIC
6e+08
3e+08
0e+00
1975
1985
1995
2005
2015

Given the difficulty of invasive eradication, the development of a proactive capacity can be one of the most important steps a country can take to protect from invasion (Early *et al.*, 2016). This means preventing new invasions and controlling newly emerging invasive species through measures such as: (i) increasing inspections at borders and ports (e.g., the Agreement on the Application of Sanitary and Phytosanitary Measures (SPS) administered by the World Trade Organization) (Mack *et al.*, 2000); (ii) conducting additional research on potential invasive species and habitats vulnerable to invasion; and (iii) educating the public, land managers, and policy makers on ways to prevent invasion and to control current invasives (e.g., IUCN, 2000). Countries also need reactive capacities (Early *et al.*, 2016) consisting of programs to control already-established invasives and educate various publics on their specific threat to human and ecosystem well-being, as well as evidence of the effectiveness of management policies. Reactive policies, as they focus on already established species, tend to be more common than proactive policies. Despite more resources, few high-income countries have proactive capacities, although there are more such programs found there than in lower income countries, and very few countries have both types of programs. Unfortunately, shortfalls in response capacities can often correlate with the greatest vulnerability to new invasions.

3.4 Climate change as a threat multiplier of degradation drivers

Recent decades have witnessed unprecedented changes in the Earth's atmospheric chemistry and climate system (IPCC, 2014a). Anthropogenic activities including fossil fuel combustion, cement production, deforestation and land-use change have resulted in significant amounts of greenhouse gases (GHGs) being emitted into the atmosphere, and atmospheric concentrations of GHGs including carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) are unprecedented in at least the last 800,000 years (IPCC, 2014a). In addition, fertilizer use, fossil fuel combustion and biomass burning have more than doubled the amount of reactive nitrogen (N), and significantly increased the amount of phosphorous (P) cycling through the Earth system (Vitousek *et al.,* 1997c, 1997b; Matson *et al.,* 2002; Dentener *et al.,* 2006; Galloway *et al.,* 2008; Bobbink *et al.,* 2010). Collectively, these increases have had significant impacts on the Earth's climate system, influencing global temperatures, precipitation patterns and the frequency and intensity of extreme events (IPCC, 2014a).

Temperature, water, CO_2 and nutrient availability exert fundamental controls on most biological processes, and thus climatic changes – such as altered temperature and precipitation regimes – and increased availability of limiting nutrients – such as nitrogen and phosphorus – can significantly impact the functioning of ecosystems, with direct and indirect effects on both land degradation and restoration processes. Climatic changes are also of particular concern because of their role in shaping the extent, severity and frequency of occurrence of other degradation drivers, with effects that in turn feedback to influence potential future climate.

3.4.1. Temperature

Between 1880 and 2012, global average temperatures increased by 0.85°C [0.65° – 1.06 °C], with the last three decades successively warmer than any preceding decade since 1850 (IPCC, 2014a). Even under conservative scenarios temperatures are projected to rise further, with temperatures at the end of the 21st century likely to be 1.5°C or more higher relative to 1850-1900 under most scenarios (IPCC, 2014a).

Many temperate and subtropical species, as well as some tropical species, have shifted their geographic ranges, and altered their phenology and migration patterns in recent years in response to ongoing climatic changes, particularly temperature (Walther *et al.,* 2002; Parmesan & Yohe, 2003; Menzel *et al.,* 2006; Parmesan, 2006; Lenoir *et al.,* 2008; Chen *et al.,* 2009; Pauli *et al.,* 2012; Telwala *et al.,* 2013; IPCC 2014a, 2014b) (also see Chapter 4, Section 4.2.8). Because species differ in their physiological, phenological and demographic responses to temperature changes, and in their ability to move and migrate to track suitable climate, this can result in spatial and temporal mismatches between interacting species and the disruption of key existing ecological interactions (e.g., phenological decoupling of plant-pollinator mutualisms) (Memmott *et al.,* 2007; Tylianakis *et al.,* 2008; Schmidt *et al.,* 2016; Settele *et al.,* 2016), or alternately, the introduction of novel interactions in food webs (Carrasco *et al.,* 2017) in the future. Climate change has been implicated as a driver of some past species extinctions (Pounds *et al.*, 2006; Pounds et al., 1999; Sinervo *et al.*, 2010), and modelling efforts suggest climate change impacts on future species extinctions are likely to be substantial (Malcolm *et al.*, 2006; Sinervo *et al.*, 2010; Thomas *et al.*, 2004; Williams *et al.*, 2003). Although modelling approaches have methodological limitations, and do not always adequately account for species interactions and the multiple responses of species to climate change, the majority of available models nevertheless indicate that many species are likely to decline and face increased extinction risks in the future due to climate change (Bellard *et al.,* 2013b; IPCC, 2014a). For example, based on projected future range distributions for over 1100 plant and animal species, Thomas *et al.* (2004) estimated that even under mid-range climate warming scenarios, between 15 - 37% of the species considered are likely to be 'committed to extinction' by 2050. High elevation range-restricted endemic species are particularly vulnerable in this regard (Dirnböck *et al.,* 2011; Dullinger *et al.,* 2012). Importantly, for taxa such as ectotherms and lowland plants that have narrow temperature ranges or are currently living close to their optimal temperature, extinction risks from warming are likely to be greatest in the tropics, where most of the world's biodiversity is located (Colwell *et al.,* 2008; Deutsch *et al.,* 2008; Huey *et al.,* 2009).

Temperature changes can also affect ecosystem service provisioning by altering the structure and architecture of ecological communities. Warming, which is particularly pronounced at high latitudes (IPCC, 2013), is resulting in a pan-Arctic shrub expansion in tundra ecosystems, with potential impacts on regional biota, ecosystem carbon-, energy- and water- budgets, as well as human activities such as reindeer herding, berry harvesting and access to traditional travel routes (Sturm *et al.,* 2001; Tape *et al.,* 2006; Myers-Smith *et al.,* 2011). In addition, warmer temperatures can also directly drive land degradation by affecting physical processes (also see Chapter 4, Section 4.2.8). Permafrost degradation and glacier retreat, as a result of warmer temperatures, can alter soil drainage and hydrology, destabilize slopes and cause mass movements such as landslides, mudflows and glacier floods, changing habitat for wildlife and vegetation, damaging infrastructure and posing threats to human life (also see Chapter 4, Section 4.2.8) (Cheng & Wu, 2007; Jorgenson *et al.*, 2010; Jorgenson & Osterkamp, 2005; Kääb *et al.*, 2005; Marchenko *et al.*, 2007; Nelson *et al.*, 2001; Schuur *et al.*, 2008).

Future temperature increase is also likely to threaten food security by negatively affecting crop yields and exposing livestock to increased thermal stress, which can reduce livestock productivity and reproductive rates (Howden *et al.,* 2008; Challinor *et al.,* 2014; IPCC, 2014b) (see also Chapter 4, Section 4.2.8). In addition to directional changes in temperature, increased inter-annual temperature variability, which has been documented in some regions such as Western Europe (Alexander & Perkins, 2013), is also a major cause for concern as it can have significant effects on a variety of ecological processes (Stenseth *et al.,* 2002).

3.4.2. Rainfall

Globally, precipitation patterns have been changing since the early to mid-1900s, with some regions of the globe witnessing increases, and others decreases (IPCC, 2014a). Future changes in precipitation are similarly predicted for large parts of the globe, but these are not likely to be spatially uniform (IPCC, 2014a).

Consistent declines in rainfall over time can reduce vegetation productivity and biomass in sites (Barbosa *et al.*, 2015; Hilker *et al.*, 2014; Malhi *et al.*, 2008; Meir & Woodward, 2010; Phillips *et al.*, 2009), while directional increases, on the other hand, can result in enhanced tree cover particularly in more arid and semi-arid biomes such as grassland and savannahs (Barbosa *et al.*, 2015; Fensham *et al.*, 2005; Sankaran *et al.*, 2005, 2008). Even where total rainfall remains unchanged, changes in rainfall climatology towards more frequent but less intense rainfall events, can result in increased tree cover (and vice versa) in these systems by altering the balance between water run-off and infiltration into soils (Good & Caylor, 2011). Woody tree and shrub encroachment can lead to degradation in grasslands and savannahs by reducing grazing potential, livestock carrying capacity and yields (Archer, 2009; Eldridge *et al.*, 2011; Mugasi *et al.,* 2000; Roques *et al.*, 2001; Sankaran *et al.*, 2004). In agricultural systems, directional decreases in rainfall over time and droughts can lead to reduced agricultural productivity (IPCC, 2014b), potentially rendering some areas unsuitable for crop production in the future. In addition to directional changes in mean rainfall, inter- and intra-annual variability in rainfall can also have important consequences for ecological processes (Stenseth *et al.,* 2002) and also directly affect human populations by leading to crop failures, food insecurity and out-migration of humans from affected areas (Afifi *et al.,* 2014).

3.4.3. Extreme events

Besides directional changes in temperature and precipitation, there has also been an increase in the frequency and intensity of extreme events including heavy rainfall events and temperature extremes in many regions over the last several decades (IPCC, 2014a). Importantly, heavy rainfall events, extreme temperatures and heat waves are predicted to become more intense and frequent in many regions (IPCC, 2013, 2014a).

Extreme rain events render areas vulnerable to floods and landslides, while high winds and dust storms can erode top soil, damage crops and infrastructure, reduce air quality and even disrupt transport networks (Michener *et al.,* 1997; Gao *et al.,* 2003; Luino, 2005; Clarke & Rendell, 2007). Similarly, severe droughts can lower crop productivity, reduce water availability for humans, livestock and wildlife, lead to the loss of biodiversity, depress plant performance and survival even in arid and semi-arid systems, and make forests more susceptible to die-offs (Allen, 2009; Allen *et al.*, 2010; Clarke & Rendell, 2007; Hoover *et al.*, 2017; Lewis *et al.*, 2011; Phillips *et al.*, 2009). At present, there are several documented instances of enhanced tree mortality throughout the globe, from modest increases above background rates to regional scale forest die-offs, with a marked increase in documented events since the 2000s (Adams *et al.*, 2010; Allen, 2009; Allen *et al.*, 2010; McDowell *et al.*, 2011; Phillips *et al.*, 2009). Such dieback events have been observed on all six vegetated continents, and across a range of woody ecosystems from monsoonal savannahs and Mediterranean ecosystems to sub-alpine conifer forests and rainforests (Allen *et al.*, 2010; McDowell *et al.*, 2011). Although episodic tree mortality can occur due to natural causes, observations of large-scale mortality from regions receiving less than 400 mm to environments that receive more than 3000 mm rainfall per year and are not normally considered water limited, suggests a common global driver, and is consistent with climate-change induced forest mortality and dieback driven by drought and heat stress (Allen *et al.*, 2010). Such die-offs can have substantial impacts on both market and non-market values of forests including timber and non-timber resource production, carbon sequestration, water quality regulation, cultural services and aesthetic values (Allen *et al.*, 2010; Anderegg *et al.*, 2013).

3.4.4. Elevated CO_2 and nutrient deposition

Atmospheric CO_2 concentrations have increased steadily since the 1950s as a result of fossil fuel combustion, industrial processes and land-use changes (IPCC, 2013), and currently exceed 400 ppm. Concurrently, humans have also drastically transformed global nitrogen and phosphorous cycles through the production of synthetic fertilizers, expansion of nitrogen-fixing crops and the mining of phosphorous compounds (Elser & Bennett, 2011; Falkowski, 2000; Fowler *et al.*, 2013; MA, 2005). The fixation of nitrogen through the Haber-Bosch process is now double that from natural sources (120 Tg N yr^{-1} vs 63 Tg N yr^{-1} as of 2010) (Fowler *et al.,* 2013), and phosphorous inputs to the biosphere have increased approximately four-fold (Falkowski, 2000; Elser & Bennett, 2011).

Elevated atmospheric CO_2 concentrations can stimulate plant production (Nowak *et al.,* 2004; Norby *et al.,* 2005), but the extent of such stimulation can differ between biomes and ecosystem types depending on the nature of limiting nutrients and climate. Typically, the stimulatory effects of elevated CO_2 are more pronounced for C_3 plants compared to C_4 plants (see Box 3.2) (Bond & Midgley, 2000; Bond, 2008; Obermeier *et al.*, 2016).

Box 3.2 C_3 versus C_4 plants

C_3 plants: These are plants characterized by a photosynthetic pathway in which CO_2 is first fixed into a 3-carbon compound. Around 95% of the plant species in the world are C_3 plants, and include trees, shrubs, forbs and grasses. C_3 plants are most efficient under moderate to cool temperatures, high water availability and high CO_2 concentrations.

C_4 plants: C_4 plants employ a photosynthetic pathway derived from C_3 photosynthesis, in which CO_2 is first fixed into a 4-carbon compound. C_4 plants have an additional mechanism by which CO_2 is actively transported and concentrated into specialized cells where photosynthesis occurs. C_4 photosynthesis is most common amongst tropical grasses, including staple crops such as maize, sorghum and millet. Although only around 3% of plant species employ C_4 photosynthesis, over 30% of the Earth's land surface area is dominated by C_4 plants, particularly tropical grasslands and savannahs. C_4 plants do well under high temperatures, low water availability and low CO_2 concentrations.

In C_4-dominated grasslands and savannahs, which cover approximately 20% of the global land surface and more than 70% of Africa's vegetated surface (Scholes & Archer, 1997; Parr *et al.,* 2014; Midgley & Bond, 2015), rising atmospheric CO_2 concentration is believed to be one of the causes underlying widespread woody encroachment, by favouring shrubs and trees over C_4 grasses (Midgley & Bond, 2015; Stevens *et al.*, 2016; Stevens *et al.*, 2017). Shrub and woody plant encroachment has been widely reported from arid and semi-arid grasslands and savannahs across the globe in recent decades (Archer, 2010; Buitenwerf *et al.*, 2012; Eldridge *et al.*, 2011; Fensham *et al.*, 2005; Roques *et al.*, 2010; Stevens *et al.*, 2017; Van Auken, 2000; Wigley *et al.*, 2010), with up to 330 million hectares in the western United States (Knapp *et al.,* 2008; Eldridge *et al.,* 2011) and 13 million hectares in southern Africa (Trollope *et al.,* 1989; Eldridge *et al.,* 2011) estimated to be impacted by bush encroachment. Although the underlying causes for woody plant encroachment are varied and debated, and include changes in land-use, resource extraction, grazing, browsing and fire regimes (Archer, 2010; Eldridge *et al.*, 2011), the consistent increases in woody cover observed across continents despite differences in land-use and management practices suggests a role for common global change drivers such as altered rainfall regimes, elevated CO_2 and nitrogen deposition (Archer, 2010; Eldridge *et al.*, 2011; Fensham *et al.*, 2005; Midgley & Bond, 2015; Stevens *et al.*, 2017; Wigley *et al.*, 2010). In addition to reducing livestock production potential, shrub encroachment can also impact native biodiversity, ecosystem hydrology and nutrient cycling (Archer, 2010; Eldridge *et al.*, 2011). In conservation areas, shrub encroachment can impact visitor numbers and satisfaction as visibility of animals is reduced, with potentially significant economic consequences (Gray & Bond, 2013). Finally, shrub encroachment can also reduce albedo an alter ecosystem energy budgets, with effects that can feedback to further influence climate (Chapin III *et al.,* 1997, Myers-Smith *et al.,* 2011).

There is also evidence to suggest that increased atmospheric deposition of nitrogen and phosphorus is driving species loss and compositional shifts across a range of different ecosystem types, particularly temperate and northern ecosystems (Bobbink *et al.*, 2010; Clark & Tilman, 2008; Duprè *et al.*, 2010; Maskell *et al.*, 2010; Stevens *et al.*, 2004). Enhanced nitrogen and phosphorus deposition can alter nutrient cycling patterns, lead to soil acidification and toxicity, and result in eutrophication of water bodies and the lowering of water quality, with potential impacts on the quality and nature of services provided by ecosystems (also see Chapter 4, Section 4.2.4) (Vitousek *et al.,* 1997a; Bobbink *et al.,* 2010). Although information is currently lacking from several eco-regions in eastern and southern Asia, Mediterranean ecoregions in California and southern Europe, and several subtropical and tropical regions of Latin America and Africa, it is likely that most ecosystems are likely to be affected by future anthropogenically-driven changes in nitrogen and phosphorus availability (Bobbink *et al.,* 2010). In fact, scientists have concluded that we have exceeded the 'planetary boundaries' for the biogeochemical flows of nitrogen and phosphorus, and are now outside the "safe operating space" for humanity where abrupt non-linear global environmental change can no longer be excluded (Rockström *et al.,* 2009; Steffen *et al.,* 2015).

In addition to the effects mentioned above, climate change can also drastically alter ecosystems by inducing wholesale biome shifts. Indeed, long-term field monitoring efforts have revealed several cases of biome shifts in the 20th century in boreal, temperate and tropical ecosystems (Penuelas & Boada, 2003; Gonzalez *et al.,* 2010), many of which appear to be driven by changes in climate, rather than land-use change or other factors (Gonzalez *et al.,* 2010). Model predictions suggest that one-tenth to nearly one-half of the global land surface may be highly (confidence 0.80– 0.95) to very highly (confidence greater than or equal to 0.95) vulnerable to biome shifts by the end of the twenty-first century (Gonzalez *et al.,* 2010). Risks associated with future climate change include both the disappearance of extant climates, as well as the creation of novel climates for which there are no current analogues (Garcia *et al.*, 2014; Williams *et al.*, 2007). Disappearing climates are likely to be concentrated in tropical montane regions and the poleward portions of continents, while tropical and subtropical regions are likely to witness novel climates in the future (Garcia *et al.*, 2014; Williams *et al.*, 2007). Such biome shifts, where they occur, can alter both the extent and kinds of services provided by these ecosystems, with the potential to influence more than a billion people who currently live in these regions (Gonzalez *et al.,* 2010).

3.4.5. Climate change as a threat multiplier

The greatest threat of future climate change arises perhaps not from its role as a direct driver of degradation, but rather from its ability to act as a threat multiplier for other degradation drivers, both by exacerbating the effects of other land degradation drivers, as well as by altering the frequency, intensity, extent and timing of events such as fires, pest and pathogen outbreaks, and species invasions (Allen *et al.*, 2010; Anderson *et al.*, 2004; Bentz *et al.*, 2010; Dale *et al.*, 2001; Dukes & Mooney, 1999; Field *et al.*, 2007; Gisladottir & Stocking, 2005; Mainka & Howard, 2010; Moritz *et al.*, 2012; Webb *et al.*, 2017).

Land degradation can increase the sensitivity of systems to climatic changes and extreme events, and reduce the effectiveness of adaptation options (Gisladottir & Stocking, 2005; Webb *et al.,* 2017). For example, sustainable agro-forestry ecosystems, which tend to be characterized by the presence of more top-soil, less erosion, and diverse and structurally complex vegetation, have been shown to suffer less damage and experience lower economic losses in response to hurricanes than conventional, less diverse or monoculture plantations (Holt-Giménez, 2002; Philpott *et al.,* 2008; Altieri & Nicholls, 2017). Similarly, degradation of grazing lands as a result of shrub encroachment, wind erosion, invasions by exotic species and the loss of perennial forage species can increase the risk of negative impacts and the vulnerability of local communities to future climate changes, while also limiting the effectiveness of adaptation strategies – for example adjusting stocking rates to suit forage availability and reducing the options available to land managers to adapt to future climates (Dougill *et al.,* 2010; Webb *et al.,* 2013, 2017; Briske *et al.,* 2015).

Future warmer temperatures and drought, in addition to acting as major direct stressors for trees, can also render them more susceptible to attack by insect pests and pathogens (Raffa *et al.,* 2008; Bentz *et al.,* 2010). At the same time, warmer temperatures can also have a direct effect on the insect pests themselves by reducing generation times, increasing over-winter survival, and allowing pests to expand their ranges into previously unsuitable habitat, thus favouring insect outbreaks (Raffa *et al.,* 2008; Bentz *et al.,* 2010). Although underlying mechanisms are complex, such interactive effects are believed to be responsible for the bark beetle outbreaks in North America in recent decades, where millions of hectares of conifer forests have been killed from Mexico to Alaska (Figure 3.11) (Raffa *et al.,* 2008; Bentz *et al.,* 2010), and for observed range expansions of the coffee berry borer in East Africa (Jaramillo *et al.,* 2011), amongst others. It is estimated that the coffee berry borer shifted its elevation range upwards by 300 m over a 10 year period in Tanzania, and is now present at altitudes greater than 1800m in East Africa (Jaramillo *et al.,* 2011). Model forecasts suggest that even a 1-2°C increase in temperatures can worsen infestations by the berry borer in *Coffea Arabica* producing areas of Ethiopia, Rwanda, Burundi, the Ugandan part of Lake Victoria, Mt. Kenya and the Mt. Elgon parts of Kenya and Uganda (Jaramillo *et al.,* 2011).

Figure 3 11 **Forest mortality induced by mountain pine beetle, *Dendroctonus ponderosae*, attack. The red foliage shows recent tree canopy mortality.**

Location: Prince George, British Columbia, Canada. Photo: courtesy of USDA Forest Service.

Because of the strong linkages between climate and fire, climatic changes can have significant effects on fire regimes, the signals of which are already becoming apparent. Between 1979 and 2013, ongoing climatic changes have resulted in fire weather seasons lengthening across nearly a quarter of the Earth's vegetated land surface (approximately 30 million km^2) (Jolly *et al.,* 2015). This has resulted in an 18.7% increase in mean fire season length, a doubling of global burnable area affected by long fire weather seasons, and an increase in the frequency of long fire weather seasons across approximately 62 million km^2 of the terrestrial land surface (Jolly *et al.,* 2015). Increases in fire season length can also reduce the time available for prescribed burning as a fire management tool. Modelling results also suggest that projected future climate changes are likely to have substantial effects on fire regimes over vast portions of the globe during the 21st century (Moritz *et al.*, 2012). Although major uncertainties in the projections remain, patterns are likely to be spatially variable across the globe with increasing fire probabilities in the mid- to high-latitudes and decreasing fire probabilities in the tropics (Moritz *et al.*, 2012). Additionally, climatic effects on fire regimes in the future can be further exacerbated when coupled with climate-induced forest dieback, pest outbreaks and other anthropogenic degradation drivers such as deforestation. Such disturbances increase litter and woody debris, and alter microclimatic and fuel conditions and can thus influence subsequent fire risk (Anderegg *et al.,* 2013).

Climate change can also alter the distribution, spread, abundance and impact of invasive species by influencing processes at all stages of the invasion pathway from introduction to establishment (Hellmann *et al.,* 2008; Mainka & Howard, 2010). These include changes in the mechanism of transport and introduction of invasive species, changes in the climatic constraints faced by invasive species, alterations in the distributions of existing invasive species and alterations in the impacts of invasive species (Hellmann *et al.,* 2008). Changes in average annual temperatures between 1900 and 2005 have been shown to be significantly correlated with establishment rates of invasive alien insects across multiple continents, even after accounting for other factors such as increase in international trade during the time period, with a 1°C increase in temperature increasing establishment rates by 0.5 species year^{-1} (Huang *et al.,* 2011). Although our understanding of climate change impacts on invasive species is far from complete (Hellmann *et al.*, 2008; Smith *et al.*, 2012), it is likely that climate-induced changes will further exacerbate the problems of invasive species in many areas (Kriticos *et al.,* 2003, Ward & Masters, 2007; Bradley, 2010; Ziska *et al.,* 2011).

Ultimately, the nature and severity of climate change impacts on biodiversity and ecosystem processes, and the importance of climate change as a degradation driver is likely to be spatially variable across the globe, contingent on the extent to which climatic variables change locally, with some combinations of climate change factors multiplying, and others offsetting, the impacts of different degradation drivers. The net effects will depend not only on the extent and velocity of change in average climates, but also on the probability of occurrence and magnitude of extreme climatic events (Garcia *et al.,* 2014). Further, different ecosystems are likely to vary in their sensitivities to current and future climatic variability (Seddon *et al.,* 2016). Understanding the underlying mechanisms and quantifying ecosystem responses to climatic change remains a challenge, but is critical for developing appropriate tools and technologies to combat and deal with land degradation in the future.

3.5 Avoidance and mitigation of land degradation, and restoration of degraded land

Land degradation is both expensive and hard to reverse. As such, measures to avoid or mitigate land degradation are preferable and are often more cost-effective than restoring land once degraded (Liniger & Critchley, 2007; Turner *et al.*, 2016; Vlek *et al.*, 2017). However, restoring degraded landscapes is often necessary, and it is estimated that between 1 and 1.5 billion people globally are already directly negatively affected by land degradation (Thomas *et al.*, 2013) (also see Chapter 5). In addition, restoration of degraded land is also needed to reduce the pressure to clear remaining areas of native vegetation (Lataweic *et al.,* 2015).

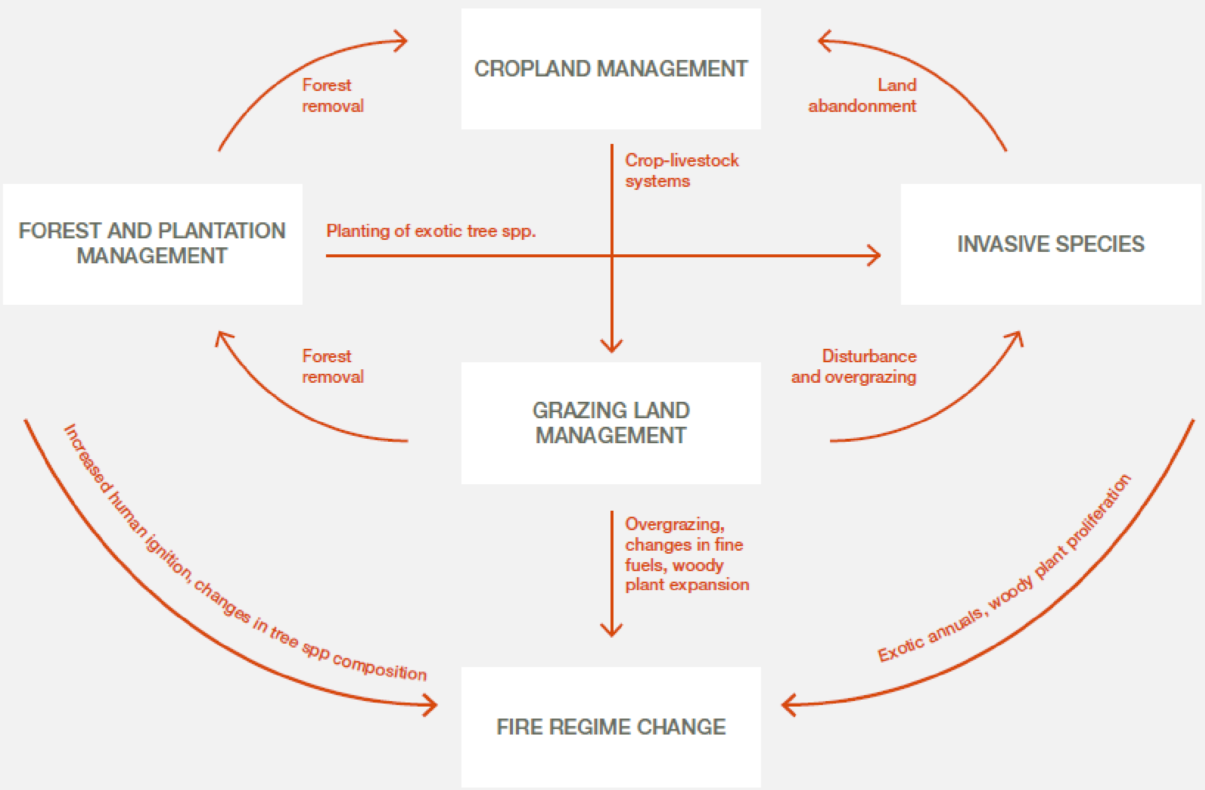
Avoiding or mitigating degradation requires both the development of sustainable land management practices and their application through institutional and legal measures (e.g., education and training, financial incentives, policies, or legislation; see Chapters 6, Section 6.4 and Chapter 8, Section 8.3). This is not always an easy or straightforward process (Winslow *et al.,* 2011). A particularly significant challenge is the alignment of government policies and regulations across multiple government departments and agendas, including the need for extensive coordination among institutions, stakeholders, and practitioners (see in particular Chapter 8, Sections 8.3, 8.4.2, 8.4.3) (Liniger & Critchley, 2007). Typically, actions that avoid or mitigate degradation have been more readily adopted when large losses of resources have been apparent, such as the loss of soil stability and carbon stocks in the USA through tillage practices (Bernacchi *et al.,* 2005; Lal *et al.,* 2007).

If prevention measures fail, then active measures to restore degraded land may be needed. However, restoration is generally very difficult, regardless of ecosystem type, and is often cost-prohibitive (Liniger & Critchley, 2007). Because restoration requires the consideration of multiple factors and institutional capacities that are site-specific, there is no overarching decision support tool available (see Chapter 8). In order for restoration to be successful, a thorough analysis of available local knowledge and published information is needed to identify the costs and benefits of different options (see Chapter 8, Section 8.2.2).

In addition, the development of an evaluation framework is needed to understand the relative importance of different indirect and direct drivers, identify priority actions and define restoration goals (Liniger & Critchley, 2007) (see Chapter 6, Section 6.3). Lastly, the institutions responsible for restoration efforts need to be identified and developed through participatory approaches (Liniger & Critchley, 2007) (see Chapter 8, Sections 8.3.2, 8.3.4).

Successful restoration has occurred where efforts are limited in space and time, and where restoration goals are clearly defined and achievable with the available resources. For instance, many non-native *Acacia* trees were, and continue to be, planted in African dryland sites (e.g., http://www.fao.org/news/story/en/item/80060/icode). Some of these species, such as *Vachellia (Acacia) reficiens,* have become highly invasive. Originally planted for fuel and forage, they soon formed impenetrable thickets, restricting livestock and human access. Sites invaded by this tree have been successfully restored in northern Kenya (e.g., http://www.nrt-kenya.org/). The restoration of invaded areas has been accomplished by removing the degradation driver (here, the planting of trees), avoiding unintended consequences of removing the driver (here, preventing soil erosion by using dead limbs), facilitating restoration goals (here, reseeding desired grass species) and setting achievable restoration targets (here, establishing grass productivity, not necessarily nutrient or carbon cycles). Due to cost and limited resources, such efforts have only been successful at a small local scale.

Figure 3 12 The management of croplands, grazing and forest lands influences changes in invasive species and fire regimes via a complex array of interactions and feedbacks.



However, restoration is often unsuccessful, even in situations where significant resources are available and where there is an in-depth understanding of degradation drivers and processes. Complex interactions among direct drivers and altered ecological processes can confound restoration efforts, creating significant challenges for restoring degraded lands unless the feedback loops among the drivers can be altered to promote restoration (Figure 3.12). For instance, soil surface disturbance resulting from grazing, cropland abandonment, and off-road vehicles have facilitated the large-scale invasion of the exotic annual grass *Bromus tectorum* (cheatgrass) in the western USA (Germino *et al.,* 2016a). The widespread presence of cheatgrass has dramatically increased fire cycles across these landscapes (Figure 3.13), resulting in extirpation of many native plant and animal species, as most were not fire-adapted, and altered nutrient, carbon and water cycles. The niches left open by the absence of the native plants have allowed for further invasion, resulting in increases in fire. Soil erosion (wind and water) following each fire has led to further site degradation. Until this feedback loop can be broken, the original structure, composition, and functional attributes of these once shrub-dominated ecosystems will be lost across millions of hectares (Germino *et al.,* 2016a). However, despite extraordinary efforts and large amounts of resources put towards breaking this fire cycle, efforts have failed.

Figure 3 13 Cheatgrass (*Bromus tectorum*) fire in a sagebrush ecosystem, California, USA.

Cheatgrass promotes more frequent fire that native perennial shrubs are not able to withstand. Large expanses of the Great Basin ecosystem of the USA have been transformed into cheatgrass monocultures. Copyright © 2015 The Regents of the University of California. Used by permission.

Effective avoidance, mitigation, or restoration of degraded lands can contribute to the attainment of multiple Sustainable Development Goals. Averting land degradation in the face of future climate change, and ensuring that global development and food security goals are achieved, will, however, require innovative management and policy solutions that simultaneously benefit land, biodiversity and climate (Webb *et al.,* 2017). It is imperative that measures to reverse degradation and mitigate climate change be based on an understanding of land degradation that is tailored to different ecosystem types and land-use systems. For example, tree planting, which is a common strategy both to restore lands and reduce CO_2 emissions through the sequestration of carbon – while ecologically reasonable in deforested landscapes – can be catastrophic when applied to grasslands and savannahs (Bond, 2016; Griffith *et al.*, 2017; Parr *et al.*, 2014; Strassburg *et al.*, 2017; Veldman *et al.*, 2015a, 2015b). Grasslands and savannahs are ancient ecosystems that are globally extensive, support significant biodiversity, and provide critical ecosystem services to an estimated one-fifth of the world’s population, but are often misperceived and undervalued for their conservation potential (Bond, 2016; Bond & Parr, 2010; Griffith *et al.*, 2017; Parr *et al.*, 2014;

Ratnam *et al.*, 2016; Veldman *et al.*, 2015a, 2015b). Failure to differentiate reforestation (i.e., planting trees on deforested land) from afforestation (i.e., planting forests where they did not historically occur) can have negative environmental consequences similar to those that occur from deforestation, and rather than restoring can lead to degradation by reducing habitat for livestock and animals adapted to open environments, reducing native biodiversity, and altering water-, energy- and nutrient cycles (Bond, 2016; Griffith *et al.*, 2017; Ratnam *et al.*, 2016; Veldman *et al.,* 2015a, 2015b).

3.6 Indirect drivers of land degradation and restoration

3.6.1 Defining indirect drivers of land degradation and restoration

Indirect drivers of how humans both use and impact natural resources, and hence determine processes of land degradation and restoration, are generally conceived in terms of five sets of factors: (1) demographic; (2) economic; (3) technological; (4) policy and institutional; and (5) cultural (Geist & Lambin, 2004, 2002) (Table 3.2). These drivers can operate either over long timescales, such as through changes in demographic variables, or much more quickly, such as through the introduction of new policies or economic incentives (Table 3.4).

There is ample evidence to suggest that many (if not most) of the changes in how land is used and managed come from individual and social responses to perceived economic opportunities, such as a shift in demand for a particular commodity or improved market access, moderated by institutional and political factors (e.g., agricultural subsidies and low-interest credit, or government-led infrastructure projects) (Lambin *et al.*, 2003; Nkonya & Mirzabaev, 2016) (also see Chapter 2). Household wealth can also have a powerful moderating effect on how economic opportunities play out in different regions and different communities, affecting for example, who is able to develop, use and profit from new technologies for managing land (Lambin *et al.,* 2003).

Technological factors are often closely associated with economic drivers of land degradation and restoration, whether through intensified farming techniques and biotechnology, high-input approaches to rehabilitating degraded land or through new forms of data collection and monitoring. Technological factors can have a transformative effect on human-environment interactions, but these can be either positive or negative depending on the social, political and economic context in which a new technology is introduced (Table 3.4).

Demographic factors generally operate across longer timescales (i.e., decadal or more) than economic factors and are manifested through a wide range of variables from the impact of changing mortality and fertility levels on background population growth rates to changes in migration patterns from rural to urban areas or between rural areas, to changes in how society is structured (e.g., in the size and interconnectedness of family units) (Royal Society, 2012). The impacts of changes in demography on land degradation and restoration are myriad and inconsistent, with evidence that population changes can, depending on the context, both ameliorate and exacerbate the extent to which natural resources are managed sustainably (Boserup, 1965; Blaikie & Brookfield, 1987). High rural population densities have been associated with both the improvement and degradation of agricultural lands (Barbier & Hochard, 2016). Migration, in its various forms, is perhaps the most powerful driver of changes in patterns of land and resource use at decadal timescales, and can precipitate a cascade of other political and economic changes (Lambin *et al.,* 2003) – seen most vividly in the context of the rapid urbanization of rural people across the developing world.

Table 3.4 Typology of drivers of land-use change and associated land degradation and restoration processes. Adapted from (Lambin *et al.,* 2003).

Rate of change of the driver	Resource scarcity	Changing opportunities created by markets and technological change	Outside policy intervention	Loss of adaptive capacity and increased vulnerability	Changes in social organization, in resource access and in attitudes
Slow	Population growth Division of family units Loss of land productivity following unsustainable use and increased migration to remaining productive areas Failure to restore depleted natural resources Incentives and efforts to increase efficiency of resource use	Increased commercialization and supply chain development Improved access to markets through infrastructure Changes in market prices to inputs and outputs Off-farm wages and employment opportunities Emergence of markets for sustainable products or for ecosystem services	Economic development programs Subsidies and fiscal incentives Frontier development to support political agendas Poor governance and corruption, or gradual strengthening of governance Insecurity in land tenure Development and implementation of land zoning plans	Impoverishment (debts, lack of access to credit, lack of employment alternatives) Breakdown of social capital Dependence on external assistance (e.g., credit, welfare payments) Social discrimination	Changes in institutions governing access to resources (e.g., shift from communal to private, decentralization) Rural-urban migration for education, health and employment benefits Lack or increase of access to information and education
Fast	Economic migrants and conflict refugees Decrease in land-availability due to competition with other land uses (e.g., protected areas) Sudden perception of crises that precipitate demands for structural changes	Capital investments Changes in national or macro-economic and trade conditions New technologies for intensification of resource use	Rapid policy changes (e.g., devaluation, moratoria or ban) Government and political instability (e.g., elections) War	Internal conflicts Health (e.g., HIV) Hazards associated with natural risks	Marginalization due to the loss of entitlements to environmental resources (e.g., through major infrastructure and conservation projects Public awareness due to NGO and other campaigns

Institutional factors often play a key moderating role in determining the relevance and impact of changes in economic and demographic variables on patterns of resource use and exploitation. Institutions encompass not only the rule of law and other legal frameworks but also other social structures that may be equally if not more important in determining how land is managed, including: formal and informal property rights regimes and their enforcement; information and knowledge exchange systems; informal institutions and social processes such as corruption and elite capture, civil society networks and movements, and local and traditional knowledge and practice systems (Lambin *et al.*, 2003; Young, 1982) (also see Chapter 2 and Chapter 8).

Cultural factors, whilst often far less recognized and understood can have a powerful and long-lasting effect on how individuals and whole human communities and nations relate to both environmental opportunities and challenges. Perspectives from psychology and sociology have illustrated the often-stark distinctions between economic and social well-being in many rural regions (Easterlin *et al.,* 2010), underpinned by the high and sometimes overriding importance that is placed on non-monetary benefits, which in turn are grounded in local and regionally relevant social and cultural values. Perceptions and the concepts with which these perceptions are interpreted (and which themselves are shaped by culture), strongly affect how humans react to changes in their natural or social environment (see also Chapter 2, Section 2.2). A key cultural factor that has a profound influence on how economic development impacts the use of natural resources is diet, and in particular, increased consumption of meat. Indeed, dietary change may override both yield-enhancing technologies and even population growth as a major driver of land requirements, and thus as a driver of the risk of land degradation (Kastner *et al.*, 2012).

3.6.2 Emergent characteristics of indirect drivers of land degradation and restoration

Indirect drivers of land degradation and restoration do not, by their very nature, lend themselves to reductionist analyses or highly prescriptive policy solutions. Rather, indirect drivers interact with each other in complex, inter-dependent ways, reaching across both short and long-term periods and geographic distances whilst also being subject to feedback effects from the direct drivers that they influence in the form of changes to ecosystem services and human wellbeing (Díaz *et al.,* 2015). Yet, the complexity through which different indirect drivers of land degradation and restoration operate is not completely irreducible, and distinct patterns or modalities of impact can often be discerned (Lambin *et al.,* 2003, Geist & Lambin, 2004; Nesheim *et al.,* 2014). Lambin *et al.* (2003) propose a typology of five high-level causes of land-use change: resource scarcity, market opportunities, external policy intervention, loss of adaptive capacity and changes in social organization. Each of these may be underpinned by multiple indirect drivers. In untangling the importance of different indirect drivers, it is possible to identify a number of emergent characteristics of the way in which indirect drivers of land degradation and restoration typically operate on the ground (and see Table 3.4).

3.6.2.1 Drivers are multiple and interacting

A defining feature of indirect drivers of land degradation and restoration is that they very rarely, if ever, operate in isolation from each other. Indirect drivers of land degradation and restoration typically operate as ‘contributory’ or ‘combinatory’ causes. Indeed, multiple factors are often necessary to determine a particular outcome, but that same outcome can also be achieved by different combinations of factors – for example different processes of forest transition (see below).

Indirect drivers commonly combine to result in complex reinforcing and dampening effects that in turn produce the enabling and disenabling conditions that shape direct drivers of degradation and restoration (Geist & Lambin, 2002, 2004) (Box 3.3). In a landmark review of the underlying driving forces of tropical deforestation, Geist *et al.* (2002) found that often 3-4 indirect drivers (e.g., economic, technological and institutional factors) underpinned the majority of direct drivers (e.g., agricultural expansion, infrastructure development and timber extraction). The same authors came to similar conclusions when assessing the drivers of dryland degradation (Geist & Lambin, 2004). The fact that land degradation processes are so commonly underpinned by a number of interacting drivers challenges popular single-factor explanations that place much of the blame for land degradation on, for example, high densities of rural poor – an interpretation that can be easy to reach when only assessing surface patterns (e.g., population density in South Africa) (Hoffman & Todd, 2000) or property sizes in the Brazilian Amazon (Michalski *et al.,* 2010), at the expense of a deeper analysis of underlying factors that may have resulted in those patterns. For example, corruption is often an important institutional driver of land degradation, as the prospects of the money that can be gained by political and administrative officials from extractive activities through corruption can encourage them to overlook or even support these activities (Cerutti *et al.*, 2013). But another study on South America showed that improvements in general indicators of governance, including corruption, can promote deforestation, likely by providing an environment more conducive to business investments (Ceddia *et al.*, 2015).

Natural environmental variability interacts with underlying human causes of land degradation and restoration in important ways. In particular, the spatial variability in environmental resources has a strong moderating effects on human activities – as manifested for instance in the patterns of road expansion into areas that are more suitable for agriculture (Chomitz & Gray, 1996). Sometimes variability in natural conditions can override the influence of socioeconomic variables. For example, Redo *et al.* (2012) found that environmental variables such as temperature, precipitation and elevation are consistently associated with patterns of deforestation and regeneration in Bolivia, outweighing the influence of many socio-economic variables, including population density. Similarly a number of studies have demonstrated the key role that changes in precipitation has had in driving land degradation in Africa, including in both the Sahel (Ayoub, 1999) (see also Chapter 4, Section 4.2 and a case study in Chapter 1) and South Africa (Wessels *et al.,* 2007). Understanding the importance of these natural factors is critically important to avoid misleading interpretations about the significance of demographic, economic and political factors. That said, it is also important to recognize that anthropogenic factors can play a dominant role in exacerbating the impact of otherwise natural drivers. The obvious example here is global climate change (see Section 3.4), but there are many others. One example that is particularly prominent in many parts of the tropics is landslides. Landslides are commonplace in many mountainous regions but are much more likely to occur in more deforested and human-modified areas (Guns & Vanacker, 2013).

Box 3.3 Synergistic interactions between indirect drivers of land degradation

It is possible to distinguish three modes of underlying causation of land degradation: (1) **single-factor causation** (one individual underlying factor driving one or more direct drivers); (2) **concomitant occurrence** (independent, separate operation of factors); and (3) **synergistic causation** (several interlinked factors acting together) (Geist & Lambin, 2004). In their meta-analyses of the drivers of deforestation and dryland degradation Geist and Lambin (2002, 2004) identified extremely few cases where it is possible to isolate a dominating influence of one indirect driver that is responsible for determining human activities that result in land degradation, concluding instead that the most common type of causation is due to synergistic interaction between multiple drivers.

In many situations indirect drivers operating at multiple spatial scales, and in different geographies, combine to shape the activities of a particular land-use sector and its implications for land degradation and restoration outcomes. Liu *et al.* (2013) reviewed the iconic case of the soybean trade between Brazil and China which provides an illustrative example of this. A superficial analysis identifies the strong demand for soy bean products, including animal feed (mostly pigs) in China as being the dominant indirect driver. However, interacting with this demand are the political influences of the Chinese government in pursuing foreign investments and the Brazilian government in developing an export market. Strong cultural preferences for soybean products underpins the economic demand from China, whilst landmark developments in agricultural technology and selective breeding by Embrapa, Brazil´s agricultural research institution, were critical in enabling Brazilian farmers to plant soy in the otherwise infertile soils of the Brazilian cerrado.

A frequently encountered situation of dryland degradation can be seen in the creation of water-related infrastructure resulting in the expansion of irrigated croplands and pastures. Underlying this expansion is a set of political, economic and technological factors that, in developing countries, are often underpinned by national policies aimed at consolidating territorial control over remote, marginal areas and attaining self-sufficiency in food and clothing (Geist & Lambin, 2004). Some of the most powerful examples of this can be found in Central Asia. For example, in Turkmenistan agriculture is almost entirely dependent on irrigation, initially established in the Soviet era and driven, in particular, by a desire to rapidly expand the production of cotton. However, flaws and inefficiencies in the design of these irrigation systems has led to widespread soil and water degradation due to waterlogging and salinization with significant implications for the country's plans to diversify its agricultural base and enable its food requirements to be met (O'Hara, 1997). This same pattern can be found across the Aral Sea drainage basin, encompassing much of Turkmenistan, Uzbekistan and Tajikistan and leading to one of the world´s worst examples of desertification (Saiko & Zonn, 2000)

3.6.2.2 The key drivers and their effects are context dependent

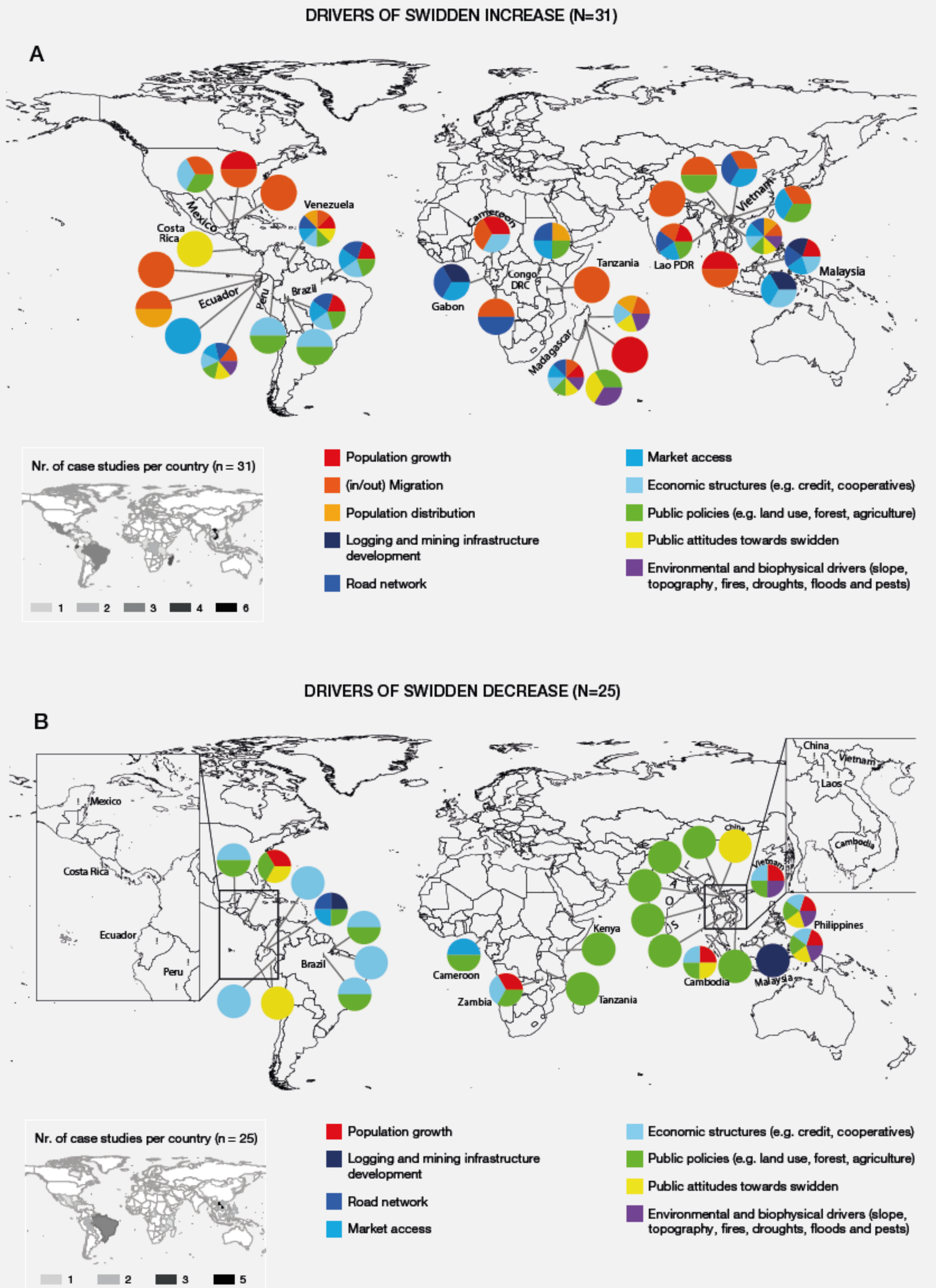
Profound differences in the biophysical, economic, social and political context of different regions of the world can mean that different drivers are varyingly important in different places. For example, at the global scale, commercial agriculture is a much more important driver of deforestation in the more economically developed countries of Latin America and Southeast Asia compared to many less developed countries in Africa, where subsistence agriculture is more important (Kissinger *et al.,* 2012a). Similarly, wood collection and charcoal production are the primary drivers of forest degradation in Africa, where the majority of the rural population still relies on biomass for household energy, while commercial logging is the major factor of forest degradation in tropical Asia (Kissinger *et al.,* 2012a). The extent to which swidden agriculture has been replaced by commercial, intensive farming techniques offers one of the most visible contrasts between different agricultural regions across the world. In more remote regions, that are poorly connected to international markets and where farmers have unequal or insecure access to investment or market opportunities, swidden agriculture remains an important way of life for millions of people (van Vliet *et al.*, 2012) (Figure 3.14).

Further, because of the predominance of interacting causes, the same driver can have very different effects depending on the presence or absence, or characteristics of other contextual factors. Such context-specific effects can be readily identified at smaller scales. In a landmark study (Brashares *et al.,* 2011) demonstrated how the same indirect driver – household wealth – can have contrasting effects on exploitation of natural resources simply dependent on location, with wealthier households consuming more bushmeat than their poorer neighbours when in settlements nearer urban areas, but with the opposite pattern in more isolated settlements.

This context-specificity of effects makes it challenging to draw general predictions on the sustainability impacts of specific indirect drivers, especially given the constantly shifting backdrop of phenomena that shape how land and natural resources are used, undermining the extent to which studies can be compared across regions and over time. Yet, it is sometimes possible to produce a contextual generalization (i.e. identify a chain of mechanisms which is valid for explaining a relatively well-bounded range of phenomena) and the conditions or contextual factors which trigger, enable or prevent this causal chain (Meyfroidt, 2016). Such types of generalization can be considered as a “middle-range” or “typological” theory (George & Bennett, 2005; Meyfroidt, 2016). One example of this can be seen in the context of agricultural intensification, driven by market growth and technological developments (see Section 3.6.3).

Figure 3 14 **Indirect drivers of increases A and decreases B of swidden agriculture across the tropics, based on a meta-analysis of 157 cases of swidden agriculture.**

In more remote regions that are poorly connected to international markets, and where farmers have unequal or insecure access to investment or market opportunities, swidden agriculture remains an important way of life for millions of people. Source: Redrawn from van Vliet *et al.* (2012).



It is also the case that certain factors can have a dominant if not overriding influence on patterns of resource use. Distance to urban centres is one such factor. Working in Tanzania, Ahrends *et al.* (2010) found that the distance from Dar es Salaam was the most important factor determining waves of forest exploitation. The highest value resources (timber in this case) were exploited first, closest to the city, followed by lower value resources (charcoal). Strikingly, after taking account of the distance from Dar es Salaam, there was no relationship between the level of degradation within a forest and management policies or institutions governing that forest (Ahrends *et al.,* 2010)

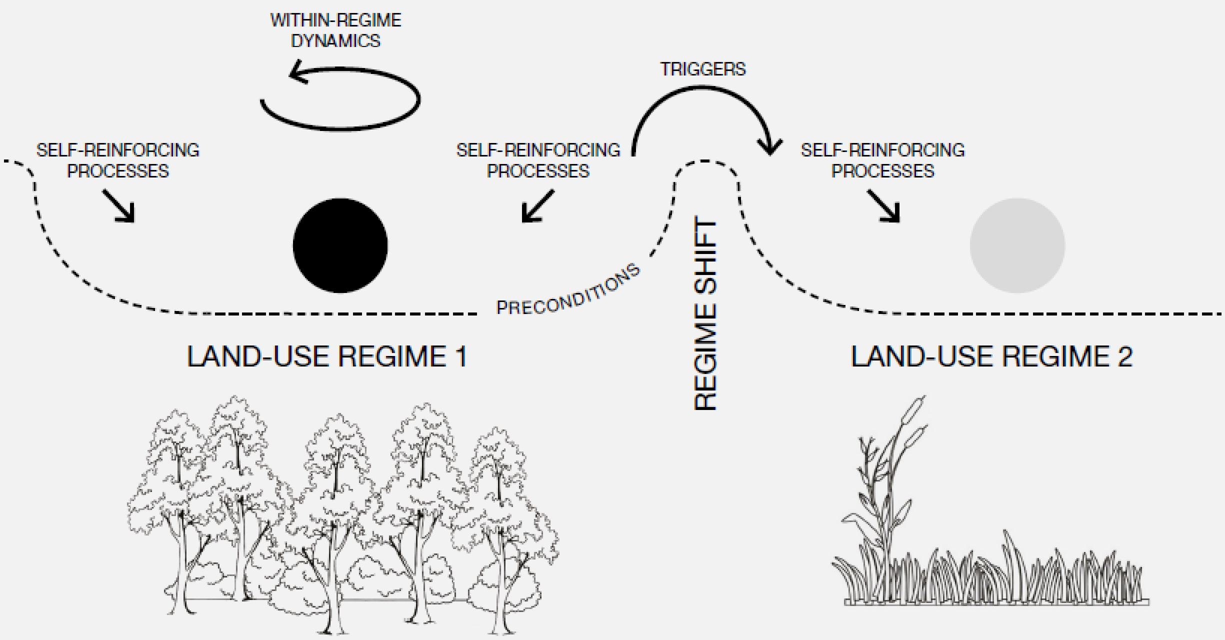
Finally, the characteristics and agency of different actors is also a very significant factor that moderates the relation between indirect drivers and actual management decisions and thus direct drivers and sustainability outcomes (van Vliet et al., 2015). Heterogeneity and agency of the actors are often overlooked in both research and policy design.

3.6.2.3 Rapidly changing drivers can lead to non-linear change

Whilst some indirect drivers of land degradation and restoration only play out over decadal or longer timescales, others are capable of changing much more rapidly and can have almost immediate impacts. Rapid changes in direct drivers are perhaps most evident in the context of an increasingly globalized and integrated economy, where changes, for example, in price, trade regulations and investor decisions of internationally traded agricultural commodities can have a profound effect on the behaviour of producers on the opposite side of the world (Meyfroidt *et al.*, 2013; and see below). Rapid changes in political and governance variables can also quickly reshape the types of challenges and opportunities that face efforts to manage land resources more sustainably. A positive example of this may be seen in the recent boom of private and public sector commitments to deliver zero deforestation agricultural supply chains (e.g., as manifested in the New York Declaration on Forests; see Climate Focus, 2015) offering an unprecedented opportunity for coordinated action to curb the deforestation and environmental degradation that is embedded in global trade.

Figure 3 15 Framework for understanding land-use regime shifts.

Many rapid, non-linear changes in how land resources are used are driven by powerful positive feedbacks, where initial disturbances or interventions precipitate, for better or worse, a cascade of further changes, resulting in a shift between different land-use regimes. Source: Redrawn from Ramankutty & Coomes (2016).



Non-linear trajectories of land-use change correspond to distinct "land-use transitions", resulting from combinations of endogenous social-ecological feedbacks (i.e., changes in land use following a decline in the provision of important ecosystem goods and services) and exogenous socio-economic change, driven by urbanization, economic development or globalization (Lambin & Meyfroidt, 2010; Müller *et al.,* 2014). Forest transitions are a prominent example of land-use transitions, and encompass a number of different processes by which large areas of land may become available for reforestation (Box 3.4). Many rapid, non-linear changes in how land resources are used are driven by powerful positive feedbacks, where initial disturbances or interventions precipitate, for better or worse, a cascade of further changes (Ramankutty & Coomes, 2016) (Figure 3.15). Such path dependent dynamics provide a level of predictability regarding the likely negative or positive impacts of a particular management intervention in a particular context. Three sets of mechanisms and reinforcing dynamics are needed to constitute a regime shift: (1) preconditions; (2) triggers (immediate cause); and (3) the self-reinforcing processes that maintain the regime in the new state (Ramankutty & Coomes, 2016) (Figure 3.15). A commonly cited example of a regime-shift is the shift from small-scale farming in the Brazilian cerrado until the early 1970's to industrial-scale soy farming. Here the development of new tropical soy varieties was a key precondition for the change. Trigger factors included the collapse of anchovy fisheries in Peru (contributing to the rise of soybeans as a substitute), and 1973 drought in the USA, which raised the price of soy on international markets. Reinforcing mechanisms of this regime shift included credit subsidies allowing farmers to adopt new technologies and machinery (Ramankutty & Coomes, 2016).

Box 3.4 Drivers of forest transitions

A forest transition describes a shift, usually assessed at the national scale, from net forest loss to net forest gain, whether through natural recovery or planted forests (Mather, 1992). Three main types of land-use dynamic explain a process of forest transition, or how land is made available for restoration of natural ecosystems (Jadin *et al.*, 2016). **First**, space can be created for reforestation by the intensification of agricultural and forestry intensification, allowing for increasing the output per unit of land (Green *et al.,* 2005). In this "active land-sparing" thesis, demand for agricultural products – related to population growth, affluence and market engagement – constitutes a necessary cause of intensification, but is moderated by other technological, institutional and socioeconomic variables, including land scarcity (Boserup, 1965; Turner & Ali, 1996). **Second**, a spatial redistribution of land use to better match land suitability following a process of progressive learning may also result in intensification, and a form of "passive land-sparing" (Mather & Needle, 1998). **Third**, international trade in land-based products may facilitate forest recovery in one place by displacing pressure on environments elsewhere, possibly as a form of leakage (i.e., displacement resulting from environmental conservation policies) (Meyfroidt *et al.,* 2010).

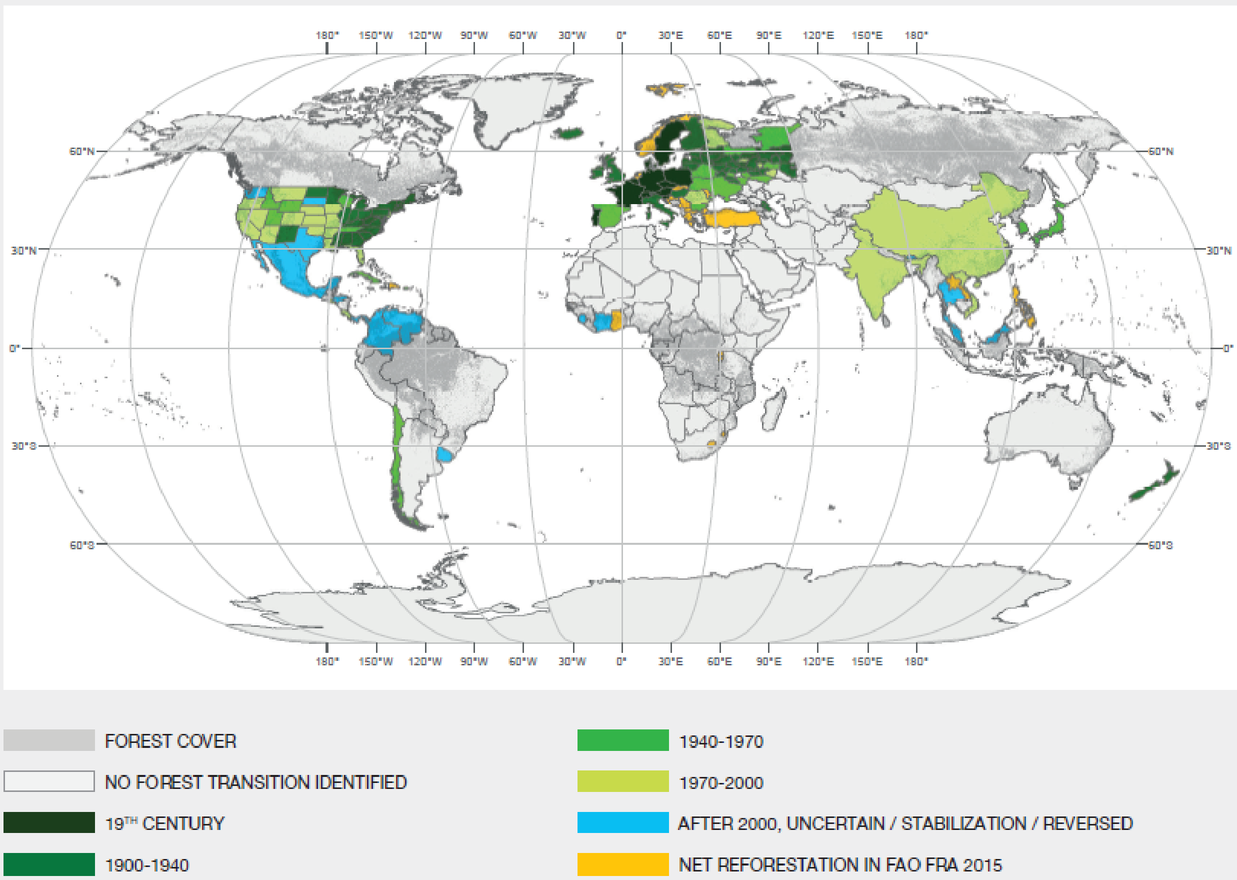
These processes typically interact with each other and are supported by broader dynamics – or underlying drivers – in the technological, economic, political, cultural and institutional spheres of human societies, that have been stylized in a series of "pathways" (Rudel *et al.,* 2005). In an **economic development pathway**, intensification occurs in the context of changing economies with urbanization and industrialization driving labour scarcity in agriculture and concentrating production on the most suitable land, thereby retiring marginal agricultural lands from production. In the **forest scarcity pathway**, economic, political, and cultural reactions to environmental degradation and the scarcity of forest products drive more active measures and policies to support forestry intensification, tree plantation and rehabilitation, set-asides and protection of remaining natural habitats (Hyde *et al.,* 1996; Rudel *et al.,* 2005; Lambin & Meyfroidt, 2010). Variants of these pathways include the **globalization pathway**, which puts an emphasis on multiple processes associated with globalization that affect forest cover, and the **state forest policy pathway**, which takes into account multiple motives, beyond forest scarcity, for which governments protect and plant forests (Lambin & Meyfroidt, 2010). Forest transitions in the 19th and early 20th centuries occurred mainly in temperate developed regions, but national-scale forest transitions have recently been observed in tropical regions as well (Figure 3.16).

As can be seen from the example of the Brazilian cerrado new agricultural frontiers can emerge very rapidly, and comparative analyses of environmental, economic and governance conditions can provide valuable insights on the potential development of newly emerging frontiers. Gasparri *et al.*, (2016) identify the enormous potential for soy expansion in Southern African dry forests and savannahs by highlighting strong similarities in environmental, institutional and other contextual characteristics between Southern Africa and South America. The expansion of soy into Southern Africa is further strengthened by emerging linkages between Southern African and the South American region, including knowledge and capital into infrastructure development, land acquisition, agricultural research, and institutional reforms (Gasparri *et al.,* 2016).

Figure 3 16 Timing of observed transition from decreasing forest area to increasing forest area, in regions where a transition has been observed.

Forest transitions in the 19th and early 20th centuries occurred mainly in temperate developed regions, but national-scale forest transitions have recently been observed in tropical regions as well. Forest cover is shown by shading, using data from the ESA CCI Land Cover project. Blue corresponds to countries where recent forest cover dynamics indicate a stabilization of net forest cover without confirmation by in depth case studies, or where a shift to net reforestation has been followed by another reversal to net deforestation. Orange corresponds to countries where FAO Forest Resource Assessment 2015 indicates a net reforestation, but without confirmation by in depth case studies. Source: Based on Meyfroidt & Lambin (2011) and additional unpublished data (Meyfroidt, personal communication).

TIMING OF FOREST TRANSITIONS



Land-use transition is sometimes presented as a quasi-deterministic process, associated with the theory of environmental Kuznets curves (EKC), where environmental degradation is expected to increase in the early stages of economic development, and then decrease once per capita income has exceeded a certain threshold, in a trajectory that can only be delayed or accelerated by policies (Barbier *et al.,* 2010). Empirical evidence of EKC, which frequently relies on cross-national approaches, remains equivocal at best (Chowdhury & Moran, 2012). A commonly stylized land-use transition trajectory suggests that land use in a particular region typically follows a series of transitions that accompany economic development - from wildlands with low human population densities to frontier clearing and subsistence agriculture with the majority of the population employed in food production for local consumption to intensive agriculture supporting mainly urban populations (DeFries *et al.*, 2004; Fischer-Kowalski & Haberl, 2007). Such stylized frameworks have been criticized as being overly deterministic and overlooking many of the complexities that define real land-use trajectories (Perz, 2007; Walker, 2008). Further, they generally treat countries as closed entities, overlooking the connections between countries that are made through trade, power relations, and exchange of information (see Chapter 2) and other drivers. Instead, the context-dependence of land-use transitions is manifest in the fact that the same underlying drivers can have opposite effects depending on their combinations and the presence of different contextual factors. Further, land-use transitions result not only from local dynamics, but also from the influence of distant drivers and complex interconnections between social-ecological systems that are separated geographically (Lambin & Meyfroidt, 2011). Land-use transitions in one place may also have direct and indirect consequences of land-use changes in other places.

To highlight only a few factors, outmigration and decreasing population pressure may release pressure on land, but increasing population densities may also trigger forms of tree-based land-use intensification in line with the narrative of "more people, less erosion" (Tiffen *et al.,* 1994; Kabanza, 2013). The dynamics of international trade may encourage improvements in the productive potential in each region, leading to agriculture being concentrated on the best farmland and potentially relieving pressure on marginal ecosystems (Kastner *et al.*, 2014). Or conversely it may encourage countries with less strict environmental regulations to exploit their natural resources or accept more polluting activities in order to serve consumption of more developed countries (de Waroux *et al.*, 2016).

3.6.3. Beyond the caricatures

A number of caricatures are popularly invoked as providing general explanations for the over-use of natural resources (see Chapter 2). Foremost amongst these are a focus on either population growth or rural poverty as singular explanations of land degradation. Whilst both factors are unquestionably important, decades of research on human-environment relationships illustrates that neither population nor poverty alone constitute the sole or even major indirect driver of land and natural resource use worldwide (Lambin *et al.,* 2001). The importance of population growth was immortalized in the academic literature on sustainability by the elegant simplicity of the IPAT equation (Impact = Population x Affluence x Technology), first proposed by Ehrlich & Holdren (1971), that combines population with the changing affluence of human populations and the human capacity for technological ingenuity to predict environmental impacts. Put simply, the IPAT formulation invokes the notion that the use and extraction of environmental resources operates within a closed system, subject to an exogenous influence of technology. This perspective places the "culprit" (e.g., high population growth) as being in close proximity to the place in question and consequently identifies placing limits, in this case, on population growth, as the solution (Lambin *et al.,* 2001). Both population growth and poverty play an unquestionably important role in shaping how humans exploit natural resources (Dietz *et al.,* 2007; Chomitz, 2007). However, a growing body of research inspired by systems thinking in land change science and research on the resilience of coupled social-ecological systems has underscored the conclusion that neither factor constitutes the sole or major cause of changes in land use and degradation for much of the world. Rather, that people´s response to economic opportunities, moderated by a complex and multi-scale set of contextual factors, including the often overriding effect of other, often higher-level and interacting institutional, political and social drivers are responsible (Hecht, 1985; Lambin *et al.*, 2001; Meyfroidt *et al.*, 2013). Indeed, a comprehensive modelling exercise exploring trade-offs among environmental conservation initiatives and food prices in a set of scenarios showed that variation in population and economic growth were much less important than resource-use and management policies in determining the eventual achievement of land resource-related Sustainable Development Goals (SDGs) (Obersteiner *et al.*, 2016).

Some underlying factors that play a key role in shaping patterns of land degradation and restoration are more usefully viewed as emergent properties of other drivers, rather than a driver in their own right, as they are commonly interpreted in general explanations. A classic example of such a factor is competition for land (Smith *et al.*, 2010). The relationship between land degradation and competition for land is multifaceted. Land degradation is often invoked as a primary cause of increased resource scarcity and competition for land (Smith *et al.*, 2010). Yet in addition, increased resource scarcity is often argued to be a key driver of the efficiency improvements that are needed to reduce degradation – a contention most famously discussed by the Danish economist Ester Boserup who described population pressure as a driver of agricultural intensification (Boserup, 1965). Central to concerns over both land degradation and restoration is the fact that competition for land is set to increase through increasing demands for land to provide non-food ecosystem services, many of which are vital for maintaining the sustainability of agricultural systems more generally, including a wide range of regulating and supporting services as well as the conservation of biodiversity (Lataweic *et al.,* 2015). Land scarcity is a key factor influencing patterns of resource use intensification and land degradation but its influence is moderated by access to other resources that can substitute for land (whether technologies, institutions, labour force, or capital).

Agricultural intensification can be a driver of land degradation, but can also contribute to spare land for conservation of natural ecosystems and allow for land restoration. However, interpretations of land sparing are often over-simplified as sparing is only likely to occur under certain conditions (Angelsen & Kaimowitz, 2001; Byerlee *et al.*, 2014; Hertel *et al.*, 2014; Lambin & Meyfroidt, 2011; Rudel *et al.*, 2009a; Villoria *et al.*, 2014). At a local scale, land sparing is only likely to occur when intensification increases the local costs of production – when the availability of capital or labour inputs are limited relative to the capital or labour intensity of the intensification process. Land sparing will also be more likely where strong biophysical or institutional (e.g., territorial land-use policies or supply chain interventions) restrictions on land-use expansion are in place. Finally, local land sparing effects will also be more likely where the demand for the product is relatively inelastic to price changes (e.g., in the case of staple caloric crops or when markets are closed).

Land-use intensification can also result in rebound-effects: the effect by which production processes become more profitable and competitive, and further expansion into natural ecosystems is thus encouraged. Rebound effects are more likely to occur where the intensification process increases input efficiency: it makes agriculture more competitive and there are few physical or institutional restrictions on land-use expansion. Rebound effects are also more likely in situations where intensification has occurred by switching to produce crops that are in high demand, where demand is elastic to price, or when markets are well integrated. Such rebound effects may be avoided, at least locally, if improvements in the efficiency of agricultural production systems are coupled with effective environmental protection measures (Meyfroidt & Lambin, 2011; Phalan *et al.*, 2016).

3.6.4 Distant drivers and globalization

In an increasingly globalized world many of the most powerful indirect drivers of land and resource use in a given region may have their origins on the other side of the planet. Understanding how to identify distant indirect drivers, mitigate their negative impacts and amplify possible benefits is therefore of central importance to tackling the challenges of land degradation and restoration.

Distant indirect drivers are often treated as exogenous variables by both researchers and conservation and development practitioners, and feedbacks are rarely given explicit consideration (Liu *et al.*, 2013). Yet it is becoming increasingly clear that distant drivers and feedback effects can fundamentally alter the distribution, extent and severity of activities that drive both degradation and restoration outcomes.

Taken together the processes of globalization have fundamentally reshaped how human activities drive land degradation across the planet, generating an increasingly complex and far-reaching web of indirect effects that challenge conventional wisdom. Globalization processes can both amplify and attenuate the direct drivers of land degradation and restoration by breaking down regional barriers and strengthening global connections and influences, such as trade tariffs and restrictions, global prices, legal conventions and access to information, over local factors, such as regional markets, extension services and local governance regimes (Lambin *et al.*, 2001; Liu *et al.*, 2013).

There is now a broad consensus in the sustainability research community that factors related to economic development, and in particular international trade and urban demand, are the dominant drivers of unsustainable levels of resource extraction and land degradation worldwide (Geist & Lambin, 2002; Meyfroidt *et al.,* 2013; Nesheim *et al.,* 2014). Indeed, following the rapid growth of international trade there are few, if any, regions of the world whose environments and natural resources are not susceptible in some way to the behaviour of humans on distant corners of the planet. Lifestyle changes and rising consumption patterns of high-income and emerging economies, including in particular shifts towards diets rich in meat and dairy products drive land degradation in regions that are often unseen by local consumers (Cassidy *et al.*, 2013; Kastner *et al.*, 2012; Kissinger *et al.*, 2012; Yu *et al.*, 2013). In particular, the export of agricultural and forest-based commodities, led by the burgeoning demands of a rapidly changing and increasingly globalized economy, and exacerbated by the propensity of weak institutions and environmental governance in many producer nations, has emerged as a key driver of deforestation and forest degradation, especially in the tropics (Kissinger *et al.,* 2012a).

The consequences of globalization and global market integration for the drivers of land degradation and restoration are profound. First, economic growth of a given country is increasingly hard to decouple from land degradation. At the global level, economic analysis of the resource dependencies of different countries that takes account of indirect consumption of material goods has demonstrated that a country's use of nondomestic natural resources is, on average, about threefold larger than the physical quantity of traded goods, and that around 40% of the global material resource extraction and use was linked to internationally traded goods and services (Wiedmann *et al.,* 2015). As countries become wealthier, pressure on the natural environment does not relent but is exported to the countries from which raw materials are originally sourced. Linked to this, there is also strong evidence to suggest that in a world increasingly interconnected economically and ecologically, the costs imposed by environmental and land degradation are felt disproportionately by low-income nations which are being increasingly depended upon as the producers of raw materials for more developed nations (Srinivasan *et al.,* 2008). One manifestation of how globalization has disproportionately impacted developing countries has been through large-scale land acquisitions or "land grabbing" to provide agricultural products for export. Such acquisitions have been shown, in many cases to have profoundly negative impacts on the livelihoods of the rural poor especially smallholder farmers, who lose access to land and water resources due to insecure land rights, unequal and un-transparent contract negotiations, and poor governance and legislation (Cotula *et al.,* 2009; Anseeuw *et al.,* 2011; Ortiz *et al.,* 2013).

A second consequence of globalization is that the drivers of land degradation and restoration have become increasingly unpredictable. For example, trade-bans on beef exports by the European Union following the outbreak of foot and mouth disease had a marked effect on rates of change in agricultural expansion and deforestation in Brazil (Nepstad *et al.,* 2006; Hargrave & Kis-Katos, 2011). Similarly, the collapse of the Soviet Union led to a marked strengthening of beef exports from Brazil to Russia, now one of the primary importers of Brazilian beef, following a collapse in domestic production in the early 1990s and a rebound of domestic consumption since late 1990s (Schierhorn *et al.,* 2016).

Another example of an even less predictable economic signal to have emerged from an increasingly interconnected global commodities market is the fluctuation of exchange rates between currencies of trading nations. For example, Richards *et al.* (2012) present evidence to suggest that the devaluation of the dollar and appreciation of the Brazilian real in the late 1990s and early 2000s, due to macroeconomic changes, counteracted a rise in global soybean prices, and in the process, spared an estimated 40,000 km^2 of new cropland in the Amazon region alone by reducing the incentives to expand soybean production for exports.

Third, the increasing levels of market integration and cross-scale interdependencies in how land and natural resources are managed also presents a fundamental challenge in containing the unwanted feedback effects of otherwise much needed interventions to alleviate poverty and enhance the conservation of native ecosystems. A good example of this can be seen from the possible outcome of a future African Green Revolution, driven by a combination of development imperatives and looming global land scarcity (Lambin & Meyfroidt, 2011). Under a scenario of globally segmented crop markets, an African Green Revolution would be both land and emissions sparing, yet when accounting for rapidly accelerating market integration, rising global demand for agricultural commodities means that such a revolution is likely to drive ongoing vegetation clearance and land degradation (Hertel *et al.,* 2014).

Fourth, the rising importance of international trade in land-based commodities has dramatically raised the profile of private sector actors and market processes over state orientated governance processes in shaping degradation and restoration outcomes (Rudel, 2007) (see also Chapter 2, Section 2.2.3). This restructuring is particularly visible in deforestation frontiers across the tropics, where well capitalized farmers and loggers producing for export markets and domestic consumers in rapidly expanding cities have weakened the historically strong relationship between local population growth and forest cover (Rudel *et al.,* 2009b), and in many places undermined the potential for urbanization to drive a forest transition (Defries *et al.,* 2010).

3.6.5 Scaling up efforts to halt, reverse and mitigate indirect drivers of land degradation

Two important conclusions can be drawn in trying to understand how to turn back land degradation and rehabilitate degraded land. First, no single global set of indicators for assessing land degradation status for a given biome or land-use system will be able to reveal the complexity of human-environment relationships that determine patterns of resource use (see Chapter 8). Second, no universal policy for halting, mitigating or turning back a specific direct degradation driver can be conceived of for a given biome of land-use system.

It is perhaps unsurprising that the majority of programs and efforts to address land degradation have focussed overwhelmingly on direct drivers (e.g., Weatherley-Singh & Gupta, 2015). Whilst there are no panaceas in the search for policy solutions to environmental challenges (Ostrom *et al.,* 2007) improvements are being made in developing more lasting and large-scale solutions that are anchored in recognition of the effects of globalization (Perrings, 2007). Indeed, an increasing number of policy debates are focusing on demand (versus supply) side drivers, and acknowledging the role of distant impacts of consumption in developed countries, including for example the European Union´s FLEGT license scheme and the USA´s Lacey Act for legal timber, and the EUs Renewable Energy Road Map and the US Renewable Fuel Standard (Meyfroidt *et al.,* 2013). In one high profile example, some 190 companies, governments and civil society organizations have signed up to the New York Declaration on Forests that commits signatories to end natural forest loss by 2030, and reduce deforestation by 50% by 2020 (Climate Focus, 2016). At national and regional scales, the last few years have also seen a marked shift in the emergence of multi-sectoral and hybrid governance arrangements, with coalitions of public and private actors having access to an increasingly rich toolbox of regulatory and voluntary measures to improve the sustainability of natural resource governance (Lambin *et al.,* 2014). Responses to both the direct and indirect drivers of land degradation and restoration are explored in more depth in Chapter 6.

3.7 Priorities for research on the drivers of land degradation and restoration

The complex and multifaceted nature of the drivers of land degradation and restoration present major challenges to the research community. In particular, there is an urgent need to advance understanding on why and how specific driver and driver combinations result in observed outcomes and most importantly, why and how interventions do and don't work in different circumstances.

3.7.1 Priorities for research on the direct drivers of land degradation and restoration

Whilst our understanding of the direct drivers of land degradation and restoration is generally better than our understanding of indirect drivers, major knowledge gaps remain.

A general challenge facing researchers working on land degradation and restoration is the lack of consistent definitions and reporting of degradation and restoration indicators at large scales and over time (e.g., for deforestation Malhi *et al.*, 2014), as well as the use of different approaches to amalgamate data (Keenan *et al.,* 2015b). Particular challenges still exist, however, in trying to reconcile estimates derived from remote sensing techniques with estimates derived from ground-based inventories, government statistics and expert opinion. Analogous challenges exist in assessing temporal trends in degradation indicators when definitions and data processing methods change over time (Keenan *et al.,* 2015b). Part of the challenge also lies in the fact that estimates of uncertainty are rarely provided for maps of global land use and land cover (Keenan *et al.*, 2015), and improvements are needed in how to estimate and represent uncertainties in spatial data. Fortunately, the availability of spatially extensive, higher resolution remote sensing products is becoming more readily available and more robust statistical approaches are being incorporated to evaluate uncertainty in spatial data. Data availability in addition to recently agreed on global land degradation indicators of land cover, land productivity, and carbon stocks to achieve land degradation neutrality are a positive move to have globally consistent measures reported over time (Orr *et al.,* 2017).

Thus, the development of commonly agreed indicators and approaches to mapping land degradation at different spatial scales, together with a concerted effort by a research community engaged in transdisciplinary research that also engages non-scientific stakeholders to develop such datasets, would constitute a major advance in our understanding of priority areas for action, as well as the underlying drivers of observed patterns. Moreover, while many remotely-sensed or modelled datasets exist for key indicators, such as patterns of deforestation or the density of livestock in grazing lands, data on specific management practices such as levels of rotational grazing in pastures, levels of fertilizer and pesticide application in croplands, and harvesting intensities of timber and non-timber resources are much sparser and inconsistent. Yet these data are critical for linking observations of land degradation and restoration to differences in actual land-management practices.

Aside from the work needed in data compilation and assessments of the status and trends of drivers of land degradation and restoration another critically important research frontier is the need to better understand interactive effects, including path-dependencies and feedback effects amongst multiple drivers, including myriad interactions between direct anthropogenic drivers and climate change. The combined effect of multiple degradation drivers carries the risk of regime shifts and transitions in how land is used and managed that, once occurred, can be difficult to reverse – as demonstrated, for example, in the combination of land-use intensification, climate change, invasive species and fire in driving large expanses of economically important grazing lands in North America into annual grass monocultures (see Section 3.7 and 3.8 and Chapter 4).

Finally, research on the effectiveness of different strategies to land restoration and rehabilitation, including the underlying economic and policy levers necessary to implement such strategies, is still very much in its infancy. Systematic appraisals and meta-analyses focused on the cost-effectiveness of different restoration strategies are urgent research priorities, as are the development of commonly accepted conceptual and analytical frameworks for measuring success – where success includes interventions across the entire mitigation hierarchy, from avoided loss to the mitigation of impacts to the recovery and rehabilitation of degraded land.

3.7.2. Priorities for research on the indirect drivers of land degradation and restoration

Sustainable restoration of degraded lands requires addressing the underlying drivers of land degradation. Establishing the causal role of a driver, however, needs to rely on two dimensions: causal effects and causal mechanisms (George & Bennett, 2005; Elster, 2007; Mahoney, 2008; Meyfroidt, 2016). A causal effect essentially amounts to the change in an outcome brought about by the change in some factor. The causal mechanism is an explanation of how the cause or combination of causes produced its effects.

Reductionist approaches to understanding indirect drivers can only make a very limited contribution to understanding the causal mechanisms. Yet, without the proper understanding of the causal mechanisms, our ability to predict under which conditions this causal effect may hold is very limited. A good example of this is in the widespread failure of land-use change models to accurately predict patterns of deforestation and broader environmental degradation. Indeed, none of the model projections available in the literature plausibly captured the overall trajectory of land use and cover change that has been observed in the Amazon over the last decade (Dalla-Nora *et al.,* 2014), a failure that is reflected in land-use change models of tropical forest regions globally (Rosa *et al.,* 2014). The main reason for why such modelling approaches are limited in their ability to make accurate predictions lies in the complexity of the underlying anthropogenic drivers that ultimately determine how land is used (Geist & Lambin, 2002; Lambin *et al.,* 2003).

Despite these challenges there are very promising avenues of investigation opening up that can help deepen understanding of actual causal pathways and processes. Many approaches for establishing causal effects build on statistical modelling approaches (e.g., Rubin causal model) which rely on a potential outcome or 'counterfactual' (Holland, 1986; Ferraro & Hanauer, 2014). In this approach, the causal effect of a given factor (called the ‘treatment’) is the difference between the value of a given outcome variable when a unit of observation is affected by this treatment and the value of that outcome when the unit is affected by an alternative control treatment. A credible counterfactual is a unit which is as similar as possible to the treated unit in ways that could influence the likelihood of being treated as well as the outcome. In experimental, laboratory sciences, the exact same experiment can be replicated with and without the treatment, providing a true counterfactual. In sciences observing the “real” world outside laboratories, building credible counterfactuals is much more challenging (Ferraro, 2009). One simple approach is to use “natural experiments” such as observing a region that is relatively homogeneous in biophysical, ecological and social characteristics but separated by an administrative border with a different policy implemented on both sides. Quasi-experimental approaches rely on more sophisticated statistical tools such as matching or synthetic control methods, or even the increasingly popular Randomized Control Trial (RCT) approach, all of which are widely used in the “impact evaluation” or “program evaluation” literature (e.g., Ferraro & Pressey, 2015; Baylis *et al.*, 2016). For example, many works have examined the effectiveness of protected areas (PAs) (Andam *et al.,* 2008). Protected areas are known to be prone to a selection bias, i.e., to be located preferentially in low suitability and low accessibility areas. These are precisely the same characteristics that are also thought to affect the likelihood of deforestation. Thus, an unbiased evaluation of the impact of protected areas on deforestation through statistical matching requires, first, selecting a random sample of observation in protected areas, and then selecting a set of counterfactuals, i.e., control observations that have similar suitability and accessibility characteristics, but are outside of protected areas. One can then compare deforestation in both the protected and the unprotected sample. Other approaches include time-series analyses or space-for-time substitution (see Chapter 4), or meta-analyses of local case studies. Other complementary approaches can be used to trace and analyse causal mechanisms and causal chains (Meyfroidt, 2016). An integrated understanding of the role of indirect drivers then requires mixed method approaches that blend statistical analyses with scenario modelling, place-based empirical studies and qualitative research (e.g., Meyfroidt *et al.*, 2013) and the inclusion of more targeted data on social, political and governance variables in land-use modelling work that have traditionally been dominated by natural scientists (e.g., McNeill *et al.*, 2014).

3.8 References

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Chapter 4

Status and trends of land degradation and restoration and associated changes in biodiversity and ecosystem functions

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Executive Summary

There is wide consensus that land degradation is a global phenomenon resulting in a substantial loss of both biodiversity and ecosystem services (*well established*). However, the global extent, severity and trends in degradation remain inconclusive. The negative impact of degradation on ecosystem services has been well established in numerous local studies. Often quoted figures suggest that four-fifths of agricultural land suffers from severe erosion, as do 10-20% of rangelands {4.1.6, 4.2.6.2}. These numbers are, however, inconclusive, mostly dated and hard to verify {4.1.6}. Many global studies focus on single, narrowly-focused indicators and do not account for the multiple forms of degradation, all of which reduce biodiversity and ecosystem services. In the case of wetlands, an estimated 75% have been lost (*established but incomplete*) {4.2.5.2}. The extent and rate of forest loss is well established, but condition changes within forests are poorly resolved {4.3.1, 4.3.4}.

Degradation is occurring in all land-cover, land-use and landscape types and in all countries (*well established*). This results in a loss of biodiversity {4.2.9} and ecosystem services through: the loss of forests {4.3.4}, rangelands {4.3.2} and wetlands {4.2.5.2}; increased erosion {4.1.1} resulting in reduced net primary production {4.2.3} and crop yields {4.3.3}; increases in destructive wildfires {4.3.6}, sometimes exacerbated by invasive alien plants {4.3.7}; increases in outbreaks of pests and diseases causing losses to natural and crop fauna and flora {4.2.7}; changes in forage quality {4.2.6.2}; and the loss of regulating services such as carbon sequestration {4.2.3} and hydrological function {4.2.5}.

Degradation takes place through a number of biophysical processes and can manifest itself in a wide variety of ways (*well established*). A single direct driver of degradation may affect a multitude of degradation processes, often through a cascading set of interactions {4.1.2}. For instance, removal of vegetation through overgrazing may exacerbate soil erosion, losses of soil organisms and soil organic matter. In combination, these impacts change soil fertility, water infiltration and the water-holding capacity of the soil. The combined effect leads to reduced net primary production, loss of biodiversity and reduced resilience of the landscape when environmental changes occur. Some impacts, such as soil erosion, are a consequence of many direct drivers, whilst others may be driver-specific, so there can be many-to-one and one-to-many links between biophysical drivers, degradation processes and final impacts on ecosystem services {4.2.1}.

Whilst many degradation processes are location specific and a direct consequence of local land management (*well established*), there is an increasing realization that many degradation impacts are a consequence of global processes and drivers. Removing or mitigating local direct drivers of degradation can be achieved through changing land management practices on a specific parcel of land {4.2.1, 4.2.2}. However, many degradation processes such as climate change {4.2.4} or pollution {4.2.8} are regional or global in nature, and occur as a consequence of off-site impacts, over which the land manager has no control. In these cases, since the on-site restoration cannot change the direct cause of the degradation, the only option is often to either mitigate or reverse the impacts of distant drivers. In general, while interventions are available to restore land, taking action before the land is degraded is more efficient {4.1.2}.

At a regional or global scale, distinguishing the impacts of climate change and variability from anthropogenic degradation remains problematic (*unresolved*). There are strong interactions between climate variability and human-induced degradation. Experience in the Sahel {4.2.6.2} suggests that observed trends, which may appear to manifest themselves as “desertification” (dryland degradation), are actually the result of medium-term variability in climate. The climate impacts interact with, and exacerbate, local degradation, as a consequence of inappropriate land management {4.1.3}. There is an urgent need to find monitoring methods that can reliably and repeatedly distinguish impacts of climate variability from anthropogenic degradation {4.1.3}.

Land degradation takes place in both natural vegetation and on previously transformed land, so choice of an appropriate baseline against which to assess change is important (*unresolved*). Land transformation can, in itself, be considered as a form of degradation. This is especially relevant when considering impacts on landscape-level processes, including biodiversity loss {4.2.6.5, 4.2.9}. Transformed land may enhance the provision of specific ecosystem services (such as agricultural crops) at the cost of biodiversity and other ecosystem services (including many regulatory services). Degradation can take place in both natural and transformed land, such as crop fields {4.1}. Furthermore, sustainable land management practices can be applied in both natural land and transformed land to ensure the sustainable provision of ecosystem services. Choosing the baseline against which degradation is measured is therefore critical {4.1.4}. Natural baselines may be meaningful when, for instance, biodiversity impacts are being considered. However, recent baselines such as the present, 10 or 20 years in the past may be far more relevant when considering zero net land degradation targets, assessing the impact of policy interventions or devising sustainable land management interventions. Restoration and mitigation of degradation without changes in current land use is likely to be more common than attempts to restore landscapes to their natural state.

Changes in soil and soil functions occur in almost all forms of degradation with profound but slow impacts on crop production *(well established*). The soil plays a critical role in supporting plant growth and net primary productivity through the provision of water and nutrients. These functions require maintenance of soil physical structure, a wide range of soil organisms and the prevention of pollution that can result from applications of chemicals. Accelerated soil erosion {4.2.1}, by water or wind, is one of the most obvious forms of land degradation. Erosion can be localized, in gullies, or affect large areas such as in the U.S. Dust Bowl. Soil erosion occurs on all non-frozen landscapes, on all continents and in all countries. Loss of plant cover is the single biggest direct cause of erosion. Enhanced erosion is a feature of almost all croplands {4.2.1, 4.3.3}. Generally, erosion is insidious, unrecognizable on an annual basis, but can lead to a total collapse of the cropping and rangeland systems over decades; thus, long-term monitoring is needed. A number of additional factors can alter the biological and hydrological function of soils. Soil acidification – due to the over-application of fertilizers and atmospheric pollutants – is affecting soils in North America, Central and Northern Europe and Southern China {4.2.2.1}. An estimated 76 million ha of mostly irrigated land has been lost to salinization {4.2.2.2}, often in association with further losses to water logging {4.2.2.3}.

Soils are the single biggest store of terrestrial carbon. The loss of soil organic carbon (SOC) has negative impacts on soil biodiversity and soil water and nutrient holding capacity (*well established*). An estimated 55 Pg C has been lost from soil organic carbon predominantly from croplands since 1800s (*established but incomplete*) {4.2.3}. Croplands can lose 50% or more of the soil organic carbon compared to natural habitats, and many forms of land degradation have negative impacts on soil organic carbon. It is estimated that 0.4-0.8 Pg C y^{-1} could be sequestrated due to improved carbon management in crop fields {4.2.3.1}. Although peatlands account for only an estimated 3% of the terrestrial land surface, they are the single biggest store of soil organic carbon. Excluding the vast and relatively intact peatlands of Russia and Canada, the remaining world's peatlands are badly degraded {4.2.3.3}.

Rangeland degradation, due to a multitude of factors, is occurring (with some exceptions) on all continents with rangelands (*established but incomplete*). Extensive loss of groundcover and often dramatic erosion are the classic depiction of degradation, especially when compared to a natural baseline {4.2.6.2}. More contemporary changes to rangelands include a multitude of other degradation processes, such as invasion by alien plant species {4.3.7}, changes in species composition to less palatable species and increases in woody plant density {4.2.6.2}. These changes are often less easily detected, especially in global monitoring products {4.1.3}, but manifest themselves in reduced livestock carrying capacity, with up to ten-fold reduction being reported {4.2.6.2, 4.3.3.2}. Nevertheless, greening, which is attributed to increasing precipitation and atmospheric CO_2, has been observed in some rangelands {4.2.3.1}.

Erosion and the leaching of agricultural chemicals due to poor land management has profound off-site impacts on wetland, river systems, coastal waters and groundwater (*well established*). Intensive agriculture has resulted in widespread eutrophication of rivers, lakes, dams and wetland systems – with hypoxic areas in waterways and at the mouths of major catchments having profound impacts on coastal fisheries resources. This is largely driven by the overuse of fertilizers and is also a consequence of industrial livestock production systems {4.2.4, 4.3.2.1}.

Wildfire is a natural occurrence in many habitats, but humans change fire frequency and seasonal timing, as well as causing fires to enter ecosystems where they naturally do not occur (*well established*). Human activities such as the drainage of peatlands {4.2.5.2}, the introduction of alien species {4.3.7} and thinning of forests {4.3.5} can allow fires to enter and permanently transform habitats {4.2.6.3, 4.3.6}. Either too frequent or infrequent fires can interfere with plant life-histories and disrupt reproduction, again changing the vegetation structure. From a human perspective, some of the most damaging fires occur due to fire suppression, which results in unnatural fuel build-ups. In the coming decades, it is likely that fire in many regions of the world will increase as a result of greater human occupation of natural ecosystems and the effects of climate changes {4.2.6.3, 4.3.6}.

Growing urbanization, infrastructure and industrial use of land is directly reducing available agricultural land, but has a far wider footprint in terms of the emission of pollutants and the urban demand for water, food, fibre and other natural resources (*well established*). Despite the spatial footprint of urban areas being less than 1% of the global land area, they house approximately half of the world's population. In addition to their local impacts, urban centres have off-site impacts including: increases in pollution of the atmosphere, land surface and waterways; increases of surface temperature; changes in the water cycle; and changes in species composition and biodiversity {4.3.10}.

Biodiversity loss – as a consequence of land transformation – is reasonably well understood. However, impacts on biodiversity from other forms of degradation are poorly resolved, especially at regional and global scales (*unresolved*). By 2005, land use and related pressures had reduced species richness by about 15% compared with what they would have been in the absence of human impacts. These losses are enough to alter ecosystem functioning substantially. However, few accurate measurements of species numbers exist for many groups of organisms, owing to difficulties in detection. Hence, many global estimates are based on a few, easily-observed groups such as higher plants and large animals that are unlikely to be representative of actual numbers, although they do allow for processes to be tested. Losses occur not only at the species level, but also in genetic diversity of individual species – a particular concern for the resources available for future breeding of crop species. The distribution of declines is not geographically uniform and losses are greater in some land-cover and land-use types than in others: mines, industrial areas, urban areas, croplands and improved pastures have the greatest decreases compared with primary ecosystems and secondary growth. The main causes of biodiversity loss are habitat loss and fragmentation, and the overexploitation of species by humans, pollution, climate change, invasive species and disease. The biodiversity of ecosystems undergoing recovery has been found to average half the natural levels {4.2.6.3, 4.2.7}. Though poorly researched, loss of soil biodiversity has profound impacts on the soil's ability to support ecosystem services {4.2.6.4}.

There is growing concern over the impacts that climate change may have on degradation (*inconclusive*). Temperature increases and precipitation changes, as well as increased CO_2 concentrations, probably have already had effects and can be expected to have widespread effects on biodiversity, net primary production and fire regimes in the future. The two-way interactions between climate change and degradation is particularly important since land degradation is a major emitter of CO_2, whilst restoration can play a significant role in increasing sequestration of CO_2 {4.2.8, 4.2.3}.

Degradation can have differing impacts on ecosystem services and in some cases, enhance some contributions at the expense of others. Productivity may even increase despite many ecosystem services being lost through degradation (*well established*). There are a number of situations where land is considered degraded – since the ecosystem services the land-user requires decrease – despite other aspects, such as net primary production remaining constant or even increasing. In rangelands {4.3.2}, invasions by alien species, increases in unpalatable plants and increases in density of woody plants may all result in increased net primary production, but with decreases in grazing potential {4.2.6.2}. Impacts from deliberately and accidentally introduced alien species have substantive impacts on natural biodiversity, ecosystem function and the flow of ecosystem services {4.3.7}. Converting forest or rangeland to cropland can result in huge increases in food, but at the cost of biodiversity and regulating services. This has important implications for degradation mapping and monitoring since different techniques and indicators are required for different forms of degradation {4.1.2}.

There is an urgent need for the development of appropriate degradation and restoration indicators and strengthening of existing measurement and monitoring programmes (*well established*). National, regional and global land degradation and restoration monitoring networks should be strengthened or established where absent. These are essential to determine the locations, extent and severity of degradation as a prelude to restoration and prevention. On-the-ground monitoring needs to complement remote sensing techniques and, in both cases, appropriate indicators need to be refined or established. Many existing indicators are flawed or not useful. Underlying ecological processes also need further investigation, particularly those subject to non-linear transitions and thresholds beyond which degradation cannot be reversed with the resources that are realistically available. The conditions in which permanent degradation occurs (and its frequency) are critical since their ecosystem services are also lost.

4.1 Introduction

4.1.1 Aims

Humans have historically modified their environment, directly and indirectly, to meet their requirements (August *et al.*, 2002; Forman, 1995; Turner *et al.*, 1994; Vitousek *et al.*, 1997). The resulting anthropogenic impacts on land have been so profound that a new geologic era has been recognized, the Anthropocene (Ellis *et al.*, 2010; Ellis & Ramankutty, 2008; Steffen *et al.*, 2015, 2016; Waters *et al.*, 2016) – generally dated from 1950 (Waters *et al.*, 2016). The concept of "planetary boundaries" has emerged to attempt to forestall irreversible, adverse impacts on the Earth (Steffen *et al.*, 2015). However, in order to avoid or mitigate the adverse effects of land degradation, there is a clear need to assess the extent, causes and processes of degradation affecting humans in the past, present and into the future. However, there has been, and continues to be, confusion over the meaning of the term "degradation". Many believe they can recognize it when they see it (in the field or with satellite imagery), yet the confusion in the literature belies this view. The IPBES definition of land degradation (see Chapter 1 and Glossary) states it clearly, but it is the implications of such a necessarily brief definition that often give rise to confusion.

There is a distinction between, on the one hand, the human causes, motivations and consequences of land degradation and, on the other, the biophysically imposed constraints. This relationship was first noted by Carl Saur and has long been recognized in geography under the title "possibilism" (Robbins, 2012). The term "biophysical" is used here to distinguish the human from the ecological perspectives, although humans are inextricably associated with the ecological, as other chapters in this assessment point out (see Chapters 1, 2 and 5). It is important to recognize that environmental processes alone can result in conditions that take the form of anthropogenic degradation (such as natural hillslope erosion), but are not anthropogenic drivers of "degradation", unless the natural process is initiated or exacerbated by humans (such as erosion following removal of vegetation). This chapter focuses on the latter.

Degradation results from a multitude of drivers (see Chapter 3) and can be manifested in many forms (see Section 4.2.), such as erosion, loss of fertility, reduced carbon stocks, and changes in hydrological regimes. It can be driven by changes in land cover caused by, for example, pollution, pests and diseases spreading as a result of climate change and through biodiversity loss. The multitude of drivers has differing impacts on different environmental systems and the drivers from Chapter 3 are mapped to impacts in Section 4.3. "Degradation" is not a single phenomenon – the term is too general. A wide range of disciplines and measurements are often involved (e.g., Symeonakis & Drake, 2004; Zucca & Biancalani, 2011). Nevertheless, the exact biophysical processes and degradation outcomes are, in many cases, insufficiently known. This presages one of the key findings of the chapter that is the dearth of data – hence, the critical need for new techniques and routine monitoring programmes.

The objective of this chapter is to assess the status and trends of the biophysical aspects of degradation to provide connecting links between: the identification and motivations of the human drivers of degradation (Chapter 3) (Millennium Ecosystem Assessment, 2005); the current status and trends of the biophysical processes on ecosystem services (this Chapter); the resultant livelihood and well-being implications (Chapter 5); and the effectiveness of existing interventions and responses to mitigate and prevent degradation or restore land (Chapter 6). This Chapter gives an overall introduction to the degradation process, detecting degradation, designation of baselines and history. In Section 4.2, the status and possible future trends of degradation processes are described. Section 4.3 takes a different perspective, which is to assess the effects of specific human activities, such as excessive livestock production, agriculture, forestry, alien species introductions, abandonment of land, mining and urbanization.

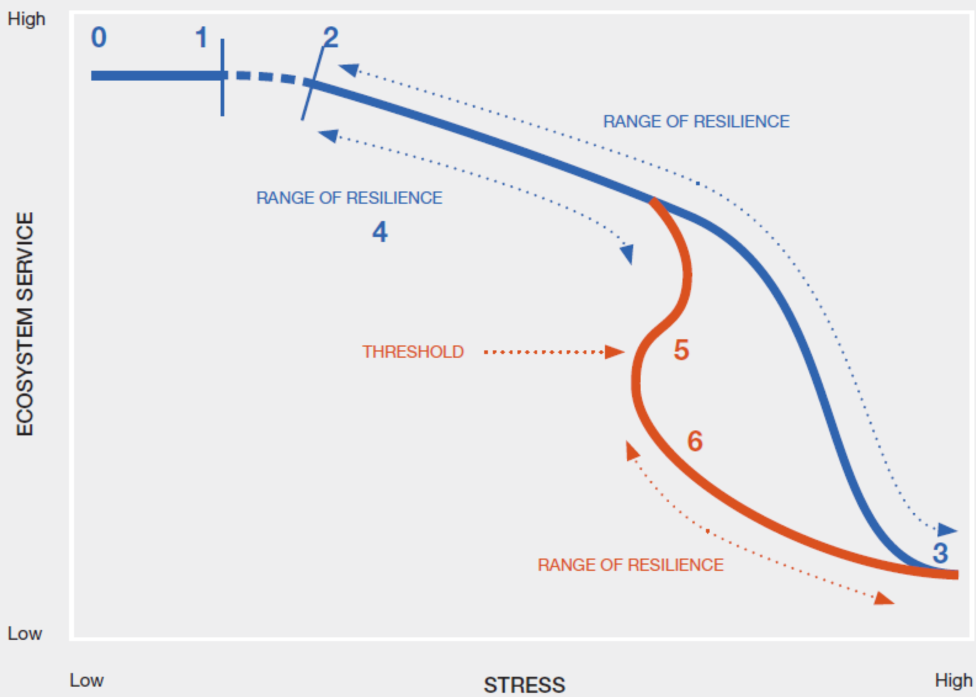
4.1.2. The degradation process

As noted above, there has been and continues to be confusion over the meaning of the term “degradation”. Many believe they can recognize it when they see it (in the field or with satellite imagery), yet the confusion in the literature belies this view. The definition of the term has led to interminable reviews (see review by Vogt *et al.*, 2011) and even the more detailed versions often give rise to confusion (Prince, 2016a, 2018).

The analogy of a cusp threshold (Figure 4.1) illustrates some of the different types of degradation. The effects of stress caused by human activities to which organisms are susceptible, and therefore the ecosystem service they provide (e.g., depleted soil nitrogen and crop production), can be envisaged as a “response curve”. This is shown by the blue curve from 1 to 2 to 3 in Figure 1. The ecosystem service responds rapidly, almost linearly to the particular stress involved (from point 2 to 3), until the stress declines (e.g., nitrogen is added in the crop example). As the stress declines from right to left in Figure 1, further increases in the service (e.g., crop yield) decrease (from 1 to 0), often reaching a plateau when additional reductions of the specific stress have no further effect (at 0). Fluctuations in the stress cause the ecosystem service to move up and down the curve in its range of resilience (2 to 3). On the other hand, there are conditions in which stress drives down the provision of the service, as illustrated by curve 5 to 6, until it reaches a threshold (point 5) (Turnbull *et al.*, 2008) at which the ecosystem service drops dramatically. This is an example of a non-linear ecological process. Most importantly the ecosystem service cannot be recovered no matter how much the stress is relieved. In this level of degradation, shown as the lower part of the red curve, the ecosystem reaches its completely degraded condition (point 3): this is the permanently degraded condition described in Vogt *et al.* (2011).

Figure 4 1 **Two types of response to stress.**

In curve 2 to 3 (blue) the degree of anthropogenic stress determines the level of ecosystem service over the full range, until point 3 when the stress is so high that it has no further effect. The second curve (5 to 6) reaches a threshold (5) at which the response to stress is non-linear and the ecosystem changes to a new state that cannot return to the upper level, no matter how much the stress is alleviated. Illustration based on Lockwood & Lockwood (1993). Source: Prince (2018).

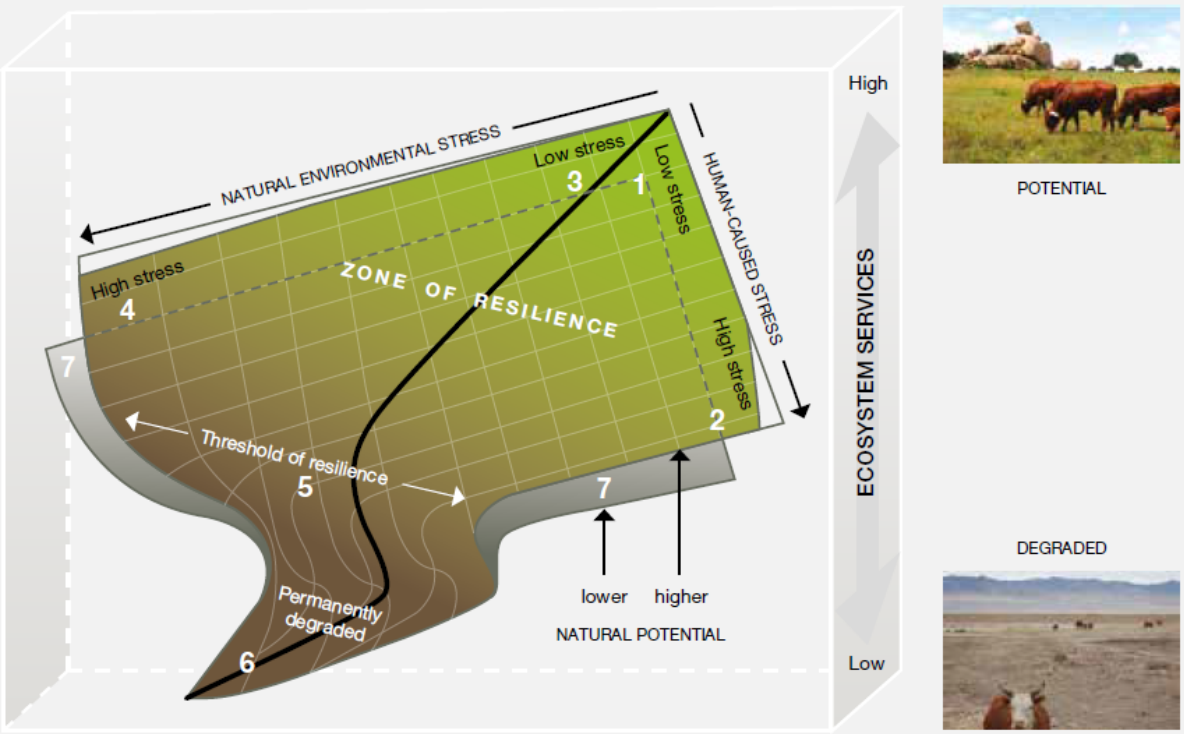


The analogy of response curves is helpful only when one anthropogenic stress is involved, but normally there are many that affect ecosystem services, such as soil type, pollution, soil compaction, loss of palatable species for livestock, and reduced productivity – all in one location. These stresses can be divided into two classes. The first is those that are caused by the physical environment with no human involvement, and the second, those that are brought about by human action alone (anthropogenic stresses). These two classes of stress frequently occur together and interact.

Figure 4.2. illustrates the additional complexity when both biophysical and anthropogenic stresses occur together. While a service may be resilient to the full range of anthropogenic stresses when there is negligible environmental stress, a moderate environmental stress moves the anthropogenic response curve closer to the threshold. A further increase in environmental stress drives the site over the cusp and into the zone of permanent degradation, from which no return is possible without drastic, expensive and lengthy artificial remediation. Typically, neither anthropogenic nor environmental stresses alone drive the site into the permanently degraded zone, but when they work together catastrophic loss of services can ensue.

Figure 4 2 Conceptual representation of the states and process of degradation and the potential contributions of anthropogenic (human-caused) and natural environmental stresses.

The ecosystem service(s) is represented by the vertical dimension and the ecosystem dynamics by movement over the surface. The higher up on the surface in the vertical dimension, the higher the ecosystem service. The top two edges represent stress from the natural environmental (left) and anthropogenic stress (right). Both stresses increase across the surface (from 1 to 2 and from 3 to 4). The fold or cusp in the surface (5) represents the threshold of a zone of permanent degradation. Sites that move over the threshold of resilience on any trajectory cannot return to the upper zone of resilience. A second surface shown below (7) represents a site that naturally provides lower environmental services, but is not initially degraded: it has all the features of the upper surface including resilience and the possibility of permanent degradation (see Section 4.1.2). Note that no trend, or no trend after environmental normalization (Bai *et al.*, 2008; Rishmawi *et al.*, 2016), could indicate land that has been degraded in the past (zone 6) or no degradation has occurred and the environment is stable (zones 1 and 3). Source: Prince (2016, 2018).



These concepts lead to recognition of six types of "degradation" shown in Table 4.1 (Prince, 2016a, 2018). Types i and iii are actually not degraded, but are often mistaken for it. Recognition of this distinction can be difficult, but it is critical when assessing the status and planning for restoration – the initial failure to recognize these two states and their difference from true degradation has caused much confusion, for example understanding of Sahelian "desertification" (see Chapter 1 and Section 4.2.6.2) (Herrmann & Sop, 2016). A lot of "degradation" mapping is actually about measuring differences in the potential of the ecosystem to provide services, not degradation of that potential (Vagen *et al.*, 2005). Similarly Type ii may have existed for a long time and might be assumed to not be degraded, but it could belong to Type vi (i.e., permanently degraded). Types v and vi are the only states that are correctly termed "degradation" (Adeel *et al.*, 2005; Spinoni *et al.*, 2014), since their condition is effectively irreversible, even when the driver of the stress is removed. The degradation below the threshold is generally not static, but also moves according to its resilience as the stress varies (Type v) (Wessels *et al.*, 2007), but never back over the threshold. Completely static degradation (Type vi) does occur, for example in heavily salinized croplands. Type iv is of greatest interest since, if the stress is alleviated, it has the capacity to recover naturally – although recovery may be accelerated by human intervention; the alternative being unremitting, further degradation to Type v or vi.

Recovery from Types v and vi is actually possible, but only with significant efforts and expenses, or over exceptionally long-time periods, generally exceeding a human life-span. Moreover, the value of the restored land rarely merits the cost of restoration or recovery. For example, the 20 million ha of the southern Great Plains of the USA that were lost to the "Dust Bowl" in the early 1930s (Baveye *et al.*, 2011; Hurt, 1986) were restored at the cost of approximately $17 billion (in 2017 dollars) and the creation of an entirely new government agency (now called the Natural Resource Conservation Service) which, by 2017, employed 12,000 people in 2,900 offices countrywide. Nevertheless, land in Type vi remains low in ecosystem services and is susceptible to renewed degradation (Romm, 2011).

Table 4.1. The six degradation states (Prince, 2016a, 2018)

Six states	Comments	Citations
(i) Appearance of degradation	• Land with low resource availability in its natural state often appears superficially similar to degraded land.	Castro *et al.* (1980); Safriel (2009); Vagen *et al.* (2005)
(ii) Degraded in the past	• Assumed to be in natural state, but actually degraded. • Lack of baseline (see Section 4.1.4.) prevents correct interpretation.	Gritzner (1981)
(iii) Susceptible to degradation	• Susceptible land owing to its natural properties and its environment, but not actually degraded.	Beinroth *et al.* (2001); UNEP (1997)
(iv) Land recovers when stressors removed	• Land apparently degraded, but within its range of resilience. • When stressors removed (e.g., drought, overstocking), the land returns to its initial, non-degraded condition.	Olsson *et al.* (2005); Tucker *et al.* (1991)
(v) Temporal trend of increase in degradation	• The degradation persists when stressors (e.g., drought, overstocking) are removed – and there is a temporal trend of increasing degradation.	Wessels *et al.* (2007)
(vi) Stable, degraded state	• Degraded land in static condition that changes little when stressors (e.g., drought, overstocking) are removed, but never recovers to the condition above the cusp.	Milton *et al.* (1994a); UNCCD (2017)

Remediation and restoration techniques (see Chapter 6) are frequently applied to control degradation. However, the recovery of the original, pre-degradation ecosystem is at best extremely slow. In cases where there are data, disturbance remained detectable over long periods. For example, some experimental sites in the USA shortgrass steppe, that still showed the degradation caused by grazing and burning 80 years earlier (Peters *et al.*, 2008), with no evidence of complete recovery. Many such cases have been recognised, a common one being soil compaction by heavy vehicles (Webb, 2002). Thus, degradation can be permanent, on century-long scales. In the ecological literature, this state is referred to as a deflected succession, a subclimax, or plagioclimax.

4.1.3 Detection of degradation

4.1.3.1 Types of data used for mapping large areas

In the past and into the present there has been a failure to agree on what ecosystem conditions should be regarded as degraded, hampering any consensus on location, severity and extent. For example, in forested areas, there is extensive mapping of transformation to other land cover types, but less recognition of the extent of degradation within untransformed forest.

Developing indicators and monitoring them are essential to any understanding of land degradation. In the report "Ecological Indicators for the Nation" the National Research Council (2000) provides criteria for selection of indicators. Anthropogenic land degradation generally consists of multiple conditions and so most monitoring programs use several indicators (Lorenz & Lal, 2016; National Research Council, 2000). The Sustainable Development Goal Target 15.3 has adopted three indices (CBD, 2016), while UNCCD uses 11 (Orr, 2011), WOCAT uses 57 (Liniger *et al.*, 2008), and GLADA uses 132 (Nachtergaele & Licona-Manzur, 2008).

The characteristics of data on land degradation that are appropriate for rigorous analysis and development of policy-relevant conclusions are the same as those that apply to all quantitative data. They have little meaning unless accompanied with explicit information on the methods used, any necessary qualifications and the variance of the reported values. For example, much of the information on the carbon cycle (Section 4.2.3) (Lorenz & Lal, 2016) has confidence limits. Qualitative data (including indigenous and local knowledge) can also have error metrics and can be combined with quantitative data and statistical methods in joint analyses known as "mixed methods" (Creswell, 2007).

Data are collected at a wide range of spatial and temporal scales: from single points or small areas of a few hectares, all the way up to global, and for one point in time to monitoring long-term trends. Methods differ for different scales. Global measurements are almost entirely made using remote sensing since they can have global coverage, spatial resolutions of a few meters and daily, monthly or annual repeat measurements. In the case of remote sensing of vegetation, the remarkable characteristics of vegetation indices (e.g. the Normalized Difference Vegetation Index, NDVI) (Bannari *et al.*, 1995) and their inter-annual trends have frequently been applied to measurement of degradation.

Although net primary production can be estimated globally (Tucker & Pinzon, 2016), it is not, alone, an indicator of degradation without attention to normalizations of weather and other non-anthropogenic factors (Prince *et al.*, 1998; Rishmawi *et al.*, 2016) and especially additional methods that are needed to separate out different types of degradation (see Table 4.1 and Figure 4.2). Global monitoring of above and below ground carbon stock is impractical. A single, large-area map has been developed based on the development of functions for upscaling point data to a full spatial extent using correlated environmental covariates, for which spatial data are available, such as Global Soil Information System (Brus *et al.*, 2017); however, the simple correlation technique's variability is too large to detect the relatively small changes involved in monitoring degradation (Lorenz & Lal, 2016).

While NDVI and related vegetation indices can be used as surrogates for vegetation production (gross primary production), they are only proxies, and can be incorrect in some conditions (Prince, 1991). Other information, such as plant diversity, generally cannot be measured directly, although some interspecific differences can be detected by seasonal phenological changes in the indices. More direct detection of species has been achieved in some cases using many spectral bands with imaging spectrometry (hyperspectral), but the "spectral diversity" often consists of more than one, not single taxonomic species (Gholizadeh *et al.*, 2018; Thenkabail *et al.*, 2012). Satellite data having the necessary multiple, narrow spectral bands do not exist at the time of writing.

Improvements in types of measurements and storage in archives is a high priority. An important aspect of data use, by which degradation can be detected and monitored, is improved access. Archives and data bases (Section 4.4.3.5.) are increasing in number and size, but tend to concentrate on data for large-areas, while more local data remain with those who made the observations. Another difficulty in the use of data is the gap between research products and adoption for routine monitoring. An example is global mapping of the extent of conversion to urban land cover for which a new method exists (Ying *et al.*, 2017), but has not been repeated for monitoring of trends. Researchers rarely have the resources for repetitive, routine monitoring – this can only be executed by designated and appropriately resourced institutions. Furthermore, access generally assumes broad-band, high speed internet which may not be available in less-developed countries, limiting local interpretation and dissemination of local data to the broader community.

Degradation generally extends over long-time scales (ie., "long-term", "permanent"), yet there are frequent attempts to account for the long-term at the scale of factors such as annual stocking rates, whereas soil formation has a time scale of many years. Both processes are relevant to degradation, but in quite distinct ways related to their scale of action (Wiegand *et al.*, 2005, 2006; Wiegand & Milton, 1996). Furthermore, many areas of current degradation, degraded prior to current satellite-based trend data, may appear as stable land in these data sets (Gibbs & Salmon, 2015). The same occurs over space – for example, deposition of wind-blown products of surface erosion can takes place over hundreds of square kilometres, and hundreds of kilometres from the source, yet cattle hoofs that compact the soil are limited to paddocks measuring hectares. The scale of national politics is another range of space and time scales.

4.1.3.2 Multi-metric indices

Since there can be no single metric of all types of degradation (see Section 4.1.1) combinations of a number of different measurements into a single, multi-metric index to summarize ecological conditions and processes have often been proposed (Symeonakis & Drake, 2004; Zucca & Biancalani, 2011). Such multi-metric indices attempt to summarize ecological subjective attributes such as "sustainability", "integrity", ecosystem "health" and others. Examples of such indices include: "Ecological Integrity" (Andreasen *et al.*, 2001); "Ecosystem Health" (Brown & Williams, 2016); "Index of Biotic Integrity" (Karr, 1991); "Living Planet Index" (World Wildlife Fund, 2016); SDG target 15.3 (CBD, 2016) and the many that combine ecological and socio-economic factors (e.g., Environmental Vulnerability Index - Pratt *et al.* (2004)).

Multi-metric indices, however, are not ideal since they can give a false impression of being founded on well-accepted knowledge of ecosystem processes when, in many cases, they are or contain, highly subjective components. In addition, just because an index is numeric does not make it ecologically sound. Specific indices have strengths and weaknesses, but all are subject to certain flaws: they are subject to loss of information in the condensation of multi-dimensional variability into a one-dimensional index (so the condition in need of remediation often cannot be identified from the index alone); they are subject to systematic bias if raw data are converted into categorical scores; they are subject to weighting, as combination of multiple data types, either implicitly or explicitly, weights the measurements of the properties by different amounts, thus emphasizing some aspects more than others (Cai *et al.*, 2011; Kosmas *et al.*, 2012). Weightings can only be justified if the processes are understood well enough to select appropriate ones to which assign greater weight (e.g., McRae *et al.*, 2017). The Sustainable Development Goal Target 15.3 has adopted an index "proportion of land that is degraded over total land area", measuring degradation with a combination of net primary production, land cover and soil organic carbon stock (above and below ground) (United Nations, 2015). It has been shown that these are appropriate metrics for measurement of some types of degradation individually; however, measurement of none of the three is possible above the local scale and the misrepresentation of the potential scale of application in the 2030 Agenda for Sustainable Development (United Nations, 2015) is regrettable, bearing in mind the probable future influence of the SDGs.

4.1.3.3 Data and models

Mechanistic models can simulate degradation and other relevant metrics using mathematical representations of biophysical processes. Many such models exist, appropriate to different aspects of degradation (e.g., Izaurralde *et al.*, 2007; Kirkby *et al.*, 2008; Tamene & Le, 2015). These models are attractive since they are designed to behave according to the same processes that determine the degradation, unlike, for example, mapping of some indicators. Model results can be very accurate when the biophysical processes are known and adequate data are available. However, the more realistic models are, the greater their complexity and their need for data. The demand for data and parameters can be prohibitive, and oftentimes default values have to be used with consequent reduction of accuracy. Rarely do such models have adequate precision to detect subtle local degradation.

4.1.3.4 Syndromes

Syndromes are descriptions of archetypical, dynamic, co-evolutionary patterns of human-environment interactions (Lambin & Geist, 2008). The concept shares some features of models since a set of *a priori* definitions based on socio-economic and biophysical factors are selected and then used to classify types of degradation. They are derived from qualitative studies of the physical and human aspects of selected degradation case studies. Syndromes have been used in relation to degradation and its socioeconomic effects (Ibáñez *et al.*, 2008) and in a predictive model (Sietz *et al.*, 2006). Geist (2005) developed an inventory of syndromes applied to dryland degradation. While attractive as summaries of the nature of specific degradation processes, the selection of types of syndromes is not based on any objective scheme, and the concept has been applied at limited scales (Geist, 2005a; Petchel-Held *et al.*, 1999).

4.1.4 Baselines

Land degradation takes place in both natural vegetation and on land transformed to an altered state and use (such as cropland and plantation forests). Although land transformation can, in itself, be considered as a form of degradation, transformed land may also enhance provisioning of specific ecosystem services, such as agricultural commodities. As such, the choice of an appropriate baseline against which to assess degradation is important. Evaluation of land degradation and restoration requires answers to the questions, "degraded relative to what?" and "progress in restoration towards what?" A reference or baseline is essential to detect and assess the magnitude and direction of any trend in degradation compared with the current conditions (National Research Council, 2000; Prince, 2016a) (see also Chapter 2, Section 2.2.1.1. and Box 2.1).

For example, the concept of "Zero Net Land Degradation" (Chasek *et al.*, 2014) is clearly dependent on baselines for adaptive management and assessment of success. Multiple types of reference states are in use to furnish a start, baseline or reference condition for comparison with the current conditions (Table 4.2). A salutary warning of the danger of a lack of baseline was given by Alexander von Humboldt in 1848, as reported by Gritzner (1981), that travellers unfamiliar with arid lands are "easily led to adopt the erroneous inference that absence of trees is a characteristic of hot climates" where in reality, the area had long been degraded by the enormous caravans that crossed the Sahara. Clearly Humboldt recognized the difference between Types i and vi degradation (Table 4.1) long before modern environmental science rediscovered the distinction.

4.1.4.1 Target condition

Ecosystem services are provided to human beings and have no meaning apart from that. They are a measure of human preference and satisfaction, so a particularly pertinent reference condition would be one that maximizes the desired mix of ecosystem services – namely, a target condition. This is similar to the "utilitarian" concept of the Millennium Ecosystem Assessment (Hassan *et al.*, 2005a). A target condition is based on a deliberate choice and is therefore context-dependent. For example, in the case of long-standing cropland agriculture, sustained and healthy crop production, rather than the natural land cover, is the target. This is perhaps the most important reference for policy purposes, since it represents a desired future state, the achievement of which can be measured and monitored. A target, however, is not static – it is an aim, and aims can change. It also usually not possible to treat a single service alone since any gain in one can cause a loss of another, so trade-offs are normal, and the choices involved can also change. Furthermore, in many regions and ecosystems, this potential is also not static because of ongoing regional and global changes such as climate change and atmospheric nitrogen deposition.

4.1.4.2 Historical baseline

The historical baseline is the condition of a site in the past. The change from the historical condition to the present time measures the trend. This provides an objective assessment, as opposed to the selection of a target condition which is an aspiration (or a natural baseline, see 4.1.4.3. below). A historical trend can indicate undesirable changes in an ecosystem and also point to the processes of degradation that have led to the current state and restoration efforts.

While highly desirable, unfortunately there are few, detailed, time-series of observations of ecosystem properties that are more than 50 years old. Examples include the Park Grass Experiment that started in 1856 (Silvertown *et al.*, 2006) and selected plant communities throughout the Netherlands that started in the 1930s (Smits *et al.*, 2002). Most repetitive measurement programs are recent. Examples include the annual North American Breeding Bird Survey (Sauer *et al.*, 2017); the many UK Biological Records Centre monitoring schemes (Biological Records Centre, 2017); the 43-year Earth-Observing satellite record (Moran *et al.*, 2012); and many "permanent plots" in which earlier surveys are repeated, often more than once (Bakker *et al.*, 1996; Kapfer *et al.*, 2017). Historical baselines have been used extensively for assessment of the status and trends of species and ecosystems (e.g., the IUCN Red List of Threatened Species - IUCN, 2017). However, few of these records are coordinated, and start dates, repetitions and types of measurements generally differ, which makes comparisons difficult. Care must be taken to avoid a false impression of more or less degradation based on different starting dates (Pauly, 1995). Furthermore, sites may have suffered degradation before the historical baseline (e.g. Gritzner, 1981).

4.1.4.3 Natural baseline

In some circumstances, particularly where human influence and degradation are low, such as in isolated areas of boreal forest, remote humid forests and some islands, it may be reasonable to infer the condition before the first human influence on the land cover (Bull *et al.*, 2014). This seems an obvious baseline from which to assess any trends in degradation and recovery, since it was before any human modification (Kotiaho *et al.*, 2016), but practical and theoretical issues weigh against it. No exact date can be given for the first human occupation in the Holocene (≤10,000 BCE) but, for practical purposes, maybe only 200-300 of the past years, or even the start of the Anthropocene (see below) is adequate. Practically, it is rare to find objective data from so far into the past (Spikins, 2000). The only data of this type are fossil deposits, pollen and also fossil parts of plants, insects and diatoms and evidence of human-induced soil erosion that can provide some indications (Hoffmann *et al.*, 2009). These can sometimes be dated or otherwise assigned to the pre-human period, but they are often too generalized to specify the state of the environment in adequate detail for comparison with existing conditions. Of course, a pre-human baseline has no use when the climate or other physical environmental conditions changed in the time between the baseline and the present time, as occurred, for example, in the Little Ice Age just 400 years ago (Matthews & Briffa, 2005).

The start of the Anthropocene (approximately 1950) (Ludwig & Steffen, 2017; Morselli *et al.*, 2018; Waters *et al.*, 2016) can be a logical starting point for a natural baseline – an "Anthropocene baseline" – since it marks, by definition, the start of the massive acceleration of human influence on the natural environment and its biota. Data availability for the last 100 years is obviously greater in number, type and accuracy. While anthropogenic degradation occurred in many places before the beginning of the Anthropocene, it was often negligible compared with the post-1950 period and is therefore a useful starting point to assess anthropogenic degradation.

However, even for an Anthropocene baseline, a significant amount of qualitative judgement is needed. A more objective method is the "space for time" substitution (Johnson & Miyanishi, 2008; Pickett, 1989), which compares similar sites in different locations and treats spatial and temporal variation as equivalent. Although this assumption has been challenged, space-for-time substitution is often used due to necessity or convenience (Pickett, 1989). This is one respect in which the use of current conditions to infer a historical baseline is helpful, since non-anthropogenic, environmental changes, such as weather fluctuations will have affected both the putative non-degraded and degraded sites, thereby eliminating some non-anthropogenic environmental changes before the present time. A more objective method for inferring a former state from the current condition is by mathematical process modelling (McGrath *et al.*, 2015; Spikins, 2000; Wang *et al.*, 2006) but data are often sparse and spatial scales are coarse. Furthermore, there are many potential errors in modelling; for example, the mathematical representation of natural processes may not apply to the entire period between the current state and the original natural state.

Table 4.2 Types of baselines for detection of trends in degradation (Prince, 2016a)**.**

Baseline type	Meaning	Data sources	Data processing	Examples
Natural	Pre-modern (≤10,000 yr. BCE)	Paleontological data. Information on environment event and trends (e.g., paleoclimate).	Expert opinion; Interpretation of fossils	Davis & Shaw (2001); Graumlich (1993)
	Pre-Anthropocene (approximately 1850-1950)	Early descriptions, images, recent archaeology, land use. Information on environment event and trends.	Expert opinion based on residual unaltered sites	Gammage (2011)
Historical	Past ecosystem records. Ecological and agricultural monitoring programmes started in the past. Typically, mid-19th century, 1950s, and early 21st century.	Ecological data. Information on environmental events and trends (e.g., meteorological variables, CO_2, land use)	Analysed with statistical methods, error measurement.	Storkey *et al.* (2015)
			Adequate data to match with key characteristics of; “Current”, “Ecological Integrity” or “Target” definitions	Buma *et al.* (2017)
			Long time-series of records allow more accurate specification of trends	Ridding *et al.* (2015)
			Measurement techniques used must be known and repeated in all subsequent data collections	Root *et al.* (2003)
"Current"	Baselines established recently	Repeatable measurement techniques. Specify land use and date of establishment. Based on observations, not derived indices. Detailed location information. Secure archive publicly accessible.	Statistically rigorous, including frequency distributions, accuracy and error.	Rogers *et al.* (1989)
Ecological integrity	Maintenance of ecological processes (Munyati *et al.*, 2013; Karr, 1996)	User specification of desirable condition. Land use at date of definition. Program of adaptive management. Specify measurement techniques		Silva *et al.* (2010)

Target	The state that is most desirable to the land user ("utilitarian" concept; Hassan *et al.*, 2005)	Land managers, farmers, foresters, biodiversity experts, environmentally-aware public and so on. Quantify targets. Specify land use and date of establishment.		Hobbs *et al.* (2009)

4.1.5 Future trends of degradation

Accurate information on future environmental conditions and human effects on the environment would assist remediation and recovery efforts. Speculation of future trends are often based on hypothetical, but realistic scenarios of future human activities (see Chapter 7) including future land cover, changes in carbon sequestration and pollution. In order to have consistency in forecasts, scenarios that provide some descriptions of how the future might unfold have been developed. Scenarios are defined as "hypothetical sequences of events constructed for the purpose of focusing attention on causal processes and decision points" (Geist, 2005; Kahn & Wiener, 1967). A range of plausible pathways, scenarios, and targets are used to capture a set of conditions for a range of land use, the efficiency of the use of land resources and products, trade and food self-sufficiency, effects of climate change, biodiversity, land use, and so on. These are potential outcomes based on an internally consistent, reproducible, and plausible set of assumptions and theories of key driving forces of change (IPCC, 2000) but they should not be interpreted as accurate forecasts.

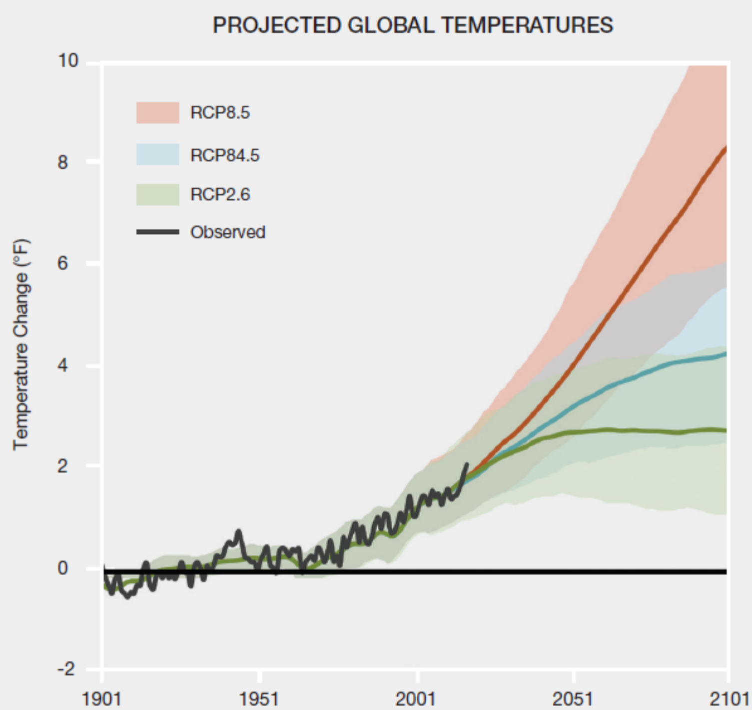
Scenarios of human activities and their effects on climate (Bjørnæs, 2015) use Integrated Assessment Models (IAMs) that estimate the combined effects of human activities (e.g., land use and fossil fuel emissions) on the carbon-climate system. IAMs such as the IMAGE model (Integrated Model to Assess the Global Environment) (Stehfest *et al.*, 2014) have been coupled with climate models (Moorcroft, 2003; Moss *et al.*, 2010) to simulate the potential interactions of human activities and climate (Bos *et al.*, 2015; IPCC, 2000; Meller *et al.*, 2015). These scenarios are called Representative Concentration Pathways (RCPs) (Bjørnæs, 2015) (Table 4.3, Figure 4.3).

Table 4.3 Four Representative Concentration Pathways (RCPs) derived from 4 integrated assessment models (Bjørnæs, 2015).

Scenario and emissions	Human activities	Anticipated results
RCP 8.5 High emissions Sometimes called "business as usual", meaning no changes occur in current factors that affect the future.	No policy changes to reduce emissions. Increasing greenhouse gas emissions that lead to high greenhouse gas concentrations over time.	• Three times today's CO_2 emissions by 2100 • Rapid increase in methane emissions • Increased use of croplands and grassland which is driven by an increase in population • A world population of 12 billion by 2100 • Lower rate of technology development • Heavy reliance on fossil fuels • High energy intensity • No implementation of climate policies

RCP 6 Intermediate emissions	Radiative forcing is stabilized shortly after year 2100.	• Heavy reliance on fossil fuels • Intermediate energy intensity • Increasing use of croplands and declining use of grasslands • Stable methane emissions • CO_2 emissions peak in 2060 at 75% above today's levels, then decline to 25% above today
RCP 4.5 Intermediate emissions	Radiative forcing is stabilised shortly after year 2100,	• Lower energy intensity • Strong reforestation programmes • Decreasing use of croplands and grasslands due to yield increases and dietary changes • Stringent climate policies • Stable methane emissions • CO_2 emissions increase only slightly before decline commences around 2040
RCP 2.6 Low emissions	Radiative forcing reaches 3.1 W m^{-2} before it returns to 2.6 W m^{-2} by 2100.	• Declining use of oil • Low energy intensity • A world population of 9 billion by year 2100 • Use of croplands increase due to bio-energy production • More intensive animal husbandry • Methane emissions reduced by 40% • CO_2 emissions stay at today's level until 2020, then decline and become negative in 2100 • CO_2 concentrations peak around 2050, followed by a modest decline to around 400 ppm by 2100.

Figure 4 3 Changes in global annual mean surface temperature relative to 1901–1960 for three Representative Concentration Pathways (RCPs) (as seen in Table 4.3) and the ranges of confidence based on +20 climate models (Table 4.3). Source: Hayhoe *et al.* (2017).



4.1.6 History of degradation studies

Land degradation predates modern written history. A well-documented example is from 2,400 BC in Mesopotamia, where irrigated agriculture in the Tigris and Euphrates valleys led to salinization (Thomas & Middleton, 1994a). Notwithstanding this long history, modern day attempts to quantify the extent and scale of land degradation have proven difficult, especially at the global scale.

Early global assessments of degradation had a narrow soil focus (e.g., Oldeman *et al.*, 1990). More recent studies have been based on loss of net primary production, often using satellite data (Jackson & Prince, 2016; Noojipady *et al.*, 2015; Prince *et al.*, 2009; Prince, 2016b). Following from the Millennium Ecosystem Assessment (Hassan *et al.*, 2005a), the emphasis has been on declines in the flow of ecosystem services. Assessment methods have ranged from estimation by specialists; detailed analysis of satellite observation products; social assessment of abandoned land; and simulation models (Prince, 2016b; Wessels *et al.*, 2008, 2012).

Comments such as 80% of the global croplands are degraded, or that 10-20% of rangeland are degraded are common and often cited (Table 4.4) (Gibbs & Salmon, 2015a; Safriel & Adeel, 2005), however, progress towards a credible measure of the extent of land degradation remains elusive. The GLASOD "world map" of desertification (Oldeman *et al.*, 1990) has been widely used, but recent reviews (Prince, 2016b; Sonneveld & Dent, 2009) have found it to be seriously flawed and it cannot be accepted as a map of desertification (Sonneveld & Dent, 2009) . Although a number of other attempts have been made at quantifying the global extent of degradation (Table 4.4) (Gibbs & Salmon, 2015), at the global scale, the spatial locations and severity of degradation remain substantially unknown (Prince, 2016b). The 3rd edition of the World Atlas on Desertification (Cherlet *et al.*, 2015) does not attempt to develop a single degradation map, but rather uses a convergence of evidence approach.

Table 4.4 Synthesis of continental and global scale estimates of degradation (ha 106) modified from Gibbs & Salmon (2015) by addition of NRCS values. Note: (i) light degradation was excluded from the estimates here; and (ii) North America includes Mexico and Central America, unless otherwise noted. Table annotations: a - does not include Caribbean; b - includes some Caribbean countries; c - total based on country areas listed in Bai *et al.*, (2008) and does not match global total listed in the same source (3,506 million ha); d - non-tropical continents not included in this study; e – many inconsistencies in Eswaran *et al.* (2001) and Eswaran & Reich (1998), between and within each.

Area	**GLASOD (Oldeman *et al.*, 1990)**	**FAO TerraSTAT (FAO, 2002)**	**Dregne & Chou (1992)**	**GLADA (Bai *et al.*, 2008)**	**Cai *et al.* (2011)**	**Campbell *et al.* (2008)**	**FAO (2001)**	**Eswaran, Lal, & Reich, (2001)**	**Eswaran & Reich, (1998)**
Africa	321	1,222	1,046	660	132	69	9	5,233	
Asia	453	2,501	1,342	912	490	118	12	124,467,900	
Australia and Pacific	6	368	376	236	13	74	d		

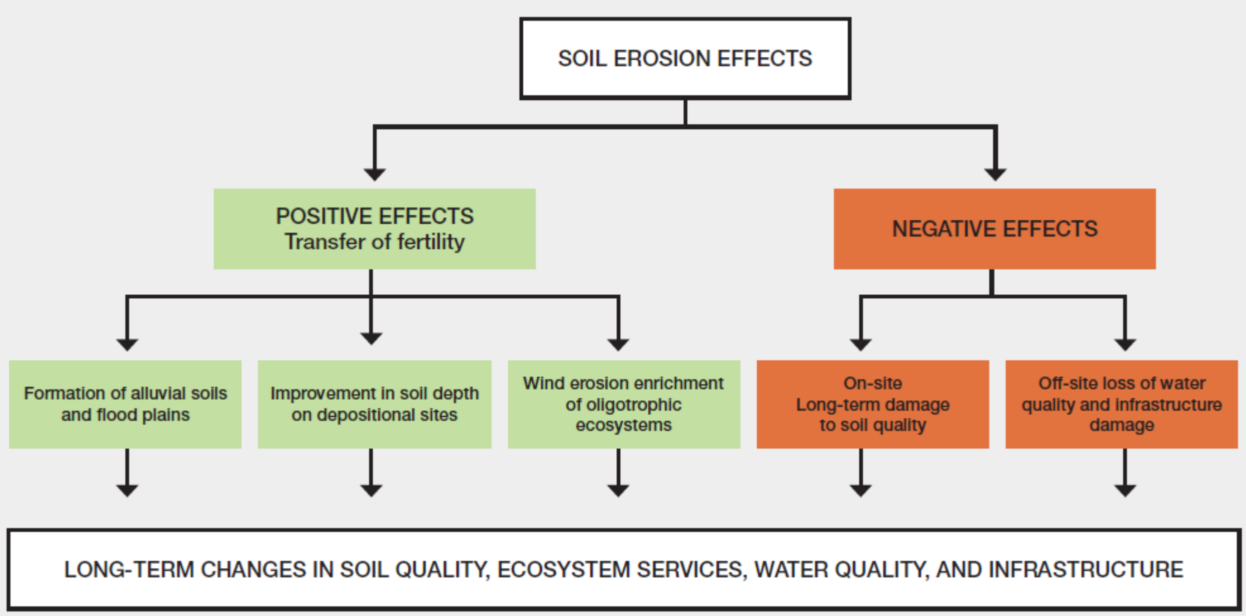
Europe	158	403	94	65	104	60	d		
North America	140	796	429 *a*	469	96	79	d		
South America	139	851	306 *b*	398	156	69	56 *b*		
World (Total)	1,216	6,140	3,592	2,740 *c*	991	470	76 *d*	57,560	15

4.2 Individual degradation processes

4.2.1 Soil erosion

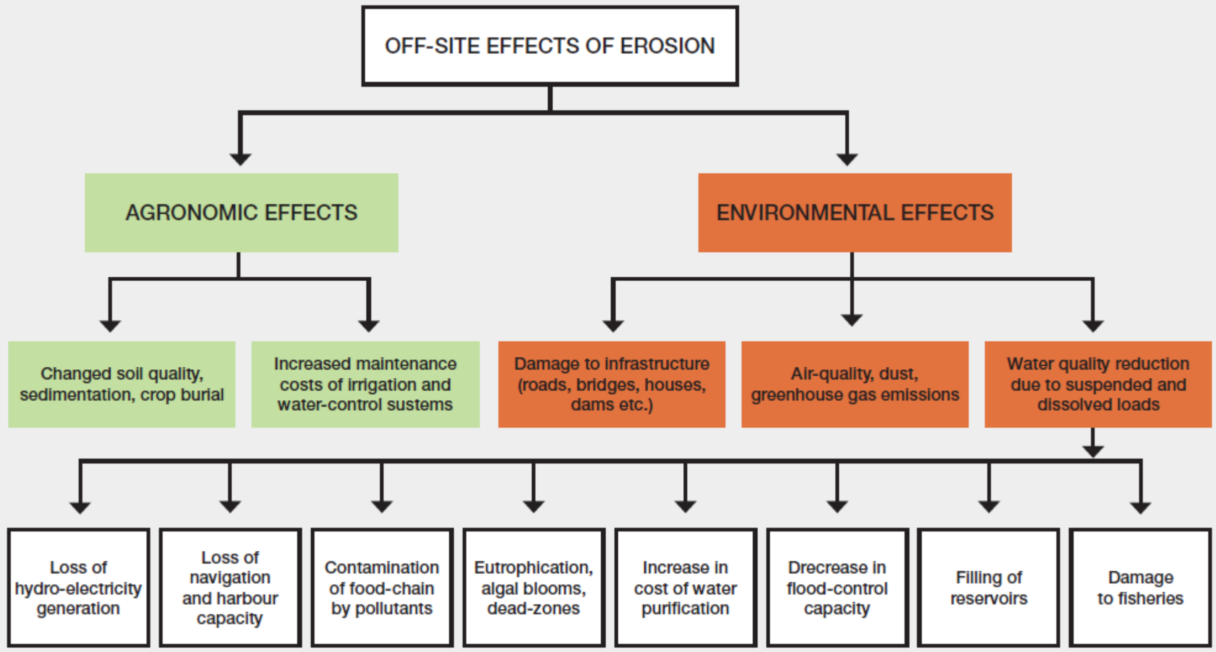
Soil is the basis for provision of many essential ecosystem (Costanza & Daly, 1987; Hassan *et al.*, 2005b) yet the soil resources of the world are finite and non-renewable in the human-time scale (Lal, 1998) and so extensive loss through erosion is a serious problem (Montgomery, 2007b). Nevertheless, the effects of soil erosion can be positive as well as negative (Figure 4.4) and off-site effects may be substantially larger than on-site (Figure 4.5) (Lal, 1998), both on productivity and on environmental quality (den Biggelaar *et al.*, 2003).

Figure 4 4 Positive and negative effects of soil erosion. Source: Modified from Lal (1998).



Erosion is a natural process, but is strongly accelerated by agriculture (Montgomery, 2007b) and mismanagement (Diamond, 2011). Nowadays the combined effects of, for example, the development of industrial cropping and urban sealed areas (see Section 4.3.10) and with an increasing population to feed, have resulted in cultivation of marginal lands, leading to significant soil erosion (Tato, 1992).

Figure 4 5 The major negative effects of soil erosion. Source: Modified from Lal (1998).



If a median value of 0.3% annual crop loss caused by erosion is valid for the period from 2015 to 2050, a total yield reduction owing to erosion of 10.25% could be projected to 2050 (assuming no other changes such as the adoption of additional conservation measures by farmers) (FAO & ITPS, 2015). This loss depends on the crop type and soil management (den Biggelaar *et al.*, 2003) and would be equivalent to the removal of 150 million ha from crop production or 4.5 million ha yr^{-1} (Foley *et al.*, 2011).

There are major regional differences in the status and trends of soil erosion (FAO and ITPS, 2015; F. Nachtergaele *et al.*, 2010). Parts of Europe, North America and the Southwest Pacific generally have recent improving trends. Sub-Saharan Africa has variable trends, whereas Asia, Latin America and the Caribbean, the Near East and North Africa have particularly negative trends. Three climatic zones where erosion rates can be particularly high are Mediterranean, monsoonal and semiarid areas (Walling & Kleo, 1979). There are erosion hotspots (Table 4.5) (Lal, 1998) but the estimates of intensities have low confidence (Boardman, 2006) because of its large temporal and spatial variation, a paucity of accurate measurements and the problem of extrapolating data from small plots to larger areas (García-Ruiz *et al.*, 2015; Stroosnijder, 2005).

Table 4.5 Global hotspots of soil erosion of natural and anthropogenic causes. Erosion rate values have been estimated. Adapted from Lal, (1998).

1. Developing countries (Asia, Africa, Latin America with 0.03 to 0.05% of yield loss/T of soil loss) more than Western Europe and North America (0.01-0.02%) (den Biggelaar *et al.*, 2003).
2. Chinese Loess plateau, the Yangtze basin and the southern hilly country. The Yellow river has by far the highest sediment load of any large rivers in the world.
3. Some mountainous areas such the Himalaya belt and the Andes, especially the central drier part with widespread badlands, stripped bedrock and sand dunes. However, the balance of natural vs. anthropogenically driven erosion is unclear.
4. South and East Asia. Moderate to extreme water erosion reported from India (10% of area), the Philippines (38%), Pakistan (12.5%), Thailand (15%) and Vietnam (10%).
5. The Mediterranean basin, Ethiopia, Lesotho, Madagascar.
6. Mountainous islands such as in the Caribbean (Haiti). Erosion mainly related to deforestation and subsequent cultivation (e.g., Haiti) or grazing (e.g., Iceland).
7. The Bodélé Depression in Chad is the largest source of dust in the world, but the erosion is natural, not caused by human activities.

Three types of soil erosion occur (Table 4.6): water, wind, and mass transportation. Water or hydric erosion is caused by running water and includes the detachment of particles by splash, transport (concentrated runoff) and deposition. Wind erosion (deflation or aeolian) occurs in areas having <250mm annual rainfall (Shao, 2008). Dust emissions from wind erosion can reach high levels in the atmosphere and impact climate (Choobari *et al.*, 2014), air quality and human health far away from the source. Mass transportation by gravity is a natural process on slopes that can be initiated and exacerbated by animals and humans who break the surface vegetation, off-road vehicles and by agricultural tillage (Van Oost *et al.*, 2000). This includes landslides which often occur on steep slopes denuded by humans, often near habitations where the results can cost large numbers of human lives (Figure 4.6). Extreme rain events can render areas vulnerable to floods, landslides, gully incisions and soil erosion by water, depending on geology, relative relief and climate (Figure 4.6) (Clarke & Rendell, 2007; Luino, 2005; Ravi *et al.*, 2010).

Each of these types is strongly affected by human activities (Morgan, 2005) and environmental conditions, including soil texture, soil structure stability – strongly affected by the amount of organic matter in the soil (4.2.3.1), surface protection by vegetation, soil crusting, stones, also slope and landscape structure (4.2.6.5).

Figure 4 6 Landslide at the Philippine village of Guinsaugon in 2006 in which half of the 2500 residents died. Photo: courtesy of Lance Cpl. Raymond D. Petersen III, USA Marine Corps.

Table 4 6 **Effects of soil erosion. Gullies, pipes, rills and stoniness are indicators of strong erosion. The risk of erosion is high if two or more indicators are present. Source: Stocking & Murnaghan (2000); Vigiak *et al.* (2005)**

VISUAL EROSION INDICATOR	WATER EROSION	WIND EROSION	MASS TRANSPORTATION
Rills	✓		
Gully, pipes	✓		
Pedestal	✓		
Armour layer, stone pavements	✓	✓	✓
Accumulation of soil around clumps of vegetation, upslope of trees, fences and barriers	✓	✓	✓
Deposit of soil in gentle slope	✓		
Exposed roots	✓		✓
Exposed stones	✓		✓
Muddy waters during/shortly after storm	✓		
Sedimentation in streams and reservoirs	✓		✓
Dust storms and clouds		✓	
Sandy layer at soil surface		✓	
Parallel furrows in clay soils or ripples in sandy soil		✓	
Bare and barren spots	✓	✓	✓
Nutrient deficiency, toxicity symptoms evident on plants	✓		
Decreasing yields	✓	✓	✓
Poor seed germination	✓		✓
Seeds washing	✓		✓
Change in vegetation species	✓	✓	✓
Restricting rooting depth	✓	✓	✓
Decrease in organic matter (lighter soil colour)	✓	✓	✓

In general, land use and land cover are the major factor in soil erosion rates (Figure 4.5) (Lal, 1998; Montgomery, 2007b). Erosion rates have been found to increase in the order: below natural forest and shrubland < planted trees < perennial plantations < annual crops < bare soils, with over 5mm yr^{-1} in extreme cases. Below trees and shrubs, erosion is complicated owing to interception by the canopy which can create a pattern of more and less erosion, while in pastures, the point-to-point variation is less (García-Ruiz *et al.*, 2015). Heavy siltation has raised river beds, increasing the risk of flooding, especially in the Yangtze river basin in China, the major river basins of humid tropics in East Asia and the Amazon Basin (Aylward, 2005;

Bruijnzeel, 2004a, 2005; Yin & Li, 2001). "Conservation" agriculture, contour line ploughing, no tillage or sowing directly into a cover crop and mulching bare surfaces can decrease soil erosion by over 80% (Montgomery, 2007). With these techniques, soil erosion on cropland in the USA declined nearly 40% between 1982 and 1997, from 3.1 to 1.9 Pg yr^{-1} even while the area of cropland remained roughly constant (FAO & ITPS, 2015; Wiebe, 2003).

4.2.2 Loss of soil fertility

4.2.2.1 Soil acidification

Occurrence

Acidic soils are found on every continent (Figure 4.7). Particularly low pH soils occur in South Eastern Asia, eastern North America, along the west coast and south-central regions of Africa, Northern Europe, portions of Siberia and the Amazon Basin of South America. These regions are vulnerable to further acidification by human disturbances. Particularly severe effects have been reported in Southern China (Guo *et al.*, 2010) due to nitrogen fertilizer application (500-4,000 kg N ha^{-1} yr^{-1}) resulting in acidification of 20-221kmol (H^+) ha^{-1} yr^{-1}), and double cropping practices which can exacerbate cation removal (15-20 kmol H^+). Acid sulphate soils are prevalent in coastal regions, particularly Australia (58,000 km^2).

Sources of acidification

Soil acidification is a natural process that occurs in regions with an abundance of precipitation and leaching, leading to accelerated weathering of soil minerals, release of base cations such as calcium, magnesium, sodium and potassium, which are removed from soil with drainage waters (van Breemen *et al.*, 1983). Sandy soils with low quantities of organic matter are most susceptible. Soils with naturally low quantities of weatherable minerals or minerals resistant to weathering are also commonly acidic. In addition to loss of base cations, inputs of strong acids can lead to mobilization of dissolved inorganic aluminium, which is toxic to plants and aquatic biota. Soils enriched in amorphous iron or aluminium oxides from acidification readily immobilize phosphorus, affecting plant availability.

Waterlogging or other mechanisms resulting in reducing conditions in soils, sediments and organic substrates can produce iron sulphide minerals, forming acid sulphate soils. If acid sulphate soils are drained, excavated or exposed to air, the iron sulphide minerals oxidize, resulting in the production of sulfuric acid and extremely acidic conditions. Acid sulphate soils are common in coastal areas, but also occur in agricultural areas with saline, sulphate-rich groundwater and in freshwater wetlands.

Biotic effects

Soil acidification can affect the supply and availability of inorganic nutrients (calcium, magnesium, phosphorus), affecting fertility and the nutritional needs of grazing animals. Soil acidification coupled with the leaching of strong acid anions (sulphate, nitrate, chloride) results in the mobilization of dissolved inorganic aluminium from soil (Cronan & Schofield, 1990), which is toxic to plants due to inhibition of root growth and function, and runoff with elevated aluminium concentrations, which is toxic to fish and aquatic invertebrates (Driscoll *et al.*, 2001; Pardo *et al.*, 2011).

Human causes

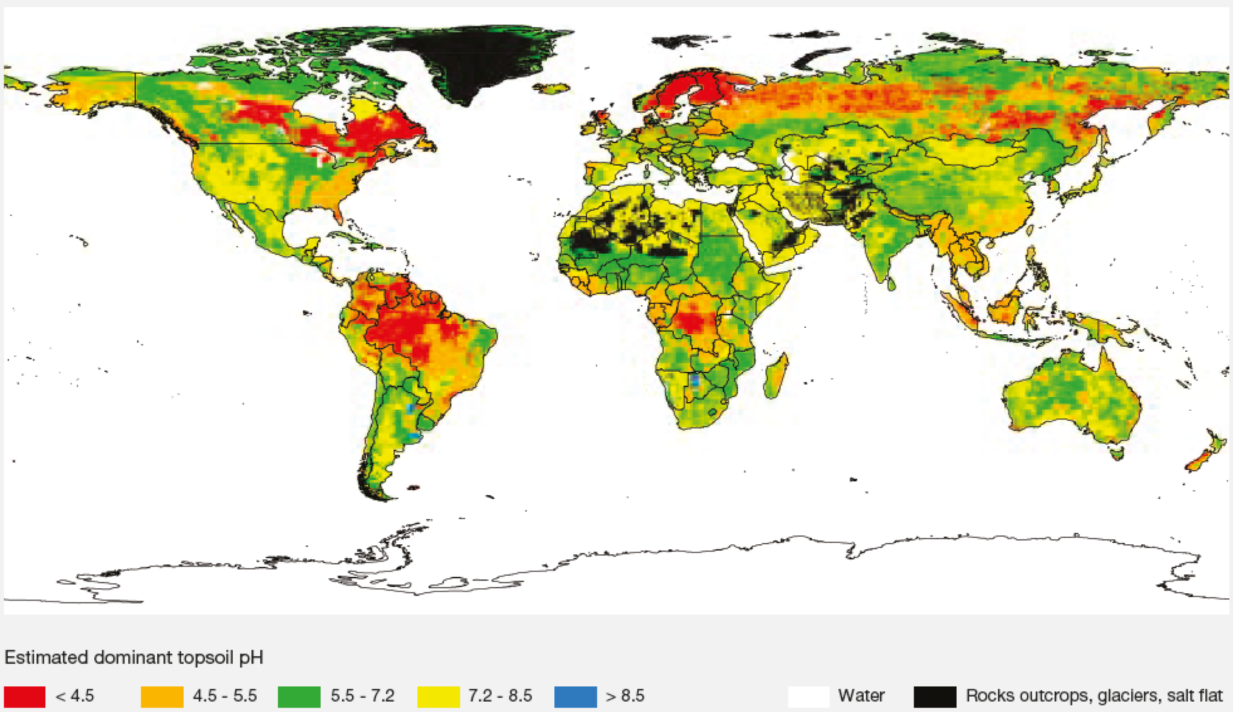
Human activities can exacerbate acidification that occurs with natural soil development. The common causes are: wet and dry deposition of acidic atmospheric pollutants ("acid rain") emitted from fossil fuel combustion; excessive application of ammonium-based fertilizers and intensive agricultural cropping; deforestation and tree harvesting; and exposure of drained acid sulphate soils.

In forests, particularly those on base poor uplands, chronic acid deposition (Driscoll *et al.*, 2001) and repeated harvesting with removal of nutrients in the biomass, especially under short rotation, can severely acidify soils (Likens *et al.*, 1998). For a few years after harvesting, elevated nitrate leaching can occur which itself reduces fertility and accelerates the depletion of exchangeable nutrient cations from the soil exchange complex (van Breemen *et al.*, 1984). Cation accumulation associated with re-growing forest biomass continues soil acidification.

Intensive agriculture with large application of nitrogen fertilizers can result in soil acidification through plant uptake of ammonium and/or ammonium oxidation and nitrate leaching (Guo *et al.*, 2010; van Breemen *et al.*, 1983). In tandem, as in forestry, the removal of crops and other biomass can exacerbate soil acidification due to the removal of nutrient cations (calcium, magnesium, potassium) (Tang & Rengel, 2003).

Figure 4 7 **Global map of pH of topsoil.**

Values < 7.0 are acidic, but only soils below 5.5 are generally unsuitable for most crops and below 4.5 are severely acidic.
Source: Based on FAO/IIASA/ISRIC-CAS/JRC (2009).



4.2.2.2 Soil salinization and alkalinisation

High concentrations of soluble salts limit the ability of plant roots to absorb soil water, decreasing plant growth and crop yields. There are three categories of salt-affected soils: saline, sodic and saline-sodic soils (Table 4.7).

Occurrence

The global areal extent of all salt-affected soils, most of which are naturally salty, is about 1 billion ha, occurring in about 100 countries (Table 4.7). Irrigated land damaged by salinization is estimated globally to be 60 million ha: in India (20 million ha), China (7 million ha), the USA (5.2 million ha) and Pakistan (3.2 million ha), also in Afghanistan, Egypt, Iraq, Kazakhstan, Turkmenistan, Mexico, Syria and Turkey (Squires & Glenn, 2011). Although there is little quantitative information, it is thought that the areal extent of naturally occurring and human induced salt affected soils are increasing due to climate change and increased use of irrigation for crop production (FAO & ITPS, 2015).

Although salinity occurs naturally, it is often exacerbated by human activities, most commonly through irrigation at rates that are not adequate to exceed evapotranspiration, so there is inadequate movement of water below the rooting zone to leach salt from the soil. Other common causes for salt-affected soils are: poor drainage or groundwater near the soil surface (< 2m) (India, Pakistan, China, Kenya, USA); use of brackish water for irrigation (Asia, Europe, Africa); intrusion of seawater near coastal areas; and shifts from deep rooted perennial vegetation to shallow rooted annual crops and pastures (southern Australia) (FAO & ITPS, 2015).

Types

Saline soils have excessive levels of soluble salts (calcium, magnesium, sodium, chloride, sulphate) and are characterized by high specific conductance values > 4 dS m^{-1} (Table 4.7). Owing to the high osmotic potential of saline soil water, plants have difficulty absorbing water, leading to drought-like conditions even though the soils are moist.

Table 4.7 salinity-sodicity classifications and criteria used by the USDA Natural Resource Conservation Service: ESP: exchangeable sodium percentage; ECse: saturated extract electrical conductance (Allison *et al.*, 1954).

Class	ESP%	ECse (dS m^{-1})	Soil pH
Nonsaline, nonsodic	< 15	< 4	< 8.4
Saline	< 15	> 4	< 8.4
Sodic	> 15	< 4	> 8.4
Saline-sodic	> 15	> 4	< 8.4

Sodic soils have high levels of sodium adsorbed on cation exchange sites (> 15%) (Table 4.7). When a large fraction of negatively charged surfaces of clay particles are occupied by sodium, they disperse (deflocculate) from the larger soil aggregates forming sodium-clays. Dispersed sodium-clays clog the soil pores, decreasing permeability to water (low hydraulic conductivity). Sodic soils are difficult to till, have reduced infiltration and drainage and are characterized by poor seed germination and restricted root growth. Furthermore, the loss of aggregates and cohesion of soil particles makes sodic soils susceptible to wind and water erosion of the soil above the impervious layer.

Saline-sodic soils have both elevated salinity and sodicity (Table 4.8). Note, saline and saline-sodic soils are characterized by higher concentrations of divalent cations (calcium, magnesium) that promote flocculation of clays, thereby reducing their tendency to disperse and resulting in better drainage than in sodic soils.

Table 4.8 Areal extent of saline and sodic soils in different regions (UNEP, 1992).

Continent	Saline soils (10^6 ha)	Sodic soils (10^6 ha)	Total (10^6 ha)
Africa	122.9	86.7	209.6
South Asia	82.3	1.8	84.1
North and Central Asia	91.5	120.2	211.7
Southeast Asia	20.0	-	20.0
South America	69.5	59.8	129.3
North America	6.2	9.6	15.8
Mexico/Central America	2.0	-	2.0
Australasia	17.6	340.0	357.6
World total	412.0	618.0	1030

4.2.2.3 Waterlogging

Waterlogging is a chronic problem in all continents, particularly in irrigated cropland causing impairment of plant productivity. While its prevalence is difficult to assess since it is usually quite localized, it can be expected to increase in relation to increases in irrigation. Degradation results from excessive input of water and/or inadequate drainage, so the water table rises towards the soil surface, leading to: depletion of soil oxygen and carbon dioxide accumulation; chemical conversion of non-toxic chemicals into their reduced form which can be toxic (e.g., sulphate reduced to sulphide); denitrification and emission of nitrous oxide (N_2O) – a major greenhouse gas; and reduction of nitrogen fixation by the nodules of legume crops and pastures, all leading to anoxic conditions. Waterlogging is frequently accompanied by salinization.

There are several drivers of large-scale waterlogging in non-wetland soils. Irrigation is probably the main contributor, due to excessive application of water and/or poor drainage due to impermeable clay layers or topography. Urbanization changes the hydrologic cycle by increasing impervious surfaces and the removal of vegetation (see Section 4.3.10). This land use has lower infiltration and evapotranspiration, increasing surface runoff and flooding. Deforestation can cause waterlogging due to decreases in evapotranspiration and increases in soil water content. Waterlogging would be exacerbated by increased precipitation, which is projected to occur under climate change (Melillo *et al.*, 2014). Remediation is normally by prevention – reduced soil water through drainage or, more locally, raised planting beds.

Human transformations of land ecosystems since the start of the Anthropocene (see Section 4.1.4.3) have contributed a net amount of about 180 $\pm$ 80 PgC to the atmosphere (Ciais *et al.*, 2013). Depending on the calculation method used, the annual carbon emission from land-use change has either been fairly constant at about 1.2 PgC yr^{-1} since 1960; or has decreased from about 1.5 PgC yr^{-1} in the 1960s to about 1 PgC yr^{-1} between 2005 and 2016 (Le Quéré *et al.*, 2016). The main processes include the loss and degradation of forests; the drying and burning of peatlands; and the decline in carbon content in cultivated soils and rangelands as a result of excessive disturbance and insufficient return of organic matter to the soil.

Despite the ongoing loss of tropical forest cover and reduced extent of other natural ecosystems, roughly a quarter of anthropogenic CO_2 emissions are sequestered annually by the terrestrial ecosystems which remain untransformed, including some recovering from former degradation. The net annual change in the terrestrial carbon stock has increased from near zero in the mid-1880s to around 4 PgC yr^{-1} between 2005 and 2016 (Le Quéré *et al.*, 2016). The magnitude of this 'land carbon sink' may be up to 1 PgC yr^{-1} larger than these estimates once harvest, grazing and tillage have been fully accounted for (Pugh *et al.*, 2015). In warm, dry years, associated especially with El Nino climate events, the global land sink weakens sharply, to the point where the land may become a small net source of carbon to the atmosphere (Ciais *et al.*, 2013).

4.2.3 Changes in carbon stocks following degradation and restoration

Human transformations of land ecosystems since the start of the Anthropocene (see Section 4.1.4.3) have contributed a net amount of about 180 $\pm$ 80 PgC to the atmosphere (Ciais *et al.*, 2013). Depending on the calculation method used, the annual carbon emission from land-use change has either been fairly constant at about 1.2 PgC yr^{-1} since 1960; or has decreased from about 1.5 PgC yr^{-1} in the 1960s to about 1 PgC yr^{-1} between 2005 and 2016 (Le Quéré *et al.*, 2016). The main processes include the loss and degradation of forests; the drying and burning of peatlands; and the decline in carbon content in cultivated soils and rangelands as a result of excessive disturbance and insufficient return of organic matter to the soil. Despite the ongoing loss of tropical forest cover and reduced extent of other natural ecosystems, roughly a quarter of anthropogenic CO_2 emissions are sequestered annually by the terrestrial ecosystems which remain untransformed, including some recovering from former degradation. The net annual change in the terrestrial carbon stock has increased from near zero in the mid-1880s to around 4 PgC yr^{-1} between 2005 and 2016 (Le Quéré *et al.*, 2016). The magnitude of this 'land carbon sink' may be up to 1 PgC yr^{-1} larger than these estimates once harvest, grazing and tillage have been fully accounted for (Pugh *et al.*, 2015). In warm, dry years, associated especially with El Nino climate events, the global land sink weakens sharply, to the point where the land may become a small net source of carbon to the atmosphere (Ciais *et al.*, 2013)).

Changes in the global terrestrial biomass and soil carbon stock have been proposed as indicators of human impact on the land, since they integrate the many underlying processes affecting productivity, respiration and disturbance (CBD, 2016; Orr, 2011).

4.2.3.1 Loss and recovery of soil carbon

Soil organic matter is a complex array of (a) fast cycling living microorganisms; (b) plant, animal and microbial debris slowly undergoing decomposition; and (c) recalcitrant organic carbon compounds collectively known as "humus". Soil organic carbon (SOC) makes up about 58% of soil organic matter and contains many other life-essential elements, some of which (nitrogen, phosphorus, and sulphur) cycle in close coupling with carbon. Because of the close relationship between soil organic carbon and soil organic matter, the terms are used interchangeably, with the former preferred for carbon balance calculations and the latter for understanding the effects of organic matter on soil properties and processes such as bulk density, water holding capacity, pH buffering capacity, biological activity, nutrient cycling and soil structure. Soil organic matter changes under degradation and restoration in both quantity and form, because of changes in the balance between carbon inputs (plant litter, manure) and outputs (product exports, mineralization, and erosion).

After the carbon held in oceans, soil organic carbon is the second largest carbon pool in the biosphere. Scharlemann *et al.* (2014) reviewed 27 global estimates of soil organic carbon stocks, which ranged from 504 to 3,000 PgC. One widely-cited estimate (Batjes, 1996) is that SOC stocks in the top 1 m soil depth amount to 1,505 ± 61 Pg, with a further 722 ± 38 Pg of inorganic carbon. Soil inorganic carbon – common in arid lands as calcrete – is less responsive than soil organic carbon to human-induced change. More recent estimates of global SOC are about 2,300 PgC in mineral soils, with a further 600 PgC in peatlands and 1700 PgC in permafrost (Field & Raupach, 2004; Lorenz, 2013; Prentice, 2001). In terrestrial ecosystems, more carbon is typically held as SOC than as biomass, although the fraction varies widely, from more than 60% in forests to more than 80% in grasslands. Tundra, permafrost deposits and peatlands (see Section 4.2.3.3) have almost all of their carbon stock in the soil. Owing to the centrality of SOC to fertility, low values and negative trends have been proposed as an index of degradation (National Research Council, 2000; Orr, 2011). The majority of the area currently under agricultural use (~1.5 billion hectares) originated from the conversion of forests and grasslands to agricultural use via deforestation, burning, and cultivation. Generally, these historical changes in land use lead to reductions in soil organic carbon stocks. SOC loss from land conversion and unsustainable land management practices over the past two centuries has been estimated, using very approximate methods, to be 8% (176 PgC) of the assumed pre-modern global SOC stock (Van der Esch *et al.*, 2017). Globally, SOC losses of 55 PgC have been estimated to have occurred since the 1800s because of cultivation (Cole *et al.*, 1997). In temperate environments, topsoil SOC losses of 25-50% have been reported after 30-70 years of cultivation (Ellert & Gregorich, 1996; Mann, 1986; Mikhailova *et al.*, 2000). Soil carbon losses in subtropical and tropical soils often match or surpass those under temperate soils (Abril & Bucher, 2001; Lal, 1996; Lobe *et al.*, 2001). Large releases of carbon have been documented in the tropics during forest clearance (Houghton, 2003) and after draining tropical peatlands for oil palm cultivation (Page *et al.*, 2002). The soil organic carbon loss in cultivated soils results from reduced carbon inputs of plant litter (since the net primary production may be reduced relative to the original vegetation, and a large fraction is harvested for human or animal use) and increased carbon outputs through heterotrophic respiration, stimulated by the action of ploughing. If left unattended, SOC losses can render soils unproductive and physically degraded. Large tracts of land exist today where agriculture is no longer practiced due to low SOC content and productivity (Gibbs & Salmon, 2015b).

Soil erosion has been responsible for significant SOC losses, including its indirect effects via reduction in productivity, but quantitative estimates of the net effect remain uncertain. Lal (1995) postulated erosion induced emissions to be a source of 1.1 PgC yr^{-1} to the atmosphere; other analyses treat soil erosion as a net sink of carbon of approximately 1.5 PgC yr^{-1}, because eroded soil may end up deposited and buried downslope or as part of waterlogged sediments where decomposition rates are low (Izaurralde *et al.*, 2013; Stallard, 1998). Recent research estimated total sediment transport and deposition globally at 0.5 ± 0.15 PgC yr^{-1} (Quinton *et al.*, 2010), with less than 2.5% of eroded SOC mineralized and released as CO_2 to the atmosphere (Van Hemelryck *et al.*, 2010). Much less is known concerning amounts and fate of SOC losses caused by wind erosion.

Soils typically lose carbon under human use (Burke, 1999) but can also recover (sequester) the lost carbon and productivity upon implementation of management practices that favour carbon inputs over outputs and reduce soil erosion. Examples of carbon-accruing practices include afforestation, agroforestry, diversified crop rotations, grazing and livestock practices, tillage, residue management, nutrient management, and erosion control (Post *et al.*, 2012; Smith *et al.*, 2008). In spite of gains in crop productivity and implementation of engineering and agronomic practices to conserve soil, the question remains: at the regional scale, are cultivated soils still losing or gaining carbon? Some studies suggest that soil organic carbon content may be increasing in some regions because of the implementation of improved agricultural practices (Janzen *et al.*, 1998; Montgomery, 2007a), regrowth of forests (Montgomery, 2007), or afforestation of croplands (Poeplau *et al.*, 2011). For example, using a meta-analysis approach, Bárcena *et al.* (2014) found that afforestation of former croplands in Northern Europe led to SOC increases, but afforestation of grasslands did not. Losses of carbon have been documented as well. For example, reductions in SOC stocks at an annual rate of 0.6% were observed in England and Wales between 1978 and 2003 (Bellamy *et al.*, 2005a)

Estimates vary for global potential of soil carbon sequestration. Cole *et al.* (1997) estimated that it would be technically possible to recover up to two thirds of the historical SOC losses (about 40 PgC) during a period ranging from 50 to 100 years by implementing improved agronomic practices. This translates to rates of 0.4-0.8 PgC yr^{-1}. Similar estimates by Lal, (2004) range from 30 to 60 PgC achievable during 25-50 years (i.e., rates of 0.6 - 2.4 PgC yr^{-1}). At field scale, observed rates of soil carbon sequestration vary from 0.05 to 1.0 MgC ha^{-1} yr^{-1} with adoption of improved agricultural practices (West & Post, 2002).

4.2.3.2 Loss of terrestrial biomass and carbon sequestration

Productivity

Net primary production is the capacity of land to produce biomass (see Box 4.1 for definitions) and is the source of energy in terrestrial ecosystems, supporting all life. Tropical forests account for 34% of global terrestrial net primary production, tropical savannahs and grasslands 26%, croplands 12%, temperate forests 8%, temperate grasslands and shrublands 7%, boreal forests 7% and drylands 6% (Beer *et al.*, 2010). Anthropogenic land degradation generally reduces net primary production, which is why it changes in net primary production can be an indicator of land degradation. There are exceptions: nutrient oversupply in polluted aquatic systems results in increased productivity (see Section 4.2.4.3). Land degradation is estimated to have reduced net primary production on 23 % of the global terrestrial area; amounting to a 5% reduction in total global net primary production (Van der Esch *et al.*, 2017). Land transformation may lead to less net primary production overall, but a greater fraction of the net primary production is useful to people, which is why the transformation was undertaken.

There are four main sources of information on terrestrial primary production: (1) direct measurement in the field by biomass increase or gas flux measurement (Brienen *et al.*, 2015); (2) remote sensing of the duration and intensity of green cover (Fensholt *et al.*, 2009); (3) seasonal changes in the concentration of CO_2 in the atmosphere (Keenan *et al.*, 2016); and (4) mathematical models of plant production (Cramer *et al.*, 2001). Method 1 is limited by the sparse and uneven distribution of studies; 3 has limited spatial resolution; 4 can only be as good as the data and understanding which informs the models. Thus, currently only method 2 has the spatial resolution coverage to monitor primary production and reveal places where land degradation is taking place (Prince, 2002), but is an inferential method sensitive to assumptions about the efficiency of the conversion of intercepted photosynthetically-active radiation into primary production, rather than a direct measurement.

Box 4.1 Terms used for the different components of primary productivity and carbon sequestration

Total terrestrial gross primary production is the total mass of carbon taken out of the atmosphere by plant photosynthesis. After return to the atmosphere of autotrophic respiration - the carbon-based energy used by plants for maintenance and growth - the remainder is manifest as the production of plant organic material, known as net primary production – sometimes called biomass productivity. The amount of net primary production left in the *ecosystem* after the additional respiration by microbes and animals is the net ecosystem production. The amount of carbon accumulating or lost in ecosystems at the regional scale is the net biome production, defined as the net ecosystem production corrected for lateral transfers of carbon to adjacent biomes, due to process such as trade in agricultural products, export of organic matter in rivers and losses due to disturbances, including land clearing and wildfire (Schulze & Heimann, 1998). In the long-term, for net sequestration of carbon to occur a positive net biome production is required.

Net biome production = Biome area x [gross primary production - plant respiration-animal and microbial respiration ±carbon containing chemicals exported or imported from biome]

Despite the general trend of direct net primary production and biomass reduction from terrestrial ecosystems under human use, there is also evidence for indirect human-induced net primary production and biomass carbon stock increases in many ecosystems worldwide. These increases are attributed to higher temperatures associated with human-caused climate change, nitrogen deposition, altered disturbance and competition regimes, and rising CO_2 levels in the atmosphere. Biomass stocks accrue within logged-over (secondary) forests, as a result of regrowth in between harvest episodes. It also increases due to stand aging if the interval between harvests is increased.

The carbon sequestration associated with the biomass growth increase described above is estimated to be 0.05 to 0.5 Mg C ha^{-1} yr^{-1} (Laurance *et al.*, 1997), not a negligible amount given the large areas involved.

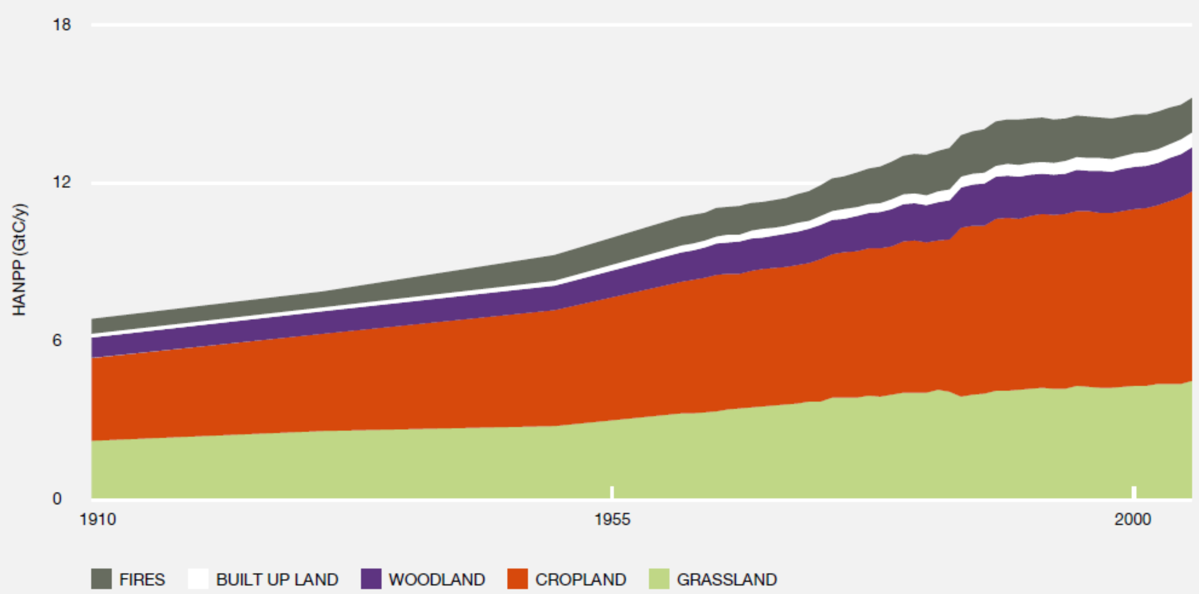
Overall, total net primary production of the terrestrial biosphere has increased by 0.02-0.04% yr^{-1} (an increase of 20 to 40 TgC yr^{-1} relative to a total global terrestrial NPP of around 100 PgC yr^{-1}) over the past several decades (many lines of evidence, including for example, Donohue *et al.*, 2013; Le Quéré *et al.*, 2016; Mao *et al.*, 2016). This increase is the net result of various trends in each biome, some down but others up. Broadly speaking, the increase in productivity since 1982 occurred over 25-50% of the terrestrial surface and a reduction over less than 20% (de Jong *et al.*, 2013; Fensholt *et al.*, 2012; Liu *et al.*, 2015; Zhu *et al.*, 2016). The growth is attributed to one or more of the following: rising CO_2 concentration in the atmosphere; warming and wetting trends in climate over some parts of the world; recovery in net primary production and biomass following past degradation (see Section 4.2.6.1), especially forest regrowth; and fertilization by anthropogenic atmospheric nitrogen deposition (Keenan *et al.*, 2016) (see Section 4.2.4.1). The factors causing the net primary production increase as discussed above have non-linear responses and will saturate over time, even if the drivers continue to rise. The tropospheric ozone content is also rising as a result of human activities, and impairs net primary production by and amount a similar to the stimulation resulting from increases in CO_2 (Ainsworth *et al.*, 2012).

In temperate regions of the northern hemisphere, net primary production reductions occurred from 1995 to 2004, in most places, followed by increases from 2005 to 2012 in many places. These increases in net primary production have been attributed to all of the factors listed above (Mao *et al.*, 2016), especially forest regrowth after almost complete deforestation of large areas of eastern North America and of Europe prior to the 20th century (see Section 4.3.4.) There is evidence of a loss of production in the Congo (Wu *et al.*, 2014) and Amazon Basins (Brienen *et al.*, 2015), attributed to forest transformation to agriculture.

Recent analyses suggest a net sink in arid and semi-arid ecosystems (Donohue *et al.*, 2013), attributed to the effect of rising atmospheric CO_2 on plant water use efficiency (and hence net primary production). There is broad agreement regarding increasing net primary production trends in many subtropical rangelands (Miehe *et al.*, 2008) (see Section 4.2.6.2.). For the period 1982 to 1994, net primary production was lower in parts of the Horn of Africa and south-central Africa, Central Asia and some dry sub-humid parts of South America; for these regions reduction in rainfall and increases in temperature associated with El Niño–Southern Oscillation events (Liu *et al.*, 2015) may have exacerbated human land-use changes and degradation due to inappropriate cropping and grazing practices (see Sections 4.2.6.2, 4.3.2 and 4.3.3).

The fraction of net primary production which is diverted directly or indirectly to human use, is termed "human appropriation of net primary production" (Haberl *et al.*, 2007; Krausmann *et al.*, 2013). For instance, the harvest of biomass from terrestrial ecosystems in Europe exceeds net primary production threefold (Schulze *et al.*, 2010). The fraction of net primary production remaining after the human appropriation is what is available to non-domesticated organisms; thus, rising human appropriation of net primary production is at the expense of biodiversity.

Figure 4 8 **Global human appropriation of net primary production (HANPP) of major land-use types and human-induced fires from 1910 to 2005. Source: Krausmann *et al.* (2013).**



During the last century, human appropriation of net primary production grew from 13% of the net primary production in 1910 to 25% in 2005, reaching 14.8 PgC yr-1 in 2005 (Figure 4.8) (Krausmann *et al.*, 2013). Human appropriation of net primary production increased at a slower rate than human population over the same period, thus human appropriation of net primary production per capita declined from 3.9 to 2.3 MgC yr-1 per person, globally averaged. The major decline occurred after 1950. The amount of biomass consumed as food by each person has remained nearly constant, but the amount of biomass energy has declined with the increase in the use of fossil fuels. A potential future increase in the use of net primary production for biomass energy will likely cause an upturn of human appropriation of net primary production (Erb *et al.*, 2017).

Carbon stocks in biomass, particularly aboveground

After soil organic carbon (4.2.3.1), the next- largest terrestrial carbon stocks are in plant biomass, estimated to be between 450 and 650 PgC. A recent estimate is 497 PgC (Scharlemann *et al.*, 2014). Soil microorganisms are estimated to contain 110 PgC. Total forest biomass has been estimated at 363 PgC, of which tropical forests account for about 60%, temperate and boreal forests about 20%, and the remainder is in savannas and other ecosystems such as mangroves (Donato *et al.*, 2011). Intact Forest Landscapes (see Section 4.2.6.1.) comprise 20% of all tropical forest, yet contain 40% of all the above ground forest carbon. These estimates may be biased because of shortcomings of the data, especially reliance on small samples and many regions without measurements (Feldpausch *et al.*, 2016; Houghton *et al.*, 2009).

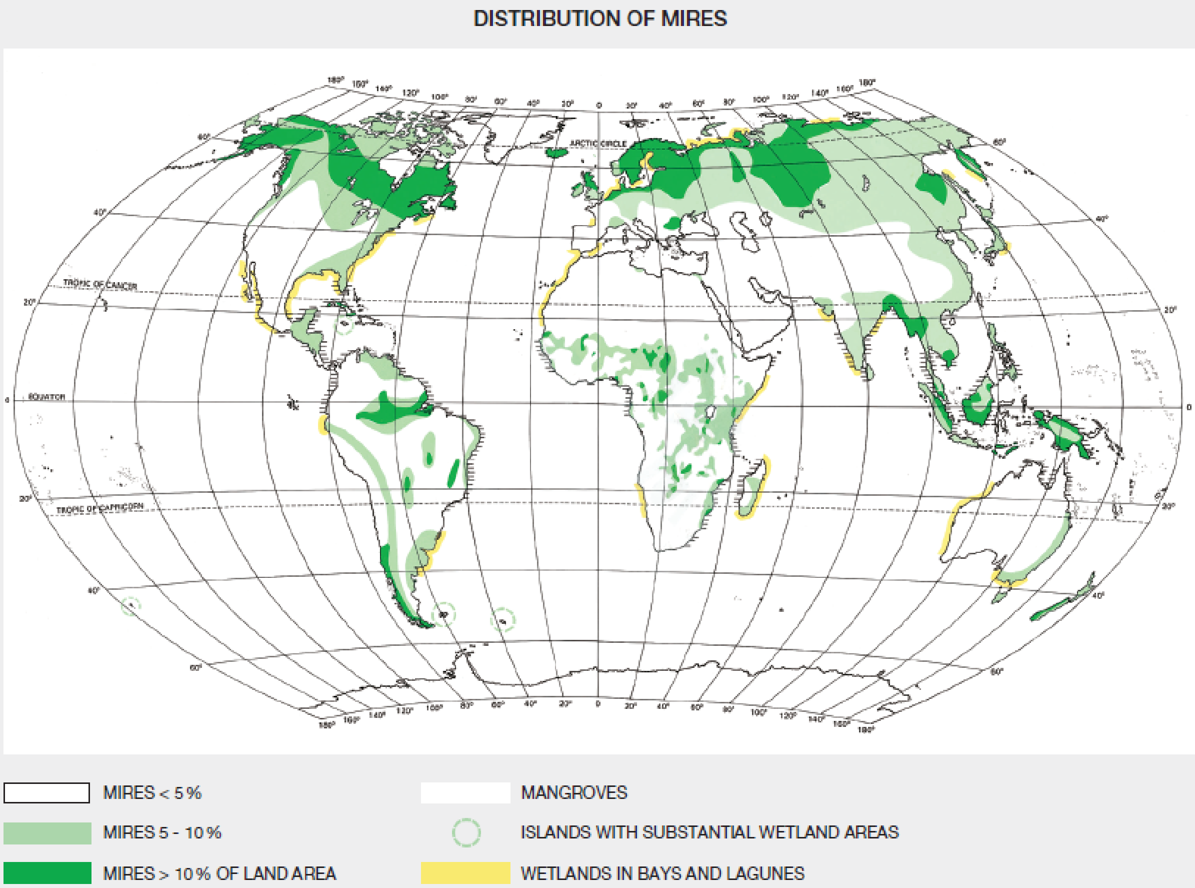
The broad control of biomass stocks is determined by changes in net primary production minus disturbances such as harvest and fire. The current growth of the land biomass stock in untransformed areas (e.g., Running *et al.*, 2004) can only result from increased net primary production, decreased in respiration by microbes or animals, or decreased fire emissions or harvest loss. Since there is no evidence of the latter processes, it is likely that the global net primary production is increasing. This does not mean that there are no areas of decrease caused by some types of land degradation, but it does constrain their extent and magnitude.

The widely-observed encroachment of woody plants into formerly more open, grassy ecosystems (see Section 4.2.6.2) – a form of rangeland degradation – contributes to the land carbon sink (Higgins & Scheiter, 2012), but the relative contribution of local changes in fire (see Sections 4.2.6.3, 4.2.8 and 4.3.6) and grazing (see Sections 4.2.6.2 and 4.3.2) and global causes (rising CO_2 and climate change) to this phenomenon is poorly quantified. Globally, fire is the largest cause of losses in the biomass carbon stock in the short term. In ecosystems with an unchanged natural fire regime, this is not a long-term net loss, since the carbon emitted is taken up in regrowth in subsequent years. In the period 1997-2004, wildfire is estimated to have accounted for 4.4% of carbon returns to the atmosphere. This fraction can rise to a 20% in frequently burned ecosystems such as savannahs (van der Werf *et al.*, 2006).

4.2.3.3 Degradation of peatlands

Peatlands are wetlands where dead plant matter (and therefore carbon) accumulates in the soils and sediments because waterlogging slows down the rate of decomposition. The accumulated mass of semi-decayed plant material is termed peat (Joosten & Clarke, 2002). Peat accumulation typically occurs around 1 mm per year, amounting to 0.08 and 1 MgC ha^{-1} yr^{-1}(Charman *et al.*, 2013; Dinsmore *et al.*, 2010; Yu *et al.*, 2009). Some peatlands have been accumulating carbon for more than 100,000 years and may be as much as 40 m deep (Rydin *et al.*, 2006). Natural peatlands are, on balance, generally greenhouse neutral or have a slight cooling effect on the global climate (Strack *et al.*, 2008), whereas damaged peatlands are substantial emitters of CO_2 (Couwenberg, 2009; Laine *et al.*, 2009; Oleszczuk *et al.*, 2008). Known peatlands cover some 3% of the Earth's land surface and are found in almost every part of the world (Figure 4.9). They are estimated to contain more than 600 PgC (Yu *et al.*, 2010). This is similar to the amount carbon held in the biomass of the world's vegetation (see Section 4.2.3.2). This is likely an under-estimate because large areas are continually being recognised as peatland having previously been categorised as other habitat types (e.g., Dargie *et al.*, 2017; Draper *et al.*, 2014). Batjes (1999) notes that peats contain at least five times more carbon than any other soil type, so even small changes in their documented extent can result in substantial changes to the known global carbon store.

Figure 4 9 Major known areas of peatland distribution. Source: Adapted from Lappalainen (1996), by permission of the International Peat Society.

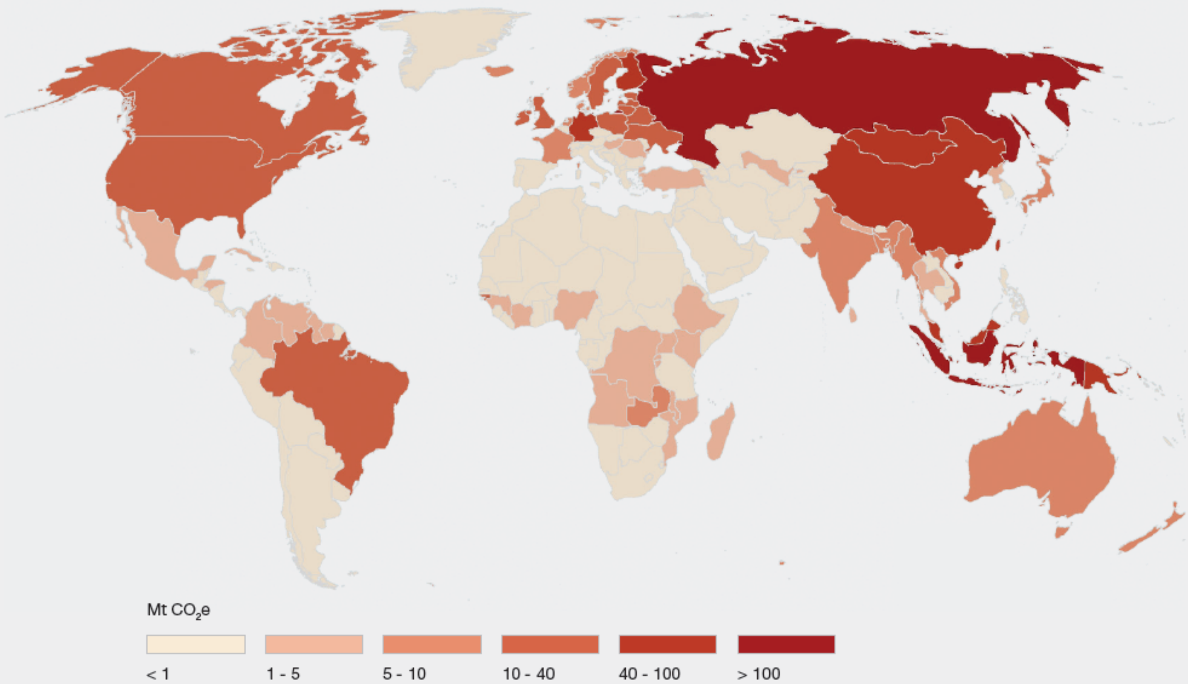


Peatlands are the most extensive form of terrestrial and coastal wetland (Section 4.2.5.2). Davidson, (2014) shows that wetland losses of 87% are typical of some regions, although Joosten, (2009) indicates that only 11.6% of the world's peatlands are currently considered to be degraded, this estimate is dominated by huge stretches of undamaged peatland in northern Canada and Russia. Even here, however, entire regions are undergoing change because of permafrost melting due to climate change (Christensen *et al.*, 2004; Voigt *et al.*, 2017).

Studies in non-boreal regions reveal as much as 99% degradation or loss of peatland habitat. The 3,400 km^2 of the UK's East Anglian Fens are now reduced to less than 10 km^2 (Darby, 1956; Sheail & Wells, 1983) in a pattern of land-use change typical across the globe for groundwater-dependent fen peatlands (Bragg & Lindsay, 2003; Williams, 1991). Bog systems (i.e., entirely rain-fed peatlands) are more challenging environments for humans to transform to agriculture because of their low nutrients and acidity (Section 4.2.4.2), but near-natural habitat has been reduced to 5% of its former extent in some regions (Grünig *et al.*, 1984; Lindsay & Immirzi, 1996). A comprehensive review of European peatlands has revealed that approximately 10% of peatlands have been lost completely while 48% of the remainder are in a degraded state (Tanneberger *et al.*, 2017). Subsidence is an inevitable consequence of peatland drainage and now threatens many former coastal peatland areas with inundation (Hooijer *et al.*, 2012).

Current estimates of annual carbon emissions (as CO_2 and CH_4) from known peatlands show a total of some 2 PgC y^{-1}, nearly twice that released annually by consumption of aviation fuel (Joosten, 2009; Wetlands International, 2015) (Figure 4.10). A single year of peatland fires in Southeast Asia is estimated to have released an amount of carbon equivalent to as much as 40% of all global fossil fuel emissions for that year (Page *et al.*, 2002).

Figure 4 10 Annual emissions from natural and damaged peatlands per country in Mt CO_2e (that is exchange of all gases including methane (CH_4) converted into values of global warming potential for equivalent amounts of CO_2) indicating countries that contribute most to global peatland emissions. Source: Map courtesy of Griefswald Mire Centre.



4.2.4 Pollution

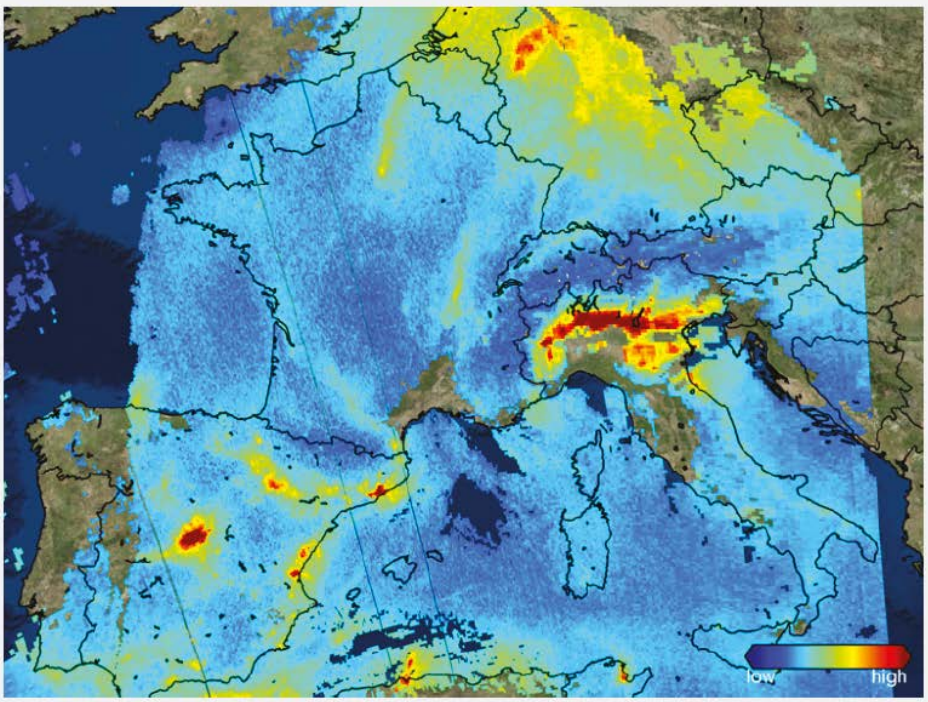
4.2.4.1 Atmospheric pollution

Over the last century human activities have increased emissions of reactive nitrogen, sulphur and mercury resulting in impacts to the environment and human health (Driscoll *et al.*, 2001, 2013; Galloway *et al.*, 2008). Oxidized nitrogen, sulphur dioxide and mercury are emitted from fossil fuel combustion, while agricultural activities largely contribute emissions of reduced nitrogen (e.g., ammonia). Emissions of reactive nitrogen, sulphur and mercury are deposited to the Earth's surface. These pollutants undergo transformation in the atmosphere and are transported far from human sources to remote unmanaged lands where atmospheric deposition dominates nitrogen inputs to nitrogen-limited ecosystems (e.g., Phoenix *et al.*, 2006); they supply mercury, causing exposure to terrestrial and aquatic biota (Driscoll *et al.*, 2013); and can acidify acid-sensitive soils and water (see 4.2.2.1) (Greaver *et al.*, 2012)

Lamarque *et al.* (2013) estimated historical and projected future global atmospheric nitrogen and sulphur deposition under the IPCC Representative Concentrations Pathways (see Section 4.1.5; Figures 4.11, 4.12, 4.13, 4.14.). In 1980, atmospheric sulphate and nitrate depositions were elevated in eastern North America, Europe, central Africa and East Asia due to intensive fossil fuel use. By 2000, deposition decreased in North America and Europe due to economic changes and air quality management, while deposition increased in east and south-central Asia due to industrialization and increases in population. Future projections assuming the Representative Concentrations Pathway 4.5 scenario suggest that these deposition trends will continue through 2030. Patterns of ammonium (reduced nitrogen) deposition contrast with sulphate and nitrate due to emissions from agricultural activities (Figures 4.11 - 4.14). Ammonium deposition is elevated in central North America, North and East-central South America, Central Africa, Europe, Indonesia and West, South-central and East Asia and projected to increase under Representative Concentrations Pathway 4.5 from current values to 2030 particularly in south-central Asia.

Figure 4 11 **Nitrogen dioxide over Europe on 22 November 2017.**

The highest concentrations are over the Po Valley in northern Italy and western Germany, likely associated with the combustion of fossil fuels from industry and road traffic. Source: McKinnon (2017).



Sulphur and nitrogen emissions deteriorate ambient air quality due to formation of ozone and fine particulate matter, contributing to cardiovascular and respiratory conditions and premature deaths. Increased near-surface ozone concentrations, largely as a consequence of nitrogen oxides, methane and non-methane volatile compounds in the presence of sunlight and exacerbated under climate change, decrease crop yields (Capps *et al.*, 2016). Ozone decreased soybean and maize production in the USA by 5% and 10%, respectively, between 1980 and 2011 (McGrath *et al.*, 2015) and was responsible for 5-11% loss in winter wheat and 3-6% in rice from 2002 to 2007 in India (Debaje, 2014). Elevated atmospheric nitrogen deposition contributes to the eutrophication of soils causing changes in plant species composition and diversity in unmanaged terrestrial ecosystems; increases in emissions of nitrous and nitric oxides; and elevated runoff of nitrate resulting in eutrophication of fresh and coastal waters (Galloway *et al.*, 2003, 2004). Atmospheric nitrogen deposition exceeding a threshold of 10 kg N ha^{-1} y^{-1} is an order of magnitude greater than natural rates and may result in adverse effects (Bouwman *et al.*, 2002; Pardo *et al.*, 2011). Sulphate, nitrate and ammonium deposition to acid sensitive regions can acidify soils and impair the health of tree species and acidify surface waters, decreasing biodiversity (see 4.2.2) (Driscoll *et al.*, 2001). Future efforts to control emissions may be offset by the growing demand for food and energy in the developing world likely increasing inputs of reactive nitrogen (Erisman *et al.*, 2008; Galloway *et al.*, 2008).

Atmospheric deposition is also the dominant pathway for mercury to ecosystems (Driscoll *et al.*, 2013). There are geogenic (natural – volcanos, soil weathering), primary human, and secondary (reemissions – soil and water emissions of previously deposited mercury, biomass burning) emissions of mercury. Mercury emissions occur as elemental mercury, which is a global pollutant due to its long atmospheric residence time (0.5-2 yrs), and oxidized mercury which is largely deposited locally. Primary human mercury emissions include artisanal gold mining (37%), coal combustion (24%), non-ferrous metal production (10%) and cement production (9%) (UNEP, 2013). Atmospheric mercury deposition can be converted to methylmercury, which is biomagnified to elevated concentrations in top predators, resulting in exposure and health effects to humans and wildlife (Driscoll *et al.*, 2013).

In addition to the direct effects of atmospheric pollution, there are effects on the regional and global energy balance owing to the reflection of sunlight from atmospheric particulates and aerosols, and by their effects on cloud cover (see Section 4.2.8).

Figure 4 12 **Total sulphur deposition in kg S ha^{-1} yr^{-1} for 1980, 2000 and 2030.**

Derived from the multi-model global datasets for sulphur deposition and climate change scenario Representative Concentrations Pathway 4.5 (see Section 4.1.2.3). Source: Lamarque *et al.* (2013).

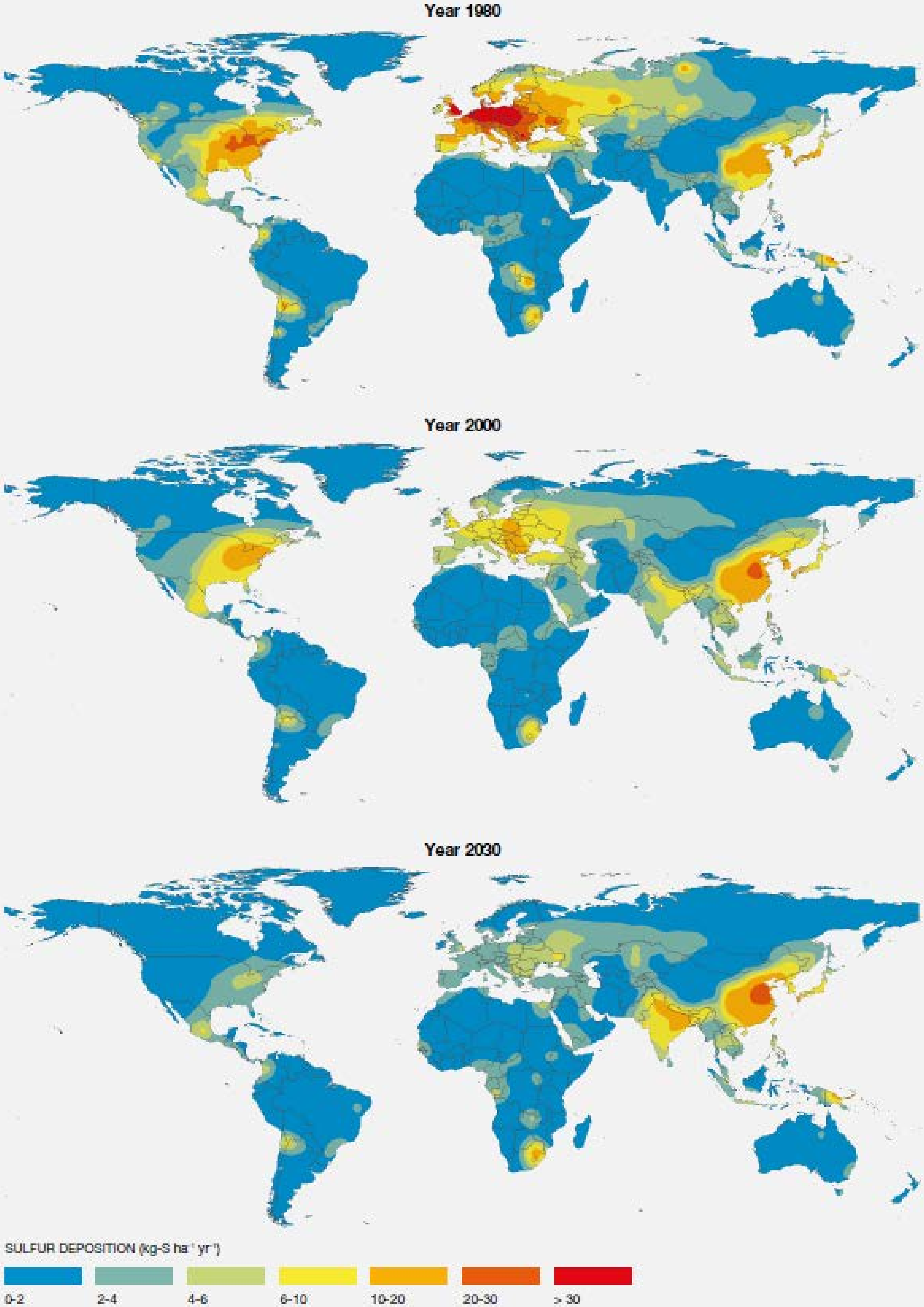


Figure 4 13 **Total nitrate deposition in kg N ha^{-1} yr^{-1} for 1980, 2000 and 2030.**

Derived from the multi-model global datasets for nitrogen deposition and climate change scenario Representative Concentrations Pathway 4.5 (see Section 4.1.2.3). Source: Lamarque *et al.* (2013).

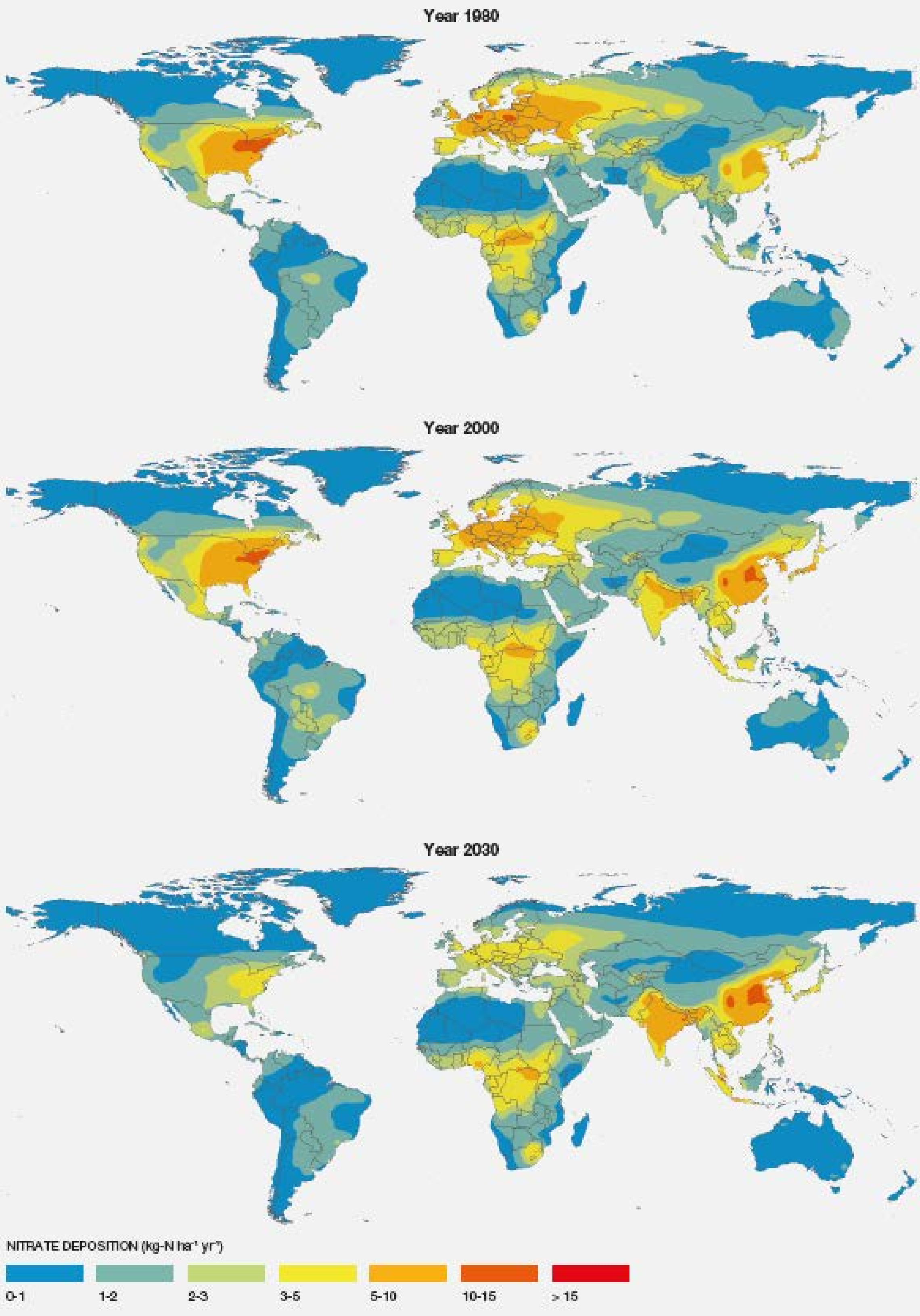
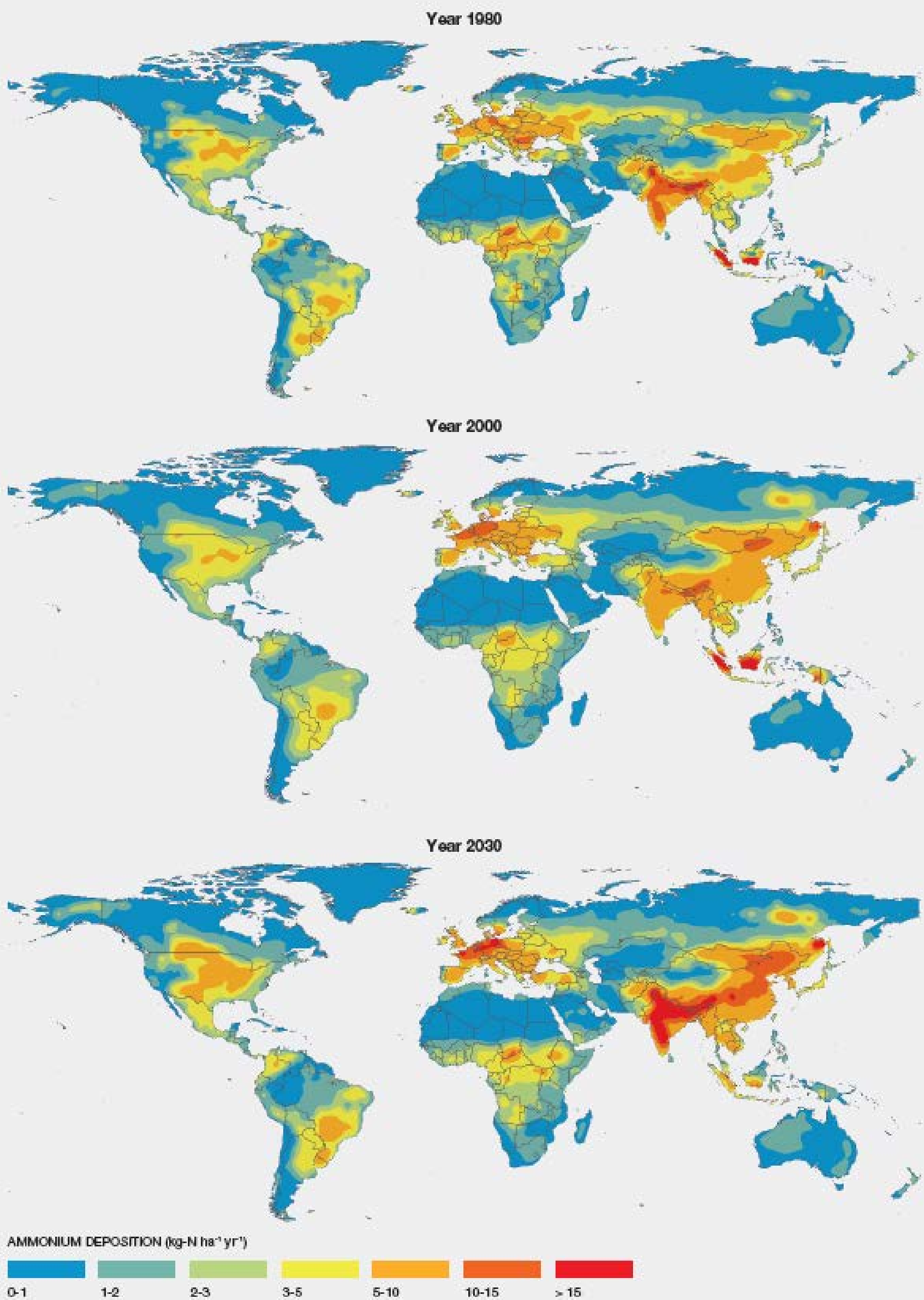


Figure 4 14 **Total ammonium deposition in kg N ha-1 yr-1 for 1980, 2000 and 2030.**

Derived from the multi-model global datasets for ammonium deposition for 1980, 2000 and 2030 Representative Concentrations Pathway 4.5 (see also Section 4.1.2.3). Source: Lamarque *et al.* (2013).



4.2.4.2 Soil pollution

Agriculture

China, India and the USA account for over 50% of global fertilizer consumption (FAO & ITPS, 2015). A global mass balance analysis (Bouwman *et al.*, 2009) shows very high rates of soil nitrogen and phosphorus accumulation occur in densely populated Europe and South Asia for 2000. A comparison of rates for the year 2000 with those of 1970 suggest that soil nutrient accumulation has decreased in Europe, but is increasing markedly in South Asia and, to a lesser extent, other developing regions including South and Central America and Africa. Hotspots of agricultural nutrient use have shifted from North America and Europe in the 1980s to Eastern Asia. Africa is expanding agricultural areas, but with a small increase in fertilizer usage (Lu & Tian, 2017).

Trends toward intensive livestock production result in large quantities of manure. Manure is a valuable source of nutrients, but due to transportation costs is typically used close to the source (Teenstra *et al.*, 2014). Manure can not only be a major source of nutrients and trace elements where generated and from over-application to farmlands, but can cause also imbalances in nutrient ratios (Miller, 2001).

Bouwman *et al.* (2009) showed that nitrogen losses by denitrification, ammonia volatilization and runoff are increasing, with consequent environmental degradation. The total runoff of nitrogen from global croplands is estimated at 35 million tonne nitrogen yr^{-1}, of which 70% (24.4 million tonne N yr^{-1}) originates from anthropogenic sources (fertilizers, manure) (Mekonnen & Hoekstra, 2015). The wide-scale use of synthetic and organic fertilizers has far reaching environmental impacts, including air pollution, soil acidification and degradation, accumulation of trace metals, crop yield reduction, and eutrophication of both inland (see Section 4.2.2) and coastal waters (Lu & Tian, 2017; Savci, 2012). Substantive improvements in nitrogen use efficiency and reductions in total nitrogen use have been achieved in some countries. In Denmark, legislative controls and adoption of best management practices have decreased the applied nitrogen by 52% since 1985, resulting in a 47% reduction in ammonia emissions (Beatty & Good, 2011; Olesen *et al.*, 2004). Fertilizer usage can be reduced by 30-50% without affecting yields, but greatly decrease air and water pollution (Beatty & Good, 2011; Hoben *et al.*, 2011; Ju *et al.*, 2009; McSwiney & Robertson, 2005). Growing crops to the economic optimum yield rate, rather than optimising total yield is both an economic and environmentally preferable option (Kim & Dale, 2008; Scharf *et al.*, 2005). Changing management practices such as tilling methods, type of fertilizer used or timing of applications can reduce pollution (Beatty & Good, 2011).

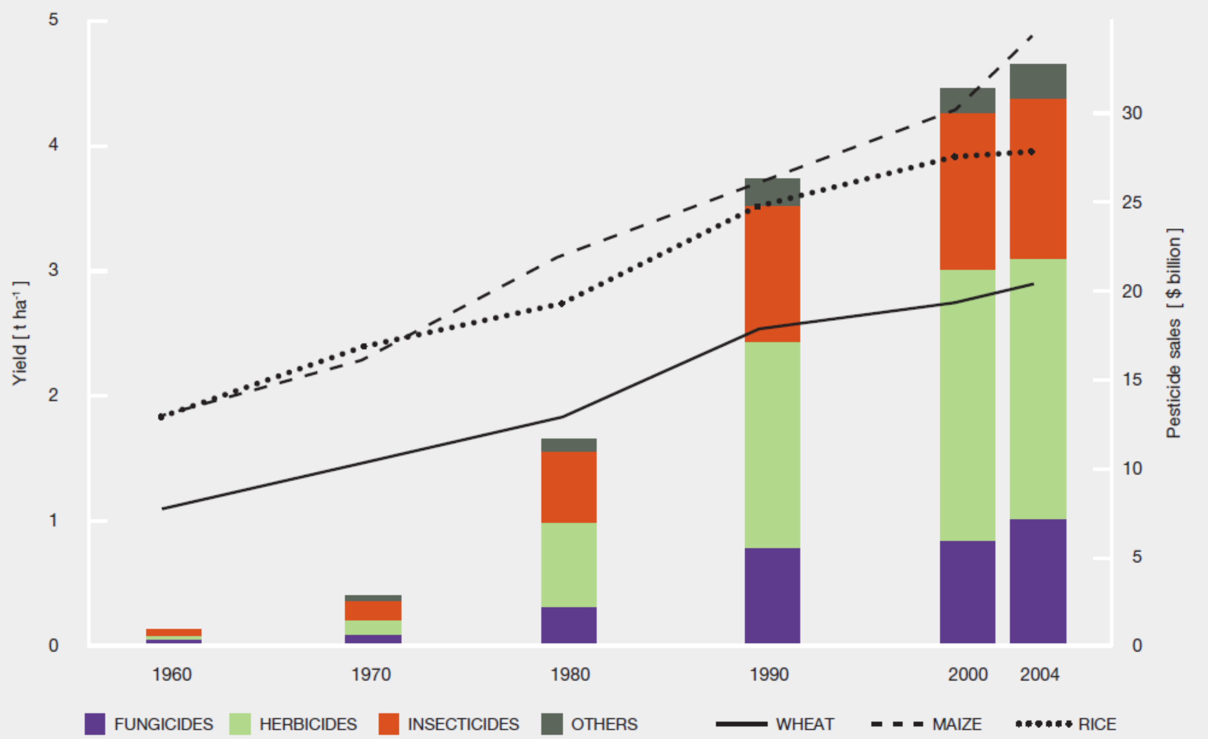
Persistent organic pollutants

Persistent organic pollutants are products or by-products of industrial activities. Persistent organic pollutants released by combustion are common. Most persistent organic pollutants are of relatively recent origin – first appearing in the mid-20th century. They comprise hundreds of organic chemicals that are used on every continent, including dioxins, furans, hexchlorobenze (fungicide), polychlorinated biphenyls (PCBs) and polycyclic aromatic hydrocarbons among many others. Some persistent organic pollutants are no longer manufactured, such as PCBs, hexchlorobenze and DDT (but still used for mosquito control in some parts of the world).

Important characteristics of persistent organic pollutants are: persistence (slow degradation and occurrence of intermediates); bioaccumulation in living tissues; toxicity (adverse effects to humans, wildlife or the environment); and long-range transport potential far from the original release.

Global crop yields have increased sharply, aided by pesticide use (Figure 4.15). While increases in pesticide use have occurred worldwide, application rates vary widely among countries. Although the use of pesticides in developed-countries has decreased markedly, their use in the developing world continues to rise. In addition to pesticides, antibiotics, which are used in livestock production, remain active in excreted biological matter (faeces, urine) and are released into the environment.

Figure 4 15 Worldwide trends in average yield per unit of area for wheat, rice and maize and pesticide sales (1960–2004). Source: Oerke (2006).



Several studies have reported pesticide residues in human food (Jardim & Caldas, 2012; Szpyrka *et al.*, 2015) and breast milk (Fan *et al.*, 2015; Honeycutt & Rowlands, 2014). The significance of quantities of pesticides in soils is uncertain since threshold values have not been established for human toxicity to single pesticides, still less for mixtures, so estimation of the risk to exposure is currently not possible.

Monitoring programs show that application of pesticides and livestock antibiotics in agricultural regions are transported to adjacent lands and downstream water bodies (Benotti *et al.*, 2009; Golovko *et al.*, 2016; Wang *et al.*, 2016). Transport pathways are atmospheric by airborne suspension from sprays, volatilization from soil surfaces and airborne dust contaminated with pesticide (Bento *et al.*, 2017), and fluvial by soil erosion or associated with dissolved organic matter.

Pesticides affect a range of soil processes, including decomposition of organic matter and infiltration of rainwater (Pelosi *et al.*, 2014). Herbicides generally are less deleterious to soil organisms than insecticides and fungicides (Bünemann *et al.*, 2006), but significantly reduce plant biodiversity (Geiger *et al.*, 2010).

Insecticides and fungicides have greater effects on soil organisms than herbicides, especially copper-containing fungicides.

No remediation strategies exist for persistent and diffuse pollution by pesticides, only prevention through sustainable cropping measures, such as Integrated Pest Management. As with pesticides, many persistent organic pollutants have been invaluable for pest and disease control, crop yields and industry and have improved the quality of life. However, deleterious effects of persistent organic pollutants have been evident for the past 30-40 years.

Trace elements

Soils are contaminated with trace elements when concentrations are high enough to disrupt ecosystem services. Of the 78 naturally-occurring trace elements, contamination by arsenic, cadmium, chrome, copper, mercury, nickel, lead, selenium and zinc are of greatest environmental concern based on potential for human, wildlife and plant toxicity and the area affected (Mulder & Breure, 2006; Pierzynski & Gehl, 2004). The loss of terrestrial primary productivity is likely the most significant impact.

Sources of trace element contamination vary considerably from naturally occurring, low level contamination associated with release from soil or weathering, to small areas with high concentrations caused by spills or poorly managed human activities (e.g., mining, smelting, industrial production), to widespread atmospheric deposition or land application of contaminated by-products including animal manures and biosolids. Due to the wide variety of sources, differences in the degree of contamination and sizes of areas affected it is difficult to assess regional and global status of trace element contamination. Furthermore, the toxicity of some elements, such as chrome and mercury, depends of their speciation, so total analysis of the contaminant provides limited insight on potential for human exposure.

4.2.4.3 Freshwater pollution

Introduction

Pollution is a major threat to freshwater services and biodiversity globally (Dudgeon, 2013). It leads to extirpation of species, changes in biogeochemical cycling and simplification of aquatic food webs. Direct inputs of industrial, mining or domestic pollutants to freshwaters are common in the developing world (e.g., Darwall *et al.*, 2011). Nonpoint inputs of sediments, fertilizers and contaminants from urban and agricultural activities (Table 4.9) are growing in the developing world but already quite high in North America, Europe and Australia. Older cities have often combined waste and storm water sewer systems that overflow and contaminate rivers during high runoff events.

Table 4 6 Dominant forms of pollution wide and the underlying causes. Source: Laws (2017); Mekonnen & Hoekstra (2015); Stehle & Schulz (2015); UNEP (2016).

FORM OF POLLUTION	AGRICULTURE	URBANIZATION	INDUSTRY	MINING
Pesticides	XX			
Herbicides	XX	X		
Nutrients	XX	X		
Silt/sedimentation	XX	X		X
Metals		X	XX	XX
Pharmaceuticals		X		
Salinization	X	X		X
Petroleum products		X	XX	

Eutrophication

Agriculture impacts surface and groundwater due to soil erosion, run-off and is the primary source of nutrient pollution in the USA. In Asia, it led to high nutrient levels in 50% of the rivers and moderate levels in 25% (Evans *et al.*, 2012). In China, direct inputs of manure from animal production contributes >60% of nutrients to northern rivers and up to 95% in the central and southern rivers (Strokal *et al.*, 2016). Most major lakes in Latin America and Africa have increasing nutrient loads due to livestock wastes and runoff of inorganic fertilizer from croplands (UNEP, 2017). Urbanization also contributes to nutrient pollution and is now considered the dominant threat globally to the integrity of water that supplies cities. McDonald *et al.* (2016) estimate that some level of water degradation has now occurred in 90% of urban source watersheds. From 1900-2005, they report an increase in the average pollutant yield of urban source watersheds by 47% for phosphorus and 119% for nitrogen.

The combination of high levels of organic wastes and high nutrient levels leads to dramatic declines in oxygen owing to microbial respiration, with cascading ecosystem effects such as hypoxic "dead zones" (Diaz & Rosenberg, 2008), leading to declines in fisheries and other aquatic organisms that are the main source of protein for many people

Pharmaceuticals and other chemicals

Pollution from pesticides and other organic pollutants occurs worldwide. Malaj *et al.* (2014) found that up to 75% of the sites sampled in river basins in the north-western region of Europe had organic chemical levels posing a very high risk (often acute toxicity levels) to invertebrates, fish, algae and other aquatic organisms. Pollution from wastewater discharge in rapidly developing countries is high with Asian river basins having the highest number of people living in wastewater-polluted river basins (Wen *et al.*, 2017). At least 38 pharmaceutical substances are found in surface and ground waters throughout the world and up to 100 in the USA and some European countries (Beek *et al.*, 2016).

Salinization

Most freshwater organisms cannot tolerate saline water and ecosystem processes including biogeochemical transformations and food web transfers are harmed. High salinity in rivers and streams can result from natural sources, but more common today from human activities, particularly agriculture, mining and de-icing of roads (see Section 4.2.2.2). About 10% of all river stretches in Africa and Asia have high salinity levels primarily associated with agricultural irrigation (UNEP, 2017); Latin American rivers have similar levels of degradation but it is primarily from industry. In the USA, winter concentrations of salts in streams can spike to approximately 25% that of seawater (Kaushal, 2016). In addition to the osmo-regulatory stress freshwater organisms experience in salinized water, they are exposed to contaminants that can be mobilized from sediments due to salinization.

Sediment pollution

Many streams and rivers naturally carry very high loads of sediment and are turbid year-round. However, degradation of freshwater ecosystems due to excessive inputs of fine sediment to streams that otherwise have low levels is occurring worldwide largely due to urbanization and farming (Naden *et al.*, 2016; Russell *et al.*, 2017) (see Section 4.2.1).

4.2.5 Changes in hydrological regime

4.2.5.1 Freshwater degradation

Overview

Land degradation associated with urbanization, agriculture and mining indirectly modifies aquatic ecosystems, affecting habitat availability and quality and agricultural food production. Land degradation is a major driver of the changes in freshwater quality and quantities, while the impacts of this extend to all ecosystem types where freshwater ecosystems are particularly vulnerable (Vörösmarty *et al.*, 2010).

Several types of land degradation can cause green water depletion. Reductions in soil organic matter (see Section 4.2.3.1) and soil depth due to soil erosion (see Section 4.2.1) directly reduce the soil water holding capacity. Degradation and reduction in vegetation cover (e.g., by agriculture, overgrazing, deforestation, or fire), exposes soil surfaces to raindrop impact, or creates physical surface crust layers that reduce infiltration rates by orders of magnitude. Increased runoff is the major cause of land degradation through gradual erosion (see Section 4.2.1) and strongly through frequent flash floods generated by reduced vegetation cover (e.g., Costa *et al.*, 2003; Pinter *et al.*, 2006). The resulting sediment and soil chemical transport leads to reduction in blue water quality through clogging of water ways and filling pools and lakes, covering the original water bed with consequent effects on water biota (Allan *et al.*, 1997).

Degradation of hydrologic regimes

Changes in surface processes affect the availability and quality of blue water resources used to meet human needs and support aquatic organisms. The creation and maintenance of habitat for aquatic organisms is directly tied to watershed-scale processes that influence the delivery of sediment and water to streams. As land is cleared of vegetation or paved-over, sediment and water fluxes to rivers and streams increase. Under these conditions, both overland and shallow subsurface flows increase rapidly during rainfall, creating high peak flow velocities in streams, ultimately causing channel scour, transport of fine materials and low retention of organic matter (Paul & Meyer, 2001).

Surface hydrologic regimes

If land degradation extends all the way to the stream channel, stream flows may not be slowed by riparian vegetation or inputs of wood. Higher streamflow rates may result in erosion potentially causing channel deepening, floodplain disconnection, loss of critical habitat for aquatic organisms and modification of important biogeochemical processing (Naiman & Décamps, 1997). Reduction in the natural input of wood (leaf litter, branches and logs) to waterways is problematic because the presence of wood in the stream channel alters flow patterns, creates scour pools in running-water systems and can serve as important habitat for many fish and other aquatic species (Gregory *et al.*, 2003). By partially restricting flow and trapping sediment, wood accumulations also help develop and maintain river-floodplain connections, which further increases habitat complexity (Wohl *et al.*, 2015).

Groundwater regimes

Aquifers supply drinking water to billions of people, water for irrigation of agricultural land and groundwater seepage into rivers, upon which many ecosystems depend (Gleeson *et al.*, 2012). Broadly, three semi-independent processes lead to the degradation of aquifers: (1) depletion of aquifer storage due to over-pumping and its effects in reducing both groundwater levels and freshwater availability to terrestrial and aquatic ecosystems, particularly during dry periods; (2) groundwater salinization when salts and nutrients are flushed from subsurface soils during recharge by rain or irrigation, and sometimes in upper estuaries when upstream freshwater inflows have been depleted and salt water intrusion occurs; this usually, but not exclusively, occurs in coastal aquifers; (3) inputs of pollutions from point sources, such as urban and industrial wastes and chemicals, or from diffuse nonpoint sources, less concentrated but widespread, including nutrients and pesticides from agriculture (Foster & Chilton, 2003; Morris *et al.*, 2003; Scanlon *et al.*, 2007). Subsidence caused by ground water extraction is increasing with human use of ground water (Galloway *et al.*, 1999) (see Section 4.2.6.4).

Status and trends in groundwater

A recent estimate of annual global groundwater storage depletion in sub-humid, semi-arid and arid climatic zones suggests that between the years 1960-2000 there was continuous depletion, more than doubling over the 40-year period (from 126 ±32 to 283±40 $km^3\ yr^{-1}$ respectively). This means that 39±10% of the yearly groundwater withdrawals were not replenished by recharge (Wada *et al.*, 2010). The global groundwater footprint – which considers that portion of water required for supporting environmental flows – is 3.5±0.7 times the actual area of aquifers (Gleeson *et al.*, 2012). An estimated 80% of aquifers have a groundwater footprint less than their area, so the net global withdrawal is driven by a few heavily exploited aquifers. Aquifers that are stressed by withdrawals an order of magnitude more than the global average include the upper Ganges, Arabians, south Caspian and Nile Delta. Gleeson *et al.* (2012) estimated that 1.76 ± 0.4 billion people live in regions where groundwater resources and/or groundwater-dependent ecosystems are under threat, with approximately 60% of them located in India and China.

Status and trends in surface water

Mass balance estimates show that the global continental freshwater discharge for a 13-year period (1994-2006) increased by 540 $km^3\ yr^{-1}$, largely attributed to an increase of global-ocean evaporation (768 $km^3\ yr^{-1}$). Recent estimates of trends in freshwater discharge show large variations in yearly streamflow in most of the world's large rivers and also in continental discharge. Inter-annual-to-multi-decadal variation in discharge was found to be directly related to precipitation (Dai *et al.*, 2009; Gerten *et al.*, 2008).

Changes in land cover and land use were second in importance in affecting discharges over the 20th century, particularly in the tropics. However, the exact effects of different land-cover and/or use changes are uncertain and experts differ on the effects of tropical deforestation (Gerten *et al.*, 2008; Gerten, 2013; Piao *et al.*, 2007). The magnitude of the effects of irrigation and storage in reservoirs and other human activities on annual global river flows is uncertain (Liu *et al.*, 2017), although it is possibly related to the fractional irrigation area of river basins. The largest areas of uncertainty are in most areas of Asia and the northern countries of the Mediterranean basin. Sustained growth of these flux rates into long-term trends would indicate an increase in the intensity of the hydrologic cycle (Syed *et al.*, 2010).

In addition to land cover, direct modification of aquatic systems has been occurring to an increasing extent since the start of the Anthropocene – wetlands have been filled (Section 4.2.5.2), streams paved over and rivers channelized. Loss of habitat associated with these activities has had a dramatic impact on aquatic biodiversity, freshwater ecosystem services and the flux of materials that influence global processes (Dudgeon, 2013; Roy *et al.*, 2005).

Although less than 10% of the total annual renewable blue-water is withdrawn for human activities (mainly irrigation, industry and drinking), 2.4 billion people live in highly water-stressed areas because of the uneven distribution of renewable blue and green water resources in time and space (Oki & Kanae, 2006; Rockström *et al.*, 2007). Nearly 80% (4.8 billion) of the world's population have low water security, accompanied by high loss of aquatic biodiversity (Vörösmarty *et al.*, 2010).

Status and trends in evapotranspiration

Global plant transpiration (green water flow) has reduced by 7.4% over a period of 30 years (1961 -1990) due to land-cover changes, mainly forest clearing for agriculture, across Europe, USA and Western and South-eastern Asia. During the same period, the global evaporation (white-water flow) increased by 9.7 % (Gerten *et al.*, 2005; Griebler & Avramov, 2015). The capacity of cropland soils to retain water in the root-zone is affected by the amount of soil organic matter and, while there are no global surveys, it has been estimated that croplands have lost 30-50% of their organic matter content (Lal, 2002) as a result of intensive tillage.

4.2.5.2 Wetland loss

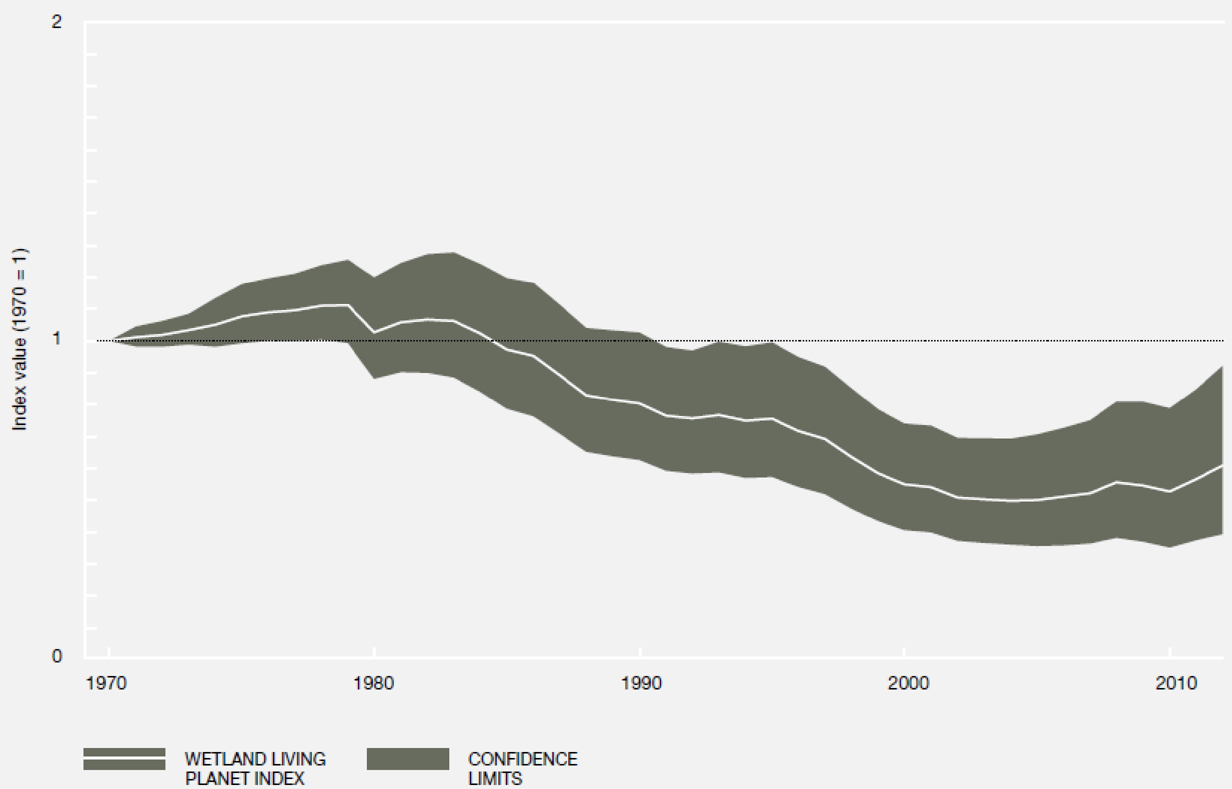
Status and trends in degradation

According to the most recent estimate, about 87% of wetlands have been lost worldwide in the last 300 years (Davidson, 2014), with 54% of the loss happening since 1900; the study included data from 189 studies on wetland loss globally. The loss was higher in inland wetlands (61%) as opposed to coastal wetlands (46%). The study shows that the annual rate of wetland loss in the 20th and 21st increased ten-fold than that before the 18th century (-0.11%). The Convention on Biological Diversity (CBD) Progress towards the Aichi Biodiversity

Targets report (Leadley *et al.*, 2013) shows that during the period between 1970 and 2008 the global relative extent of wetlands diminished by 53% and 73% in Europe. Although the trend between 1970 and 2008 shows higher losses from Europe and Asia, the overall loss during the 20th and 21st was largest in Europe and North America; 56% loss relative to 1900. A similar trend is found in the Living Planet Index (World Wildlife Fund, 2016) for wetland-dependent species, where species abundance decreased 39% (range: -8 to -60%) between 1970 and 2012 (Figure 4.16). The global trends of wetland extent between 1970 and 2008 included more than 1,000 wetlands from 170 studies (Leadley *et al.*, 2013; Liu *et al.*, 2017; Ramsar, 2013).

Figure 4 16 **Global trends of the Living Planet Index for wetland-dependent species.**

The Living Planet Index includes data on population abundance for 706 inland wetlands populations of 308 freshwater species monitored across the globe between 1970 and 2012. Source: Living Planet Report (WWF, 2016).



Description of the process

Wetlands have been drained, filled, logged, polluted or degraded in some way for millennia (Davidson, 2014). Wetland degradation usually involves an alteration of the hydrological regime, either completely disrupting it (e.g., drainage) or changing it (e.g., isolation from the tides or from the river flow). It also involves a complete removal of vegetation and animal aquatic communities or a substantial change in them due to altered hydrological dynamics. Degradation can also be consequence of eutrophication by urban and agricultural sources.

Impact on biodiversity, ecosystem process and function

A meta-analysis comparing restored and undisturbed wetlands found that wetland hydrological dynamics recovered to reference levels right after restoration happened (Moreno-Mateos *et al.*, 2012). However, species richness and abundance, recovered to only 77% (on average) of reference values, even 100 years after restoration. After 50 to 100 years, restored wetlands recovered to an average of 74% of their biogeochemical functioning relative to reference wetlands. Nitrogen cycling was below reference levels for 30 years and carbon cycling was only 50% of the reference after 50 years. This study reported that different recovery metrics could have very different recovery times. Specifically, it showed that while recovery of vertebrate diversity and abundance could happen within 10 years, plant recovery was still below the reference after 100 years (Figure 4.17) (Moreno-Mateos *et al.*, 2012). Similarly, carbon stored in soils only recovered to 50% after 50 years after restoration while phosphorus did not change. The study also reported faster recovery in warm climates than in cold ones, and in wetlands over 100 ha than in smaller wetlands.

Wetlands are key habitats, connected with processes occurring over a much wider territory. The biotic connection through dispersal mechanisms among wetlands indicates that preservation of isolated sites that are considered to be of special importance (e.g., concentrations of migratory water birds), has another aspect (e.g., water bird migration). This interconnected element calls for a regional approach to wetland management within a continental and global context (Amezaga *et al.*, 2002).

Figure 4 17 Synthetic chrono-sequence of the evolution of different metrics after wetland restoration.

Response ratio was the results of comparing metrics at restored and reference sites. Upper panel includes measurements of species richness and abundance of the groups represented. Lower panel includes measures of carbon, nitrogen and phosphorus in soils. Dots and error bars represent average values and standard errors. Dashed line at the zero of the Y axis represents undisturbed reference wetlands. The numbers on the X axis (in black) indicate years since restoration.
Source: Moreno-Mateos *et al.* (2012).

4.2.6 Changes in land cover

4.2.6.1 Land-cover conversion

Land cover refers to the physical and biological cover of the surface of the land, including water, vegetation, bare soil, habitations, and impervious surfaces. Land use is more complicated, consisting of human activities such as agriculture, forestry, grazing, and building construction. For example, areas covered by woody vegetation may be an undisturbed natural shrubland, a forest preserve, regrowth following forestry, a plantation, fallow swidden agriculture plots, or an irrigated tea plantation. Different types of land cover can be managed or used quite differently. Changes in cover can have fundamental effects on the global environment (Leemans & Zuidema, 1995).

Types of land-cover degradation

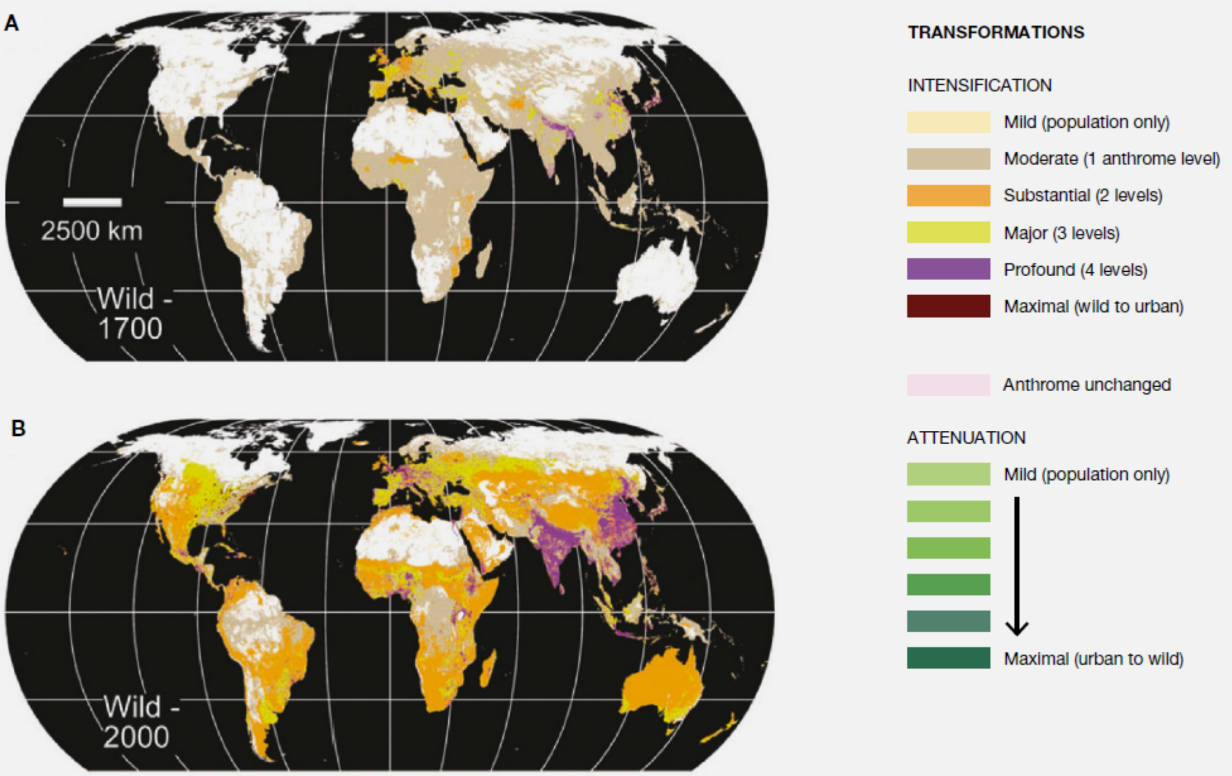
Land-cover changes are pervasive and, when aggregated globally, they may significantly affect basic processes of the global system's functioning (Lambin & Geist, 2006). They encompass the many types of deforestation, conversion of forests, grasslands and drained wetlands to cultivation as well as changes between types of agriculture, such as annual crops, perennial crops, and orchards. Particularly important changes that have strong effects are crop irrigation and urbanization, which often results in creation of large impervious surfaces (see Section 4.3.10). In more subtle ways, degradation can arise from changes in land use, such as salinization (see Section 4.2.2.2) caused by over irrigation, and erosion following deforestation (see Section 4.2.1).

Extent of change

Human alteration of terrestrial ecosystems by hunting, foraging, land clearing, agriculture, and other activities started about 12,000 years ago (UNCCD, 2017). Land-cover change increased dramatically from the start of the industrial era (Ellis *et al.*, 2010; Hurtt *et al.*, 2011) (Figure 4.18). Currently, most land with no anthropogenic pressure is in places that are unsuitable for agriculture, such as deserts. While conservation of all types faces multi-faceted challenges in developing countries, in developed countries there is a positive correlation between increased Human Development Index and decreasing pressure on protected areas (Geldmann *et al.*, 2014).

Figure 4 18 Global patterns of human transformation of land cover.

A Estimated land cover in 1700, before the industrial age; B Land cover in 2000. Colour bar shows the intensity of modification of land cover indicated by the level of anthrome conversion. Source: Ellis *et al.* (2010).



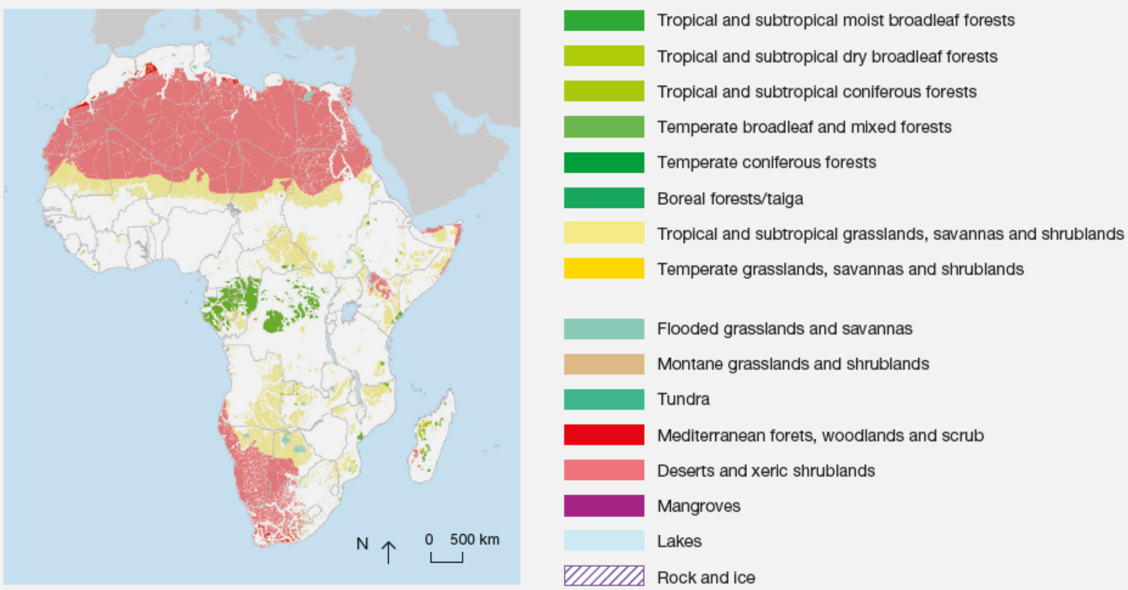
Over the past 300 years, more than 50% of the land surface has been substantively altered by land-use activities, over 25% of forests have been permanently cleared, over 30% of the land surface is occupied by agriculture, and 10–44 10^6 km^2 of land is globally recovering from previous human land-use activities (Hurtt *et al.*, 2006, 2011; Turner *et al.*, 1990; Vitousek *et al.,* 1997; Waring & Running, 2010). As examples: less than 0.1% of tropical deciduous dry forests in Central America's Pacific Coast and less than 8% in Madagascar remain (Laurance, 1999); 10-20% of the world's drylands, which include temperate grasslands, savannas, shrublands, scrub, and deciduous forests, have been somewhat degraded (although there are exceptions such as tallgrass prairies of North America) that have less than 3% of natural habitat remaining; farming and logging have severely disturbed at least 94% of temperate broadleaf forests; more than 50% of wetlands in the USA have been destroyed in just the last 200 years (Erb *et al.*, 2009); and between 60% and 70% of European wetlands have been completely destroyed (Stein *et al.*, 2000). Boreal forests have a relatively short history of large-scale human activity: localized degradation started around 16th century but more recently there has been large-scale logging, initially for tar production and later for shipbuilding, charcoal and so on (Wallenius *et al.*, 2010). Currently, logging for lumber and biomass harvesting for power generation are the most important uses which, together, are now very extensive. For example, in Fennoscandia, more than 90% of the productive forests are under intensive forest management, often at the expense of other ecosystem services (Gamfeldt *et al.*, 2013a; Hansen *et al.*, 2013a). Opportunities for land expansion without damaging forests and natural ecosystems are increasingly limited around the world and future increases in agriculture and grazing systems production will need to come mainly from increases in productivity (Godde *et al.*, 2017).

Pattern of land cover

The removal of native land cover and repurposing of land modified at an earlier date has created an intricate mosaic of land cover and land uses (see Section 4.2.7). Forest loss and conversion of grasslands to cropping are clear cases, but less obvious changes such as in types of crops can be equally significant. The expansion of cultivation into formerly natural vegetation is often along roads (Geist & Lambin, 2002) and around settlements, not along a broad front. The result is fragmentation of the natural land cover which leads to changes in conditions and diversity within the residual patches (see Section 4.2.6.5) (Broadbent *et al.*, 2008; Gascon *et al.*, 2000; Murcia, 1995; Skole & Tucker, 1993). The global extent of this loss has been demonstrated in a map of "the last of the wild" (Figure 4.19) (Sanderson *et al.*, 2002).

Figure 4 19 The Last of the Wild map of Africa.

The colours indicate least influenced (most wild) areas and their natural land cover. Source: Based on Sanderson *et al.* (2002). Image is licensed under a Creative Commons 3.6 Attribution License.



In most forms of cropping, except in subsistence agriculture, there is a trend towards increasingly large areas planted not only to the same species but often of the same genotype. Monocultures have advantages in management, such as more efficient deployment of agricultural machinery, but a result is increased susceptibility to eruptions of pests and diseases that would otherwise be limited by the distance between fields of food species. The decline in the practise of crop rotation, aided by use of fertilizers and pesticides, encourages pests and diseases that can become endemic (Plantegenest *et al.*, 2007).

Rates of change

Human changes in land cover typically take place in short periods of time but, where recovery is allowed, it is generally very slow. For example, in the Mid Atlantic of the USA, where all accessible forest was felled by 100 years ago, the occasional but rare patches that were not felled ("old growth") provide a baseline for comparison. The findings are that the original condition has not been restored even over 100 years. This is a case of permanent degradation in the sense of the IPBES definition (see Section 4.1.2).

Erosion

Loss of vegetation cover can lead to accelerated erosion with related productivity impacts. Erosion has been extensively discussed in Section 4.2.1, and to avoid repletion, we place a reference to that Section here.

Biodiversity loss

When habitat is changed or lost, in addition to the biodiversity lost from the converted land, the smaller areas of original habitat generally support fewer species (see Section 4.3.1), especially for species requiring undisturbed, core habitat. Fragmentation can cause local and even general extinction. Species invasions by non-native plants, animals and diseases may occur more readily in areas exposed by land use and land-cover change, especially in proximity to human settlements (see Section 4.3.7).

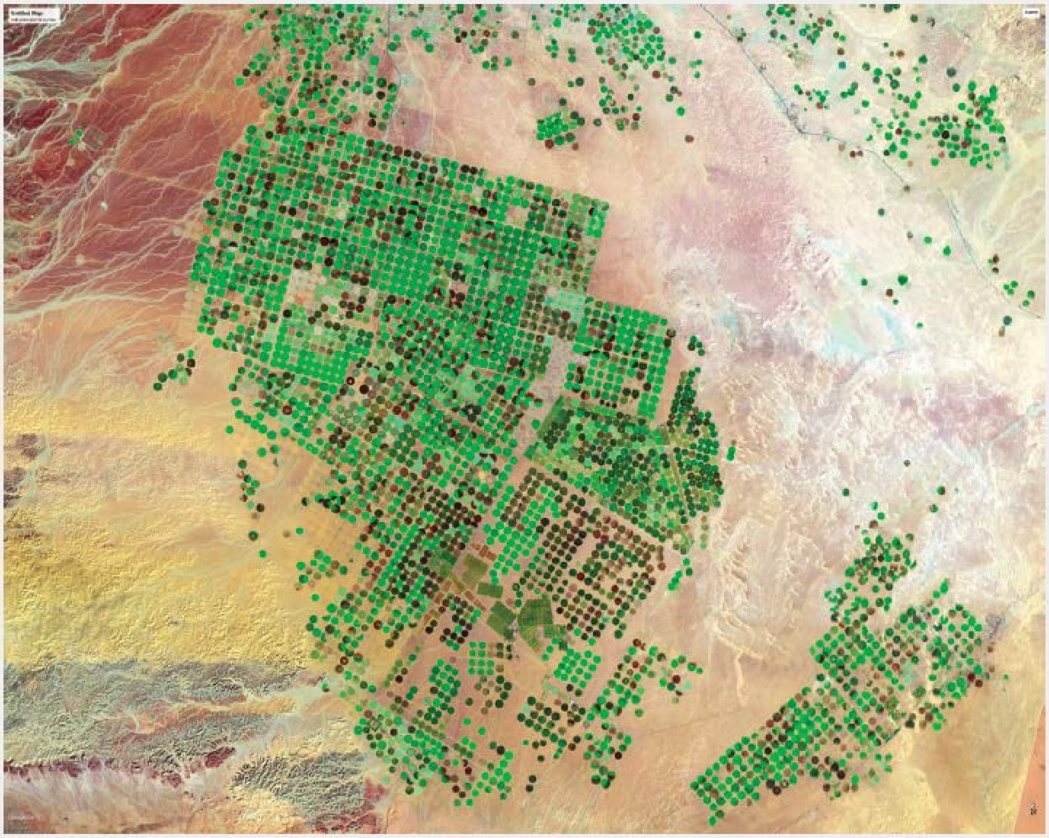
Climate

Land cover has large effects on the atmosphere, influencing climate at local, regional, and global scales (Pielke, 2005). Physical changes of the land surface affect surface albedo, latent and sensible heat exchanges generation of atmospheric aerosols and greenhouse gases (Figure 4.20). The combined effects of these changes have been estimated to cause 40% ± 16% of the human-caused global radiative forcing from 1850 to present day (Wuebbles *et al.*, 2016). However, the complexity and dynamic interplay of land processes and therefore the net effects are currently poorly known. Land cover not only affects climate directly, but itself responds to climate, creating a feedback which can be positive (Nicholson, 2000; Pielke *et al.*, 1998).

Land-cover changes can have multiple, significant effects on the troposphere. For example: dew point temperatures have increased due to a change in land cover to agriculture in USA; warmer temperatures occur in urban versus rural areas (see Section 4.2.8.); regional daily maximum temperatures can be lowered due to forest clearing for agriculture; temperature can increase following regrowth of forests on abandoned agricultural fields; conversion of rain-fed cropland to irrigated agriculture cools temperatures directly over croplands and at great distances (10°C to 32°C in California’s Central Valley), it can increase relative humidity by 9% to 20% and affect precipitation at a regional scale (detected 1,000km away in central USA); urban landscapes can affect the formation of convective storms and change the location and amounts of precipitation compared to pre-urbanization. Figure 4.20 shows a source of a “water island” that has large down-wind effects.

Figure 4 20 Irrigation near Tubarjal in the Nahud Desert, Saudi Arabia.

In this extreme case, the land-cover change to the irrigation forms a "water island" that can have large down-wind effects. Source: Google Earth.



Hydrology

Soil hydrology is strongly influenced by land use and land cover (D'Odorico *et al.*, 2007). The absence of a protective vegetation cover can lead to soil sealing and soil crust formation due to impact of rain drops, which increases run-off. Furthermore, reduced organic matter in the surface (living plants and litter) reduces water holding capacity of the soil, and leads to a wetter land surface and more run-off during rainy periods and to a dryer surface during dry periods. The water holding capacity of soil is especially relevant where rainfall is erratic and the buffering capacity of soils to store water is an important factor. Runoff has major effects in rivers since the rapid changes in run-off, as measured in the river hydrograph, affects erosion and freshwater biota. Over longer periods, land-cover change may amplify or moderate these effects of climate change on water flows and on the risks of flooding and drought.

4.2.6.2 Drylands

Definitions and incidence

The UNCCD (1994) defines drylands as area where the aridity index is less than 0.65. Drylands are globally important, accounting for 41% of the land surfaces (White & Nackoney, 2003) and are home to approximately one third (2 billion) of the global human population, most of which (~90%) is located in developing countries (Safriel *et al.*, 2005). Four subtypes are usually recognized amid drylands: hyper-arid, arid, semiarid, and dry-subhumid, and their boundaries vary depending on the definitions used (Nicholson, 2011; Safriel & Adeel, 2005). Dryland are considered particularly vulnerable to environmental change, with the UNCCD using the term "desertification" to denote land degradation within drylands. Climate change is causing an increase in the global area of drylands, observational data suggesting the area has already increased by 4% since the 1948-1962 period. Estimates suggest that by 2100 the drylands will have increased in spatial extent by 11 to 23%, constituting up to 56% of the global land surface (Huang *et al.*, 2017).

Desertification

Desertification is defined as the loss of biotic productivity in arid, semi-arid and dry sub-humid lands UNCCD (1994); in other words, a form of land degradation specific to the drylands (excluding the hyper-arid areas). The term "desertification" has come to evoke an image of the advancing desert, with grazing and arable lands turning into deserts. There are numerous examples of past cultural declines associated with the spread of desert-like conditions, such as the decline of Saharan civilizations some 3,000-4,000 years ago when the climate changed rapidly, leading to a change from savannah to desert (Nicholson, 2011). The UNCCD (1994) stated that 25% of the Earth's land surface was affected by desertification. It is now realized that desertification is a subtle and complex process at the nexus of people, climate and the environment (Miehe *et al.*, 2010). If defined as permanent loss of productive potential (see Section 4.1.2.1), desertification is not nearly as widespread as previously thought (e.g., Prince, 2002; Prince *et al.*, 1998), but it does exist (Rishmawi & Prince, 2016a)

Part of the sometimes discordant debate about desertification (Behnke & Mortimore, 2016; Thomas & Middleton, 1994b) derives from problems in differentiating desertification from drought which has similar immediate impacts. The 1970s and 1980s droughts in the Sahel highlighted a phenomenon common throughout global drylands where bad management during droughts leads to long term land degradation. A further example is the Dust Bowl days of the 1930s in the Great Plains of the USA, when farmland was ruined and soil was eroded, triggered by some of the worst drought conditions on record in the region. The Dust Bowl days coincided with the expansion of inappropriate agricultural techniques onto marginal lands, related to the high value of wheat (Egan, 2006), and the decline in the number of sheep in New South Wales from 13 million in 1890 to 4-5 million in 1900, associated with a drier period (Graetz, 1991).

Currently, unravelling the processes, consequences, severity and extent of drought versus degradation, even in the iconic and well-studied Sahel region, remains contentious. The maps that show the locations and intensity of desertification have all serious shortcomings since they are based either on subjective assessments by experts, or on unproven methodology, and therefore cannot be applied globally nor used in future for monitoring (Gibbs & Salmon, 2015; Prince, 2016). This problem is partly because a range of distinct environmental processes are often lumped together under the term desertification, e.g., sheet erosion, productivity, loss of palatable species, bush encroachment (Nicholson, 1996; Nicholson *et al.*, 1998; Prince, 2002, 2016). Even when a distinct process is addressed, suitable metrics can be difficult or impossible to apply spatially (Bunning *et al.*, 2011), especially over large areas.

Susceptibility to grazing

Managed grazing of rangeland is globally the single largest land use, covering more than a quarter of the global land surface, and 65% of the drylands, typically in area with marginal bioclimatic and edaphic conditions (U. Safriel & Adeel, 2005). Mismanagement of rangelands, leads to compaction of soils, loss of carrying capacity, erosion, woody encroachment and deforestation (see Section 4.3.2.). This degradation has widespread effects on the vegetation, soils, biogeochemistry, hydrology and biosphere-atmosphere exchange. In combination, they are major causes of global environmental change (Asner *et al.*, 2004). Despite this, some drylands are extremely resistant to long term overgrazing, bouncing back rapidly after droughts (e.g., Hiernaux *et al.*, 2009).

Invasion by weeds and increases of unpalatable species

Expansion of invasive plants (see Section 4.3.7) on drylands has been studied over a long period (Richardson & Pyšek, 2008) and has been attributed to many factors, including traits of the vegetation and physical ecosystem properties. The effects can be catastrophic. In the intermountain west of the USA, for instance, many of the ecosystems that *Bromus tectorum* (cheatgrass) has invaded are seriously altered, and no longer support the vegetation of the potential natural community (Zouhar, 2003). Invasive plant traits may include genetic variation and plasticity that enhance invasion. Also, high seed production and dispersal ensures propagule spread. Once invasive plants become established and spread within a site, the chance of successfully controlling them is greatly reduced and becomes extremely costly over the long term. Therefore, early detection and containment is critical for preventing the introduction, establishment, and spread into new sites. In addition to forbs, invasions of woody species that are toxic to livestock if ingested frequently occur in overgrazed rangelands (see below).

Bush encroachment (woody densification)

Encroachment by bushes and small trees into formerly herbaceous rangeland (bush encroachment, woody densification) dramatically reduces grazing and hence livestock carrying capacity, habitat structure, biodiversity, fire regimes and hydrology (Abrahams *et al.*, 1995; Archer, 2010; Desta & Coppock, 2002; Safriel, 2009; Scholes & Hall, 1996). In extreme cases, this can reduce grazer carrying capacity by up to 90% (de Klerk, 2004). It has been estimated that increases in woody cover affects 10-20% of rangelands worldwide (Reynolds *et al.*, 2007) and 335 million ha (40%) of the United States (Pacala *et al.*, 2001). Densification has been reported globally (Archer, 1995; Archer *et al.*, 1995; Asner & Heidebrecht, 2003; Britz & Ward, 2007; Fensham & Fairfax, 2005; Skarpe, 1990; Van Auken, 2000; Wigley *et al.*, 2010) and has been estimated to be expanding at between 0.5% and 2% worldwide per year (Archer *et al.*, 1995). Even though woody densification does not reduce primary production, it meets the IPBES definitions of degradation through long term reductions in some ecosystem services and biodiversity. Woody densification has mixed impacts on carbon stocks and results are inconclusive. In the southwestern USA, in semi-arid and subhumid regions of >336 mm rainfall, encroachment has been shown to increase above-ground carbon sequestration by 0.7 g C m^{-2} $yr^{-1}mm^{-1}$ rainfall and soil organic carbon gains averaged 385 g C m^{-2} yr^{-1}. In arid regions (< 336 mm), there were decreases in both above and below ground of 6,200 g C m^{-2} (Barger *et al.*, 2011). Jackson *et al.* (2002) reported the opposite with moist sites losing soil organic carbon but this being offset by the above ground carbon gains.

The process of woody densification is not fully understood, but likely causes are heavy grazing that leads to loss of grass cover and reduces fires, reducing the competition for woody plants to establish. In addition, there is also growing evidence that densification may be facilitated by increased atmospheric CO_2 fertilisation effects, which benefits C_3 tree growth more than that of C_4 grasses (Archer *et al.*, 1995, 2001; Bond & Midgley, 2000; Higgins & Scheiter, 2012; Kgope *et al.*, 2009; Macinnis-Ng *et al.*, 2011; Midgley & Bond, 2015).

Sahel desertification case study

From 1968 to 1974 and again in the early to mid-1980s, severe famines struck the Sahel – the strip of land bordering the Sahara Desert that extends approximately 5,000km from Somalia in the east to Senegal in the west and 500km from the desert to humid regions to the south. There are many estimates of the effects of these and subsequent famines on the human population (Thomas & Middleton, 1994; UNCED, 1992; WFP, 2012) including decimation of livestock, failure of crops, mass migration to refugee camps and urban areas, epidemics, starvation and lengthy dependence on food aid (Mortimore & Adams, 2001). The severity of the disaster shocked the world and ultimately vast relief campaigns were mounted followed by many development programs.

In tandem with the international outpouring of concern and funds, environmentalists began to suspect a progressive southerly movement of the Sahara Desert was in progress, along the entire length of the Sahel. Evidence was drawn from many sources, some of which were anecdotal (Thomas & Middleton, 1994b). Causes were mostly attributed to over-stocking of livestock and over-cultivation of the land during a drought, leading to bare ground which, in turn, set in motion a positive feedback of reduced rainfall leading to further loss of vegetation (Nicholson, 2000; Pielke *et al.*, 1998).

Such was the level of alarm that the United Nations Conference on Desertification was convened in 1977, ultimately leading to the present UN Convention to Combat Desertification (UNCCD), a legally-binding agreement with 196 national signatories. The term “desertification” entered the popular vocabulary, fed by images of undernourished farmers standing in landscapes of bare ground, suggesting crop failure and reduced capacity for livestock. A search on the internet for “desertification” yields many such pictures (e.g., WFP, 2012) which continues to fuel the popular imagination.

In the early 1980s, data from sensors on earth-orbiting satellites that could measure the amount of vegetation on the ground, crops, natural woodlands and herbaceous cover for the entire world, with approximately 9-day repetition, became available (Herrmann & Sop, 2016) (Figure 4.21). Later field studies linked vegetation at the satellite and field scales (Dardel *et al.*, 2014). By the mid-1980s an inter-annual time-series began to accumulate (Figure 4.22), to which archived data starting in 1981 were added. Analyses of the relatively short time-series did not support the notion of progressive southerly movement of the desert; in fact, the location of the boundary of measurable vegetation varied from year to year, some years to the south and other years shifting to the north (Tucker & Nicholson, 1999).

By 2000 enough data were available to detect longer-term trends which showed that from the late 1980s there was not a progressive southerly movement of the desert but rather a gradual increase in vegetation, in lock-step with a gradual increase in rainfall (Figures 4.21-4.24), leading to the conclusion that a “greening of the Sahel” was occurring (Dardel *et al.*, 2014) . It had not been “desertification”, but rather a drought (see Section 4.1.2). Thus the Sahel fell from being the icon of desertification to an example of the response of dryland vegetation to rainfall, although Alexander von Humboldt recognized this distinction in 1878 (Gritzner, 1981). There are a few examples of restoration actions that have increased greenness above what would be expected with the higher rainfall (e.g., Herrmann & Sop, 2016). This is not to say that “desertification” in the sense of the current UNCCD definition does not exist in the Sahel, just that it is localized and not sub-continental in scale (Figure 4.24).

Figure 4 21 Productivity (net primary production) images for one month in the Sahel growing season (September) in 2015 (dry year, above) and 2004 (wet year, below).

Rainfall values are deviation from the 1980-2009 June to October average in mm. Rainfall data from Joint Institute for the Study of the Atmosphere and Ocean (JISAO). Source: Janowiak (2015). Images from NASA Earth Observatory (2018).

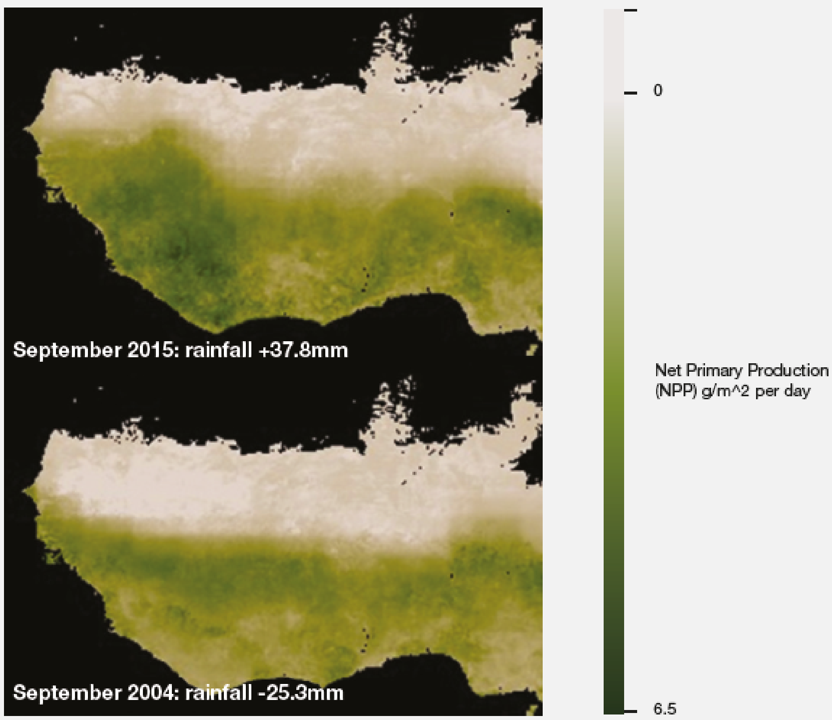


Figure 4 22 Temporal profiles of A field observations of herbaceous mass and B GIMMS-3g normalized difference vegetation index (NDVI) over the Gourma region of Mali.

Panel b) shows the normalized difference vegetation index GIMMS-3g for the exact same years when field observations are available (in orange) and for all years when normalized difference vegetation index data are available (1981-2011, in purple). Source: Dardel *et al.* (2014).

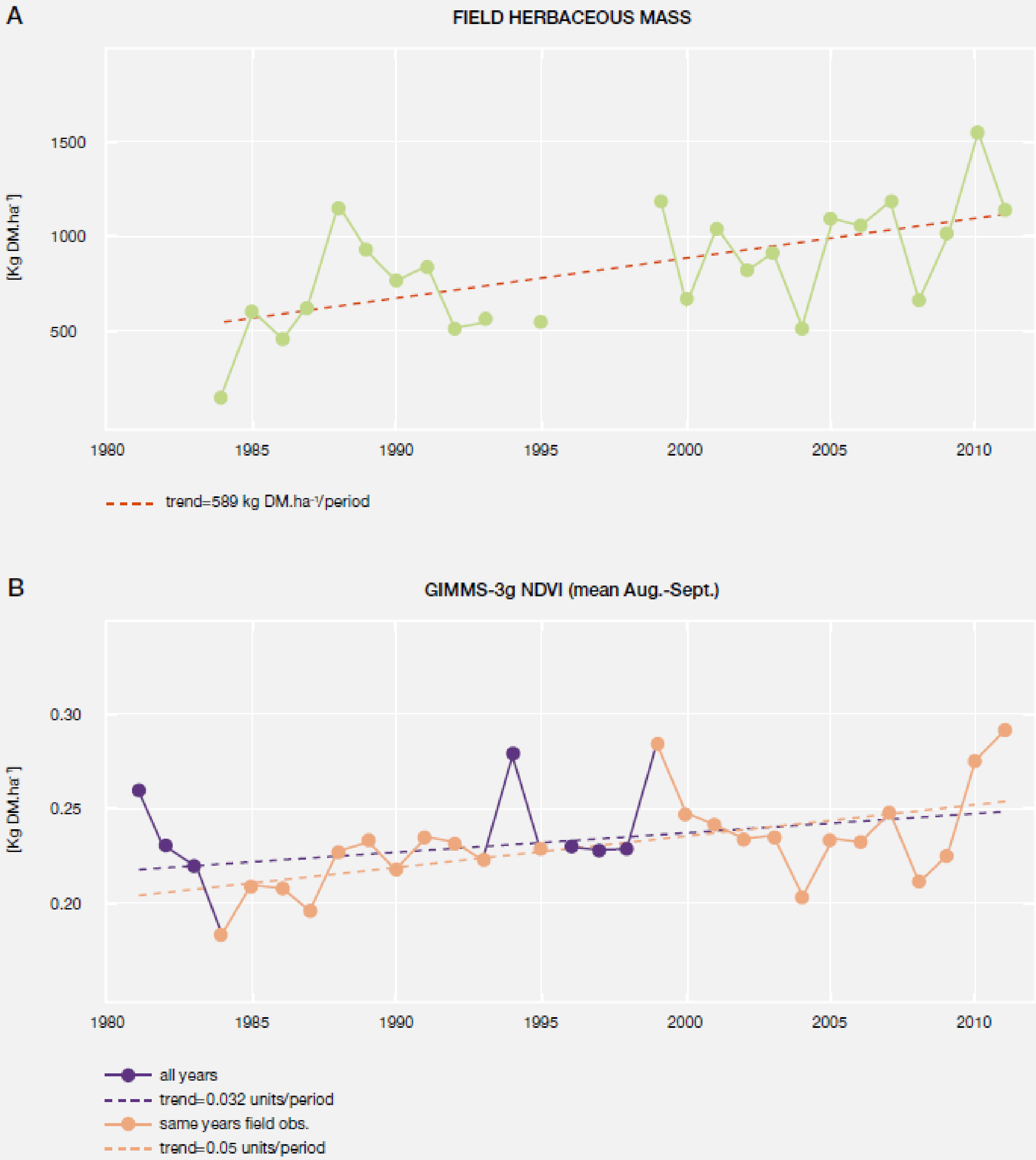


Figure 4 23 Sahel precipitation June–October from 1900 to 2011 shown as anomalies (deviations) from the mean of all dates.

Data from National Oceanic and Atmospheric Administration Global Historical Climatology Network gridded rain gauge precipitation anomalies for 10°-20°N and 20°W-10°E and National Oceanic and Atmospheric Administration AVHRR normalized difference vegetation index anomalies for the same region from 1982 to 2011 for the three decades of overlap. Pearson's linear correlation coefficient: 0.82. Source: Precipitation data from Janowiak (2015); NDVI data and statistics from Herrmann & Sop (2016).

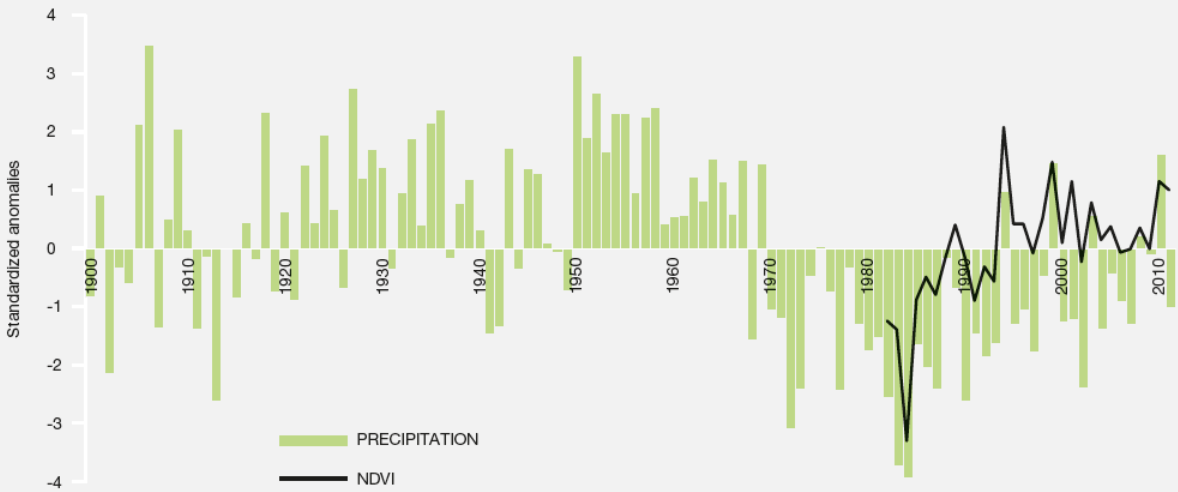
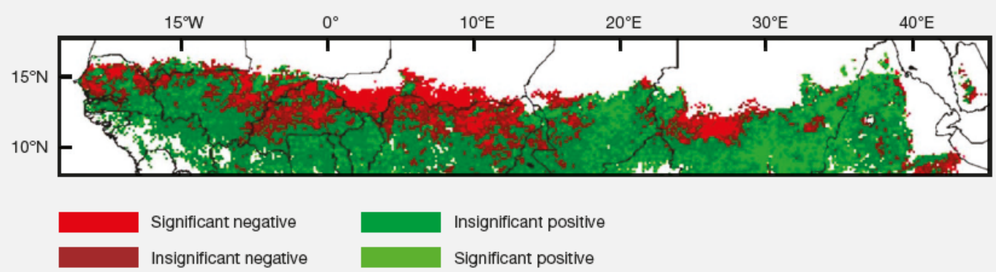


Figure 4 24 Potential areas where anthropogenic degradation may be in progress in the Sahel (shown in red).

Map derived from residual trend (RESTREND) analyses from 1982 to 2006. Source: Rishmawi & Prince (2016).



4.2.6.3 Fire and associated degradation

The major effect of fire is the reduction in vegetation cover, able to remove 80% of above-ground net primary production (Bond & van Wilgen, 1996; Bond & Keane, 2001). During this process fire plays a major role in the cycling of nutrients (Zavala *et al.*, 2014). Fire is both natural and critical to many ecosystems (Whelan, 1995). For example, infrequent, intense fires can trigger the release of seeds from the fruits of some fire-adapted species (e.g., serotinous pine, *Hakea* and *Protea* cones), thereby timing regeneration to a period of reduced competition by established vegetation and placing them in an ash bed that favours successful establishment (Bradstock *et al.*, 1994; Johnson & Gutsell, 1993; van Wilgen & Viviers, 1985). In contrast, frequent, smaller events can kill saplings. Suppression of fire can lead to unnatural changes in vegetation, but is also often responsible for the intense fires that occur when the area eventually burns.

It is important to separate natural fires regimes from fire impacts that can be considered as degradation. Vegetation types react differently to fires, with tropical grasslands and savanna as well as some

Mediterranean climate vegetation being both fire adapted and fire dependent (Bond & van Wilgen, 1996; Bond *et al.*, 2004; Bond & Keane, 2001; Head, 1989; Zavala *et al.*, 2014). Many forest types are fire adapted or fire dependent, though fire return times may be as long as 300 years or more; tropical rain forests by contrast seldom, if ever, experience natural fires. The extensive dry forests of Africa, the miombo, experience fires every few years (Frost, 1996) whilst boreal forest may only burn occasionally. Fires occurring in their natural frequencies are clearly not a form of degradation, in fact they are required to maintain the natural biodiversity and functioning of the ecosystem; however, changes in fire frequency, intensity or season (see Section 4.3.6) can have major impacts on the resultant vegetation and the provisioning of ecosystem services. The impacts of fires vary depending on the intensity and seasonal timing. For example, low intensity, smouldering fires are beneficial in the wetland ecosystems in the Big Cypress Preserve in Florida (Watts *et al.*, 2015).

Differentiating between fires impacts on the biodiversity of natural vegetation versus the impacts that fire can have on the flow of ecosystem services is also important. For instance, managed or plantation forests that are being maintained specifically for their provision of wood products can be totally destroyed by fire, with high financial loss to humans. Fire can also have devastating impacts on human habitation, livestock and infrastructure – ironically, often as a consequence of supressing fires in fire-prone regions, which allows for unnatural build-up of flammable material.

Fires create a landscape where young forest cohorts are overrepresented compared to natural forests (Bergeron *et al.*, 2001). In North America, intensive logging has changed the whole landscape structure (Cyr *et al.*, 2009). On the other hand, abandonment of Soviet era agricultural land has caused quite extensive reforestation that partly counteracts forest losses due to fire (Prishchepov *et al.*, 2013).

Fires can induce change in physical, chemical and biological properties of soils, which can last from days to decades. The severity of the impact on soils is a function of many variables including fire intensity (energy release rate), moisture content, humus layer and duration. Typically only a small proportion of thermal energy enters the ground, and seldom effects more than the top few centimetres (Certini, 2005; Zavala *et al.*, 2014), though this can have substantive impacts on aspects such as permeability and the release of nutrients. Fires can reduce water infiltration (DeBano, 2000) depending on the temperature of the fire, the type of vegetation, soil organic matter and soil type (Fox *et al.*, 2007; Zavala *et al.*, 2010), leading to enhanced overland flow and erosion (Shakesby, 2011).

Fires affect more than ecosystem biomass. Crutzen *et al.* (1979) found that the production of trace gases by tropical forest fires influences atmospheric chemistry and biogeochemical cycles. For example, during Indonesia's widespread fires in 2015, the resulting air pollution was so extensive and intense that schools closed, air travel was banned, airports were closed and states of emergency were imposed in neighbouring Southeast Asian countries, including Malaysia and Singapore.

Weather fluctuations cause large differences in inter-annual fire frequency. Between 2001 and 2007, the average area of fires in Canada was 5,930 ha and 1,312 ha in Russia (de Groot *et al.*, 2013), but Russia has the most extensive overall forest loss (Hansen *et al.*, 2013b). In Western Russia alone, 1.5% of forest cover was lost from 2000 to 2005 (Potapov *et al.*, 2011). In north western USA and Canada the combination of large bark beetle outbreaks and subsequent fires are comparable in extent (Bentz *et al.*, 2010; Hansen *et al.*, 2016; Simard *et al.*, 2011).

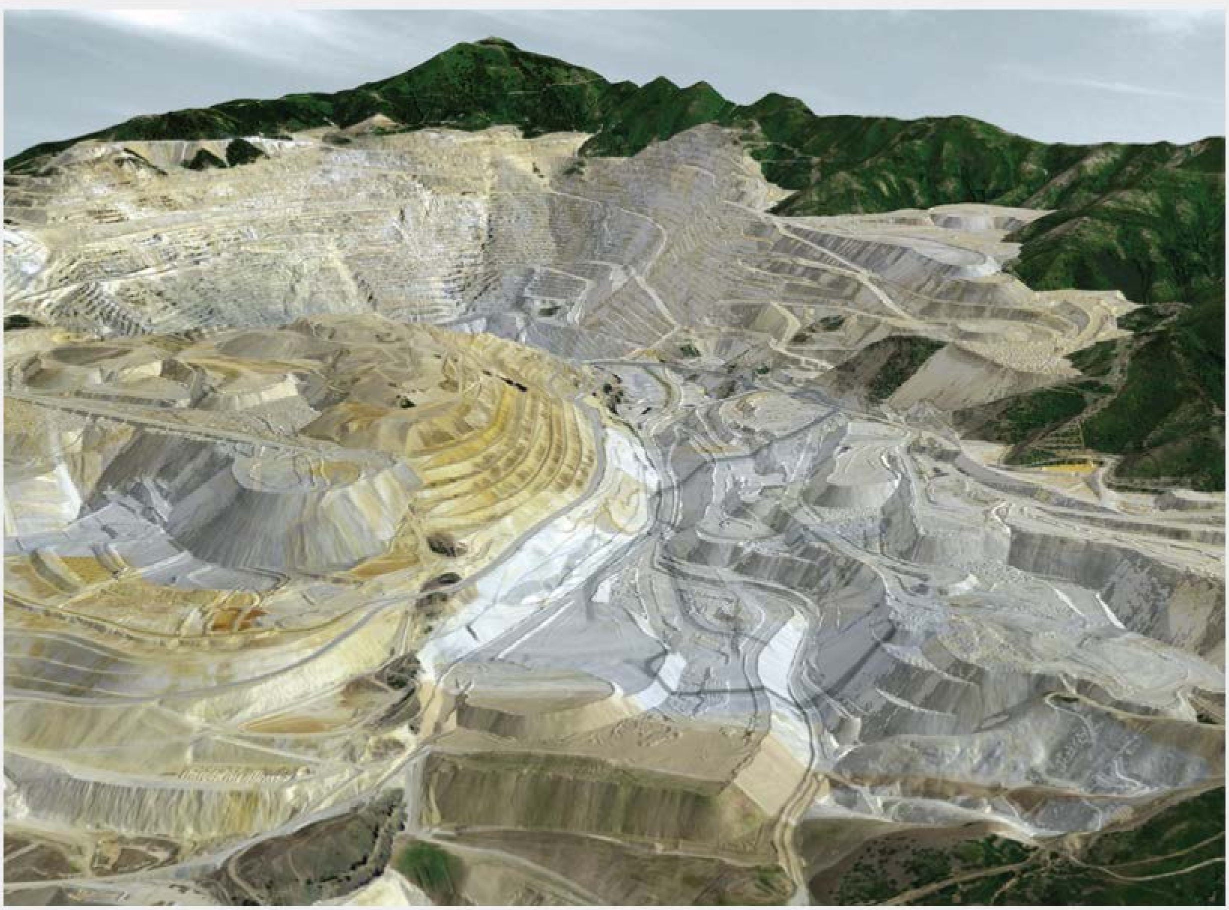
As discussed in Section 4.2.8, climate change is anticipated to have major, but uncertain, impacts on fire regimes.

4.2.6.4 Disruption of topography

Human activities have had dramatic effects on the topography of the Earth (Tarolli *et al.*, 2017) (Figure 4.25.), even initiating earthquakes. For example, it has been estimated that mountaintop removal and valley fills in the US Appalachians are responsible for burying and polluting more than 3,200 km of headwater streams (EPA, 2011). In many areas, excavation and earth-moving have changed flood patterns, created barriers to runoff and erosion, funnelled sediments into new deposition areas, created unstable spoil heaps, and dredged sediment from water bodies to create new land with consequent starving of existing beaches.

Subsidence, sometimes over vast areas, can be induced by reduction of over-burden for open-cast mining, drainage of organic soils, and human induced thawing permafrost. For example, over 44,000 square kilometres in the United States have been directly affected by subsidence, with over 80% the result of groundwater extraction (Galloway *et al.*, 1999). Mining, in particular, can have sudden catastrophic effects through fracture below ground structures. Coal mining (Loupasakis *et al.*, 2014), oil wells (Frohlich *et al.*, 2016) and hydraulic fracturing for gas extraction (Ellsworth, 2013) often initiate ground movements.

Figure 4 25 Three dimensional view of Bingham Canyon Mine, showing the extent of a human-made topographic feature. Source: Tarolli *et al.* (2017).



4.2.6.5 Landscape-scale degradation

Large, diverse areas of land are more than a collection of individual cover types, each type with its individual set of characteristic processes. More than 50% of the ice-free Earth surface has been completely modified or replaced by human activities and much of the remaining semi-natural areas are also highly modified, not only changing land cover but also creating new mosaics of original or novel land-cover types (Figure 4.26). The members of the mixture or mosaic of cover types generally interact, resulting in properties that are distinct from any of the individual component cover types. These interactions result in emergent properties, in addition to those of the individual landscape components. Degrading the landscape can cause thresholds to be exceeded, causing abrupt changes in landscape processes that are often irreversible and beyond which unexpected changes occur (Hanski & Ovaskainen, 2000) (also see Section 4.1.2.) which can lead to catastrophic shifts in land cover and functions (Scheffer *et al.*, 2001) (also see Section 4.2.6.2). These relationships, however, are poorly understood. The landscape scale is a critical component of the links between local and global scales.

Figure 4 26 An aerial view of a landscape mosaic of mostly human-created patches around the village of Glenridding, and the southern part of Ullswater in the Lake District National Park, Cumbria, North West England.

Patches consist of habitation, fields, secondary forest, deforested mountains, mountain footpaths, natural erosion, recreational boating and some forest plantations in the background. Photo: courtesy of David Iliff. License: CC-BY-SA 3.0.

The properties and processes in landscapes can be considered under four headings: (1) composition; (2) spatial configuration; (3) connectivity; and (4) disturbance. These are described further below.

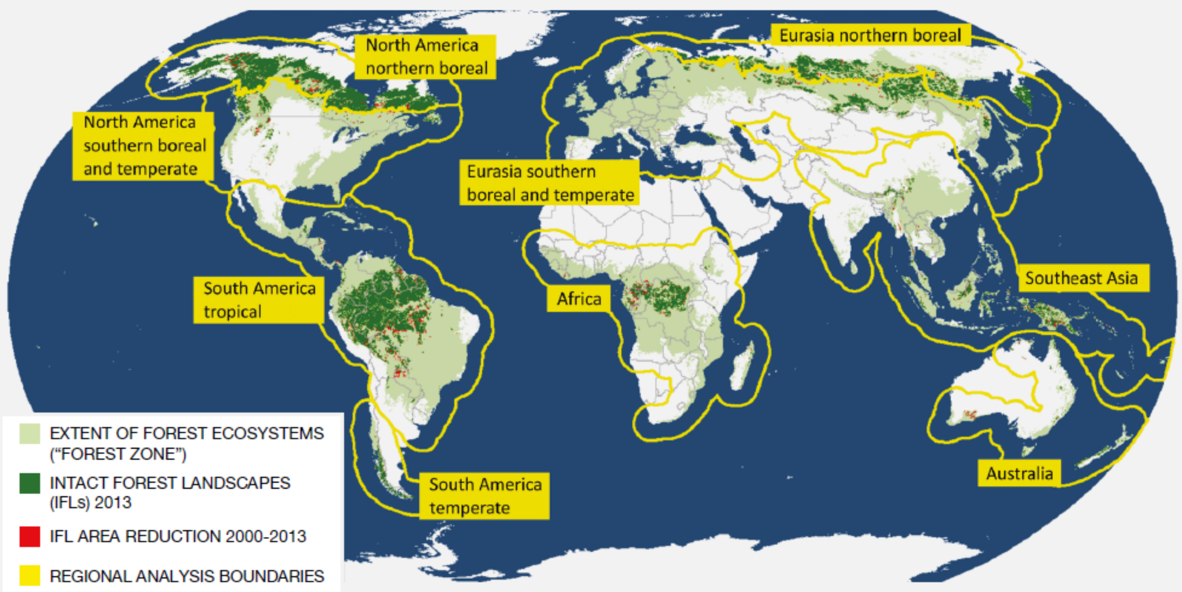
Composition

Degradation usually changes the landscape composition, often involving a reduction in native land-cover patches and an increase in human-dominated land uses (see Section 4.3.1). Where cover types are dependent on one another for their fundamental processes, the loss or degradation of one can have cascading effects on others. Many bird species feed and nest in different locations; degradation of one of these will cause a loss of the bird from other as well (Cornelius *et al.*, 2000).

Spatial configuration

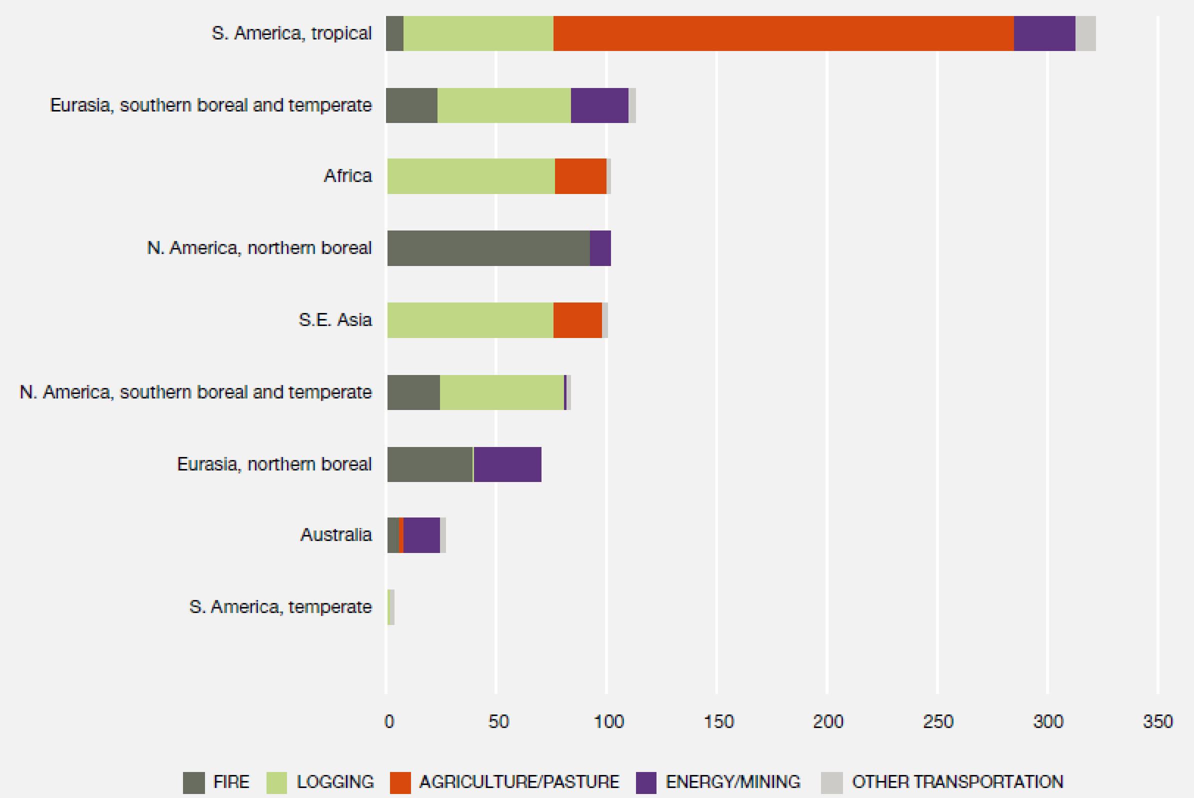
An individual land-cover type in a landscape can form one patch or many fragmented patches and the patches themselves can have complex shapes and, therefore, boundaries between them and the other land-cover types. The degree of fragmentation has important effects on biodiversity and some aspects of the physical environment. Many species have minimum territory sizes and, while the total area of their habitat may be adequate, if it is fragmented the intact habitat types may be inadequate, unless the species can cross the intervening land-cover types. Fragmentation also creates a greater length of boundaries or edges between different habitat patches. Edges typically are different from the interior of a patch (Batáry *et al.*, 2014), affecting microclimate, species presence and other factors such as water drainage (Collinge, 1996; Haddad *et al.*, 2015; Laurance *et al.*, 2007). Fragmentation is pervasive in heavily-altered landscapes – 30% of the EU's territory is highly fragmented (Jongman, 2002; Tillmann, 2005).

Figure 4 27 Global intact forest landscapes (IFL) extent in 2013 and reduction from 2000 to 2013. Source: Potapov *et al.* (2017). http://www.intactforests.org/



A global survey of fragmentation of land cover (Potapov *et al.*, 2017a) measured the area of intact forest landscape, defined as land in a seamless mosaic of ecosystems with no signs of human activity and a minimum area of 500 km^2 (Figure 4.27). It was found that IFLs comprise only 20% of tropical forest area. Only 12% of global intact forest landscapes are protected (Potapov *et al.*, 2017a). Globally, the average rate of reduction in intact forest area over 14 years was 7.2% with an extreme of 80% (Figure 4.28). The International Union for Conservation of Nature Forest protection activities slowed the reduction of intact forest landscape area from timber harvesting, but was less effective in limiting agricultural expansion, while, in the Congo Basin, the certification of logging concessions under responsible management had a negligible impact on slowing intact forest landscape fragmentation (Potapov *et al.*, 2017a). The causes of the declining intact forest landscape include logging, fire and conversion to agriculture, but with large differences between regions (Figure 4.28).

Figure 4 28 Regional reduction of intact forest landscape (IFL) area (km²x 10³) and causes of change. Source: Potapov *et al.* (2017). http://www.intactforests.org



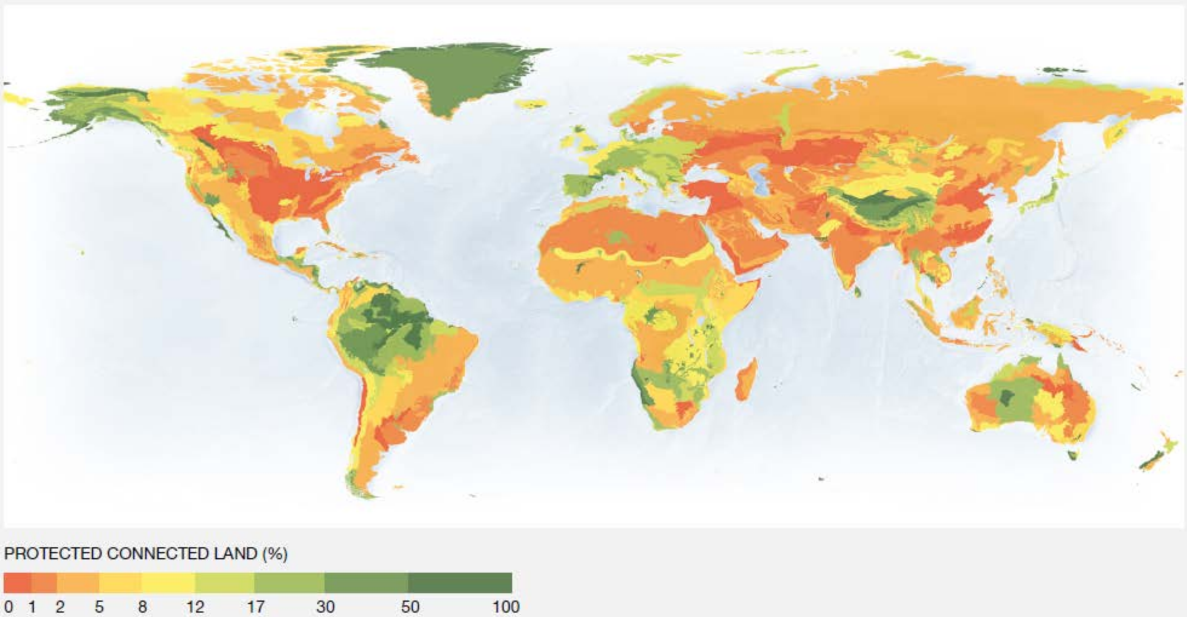
Based on a review of seven large-scale fragmentation experiments, running in five continents through the last 35 years, Haddad *et al.* (2015) estimated that habitat fragmentation reduces biodiversity by 13% to 75%. The ecosystem services that depend on native species, such as pollination, pest control or diseases regulation decline in their turn (Mitchell *et al.*, 2015). As a consequence, agricultural productivity can be significantly reduced (IPBES, 2016).

Fragmentation increases the edges where two land-cover types abut and thereby expands the area of the original land cover that is affected. For example, micro-climate and altered disturbances regimes in forest edges (Laurance *et al.*, 1997, 2002) tend to accelerate the loss of biomass that would occur due only to deforestation (Pütz *et al.*, 2014). Edge-effects include changes that extend into the original land-cover patches such as microclimate and the propagation of fires from grassland into what was continuous forest. Edges have been found to contribute 31% more carbon emissions and thus are large enough to affect the global carbon balance (Brinck *et al.*, 2017). The expansion of agricultural activities in the Amazon intensifies forest fire regime and drought, which in turn accelerate forest degradation and loss, creating a positive feedback (Nepstad *et al.*, 2008). Only 9.3% of global natural vegetation has at least a 10km buffer of functionally-connected land and only a third of the 827 terrestrial ecoregions meets the Aichi Biodiversity Target of 17% of well-connected protected areas (Figure 4.27.) (Saura *et al.*, 2017). Haddad *et al.* (2015) estimated that ca. 20% of the world's remaining forest is within 100m and 70% is within 1km of a forest edge. In some regions, such as the Brazilian Atlantic Forest, less than 10% of the remaining habitat is more than 1km away from human occupations (Ribeiro *et al.*, 2009). Similar patterns can be expected for all areas affected by agricultural expansion.

Connectivity

Most species are confined to a specific range of habitat and may be unable to cross disturbed areas such as cultivation and roads that divide their habitats (Bélisle *et al.*, 2001; Taylor *et al.*, 1993). Where connectivity is not enough to allow recolonization at a sufficient level to compensate local species extinction, these species are lost, even though some of their habitat remains (Tscharntke *et al.*, 2008). However, although fragmentation has often been shown to be important, the quality of the remaining habitat is also a factor (see Section 4.2.6.1). In a study of 19,432 vertebrate species worldwide, the quality of the remaining habitat was more important than fragmentation (Betts *et al.*, 2017). Connectivity affects the functioning of the landscape as well as the species present (Tischendorf & Fahrig, 2000).

Figure 4 29 Percentage of protected areas connected by at least a distance of 10 km. Source: Saura *et al.* (2017).



Disturbance

Extensive land-cover conversion generally leads to a narrower range of ecosystem types and these tend to be less resilient and more subject to catastrophic shifts in their state under stress (Scheffer *et al.*, 2001). This can have spill-over effect on agricultural productivity by significantly reducing regulation services, as recently shown by the IPBES assessment on pollination (IPBES, 2016), which estimates 35% of global crop production depends on pollination, representing an annual market value of $235 billion-$577 billion worldwide.

The change in landscape often leads to replacement of climax species by generalist species (Banks-Leite *et al.*, 2014). This pattern was observed for different groups of vertebrates and plants in tropical forests (Atlantic forest, Amazonia) (Lima & Mariano-Neto, 2014; Martensen *et al.*, 2012; Ochoa-Quintero *et al.*, 2015; Pardini *et al.*, 2010; Rigueira *et al.*, 2013), forest-savannahs ecotones (Muylaert *et al.*, 2016) and for different types of vegetation in temperate region (Canada, Australia, USA) (Maron *et al.*, 2012; Radford *et al.*, 2005; Richmond *et al.*, 2015; Yeager *et al.*, 2016) with thresholds varying from 50% to 20% of remaining native habitat.

Another emergent property occurring at the landscape-level is interactions of degradation processes, sometimes leading to cascades of multiple landscape components. For example, habitat degradation at forest edges can result in synergetic process that lead to additional forest loss and carbon depletion (Pütz *et al.*, 2014). Habitat loss and fragmentation might lead to retrogressive succession, particularly when habitat cover is below the threshold (Rocha-Santos *et al.*, 2016). An interaction of insect outbreaks and fire has been noted in subalpine forests under moderate burning conditions, in which the severity of bark beetle outbreaks affects fire severity and then post-fire tree regeneration (Harvey *et al.*, 2014).

Anthropogenic landscape-level disturbances include transportation, mining, cropping, livestock production, logging and fire (Aragão & Shimabukuro, 2010; Archibald *et al.*, 2013; Potapov *et al.*, 2017a). Other causes are flooding (Kingsford & Kingsford, 2000), pest eruptions in agricultural and forest landscapes (Wermelinger, 2004) and disease outbreaks in human dominated landscapes (Reisen, 2010). Depending on the intensity and frequency of those disturbances, different landscapes can emerge. If the new regime imposed by human activities is characterized by frequent or intense disturbances, then landscapes will become more dynamic, less stable and dominated by disturbed or early-successional ecosystems (Turner *et al.*, 1993).

4.2.7 Pests and diseases

Human diseases

Infectious diseases are a product of the pathogen, vector, host, and environment. Thus, understanding the nature of epidemic and endemic diseases and emerging pathogens is essentially a study of the population biology of these three types of organisms, as well as of environmental factors. In a meta-analysis of 1,415 species of infectious organism known to be pathogenic to humans Taylor, Latham and Woolhouse (2001) found 217 viruses and prions, 538 bacteria and rickettsia, 307 fungi, 66 protozoa and 287 helminths. 61% were zoonotic (a disease that normally exists in animals but can infect humans). The major vector-borne diseases are focused in the tropics.

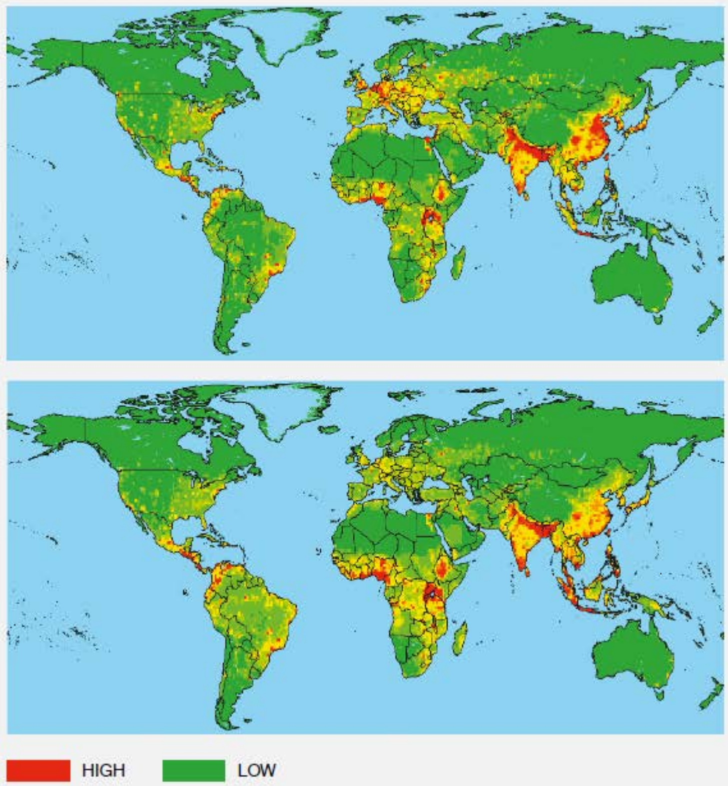
Most emerging diseases are driven by human activities that modify the environment or spread pathogens into new ecological niches (Table 4.10). The magnitude and direction of altered disease incidence due to anthropogenic disturbance differ globally and between ecosystems (Figure 4.30). Biophysical drivers that especially affect infectious disease risk are shown in Table 4.11 (Patz & Confalonieri, 2005).

Table 4.10 Principal drivers of increases in risk of infectious disease.

1.	Altered habitats or breeding sites for disease, destruction of or encroachment into wildlife habitat, uncontrolled urbanization or urban sprawl, deforestation, leading to changes in the number of vector breeding sites or reservoir
2.	Host distribution
3.	Increased contact of humans with natural ecosystems containing pathogens and their vectors increases the risk of human infections, particularly zoonotic pathogens (Jones *et al.*, 2008), that is those transmitted between humans and animals. Poor water supply and sewerage systems leading to cholera-type epidemics
4.	Hydrological modifications such as dam construction and irrigation which provide habitat, for intermediate host species and breeding habitats for vectors

5.	Agricultural land-use changes, including livestock raising and cropping; use of sub-therapeutic doses of antibiotics
6.	Climate change
7.	International travel and trade; and either accidental or intentional human introduction of pathogens

Figure 4 30 **Global distribution of relative risk of an emerging infectious disease. Source: Jones *et al.* (2008).**



The biophysical mechanisms that drive increases in human diseases are largely related to changes in land use (see Sections 4.2.6 and 4.3.1). Intact ecosystems play an important role in regulating the transmission of many infectious diseases. There is evidence that habitat fragmentation (see Section 4.2.6.5) increases the prevalence of the many diseases. Intact ecosystems maintain a diversity of species in equilibrium and, if degraded, may no longer regulate disease organisms or their vectors. Reduced predation of potentially disease-causing agents by increasing transmission, invasion or maintenance, is an obvious example (Table 4.11). However, there are cases where natural systems are a source of pathogens, and destruction sometimes reduces the prevalence of a disease (see also Chapter 5, Section 5.4).

Table 4.11 Biophysical mechanisms that may lead to increases in disease transmission in different types of ecosystems (Patz & Confalonieri, 2005).

Mechanisms	Cultivated Systems	Dryland Systems	Forest Systems	Urban Systems	Coastal Systems
Habitat alteration	Schistomiasis, Japanese encephalitis, malaria	Hantavirus, Rift Valley fever, meningitis	Malaria, arboviruses (e.g., yellow fever), onchocerciasis	Lymphatic filariasis, Dengue fever, malaria	Cholera
Niche invasion or host transfer	Nipah virus BSE (mad cow), SARS, influenza		HIV (initially)	Leishmaniasis	
Biodiversity change	Leishmaniasis	Onchocercasis	Rabies, onchocerciasis	Lyme disease	
Human-driven genetic changes	Antibiotic-resistant bacteria		Chagas disease	Chagas disease	
Environmental contamination of infections agents	Cryptosporidiosis, leptospirosis			Leptospirosis	Diarrheal diseases

Pests and diseases of crops and ecosystems

Since the beginnings of agriculture about 10,000 years ago, growers have had to compete with harmful organisms – animal pests (insects, mites, nematodes, rodents, slugs and snails, birds), plant pathogens (viruses, bacteria, fungi, chromista) and weeds (i.e., competitive plants) – collectively called pests for crop products grown for human use and consumption. Annual losses of crops caused by pests and diseases are estimated at about 20% to 40 % globally (Oerke, 2006) with about 15% to 26 % attributed to insect pests (Culliney, 2014).

Forests are particularly susceptible to insect pests (van Lierop *et al.*, 2015a). Temperate forests account for the largest area of forest damaged by insect pests leading to massive die-backs and disturbance. Dale *et al.* (2001) found that in the temperate forests of North America insect pests and diseases affected annually almost 50 times as much forest as burning (Jones *et al.*, 2008). Most global climate change scenarios favour an increase in incidence of outbreaks in temperate forests in the future (Logan *et al.*, 2003), especially of bark beetles (Hicke *et al.*, 2012).

4.2.8 Climate Change impacts

It has been established with high certainty, that the main cause of climate change is anthropogenic (IPCC, 2007) (see also Chapter 3, Section 3.4). Global and regional climates have experienced shifts and new conditions have arisen, driven by unprecedentedly high atmospheric concentrations of greenhouse gases (GHG), combined with the effects of land-cover changes. Since 1980, the rates of warming of the land have averaged about 0.03°C per year and, in the Northern Hemisphere and the three decades from 1983 to 2012 were the warmest of the last 1,400 years (Yang *et al.*, 2017; IPCC, 2013). Cultivation of crops, livestock management, deforestation and other land-use changes are substantial contributors of human-induce GHG emissions, accounting for 24% of 2010 global GHG emission (Field *et al.*, 2014). Land conversion contributes to climate change as croplands tend to store and sequester less carbon than the ecosystems being replaced. Each year, land conversion results in emissions of approximately one billion metric tonnes of carbon (1 PgC yr^{-1}), some 10% of emissions from all human activities (Friedlingstein *et al.*, 2010). The knowledge of the consequences of climate change is expanding rapidly. The effects are multifarious and include physical environment, biota and humans.

The physical environment

The effects of climate changes on the physical environment have been more rapid and severe than expected. The negative impacts far outweigh the positive. These include increases in occurrence of high temperatures, increased frequency and severity of storms and other extreme weather conditions (Coumou & Rahmstorf, 2012); increased fire frequency (see Section 4.2.8); and longer periods of drought (see Section 4.2.6.1). Many types of anthropogenic degradation will increase. These include: water and wind erosion (Cui & Graf, 2009; Ravi *et al.*, 2010) (see Section 4.2.1); higher temperatures and increased use of irrigation with its consequent effects on fertility (see Section 4.3.3); exacerbated effects of clearance of tropical forests (see Section 4.3.4); land loss by inundation of wetlands due to sea-level rise (see Section 4.2.5.2). Directional declines in rainfall amounts over time, as has been observed over large parts of Amazonia, can reduce greenness, terrestrial water storage, ecosystem productivity and carbon uptake, and alter fire risk, with cascading implications for global carbon cycling and climate (Barbosa *et al.*, 2015; Hilker *et al.*, 2014; Malhi *et al.*, 2008; Meir & Woodward, 2010; Phillips *et al.*, 2009). All of these can have effects at higher trophic levels; for example, increasing dry season lengths have been linked to decreased population growth and viability in birds (Brawn *et al.*, 2016).

Animals, insects

Climate change is affecting the phenology of many organisms. Warming impacts the rate and timing of the development of many ectothermic organisms, favouring some by lengthening the season and increasing the number of reproductive cycles (Peñuelas *et al.*, 2013), while exposing others to disruption of development (Van Dyck *et al.*, 2015). Insects are particularly vulnerable (Bale *et al.*, 2002). Warming tends to advance the onset of flowering of plants and the dates of first appearance of pollinators (Fitter & Fitter, 2002) and sometimes causes temporal mismatches in mutualistic plant-pollinator relationships (Bellard *et al.*, 2012). Temporal mismatches are also beginning to be found in predator-prey relationships (Laws, 2017).

Plant growth and crop yields

Climate and weather conditions are the primary controlling factors of plant productivity. Aspects of climate change that can be expected to enhance productivity include: moderate increases in temperature in places currently below the optimum for plant growth; increases in precipitation in drylands; and longer frost-free season and growing seasons. However, these simple relationships are complicated by many other factors. Temperature has nonlinear effects on metabolism and different physiological process can react differently (Dillon *et al.*, 2010). Negative effects of climate change on agricultural productivity are expected through unfavourable temperatures, reduced rainfall in some areas, less reliable rainfall and pests (Lobell & Field, 2007; Rosenzweig *et al.*, 2001). For example, from 1980 to 2008, the global maize and wheat yields have been estimated to have declined by 3.8% and 5.5% respectively, related to the climate trends (Lobell *et al.*, 2011). Global yield loss for wheat and maize could be up to 20% and more than 30%, respectively, under Representative Concentrations Pathway 8.5 (Müller & Robertson, 2014) – although this does not take the CO_2 fertilization effect into account.

There are also important direct effects associated with the rise in atmospheric CO_2. Higher CO_2 concentrations may increase photosynthetic rates directly (CO_2 fertilization) and also water-use efficiency, thereby reducing drought susceptibility (Li *et al.*, 2017; Long *et al.*, 2004). However, increased plant growth can eventually be reduced by nitrogen limitation (Beier *et al.*, 2008) and ultimately the vegetation may succumb to direct negative effects of changes in mean and extreme climate (Lobell *et al.*, 2011; Lobell & Gourdji, 2012). Climate change results in changes in soil processes which can also lead to changes in productivity (Beier *et al.*, 2008; Várallyay, 2010). In cold and wet areas, warming increases the decomposition rate of soil organic matter and availability of soil nutrients (Goldblum & Rigg, 2010; Zhao & Running, 2010).

Terrestrial stored carbon is vulnerable to loss back to atmosphere under the influence of climate-induced disturbance, such as droughts (Corlett, 2016), heat waves (Qu *et al.*, 2016), permafrost melt (Schuur & Abbott, 2011), wildland fires (Yue *et al.*, 2015), and pest and pathogen damage (Hicke *et al.*, 2012). It is likely that many forests will become increasingly vulnerable to die-off events (Allen, 2009; Allen *et al.*, 2010; Lewis *et al.*, 2011; Phillips *et al.*, 2009).

There are clear risks of the decline of vegetation carbon sinks (Brienen *et al.*, 2015) and increase of soil carbon release (Crowther *et al.*, 2016) in many regions, even shifts from a sink to a source of carbon (Cox *et al.*, 2000; Kurz *et al.*, 2008). Additional releases of greenhouse gases from terrestrial biosphere into atmosphere will, of course, accelerate global warming.

Ecosystem composition and migration

Studies in a wide range of ecosystems have reported shifts in compositions attributed to changes in climate (Settele *et al.*, 2015). These include, for instance, studies in: tundra (Bosio *et al.*, 2012); boreal forests (Bonan, 2008); Mediterranean forests, woodlands and scrub (Sarris *et al.*, 2011); tropical grasslands, savannah and forests (Higgins & Scheiter, 2012); and peatlands (Limpens *et al.*, 2008). Ecoregions located in Southern and South-eastern Asia, Western and Central Europe, Eastern South America and Southern Australia are thought to be particularly vulnerable (Watson *et al.*, 2013).

In general, warming can be expected to cause poleward and upward altitudinal shifts of species distribution (Peñuelas *et al.*, 2013), especially birds, insects and plants (Bellard *et al.*, 2012; Virkkala 2016). Based on an analysis of more than 1,700 Northern Hemisphere species, an average speed of the northward shifts has been calculated to be about 6.1 km per decade (Parmesan & Yohe, 2003). Species with small population, limited dispersal capacities, narrow ecological niches, isolated suitable habitat patches, and those dependent on the presence of other species all have higher risks of decline. Climate-induced ecosystem shifts are causing declines of biodiversity (Dullinger *et al.*, 2012; Newbold *et al.*, 2015). Since there is evidence for positive relationship between species richness and ecosystem services (Isbell *et al.*, 2011; Cardinale *et al.*, 2012), changes in species composition may affect the stability of entire ecosystems, especially when keystone and dominant species are affected. In some cases, losses will create empty niches that, at least until stabilization of climate change occurs, will lead to an increased risk of weedy and alien species invasion (Blumenthal &

Kray 2014). The future rates of species extinction due to global climate change are predicted to be even higher than the current rates (Bellard *et al.*, 2012; Foden *et al.*, 2013).

Pest and disease incidence

Severe outbreaks of pests and diseases have been linked to climate change and are on the increase. Milder and shorter winters allow for greater overwintering survival of pests and their vectors (Bale *et al.*, 2002). Warmer temperatures can stimulate faster growth and shorter life cycles of many pest and disease species (Deka *et al.*, 2011), and is also likely to allow for the expansion of pest species' geographical ranges. An average of 612 observations of poleward shift of crop pests and pathogens since 1960 has been reported to be about 2.7±0.8 km yr^{-1} (Bebber *et al.*, 2013). Some weed species which were historically restricted to USA have invaded Canada, such as the toxic jimsonweed (*Datura stramonium*), the pasture weed barnyard grass (*Echinochloa crusgalli*) and the crop competitor proso millet *(Panicum mileacium*) (Clements & Ditommaso, 2011). Changes in climate may even raise currently benign species to pest status.

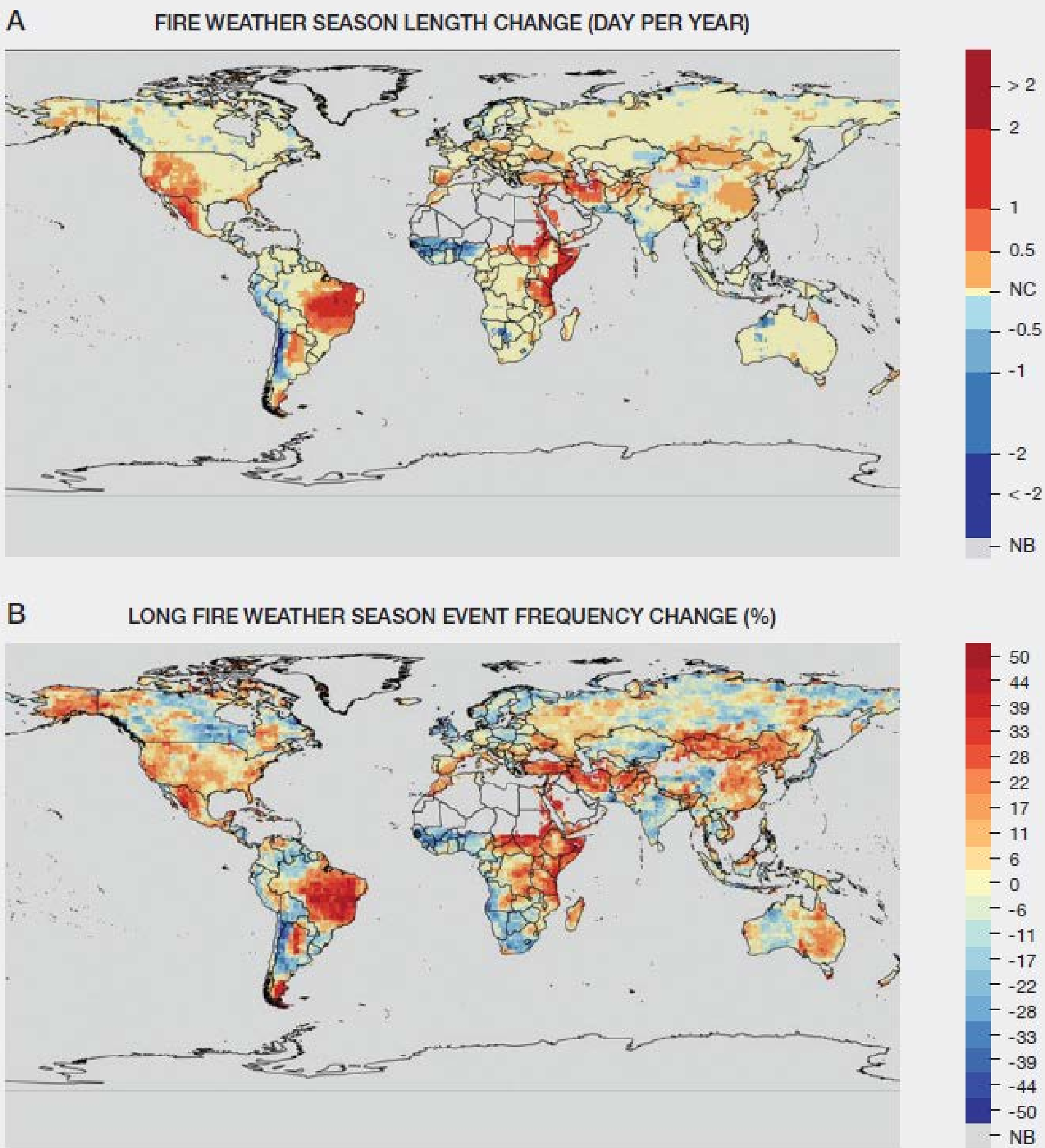
More frequent extreme weather events can reduce the resistance and defences of many organisms and make them vulnerable to diseases and predation that normally cause little harm. Damaged plants can facilitate transmission of viruses and bacteria (Mina & Sinha, 2008). Spider mites, grasshoppers and aphids cause even more severe damage (Canerday & Arant, 1964; Smith, 1954; Starý & Lukášová, 2002; Wainhouse & Inward, 2016). Climate conditions during El Niño-Southern Oscillation events have been correlated with wheat disease in the USA (Rosenzweig *et al.*, 2001). In addition, some arthropod pests favour hot and dry weather because of changes in the nutritional quality of the host plants. See Section 4.2.7 for further discussion of degradation by pests and diseases.

Fire impacts on degradation processes

Kasischke *et al.* (1995) concluded that in boreal forests – a biome which contains between a quarter to a third of the Earth's terrestrial carbon – increased fire frequency and intensity due to warming climate would result in large amounts of carbon released into the atmosphere.

Figure 4 31 Global patterns of fire weather season length changes from 1979 to 2013.

A Areas with significant trends in fire weather season length. B Regions that experienced changes in the frequency of long fire weather seasons (>1 standard deviation of historical mean) from 1996 to 2013, compared with 1979 to 1996. Reds indicate where fire weather seasons have lengthened or long fire weather seasons have become more frequent. Blues indicate areas where fire weather seasons have shortened or long fire weather seasons have become less frequent. Areas with little or no burnable vegetation are shown in grey. Source: Jolly *et al.* (2015).



In the coming decades, it is likely that fire in many regions of the world will increase as a result of climate changes (Figure 4.31) (IPCC, 2007). Climate and wildfire are closely coupled, although there are feedback loops that are not fully understood. Climate change is expected to have complex and nonlinear effects on, for example, fuels, both increasing and decreasing availability.

Changes in climates can be expected to affect fires in different ways. For example, Westerling *et al.* (2006) established clear connections between increased spring and summer temperatures and earlier snowmelt, which result in longer lasting wildfires and fire seasons and greater large-wildfire frequency. de Groot *et al.* (2013) modelled future boreal fire regimes in western Canada and central Russia using several global climate models and three climate change scenarios. Their results pointed to more severe fire weather with subsequently greater potential for extreme fire events.

Wildfire models attempt to simulate reality to estimate outcomes such as probability, spread, intensity, emissions, and impacts to the landscaped. Fire prediction modelling is based on numerical simulations of wildfires to describe the probability of an event occurring, and the behaviour and spread of potential or current fire event. The modelling is based on numerous components such as fuel conditions, weather, and terrain, the ensemble of which is often referred to as the fire environment. Ignition because of lightning is sometimes considered using a lightning ignition efficiency factor (Latham & Schlieter, 1989). However, the human component of fire ignition is difficult to predict, and while lightning causes many fires, in populated areas humans are responsible for most fires – namely, 90% according to the National Interagency Fire Center (2018).

Various models exist around the globe to improve our knowledge of past and future events and inform preparations, policy, and operational fire management, but do not necessarily integrate all components. Future development of models will necessitate both big-data computing power and better understanding of the physics of the fire ignition and propagation processes.

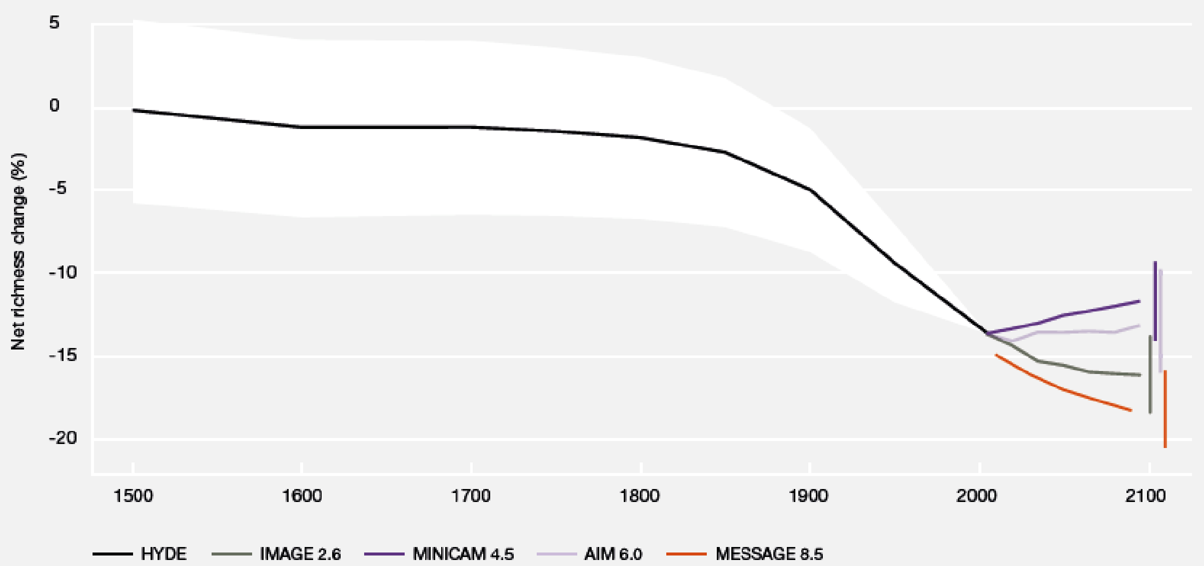
4.2.9 Biodiversity Loss

Trends

It has been proposed that we are in a sixth mass extinction of the Earth's species, following five others in the past 540 million years (Barnosky *et al.*, 2012; Ceballos *et al.*, 2015; Wake & Vredenburg, 2008). There are an accumulating number of studies that almost universally support this conclusion, although often with caveats owing to the paucity of data. For example, a meta-analysis reported that by 2005 land use and related pressures had reduced local species richness (including all kinds of organisms) by an average of 13.6% (95% confidence interval: 9.1-17.8%) (Figure 4.32) and total abundance (i.e., measured as density, cover, or biomass) of plants and animals by 10.7% (95% confidence interval: 3.8% gain to 23.7% reduction) compared with what they would have been in the absence of human effects (Newbold *et al.*, 2015). Current rates of species extinction are estimated to be about 1,000 times the background rate (rate without the presence of human pressures) (Pimm *et al.*, 2014). The IUCN Red List documents 25,360 species as threatened or extinct (IUCN, 2017b), and repeat assessments of entire taxonomic groups show that extinction risk is increasing over time, albeit at widely varying rates (Butchart *et al.*, 2007). A recent study of genetic diversity in 4,675 species estimated the spatial distribution of genetic diversity present in grid cells sampled globally and found lower genetic diversity in habitats more affected by humans than in wilder regions (Miraldo *et al.*, 2016). A meta-analysis suggests that by 2005 land use and related pressures had reduced local species richness by an average of 14% (going up to 32% in vast areas of the globe) (Figure 4.33).

Figure 4 32 **Estimated decrease in species richness between 1500 and 2005 and projected trends.**

The study uses data from the oldest available data, some extending back to 1500 as the reference to estimate the net change (Newbold *et al.*, 2015). Projected future trends are the results of five different models using the four IPCC Representative Concentration Pathway scenarios (Hurtt *et al.*, 2011).



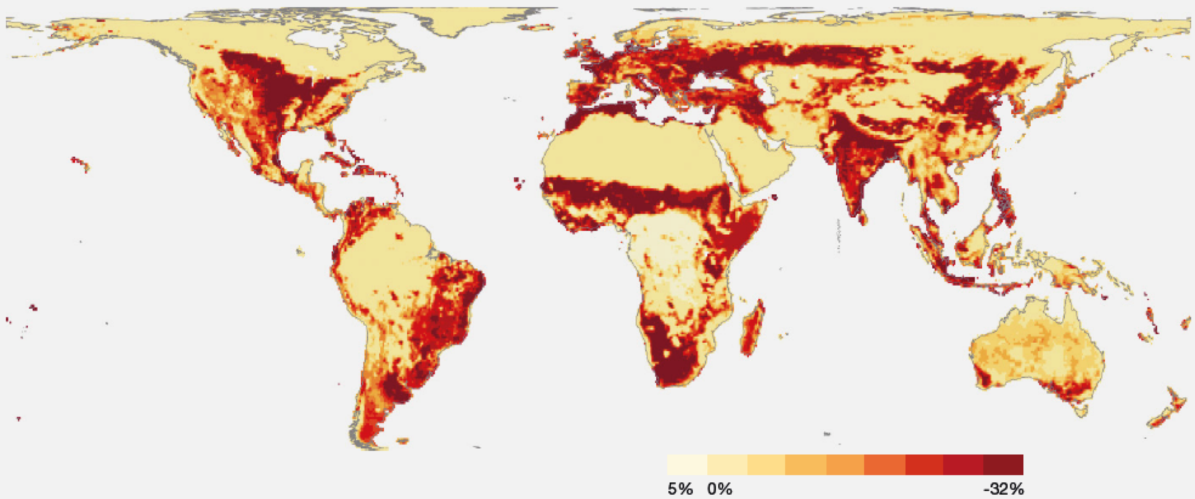
The distribution of declines in biodiversity is not geographically uniform. Croplands, pastures and urban areas have been found to have suffered the highest decrease in species richness and abundance compared to primary ecosystems and secondary growth, in a review of data from 284 publications including 26,953 species (1.4% of the 1,900,000 known species) (Chapman, 2009). Many of these estimates, however, are based on extrapolations from field studies and simple modelling.

Taking Finland and Sweden boreal forests as an example, the last Red List (IUCN, 2017b) has 1,880 and 1,992 listed species for the two countries, respectively (IUCN, 2017b; Rassi *et al.*, 2010). In Finland 56% of all forest habitat types are endangered, especially herb-rich and other highly productive forests (Raunio *et al.*, 2008). In some boreal forests, the loss of species seems to have slowed recently. However, the Red List indices of Finnish forest species is not decreasing, suggesting more stable diversity – or that the species with negative trends are compensated for by others with positive trends (Juslén *et al.*, 2016).

Among well-studied taxa, species with very small ranges are disproportionately threatened (Ceballos *et al.*, 2017; Pimm *et al.*, 2014). For example, the highest numbers of bird species live in the lowland Amazon, whereas small-ranged species concentrate in the Andes.

Figure 4 33 **Net change in local richness caused by land use and related pressures by 2000.**

Net change between 1500 and 2000 of within sample species richness is modelled according to an IMAGE 2.6 reference scenario (Hurtt *et al.*, 2011). The baseline landscape was assumed entirely uninhabited, unused primary vegetation. Source: Newbold *et al.* (2015).



A distinction must be made between species extinction and declines in population size (Table 4.12). Extinction is hard to verify but changes in geographical range are more reliable. The Living Planet Report 2016 (WWF, 2016) included data for 4,658 monitored individual populations of 1,678 terrestrial species and reported that population sizes of the species assessed have declined by 38% since 1970 with an average annual decline of 1.1%. The equivalent figures for grassland and freshwater, respectively, were 18% and 81% (Figure 4.34). In a sample, comprising nearly half of known vertebrate species, 32% (8,851/27,600) were reported to be decreasing – that is, they have decreased in population size and range. For 177 mammals for which detailed data are available, all have lost 30% or more of their geographic ranges and more than 40% of the species have experienced severe population declines (>80% range shrinkage) between 1900 and 2015 (Ceballos *et al.*, 2017). Although this decline is markedly larger than the one provided by Newbold (11%) (Newbold *et al.*, 2015), both show a consistent decline in the number of species per site and in the sizes of individual populations due to anthropogenic disturbances. Eighty percent of Earth's land animals and plants live in forests, and many cannot survive the deforestation that destroys their habitats (Brooks *et al.*, 2002).

Figure 4 34 **Living Planet Index (LPI) global and for terrestrial and freshwater species. Monitored between 1970 and 2012.**

Mean shown with a solid line, 95% confidence limits with fill colour. Green trends show the LPI for all groups, orange trends show LPI calculated without less represented taxa. Source: McRae *et al.* (2017).

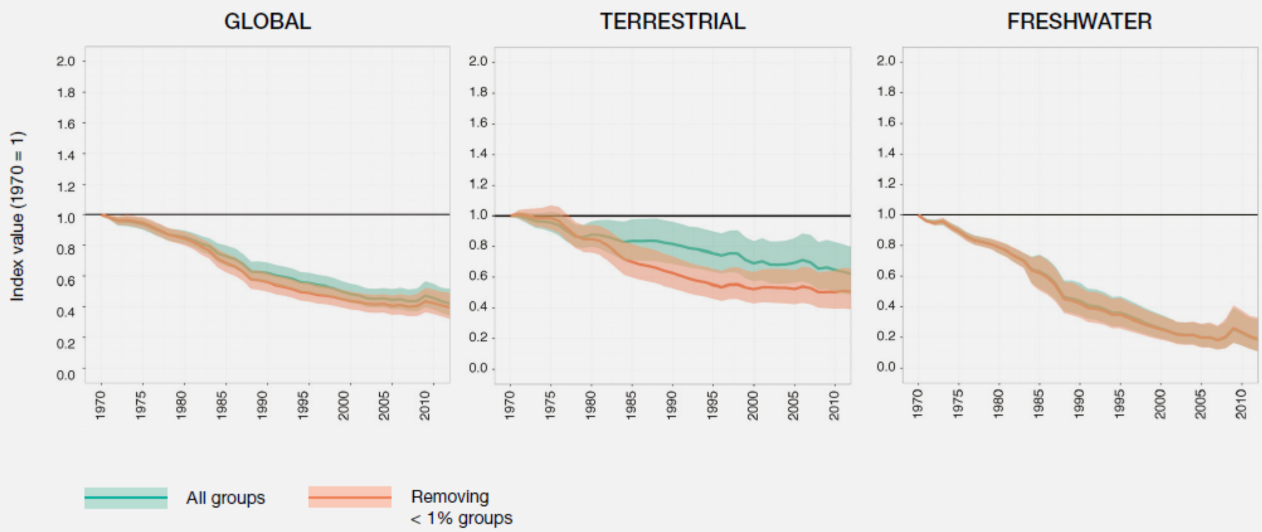


Table 4.12 Distribution of species facing imminent extinction (i.e., trigger species) and historically extinct species among taxa and island, mountain and low mainland areas (from Ricketts *et al.*, 2005). Trigger species meet the criteria necessary to trigger sites for this analysis. Historically extinct species are known to have become extinct since 1500 (IUCN, 2017b) and are mapped according to their last recorded location.

	Islands*		**Mountains[1]**		**Low mainlands[2]**		**Total**	
Taxon	Trigger spp.	Extinct spp.	Trigger spp.	Extinct spp.	Trigger spp.	Extinct spp.	Trigger spp.	Extinct spp.
Mammals	80	49	35	5	16	19	131	73
Birds	128	121	51	1	38	7	217	129
Reptiles[3]	7	8	0	0	8	1	15	9
Amphibians	88	19	268	11	52	4	408	34
Conifers	9	0	12	0	2	0	23	0
Total	312	197	366	17	116	31	794	245

*Islands are defined as landmass smaller than Greenland (New Guinea being the largest island) and include mountainous sections of islands.

[1] Mountains exclude mountainous sections of islands and are defined on the mainland by using classification from the Millennium Ecosystem Assessment (Körner *et al.*, 2005).

[2] Low mainland regions are neither on islands nor in mountainous regions of continental mainlands.

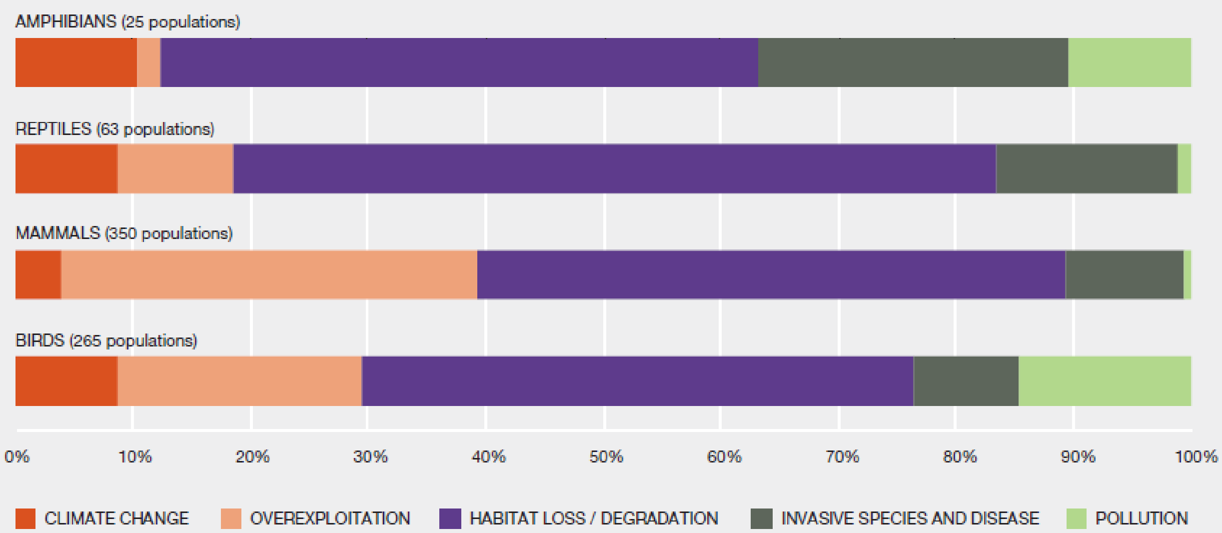
[3] Reptiles include only taxa that have been globally assessed by the 2004 IUCN Red List: order *Testudines,* order *Crocodylia*, and family *Iguanidae*.

Processes of biodiversity loss

The main causes of biodiversity loss due to human pressures mostly involve changes in the land uses, clearing primary or already disturbed land for agriculture. Other causes are species overexploitation, climate change, pollution, and invasive species and disease (Figure 4.35) (WWF, 2016). The particular change often differs between species, for example, the most common threats to amphibians is habitat degradation (Figure 4.35), but in many regions or for many species the most common threat is disease generated by species invasion (e.g., *Chytridiomycosis*). Habitat loss can involve the partial or complete destruction of the plant cover, with the consequent removal of almost all animal and plant diversity. This situation is usually caused by habitat transformations for agriculture or mining. Another common way biodiversity is reduced is by the selective removal of species, for example, trees for timber (silviculture) or animals for food or recreational purposes, like fishing and hunting, either legal and illegally. Removal of a species often disrupts the structure of interaction networks of ecosystems and can lead to new network structures more vulnerable to further pressures. The removal of large animals, for example, has been found to have major implications for the overall ecosystems functioning because it may change how plants species compete or disperse in the landscape (Malhi *et al.*, 2016). In general, there are sound theoretical reasons to infer that as biodiversity declines so does ecosystem functionality and thus the supply of ecosystem services, but the evidentiary base remains incomplete (Cardinale *et al.*, 2012) (also see Section 4.2.6.3). Other causes of biodiversity loss include pollution by toxic trace elements, POPs (persistent organic pollutants, see Section 4.2.4.2) (Mulder & Breure, 2006), nutrients (Carpenter *et al.*, 1998) and systemic pesticides (van der Sluijs *et al.*, 2015). Finally, climate change has major impacts on species phenology, species ranges and also on biological interactions, such predator-prey relationship, plant-herbivore interaction, or pollination de-synchronisation (Walther *et al.*, 2002).

Figure 4 35 The most common drivers of biodiversity loss amongst some animal taxa.

Data includes 703 populations from the Living Planet Report. Source: WWF (2016).



Impact on ecosystem process and function

A meta-analysis comparing multiple experimental results with previous meta-analysis on the effects of major anthropogenic disturbances on ecosystem productivity and decomposition found that intermediate levels of species loss (21-40%) reduced plant production by 5-10% (Hooper *et al.*, 2012).These results were comparable to the effects of more intense ultraviolet radiation and climate warming. Higher levels of extinction (41-60%) had effects similar to ozone depletion, acidification, elevated CO_2 and nutrient pollution. At intermediate levels, species loss generally had equal or greater effects on decomposition than did elevated CO_2 and nitrogen addition. More specifically, a large scale experiment of 150 grasslands found that high richness in multiple trophic groups had stronger positive effects on ecosystem services than richness in any individual trophic group (Soliveres *et al.*, 2016). Thus, biodiversity protection and restoration may require restoration of multiple trophic groups rather than absolute diversity within one group.

The loss of soil biodiversity

Soil consists of biotic and abiotic components linked together by complex interactions based on conversions of energy and materials. The soil flora and fauna have been described as "the biological engine of the Earth" (Ritz *et al.*, 2004) responsible for and modulating many of the processes which occur in the soil system. Soil organisms are largely responsible for cycling nutrients in terrestrial systems, processing carbon and nitrogen through decomposition, mineralisation, immobilisation and volatilization. The multiple functions of ecosystems are heavily dependent on soil (Delgado-Baquerizo *et al.*, 2016; Soliveres *et al.*, 2016) and so it follows that degradation of the soil biota will compromise functionality throughout trophic levels and be a general threat to ecosystem sustainability (Wagg *et al.*, 2014).

The precise relationship between land use, vegetation and soil biodiversity is a complex one. Prober *et al.*, (2015) demonstrated that plant diversity in grasslands worldwide predicts beta diversity (number of species in two habitats that do not occur in both), but not alpha diversity (number of species within a single habitat). Delgado-Baquerizo *et al.* (2016) have shown that the ratios of C:N:P drove bacterial diversity and composition, while other factors (climate, soil heterogeneity, soil pH, root processes and total microbial biomass) were secondary factors, although still important. Fierer and Jackson (2006) have highlighted the importance of factorsm, such as pH, in determining soil microbial biogeography and suggest that this is fundamentally different from macro-organisms. Food production is dependent on soil with a stable and fully functional biotic community. Earthworms and other macroinvertebrates, microarthropods, nematodes and microbial communities are known to be affected by the disturbances and stresses of intensive agricultural, extractive industries, urbanisation, non-point and point pollution (Ponge *et al.*, 2013). However, intensive agricultural production has long been recognised as disrupting and reducing the soil biota (Culman *et al.*, 2010), so that maintenance of yields requires artificial substitution for those processes by cultivation and application of man-made chemicals. However, treatment with biocides (e.g. Cortet *et al.*, 2002; Frampton *et al.*, 2001; Rebecchi *et al.*, 2000) and fertilizers (e.g. Cole *et al.*, 2005; van der Wal *et al.*, 2009) often leads to losses in soil biodiversity. Several studies have shown the decline of soil organic matter in croplands, especially in regions with intensive agriculture since the mid-20th century (e.g. Bellamy *et al.*, 2005).

Soil structural stability is impacted by intensive mechanized agriculture, earth-moving for civil engineering and soil compaction (e.g., Cluzeau *et al.*, 1992; Heisler & Kaiser, 1995). Soil biota create open soil structures, aerate the soil and maintain a fertile mix of mineral materials, allowing and modulating gaseous exchange, water storage and movement, without which plant growth would be compromised (van der Putten *et al.*, 2004). Tillage affects soil structure, for example creating hard layers where fine materials washed from the tilled horizons are deposited immediately below the plough depth. The effects on soil biodiversity have been demonstrated in several studies (Cortet *et al.*, 2002; Krogh *et al.*, 2007; Lagomarsino *et al.*, 2009). This decline in diversity is correlated with changes in biogeochemical cycles, but not directly, since there can be a strong biological activity even with poor microbial biodiversity and vice versa. Nitrogen transformations become disconnected with the result that much inorganic nitrogen can be lost from the system into ground and surface waters in the form of nitrates, or through volatilisation as ammonia and dinitrogen oxides.

This leads to a vicious circle in which declining yields, caused by artificial soil management, can only be maintained by greater applications of artificial treatments. Several approaches can be adopted to mitigate these effects, the one most familiar to western agricultural practices being crop rotation and fallowing. In this way complexity, heterogeneity and diversity can be exploited to secure productive and resilient food chains. However, alternative approaches are emerging as a result of a better understanding of agro-ecology and the role of biodiversity (Altieri, 1999) and its significance in integrated farming systems (e.g. Edwards *et al.*, 1993).

Soil biodiversity can provide signals as to the extent of degradation and the success or failure of restoration programmes (Harris, 2003, 2009; Wubs *et al.*, 2016) but there have been no global-scale assessments of the extent of soil biodiversity under different types and degrees of degradation. In the Global Soil Biodiversity Atlas, Orgiazzi *et al.* (2016) developed some “potential threat” maps, but further progress requires significant validation and model development before it could be used to assess the status and trends of soil biodiversity at scales beyond an individual field.

4.3 Degradation impacts in response to human drivers

This section considers the impacts in response to the drivers of degradation as identified in Chapter 3. It draws on the cross-cutting processes as discussed in Section 4.2 above, and considers the combined impact they have on the environment.

4.3.1 Native habitat loss

Habitat loss is the primary cause of species extinctions (Mace *et al.*, 2005; Hurtt *et al.*, 2011). Ramankutty & Foley (1999) estimated that there has been a net loss of 11.4 million km^2 of forests/woodlands and 6.7 million km^2 of savannahs/grasslands/steppes since 1850. In fact, worldwide, agriculture has already cleared or converted 70% of the grassland, 50% of the savannah, 45% of the temperate deciduous forest, and 27% of the tropical forest biome (Foley *et al.*, 2011). Temple (1986) found that 82% of endangered bird species were significantly threatened by habitat loss. Most amphibian species are also affected by habitat loss, and some species are now only breeding in modified habitats. Eighty percent of Earth’s land animals and plants live in forests, and many cannot survive the loss of their habitat (WWF, 2016). In the USA, less than 25% of native vegetation remains in the East and Midwest. Only 15% of land area remains unmodified by human activities in all of Europe. Nevertheless some species are pre-adapted to new habitats (e.g., fox, deer, rats) (Luniaj, 2004), where they may multiply to the point when they become pests (see Section 4.2.7).

Tropical rainforests have received most of the attention concerning the destruction of habitat. From the approximately 16 million km^2 of tropical rainforest habitat that originally existed worldwide, less than 9 million km^2 remain today. The current rate of deforestation is 160,000 square km^2 yr^{-1}, which equates to a loss of approximately 1% of original forest habitat each year. In an assessment of 152 cases of net losses of tropical forest cover, the proximate causes were agricultural expansion (96%), infrastructure expansion (72%), and wood extraction (67%) (Geist & Lambin, 2002).

Habitat loss is rarely absolute, rather the pre-disturbance area is dissected (see Section 4.2.6.5) and patches of different sizes are created – for example, residual patches of forest and wetlands surrounded by cultivation. Larger patches tend to contain larger numbers of species, and the relative numbers often follow systematic mathematical relationships with area - the species - area curve (Rosenzweig, 1995; Losos & Ricklefs, 2010). The species-area relationship may take time to re-establish after a sudden change in habitat – the so-called relaxation effect – which could give a false impression of the equilibrium number of species (see also Chapter 2, Section 2.2.1.3).

In addition to patch size, distances between residual patches increase as habitats are dissected by land-cover changes, so the residual patches of native habitat become land "islands" in an ocean of unsuitable habitat. These can be quite small patches and therefore more susceptible to conversion to agriculture (Mabey & Watts, 2000) or other land use. Communities in these islands are subject to occasional losses of individual species caused by random community effects and deliberate or unintended actions by humans. These losses can be reversed by immigration from nearby islands in which species are still present. Thus, a dynamic equilibrium is established between the two processes, as described by the equilibrium theory of island biogeography plants (Losos & Ricklefs, 2010). As with the species-area curve, the distance between habitat patches and species number is generally not linear.

The status of a specific, individual species can be different depending on their susceptibility to local extinction and dispersal capabilities, for example large-seeded versus wind-dispersed plants (Losos & Ricklefs, 2010). Organisms can be broadly categorized according to their functional type (Smith *et al.*, 1997), one aspect of which is ability to disperse. Large numbers of propagules that spread widely are designated r-selected, while poor disperses are called K-selected. The connectivity of the landscape varies between species, depending on the mobility of a species and the type of the available habitat and its configuration in the landscape (Bloemmen & Van der Sluis, 2004). A special case is that of migratory species that depend on island "stopovers", in which they feed before continuing their migration; their habitat consists of winter, summer and migration stopovers and all three are equally important. However, they are only temporary visitors at stopover sites and the significance of loss of these habitats can easily be overlooked.

The behaviour of single species has been compared with the spread and ultimate disappearance of an epidemic (Carter & Prince, 1981). Fundamentally a dynamic equilibrium is set up between disappearance of the species in a patch and the distance between patches – unlike population dynamics, the population is of patches, hence it is known as metapopulation dynamics. Surprisingly the relationship between invasion of new patches and disappearance from patches creates the condition for sudden complete loss of a species – a non-linear or threshold behaviour.

One important aspect of habitat loss is the potential for loss of locally-adapted crop species, known as landraces. Landraces arise because isolation of habitat patches can provide adequate breading barriers that result in divergent evolution. The differences between finches of the same species on the different Galapagos islands was remarked upon by Darwin and was one of the pieces of evidence that led to his theory of evolution. Loss of landraces can affect the development of new varieties of crops that can resist diseases or cope with harsh environments (Brush, 1995).

4.3.2 Grazing land degradation

4.3.2.1 Intensive grazing

An estimated 76-79% of pork and poultry produced is from intensive livestock production systems, also referred to as industrial, landless or concentrated animal feeding operations (Herrero *et al.*, 2013). For ruminants, the degree of intensification is slightly less, often with a mixed production models using a combination of pastures together with feedlots. Only about 2% of cattle are raised in fully landless systems, with 40% in rainfed mixed farming systems, 29% in mixed irrigated systems and 26% in fully grazing systems (Steinfeld *et al.*, 2006) There is a gradient in livestock intensification from natural pasture to improved pastures, irrigated pastures, to fully stall-fed production based on purposefully grown fodder. In general this increased intensification is linked with a decrease in biodiversity on the land where it takes place (Rook *et al.*, 2004). Intensively managed pastures are the norm in the EU, North America, Japan and the Republic of Korea. These systems have mineral fertilizer inputs and a greatly reduced biodiversity, compared with the natural pastures or forest they replaced (Steinfeld *et al.*, 2006).

Animal feed required for meat production accounts for an estimated 33-39% of all crop production (Manceron *et al.*, 2014; Paillard *et al.*, 2010; Steinfeld *et al.*, 2006), though this has reduced slightly from the 37-42% of the 2003-2009 period due to high protein soybean replacing less energy dense grain crops (Manceron *et al.*, 2014) (see also Chapter 3, Section 3.3.1). Concentrated animal feeding operations therefore have a high off-site footprint that includes land transformation to agricultural cropland with all its related environmental consequences (see Section 4.3.3).

Concentrated animal feeding operations result in high concentrations of excreta and other waste, resulting in high nitrogen and phosphorus pollution (Miller, 2001) (see Section 4.2.4.2). These are the biggest cause of phosphorus eutrophication in some river systems (Kellogg & Lander, 1999; McFarland & Hauck, 1999). Much of the manure is used as a nutrient supplement on surrounding farmland, but manure applicate based on nitrogen demand may lead to phosphorus build-up over time (Miller, 2001). Pig manure has the highest nitrogen concentration, with poultry the highest phosphorus concentration (Miller, 2001). A number of techniques are available for managing and preventing phosphorus and nitrogen contamination from intensive livestock (Borhan *et al.*, 2012; Provolo *et al.*, 2013; Sharpley *et al.*, 2006). These largely focus of sound waste management, and can also include techniques such as biogas production from waste.

From a GHG emissions perspective, it is the waste management in concentrated animal feeding operations systems that differentiate them from other livestock systems. The manure and other waste can be a major source of methane and nitrogen emissions, especially if stored in anaerobic conditions (Borhan *et al.*, 2012; Hongmin *et al.*, 2006; Provolo *et al.*, 2013), with estimates of methane emissions from manure management, being 0.25 Pg CO_2 eq, and N_2O emissions, 0.21, and 0.49 Pg CO_2 equivalent from manure management and manure application respectively (Herrero *et al.*, 2013) . Intensive production systems help reduce emissions due to their efficiency in converting fodder to animal protein, which greatly reduces the time-period from birth to slaughter mass (Scollan *et al.*, 2010).

4.3.2.2 Extensive grazing

Livestock over-stocking (see Chapter 3, Section 3.3.1) and poor herd management are major causes of degradation in rangelands, although other factors may also be important – such as fire regimes or selective extraction of products other than livestock (see Sections 4.2.6.5 and 4.3.5). The severity of land degradation is highly dependent on the ecosystem's vulnerability, with overgrazing increasing this vulnerability (Weber & Horst, 2011). The high variability in rainfall in drylands means that appropriate stocking rates for a specific area fluctuate year to year, and stocking at a density to exploit all the forage in a good year will exceed the carrying capacity in average or poor years (Behnke & Abel, 1996; Behnke *et al.*, 1993; Vetter, 2005). An often-neglected component of grazing are native and feral herbivores, such as horses and deer in southwestern USA, kangaroos, goats and rabbits in Australia, and locusts especially in Asian and African drylands which compete with livestock for fodder. For example, in Australia, the annual losses owing to competition of kangaroos with livestock are estimated at AUS $27.46 million (McLeod, 2004). There is evidence that locust plagues are associated with over-grazing (Cease *et al.*, 2012).

Heavy grazing clearly is the cause of most rangeland degradation, for example, in the over-populated, communal areas in southern Africa (Prince *et al.*, 2009), despite the fact that lower stocking rates can give better long-term financial returns (Behnke & Abel, 1996; Behnke *et al.*, 1993). The most direct impacts of overgrazing are trampling and the removal of ground cover leading to erosion (see Section 4.2.1). Grazing animals select the more palatable species and, at high stocking rates, this can lead to changes in the composition of the vegetation (Todd & Hoffman, 1999), favouring less palatable species (“increasers”) (Abule *et al.*, 2005; Vesk & Westoby, 2002) and changing grass-to-woody plant ratios (see Section 4.2.6.2) (Wigley *et al.*, 2009, 2010). Composition changes often include a shift from perennial to annual grass species (Kelly & Walker, 1976; Milchunas & Lauenroth, 1993; Parsons *et al.*, 1997), or to shrubby unpalatable woody perennials (Milton *et al.*, 1994b), which reduces forage value while making the area more susceptible to fire (Balch *et al.*, 2013). Invasive species are causing increasing damage to rangeland worldwide. In the United States, about 300 rangeland weed species cause an estimated loss of $2 billion annually (DiTomaso, 2000). In South Africa, about 161 invasive rangeland plant species are recorded, which impact about 10 million hectares or 8% of the country (Richardson & van Wilgen, 2004). In Australia, about 622 non-native naturalized rangeland plant species are recorded, 26% of which are posing threat to rangelands (Martin *et al.*, 2006). While light grazing may improve biodiversity, heavy grazing reduces biodiversity (Borer *et al.*, 2014; Lunt *et al.*, 2007). Periods of rest from grazing intensity may, however, be important for recovery.

The global extent of rangeland degradation remains contentious (see Sections 4.1.3, 4.1.6 and 4.2.6.2). Many measures emphasise erosion (see Section 4.2.1) or net primary production (see Section 4.2.3.2) and omit shifts to less palatable species and impacts from alien invasive species (see Section 4.3.7). However, at national and local levels the impacts of rangeland degradation on livestock carrying capacity is well-documented. Adeel *et al.* (2005) reported that overstocking and range mismanagement led to a decline in livestock numbers after peaking at the beginning of the twentieth century. National level reported losses in livestock carrying capacity include a 40% loss in New Mexico (Fredrickson *et al.*, 1998), 45% loss in western New South Wales (Mitchell, 1991; Rietkerk *et al.*, 1997), 60% loss in Prince Albert District of South Africa (Milton & Dean, 1996) and a 47% loss in Namibia (de Klerk, 2004). Furthermore, rangelands throughout the world are being lost to cropland expansion (see Section 4.3.2) and other human uses (see Section 4.3.10). This, in part, drives the expansion of intensive livestock systems (see Section 4.3.2.1), but has also resulted the conversion of forests to rangelands. In Brazil, 70–80% of total deforestation is estimated to have resulted from the development of extensive livestock systems (Tourrand *et al.*, 2004). However, recent data suggest that the rate of Amazonian deforestation as a direct or indirect consequence of cattle and soy production has decreased substantially (Foley *et al.*, 2007; Gibbs *et al.*, 2016; Nepstad *et al.*, 2006) (see Section 4.3.1 and 4.3.4.1).

4.3.3 Cropping Systems

Croplands may inadvertently degrade the very ecosystem services on which they rely through eutrophication of water bodies by fertilizers, toxic effects of pesticides and fungicides, pest and disease control on non-target species and erosion. While crop intensification dramatically increased crops yield during the past decades, it also accelerated pollution of soil and water (Gisladottir & Stocking, 2005). In the last 50 years, the world's irrigated cropland area roughly doubled, but global fertilizer use increased by 500%, overloading global nitrogen and phosphorus sequestration (Chesson *et al.*, 2001; Tilman *et al.*, 2001) (see also Chapter 3, Section 3.3.2). While nutrient excess causes pollution in some regions, it is currently less so in poorer regions, such as Kenya (Russo *et al.*, 2017) and Brazil (Riskin *et al.*, 2017). However, fertilizer use is likely to increase with development (Tilman *et al.*, 2002) and can be expected to further increase global pollution without concomitant extension of control techniques.

Irrigation by water extraction from aquifers can exceed recharge rates (known as over-drafts) in many regions worldwide (Siebert *et al.*, 2010), such as Northeast India and Northwest Pakistan (Rodell *et al.*, 2009), and California's Central Valley (Famiglietti *et al.*, 2011). Water used for irrigation can contain salt and brings salts deeper in the soil profile to the rooting zone (see Section 4.2.2.2). The re-routing of surface waters into dams and reservoirs alters regional hydrology, with cascading consequences for downstream ecosystems.

Tillage creates bare soil that is susceptible to erosion – before planting, between plants and between seasons. Soil can be compacted by tractors and other equipment which also leads to erosion (see Section 4.2.1), poor soil drainage, enhanced runoff, water-logging (see Section 4.2.2.3), breaking down soil aggregates and reduction of the ability of soil to retain moisture. It also increases decomposition rates, which can increase the release of mineral nutrients at times when there may not be a crop present to utilize them and promotes carbon dioxide release from soil organic matter oxidation.

As populations grow, fallow periods usually shorten or can cease, increasing periods of bare soil, which leaves soils vulnerable to all the consequences of bare soil. It also reduces yields. In developed countries, fields and even large regions are often planted with the same crop (monoculture), which can increase pest and disease pressure through loss of natural control processes, especially in fruit and vegetable crops. Monocultures also require heavy pesticide treatment, which can degrade soils and water quality.

Fertilizers and manures improve yields; however, high rates of applications can lead to a host of environmental consequences including pollution of ground and surface water (Carpenter *et al.*, 1998; Fließbach *et al.*, 2007; Galloway *et al.*, 2003) (see Section 4.2.2.1) and hypoxic coastal water (see Section 4.2.4.2). Furthermore, synthetic fertilizers contain no organic component, which leaves soils vulnerable to erosion and reduces water- and nutrient-holding capacity. The use of organic fertilizers such as farmyard manure is always superior, but the materials are generally not available in adequate quantities.

Chemical pest and weed control has been linked to, for example, water pollution, declines in bird and bee populations and other negative effects on ecosystem services, including human health (Hernandez *et al.*, 2011; Potts *et al.*, 2016). A growing dependence on chemical pest control has created a "pesticide treadmill," where pests develop resistance to one pesticide and so new ones have to be developed if possible (see Section 4.2.4.2).

The effects of cropping are at multiple spatial scales. At the farm-scale, practices such as tillage, irrigation, crop rotations, fertilizer use and chemical pest and weed control can all cause land degradation. The same factors also have consequences at the landscape, regional and global scales, although the connections are less obvious. Over larger areas, the percent of land cleared for agriculture, the degree of fragmentation, the heterogeneity of crops and land-use systems, mainly affect biodiversity beyond the local habitat scale (see Section 4.3.1) and can influence regional climate through CO_2 emissions (i.e., 10-12% of global carbon emissions are from agriculture).

4.3.4 Forest degradation

4.3.4.1 Deforestation and forest degradation

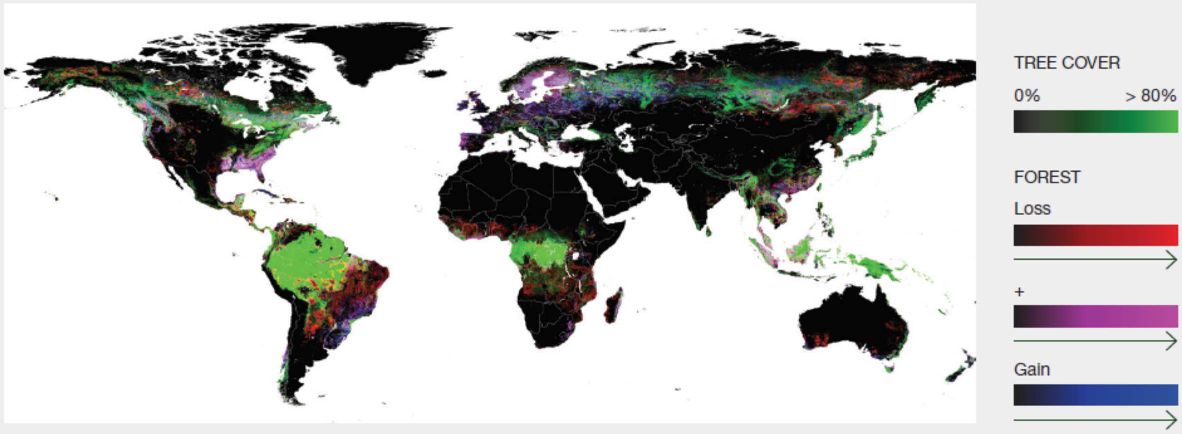
Forests worldwide are in a state of flux, with accelerating losses in some regions and gains in others (see Chapter 3, Section 3.3.3 for additional information on drivers). From 2000 to 2012 there was a net loss in global forest area of 2.3 million km^2 and a gain of 0.8 million km^2 (Hansen *et al.*, 2013b). From 2000 to 2012 the extent of undisturbed forest (IFL; see Section 4.2.6.5) fell by 7.2% (Figure 4.36) (Potapov *et al.*, 2017b). Another method that did not exclude forest borders – and therefore may have counted other cover types – reported 18% of the global hinterland forests disappeared between 2007 and 2013 (Tyukavina *et al.*, 2015). Losses have been unevenly distributed, for example a decline in Brazil's deforestation was offset by increases in Indonesia, Malaysia, Paraguay, Bolivia, Zambia, Angola, and elsewhere (Hansen *et al.*, 2013b). Intensive forestry in subtropical forests has resulted in the highest rates of forest change globally (Malhi *et al.*, 2014). Boreal forest losses are second to those in the tropics, largely due to fire and forest utilization. They have a relatively short history of large-scale human settlement: localized degradation started around 16th century, but more recently there has been large-scale logging, initially for tar production and later for shipbuilding, charcoal and so on (Wallenius *et al.*, 2010). Currently, logging for lumber and biomass harvesting for power generation are the most important uses which, together, are now very extensive. For example, in Fennoscandia, more than 90% of the productive forests are under intensive forest management, often at the expense of other ecosystem services (Bouget *et al.*, 2012; Gamfeldt *et al.*, 2013b; Hansen *et al.*, 2013b).

Future losses of forests are estimated at 170 million ha by 2030 (WWF, 2016). The main deforestation fronts are shown in Figure 4.37. Mosaics comprised of trees outside forests, remnant forest patches, and young regenerating forests constitute a modest proportion of the tropical forest estate, and lack most of the processes of continuous forests.

Forest expansion continues to occur in most industrialized countries, on lands abandoned by farming and animal husbandry and areas that continue to mature on land that was deforested in the past century but have not been converted to a different land use since then (Keenan *et al.*, 2015). Some middle income tropical countries are also transitioning to the forest gain stage. The 2015 Global Forest Resources Assessment (Keenan *et al.*, 2015) indicates that, between 1990 and 2015, 13 tropical countries may have either passed through their forest transitions from net forest loss to net forest expansion (Rudel *et al.*, 2005), or continued along the path of forest expansion that follows these transitions.

Figure 4 36 **Global tree cover, forest loss, and forest gain from 2000 to 2012.**

The colour composite shows tree cover in green, forest loss in red, forest gain in blue, and forest loss and gain in magenta. Loss allocated annually. All map layers resampled for display purposes from the 30-m observation scale to a 0.05° geographic grid (Hansen *et al.*, 2013). Forest-area estimates of the Global Forest Resources Assessment 2015 (FRA) (Keenan *et al.*, 2015) are close to satellite-derived estimates, with deviations of ±7% globally and ±17% for the tropics.



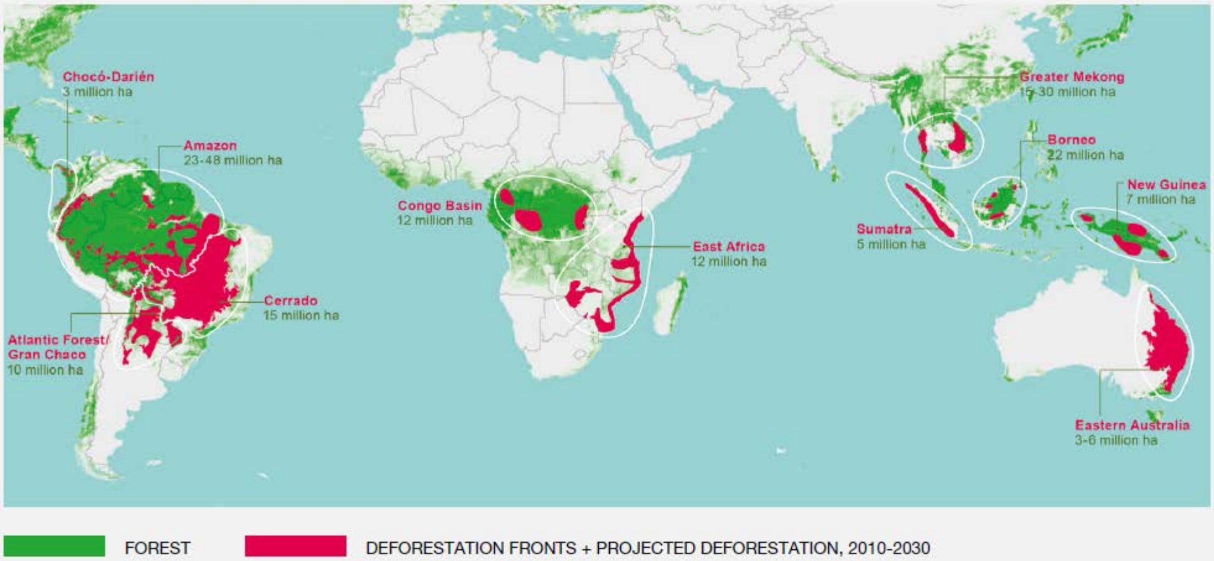
Planted forests (see Box 4.2) account for 25-100% of gains and increasingly substitute for natural forests, particularly in Africa. The global rate of planted-forest expansion since 1990 is close to a target of 2.4% per annum necessary to replace wood supplied from natural forests in the medium term, although the rate had declined to 1.5% since 2005 (Sloan & Sayer, 2015). Multiple-use forests where both production and conservation are permitted, account for 26% of the global forest and 17% of the tropical forest area, having increased by 0.81 M km^2 or 8.5% globally since 1990, with most gains in the tropics.

Forests are the largest single terrestrial sink of carbon (Watson *et al.*, 2000) (see Section 4.2.3.2). It is estimated that more than 1.5 x 10^{12} g of CO_2 are released to the atmosphere due to deforestation every year, mainly due to cutting and burning (DeFries *et al.*, 2007; Houghton, 2005), approximately equal to 25% of emissions from combustion of fossil fuels (Andrasko, 1990). A recent study found that Intact Forest Landscapes (see Section 4.2.6.1) comprise 20% of all tropical forest, yet contain 40% of all the above ground forest carbon, and have diminished in area by 7.2% between 2000 and 2010 (Potapov *et al.*, 2017b).

There is an important distinction between the terms "deforestation" and "forest degradation" used here and elsewhere. There is no deforestation if clear felling is on an area that, in time, will regenerate to forest. Degradation, on the other hand, does not involve a reduction of the forest area, but rather a reduction in its condition within an existing forest (Cannon, 2018; Lanly, 2003; van Lierop *et al.*, 2015b), such as changes in canopy vertical and horizontal structure (crown cover), exposure of the field layer, and a decrease in shade (Souza *et al.*, 2005), or a loss of fauna (see Section 4.3.5). In the Democratic Republic of Congo studies have shown that, while core forest diminished between 3.8% and 4.2%, isolated forest incursions almost doubled during the 2000-2010 period, increasing forest fragmentation and hence reducing biodiversity habitat (Harris *et al.*, 2017; Molinario *et al.*, 2015; Potapov *et al.*, 2017b).

Figure 4 37 Areas where the bulk of global deforestation is expected to take place from 2010 to 2030, under business-as-usual scenarios (see Section 4.1.3) and without interventions to prevent losses.

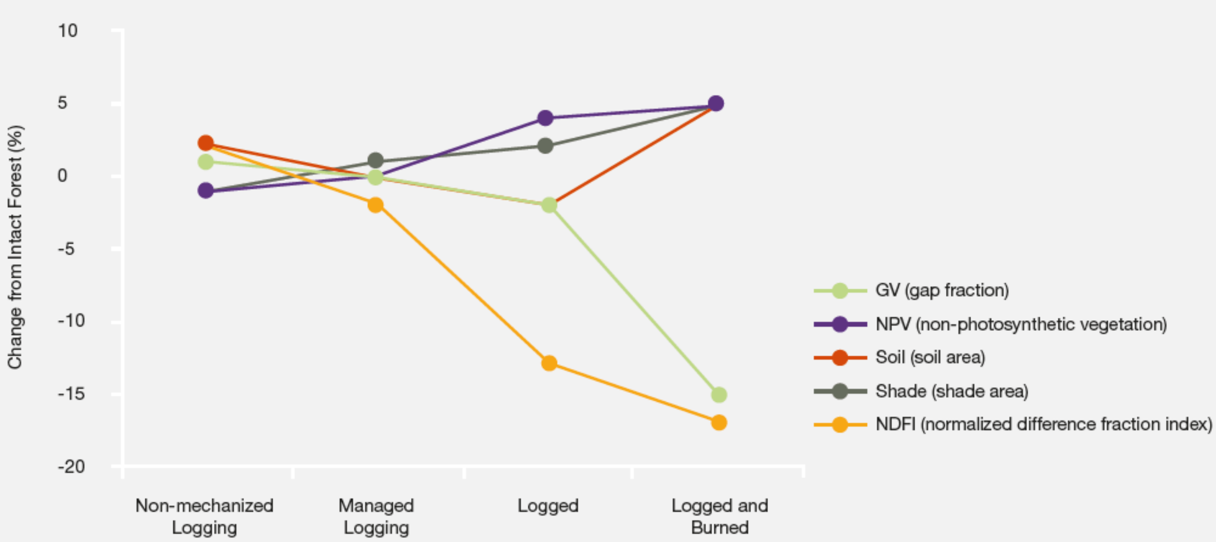
The 11 regions where the losses are expected to be greatest are circled. Source: WWF (2015).



Forest degradation includes fragmentation which has important effects beyond the proportion of area cleared (see Section 4.3.1) (Broadbent *et al.*, 2008; Gascon *et al.*, 2000; Murcia, 1995; Skole & Tucker, 1993). For example, the relationship between species extinctions and residual patch size is often non-linear (see Section 4.3.1) (Broadbent *et al.*, 2008; Gascon *et al.*, 2000; Murcia, 1995) . While deforestation has been large, degradation is generally agreed to be higher. The World Resources Institute (WRI) estimated that about 20% of global forest has been degraded compared with 30% that has been completely cleared (Minnemeyer *et al.*, 2011).

Figure 4 38 Effects of different types of logging on forest measures.

All measured relative to intact forest. Source: Souza *et al.* (2005).



Deforestation is relatively easy to detect with remote sensing (Hansen *et al.*, 2013b), but degradation of forest interiors is much more difficult (Dudley *et al.*, 2005; Souza *et al.*, 2005). A remotely-sensed index of forest canopy damage caused by selective logging and associated forest fires has been developed to measure forest degradation (the Normalized Difference Fraction Index) (Souza *et al.*, 2005) (Figure 4.38) and LiDAR remote sensing techniques are likely to make an important contribution in the near future (Donoghue *et al.*, 2007; Dubayah & Drake, 2000).

There are many types of deforestation and forest degradation that must be distinguished in order to understand their causes and effects (Chakravarty *et al.*, 2012; Davidar *et al.*, 2010; Earth Eclipse, 2018). These include managed logging (see Section 4.3.4.2); agroforestry (Box 4.3), firewood collection; livestock browsing; and clearing for hunting, each one of which can have different types and intensities of impacts. In addition to these, there are many anthropogenic activities that lead to inadvertent forest loss, such as pollution of air (see Section 4.2.4.1) and land (see Section 4.2.4) leading to, for example reduced vigour of vegetation; damage to soil properties and organisms; acid rain (Earth Eclipse, 2018); creation of favourable conditions for pests and diseases (see Section 4.2.7.2); soil erosion and sedimentation (see Section 4.2.1); and disturbances caused by recreation and tourism.

Forest loss and degradation have many effects on the broader environment (Chomitz *et al.*, 2007). Clearly, reduced net primary production results in loss of carbon sequestered in biomass and an increase in greenhouse gases (see Section 4.2.8), but there are many other impacts. An important one is the loss of habitat (see Section 4.3.1). Eighty percent of Earth's land animals and plants live in forests, and many cannot survive elsewhere. Removal of large, old trees and woody debris during clear-cutting leads to declines of many species (Oldén *et al.*, 2014; Stokland *et al.*, 2012). Several species typical of mature forests can take decades or even centuries to recover (Josefsson *et al.*, 2010; Paillet *et al.*, 2010). In addition, there can be changes in local and regional climate. Reduced evapotranspiration, infiltration rates and water-holding capacities can cause increased runoff and a decrease in watershed protection, leading to an increase in flooding, erosion (Bruijnzeel, 2004b) and reduced water supply for human use (see Section 4.2.5.1) (Chakravarty *et al.*, 2012; Dudley *et al.*, 2005). Furthermore, beyond the forested region itself, deforestation and forest degradation can disrupt normal weather patterns, creating hotter and drier weather thus increasing drought, crop failures, and displacement of major ecosystems, modifications of wind, water vapour content and mixing of the lower atmosphere. For example, deforestation on lowland plains has been shown to shift cloud formation and rainfall to higher elevations (Lawton *et al.*, 2001).

Box 4.2 Planted Forests

Planted forests established primarily for timber, fibre, fuelwood or environmental protection may have negative or positive impacts on processes land degradation, depending on their local and landscape context, the condition of the land prior to their establishment, species selection, and management practices used for their establishment and maintenance (Brockerhoff *et al.*, 2008, 2013; Hunter Jr *et al.*, 2016; Lindenmayer *et al.*, 2006; Thompson *et al.*, 2014; Waterworth *et al.*, 2007). The replacement of natural or secondary forests or grasslands by plantations typically results in lower rates of soil formation, lower potential for water purification and waste treatment and poorer habitat quality for a wide range of grassland and forest plant and animal species (Brockerhoff *et al.*, 2013; Fletcher *et al.*, 2011). However, where plantations are established on previously degraded lands (e.g., abandoned croplands and pastures, eroded soils, derelict sites resulting from mineral extraction or infrastructure development) (see Section 4.3.8), they may lead to significant improvements in biodiversity (Brockerhoff *et al.*, 2008; Carnus *et al.*, 2006; Parrotta *et al.*, 1997) and other ecosystem services (Brockerhoff *et al.*, 2013; Lamb *et al.*, 2005; Pawson *et al.*, 2013; Thompson *et al.*, 2014). The evidence for this is mixed (Griscom *et al.*, 2017), particularly in light of the risks associated with climate change (Payn *et al.*, 2015). There are also concerns regarding the impacts of some commonly used plantation species that can, in many situations, become invasive (e.g., *Acacia* and *Pinus* species) (Padmanaba & Corlett, 2014; Richardson, 2008).

Box 4.3 Agroforestry

Agroforestry, sometimes known as alley cropping or intercropping with trees, is the simultaneous cultivation of woody plants (trees or shrubs) and herbaceous crops, replacing treeless monocultures. The understory may consist of annual (e.g., maize, cassava) or perennial (e.g., coffee or cacao) crops. Trees are planted on farms for many reasons: often for supplementary income (e.g., fruit or timber), but also for conservation-related purposes such as wind breaks, runoff reduction (in one case by 28-56% according to Lamichhane (2013)) and sediment trapping to minimize erosion. Trees can also capture nutrients that might otherwise be lost to leaching (by 20-40%) (Babbar & Zak, 1995; Mekonnen *et al.*, 1997; Udawatta *et al.*, 2002) and so reduce nitrogen loading in streams (Lamichhane, 2013; Udawatta *et al.*, 2002, 2011). Agroforestry practices can sequester carbon and enhance microbial biomass and enhance water-holding capacity compared to monoculture (Tully & Ryals, 2017). Nitrogen-fixing leguminous trees, such as *Erythrina poeppigiana,* can be used to provide organic material with a high nitrogen content (Harmand *et al.*, 2007; Tully *et al.*, 2013). The orientation and management of the trees plays a major role in their functioning.

4.3.4.2 Timber production

Managed logging for round wood (see Section 4.3.4.2), is often in clear-cut parcels which are susceptible to erosion and, later, burning of discarded branches. Logging often leads to degradation caused by heavy vehicles, construction of access road, and burning forest residue. Some of these are alleviated by non-mechanized forest product extraction, selective logging for one or a few species or the most mature individuals, and replanting (Souza *et al.*, 2005). Intensive logging (see Section 4.3.6) creates a landscape where young forest cohorts are overrepresented compared to natural forests (Bergeron *et al.*, 2001). In North America, intensive logging has changed the whole landscape structure (Cyr *et al.*, 2009). On the other hand, abandonment of Soviet-era agricultural land has caused quite extensive reforestation that partly counteracts forest losses due to fire (Prishchepov *et al.*, 2013).

4.3.5 Non-timber forest use: woodfuel, bushmeat, edible plants, and medicinal herbs

The term non-timber natural resource extraction is used to describe a multitude of practices resulting in the selective harvesting of specific species for subsistence and commercial purposes (Cowlishaw *et al.*, 2005) (see Chapter 3.3.4 for a more detailed description of drivers). The main concern of non-timber natural resource extraction is that specific forest species (or groups of species) are harvested at rates beyond the natural regeneration rates (Bennett *et al.*, 2007; Bennett & Robinson, 2000; Nasi *et al.*, 2008). In addition to changing the species mix, this can result in structural changes to the habitat (Ndegwa *et al.*, 2016).

The degree to which any non-timber natural resource extraction degrades the environment globally is poorly understood, though there are many local cases studies suggesting that local level impacts can be huge (Chidumayo & Gumbo, 2013; Ndegwa *et al.*, 2016). However, there are also data suggesting that most practices can be sustainable, if properly regulated and managed (Benjaminsen, 1993; Chidumayo & Gumbo, 2013; Cline-Cole, 1998; Ribot, 1999). Although there has been an increased focus in both these subjects over the past 10 years, data sources are still few and scattered.

Overharvesting of non-timber products impacts primarily on the product harvested, though there may be a number of secondary impacts on ecosystem services. Many species can survive high offtake levels. However, for slow breeding species even a low offtake can be devastating to population dynamics (Van Vliet *et al.*, 2010; Van Vliet & Nasi, 2008).

Woodfuels

Fuelwood harvesting, can result in overall structural changes of the vegetation, converting a forest or woodland area into shrubland or grassland, with impacts on productivity, soil erosion and biodiversity (Ndegwa *et al.*, 2016). It can have secondary impacts on fire regimes which may restrict woody plant re-generation (Chidumayo & Kwibisa, 2003).

For sustainable woodfuel use, there is no net overall emission of carbon since the harvest is not fully compensated by regrowth. However, where woodfuel is unsustainably harvested, leading to deforestation, the emission from this land-use change is potentially the largest single carbon emission as was found for Zambia (Kutsch *et al.*, 2011).

Ecosystem processes directly impacted through woodfuel harvesting include: increased soil erosion, change in forest/woodland structure, change in woody plant to grass ratios, change in fire regimes, loss of biomass and sequestered carbon, change in soil properties, especially at charcoal pits where extreme temperatures have lasting impacts on soil, change in hydrology, and possibilities of increased flooding (Chidumayo & Gumbo, 2013).

Medicinal plants

Medicinal plant harvesting impacts on species specific such as the African cherry (*Prunus africana*) (Stewart, 2003), driving individual species to near extinction as in the case of *Warburgia salutaris* (pepper bark) and

Ocotea bullata (stinkwood) in South Africa (Botha *et al.*, 2004; Geldenhuys, 2004) (see Chapter 5 for further discussion on non-timber forest use).

Bushmeat

Bushmeat harvesting leads to the selective loss of a large proportion of the mammalian and avian species (Bennett *et al.*, 2007). Redford, (1992) termed this “the empty forest” phenomenon – forests maintaining their mature tree structure, but being devoid of larger vertebrates. These species play an important role in the forest dynamics including pollination, seed dispersal and seedling predation (Connell, 1971; Janzen, 1970; Swamy & Pinedo-Vasquez, 2014; Terborgh & Estes, 2010). Furthermore, there could be impacts on the principle predators (either through direct hunting) or through lack of prey (Henschel *et al.*, 2011). The loss of keystone species can have ripple effects into the overall vegetation dynamics (Campos-Arceiz & Blake, 2011; Fragoso, 1997; Keuroghlian & Eaton, 2009; Terborgh *et al.*, 2001; Terborgh & Estes, 2010). This is not only a developing world or tropical forestry effect, as the re-introduction of wolves into Yellowstone National Park in the USA illustrates (Hermans *et al.*, 2014). There is evidence that forest restoration without the re-introduction of forest vertebrates may be impossible (Brodie & Aslan, 2012; Chapman & Onderdonk, 1998).

The extent to which bushmeat harvesting is unsustainable is poorly researched. Bushmeat harvesting is largely opportunistic and rare species are seldom specifically targeted, representing a small percentage of the total offtake (Abernethy & Ndong Obiang, 2010; Nasi *et al.*, 2011; Van Vliet *et al.*, 2010). Despite this, a number of primate species are in a threatened or vulnerable state largely due to overharvesting.

Hunting has reduced mammalian density by between 13% and 100% (i.e., local extinction) in areas hunted in Central and West Africa (Hart, 2000; van Vliet & Nasi, 2008) and accounts for a 50% decline in apes in Gabon over two decades (Walsh *et al.*, 2003). Hunting is a primary threat to about 85% of the primates and ungulates that are endangered or critically endangered according to the IUCN Red List (Swamy & Pinedo-Vasquez, 2014). Bushmeat hunting can lead to the local and potentially total extinction of some species, the great apes being particularly vulnerable (Abernethy & Obiang, 2010; Oates *et al.*, 2000). Galliform birds are highly threatened by direct pressure from hunting globally, though are seldom hunted in the tropical and Neotropical forests (Keane *et al.*, 2005). Peres and Palacios (2007) identified 11 Amazonian vertebrate species with over a 68% reduction in abundance, with the abundance of Uakari monkey (*Cacajao calvus*) reduced by 90-97% from overhunting.

Regions of specific concern from bushmeat extraction are the Congo basin and Madagascar. It is estimated that between 1 (Wilkie & Carpenter, 1999) and 5 (Fa *et al.*, 2003) million tonnes of bushmeat is harvested annually from the Congo basin alone. The Congo basin and West Africa appears to be under greater threat than the Amazon from hunting, largely due to the high demand for bushmeat from urban centres in Africa versus South America (Swamy & Pinedo-Vasquez, 2014). A reduction in the global forest extent (see Section 4.3.5) means that bushmeat hunting is being concentrated into ever smaller forest areas.

4.3.6 Changes in fire regimes

Negative impacts associated with uncontrolled fires has increased over the past few decades, and was especially noticeable during the drought period initiated by the strong El Niño conditions of the 1997-1998 period when an estimated 20 million ha of forest were impacted globally (CBD, 2001). As emphasized in this section (see Section 4.2.6.3 and Chapter 3, Section 3.3.7), this does not mean that all areas effected should be seen as degradation, as periodic fires are a feature of many forest types.

Human use of fire is thought to have been a factor that has caused major change in the dominant vegetation of many areas. For instance there is evidence that the Mediterranean had a far higher dominance of oak forests in the past, but a human induced, altered fire regime from around 7000 years ago has now lead to a dominance of fire tolerant conifers (Zavala *et al.*, 2014). There is evidence that aboriginal use of fire in Australia is what has led to a dominance of fire tolerant eucalyptus over more fire sensitive species. The European settlers in Australia, prevented fires which caused changes to both the vegetation and fire regimes, and this may be responsible for some of the more recent devastating fires (Bowman, 1998; Head, 1989). In the miombo regions of Africa, thinning of trees (for timber, fuelwood or agriculture) leads to increases in grass density, and hence more intense fires. This can then further damage the remaining late succession and fire intolerant trees, resulting in a grassland or open woodland, dominated by early succession, fire tolerant trees (Frost, 1996). In the Great Smoky Mountains of Tennessee, USA., Flatley *et al.* (2015) showed that, over the past few centuries, humans have altered forest succession through active fire suppression. Fire is used as a management tool in many vegetation types to stimulate forage production for livestock, or to alter the ratio of tree to grass (Archibald *et al.*, 2013; Frost, 1996). The baseline (see Section 4.1.4) against which fire impacts are measured will therefore be both critical and complex.

Changing of fire frequency, timing or intensity can change vegetation structure and biodiversity, even in fire tolerant ecosystems. However, of greatest concern is when human activities allow for fires to penetrate biomes where they are not typically present such as tropical forests and peat beds. Peat fires as a result of peatlands being drained have been a major concern in Indonesia. The burning peat can kill all seedlings, sprouts, lianas and young trees as well as overheat stems and roots of mature trees, leading to their death (Nepstad *et al.*, 1999). For example, an estimated 24000 km^2 of peatland burned in Indonesia during the 1997-1998 El Niño-Southern Oscillation drought (Page *et al.*, 2011).

Forest fires in closed tropical rainforest are almost impossible, except during extreme drought conditions. However, human activity such as logging and opening up of the forest, can greatly increase the likelihood of fire. Burning also increases the likelihood of further burning as dead trees topple, increasing the fuel load (CBD, 2001). In some instances, destroyed forest can be replaced with fire tolerant grasslands, which makes forest recovery almost impossible.

Large-area forest fires are the main cause of forest loss in boreal forests. Weather fluctuations cause large differences in interannual fire frequency. Between 2001 and 2007, the average area of fires in Canada was 5,930 ha and 1,312 ha in Russia (de Groot *et al.*, 2013), but Russia has the most extensive overall forest loss (Hansen *et al.*, 2013a). In western Russia alone, 1.5% of forest cover was lost from 2000 to 2005 (Potapov *et al.*, 2011). In North-western USA and Canada, the combination of large bark beetle outbreaks and subsequent fires were comparable in extent (Bentz *et al.*, 2010; Hansen *et al.*, 2016; Simard *et al.*, 2011). de Groot *et al.* (2013) modelled the future of boreal fire regimes in western Canada and central Russia using several global climate models and three climate change scenarios. Their results pointed to more severe fire weather with subsequently greater potential for extreme fire events.

Fire frequency and severity may interact leading, for example, to the population collapse of alpine ash (*Eucalyptus delegatensis*) in the Australian Alps (Bowman *et al.*, 2014). Fire suppression can also lead to unnatural changes, for example forest succession in the Great Smoky Mountains of Tennessee, USA has been altered by active fire suppression over the past few centuries (Flatley *et al.*, 2015).

Increases in fire can be expected to increase loss of life and property and increased financial burden to protect against and suppress fires (Williams *et al.*, 2009). In the United States alone, fire suppression costs have exceeded $1 billion per annum for most of the last 10 years, with last year exceeding $2 billion. The human contribution, beyond climate forcing, needs attention. In the United States, human-caused fires average about 62,000 per annum compared to just 10,500 from lightning. It should also be noted that 66% of the human-ignited fires in the USA occur in the eastern and southern states and many are likely associated with pine plantations. The Chilean fires in January 2017 scorched more than 300,000 hectares, killed at least 11 people, and caused more than $300 million in damage.

4.3.7 Invasive species

Invasive alien species threaten native species and ecosystems on a global scale (World Conservation Union Species Survival Commission, 2000) and pose one of the biggest threats to biodiversity worldwide (D'Antonio & Kark, 2002; Sala *et al.*, 2000) (see also Chapter 3, Section 3.3.8). Any introductions, even in carefully planned biological control programs, are risky, but risk assessment is difficult because it is hard to predict community and ecosystem-wide impacts of introduced species and because introduced species often disperse and may evolve after arrival (Simberloff & Stiling, 1996). Not all invasive aliens have negative effects, some indeed are beneficial (Schlaepfer *et al.*, 2011), but interactions between invasive alien and native species are generally undesirable (Richardson, 2011). The types of invaders include: plants, vertebrates, insects, mites, nematodes, weevils, parasitoids, pathogenic bacteria, fungi, viruses, and algae. Damage can be caused by predation, competition for resources such as space, food and breeding sites (Baillie *et al.*, 2004) above and below ground, and by causing diseases (e.g., Bhaumik, 2013). Not only do invasive aliens affect native species diversity, but they can also modify ecosystems (e.g., Haile, 2016) and cause direct damage to ecosystem services, especially food production (Seguin *et al.*, 2007) (Figure 4.39) and by altering wildfire regimes (e.g. Brooks *et al.*, 2009; van Wilgen *et al.*, 2008).

Figure 4 39 Two invasive aliens.

A Feral cat with bird prey. Feral cats threaten 40 native mammals, birds and reptiles in Australia alone (Dickman, 1996). Photo: courtesy of Vasiliy Vishnevskiy. B Dense, floating water hyacinth (*Pistia stratiotes*), in the Burigana river, Bangladesh. Water hyacinth often clogs waterways and water intakes, deoxygenates water killing most aquatic biota and enhances breeding of insects and diseases harmful to humans (CABI, 2017). Photo credit: www.eniday.com under CC BY-NC-SA 3.0 IT.

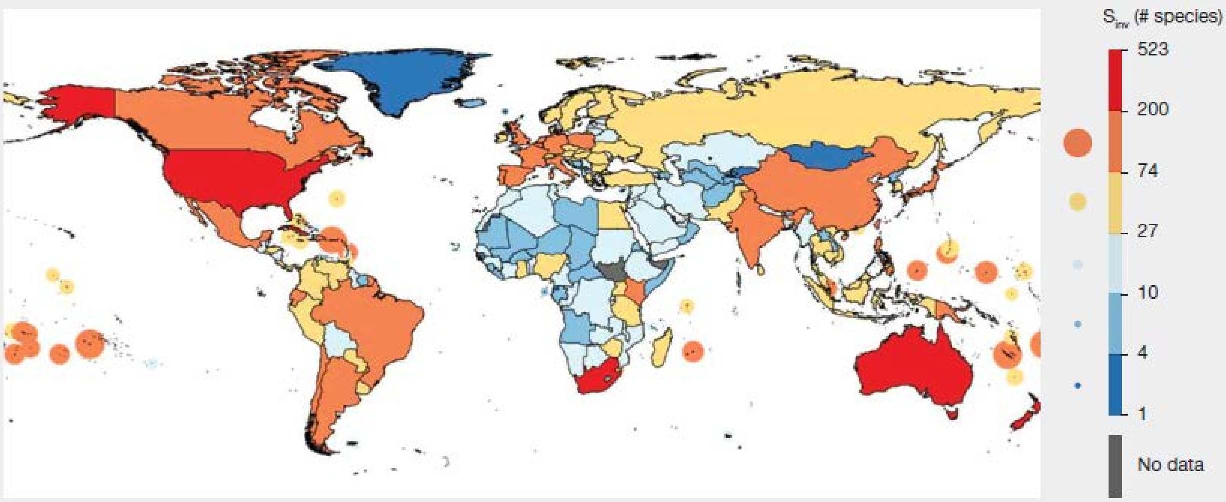
A

B

Human-mediated introductions now dwarf natural dispersal, either intentionally (e.g., introduction into New Zealand of possum, rats, mice, ferrets, stoats, weasels and rabbits, wilding conifer, gorse, crack willow trees, lupines) or, more often, unintentionally – an inevitable consequence of global travel by humans and trade. Bioterrorism may also involve invasive aliens, in most cases pathogenic microorganisms (Meyerson & Reaser, 2003). In total, 13,168 plant species, corresponding to 3.9% of the extant global vascular flora, or approximately the size of the native European flora, have become naturalized somewhere as a result of human activity (van Kleunen *et al.*, 2015) (Figures 4.4 & 4.41). Worldwide, 27% of all threatened animals are imperilled by invasive organisms (Bellard *et al.*, 2016). Invasive alien species are responsible for the stresses on 30% of threatened birds (and as much as 67% on islands), 11% of threatened amphibians, and 8% of threatened mammals sites (Baillie *et al.*, 2004). About 42% of the species on the US Threatened or Endangered species lists are at risk primarily because of alien-invasive species (Pimentel *et al.*, 2005). In the United States alone, there are approximately 50,000 invasives and the number is increasing. The cost to all aspects of the economy in the USA has been estimated at almost $120 billion per year (Pimentel *et al.*, 2005).

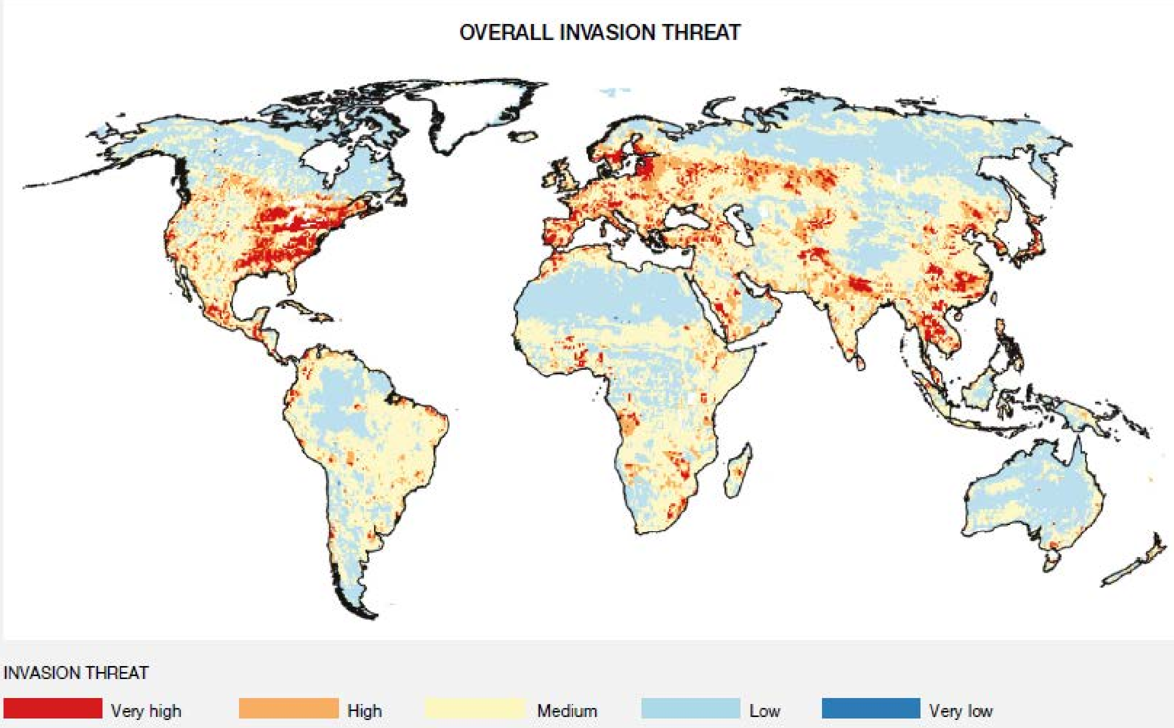
Figure 4 40 Number of invasive alien species per country, excluding overseas territories.

Based on the Global Invasive Species Database (GISD, 2016) and the CABI Invasive Species Compendium (CABI ISC, 2016). Map source: Turbelin *et al.* (2016).



The success of an invasion depends on the ecological characteristics of the potential invader (Moravcová *et al.*, 2015) and also the invasibility of the ecosystem (Olyarnik *et al.*, 2009). While the number of invasives and their impact is large, as a percentage of the native species where they invade the number is small – in fact most invasions fail (Williamson & Fitter, 1996). Invaders often have certain characteristics (Kolar & Lodge, 2001), including: fast growth, rapid reproduction, high dispersal ability, phenotypic plasticity, tolerance of a wide range of environmental conditions, ability to live off of a wide range of food types, association with humans, and ability to occupy inhospitable locales. Global changes, such as climate, land-use change and changes in the nitrogen and carbon cycles, can be expected to open new regions to invasives and allow previously benign species to become invasive (Masters & Norgrove, 2010; Hebertson & Jenkins, 2008).

Figure 4 41 Global invasion threat for the twenty-first century. Source: Early *et al.* (2016).



Invasibility is often associated with anthropogenic disturbance. For example, in China, reclamation of coastal wetlands has contributed towards to invasion by the alien grass *Spartina alterniflora* with serious consequences including indirect impact on bird communities (Yuan *et al.*, 2014), similar to the invasion by *Phragmites australis* in the USA Mid-Atlantic (Saltonstall, 2002). Higher ecosystem diversity is associated with resistance to invasive species (Naeem *et al.*, 2000), but not always (Holle & Simberloff, 2005). Efforts to identify future invaders based on their ecological characteristics have often been ineffective but there is some success in predicting susceptible locales for future invasions (e.g., Korzukhin *et al.*, 2001).

A recent success in biological control is the virtual elimination of a mealy bug (*Phenacoccus manihoti*), from South America, accidentally introduced into Africa where it became a pest of cassava (*Manihot esculenta*), spreading rapidly through many countries. A search in South America found a parasitoid (*Epidinocarsis lopezi*) a natural enemy. After its first release in Nigeria in 1981, *E. lopezi* spread rapidly through neighbouring African countries with enormous economic benefits (Neuenschwander, 2001) and is now regarded as one the most successful programmes in biological control.

However, there are many examples where introductions, intended for biological control, unexpectedly affect non-target species, sometimes creating a worse problem than they were supposed to solve (Louda *et al.*, 2003), such as, for example, the disaster of the cane toad (*Rhinella marina*) in Australia. This animal was intentionally introduced to Australia to control the greyback cane beetle (*Dermolepida albohirtum*) and other pests of sugar cane. It was later discovered that the toads were unable to eat the cane beetles but it thrived by feeding on other insects. They spread rapidly, taking over native amphibian habitat and introduced alien diseases to native species. When threatened or handled, the toad releases a poison harming or killing native species such as goannas, tiger snakes, dingos and northern quolls. Control programs have had limited success (Department of the Environment Water Heritage and the Arts, 2010).

4.3.8 Land abandonment

Land abandonment is a process whereby human control over land (e.g., agriculture, forestry) is given up (FAO, 2006; Munroe *et al.*, 2013). It typically occurs on remote, less productive land of lower agricultural profitability (Munroe *et al.*, 2013), but can also occur on land not considered marginal (Hatna & Bakker, 2011). Trends of land abandonment vary strongly by region (Munroe *et al.*, 2013). Land abandonment has important effects on the provision of ecosystem services (Benayas *et al.*, 2007).

The cover of abandoned land is not static, but rather a succession start – a sequence of changes of the vegetation and soils on land previously disturbed by humans. The actual sequence of changes through a succession is determined by climate and soil type and, in the case of secondary succession, the prior land cover and land use (e.g., cropping, livestock grazing) (Bowen *et al.*, 2007; Plieninger *et al.*, 2014; Queiroz *et al.*, 2014). A relatively steady state after the progressive changes during a succession slow down, known as the "climax", typically has the maximum biomass and biodiversity in the succession, but there are exceptions – maximum carbon sequestration often occurs before the climax. Secondary successions rarely reach the same climax state as a primary succession, and are distinguished by the term "plagio-climax". An example is the impacts of cropping in Mongolia which persists for a long time (Venter *et al.*, 2016). However, the initial disturbance and any subsequent anthropogenic effects (Meyfroidt *et al.*, 2016; Munroe *et al.*, 2013) can keep a succession in an intermediate or even different state, known as a "sub climax". To the extent that cessation of the disturbance is not followed by continued progress to the plagio-climax, the land can be considered degraded (see Section 4.1.2).

The consequences of land abandonment for biodiversity are diverse (Queiroz *et al.*, 2014). It may be followed by passive landscape restoration (Bowen *et al.*, 2007) or ''rewilding'' (Navarro & Pereira, 2012), facilitating the restoration of natural ecosystem processes and species previously excluded by anthropogenic disturbances (Peco *et al.*, 2012). For example, some Mediterranean woodland bird and large mammal populations have benefited from large-scale land abandonment (Blondel *et al.*, 2010). Processes induced by land abandonment include habitat loss (see Section 4.3.1), decrease in habitat patchiness (see Section 4.2.6.5), competitive exclusion of certain species, erosion of newly exposed soil, invasions of non-native plants (see Section 4.3.7), litter accumulation and increased carbon sequestration, soil carbon and carbon stocks (see Section 4.2.3.2), increased wildfires (Benayas *et al.*, 2007) and changes in the local and regional climate (see Sections 4.2.6.1 and 4.2.8). However, abandonment has been found to have mainly negative biodiversity outcomes in Europe and Asia, while positive effects were most common in the Americas (Queiroz *et al.*, 2014).

From the 1700s to 1992 cropland abandonment affected an estimated 1.47 million km^2 worldwide and the rate has greatly increased since the 1950s (Ramankutty & Foley, 1999). Agricultural abandonment has been substantial throughout the 20th century in the Eastern United States, in China, South America and the former Soviet Union (Gutman & Radeloff, 2017), followed by the Western United States, Southern Asia, Europe, Canada, the Pacific developed nations, and Africa (Cramer *et al.*, 2008). Some lands are permanently abandoned, while others may be re-cultivated. Land abandonment is projected to continue under different future scenarios (see Section 4.1.2).

4.3.9 Mining

Mechanisms of land degradation by mining

Mining is the cause of some of the highest intensity anthropogenic landscape transformations, which are in most cases irreversible (Alvarez-Berríos & Mitchell Aide, 2015; Murguía *et al.*, 2016; Sonter *et al.*, 2015a). Mineral extraction is a major driver of land disturbance and contamination to aquatic and terrestrial ecosystems at multiple levels (de Castro Pena *et al.*, 2017; Murguía *et al.*, 2016; Sonter *et al.*, 2015a). Although mining operations are temporary, they create degradation legacies that persist beyond the temporal and spatial boundaries of their direct impacts through the mine life-cycle (Jordan & Szucs, 2011; Lecce & Pavlowsky, 2014; Skaloš & Kašparová, 2012).

The operational life of a mine consists of several phases, each with specific impacts that can occur in sequence or together and often interact cumulatively. These include: geological exploration; construction of infrastructure (e.g., access roads and conveyors, industrial plants for processing and smelting, waste storage, energy facilities, urban services for labour-force); ore extraction by subterranean tunnels, shafts, drifts, pits, surface or mountain top removal, or alluvial dredging (see Section 4.2.6.7); processing (comminution, hydro and pyrometallurgy for concentration, extraction, recovery and refining); waste disposal; rehabilitation and mine closure (Adiansyah *et al.*, 2015).

The risks associated with each phase and the severity of degradation and contamination potential to land and water ecosystems are determined by geologic, geographic and environmental factors (Marsden & House, 2006; Zyl *et al.*, 2002). The geographic location, size of ore reserves and their grades (i.e., ratio between valuable versus undesirable minerals) ultimately determine the footprint of exploration disturbance and of mine waste deposits (Lottermoser, 2010; Sonter *et al.*, 2015b). The geochemistry and mineralogy of ores, metallurgical methods and chemicals utilized for processing and environmental management systems determine the ecological risks of mining waste effluents releases (see Section 4.3.9.2), resilience of disturbed sites (see Section 4.1.2) and challenges for rehabilitation.

In more than 80 countries, “artisanal and small-scale mining” represents a significant source of land degradation and chemical contamination (Swenson *et al.*, 2011). In the world’s poorest regions, this largely informal sector directly and indirectly supports 100 million (Seccatore *et al.*, 2014; Veiga & Hinton, 2002). The rudimentary nature of most artisanal and small-scale mining practices has severe impacts on the structure and chemistry of soils and riverine systems (Figure 4.42). Besides a few local studies, mostly in the Amazon region, (e.g., Alvarez-Berríos & Mitchell Aide 2015; Swenson *et al.*, 2011), there are no global estimates of land degradation by artisanal and small-scale mining. Measuring small-scale forest degradation is challenging due to variable footprint scales (from <10ha to >1000ha) (Austin, 2002). Owing to its widespread occurrence in often remote and pristine ecosystems, and the absence of environmental management (e.g., impact mitigation and mine-closure planning), the severity of disturbances and contamination potential by informal mining is probably as high as by large-scale mining (Sousa *et al.*, 2011; Veiga & Hinton, 2002).

Figure 4 42 Impacts of Artisanal and Small-Scale Gold Mining on floodplains of the Madre de Dios River, in the Peruvian Amazon. Photo credit: Carnegie Airborne Observatory.

Mining Waste

Waste generation is an unavoidable aspect of mining (Zyl *et al.*, 2002) (Table 4.13). Waste materials usually account for more than 99% of the volume of rock extracted (Zyl *et al.*, 2002). The impacts of environmental releases of hazardous waste materials are often considered the most serious aspect of the extractives industry (Martin *et al.*, 2002). Toxic tailings dams are a hazard to local wildlife when not properly maintained (Donato *et al.*, 2007). Releases of hazardous tailings and acid mine drainage effluents from rock spoil dumps have occurred on many occasions throughout the world (Caldwell & van Zyl, 2011; Rico *et al.*, 2008). An analysis of tailings dam failures in the last three decades indicates that, although the overall number of failures has decreased, the number of serious failures has increased (Azam & Li, 2010). Depending on volume, physical properties and chemical composition of the released material, the resulting impacts can be catastrophic (Fernandes *et al.*, 2016; Turner *et al.*, 2008). Irreversible effects occur when large volumes of toxic aqueous slurries and sediments are released into aquatic systems after tailings dam bursts. Immediately after these events, water flow, sediment deposition and toxic effects degrade riparian and aquatic ecosystems locally and downstream of the mine site (Fernandes *et al.*, 2016; Kossoff *et al.*, 2014; Moore, 2015).

Table 4.13 Characteristics of mining wastes generated in each phase of the mining lifecycle, disposal techniques, potential impacts to ecosystems and mitigation actions.

Mine Phase	Waste type	Characteristics	Disposal	Risks to ecosystems	Best management practices
Exploration and extraction	Soils and biomass	Suppressed vegetation and organic soils (horizon A and B) containing nutrients, seed banks, mycorrhiza and pedo-fauna.	Waste dumps or stockpiles	If stored improperly, organic materials may emit greenhouse gases during decomposition	Biomass used for fuel or timber. Rescued germplasm and soils used for reclamation of pits, quarries and waste disposal facilities
	Overburden and spoiling rocks	Underground minerals removed to access the ore	Waste dumps	Large footprint of sterile dumps. Sediments runoff and dust emissions to adjacent terrestrial and aquatic habitats. Seepage of ARD to surface and ground waters	Used for topographic re- conformation of exhausted pits Backfilling of underground mining tunnels Building tailings structures.
Processing, concentration and recovery	Tailings	Gangue separated from the valuable minerals and process chemicals	Tailings storage facilities (dams, heaps).	The Large footprint of sterile and toxic fine materials. Fugitive emissions of volatile toxics. Leakage of toxic chemicals to surface and ground waters	Dry stacking. Degradation or stabilisation of toxic chemicals (e.g., photodegradation or bioengineering). Reprocessing to recover refractory valuable minerals. Reutilization of tailings (e.g., construction materials)
Smelting and Refining	Slags				

In addition to direct impacts of solid sediments to ecosystem structure, hazardous substances and process chemicals in waste sediments and mine waters have long-term effects on watersheds (e.g., cyanide, and heavy metals in sediments or in acid mine drainage effluents) (Macklin *et al.*, 2003). Amalgamation and cyanidation are methods commonly used in Artisanal Gold Mining and a lack of management systems for tailings have allowed the release of mercury and cyanide laden effluents to river systems throughout the developing world (Drace *et al.*, 2016). Artisanal Gold Mining alone released over 800 Tg yr^{-1} of mercury, a neurotoxic heavy metal, to land and water and emitted 700 Tg yr^{-1} of vapours to the atmosphere, representing 37% of the total global mercury emissions (UNEP, 2013). Long term remobilization and transformations of accumulated hazardous substances often create toxicity legacies that may affect both human populations and wildlife for extended periods of time, up to hundreds of kilometres downstream of pollution sources (Guimaraes *et al.*, 2011; Macklin *et al.*, 2006). Prevention and remediation are particularly problematic in the case of transboundary contamination. Although there are no comprehensive reviews of the subject, there have been cases in many parts of the world that have led to international litigation.

4.3.10 Infrastructure, industry, urbanization

Between 2000 and 2040, urban land is anticipated to increase from 2.13 M km^2 or 2.06% to 6.21 M km^2, or 4.72% of all the Earth's terrestrial surface (see Chapter 3, Section 3.3.6 for drivers) (Figure 4.43). The forecast is for the growth to be disproportionally located on land that is suitable and currently available for crop production. This growth would cause the loss of almost 65 Tg of crop production, which may require up to 350,000 Km^2 of new cropland to replace the lost yield. The share of urban land take in cropland areas is highest in Europe, the Middle- East, Northern Africa, and China, while it is relatively low in Oceania and Sub-Saharan Africa (Figure 4.43) (Seto *et al.*, 2012; van Vliet *et al.*, 2017).

Urban agriculture and gardening is an increasing trend, but some of the sites that are being planted were previously used for industrial activities and the soil may contain residual chemicals at a level that could pose health risks. Lead, cadmium, arsenic, zinc, and polycyclic aromatic hydrocarbons are contaminants commonly found in any urban environment (see Section 4.2.4.2) (Heinegg *et al.*, 2002).

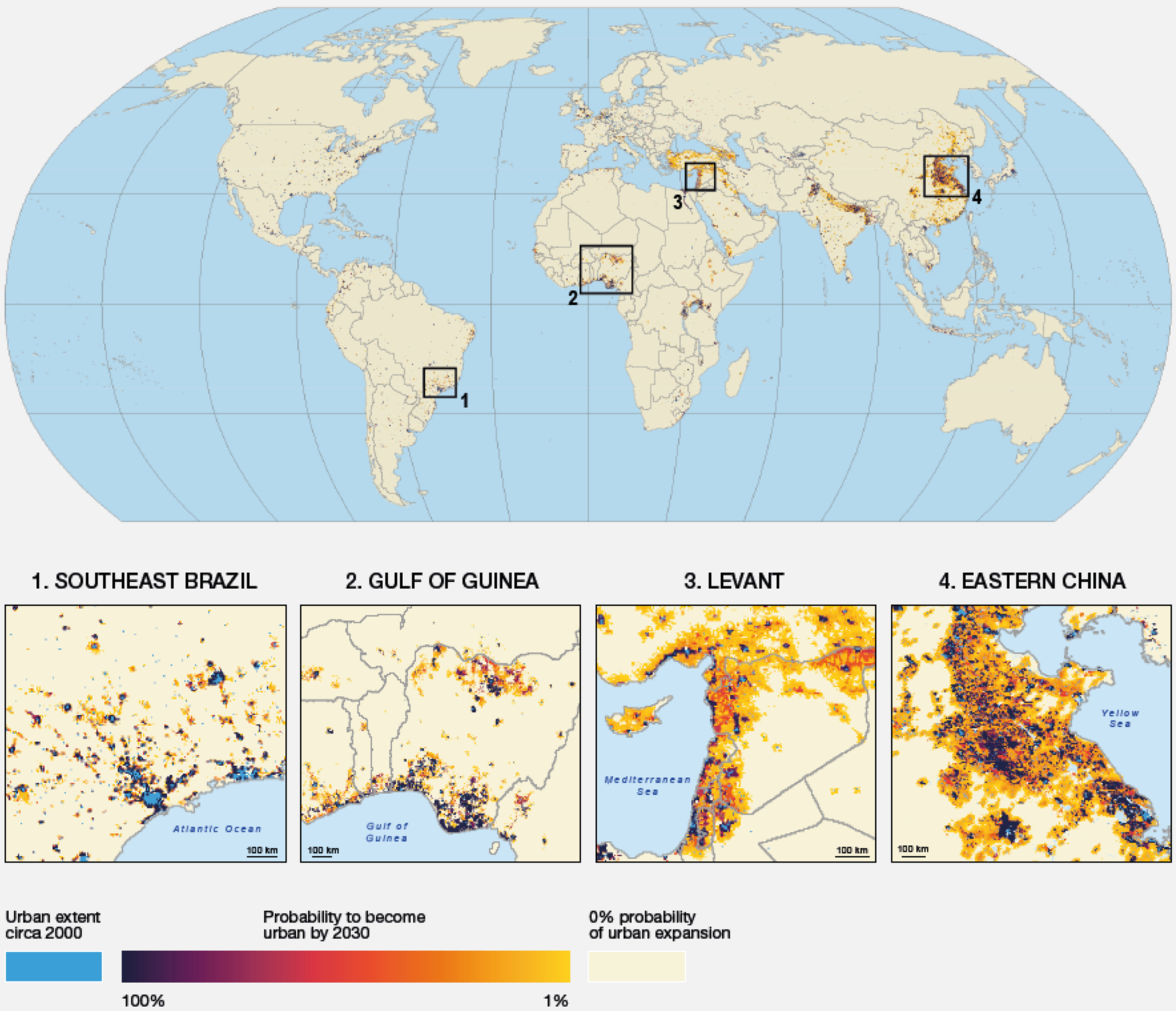
Particularly in richer countries, urban and suburban development has led to high nutrient loads in many streams and rivers due to run off from over-fertilization of lawns and golf-courses, faulty septic systems and cracked sewer pipes.

While urban areas occupy a small share of global land surface (0.5%), they are one of the major sources of carbon emissions (78%), residential water use (60%), wood used for industrial purposes (76%) (Grimm *et al.*, 2008) and various other losses of ecosystem service functions (Wan *et al.*, 2015). The effects are both local and regional – even global.

Urbanization can increase or decrease species richness. Direct causes of biodiversity loss include habitat loss, homogenization, fragmentation, heat island effects, environmental pollution and exotic species introductions and invasions (Fan *et al.*, 2017; Grimm *et al.*, 2008; Kaufmann *et al.*, 2007; Goldewijk & Ramankutty, 2004; Zhang *et al.*, 2008). Changes in landscape configuration as a result of urbanization affects the ranges of species and can enhance local extinction through loss of connectivity (Mitchell *et al.*, 2013; Ng *et al.*, 2013) (see section 4.2.7).

Figure 4 43 **Global forecasts of probabilities of urban expansion 2000-2030.**

There is significant variation in the amount and likelihood of urban expansion. Some regions have high probability of urban expansion in specific locations (1 and 2), and others have extensive, high probabilities of urban growth (3). Much of the forecasted urban expansion is likely to occur in eastern China (4). Source: Seto *et al.* (2012, 2015).



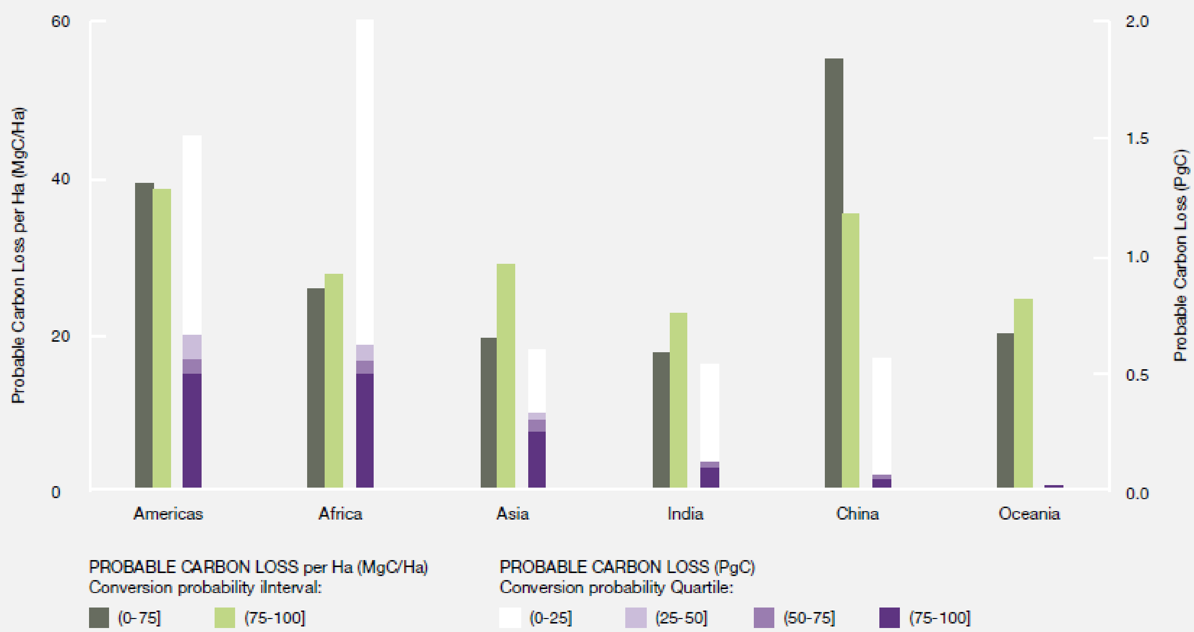
Although urbanization is a major cause of native species extinction (Czech *et al.*, 2000), the nature of urban land use can have a complicated influence on local biodiversity (McKinney, 2002). Some aspects of urbanization cause the loss of species diversity by replacement of the natural biota, while others can promote biodiversity, albeit by the addition of non-native species (McKinney, 2002, 2006) and common weeds. About 65% of studies of plants, 30% of studies of invertebrates and about 12% of non-avian vertebrates found increases in species richness with moderate urbanization (Hope *et al.*, 2003; McKinney, 2008). Urban-rural gradient studies show that, for many taxa, the number of non-native species increases toward centres of urbanization, while the number of native species decreases (see Sections 4.2.6.1 and 4.2.6.3). While diversity in terms of numbers may increase, this is accompanied by homogenization, which threatens to reduce the biological uniqueness of local sites (Blair, 2001; McKinney, 2002).

Interactions between urbanization and ecosystem service provision are multifaceted (e.g., Bennett *et al.*, 2009). Air quality, local and global climate, flood protection, erosion, pollination and recreation can all be changed (Tardieu *et al.*, 2015; Wan *et al.*, 2015). Generally, urban soils are young (Pouyat *et al.*, 2007), having been drastically disturbed and formed of low fertility and imported building materials (Craul & Lienhart, 1999).

At the regional and global scales, ecological processes are affected mostly by atmospheric dispersal of pollutants, but also through water and human transportation. Generation of nitrogen gases such as NO, NO_2, (Grimm *et al.*, 2008; Ramalho & Hobbs, 2012) (see Section 4.2.4.1), increases of CO_2 and other greenhouse gases, as well as trace gases such as O_3, SO_2, HNO_3 and various organic acids (Pataki *et al.*, 2006) (see Section 4.2.4.1) have effects beyond their point sources. In some regions, such as east coast of the USA, deposition of atmospheric nitrogen originating from urban areas as much as 500km away accounts for a substantial portion of the total nitrogen deposited in the catchment feeding the Chesapeake Bay (EPA, 2010).

Net primary productivity is particularly sensitive through loss of vegetation cover on the one hand, but this is somewhat offset by increases in nitrates and CO_2 concentration (see Section 4.2.3.2). For example, in the urban region in the Yangtze River Delta, net primary production decreased significantly due to urbanization processes from 1999 to 2010. Lu *et al.* (2010) and Wu *et al.* (2014) showed with a probability greater than 75% that infrastructure has a strong linear relationship with net primary production over the South-eastern China. Globally, between 20 and 40 MgC/ha of primary production are forecast to be lost (Figure 4.44).

Figure 4 44 Average primary production (MgC ha^{-1}) and total carbon (PgC) loss in 2030 by region within the pan-tropics based on the probability of conversion. Source: Seto *et al.* (2012).



Urbanization has become one of the main drivers of the threat to global biodiversity. Sustainable urban development, including managing and designing urban biodiversity, is therefore of crucial importance to the future of global biodiversity. Good urban planning and the pattern of urban development can reduce the loss of ecosystem services and biodiversity. To promote urban biodiversity and sustainable urban design, the Urban Biodiversity and Design scientific network was founded (Fan *et al.*, 2017; Heinegg *et al.*, 2002; Müller & Kamada, 2011; van Vliet *et al.*, 2017).

4.4 The way forward

4.4.1 Status of biophysical knowledge of land degradation

Since the mid-20th century progress in understanding ecosystem processes has been remarkable – even the term "ecosystem" was adopted less than 100 years ago (Tansley, 1935). Such has been the pervasive use of the term that the non-specialist might reasonably assume that "ecosystem processes" are well-understood. The truth is otherwise; the "ecosystem" has emerged as an extremely complex system, encompassing parts of many fields of the biological and physical sciences. Much is known, but much remains to be discovered. In the context of this Land Degradation and Restoration Assessment, disciplines such as socio-economics, environmental politics and human development need to be aware that the basis of their contributions to the Assessment, that is "degradation", its properties, location, severity and trends, is not a finished story in the biophysical realm and new developments are certain to affect our grasp of its human dimensions. Therefore, there is an urgent need for development of appropriate land degradation and restoration indicators and strengthening of existing measurement and monitoring programmes

Measurement and monitoring of some processes, however, is difficult with current capabilities. This is particularly a problem at scales beyond a single farm or small forest at provincial, national, regional and global scales. As a result, the spatial extent, severity and trends in degradation are largely unknown. The technical capability exists to expand measurement of some aspects of degradation, including monitoring the health of ecosystems, as well changes in their areas (see Sections 4.3.1 and 4.3.4). Satellite-based remote sensing remains the principal means to address the extent and severity of degradation, especially at coarser scales but increasingly at scales approaching 1m. Although, alone, remote sensing will not and cannot, provide all the necessary monitoring, the current phase of rapid development of techniques that use remote sensing is encouraging (Hansen *et al.*, 2013; National Academies of Sciences, Engineering, and Medicine, 2018).

Unfortunately there is a pervasive and alarming trend toward even sparser coverage and even losses of complete environmental and ecological monitoring networks, for example, more than half the global hydrological stations reporting in 1970 were not operating in 2000 (Wahl *et al.*, 1995). A lack of stable, long-term commitment to observations, and lack of a clear transition plan from research to operations, are two frequent limitations in the development of adequate responses to land degradation (Hansen *et al.*, 2013; Karl *et al.*, 1995). This shortage of data is exacerbated by uneven distribution of observation locations. The problem is not unique to poor or developing nations: in many developed countries, long-term monitoring is declining (e.g., Wahl *et al.*, 1995). In addition to this loss of stations, there is an insidious loss of stations having at least 30 years records. These are exactly the stations most needed for detection of trends in the context of climatic change. Clearly, strategies need to be developed and implemented that reverse the declines, fill existing gaps and preserve data with long-records.

These issues are illustrated in the case of extensive livestock production (see Section 4.3.2), which has declined by 50%. Since there are no global maps of stocking or carrying capacity, the location and severity and causes cannot be known – has fodder quality declined or has rangeland been lost to other uses, or a combination of both? For crop agriculture the opposite occurs, global crop yields have increased despite reports of widespread cropland degradation. In this case it is probable that increased use of fertiliser and improved crop varieties may be the cause, not alleviation of degradation, but the answers to these questions are unknown and unknowable with current data.

This section is focussed on the significant obstacles that have to be overcome to improve the current knowledge of the biophysical processes that are at the heart of land degradation.

4.4.2 Gaps in understanding of processes of degradation

4.4.2.1 Types of degradation

It needs to be emphasised that the convenience of the term “degradation” can result in an unconscious notion that it is a phenomenon unto itself. In fact, there is not a single condition, rather there are multiple forms of degradation that reduce ecosystem services: sheet erosion in agricultural fields, water pollution, landscape fragmentation, extinction of species, to name a few, have little in common in their causes by or effects on humans.

Furthermore, there is often confusion over what ecosystem conditions are actually the result of anthropogenic degradation (see Section 4.1.2 and Table 4.1.2). This assessment’s definition of degradation assumes it is anthropogenic in origin and functionally permanent (or in a trend towards permanence) and cannot be restored without massive and uneconomic efforts over decadal time frames. This is a serious consideration and is a critical issue for this assessment (see Section 4.1.2). However, there are other conditions that are frequently misnamed degradation. These include land which is naturally less productive or has a naturally lower biodiversity, land which is susceptible to degradation but not actually degraded, and degradation which is entirely natural, caused by environmental changes that reduce ecosystem services with no human driver. A further cause of confusion is land which is stable, maybe responding to environmental changes and apparently not degraded but which, in fact, entered a state of permanent degradation in the past, prior to monitoring records. In the case of environmental components, there is an urgent need for methods that can reliably decouple impacts of, for example, climate fluctuation from anthropogenic degradation (see Section 4.1.2).

4.4.2.2 Deficient ecological knowledge

Gaps in knowledge of processes

There are many cases where well-known ecological theory is relevant to the processes of degradation, but there is deficient or complete lack of knowledge of the aspects relevant to its degradation, hence how to avoid and reverse it. Examples of key questions for which there are no or only partial answers for many forms of degradation are listed in Table 4.14 (Horne *et al.*, 2017).

Table 4.14 Research priorities to improve ecological knowledge and capability to avoid or restore degraded land.

Key questions
1. How quickly and for how long are ecosystem services perturbed by specific types and durations of disturbance?
2. How are ecosystem services affected by multiple stressors? How should multiple stressors be considered?
3. When is it appropriate to transfer an understanding of biophysical degradation between ecosystems? How do we extrapolate monitoring and evaluation outcomes from one area to another area that has not been monitored?
4. What is an appropriate reference condition in an altered system?
5. Can we determine ecosystem resilience, thresholds that lead to a major change in ecological functioning and condition, and under what circumstance might these occur?
6. Are organisms adapting to degradation? Losses of natural conditions are often assumed to permanently diminish performance, but is there evidence for this?
7. Can measurements at one scale be used at others to match information to user's scale of interest? Are global level data products reliable if they are simply the sum of national and regional-level products? What research methods will allow us to use site-scale data to inform large-scale responses?
8. How can regional or global causes, such, as climate change (4.2.4) or pollution (4.2.8) be included with local causes?
9. Can integrated Assessment models (see Section 4.1.3) predict future human activities that lead to degradation?
10. Changes in the spatial properties of ecosystems can often be measured, but how can deleterious changes in species composition in ecosystems be detected (e.g., agricultural weeds, unpalatable species for livestock, ecological and commercially valuable forest species, and biodiversity changes)?
11. Can below-ground and aquatic biota and environmental conditions (e.g., soil organic carbon, nutrient content, macro-invertebrates in aquatic ecosystems) be developed for regions, beyond the local scale?

Combined use of observations and modelling

To address the functioning, predictability, and projected evolution of the many components of degradation, improved data and its coupling with mathematical models are equally needed (Simmons *et al.*, 2016). These two often reinforce each other – modelling is dependent on observations for development, evaluation, calibration, validation and parameterization and can provide estimates of conditions in places where local measurements are not possible. Current land surface models mostly do not include degraded conditions, and can have both a spatial and temporal resolution that are too coarse for application to small-scale degradation., Advances in Integrated Assessment Modelling that include anthropogenic degradation would be of great value.

4.4.3 Measurement, monitoring and trend detection

4.4.3.1 Routine monitoring

The most direct improvement in assessment of degradation would be a dramatic increase in routine, regular monitoring. The current situation is inadequate. The most basic information about many forms of degradation is rarely available. An apparently simple question such as "what is the biodiversity of an ecosystem?" often can only be answered for limited types of species and few locations. Furthermore, much of the existing information is suspect, mostly based on dated and hard to verify data (Chomitz, 2006) (see Section 4.1.6). Without improved information, assessments are inconclusive. Consequently, policymakers have no objective basis for interventions and interest groups lack a solid basis for dialogue. The meteorological community is far in advance of the data resources available for land degradation and restoration.

Few accurate measurements of species numbers exist for many groups of organisms owing to difficulty in detection (e.g., fungi, beetles, lichens, soil insects). Hence, many global estimates of biodiversity are based on a few, easily observed groups – such as, higher plants, Lepidoptera, birds and larger animals – that are unlikely to be representative of other types of organisms, although they do allow for processes to be tested (see Section 4.4.2). Many biodiversity surveys use habitat as a predictor of species presence (Franklin, 2009), although clearly this is an approximation since even suitable habitats may be unoccupied. The data that are collected by different agencies frequently use widely different methods, such as national crop export statistics and interpolated field measurements that differ in quality and standards. Consistency is critical for application (Weatherhead *et al.*, 2017). While the use of data provided by countries themselves is clearly preferable to override by outside agencies, there are pitfalls to "democratization" of data collection. This issue is recognised by several agencies, such as WOCAT (Nachtergaele *et al.*, 2011) and the Global Soil Organic Carbon Map "Cookbook Manual" (Brus *et al.*, 2017), which propose measurement and record-keeping techniques.

4.4.3.2 Scale

There are inherent problems in extrapolating field measurements at one location to areas. Naïve fitting together of national data at a resolution suitable for global-level assessments can be seriously misleading. In some cases, spatial data of correlated factors can be used as covariates (GSP, 2016), but generally conversion of point data to maps is still primitive**.**

4.4.3.3 Trends and baselines

"Degradation" is a comparative term and implies a comparison with a non-degraded condition. Clearly reference conditions are integral to detection of degradation and trends. Such baselines of ecosystem extent and condition must be explicit (see Chapter 2, Section 2.2.1.1 and Section 4.1.4). Furthermore, attention is needed to the precision of both the baseline as well as the new measurements, so that the statistical significance of comparisons and trends can be known. This is especially important in the case of degradation that is slow and insidious, unrecognizable on an annual basis, but which can lead to total collapse over decades (e.g., declines in biodiversity, gradual invasion by aliens, changes associated with climate change), and which can go undetected or be exaggerated without specifying statistical probability.

4.4.3.4 Degradation indicators

Given the enormous number of ecosystem properties that can be measured even for one type and location of degradation, some method to summarize these into a few key properties is clearly desirable (see Section 4.1.3.1 and 4.1.3.2.). In some cases, this is accomplished by selecting key properties that are themselves affected by contributory factors, including: net primary production which is a result of soil, weather, grazing and other factors; sediment yield which is a consequence of several finer scale erosion factors; a decline in the number of species which reflects aspects of ecosystem degradation; and many others, some for specific purposes (e.g., Hunter Jr *et al.*, 2016). A key, common requirement for these types of indicators is that they are actual ecosystem properties and can, in principle, be measured directly.

A different method for summarization of degradation properties is the use of synthetic indices. These are expressed as numbers or class-membership, as with single-variable indices, but are based on some aggregation of factors. Examples abound: summation of a large number of variables, sometimes normalized (Kumar *et al.*, 2016), sometimes summarized in components from multivariate analysis (Salvati *et al.*, 2015), diversity indices (Weisberg *et al.*, 1997) and so on. There are several reasons why these should be avoided: they are not an actual condition or process, they cannot be measured directly, and do not allow the biophysical or anthropogenic process underlying the degradation to be identified to guide restoration.

4.4.3.5 Data availability

Data users often find it difficult to locate and obtain consistent and comparable data, even within a single country. Nearly all nations collect some data – often in more than one agency – but these frequently have different procedures and rules for making data available, or cannot do so at all since data distribution is not their mission. Some public data archives have been established by international organizations (Biancalani *et al.*, 2013; GEO, 2017; Global Observing System, 2018; UNEP, 2006; WOCAT, 2015), and several national agencies (e.g., ESA, 2017; Government of Canada, 2017; NASA, 2017; NCEI, 2017) and also more specialized agencies (e.g. GFOI, 2017; ISMN, 2017; Ulloa *et al.*, 2017)

However, there is a critical need to expand data collection and monitoring, to enhance the types and coverage of data collected, and proactively to search for existing data and to make them accessible. The current status of national to global biophysical data and its availability for land degradation and restoration is unacceptable. Only with new, intensive, focussed programmes at national and international levels will biophysical research and applications to control degradation advance.

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Chapter 5

Land degradation and restoration associated with changes in ecosystem services and functions, and human well-being and good quality of life

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Executive Summary

Changes in ecosystem services do not affect all people equally; often the poor and vulnerable social groups are those most hurt by land degradation *(well established)*. Landscape transformation often leads to increased wealth by some individuals but loss of subsistence activities and decreased quality of life by others {5.2.2.1, 5.3.2.5, 5.3.2.7}. In these cases, land degradation can exacerbate inequalities between haves and have-nots in both income and in resource access *(established but incomplete)* {5.2.2.1}. In some cases, increasing inequity may increase the risk of conflict *(unresolved)* {5.6.1.2}.

Ecosystem services respond differently to land degradation. Rather than declining uniformly with landscape change, ecosystem services may be lost or added and while the amount most often decreases, some may increase *(well established)* {5.1.1}. Levels of food production, for example, may remain relatively stable but other aspects may experience decline, such as the loss of cultural identity and erosion of indigenous and local knowledge (ILK) for groups who live in close connection to biodiversity and nature {5.3.2.7, 5.9.2}. Alternatively, an increase in cash incomes from harvested wood products may be accompanied by an increase in the disease burden from vector-borne pathogens {5.4.1}.

People live in complex societies – any impact of ecosystem services on quality of life is mediated through institutions and social structures *(established but incomplete)*. Technology and markets can lead to different degrees of substitution for ecosystem services on local or regional scales. Social safety nets and migration for labour opportunities can mitigate the impact of a reduction in ecosystem services {5.2.1, 5.3.3.1, 5.6.2}.

Food security of poor and vulnerable social groups is threatened by land degradation *(well established)*. The conversion of natural ecosystems into agricultural land has generally increased food production, suggesting positive links between degradation and food security (5.3.1). However, the benefits of increased agricultural production are not evenly distributed, while at the same time, access to, and knowledge of, wild food sources has decreased with the same clearing of natural ecosystems {5.3.2.7, 5.9.2.3}. Since indigenous peoples, local communities and poorer rural populations are more dependent on wild-harvested goods, they are likely to be the ones whose food security is most threatened by land degradation {5.2.2, 5.3.2}. Connections between food security, land degradation and the rights and well-being of indigenous peoples and local communities, need to be addressed in the local or place-based contexts where these connections are operating.

Land and environmental degradation is leading to increasing poverty and worsening inequality by negatively affecting the agricultural sector and by reducing access to environmental incomes upon which poor populations are relatively more reliant *(well established)* {5.2.2}. The agricultural sector is disproportionately important for poverty reduction among the rural poor; when land degradation reduces production and employment in agriculture, it will be particularly harmful to the poor *(established but incomplete)* {5.2}. In addition, environmental incomes – incomes from the harvest of non-cultivated products in the natural environment – are relatively more important to the poor in terms of their proportional contribution to household income *(established but incomplete)* and in terms of their ability to act as a safety net for households in times of hardship *(well established)* {5.2.2.1}. When land degradation reduces access to or availability of environmental incomes, poor households will be harmed to a greater degree than wealthier ones. Environmental incomes have generally been shown to reduce inequality within communities; reducing access to them will tend to exacerbate inequality *(established but incomplete).* There is a long-standing argument about what is called the “downward spiral” relationship between poverty and land and environmental degradation: this idea posits that poverty causes land and environmental degradation which in turn worsens poverty {5.2.2.2}. Although there is evidence suggesting that, in some cases, poverty and degradation are indeed inextricably linked, this is not always the case *(unresolved).* Social, economic, and political institutions at local and national scales can influence processes and outcomes of the relationship between poverty and degradation *(well established).*

Human security is negatively affected by land degradation, particularly in cases where degradation leads to involuntary migration or where it exacerbates the risk of violent conflict *(established but incomplete)*. Land degradation is rarely a direct cause of violent conflict and the main cause of involuntary migration; however, it can act as a threat multiplier and increase the likelihood of both occurring. In populations dependent on dryland pastoralism, a declining resource base driven by land degradation has been shown to be associated with higher rates of violence {5.6.1}. Land degradation that reduces livelihood opportunities can drive involuntary migration, further reducing quality of life {5.6.2}.

Human health is affected in many ways by land degradation, including by an increasing burden of infectious disease (especially vector-borne diseases like malaria), an increase in unsafe drinking water as pollutants are released and the purifying services from forests and wetlands are lost, and a loss of future potential to find new pharmaceuticals *(established but incomplete)* {5.4}. The loss of biodiversity may have an irreversible cost to human health, as the benefit it provides through the dilution of infectious disease emergence is lost {5.4.2}. Short-term health costs of development projects may be outweighed by improved access to healthcare in the long term, but health burdens, as with many changes in quality of life, will disproportionately affect poorer segments of society as well as cultural minorities who have less access to quality medical care, and benefit less from development in the long term {5.4}. Impacts of land degradation on mental and physical health are difficult to study, but almost certainly have subtle costs that are particularly severe in urban environments without green spaces {5.4.6}. At any stage of development, the restoration of healthy ecosystems is likely to reduce infectious and non-infectious disease burden, buffer the emergence of new diseases, and improve mental and physical health.

Being connected to nature or even simply viewing natural scenes positively affects psychological well-being in many ways (*well established*). Stress levels are decreased by looking at natural sceneries or by walking in green spaces compared to those in urban environment (*established but incomplete)* {5.4.6}. Also, working memory and self-esteem are increased by engaging in green areas such as parks and gardens (*established but incomplete*). Land degradation, especially rapid urbanization, is affecting the mental health of urban dweller by reducing connection to nature *(inconclusive)* {5.4.6.1}. In rural areas, land degradation affects mental health by loss, disconnection or degradation of nature *(inconclusive).*

Water security is directly linked to human health, food and energy security, yet ecosystems that help maintain freshwater supplies, such as forests and wetlands, continue to be degraded *(well established)*. The global supply of renewable freshwater is highly geographically variable and is declining overall {5.8.1}. Land degradation reduces freshwater supplies and quality, and compromise human health through activities related to intensive agriculture, overgrazing, and urbanization. The conversion of forests and wetlands, which cumulatively supply an estimated 75% of the world’s freshwater, increases risks to water security. Currently, an estimated 80% of the world’s population lives with incident threats to water security (*established, but incomplete*), and 66% of the global population face severe water scarcity at least one month per year. Sub-

Saharan Africa has some of the lowest access to water and sanitation, lagging behind other developing countries. Globally, the impacts to human health are enormous, with an estimated 1.6 million deaths per year due to a lack of safe drinking water and poor sanitation and hygiene, and 1.8 billion people are exposed to drinking water contaminated with faeces *(well established)* {5.8.1, 5.8.2.2}. Urbanization intensifies demands for water and sanitation, leading to reduced supplies of ecosystem services per capita {5.8.2.4}. Climate change is expected to exacerbate this, increasing water shortages for an estimated 100 million additional urban dwellers by 2050. Restoration of degraded lands through reforestation, increasing soil organic matter in agricultural lands, and the wise use and restoration of wetlands, floodplains and riparian zones can help reverse trends in water scarcity and security {5.8.3}.

Access to energy improves human well-being and quality of life, but the type of energy and the mode of access determines the severity and location of the associated land degradation *(well established).* Centralized, large-scale fossil fuel-based electricity generation have significant benefits to grid users, but also intense local negative impact through extractive mining, and regional and global impacts through pollution and greenhouse gas emissions {5.7.1}. There are 2.7 billion people worldwide without access to grid electricity that uses traditional biomass, exposing themselves to significant health impacts through indoor air pollution {5.7.2.1}. Harvesting biomass for fuel use has a significant negative impact on ecosystem services from forests; in addition, the labour burden of biomass harvesting falls disproportionately on women {5.7.2.1}. Biofuels may increase agricultural commodity prices but the extent to which this effect materializes is dependent on policy implementation, and projections remain contested due to the complex nature of the models used and the lack of data on supply and demand elasticity in developing countries *(established but incomplete)* {5.7.2.2}.

Land degradation can negatively affect cultural identity of communities *(well established)*. Well-conserved ecosystems play symbolic and identity-supporting roles to indigenous peoples and local communities (IPLC), other cultural groups, and individuals around the globe {5.9.2}. The degradation of these ecosystems can affect individuals and groups in their sense of self and in their spiritual and psychological well-being {5.9.1}. Interaction with nature is central to the traditions and identity of many cultures around the world; land degradation in its extreme form threatens cultural identity and the interlinked social, cultural and physical reproduction of these groups at a fundamental level {5.9.2.3, 5.10}.

Case studies of land restoration have shown important livelihood benefits *(well established)*. Community-based restoration initiatives can be cost-effective as well as socially and ecologically successful, showing both improvements in livelihoods and in community support {5.2.3}. Restoration efforts that have best incorporated indigenous and local knowledge in their design and implementation have often shown the greatest success {5.2.3.3, 5.3.3}.

5.1 Introduction

Over the past century, quality of life has greatly improved for the vast majority of individuals while, at the same time, humans have become the driving force shaping Earth's climate and land surface. In the last quarter century alone, the Human Development Index has increased in all regions in the world (Ciara Raudsepp-Hearne *et al.*, 2010), while at the same time the amount of extant wilderness has decreased to less than 25% of the Earth's land surface (Watson *et al.*, 2016). While the impacts of anthropogenic climate change on human quality of life are the focus of Intergovernmental Platform on Climate Change (IPCC) assessments (see in particular Working Group II on Impacts, Adaptation, and Vulnerability of the IPCC's Fifth Assessment Report (IPCC, 2014)), the focus of this chapter is on understanding how humanity's interactions with the Earth's land surface aids, impacts, and influences human quality of life.

"Land degradation" is defined, for the purposes of this assessment, as the many processes that lead to a decline or loss in biodiversity, ecosystem functions, or ecosystem services in any terrestrial and associated aquatic ecosystems that cannot fully recover unaided within decadal time scales (see Chapter 1 for full definitions related to land degradation). "Degraded land" takes many forms: in some cases, all biodiversity, ecosystem functions and services are adversely affected; in others, only some are negatively affected, while other ecosystem services have been increased. Transforming natural ecosystems into human-oriented production ecosystems – for instance agriculture or managed forests – creates benefits to society but simultaneously results in losses of biodiversity and non-prioritised ecosystem services.

To most effectively and easily understand and quantify the role that human's interactions with the land surface play in impacting quality of life, the chapter is oriented around socially and politically important aspects of human well-being: poverty (Section 5.2), food security (Section 5.3), physical and mental health (Section 5.4), hazards and disasters (Section 5.5), human security (Section 5.6), energy security (Section 5.7), water security (Section 5.8), and non-material benefits from nature and culture (Section 5.9).

What emerges in terms of the relationship between land degradation and restoration on human quality of life is that degradation has diverse and wide-reaching impacts on quality of life that cause declines in economic opportunity, food security, physical and mental health, water security, safety from conflict, and personal and cultural identity. These impacts, however, are not evenly distributed; they tend to affect poor and marginalized populations in particular, because those populations are most dependent on direct use of environmental resources and tend to have worse access to social safety nets and to market alternatives (discussion of terms "poor" and "poverty" as used in this Chapter in Section 5.2). Patterns and impacts of land degradation are also mediated by social and political institutions that can serve to mitigate the negative effects of degradation or can serve to further marginalize those who are worst-affected. Restoration can be an effective way of reducing or reversing some of the effects of land degradation on populations. When done effectively and with local engagement and buy-in, restoration can improve both ecological function and human quality of life.

While these findings certainly support the importance of the integrity of land-based ecosystems in maintaining human well-being, given the global footprint of humanity, they suggest a need for a larger conversation concerning how humanity should discuss and conceive of its relationship to nature. We return to this discussion at the end of this chapter (see Section 5.9.3).

5.1.1 Emergent themes

The assessment that follows of the relationship between land degradation, restoration, and quality of life led to several emergent themes. Four of those themes, in particular, are common threads throughout this discussion. They are detailed below.

First, there are many aspects to a good quality of life that are influenced by multiple factors, including those outside of natural systems (Pascual *et al.*, 2017). In many cases, anthropogenic assets as well as institutions and governance play a central role in mediating how land degradation and restoration impacts human quality of life, and in particular, whose quality of life is impacted. Social safety nets, labour markets, and commodity markets will allow many individuals to insulate themselves from the negative impacts of land degradation; however, poor and marginalized populations will often be unable to similarly insulate themselves, and will be more severely affected by degradation. Land degradation may thus serve to exacerbate inequality as it negatively affects the vulnerable, while leaving wealthier populations less affected (see in particular Sections 5.2 and 5.4, and Figures 5.1 and 5.6). In addition, land degradation may affect different aspects of an ecosystem to varying degrees, and this itself may lead to varied impacts among people. Landscape conversion from forest to agricultural production may increase local labour opportunities and food production, but will decrease the availability of wild-harvested resources. Because some populations are more dependent on the harvest of wild resources while others are better able to take advantage of agricultural employment, the benefits of the conversion will vary greatly.

A second key theme, as is made clear in Chapter 4, is that land degradation does not affect nature in a uniform way. Thus, the impacts of land degradation on ecosystem services and the resulting quality of life is also not uniform among ecosystem services and their impacts. Ecosystem services can best be thought of as “bundles” (Raudsepp-Hearne *et al.*, 2010) with groups of services working together to impact quality of life. In this conception, the total effect of land degradation on a given aspect of human quality of life – for example, food security – will be determined by the combined and synergistic effect of changes in several different ecosystem services, such as soil health, water availability, and pollinator abundance. In some circumstances, land degradation can lead to improvements in some aspects of human interactions with the environment (e.g., decrease in malaria vectors with the elimination of wetlands) while at the same time decreasing benefits received from nature (e.g., loss of the water filtration potential of the same wetland) (Horwitz *et al.*, 2012). These situations will result in a complex pattern of change in human quality of life, with some people benefiting while others losing out.

The third theme that must be considered when assessing the impact of land degradation and restoration on quality of life is the role of an individual’s or society’s worldview. As is made clear in Chapter 2, worldview strongly impacts conceptions and perceptions of land degradation and restoration. In addition to the aforementioned relationships among land degradation and restoration, ecosystem services, and human quality of life, it is necessary to consider how differences in worldview will affect the aspects of life that are most valued. Ecosystems in an undegraded condition may have cultural and spiritual importance that goes beyond a discussion of material benefits. The importance of worldview in determining how land degradation and restoration impacts quality of life is made most clear in the section on the non-material and cultural benefits of nature (see Section 5.9).

Finally, to give a full account of the impacts of land degradation and restoration on human quality of life requires incorporating knowledge and information that goes beyond that found in the scientific peer-reviewed literature primarily published in English. It requires integrating the wealth of knowledge found in indigenous and local knowledge (ILK) systems of indigenous peoples and local communities (IPLC) (see Box 5.1 for definitions). Thus, throughout the chapter, we have included numerous examples and case studies from local communities and cultural minorities that illustrate the material and non-material impacts that land degradation and restoration is having on these peoples. While not a complete and comprehensive assessment, these examples and case studies provide key, often unreported, information on the profound ongoing impacts of land degradation on the livelihoods of hundreds of millions of individuals living around the globe.

Box 5.1 Definitions used for indigenous and local knowledge systems

For the purpose of this chapter, we adopt the approach to working with indigenous and local knowledge in the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES), approved during its fifth session, which includes the following definitions, key terms and concepts (IPBES, 2017b):

Indigenous and local knowledge systems are understood to be dynamic bodies of integrated, holistic, social and ecological knowledge, practices and beliefs pertaining to the relationship of living beings, including people, with one another and with their environment. Indigenous and local knowledge is grounded in territory, is highly diverse and is continuously evolving through the interaction of experiences, innovations and different types of knowledge (written, oral, visual, tacit, practical and scientific). Such knowledge can provide information, methods, theory and practice for sustainable ecosystem management. Indigenous and local knowledge systems have been, and continue to be, empirically tested, applied, contested and validated through different means in different contexts.

Maintained and produced in individual and collective ways, indigenous and local knowledge is at the interface between biological and cultural diversity. Manifestations of indigenous and local knowledge are evident in many social and ecological systems. In this context, the approach understands “biocultural” as describing a particular state resulting from the interaction of people and nature at a given time and in a given place and “biocultural diversity” as a dynamic, place-based aspect of nature arising from links and feedback between cultural diversity and biological diversity.

5.2 Poverty and livelihoods

Land degradation is a major contributor to rural poverty in developing nations, in part because the poor depend disproportionately on agriculture and livestock-rearing for their livelihoods. Degradation constrains agricultural potential and therefore limits the potential for livelihood improvement by individuals living on degraded lands. Degradation also affects poverty via impacts to livestock that can be pronounced, especially in drylands. The livelihoods of roughly one billion of the world’s poorest people depend to some extent on livestock, while about 1.1 billion people in total are estimated to be employed by the livestock sector (Rojas-Downing *et al.*, 2017). In addition to the impact of land degradation on agricultural and livestock sectors, degradation negatively impacts natural ecosystems and thus limits potential for the extraction of non-cultivated goods. Extraction of non-cultivated goods from natural ecosystems, so-called “environmental incomes” are disproportionately important to the rural poor; to the extent that it limits environmental incomes, the negative impacts of land degradation will be felt most keenly by the poor (Angelsen *et al.*, 2014; Chhetri *et al.*, 2015; Jagger, 2012; Pouliot *et al.*, 2012; Vedeld *et al.*, 2007).

Throughout this chapter, we will use the terms "poor" and "poverty" to describe a state of deprivation that is multi-dimensional in nature. Although some international definitions of poverty are based solely on income – the $1.25 per day poverty line, for example – poverty in a broader sense includes other manifestations of deprivation and exclusion such as hunger and malnutrition, limited access to education and other basic services, social discrimination and exclusion, and constrained participation in decision-making. Estimates of poverty that are based on income and other economic measures are often the easiest to measure and as such, the data that are available to report often shows a bias towards those aspects of poverty. However, the discussion in this chapter will take a broader view that is consistent with the approach taken by the Sustainable Development Goals (UN General Assembly, 2015)

National estimates of the economic impact of land degradation are not available for all countries, but in those countries where data is available, the impacts of degradation are immense. Degradation costs $2.5 billion annually in Tanzania and $300 million in Malawi, representing 15% and 10% of GDP respectively (Kirui, 2016). In India, degradation has been estimated to cost the national economy 5.7% of its GDP annually (World Bank, 2013). In Central Asia, land degradation costs the countries of Kazakhstan, Kyrgyzstan, Tajikistan, Turkmenistan and Uzbekistan a total of about $6 billion annually (Mirzabaev *et al.*, 2016). In Ghana, land degradation decreased agricultural incomes by $4.2 billion between 2006 and 2015 while increasing the national poverty rate by 5.4% (Diao & Sarpong, 2011). Conserving biodiversity and maintaining ecosystem integrity has also been shown to have a large potential for reducing poverty, exceeding $1 per person per day for more than 300 million of the world's poorest people (Turner *et al.*, 2012).

The centrality of agriculture as an income source for the rural poor means that the constraining effect of land degradation on agricultural incomes has a particularly pronounced impact on poverty rates. Part of this issue results from the frequent geographic overlap of poverty and degradation: in 2003, one quarter of individuals living in poverty worldwide – 1.3 billion people – lived on fragile lands (i.e., areas that were particularly vulnerable to degradation) (World Bank, 2003). It has been shown in data from 42 developing countries that gains in the agricultural sector are 2.5 times more important to incomes of individuals in the bottom 30% of the income distribution than are gains in the rest of the economy (World Bank, 2008). Improvements in rural poverty rates have had a central contribution to reduction in poverty rates generally: 45% of the reduction in poverty globally between 1993 and 2002 resulted from the reduction in rural poverty (World Bank, 2008). This is largely because of the concentration of poverty in rural areas, with 75% of poor people residing in rural areas while only 58% of the total population is rural.

5.2.1 Spatial association between poor people and marginal land

Part of the evidence for a relationship between poverty and land degradation is the fact that poverty tends to be concentrated on land with lower productive potential. Additionally, evidence suggests that poverty is also more prevalent on land that is particularly vulnerable to further degradation in productivity. An analysis of 76 developing countries found that those countries with a higher proportion of the population living on fragile lands – land vulnerable to degradation – had a higher overall proportion of rural poverty (Barbier, 2010; Barbier & Hochard, 2016). In countries with less than 20% of the population living on fragile lands (12 countries), the average rate of rural poverty was 36.8%. In contrast, in countries with more than 70% living on fragile lands (three countries), the rural poverty rate was 54.7%. There is also an association between proportion on fragile lands and national GDP per capita: in the categories mentioned above – less than 20% and more than 70% on fragile lands – the average GDP per capita was $3,326 and $671, respectively. The proportion of population living on fragile lands varies greatly, from only 11.1% in the relatively wealthy OECD countries to 39.3% in Sub-Saharan Africa. The developing country average is 25% of the population living on fragile lands (Barbier, 2010). In Sub-Saharan Africa, the poor are over-represented in drylands: while about 50% of the total population of Sub-Saharan Africa lives in drylands, the percentage is of the poor who live in those areas is 75% (Walker *et al.*, 2016).

Within countries, populations living in areas that are remote from urban centres tend to have a much higher poverty rate than populations living in less remote areas (Bird & Shepherd, 2003; Sunderlin *et al.*, 2005; World Bank, 2008). However, the link between poverty and specific environmental problems is complex and context-dependent. In a study in Southeast Asia, Dasgupta and co-authors (2005) found that patterns were quite different among Cambodia, Vietnam, and Lao PDR. In Lao PDR, poverty was indeed spatially correlated with all indicators of environmental damage measured: deforestation, risk of soil erosion, water pollution, and outdoor air pollution. The spatial links were not as strong in Cambodia, where there was no correlation with erosion risk, or in Vietnam, where the only strong correlation was between poverty and risk of soil erosion due to slope.

The fact that rural poverty tends to be worse in areas with more degraded land does not necessarily prove that land degradation causes rural poverty. The two issues – degradation and poverty – are both caused by a complex set of physical, social, and economic processes that may themselves be linked spatially. The spatial association between poverty and degraded lands could also be explained by the fact of other external factors leading to both poverty and degradation. For example, poor governance can lead to poor environmental outcomes as well as to more limited economic opportunities (Black *et al.*, 2011). We might therefore expect that both and degradation and poverty would tend to be concentrated in areas with low capacity for effective governance.

The poor governance scenario above would explain a spatial association between poverty and degradation without implying any causal relationship between the two issues. However, there are also reasons to believe that poverty may cause degradation and vice versa. As an overview, there are four broad reasons that may explain the association of poverty with degradation (Duraiappah, 1998; Markandya, 2001):

1. Higher rates of poverty result in more degradation;
2. Poor people are more likely to live in areas that are degraded, and the degradation itself contributes to continuing poverty;
3. External factors – for example market or institutional effects – lead to both higher poverty and higher degradation; and
4. Policies that result in land degradation disproportionately hurt the poor.

There are lines of evidence supporting each of these four effects, and likely elements of each is true in some situations. Throughout the following text, examples and evidence supporting each will be discussed.

5.2.2 Importance of environmental incomes

Environmental income is the livelihood benefit to households stemming from the consumption, barter, or sale of goods harvested freely from the non-cultivated environment (Angelsen *et al.*, 2014; Jagger, 2012). Environmental incomes are an important part of household livelihood portfolios in many rural areas, and are particularly important to the poor and to populations that have recently experienced a livelihood shock. Various studies have assessed the contribution of environmental incomes to rural livelihoods. Among 521 households across seven districts in western Uganda, households derived 26% of their income from forests, fallows, wetlands, grasslands, and non-crop species on agricultural land (Jagger, 2012). In Ghana and Burkina Faso, households derived averages of 23% to 36% of their incomes from environmental sources, with collection of wild foods, fodder for livestock, and fuelwood being the most important individual sources (Pouliot *et al.*, 2012). Meta-analyses suggest that these examples are not atypical for the average level of dependence of rural households on environmental incomes. In a survey of 51 studies from 17 countries on forest incomes specifically, it was found that an average of 22% of household incomes came from forests (Vedeld *et al.*, 2007). In one very large dataset – 7,978 household interviews in 333 communities in 24 tropical and sub-tropical developing countries – households derived 27.5% of their incomes from the environment, generally with 21.1% from forests and 6.4% from non-forest environments (Angelsen *et al.*, 2014). Forest incomes are shown to be particularly dominant in many studies of environmental incomes; however, they are not always so. The study in Ghana and Burkina Faso, for example, found about twice as much dependence on non-forest environmental resources than on forest-based ones (Pouliot *et al.*, 2012).

Although the use of environmental incomes varies greatly, there are some common patterns that emerge when studies from around the world are compared. One of the most consistent findings is that relative reliance on environmental incomes is highest among the poor (Shackleton & Shackleton, 2006; Hunter *et al.*, 2007; Barbier, 2010, 2015; Chhetri *et al.*, 2015; Jagger, 2012; Pouliot *et al.*, 2012; Vedeld *et al.*, 2007). This is not to say that the poor extract the highest total environmental incomes – in many cases, the total income that households gain from harvest of environmental resources is higher for wealthier families – but rather that the poor generally obtain a higher proportion of their total incomes from non-cultivated environmental sources. Additionally, the types of incomes obtained from the natural environment often vary between poorer and wealthier households. Wealthier households are more likely to extract high-value processed products, whereas poorer households are more likely to extract goods for household consumption that have lower market values (Jagger, 2012). Generally, wealthier households are more likely to have a higher proportion of cash income from the environment, but poorer households are more likely to have a higher proportion of subsistence income from the environment (Angelsen *et al.*, 2014). This distinction between high market value products extracted for cash and low market value extracted for subsistence has important implications for the livelihood significance of environmental incomes as well as for the sustainability of the harvest of wild resources. Those implications will be discussed in depth below.

The fact that environmental incomes provide a greater proportion of household incomes for poor households than for wealthier ones means that these incomes play an equalizing role within communities (Angelsen *et al.*, 2014; Vedeld *et al.*, 2007). This equalizing role of environmental incomes has been shown in Mexico (López-Feldman *et al.*, 2007), Malawi (Fisher, 2004), Zimbabwe (Cavendish & Campbell, 2005), Uganda (Jagger, 2012), and Nepal (Chhetri *et al.*, 2015). The Gini index – a common metric of wealth or income distribution where higher values represent higher inequality – was shown to increase significantly in every region of a global study (Asia, Africa, and Latin America) when environmental incomes were excluded (Angelsen *et al.*, 2014). When types of environmental incomes are examined, the resources that are the most important to the poor are consistently the most equally distributed: cash income from processed forest products (incomes captured disproportionately by the wealthy) has been shown to have the highest Gini indices (most unequal), while the subsistence resources and unprocessed forest resources that the poor depend upon have much lower Gini indices (Jagger, 2012).

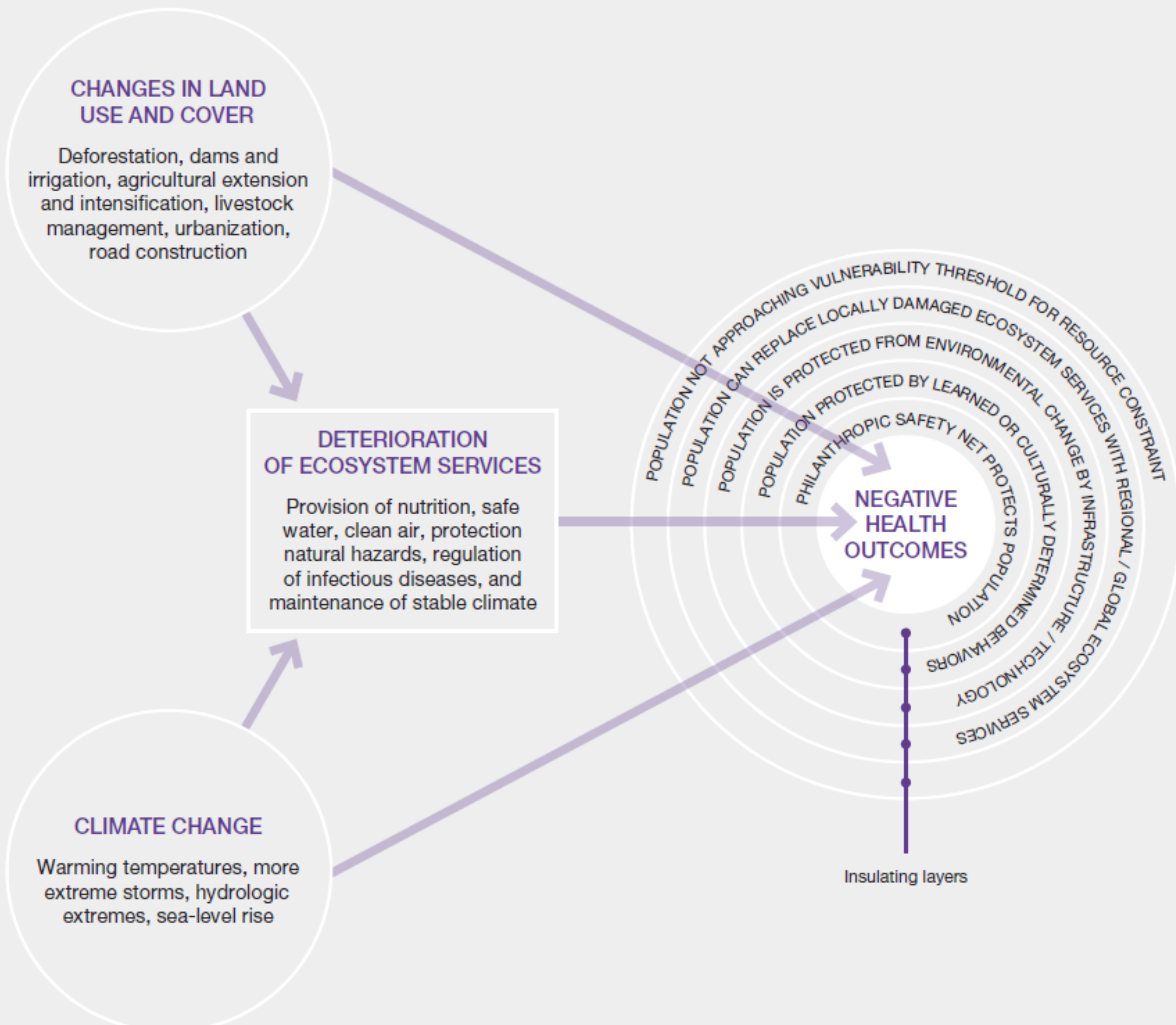
Degradation of land resources that households use as a source of environmental incomes, because of the equalizing nature of those incomes, leads to an increase in within-community inequality and poverty. A reduction in the availability and quality of natural resources for harvest tends to harm the poor significantly more than the wealthy (Barbier, 2010; Dasgupta & Mäler, 1996; Perrings, 2014).

5.2.2.1 Environmental incomes: safety nets and potential pathways out of poverty

Environmental incomes can play three different roles in supporting livelihoods: (i) they can provide safety nets that insulate households from shocks; (ii) they can support current consumption; and (iii) they can provide a pathway to alleviate and potentially escape poverty (Angelsen & Wunder, 2003; Cavendish, 2003; Vedeld *et al.*, 2007). The first of these roles, the ability to buffer households from shocks, is particularly important for the poorest households who have few other assets to draw upon in times of need (Barbier, 2010, 2015). This pattern is seen in many contexts. Households that had experienced income shocks in preceding 12 months were found to have higher environmental reliance, both from more environmental income and from lower total income (Angelsen *et al.*, 2014). Following extreme shocks, for example the loss of an adult family member, goods harvested from the environmental can play a critical role in maintaining household food security (Hunter *et al.*, 2007). Environmental incomes play an important role as gap-fillers for households who are between seasons of agricultural production (Wunder *et al.*, 2014). The causal relationship between shocks and reliance on environmental incomes has been demonstrated as well (i.e., to illustrate that it is not simply a correlation): experiments have shown that individuals who were randomly assigned to receive an income package (i.e., a reverse shock) extracted less environmental incomes (Fisher & Shively, 2005). This is a parallel but reverse finding to other work that has shown adverse shocks lead to households increasing their rates of forest product extraction, although only temporarily (Pattanayak & Sills, 2001; Takasaki *et al.*, 2004).

Figure 5 1 Impacts of land degradation on quality of life are mediated by household assets, social structures, and institutions.

Some population groups have layers of institutions and resources that are able to successfully buffer them from losses of ecosystem services. Other population groups that do not have these insulating resources and institutions will be more negatively affected by deterioration of ecosystem services as caused by land degradation. Source: Myers *et al.* (2013).



The role that environmental incomes play as buffers against shocks has been shown to be relatively more important for poor households than wealthier ones (Hunter *et al.*, 2007; Angelsen *et al.*, 2014; Wunder *et al.*, 2014). Poor households are generally less able to recover from environmental change, as has been shown by recovery rates from natural disasters in both Ethiopia and Honduras (Carter *et al.*, 2007). One reason for this reduced ability to cope with shocks is simply that the poor are less able to engage in a common coping strategy during times of hardship: selling assets for additional income (Wunder *et al.*, 2014). A lack of assets means that the poor have a limited cushion to rely on when their usual livelihood activities are curtailed (Scherr, 2000). This has been demonstrated in the field with poor households observed as being more likely to turn to forest product collection after a family shock such as illness (McSweeney, 2004), more likely to turn to non-timber forest product collection after a policy that reduced income (L'Roe & Naughton-Treves, 2014), and more likely to turn to forest product gathering and to fishing to smooth incomes crop losses due to floods (Takasaki *et al.*, 2002, 2010).

The safety net function played by environmental incomes – the ability to buffer from shocks and to prevent further hardship – is in many contexts important to the rural poor and should be safeguarded. However, it is less clear that environmental incomes are generally effective at reducing poverty (Wunder, 2001). Indeed, one of the characteristics that makes many environmental incomes “pro-poor” is that they have low barriers to entry for harvest (Sunderlin *et al.*, 2005). However, this low cost of extraction is associated with, and likely partly the cause of, relatively low economic returns of many harvested products, for example non-timber forest products (Angelsen *et al.*, 2014). Some researchers have therefore suggested that in some cases, reliance on environmental incomes can even serve as a “poverty trap,” a situation where poverty leads households into a livelihood mode that is very likely to perpetuate poverty (Barbier, 2010).

5.2.2.2 The “downward spiral” and its critiques

The particular reliance of the poor on resources directly harvested from the natural environment is one part of the relationship between poverty and land and environmental degradation. It has been suggested that there is a deterministic relationship between poverty and degradation, where poor people are forced to degrade the land base out of necessity, and then the degraded land base further exacerbates poverty. This “downward spiral” narrative was dominant for a period in much of the thinking about land and environmental degradation, particularly in the 1980s and early 1990s. It was argued with particular impact by the Brundtland Report, *Our Common Future,* produced by the World Commission on Environment and Development (WCED, 1987), and promoted over the following decade by other international organizations (UNEP, 1995; World Bank, 1992).

There is indeed a spatial association in many situations between poverty and land degradation, as described above (see Section 5.2.1). In some cases, poverty can lead to less sustainable land management. One reason is that poorer households are less likely to have secure land tenure, and land conversion itself – from a natural state to agriculture – can be a way for people to establish tenure through a process that has been called “clearing to claim” (Southgate, 1990) (Box 5.11). In addition, the poor are generally more dependent on common lands rather than on private property (Scherr, 2000). While land owners may have more incentive to engage in sustainable practices on land they own, it is often thought that an over-reliance of people on common lands will lead to a “tragedy of the commons” (Hardin, 1968) where individuals all maximize their own incomes from common lands but in so doing damage it irreparably. This dynamic can negatively affect the poor most of all, as it has been shown that wealthier landholders who have access to private land may nonetheless choose to use and capture benefits from common land first; once common lands have been degraded, wealthier landholders can turn to private landholdings while poor households only have access to the now-degraded common lands (Frimpong, 1986). In addition to being more likely to own property formally, wealthier landholders also have more resources available and are likely to invest in higher-input agricultural techniques, which generally lead to higher vegetative cover (Bahamondes, 2003). Poorer landholders may be less able to invest in similar activities that result in long-term productivity gains if they are accompanied by short-term losses. This is an illustration of a general pattern where the poor have been observed to more highly value present incomes relative to future incomes (i.e., they have a higher discount rate) (Holden *et al.*, 1998; Perrings, 2014).

Some research, however, has shown poor rural landholders may make conservative decisions to maintain future consumption at the expense of present consumption, suggesting that the general conclusion of the poor being less likely to make decisions that prioritize long-term sustainability may not always hold (Moseley, 2001). In some cases, resource limitation and degradation of the land base can actually lead to improved sustainability of management as communities and individuals are pushed to use intensification technologies instead of relying on extensive techniques (Boserup, 1965).

It has been observed that for certain types of land and environmental degradation, wealthy households in fact degrade more, particularly when in the case of extraction of high-value products such as hunted game and high value timber (Duraiappah, 1998; Scherr, 2000). This distinction among types of environmental incomes – those supported by the extraction of products with relatively low market value and those supported by extraction of products with high market value – is in fact at the centre of the poverty-environment relationship. Although the poor rely more on environmental as a proportion of their total income, wealthier households extract more in absolute terms, which is particularly the case for high-value products (Chhetri *et al.*, 2015). While environmental incomes generally reduce inequality (as discussed above in Section 5.2.2), more processed and higher-value harvested goods have been shown to have the highest levels of inequality among environmental incomes (Jagger, 2012).

One of the flaws in the “downward spiral” perspective described above may be that it focused too much on a subset of types of environmental degradation. Rather than thinking of poverty leading to environmental degradation in general, we should think of poverty leading to certain types of degradation, while wealth leads to others (Duraiappah, 1998). For example, poor households may be more likely to expand agriculture into marginal lands, resulting in deforestation, erosion, and declining soil fertility (Ravnborg, 2003; Scherr, 2000). However, other aspects of land and environmental degradation – notably biodiversity loss and chemical pollution – may result more frequently from the higher-input activities of wealthier land users (Ravnborg, 2003).

5.2.2.3 Community resource management and the poverty-degradation link

In some cases, relative poverty (i.e., the level of poverty as compared to others in the community or the country) may determine levels of land degradation as much as absolute poverty does. Inequality in wealth, income, and land often lead to less effective community resource management and worse overall environmental incomes (Boyce, 1994; Ostrom, 1999; Varughese & Ostrom, 2001). When inequality is higher, communities may have more trouble finding the common ground that is required for the establishment of effective resource management institutions. In areas of Nicaragua, co-operative efforts to establish land management institutions have been blocked by a coalition of a small number of extremely large land holders and a much larger number of the very poorest farmers who are trapped in a near-feudal relationship with those same wealthy landholders (Ravnborg, 2003). Inequality in land holdings in the Nicaragua case, and the patron-client relationships that have developed as a result, has made sustainable land management in the area much more difficult. In certain cases, conservation efforts themselves have exacerbated inequality within communities. Conservation payments, for example, have been shown in some cases to disproportionately benefit the wealthy because of their greater control over land and resources (de Koning *et al.*, 2011; Muradian *et al.*, 2010).

One of the strongest messages to come out of research on community resource management is simply that it can work very effectively to maintain ecosystem and livelihood sustainability (Chhatre & Agrawal, 2008; Ostrom, 1999; Varughese *et al.*, 2001; Wollenberg *et al.*, 2007). This is not to say that common property management is always successful in achieving sustainability; indeed, there are many challenges (Campbell *et al.*, 2001). Institutional structures and cultural factors are central to determining whether or not common property management succeeds or fails in achieving sustainability (Feeny *et al.*, 1990). However, what experience makes clear is that neither the tragedy of the commons nor the downward spiral from poverty to land and environmental degradation is a forgone conclusion. Rather, institutions, policies, markets, and social structures at local levels and at larger scales play a central role in determining the relationship between poverty and land and environmental degradation (Barbier, 2010; Dasgupta *et al.*, 2005; Scherr, 2000).

5.2.3 Land restoration and poverty reduction

Land restoration has the potential to successfully reduce many of the negative impacts of land degradation on rural poverty. Restoration can stabilize ecosystem functions, diversify livelihoods, raise incomes, and reduce gender disparities. Project design needs to prioritize among biophysical and socio-economic concerns; to do so effectively requires an understanding of the socio-economic context where the project is situated as much as an understanding of its biophysical context.

5.2.3.1 Protected areas and poverty

Ecosystem conservation and ecosystem restoration are twin processes. Certainly, the most cost-effective way to ensure the maintenance of ecological function in a landscape is to avoid degrading the landscape in the first place; however, restoration can be an important tool to improve ecological function on a landscape post-degradation (Mansourian & Vallauri, 2014). Conservation efforts such as protected areas (e.g., national parks, protected forests, marine protected areas, etc.) have had mixed effects on poverty (Gibson & Marks, 1995; Cernea & Schmidt-Soltau, 2006). In some cases, conservation has been accomplished by excluding local people from the natural resources upon which they depend for their livelihoods, with the predictable effect of worsening poverty (Cernea & Schmidt-Soltau, 2006). However, protected areas can have positive effects on livelihoods by creating employment opportunities, improving local infrastructure, and sustaining the resources that people obtain from natural landscapes (Brockington & Wilkie, 2015).

Research looking at the effect of protected areas in Costa Rica and Thailand has shown that communities near protected areas have generally lower rates of poverty than communities that are not (Andam *et al.*, 2010). Further research, again in Costa Rica, has suggested that this positive effect is generally due to the increased income and employment from the tourism that results from the presence of protected areas (Ferraro & Hanauer, 2014). Two other mechanisms examined for the impact of protected areas – changes in infrastructure and changes in ecosystem services provision and access – had no discernible effect on poverty rates.

5.2.3.1 Implications of restoration and rehabilitation for poverty

Restoration and rehabilitation of ecosystems post-degradation can improve livelihoods and reduce poverty. Projects to restore or rehabilitate degraded ecosystems have been shown to improve employment opportunities, agricultural income, environmental incomes, and other aspects of well-being such as health, equity, livelihood resilience, empowerment, and livelihood diversification (Adams *et al.*, 2016 and references therein). There can be run-on benefits of restoration projects to other sectors. For example, the re-establishment of nitrogen-fixing native Acacia species in four West African countries has resulted in increases in grain yields of up to 100 kg per hectare in neighbouring agricultural fields (Reij, 2009; Reij & Garrity, 2016). Additionally, restoration of mangrove ecosystems has been shown to have a large benefit for the fishing sector and for associated livelihood benefits; large-scale (800 km^2) restoration of mangroves in the Indian state of Gujarat is estimated to contribute $570 million annually to the state's fishing sector (Das, 2017).

Restoration has the potential to mitigate gender disparities, for example by improving access to fuelwood, which in turn tends to have the largest positive benefit on women and the poor, meaning that it may lead to an increase in gender equity (Sendzimir *et al.*, 2011) and in general economic equity (Liyama *et al.*, 2014). Sendzimir and colleagues (2011) found that foraging time for fuelwood for women was reduced from 3 hours per day to 30 minutes per day. Reductions in the amount of time foraging for wood has been shown to increase the amount that women spend on their children's education, time spent on their own education, and an increase in the amount of childcare (Reij, 2009; Sendzimir *et al.*, 2011; Wang & Maclaren, 2012; Weston *et al.*, 2015).

Cash income and employment have often risen in rural areas as a result of efforts to restore forest land. One of the most important determinants of rising incomes was an increase in livelihood diversification; indeed, diversification is one of the most frequently-reported benefits of restoration projects (Adams *et al.*, 2016). Although a review of forest restoration literature found that the majority of studies resulted in rising incomes, there were a few cases where incomes actually declined post-restoration (Adams *et al.*, 2016). In some cases, these declines resulted from program designs that did not sufficiently account for the opportunity cost to farmers of lost agricultural production on land where restoration was increasingly prioritized, or else it resulted from the fact that compensation for restoration came slowly relative to when the initial cost was imposed (Wang *et al.*, 2012; Xu *et al.*, 2007). In some cases, this was a problem with program design. However, in many cases, economic losses to certain households from restoration projects resulted from the fact that different households have different opportunity costs, livelihood portfolios, and labour availability, and even a well-designed project may result in both winners and losers (Liang *et al.*, 2012; Tschakert *et al.*, 2007). This speaks to the need for projects to carefully evaluate heterogeneity among households and to not assume a one-size-fits-all model will affect all community members in the same way.

5.2.3.3 Prioritizing restoration for livelihoods

Different approaches to restoration will have different implications for ecosystem function and for livelihoods. A comprehensive review of forest restoration projects found examples of projects that prioritized each of the following: hydrologic function, coastal protection, erosion protection, carbon sequestration, species diversity, landscape diversity, and livelihoods (Stanturf *et al.*, 2014). Although most restoration projects will have a range of benefits, the extent of each of these benefits is dependent on the design of the restoration program itself and the ecosystem functions that are prioritized (Bullock *et al.*, 2011; Lamb *et al.*, 2005; Stanturf *et al.*, 2014). Although win-win solutions are the most popular to promote, it is important to recognize that a restoration effort that seeks to maximize biodiversity will not necessarily be the effort that is most effective at reducing poverty (Lamb *et al.*, 2005). The design process for a restoration project should make explicit the trade-offs among different potential ecosystem service at a given site and the livelihood priorities of the people who will be affected by the project (Stanturf *et al.*, 2014). Prioritizing restoration by biophysical characteristics only is unlikely to maximize potential livelihood benefits and poverty reduction, and may also be less effective ecologically. A prioritization that also includes social and economic considerations (e.g., agroforestry dependence, local agronomic preferences, institutional structures) will likely lead to a more effective result (Budiharta *et al.*, 2016). One example of a framework that seeks to balance biophysical and socio-economic priorities is the Satoyama Initiative in Japan that builds on traditional land-use practices to achieve modern landscape-scale goals (Takeuchi, 2010).

The ecological and economic success of restoration efforts is greatly dependent on the effectiveness and fairness of socio-political institutions (both formal and informal). In a survey of 46 research studies on the livelihood impacts of restoration projects, 60% of studies identified governance structures as being key to socio-economic outcomes (Adams *et al.*, 2016). Several institutional factors are important for ensuring positive outcomes from restoration, in particular, clear access and use rights to land, an effective identification of local livelihood needs, and the early engagement of local stakeholders (Budiharta *et al.*, 2016; Mansourian *et al.*, 2014; Widianingsih *et al.*, 2016). As an example of identifying local livelihood needs, it has been observed that reforestation programs that incorporate fallow systems of shifting cultivators (i.e., a “land sharing” system as described in Chapter 3, Section 3.3.2.2) generally have better livelihood outcomes for local people and may be more ecologically effective; in addition, they draw on the traditional and local knowledge of shifting cultivators (Chazdon & Uriarte, 2016; Mukul *et al.*, 2016).

5.2.3.4 Costs of restoration versus benefits to livelihoods

The cost of restoration varies widely depending on the technique used and the type of ecosystem in question (Chazdon *et al.*, 2016). The most cost-effective form of land restoration is, of course, avoidance of land degradation in the first place. Land degradation itself often imposes an economic and livelihood cost that is greater than the cost of management efforts that would deter it. National-scale studies in Malawi and Tanzania have found that over a 30-year time-period, the cost of inaction against land degradation is, respectively, 4.3 times and 3.8 times higher than the cost of action (Kirui, 2016). A survey that combined data from 42 African countries found that efforts to reduce soil erosion had the potential to result in net benefits of more than $62 billion annually as they would mitigate losses of $127 billion in grain annually to erosion and degradation in those same countries. An average of surveys in Central Asian countries finds that the cost of inaction in the face of land degradation is roughly five times higher than the cost of action (Mirzabaev *et al.*, 2016).

The cost-effectiveness of activities to restore an ecosystem post-degradation often depends on how passive or active the form of restoration is. A study of four dryland sites in Latin America – in Mexico (2), Chile, and Argentina – found that passive restoration options were the most cost effective when compared to benefits to livelihoods (Birch *et al.*, 2010; Bullock *et al.*, 2011). Simply reducing or eliminating livestock grazing in the area – the passive option – was cost effective when the opportunity cost of lost grazing was compared against the benefits to four ecosystem services: timber harvest, non-timber forest product harvest, tourism, and carbon sequestration. Passive restoration with additional protection added – fences and fire protection – often ceased to be cost effective, although in some cases it was cost neutral. Active restoration, where planting of native plants was added to the passive restoration and the protection, was not cost effective under any scenario at the four sites. These results, however, are site-specific. A study in South Africa that similarly assessed the cost-effectiveness of restoration in a dryland grazing area, found that active restoration on that site could, in fact, be cost effective when assessed against the same set of ecosystem services

(Blignaut *et al.*, 2010; Bullock *et al.*, 2011). In some cases, partial rehabilitation of a degraded site may be cost effective even when complete restoration of the ecosystem to its original state may be cost-prohibitive or simply not possible under a given timeframe (e.g., Qadir *et al.*, 2014). In general, the most cost-effective restoration programs are often those that rely on assisted natural regeneration rather than on planting (Reij *et al.*, 2016). Assisted natural regeneration projects, because of their lower costs, also have much greater potential to be adopted beyond a project's borders.

Active restoration based on planting is expensive in most contexts, and some more than others. For example, in the highly diverse Karoo landscape in South Africa, restoration is extremely expensive, and is generally not cost-effective using an exclusively economic cost-benefit analysis. Even when compared against a scenario where the grazing potential of the landscape had to be replaced entirely with expensive purchased fodder, it remained cheaper to purchase the fodder than to actively restore the ecosystem (Bourne *et al.*, 2017). In this situation, arguments for the benefit of restoration may be entirely valid from an ecological perspective, but are unlikely to be successful when framed in terms of livelihoods and economic returns.

As is the case with land degradation in general, the relationships among restoration efforts, livelihoods, and poverty is complex. The effectiveness of any restoration effort will be affected by biophysical and cost considerations, while at the same time, its livelihood impacts will be mediated by livelihood portfolios of households affected, local resource use rights and institutions, and socio-political structures. Restoration projects attempt to balance many priorities at once. While trade-offs are inevitable, a careful analysis of socio-economic context along with an analysis of the biophysical context will ensure that the best possible livelihood outcomes are achieved.

5.3 Food security

Food security is a multi-dimensional phenomenon and is a critical factor in achieving human well-being and quality of life. Historically, it has had several research definitions and policy usages (FAO, 2003). For this report, we adopt the definition drafted by FAO (2002) in the State of Food Insecurity: "food security [is] a situation that exists when all people, at all times, have physical, social and economic access to sufficient, safe and nutritious food that meets their dietary needs and food preferences for an active and healthy life."

5.3.1 Status and trends in food security

The UN Sustainable Development Goal 2 "Zero Hunger" is aiming at ending hunger in all its forms by 2030, achieving food security and improved nutrition and promoting sustainable agriculture (UN, 2016). Globally the proportion of undernourished people has declined from 15% to 11%; however, 795 million people still lack regular access to adequate food. There are enormous geographic differences in undernourishment where the highest prevalence of hunger existing in countries of Sub-Saharan Africa and Southern Asia (Figure 5.2) (UN, 2016; Wheeler & Braun, 2013). 232.5 Million people in Africa and 511.7 million in Asia still suffer from hunger, but also 14.7 million people in the USA and Europe, 34.3 million in Latin America and Caribbean, and 1.4 million in Oceania (FAO, 2015; UN, 2016). 158.6 Million children under the age of five have suffered from stunted growth (i.e., inadequate height for age) and chronic undernutrition in 2014, which puts children at greater risk of dying from infections and increases frequency and severity of infections as well as reduced cognitive abilities and school and work performance (UN, 2016) (see also Section 5.4). While globally the proportion of undernourished children has decreased in almost all world regions since 2000, especially in

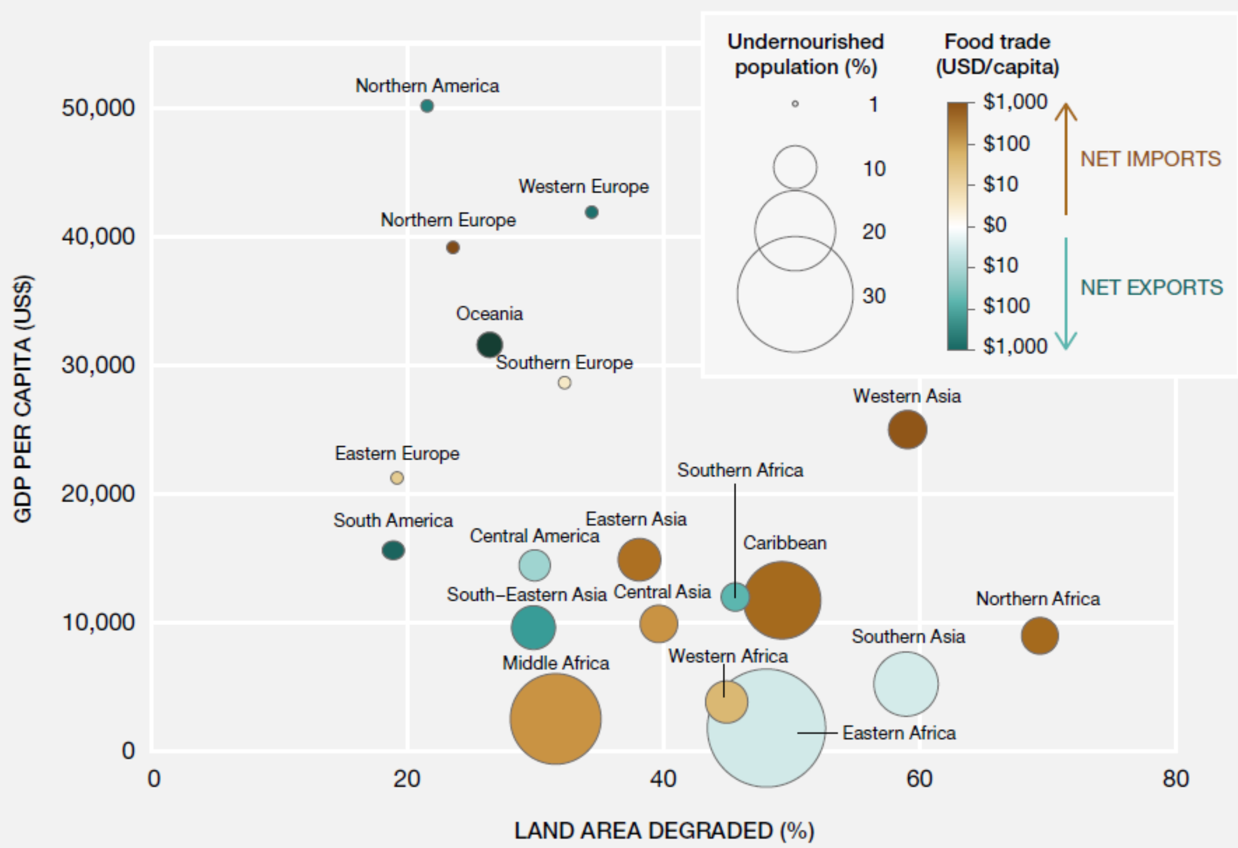
Southern Asia, the total number of stunted children in sub-Saharan Africa has increased as population growth outpaced the progress made in food availability (UN, 2016).

According to FAO (2003), while national and global assessments of food security status and trends are important development indicators, there are significant gaps in analyses of food insecurity at sub-national levels. This “meso-scale” gap is reflected in national-scale averages that do not fully reflect distributional patterns within countries, and is most apparent for larger countries such as Brazil, India, Nigeria or the Russian Federation. In the case of land degradation, processes cross national borders and patterns are often most apartment at regional scales. Regional trends might be more permanent (e.g., exhaustion of fertile soils in Sub-Saharan Africa) or temporary, due to natural disasters and climatic events. Thus, the relationship between land degradation and food security and poverty may not be fully evident at a national level. However, this gap is beginning to be bridged by the development of new tools such as the Famine Early Warning System Network (Brown, 2008). Critically, inter-linkages between agricultural liberalization, trade, land degradation and food security need to be recognized in terms of their implications for protecting the rights of diverse human populations and cultural minorities; conserving biodiversity and maintaining ecosystem services. Rural people, children and women have been disproportionally affected by land degradation and its direct impacts on human well-being, including important food security and health consequences (Rogge, 2000; MEA, 2005; ADB and FAO, 2013; FAO, 2012).

Social scientists and social movements have criticized the adoption of the concept of food security dislocated from political issues of food production and control by nation states and the private sector. Patel (2009: 665), argued that "critically, the definition of food security avoided discussing the social control of the food system. As far as the terms of food security go, it is entirely possible for people to be food secure in prison or under a dictatorship." Via Campesina, a global social movement working on defence of local food systems and human rights to food, has declared the concept of “food sovereignty” as a precondition for the existence of food security. According to the organization, food sovereignty is the right of each nation to develop and maintain its own systems to produce its basic foods respecting cultural contexts. According to the International Indian Treaty Council (IITC, 2017), food sovereignty for indigenous peoples is recognized as the “right of Peoples to define their own policies and strategies for sustainable production, distribution, and consumption of food, with respect for their own cultures and systems of managing natural resources and rural areas”.

Figure 5 2 Regional relationship between economic development and land degradation.

Economic development is measured using gross domestic product per capita in 2013 while land degradation is measured by the percent of land area in 2011 in class one under the Global Land Degradation Assessment (GLADIS; Nachtergaele *et al.*, 2011). Although it should not be interpreted as a simple causal relationship, the plot illustrates that the areas of highest land degradation are generally those that are less economically developed. The plot also illustrates the percent of the population that is undernourished (size of bubbles; FAO, 2013). Undernourishment is strongly associated with less-developed economies, but is not consistently associated with land degradation. The lack of a consistent relationship between land degradation and food security is partly the result of international trade in food; net flows of traded food is indicated by the colour of each bubble (FAO, 2013), i.e., some regions with large undernourished population like East Africa has net exports while middle Africa has net import.



National and international efforts to support food security have been, in some cases, connected to private sector and political interests, which have offered biotechnology solutions to achieve food security. Some have argued that biotechnology has resulted in a loss of food sovereignty and that it may, in the long term, compromise food security, biodiversity conservation, indigenous and local knowledge systems, and ecosystem services altogether (Bawa & Anilakumar, 2013; Jia, 2010; Macnaghten & Carro-Ripalda, 2015; Scoones, 2005). In Mexico, the introduction of genetically modified crops, such as varieties of cotton and corn, has resulted in loss of indigenous crop variety diversity, and at the same time limited the access to seeds by indigenous and local farmers (Dalton, 2001; Massieu-Trigo, 2009; Turner, 2009). In Asia, the global centre of origin of rice, the introduction of transgenic rice varieties may pose a threat to local indigenous varieties and associated knowledge; in some cases, indigenous varieties may be more resistant to diseases and climatic changes due to their local adaptation and genetic diversity (Jia, 2010).

Diverse international policy instruments have been developed to support national countries in addressing land degradation, social inequality, and food security while protecting human rights. International guidelines linking land tenure with forests, fisheries, food security and human rights were established by FAO in 2012. The United Nations Convention to Combat Desertification (UNCCD) published the Advocacy Framework on Gender in 2011, recognizing the role of women in ecological restoration and food security, and the necessity to create specific programs, policies and platforms to support gender inclusion in decision-making, access to information, funding and resources, and mainstreaming gender in regional and national programs, plans and policies (UNCCD, 2011). For additional details on governance and related policies addressing these issues, please refer to Chapter 6, Section 6.4.

Linkages between food security, land degradation and indigenous peoples' well-being and rights need to be addressed in the context of national and international policies established and signed by diverse countries, and in the local or place-based contexts where these connections are operating and/or at risk from multiple drivers of land degradation. As a starting point, it is important to acknowledge that for traditional and indigenous peoples, the right to food is inseparable from rights to land, territories, resources, culture and self-determination (Damman *et al.*, 2013). While the contribution of traditional food to nutrition status can be substantial, assessments of the impact of food insecurity among indigenous peoples, family farmers and other traditional social groups can be hindered when assessment tools consider only monetary access to market foods (Turner *et al.*, 2013) (Box 5.2, Figure 5.3).

Box 5.2 Indicators of food security, food sovereignty and sustainable development according to indigenous peoples

Cunningham (2013), lists five main indicators of food security, food sovereignty and sustainable development according to indigenous peoples, which help us to understand the importance of linkages between traditional knowledge and traditional foods (Figure 5.3):

1. Access to, security for, and integrity of lands, territories, natural resources, sacred sites and ceremonial areas used for traditional food production;
2. Abundance, scarcity and/or threats regarding traditional seeds, plant foods and medicines, food animals, and the cultural practices associated with their protection and survival;
3. Use and transmission of methods, knowledge, language, ceremonies, dances, prayers, oral histories, stories and songs related to traditional foods and subsistence practices, and the continued use of traditional foods in daily diets;
4. Indigenous Peoples' capacity for adaptability, resilience and/or restoration regarding traditional food use and production in response to changing conditions;
5. Indigenous Peoples' ability to exercise and implement their rights to promote their food sovereignty.

Figure 5 3 Indicators of food sovereignty among indigenous peoples and local communities (IPLCs), which may be considered in policies and programs to address land degradation and promote land restoration.
Source: Adapted from Cunningham (2013).

5.3.1.1 Causes of food insecurity and hunger

Hunger and food insecurity is not simply a matter of food availability, but rather of natural and human-induced disasters, socio-economic inequality, or political instability (Alexandratos, 1999; UN, 2016). In addition, large areas of cropland are allocated to animal feed and biofuel production, which could potentially feed another 4 billion people (Cassidy *et al.*, 2013; West *et al.*, 2014) (also see Section 5.7.2.2). While global agricultural areas have the capacity to produce sufficient food to feed the world population, many developing countries still suffer food insecurity and lack of food self-sufficiency (Alexandratos, 1999; Cassidy *et al.*, 2013; Erb *et al.*, 2016; Mauser *et al.*, 2015). The lack of food self-sufficiency in many developing regions, for instance in sub-Saharan Africa countries, is often attributed to an underutilized and underfinanced agricultural sector (e.g., limited infrastructure, technology, and external inputs) unable to close "yield gaps" and to keep pace with the growing population and changing consumption patterns of urban populations (Jayne *et al.*, 2010; Sayer & Cassman, 2013).

Recent trends in governmental expenditures for agriculture (e.g., agricultural share of GDP and agricultural orientation index or AOI) have shown that governmental investments in the agricultural sector have decreased between 2001 and 2013, primarily in developing countries (Nelson *et al.*, 2010; UN, 2016). However, low resource use efficiency and consumption patterns, primarily in high-income countries (Erb *et al.*, 2016; Fader *et al.*, 2013; Foley *et al.*, 2011; Garnett *et al.*, 2013) as well as food waste (especially in production, handling and storage in low-income countries) contributes to insufficient food availability and distribution (Affognon *et al.*, 2015). However, land degradation, in its different forms, strongly impacts food availability and distribution and constitutes a major driver of food insecurity and hunger in many world regions (Bindraban *et al.*, 2012; Bossio *et al.*, 2010; Foley *et al.*, 2011; Godfray *et al.*, 2010; Nkonya *et al.*, 2016a; Ortiz *et al.*, 2013; Tscharntke *et al.*, 2012; Vira *et al.*, 2015) (see Section 5.3.2).

An important driver that we discuss in Section 5.3.2.5 is the globalization of the food systems which has led to a shift away from subsistence agriculture for local consumption and towards the production of commodity crops for export. Le *et al.* (2016) recently estimated that land degradation hotspots cover 29% of the global land area with impacts on 3.2 billion people living in those degraded areas. 40% of the agricultural land is degraded to the point that crop yields have been strongly reduced and another 9% cannot be reclaimed for crop production anymore (Bossio *et al.*, 2010) (for a further discussion of the state of global degradation as measured by a range of metrics, please refer to Chapter 4, Section 4.2). Almost all biomes in both developed and developing regions are affected, although the severity of impacts varies substantially between world regions with most severe impacts on the livelihood of the poor (Le *et al.*, 2016; Nkonya *et al.*, 2016a; Nkonya *et al.*, 2016c). On the contrary, land restoration efforts, such as environmentally friendly, sustainable production practices show large potentials of mitigating the negative impacts of land degradation on food production and can help to contribute to future food security in many regions (Bommarco *et al.*, 2013; Bossio *et al.*, 2010; Foley *et al.*, 2011; Nkonya *et al.*, 2016c; Power, 2010; Pretty *et al.*, 2006; Stavi *et al.*, 2015; Tilman *et al.*, 2011) (also see Section 5.3.3).

5.3.2 Impacts of land degradation on food security

5.3.2.1 Land-use and land-cover change

Land-use and land-cover change is one the most important processes triggering land degradation, and accounts for the largest share of its global costs (Nkonya *et al.*, 2016c). Deforestation and clearance of native vegetation, habitat destruction, and unsustainable management practices on cropland and pastures, especially agricultural intensification, are among the most important drivers of degradation with strong implications on food security (Foley *et al.*, 2011; Mirzabaev *et al.*, 2016; Nkonya *et al.*, 2016a); other factors include urbanization, infrastructure development, resource extraction, land abandonment, and others (see also Chapter 3 for detailed discussion on drivers). In Sub-Saharan Africa, which has experienced the most severe land degradation worldwide, deforestation and the conversion of natural grassland to cropland have been identified as important forms of land-use and land-cover change and degradation (Nkonya *et al.*, 2016b). Forests contribute to food security and human health in multiple ways: broadly from the provision of ecosystem services, and more specifically, through provision of a diversity of healthy foods and products (e.g., food high in micronutrients and fibre and low in sodium, refined sugar and fat) which are often culturally valued, integral to local food systems and food sovereignty (Arnold *et al.*, 2011; Ferraro & Hanauer, 2011; Vinceti *et al.*, 2013). Forests help households fill seasonal and other cyclical food gaps, acting as buffers or safety stocks in times of shortages due to climate or market related changes and impacts, such as drought, crop failure, illness or other kinds of emergency or external shock (Arnold *et al.*, 2011). Deforestation and land-use intensification have also contributed substantially to climatic change and degradation of soil and water resources, which is one of the main causes of low yields and stagnating crop production in many regions, and thus, constitutes a major driving factor for food insecurity and hunger (Bindraban *et al.*, 2012; Bossio *et al.*, 2010; Tscharntke *et al.*, 2012).

Infrastructure development and land-use intensification derived from extractive industries are important drivers of land, water and soil degradation, especially in biodiversity-rich tropical countries, with implications for human well-being and food security (Killeen, 2007; Nobre *et al.*, 2016; Finer *et al.*, 2012) (see also Chapter 4, Section 4.3.10). In many Latin American countries, indigenous lands and protected areas are becoming islands of biodiversity surrounded by multiple forms and drivers of land-use and land-cover change. In the Amazon, for instance, in addition to mechanized agriculture and cattle ranching, infrastructure development (roads, ports, highways, hydroelectric dams) and extractive industries (mining plants, palm oil plantations, petroleum extraction) have been identified as a major threat to biodiversity conservation and protection of traditional livelihoods among indigenous and other local social groups such as riverine populations, rubber tappers, African descendent cultural groups (such as “quilombolas” in Brazil), and others (Barber *et al.*, 2014; Finer *et al.*, 2013, 2015; Killeen, 2007; Nobre *et al.*, 2016; Oldekop *et al.*, 2016).

Roadways, while opening up avenues for people to sell forest goods and agricultural products, can lead to rising rates of deforestation, unsustainable off-take of high value forest goods and decreased reliance on forest goods by locals (Arnold *et al.*, 2011). The Brazilian Amazon and the Congo Basin provide examples of the trade-offs existing between road construction, access to markets, deforestation and food security (Megevand & Mosnier, 2013; Soares-Filho *et al.*, 2004). In Congo, improved infrastructure through road building and paving has led to increased pressure on forests and agricultural production, while presumably has improved food security (Megevand *et al.*, 2013). Nevertheless, it is important to monitor the long-term sustainability of these trends, since agricultural intensification with lack of social capital and technical support can lead to land degradation and migration, re-configuring frontier regions. These are typical of developing countries in the tropics, where local development follows a boom-and-bust pattern of economic growth followed by a collapse phase resulting from exhaustion or over-exploitation of natural resources such as timber or productive land (Rodrigues *et al.*, 2010).

The construction of hydroelectric dams in many Amazonian tributaries presents an example of cumulative impacts of infrastructure development and deforestation on land and water degradation, which has affected local livelihoods, well-being and food security among indigenous peoples and riverine communities (Almeida, 2014; Athayde, 2014; Doria *et al.*, 2017; Fearnside, 2016; ISA, 2015). The case of the Enawene-Nawe indigenous people of the Brazilian Amazon, which illustrates the diverse facets and impacts of land and water degradation on ecosystem services, territorial management, and food security understood from ritual and subsistence perspectives (Almeida, 2014).

5.3.2.2 Deforestation and clearance of native vegetation

Deforestation and clearance of natural vegetation has increased the availability of food worldwide as the area of cropland and pasture land has substantially increased since the 18th century (Foley *et al.*, 2011; Godfray *et al.*, 2010; Goldewijk & Ramankutty, 2004). This is true for most world regions where forests and other natural habitats, such as prairies, steppes, and savannahs, have been replaced by agriculture. Brazil for instance, had among the highest deforestation rates in the world until 2005, before strict law enforcement and interventions in soy and beef industries have significantly reduced forest loss (Hansen *et al.*, 2013; Nepstad *et al.*, 2006b). At the same time, it became one of the world's leading soy and beef exporters (Nepstad *et al.*, 2006b; Ortiz *et al.*, 2013). Globally, growth in population and consumption has led to an increase in food demand, resulting in scarcity of agricultural land (Rulli *et al.*, 2013; Yu *et al.*, 2013). High income countries use land abroad to increase their agricultural land (also referred to as "virtual land use", "displaced land use", or 'telecoupling' (see Section 5.7.2.2 for other examples of such indirect land-use changes), causing displacement in land and water resources needed for food production, and "transferring" the environmental impacts to the source low-income producing countries, such as in the case of soy exports to Europe by Brazil and Argentina (Boerema *et al.*, 2016; D'Odorico *et al.*, 2013).

Recent studies have highlighted the complex and non-linear interlinkages between deforestation, climate change, biodiversity loss and agricultural decline in the Amazonian region, and their implications to global climate stability and agricultural productivity, at small and large scales (Coe *et al.*, 2013; Malhi *et al.*, 2008; Nobre *et al.*, 2016). According to Lawrence and Vandecar (2014), future agricultural productivity in the tropics is at risk from a deforestation-induced increase in mean temperature and the associated heat extremes and from a decline in mean rainfall or rainfall frequency. For a more complete treatment of interlinked drivers of degradation, see Chapter 4, Section 4.2.

Especially for intensive agricultural production systems, increases in food provision have been typically high (Grassini *et al.*, 2013; West *et al.*, 2010) and formed the basis for health, well-being and livelihood security for a large proportion of the world population. However, it has also been shown that recent agricultural expansion through deforestation has contributed little to food security and that most yield improvements were achieved through intensification rather than expansion (see below; Foley *et al.*, 2011). Moreover, it has been shown that tropical regions that have been primarily affected by agricultural expansion during the last decades (Hansen *et al.*, 2013), typically reach only half of the crop yields of the agricultural land in temperate regions (West *et al.*, 2010). Many people, especially rural communities in the tropics and subtropics, do not have sufficient access to food (Foley *et al.*, 2011; Stocking, 2003). Although the Amazon region in Brazil is a net exporter of food, a large proportion of the rural population still suffers from food insecurity (Ortiz *et al.*, 2013). The benefits and costs of these activities are not equally distributed among the population, and have further implications for human well-being locally and globally. It was estimated that in 2004 about one third of the population in the Amazon basin were medium to seriously food insecure (Ortiz *et al.*, 2013; UNEP, 2009).

As forests and other natural ecosystems also provide food in the form of wild plants and animals, deforestation and clearance of native vegetation has led to reductions in food availability, for example, for rural, traditional and indigenous populations around the world, whose livelihoods are closely tied to these resources (Arnold *et al.*, 2011; Huambachano, 2015; Kuhnlein, 2007; Woodley *et al.*, 2006). It is estimated that approximately 1.2 to 1.5 billion people, including about 60 million indigenous people, are dependent on forests (Vira *et al.*, 2015). Terrestrial and aquatic wildlife are important protein and nutrient sources for many people throughout developing countries and play an important role for human health (see Section 5.4) (Golden *et al.*, 2011; Myers *et al.*, 2013). For many of those communities, the relationship linking deforestation and land clearing to increased food provision generally does not hold. Different studies have demonstrated that deforestation, habitat loss, and losing access to wildlife food sources have negative impacts on food availability and nutrition of many local populations who live distant from urban centres (Ickowitz *et al.*, 2014; Kleinschmit *et al.*, 2015). For more details on changes in non-timber forest resources please see Chapter 4, Section 4.3.5.

In addition, many of the communities do not only lose their basis for health and food security, but also other aspects of quality of life, such as identity, autonomy and diversity and options. Traditional knowledge and lifestyles are lost with land change through loss of access to important sites necessary for food-related rituals and cultural practices, and the replacement of their traditional food resources and associated knowledge (e.g., Almeida, 2014; Arnold *et al.*, 2011; Athayde, 2014; Dounias & Froment, 2011; Finer *et al.*, 2008; Fisher, 2013; Laird *et al.*, 2011; Reyes-García *et al.*, 2005).

Although globally the proportion of people that completely depend on food from forests and other natural ecosystems is modest, wildlife food sources play an essential role for income generation and diverse and healthy diets for many people outside forest areas, especially in developing regions (Jamnadass *et al.*, 2015; Parrotta *et al.*, 2013) (Box 5.3; also refer to Section 5.2.2.1 on the importance of environmental incomes). Natural vegetation, forests, and tree-based systems play a key role in agricultural production and provide an important nutritional source during periods of food shortages (Jamnadass *et al.*, 2015). Their loss exacerbates a large problem, not only through the loss of fruits, vegetables, bushmeat, medicinal plants, and other tree-based products, but also through the decline of ecosystem services that are essential for neighbouring crop and livestock systems (e.g., animal feed and green manure) (Jamnadass *et al.*, 2015; Parrotta *et al.*, 2013; Vira *et al.*, 2015).

For many low input subsistence farming systems in poor rural areas, the loss of forest and native vegetation can have adverse impacts on the production of food as they often depend on many of those services from the surrounding landscape, such as pollination, biological pest control, and water provisioning (Foley *et al.*, 2011; Ortiz *et al.*, 2013; Tscharntke *et al.*, 2012; Vira *et al.*, 2015). Crop pollination, for instance, can be a critical factor for the production of nutrients and calories and accounts for roughly one third of the global calories consumed (Klein *et al.*, 2007; Myers, Gaf, *et al.*, 2013). Losing access and availability of non-food forest products, such as firewood for cooking and heating, can have additional negative nutritional consequences, such as shifts in diets to less nutritional food or consumption of raw products (Powell *et al.*, 2013; Vira *et al.*, 2015).

Arnold *et al.* (2011), highlight the fact that decline in the use of forest food can also occur due to decline in knowledge about its use. Among indigenous and local communities, as children spend more time in school, rather than in the fields and the bush, opportunities to learn about wild foods may be reduced, especially if school curricula do not include place-based indigenous or local knowledge about local food sources and associated ecosystems. A move to a more settled lifestyle is a widespread change that can separate people from knowledge about traditional natural resources and food sources (Alexiades, 2009; Dounias *et al.*, 2011; Nabhan & Antoine., 1993). Poorer knowledge constrains people’s use of these foods leading to dietary simplification and negative repercussions on human health, even when the latter are still available and important for dietary balance (Arnold *et al.*, 2011).

Box 5.3 Implications of deforestation for food security and nutrition among indigenous peoples and local communities (IPLCs)

Different studies have demonstrated that deforestation, habitat loss, and losing access to wildlife food sources have negative impacts on food availability and nutrition of many local populations who live distant from urban centres (Ickowitz *et al.*, 2014; Kleinschmit *et al.*, 2015).

In the Amazon basin, for example, which is among the world's largest deforestation hotspots, loss of forest and native vegetation has been a major driver of food insecurity for many indigenous communities (Ortiz *et al.*, 2013). Smallholder farmers in Amazonia depend to large extents on services provided by natural vegetation to secure their food, health, and livelihoods (Ortiz *et al.*, 2013).

In another study in Madagascar, Golden *et al.* (2011) have shown that the loss of access to wildlife would increase the number of children suffering from anaemia by 29%. Fisher (2013), for example, reports for Australia that many traditional food sources of Aboriginal communities have substantially narrowed since the 19^{th} century due to a combination of different factors including land clearing, habitat degradation, changing lifestyles, and the loss of traditional knowledge. For the Martu community in Western Australia, for instance, habitat loss, invasive alien species, and changing fire practices, have contributed to a decline of 75% of the plant species and 70% of the animal species that have formed their traditional food system (Fisher, 2013; Walsh, 2008).

Diets and diseases are sensitive indicators of the ecological and cultural costs that former hunter-gatherers currently pay to achieve their share of modernity. Examples from the Boka and Kola pygmies of Cameroon (West Africa) and the Tubu Punam of Borneo chronicle the impact which drastic alterations of forest ecosystems have had on forest-reliant hunter gatherers, affecting their diets, bringing new diseases, and spread of intergenerational mal-nutrition (Dounias *et al.*, 2011).

As many communities may not be readily able to substitute wildlife with domesticated food sources, its loss may represent a nutritional crisis for many of those people (Myers *et al.*, 2013). In addition, communities do not only lose their basis for health and food security, but also other aspects of quality of life, such as identity, autonomy and diversity and options, as traditional knowledge and lifestyles are lost with land change through loss of access to important sites necessary for food-related rituals and cultural practices, and the replacement of their traditional food resources and associated knowledge (e.g., Almeida, 2014; Arnold *et al.*, 2011; Athayde, 2014; Dounias *et al.*, 2011; Finer *et al.*, 2008; Fisher, 2013; Laird *et al.*, 2011; Reyes-García *et al.*, 2005).

5.3.2.3 Land use and management intensification

Management intensification of cropland and pastures have strongly increased food availability primarily due to technological assets first produced during the "Green Revolution" (e.g., industrial fertilizers, irrigation technology, and the use of pesticides) since the middle of the 20^{th} century (FAO, 2011; Foley *et al.*, 2005; Tilman *et al.*, 2001). Agricultural modernization has increased per capita food supply since the 1950s with increasing quantities of food rich in calories, protein, and fat (Khoury *et al.*, 2014). Production has increased between 2.5 to 3 times over the last 50 years, while the increase in agricultural land was only 12% (FAO, 2011). Vitousek *et al.* (2009) and Ju *et al.* (2009) report that increasing fertilizer application along with other technological changes has strongly contributed to yield improvements, for instance significantly increasing grain yields in China since 1975 and high yield increases in the USA since the 1940s. However, intensive agricultural production and excess nutrient application has had clear negative environmental impacts, such as degradation of soils as well as water quantity and quality (see below; FAO, 2011; Foley *et al.*, 2011; Pretty *et al.*, 2011; Tilman *et al.*, 2001; West *et al.*, 2014). Additionally, while increased food provision resulting from technological agricultural intensification generally benefits the poor, this is not always the case, as some social and institutional structures may concentrate benefits of intensification within wealthy groups to the exclusion of the poor (Binswanger & von Braun, 1991; Béné & Obirih-Opareh, 2009; Pingali, 2012). In addition to cropland intensification, the livestock sector also experienced significant intensification, most notably through increasing grazing intensities in both developed and developing region, although management practices are considerably different (e.g., industrial livestock production in developed and traditional grazing systems in many developing regions) (Delgado *et al.*, 1999). Although livestock production provides an important source of protein and nutrients, and contributes additionally to large amounts to the income of rural smallholders in developing countries, it has been shown that increasing livestock densities contribute to additional land clearing and degradation of soil and water resources, and moreover, require large areas of land for animal feed (Cassidy *et al.*, 2013; Delgado *et al.*, 1999; Nepstad *et al.*, 2006b) (see also Chapter 3, Section 3.3.1).

Although industrialized agricultural intensification has led to global increases in total food provision, many people still suffer insecure food supply and inadequate diets (Foley *et al.*, 2011; Tilman & Clark, 2014). Populations that live in regions where land degradation can be severe and where access to productive land or technological assets is missing or limited face dramatic losses in health, well-being, livelihood, and security (Stocking, 2003; Tscharntke *et al.*, 2012). Poor populations in tropical regions, such as in many sub Saharan African countries, are affected particularly strongly by the negative consequences of land use, primarily because land per person ratios and agricultural suitability are generally much lower than in high-income countries (FAO, 2011). Most of the existing yield gaps are due to nutrient and water limitation (Mueller *et al.*, 2012; Vitousek *et al.*, 2009) and improvements in agricultural production in sub-Saharan Africa were primarily due to cropland expansion rather than improving yields (Dawson *et al.*, 2016; Pretty *et al.*, 2011). Compared to its success in Asia, the Green Revolution did not succeed to the same degree in sub-Saharan Africa because it represented a radical change of the traditional agricultural practices (Dawson *et al.*, 2016). The poorest can often also not make the transition from traditional nature-based to technological production systems, primarily due to low income and limited access to agricultural inputs, infrastructure, and markets (Alexandratos, 1999; FAO, 2011; Myers *et al.*, 2013). Innovation costs for smallholders are generally high and crop production remains small and lags behind population growth (Dawson *et al.*, 2016).

Stagnation and decreases in food provision have already been observed even in high input agricultural systems in temperate regions (e.g., North America and Europe) (Grassini *et al.*, 2013; Ray *et al.*, 2012). It has been shown that regions with intensive and high efficient production systems may, in some cases, even experience greater losses through degradation, as seen for instance in a five times higher decline in milk production due to losses of grazing biomass in North America compared to sub-Saharan Africa (Kwon *et al.*, 2016). However, farmers can compensate these losses by high external inputs, often masking the negative impacts, while production losses in many developing countries show much more severe impacts, as livestock plays a much larger role for food, nutrition, and income for many people living below the poverty line (Kwon *et al.*, 2016). In those regions, where the negative impacts of land degradation can often not (or only insufficiently) be compensated by technological assets, it has been shown that declining soil and water quality has substantially contributed to a reduction in food provision (see below; FAO, 2011; Godfray *et al.*, 2010; Ray *et al.*, 2012; Stocking, 2003; Tscharntke *et al.*, 2012). However, it should be noted that the extent and severity of the impact of land use and management intensification on food provision can vary substantially between and within regions and depends not only on the type and intensity of the production systems, but also on biophysical conditions, such as climate and soil quality (Godfray *et al.*, 2010; Stocking, 2003).

Another important aspect of agricultural intensification is the decline of global crop diversity (Khoury *et al.*, 2014). Khoury *et al.* (2014) have reported that, among other factors, modernization and international trade have contributed to a global increase in the homogeneity of food composition and a decline in the variability of consumed crop products. This implies a decline in the importance of a diversity of local food crops (Khoury *et al.*, 2014), and is accompanied with a gradual replacement of food that is culturally important for local communities with "western" food products as well as the loss of indigenous and local knowledge about the characteristics and uses of native species as food and medicinal resources (e.g., Fisher, 2013; Parrotta *et al.*, 2013).

In addition, the increasing agricultural commercialization and associated intensification, for instance the establishment of large-scale monoculture soy and cattle production systems in Brazil, have also led to dispossession of smallholder family-based farmers in the Amazon and the loss of many traditional tree-based production systems (e.g., savannah woodlands and agroforestry) which have played an important role in food provision and livelihoods of many rural communities (Kleinschmit *et al.*, 2015; Ortiz *et al.*, 2013; Parrotta *et al.*, 2013). Li (2011) argues that large-scale industrial agriculture plantations often cause the dispossession of local peoples' livelihoods, and, contrary to claims that these initiatives contribute to poverty reduction, this is generally not the case. In fact, land grabbing for industrial development may worsen poverty at the local and regional scale, and may not provide enough job opportunities to justify the amount of land used, in many cases degrading or contaminating this land with pesticides.

5.3.2.4 Soil and water degradation

Soils are a fundamental resource for essential contributions of nature and for quality of life, most notably through providing the basis for food production and water regulation (Bouma & McBratney, 2013; FAO, 2015; FAO & ITPS, 2015; McBratney *et al.*, 2014) (see also Chapter 4).

Natural ecosystems and healthy soils contribute substantially to other contributions of nature relevant for food and livelihood security (Daily *et al.*, 1997; Robinson *et al.*, 2013), most importantly to the provision of freshwater by regulating the hydrologic cycle and by removing sediments, nutrients, and other pollutants from surface water and groundwater (Brauman *et al.*, 2007) (see also Chapter 4, Section 4.2). Deforestation and agricultural intensification alters the hydrologic cycle in many watersheds and, thus, the quantity and quality of freshwater (Brauman *et al.*, 2007; Compton *et al.*, 2011; Vitousek et al, 2009). Intensive production in upstream areas, for instance, may result in locally increasing yields, but can have substantial impacts on food production in downstream areas, when access to water becomes limited or when irrigation facilities are unusable due to sedimentation from upstream soil erosion (FAO, 2011; Swallow *et al.*, 2009). Land under irrigation, for instance, has more than doubled since the 1960s and the use of groundwater for irrigation is expanding rapidly, leading to increasing competition for water and water scarcity in many regions (FAO, 2011;

West *et al.*, 2014). The degradation of water resources, such as declining aquifer levels and groundwater pollution, constitutes a major risk for food production systems in arid and semi-arid regions that highly depend on irrigation (Bindraban *et al.*, 2012; Bossio *et al.*, 2010; FAO, 2011). In addition, unsustainable production practices, excess fertilization, and associated water pollution (e.g., nitrogen leaching into surface and groundwater) are causing health problems and generate substantial societal costs, as reported for example for Europe and North America (Compton *et al.*, 2011; van Grinsven *et al.*, 2013; Vitousek *et al.*, 2009).

In regions, where soil and water degradation have caused severe or irreversible losses of nature's contributions, for instance through severe water scarcity and pollution, flooding events, and productivity losses (Pimentel, 2006), consequences for food and water security, health, and livelihoods can be dramatic. Highly degraded soils may not respond to fertilizer inputs anymore and may require substantial management measures (e.g., by adding organic matter) to recover productivity, if possible at all (Young, 1994; FAO & ITPS, 2015). Impacts are most severe for poor populations living in many tropical regions that have already limited access to productive land and clean water, and low possibilities to compensate yield losses through technological assets (Young, 1994; FAO, 2011; Stocking, 2003; Tscharntke *et al.*, 2012). Some of these groups (e.g., indigenous people and smallholder subsistence farmers) may also experience losses in other quality of life aspects, such as diversity and options as well as identity and autonomy, for instance, when the degree of degradation no longer allows for sufficient production to keep their traditional lifestyles. Soil and water degradation and low crop productivity has also contributed land abandonment and to out-migration of rural communities (Young, 1994; Gray, 2011) (see Section 5.6.2). However, it has been shown that many smallholder farmers in the tropics, for instance in Africa and Asia, manage soil and water resources sustainably and productively through sustainable cultivation practices that can increase soil organic matter, nutrients, and soil biota (Godfray *et al.*, 2010; Stocking, 2003). By utilizing traditional local knowledge, they can achieve higher yields and sustain other contributions of nature, with positive impacts on their well-being and livelihoods (Godfray *et al.*, 2010; Stocking, 2003; Thierfelder & Wall, 2012) (see Section 5.3.3.2).

5.3.2.5 Globalization, production conflicts, and socio-economic inequality

Economic growth and globalization has interconnected production regions worldwide to the global market and has led to agricultural specialization of many regions and to strong disconnections of production and consumption areas (Fader *et al.*, 2013; Kissinger & Rees, 2010; Lambin & Geist, 2006; Yu *et al.*, 2013) (also see Chapter 3, Section 3.6.4). Although international trade of agricultural products has helped to increase food availability for many food-insecure countries and has large potentials to overcome food shortages (Fader *et al.*, 2013), it has been shown that primarily high-income countries, such as the European Union (EU) and Japan, require large amounts of land outside their territories, and thus, displace immense pressures on ecosystems, land, and water resources to other regions, especially low income-countries (Mekonnen & Hoekstra, 2012; O'Bannon *et al.*, 2014; Weinzettel *et al.*, 2013; Yu *et al.*, 2013). Increasing interconnectedness together with lifestyle and consumption patterns of high-income countries (e.g., diets rich in meat and dairy products and consumption of non-agricultural products) drive land degradation in many world regions, often unseen by local consumers (Cassidy *et al.*, 2013; Kastner *et al.*, 2012; Kissinger *et al.*, 2011, 2010; Yu *et al.*, 2013) (see also Chapter 7).

Deforestation, intensification, and unsustainable production practices can cause substantial losses of many regulating services, such as erosion control and nutrient and pollutant retention (see Section 5.3.2.1), resulting in soil and water quality degradation in many exporting countries (Kissinger *et al.*, 2010; Mekonnen *et al.*, 2012; O'Bannon *et al.*, 2014). This may, in the long run, negatively affect crop productivity and food prices. Developing countries may experience more severe impacts on ecosystem services due to globalization as they have less access to international markets and abilities to externalize their production compared to wealthier nations (O'Bannon *et al.*, 2014; Seekell *et al.*, 2011; Weinzettel *et al.*, 2013). On the other hand, market liberalization and subsidized agricultural commodities have led to increased imports in many developing countries, for instance in Sub-Saharan Africa, where domestic production primarily of smallholder farmers is often displaced by cheap imported products (Jayne *et al.*, 2010; Prášková, 2013). The increasing land demand, especially of wealthier nations, place additional pressure on local food and livelihood security as it may often be accompanied by intensification or direct and indirect forms of "land grabbing" (Foley *et al.*, 2011; Godfray *et al.*, 2010; Ortiz *et al.*, 2013; Tscharntke *et al.*, 2012).

Land grabbing (i.e., large-scale acquisition of land especially in developing countries), driven primarily by concerns about food and energy security of high-income countries and often executed by the private sector (Anseeuw *et al.*, 2011), have shown, in many cases, negative impacts on the livelihood of the rural poor, especially smallholder farmers, who lose access to land and water resources due to insecure land rights, unequal and non-transparent contract negotiations, and poor governance and legislation (Anseeuw *et al.*, 2011; Cotula *et al.*, 2009; FAO, 2011; Marselis *et al.*, 2017; Ortiz *et al.*, 2013). As smallholder farmers are the most important food source in developing countries (see Section 5.3.3.2), the increasing pressure and competition for land and other resources, production conflicts (e.g., cash crops and bioenergy crops), as well as an increasing vulnerability of poor families to global food price changes will constitute major threats for food security in many developing regions, especially in sub-Saharan Africa countries (Cotula *et al.*, 2009; Fian, 2010; Foley *et al.*, 2011; Prášková, 2013; Weinzettel *et al.*, 2013) (Box 5.4). On the contrary, it has been shown that the dietary changes of the wealthy nations have increasingly negative impacts on health and well-being for the populations in developed regions (Khoury *et al.*, 2014; Tilman *et al.*, 2014).

Box 5.4 International trade, land degradation, and food security in sub-Saharan Africa

Changes in international trade and food markets have strong impacts on the status of food security in many sub-Saharan Africa countries. Kenya, for instance, is largest economy in the East African Community (EAC) and has successfully established markets for cash crops such as coffee, tea, tropical fruits, vegetables, and cut flowers for export, mainly to the USA, the EU, Pakistan, and Egypt (Dietz *et al.*, 2014; Pannhausen & Untied, 2010; Prášková, 2013). On the other hand, it shows higher proportions of malnutrition and poverty rates than some of its neighbours and is highly dependent on imports of food crops from regional and global markets (e.g., corn, wheat, rice from Tanzania, India, Pakistan, and Russia) (Pannhausen *et al.*, 2010; Prášková, 2013). Increases in agricultural production since the 1960s remained relatively small and could not keep pace with population growth resulting in strongly declining per capita food availability and dramatically increasing undernourishment, food shortages, and poverty among the rural population, primarily in the Lake Victoria Basin (Daniel, 2011; Dawson *et al.*, 2016; Dietz *et al.*, 2014; Jayne *et al.*, 2010; Mulinge *et al.*, 2016; Swallow *et al.*, 2009). The lack of food self-sufficiency of Kenya is often attributed to an underutilized agricultural sector (Daniel, 2011; Nolte & Väth, 2015; Sayer *et al.*, 2013), but market liberalization associated with increasing, often subsidized, imports have also contributed to displacements of local producers and forced many smallholder farmers out of business (Jayne *et al.*, 2010; Prášková, 2013). Although cash crop cultivation plays an essential role for farm income of many smallholders (WRI, 2007), the transformation of food to cash crops in the course of land acquisition projects constitutes a major problem for food availability of poor families (Cotula *et al.*, 2009; FIAN, 2010). Kenya has become an important target for such foreign land acquisitions (i.e., land grabbing) that replaces many small and medium sized farms by large-scale monoculture plantations for export crops, and thus, makes more and more of the rural population highly dependent on outside food sources and vulnerable to changes in the global food price, as seen in the 2007/2008 food price crises (Daniel, 2011; FIAN, 2010; Nolte *et al.*, 2015; Prášková, 2013). Increasing landlessness, inequality in land distribution, and decreasing land per person ratios exert higher pressure on the remaining limited land resources suitable for cultivation resulting in soil degradation and increasing probabilities of crop failures that pushes many smallholder farmers into poverty (i.e., "environment-poverty traps") (FAO, 2011; Jayne *et al.*, 2003, 2010; Mulinge *et al.*, 2016; Swallow *et al.*, 2009). Without interventions, such as reforming international trade environments and the efficient transformation of the agricultural sector that addresses especially the role of smallholder farming and secures their land rights and access to infrastructure and markets, this development may constitute a major threat for future food security and poverty alleviation, not only in Kenya, but also other sub-Saharan Africa countries (Dawson *et al.*, 2016; FIAN, 2010; Jayne *et al.*, 2010; Prášková, 2013).

5.3.2.6 Climate change, land degradation and food security

Climate change and natural disasters will exacerbate food insecurity and inequality of access to quality food (Nelson *et al.*, 2010; Wheeler *et al.*, 2013). Increasing temperatures, heat waves, droughts and the probabilities of extreme events in the course of climate change will have negative effects on crop yields and food security in most world regions without adequate adaptation strategies (Battisti & Naylor, 2009; FAO, 2011; Lobell *et al.*, 2008; Power, 2010; Wheeler *et al.*, 2013) (see also Chapter 3, Section 3.4). Although some studies suggest increasing yields due to elevated atmospheric CO_2 concentrations for some crops (Högy & Fangmeier, 2008; Prior *et al.*, 2008), these effects may be by far outweighed by the impacts of weather extremes (Nelson *et al.*, 2010; Wheeler *et al.*, 2013). Moreover, it has been reported that atmospheric CO_2 enrichment can result in lower grain quality including lower nutrient and protein contents of the major food crops (Högy *et al.*, 2008; Loladze, 2002; Taub *et al.*, 2008).

Poorer and vulnerable cultural groups in tropical regions will be primarily affected by food insecurity as a consequence of climate change due to increasing temperatures and droughts by the end of the 21st century (Battisti *et al.*, 2009; Lobell *et al.*, 2008; Wheeler *et al.*, 2013). It is projected that regions which are already food insecure such as Africa and South Asia will experience yield losses between 5% and 17% by 2050 based on the analysis of a wide-range of climate models and emission scenarios (Ahmed & Suphachalasai, 2014; Knox *et al.*, 2012; Lobell *et al.*, 2008). Schlenker and Lobell (2010) also report high yield losses between 8% to 22% and increasing probabilities of crop damages for sub Saharan Africa under the IPCC A1b scenario for mid-century (2046-2065), and Roudier *et al.* (2011) find a mean yield loss of 11% for countries in West Africa due to climate change across a wide-range of IPCC climate scenarios. On the contrary, it has been shown that for temperate and cooler regions such as Northern Europe and North America, increasing temperatures can result in higher crop yields (FAO, 2011; Olesen *et al.*, 2011; Power, 2010; Reidsma *et al.*, 2010; Schlenker & Roberts, 2009). However, those regions will also be vulnerable and may experience yield decreases due to extreme events if no climate adaptation strategies for the agricultural sector are implemented (Schlenker *et al.*, 2009). For Europe, for instance, most climate change projections predict longer dry periods, heat waves, and increasing probabilities for heavy precipitation events (Beniston *et al.*, 2007; Huang *et al.*, 2015), which will also cause mostly negative impacts on agricultural production such as higher yield variabilities and probabilities of crop failures (Olesen *et al.*, 2011; Reidsma *et al.*, 2010). Fuss *et al.* (2015) have estimated that increasing global yield variability in the course of climate change will require strong increases in cropland area in order to meet future food demand, which can result in higher crop prices and additional land degradation due to land-use and land-cover change which can create an additional risk for food security in many regions.

Climate change as a driver of land degradation often interacts, at local and regional scales, with biophysical processes, exacerbating existing productivity challenges and existing risks to local productive systems and food security. Prolonged droughts in certain types of drier ecosystems and forests may turn them more susceptible to wildfires, which may cause further land degradation and vulnerability to subsequent droughts and fires in a vicious cycle (Soares-Filho *et al.*, 2012).

In the Amazon, interactions between deforestation and climate drive the frequency and magnitude of wildfires (Brando *et al.*, 2014), with implications for indigenous and local communities' livelihoods and food security. In the Xingu watershed in Brazil, the transition region comprising between the Amazonian forests to the north and drier savannahs to the south has become extremely vulnerable to fires due to compounding effects of climate change, deforestation, soy and cattle production (Schwartzman *et al.*, 2013). Indigenous peoples who historically have inhabited the headwaters of the Xingu river recognize changes in wildfire and rainfall regimes, which has critically worsened in the last decade (Schwartzman *et al.*, 2013). Combined with climate change, wildfires and droughts, community sedentarization and shifting cultivation have put local food production systems at risk by soil overuse and degradation, associated with uncontrolled fire events (Athayde & Silva-Lugo, 2018). Certain local and indigenous crop varieties, such as peanuts, do not grow well in degraded soils, and may be threatened of disappearing in a continuing climate change scenario (Silva, 2002). These processes have critical implications not only for food security and conservation of agrobiodiversity, but also for the protection of biocultural diversity and ecosystem services provided by indigenous lands and protected areas in the Amazonian, both locally and globally (Nepstad *et al.*, 2006a; Walker *et al.*, 2014).

5.3.2.7 Indigenous and local food systems, associated knowledge and cultural practices

Globally, place-based food has strong cultural significance, as well as health and environmental implications. Shifts in food production from locally-oriented crops, non-timber forest products, or hunted meat towards commercial crops may have negative impact on several cultural groups in both industrialized and non-industrialized countries, even if overall levels of food availability do not change (Barthel & Isendahl, 2013; Cunningham, 2013; Ibarra *et al.*, 2011; Vinceti *et al.*, 2013). The deterioration of natural ecosystems and consequent loss of biodiversity and associated knowledge present a threat to the social-ecological resilience not only to indigenous peoples, family farmers, and local traditional communities, but to humanity's well-being at large (Huambachano, 2015; Turner *et al.*, 2013). One important contribution of local food systems is the conservation of crop genetic diversity, also known as agrobiodiversity. The maintenance of local knowledge and cultural traditions regarding food is a matter not only of cultural identity and transmission, but also of maintaining food security and nutritional health (Turner *et al.*, 2013). The importance of locally based agrobiodiversity and wild food plants for food security, food sovereignty, re-establishing people connections with nature, and in shaping alternative models of consumption must be emphasized (Turner *et al.*, 2011).

Across European countries, many small-land holders and farmers still keep traditional knowledge of wild food gathering used for food and medicinal purposes, despite the crescent encroachment of their lands, as well as cultural and environmental change resulting from globalization, climate change, and soil and water contamination (Łuczaj *et al.*, 2012). The use of wild plants in Europe may be associated with times of famine or food scarcity, as well as for diet diversification, and religious traditions (Łuczaj *et al.*, 2012; Pieroni, 2001). Food substitution is the most common individual subsistence strategy in times of want and starvation. In the north of Portugal, the addition of different aromatic wild species, such as *Foeniculum vulgare* Mill., *Pterospartum tridentatum* (L.) Willk., *Calamintha nepeta* (L.) Savi, *Lavandula stoechas* L. or *Thymus mastichina* (L.) L., for seasoning soups and purees has helped to diversify a monotonous diet (Pardo-de-Santayana *et al.*, 2007).

A global assessment of indigenous peoples and food systems for health conducted by the Centre for Indigenous Peoples' Nutrition and Environment (CINE, 2017) revealed that land and environmental degradation is a major aspect of indigenous peoples' declining use of their indigenous food. Interconnected concerns across different regions include biodiversity loss of wild species and of cultivated species and varieties; hydroelectric dams and their impacts on fish and other foods; contamination of water and food from a host of chemical, radioactive and biological pollutants; and climate change, with its accompanying uncertainties and instabilities, as the main drivers leading to the insecurity of food systems (Turner *et al.*, 2013).

Case studies from the Centre for Indigenous Peoples' Nutrition and Environment initiative include both constraints to indigenous peoples' food security and sovereignty, as well as examples of local innovation and hybridization of local diverse food systems with new practices and co-management of forested ecosystems and cultivated fields. In Papua New Guinea, the degradation of soil and vegetation has led to an overdependence on sweet potato on the high-altitude plateau and the dry grasslands, with women and children being more vulnerable to reduced dietary diversity (Bayliss-Smith, 1991). In Western Amazon, indigenous peoples such as the Sacha Runa (Ecuador), the Ingano (Colombia) and the Awajún (Peru) actively cultivate biodiversity, and utilize both wild forests and cultivated fields for sustaining resilience in their ecosystem, to support their food security, medicinal care and cultural heritage (Correal *et al.*, 2009; Turner *et al.*, 2013). In the Brazilian Amazon, the Kaiabi indigenous people, through a community-based project in partnership with a local indigenous association and the Instituto Socioambiental (ISA), a Brazilian NGO, were able to recover the diversity of peanut varieties, threatened with disappearing by soil degradation and overexploitation (Figure 5.4). Leadership by a respected shaman (Tuiat Kaiabi), collective action, women's stewardship and ritual practices were important factors for the success of the project (Silva, 2002).

In Japan, the construction of the Nibutani dam has meant radical changes in Ainu people's livelihoods and culture, through restrictions on traditional hunting, fishing and farming (Iwasaki-Goodman *et al.*, 2009). According to Turner *et al.* (2013), among the Ainu of Japan, strong assimilation policies by the Japanese government have stopped food insecurity (according to the usual definition of inadequate access to enough food) from being a problem, but have also resulted in loss of cultural identity and knowledge of traditional foods and dishes. A recent effort aims to re-identify traditional Ainu foods and culture before the relevant knowledge is lost.

In the Brazilian Amazon, the construction of small and large hydroelectric dams has caused ecosystem degradation and habitat disruption for fish and associated aquatic fauna, impacting indigenous livelihoods and cultural traditions in different ways. Among the Enauenê Nauê indigenous people of Mato Grosso state, the construction of a small hydropower plant caused loss of ritualistic fishing practices, which confer health, spiritual stability and well-being to the group. During the *Ykaowa* ritual, abundant fish is offered to the spirits of the rivers, to appease their temper and promote peaceful co-existence with Enawenê-Nawê communities. After the construction of the dam, fish became very scarce in the region, causing socio-ecological disruption. The community has bought commercial fish in the cities to promote the ritual, when possible (Almeida, 2014). The ritual and associated knowledge and practices were recently declared an immaterial national patrimony by the Brazilian cultural and archaeological institute IPHAN. A video of the ritual as traditionally performed can be found online (IPHAN, 2009).

Figure 5 4 Restoration of peanut varieties by the Kaiabi indigenous people through community-based projects in Xingu Indigenous Park, Brazilian Amazon. Photo: courtesy of Geraldo Silva.

5.3.3 Land restoration for food security and biodiversity conservation

5.3.3.1 The role of indigenous and local knowledge systems

Despite the widespread recognition of the importance of forest restoration for social-ecological resilience and sustainability, not all parties have the same restoration objectives. Dudley *et al.* (2005b) highlight that there has been a mismatch between social and ecological goals for forest restoration: either it prioritised social or economic needs while ignoring its wider ecological impacts, or it has had a narrow conservation focus without taking into consideration people’s needs.

While there is much to be learned about reconciling nature and human needs, and about planning restoration areas within larger scales, there is solid evidence supporting the claim that indigenous peoples and local communities around the world hold context-specific knowledge and practices that form the pillars for initiatives aimed at forest and ecosystem restoration, towards improving ecosystem services, food security and sovereignty, and good quality of life (Altieri, 2002, 2004; Berkes *et al.*, 1994; Brondízio, 2008; Denevan, 1995; Parrotta *et al.*, 2016; Sillitoe, 1998) .

For centuries, indigenous peoples and traditional farmers have developed diverse and locally adapted agroforestry systems, and managing these systems through practices that often result in community food security, biodiversity conservation and social-ecological resilience (Altieri, 2004; Parrotta *et al.*, 2016; Walker *et al.*, 1995) (see Chapter 4, Section 4.3.4.6 for definition of agroforestry as used here). Structural-, functional- and species diversification minimizes risk, stabilizes yields, promotes dietary diversity, and maximizes returns using low levels of technology and limited resources (Altieri, 2004).

Nair (2007) makes the case that agroforestry offers a unique set of opportunities and tools for addressing land degradation while providing food security and ecosystem services in both low-income and industrialized nations. For developing countries, the emphasis is on alleviating poverty, providing nutritional security and arresting land degradation under resource-limited conditions and lower-input situations, covering an estimated 1.9 billion hectares of land and 800 million people. The author explains that in industrialized nations, the principal role of agroforestry is to provide ecosystem services, including water provisioning and quality control, carbon sequestration, biodiversity conservation, and good land ethics and aesthetics. According to van Noordwijk (2014), opportunities for ecological intensification utilizing trees in agricultural landscapes may vary along stages of a tree cover transition from forest alteration and deforestation followed by agroforestation, which may provide opportunities for food security and access to markets by local residents.

In the tropics, agroforestry has been a key land restoration strategy for more than 20 years. Areas managed by indigenous and local farmers cover approximately 10 million ha worldwide, providing cultural and ecological services not only to rural inhabitants, but to mankind at large. Such services include the preservation of indigenous and local farming knowledge, local crop and animal varieties, and native forms of sociocultural organization. Innovative agroecosystem designs have been modelled on successful indigenous and local farming systems (Altieri, 2004; Altieri *et al.*, 2011; Brondízio, 2008).

Despite the incredible potential that agroforestry systems offer to reconcile land restoration while promoting biodiversity conservation and food security, achieving these gains are not easy. Davis and Palm (2014) argue that the process takes time and effort, appropriate policies and enforcement, and enough investments to help small-scale indigenous communities and farmers. Secure land tenure and land rights are also critical, as often these restoration investments take years to pay off. Finally, establishing participatory planning and monitoring processes before, during and after such programs are developed is essential, to ensure that interventions achieve desired social-ecological goals, and make adjustments as needed (Davis *et al.*, 2014; Oba *et al.*, 2008; Reed *et al.*, 2016).

Experiences from diverse tropical and sub-tropical regions around the world testify the role of small-scale indigenous and local agriculture and agroforestry systems on land and forest restoration. Millions of hectares are covered with agroforestry systems around the world, such as 2.8 million ha of jungle rubber forest in Indonesia; 7.8 million ha of cocoa agroforests worldwide; 9.2 million ha of silvo-pastural systems in Central America; and 5.1 million ha of diverse agroforestry systems in Mali (Davis *et al.*, 2014; McIntyre *et al.*, 2009). There many agroforestry initiatives that have contributed to both food security and land/forest restoration, especially home gardens, forest plantations or enrichment, enrichment of secondary forests in shifting cultivation systems, and watershed restoration through networks for exchange of seeds and/or reforestation with native trees. Altieri and Toledo (2011) present a review of agroecological initiatives and innovations in Latin America, highlighting five geographical areas where what they call "the agroecological revolution" has taken hold, and which can be considered case studies of technological, cognitive and/or social innovation: Brazil (formal recognition and application of agroecology in education and agriculture/agroforestry), Cuba and central America (farmer-to-farmer systems and technical innovation), the Andean region (integration of traditional knowledge and practices with scientific agroecology) and Mexico (coffee-forestry systems and ecosystems services).

These initiatives have some features in common, which are relevant for the establishment and monitoring national and international policies for land and landscape restoration and biodiversity conservation (Altieri, 2004; Altieri & Toledo, 2011; Camacho *et al.*, 2016; Chirwa & Mala, 2016; Laird *et al.*, 2011; Nair, 2007; Norton *et al.*, 1998; Ouédraogo *et al.*, 2014; Parrotta *et al.*, 2016; Powell *et al.*, 2013; Rossier & Lake, 2014; Senganimalunje *et al.*, 2016; Walker *et al.*, 1995). These common features are: (i) indigenous and local agroecological knowledge has been formally recognized and incorporated in policies and programs for agroforestry development; (ii) participatory planning and monitoring processes guarantee that such initiatives attend to both social development and ecological conservation and/or restoration objectives; (iii) long-term policies have been established by different countries, providing the necessary financial, technical and political support for such programs; (iv) cross-sectorial collaboration between local communities, governmental agencies, NGOs, universities and research institutions have helped to sustain and scale-up successful place-based experiences (see Chapter 6, Section 6.4 for additional information on agroforestry governance and associated policies).

Despite the recognized importance of agroforestry for reforestation and restoration of degraded lands, Veldman *et al.* (2015) highlighted the importance to distinguish between reforestation and afforestation, the later meaning the conversion of historically non-forested land to forest or tree plantations. According to the authors, afforestation of originally grassy biomes can compromise ecosystem services and reduce biodiversity, a phenomenon referred as "the tyranny of trees". It may also have important implications for food sovereignty, by transforming local food habits and diets tied to grassy environments to other crops or forest-based foods, and thus potentially driving erosion of bio-cultural diversity and associated knowledge.

Restoration projects based on indigenous and local knowledge may include diverse management strategies, such as prescribed fire, enhancement of native species, agroforestry systems, soil enrichment and managing ecosystem patchiness and mosaics; and may span a wide range of ecosystems, such as fisheries, riverine and estuarine environments, forests, savannahs, and so on (Senos *et al.*, 2006; Turner, 2016). There are several co-management initiatives around the world articulating indigenous and local knowledge with scientific knowledge for ecosystem restoration. These initiatives may not only enhance ecosystem structure, functioning, and the overall provision of services such as clean water, but also contribute to sustaining indigenous and local people's economies and cultural practices (Senos *et al.*, 2006). Box 5.5 includes some examples of co-management and participatory experiences for ecosystem restoration developed by indigenous and local communities in partnership with governmental and non-governmental institutions, which have contributed to enhance food security and sovereignty and human well-being at local and regional scales. These arrangements, which, by design, incorporate cultural aspects into ecological restoration, have been referred to as ecocultural restoration (Higgs, 2003).

Box 5.5 Indigenous Peoples and Local Communities (IPLCs) experiences on ecosystem restoration enhancing food security and human well-being

Case studies from Finland, Mexico, United States and Canada exemplify the important role and potential that co-management and participatory management arrangements might offer to ecocultural restoration enhancing food security and human well-being.

In Finland, the Project "Skolt Sámi Survival in the Middle of Rapid Change" was implemented in the Neiden watershed through a cooperative arrangement between the Skolt Sámi and other Sámi communities, the Saa'mi Nue'tt cultural organization, and the Snowchange Cooperative, with support from the Sámi Council and the Indigenous Peoples Climate Change Assessment (IPCCA) that is being coordinated by a Peru-based non-profit organization, ANDES, and supported by the United Nations University. A community-based climate change adaptation plan was developed, identifying the salmon fishery as their greater concern, related to populational decrease, habitat degradation, and additional threats triggered by climate change phenomena. The comprehensive social-ecological management plan articulated watershed governance issues with Sámi's history, customary laws and traditional practices for salmon and riverine ecosystem management, as well as scientific data on salmon biology and ecology, and climate change. The management plan's proposed actions have been implemented, involving shared responsibilities, conflict resolution mechanisms, and identification of risks (Mustonen & Feodoroff, 2013).

In Mexico, the multiple-use strategy of tropical forest management practiced by indigenous and local peoples, may be understood as actively used and restored landscape units, which enhance biodiversity, food security and sovereignty, while providing additional ecosystem services and human well-being (Toledo *et al.*, 2003) (Figure 5.5). The authors argue that this multiple-use strategy of landscape management and restoration represents an endogenous reaction of some indigenous communities to the intensification of natural resource use, responding to more recent changes, and in conjunction with external agents such as NGOs and government agencies. In the Brazilian Amazon, the Kayapó indigenous peoples have historically practiced landscape restoration through creation of patches of productive forests (Posey, 1985). Long-term transplanting and selection of plants suggest an actual semi-domestication of many species, and overall management strategies also include the manipulation of animal species used as food.

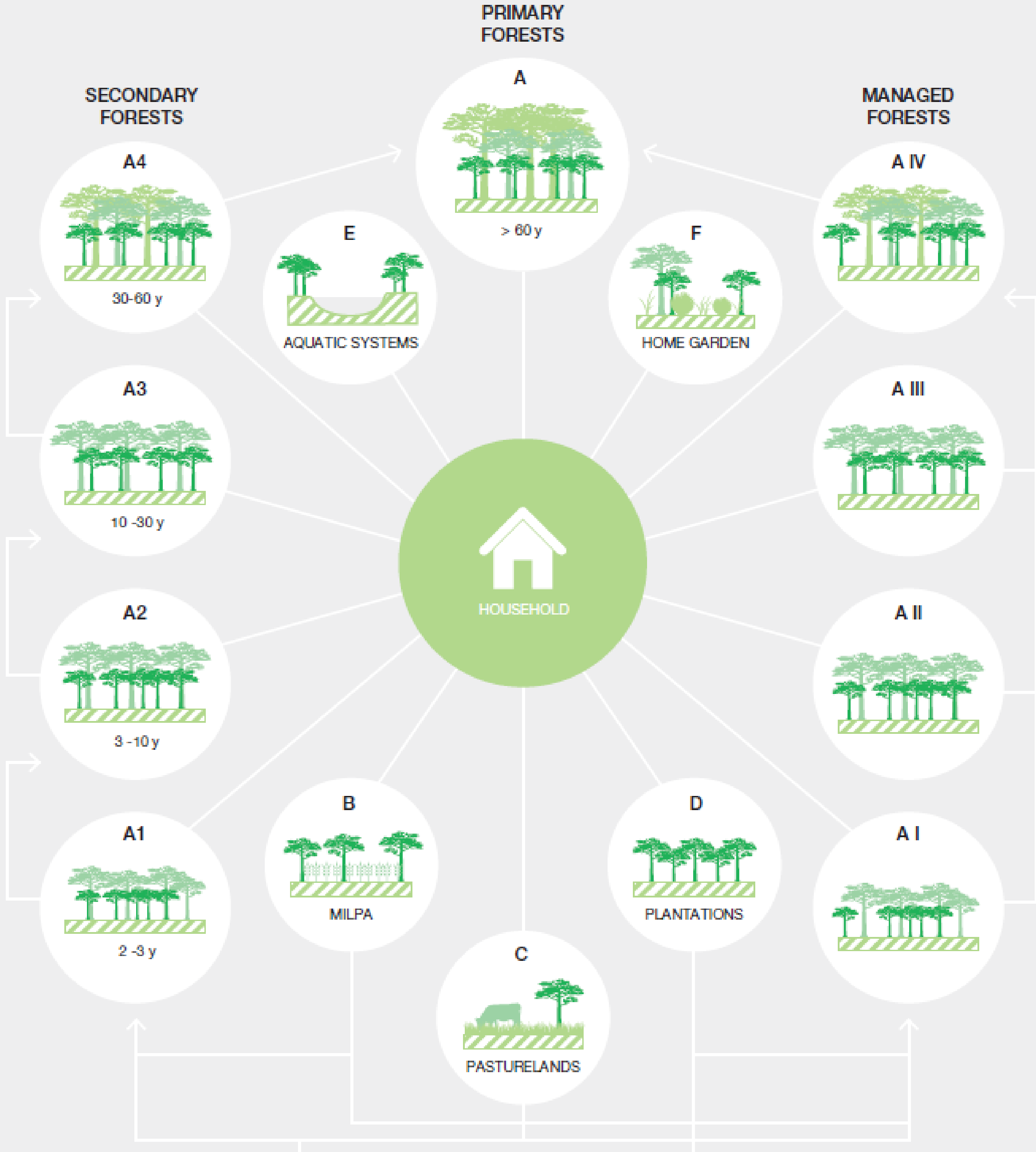
In Canada, the Songhees Nation of Vancouver Island (a Salish community) have been negatively affected by the transition from a traditional nutrition, to a more industrialized-based diet. Recently, many people have developed diabetes, while in a not so far past, this was a rare disease (Senos *et al.*, 2006). The Songhees have lost their connection with traditional food linked to ecosystem management, such as the camas bulbs, which once were a cultural keystone species, very important to the communities' socio-cultural identity. Senos *et al.* (2006) describe how the tradition of managing, preparing and consuming camas bulbs (*Camassia quamash* and C. *leichtlinii*) are being restored on patches severely degraded by sheep grazing, exotic species introduction and fire suppression, allowing the encroachment of coniferous forests on savannah prairie's ecosystems, camas' natural habitat. The camas' restoration project had the important leadership of young Songhees leaders, and marked the beginning of a series of focal restoration projects based on Coast Salish traditional knowledge.

In the United States, a cooperative project between the USA Forest Service, the Mt. Hood National Forest, the Oregon State University, and the Confederated Tribes of the Warm Springs, combined indigenous and local knowledge with Western forestry practices were applied to restore productive patches of huckleberry

(*Vaccinium* spp). Across the Pacific Northwest, other tribes and First Nations groups are incorporating traditional huckleberry management practices with forest restoration to increase the production of huckleberries, a food source important not only for human consumption, but also for wildlife (Senos *et al.*, 2006).

Figure 5 5 **Scheme of the multiple-use restorative production system developed by indigenous communities of tropical humid lowlands of Mexico, and its landscape (or land-use) units.**

The system includes the use and management of mature forests (A), secondary forests and their stages (A1 to A4), managed forests resulting from the manipulation and introduction of species in the mature forests (A-I to A-IV), milpa or corn fields (B), potreros or cattle-raising areas (C), cash-crops or agricultural fields other than milpa (D), water bodies (E), and home gardens (F). Source: Toledo (2003).



5.3.3.2 Sustainable land management and conservation agriculture

Land restoration efforts, such as environmentally friendly sustainable management practices (often referred to as “sustainable or ecological intensification”) can increase food provisioning while conserving regulating services, such as pollination, pest control, soil protection and fertility, nutrient cycling, and hydrological services (Pretty *et al.*, 2006; Power, 2010; Foley *et al.*, 2011; Tilman *et al.*, 2011; Mueller *et al.*, 2012; Bommarco *et al.*, 2013; Garnett *et al.*, 2013). Restoration of degraded land in combination with conservation practices has a large potential to halt land degradation and to sustain long-term food security and human well-being (Bossio *et al.*, 2010; Stavi *et al.*, 2015). It has been shown that many of the ecological shortcomings of industrialized agriculture can be reduced by diversified agroecological production systems with low external inputs and integration of ecosystem services (e.g., biological pest control and pollination) while increasing the availability and access to food (IPES-Food, 2016). Impacts of intensification are complex, particularly when indirect land-use effects are considered. For a fuller discussion of their ecological impacts, please see Chapter 7, Sections 7.2.1.2 and 7.3.1.

The conservation of soil resources through sustainable production practices plays an essential role for future food and water security, since soil functions underpin the provisioning of many ecosystem services that are relevant for food production (Mcbratney *et al.*, 2014). Resource-conserving agricultural practices such as conservation agriculture (e.g., minimum or zero tillage) and agroforestry reduce runoff and soil erosion (Montgomery, 2007; Palm *et al.*, 2014), water pollution, increase water holding capacities of the soil, and thus, increase water use efficiency and productivity of cultivated crops (Bossio *et al.*, 2010; Palm *et al.*, 2014) (see Chapter 6, Section 6.3.3.2). Organic farming has significantly increased during recent decades covering 37.5 million hectares in 2012 with the largest share in Australia (32%), Europe (30%), and South America (18%) (Casamiquela *et al.*, 2014; Willer *et al.*, 2014). Pretty *et al.* (2006) have shown that sustainable agricultural interventions including the above management measures have increased average crop yields by 79% for their study sites (covering 3% of farmland in developing countries). These results illustrate the potential of sustainable agriculture systems, although at present, this potential is not always achieved; a recent global-scale analysis of organic agriculture showed that typical yields from organic agriculture can be about 34% lower than those from conventional agriculture, although under certain conditions this loss is only 5% to 12% (Seufert *et al.*, 2012). Other conservation practices such as the establishment of grass strips, terraces, and soil and stone bunds, as well as the application of green manure and mulching have also shown benefits in soil protection (e.g., Pansak *et al.*, 2008) and other ecosystem services and have contributed to improved crop yields (Bayala *et al.*, 2012; Bindraban *et al.*, 2012; Kato *et al.*, 2011; Pimentel, 2006; Wickama *et al.*, 2014). Improved grassland management with controlled grazing and cultivation of legumes can also increase livestock productivity and a number of other ecosystem services (Kwon *et al.*, 2016; Nkonya *et al.*, 2016c). Farm multifunctionality, for instance through crop diversification, including food and cash crops in combination with livestock and aquaculture, can increase yields and provides diverse other benefits and income sources, and thus, can contribute to food and livelihood security of the poor and reduce their vulnerability (Bossio *et al.*, 2010; FAO, 2011; IPES-Food, 2016; Pretty *et al.*, 2006; Pretty *et al.*, 2011).

Sustainable production systems, to the extent that they enable stable or increased yields, also reduce pressure on forests and contribute to reduced deforestation rates, increasing forest cover, and climate change mitigation (Nepstad *et al.*, 2014; Paustian *et al.*, 2016; Rueda *et al.*, 2014). However, while many studies provide evidence that these sustainable management practices increase food provision, primarily for tropical regions where previous food productions have been low (Bayala *et al.*, 2012; Kato *et al.*, 2011; Palm *et al.*, 2010; Pretty *et al.*, 2006; Pretty *et al.*, 2011; Thierfelder *et al.*, 2012; Wickama *et al.*, 2014), other studies suggest that practices such as organic farming and conservation agriculture can result in lower average yields and higher land requirements, although variations are high and strongly site- and crop-specific (de Ponti *et al.*, 2012; Garnett *et al.*, 2013; Palm *et al.*, 2014; Pansak *et al.*, 2008; Seufert *et al.*, 2012; Tuomisto *et al.*, 2012; van den Putte *et al.*, 2010). Importantly, the improvements in regulating services (e.g., pollination, erosion control, nutrient retention, or water purification) may show stronger positive impacts on long-term crop productivity compared to conventional intensively cultivated production systems (Bommarco *et al.*, 2013; Lal, 2010; Power, 2010). Moreover, diverse multifunctional production systems may be more resilient and less vulnerable to crop failures and disasters compared to intensive production systems and therefore better adapted to climate change (Bommarco *et al.*, 2013; Foley *et al.*, 2011; IPES-Food, 2016; Palm *et al.*, 2010) (see Section 5.3.2.3).

However, the successful implementation of sustainable production practices that can secure future food provision requires effective governance and support by policies and local institutions (Nkonya *et al.*, 2016b; Tscharntke *et al.*, 2012). Moreover, sustainable intensification may require radical reforms of existing food production systems as well as consumption patterns (Garnett *et al.*, 2013). Especially for developing regions, sustainable food production is strongly linked to rural populations and smallholder farmers (Garnett *et al.*, 2013; Tscharntke *et al.*, 2012). Many traditional smallholder farmers in those countries perform in many cases already sustainable and resource efficient production practices compared to large-size intensive farms (Bossio *et al.*, 2010; Godfray *et al.*, 2010; Stocking, 2003; Tscharntke *et al.*, 2012) (Box 5.5). Small-scale farming accounts for the largest share of food production in developing regions; due to their importance for food provision for most of the undernourished people in the world (Bossio *et al.*, 2010), especially those farming systems should be supported through securing their income and access to land, resources, and markets (Garnett *et al.*, 2013; Hazell *et al.*, 2007; UN, 2016).

Box 5.6 Smallholder farming and food security

On the global scale, 85% of the farms are of less than two hectares in size that amount to 1.5 billion smallholder farmers, primarily in Asia and Africa, which carry out 60% of the global agriculture and provide 80% of the food in developing countries (Bossio *et al.*, 2010; Cosgrove & Rijsberman, 2000; Daniel, 2011). Numerous studies indicate that these smallholder farms tend to be more productive and show higher yields on a per unit area basis as compared to large-scale intensive cultivation systems, mostly due to their higher resource and labour use efficiency, but also because of higher crop diversity, intercropping, and combinations with livestock (e.g., Ali & Deninger, 2014; Daniel, 2011; Hazell *et al.*, 2007; Li *et al.*, 2013; Manjunatha *et al.*, 2013; Nkonya *et al.*, 2004; Verschelde *et al.*, 2013; Wiggins *et al.*, 2010).

Smallholder farming includes, but is not limited to, subsistence farming. Thus, agricultural practices under smallholder farming systems, and not large-scale intensive farming, are mostly seen as the “backbone” of food security and poverty reduction in developing countries (Hazell *et al.*, 2007; Ortiz *et al.*, 2013; Tittonell & Giller, 2013; Tscharntke *et al.*, 2012; Wiggins *et al.*, 2010). Many smallholder farmers in traditional production systems often already perform some sort of ecologically sustainable farming including manure application, crop diversification, and precision agriculture through targeted fertilization, weeding, and a variety of crop types adapted to different states of soil fertility (Daniel, 2011; Dawson *et al.*, 2016; Tittonell *et al.*, 2013). In addition, small farms tend to have higher biodiversity values and regulating services such as pollination, biological pest control, erosion control, and soil fertility as they are often associated with higher landscape heterogeneity and natural elements such as hedgerows and vegetated field margins (Belfrage *et al.*, 2015; e.g., Dawson *et al.*, 2016; Hedström *et al.*, 2006; IPBES, 2016a; Kremen *et al.*, 2004; Marini *et al.*, 2009; Rodríguez & Wiegand, 2009; Souza & Ikerd, 1996; Thies & Tscharntke, 2013; van Apeldoorn *et al.*, 2013). However, crop production and yields of smallholder farmers in developing regions, especially sub-Saharan Africa, are unable to keep pace with population growth and to achieve food self-sufficiency and food security (Egoh *et al.*, 2012; Jayne *et al.*, 2010; Nolte *et al.*, 2015; Sayer *et al.*, 2013; Tittonell *et al.*, 2013). Moreover, many smallholder farmers are suffering from increasing imports and foreign land investments that force many of them out of business (Daniel, 2011; Dawson *et al.*, 2016; Jayne *et al.*, 2010; Nolte *et al.*, 2015).

Instead of solely focusing on innovation and modernization of the agricultural sector to feed increasing urban populations, land use and development policies should recognize the importance of traditional knowledge and farming systems for food security, support existing smallholder farming practices, and especially assure their equal access to land, infrastructure, and markets (Dawson *et al.*, 2016; Egoh *et al.*, 2012; Hazell *et al.*, 2007; Tittonell *et al.*, 2013; United Nations, 2016) (see Chapter 6, Section 6.4).

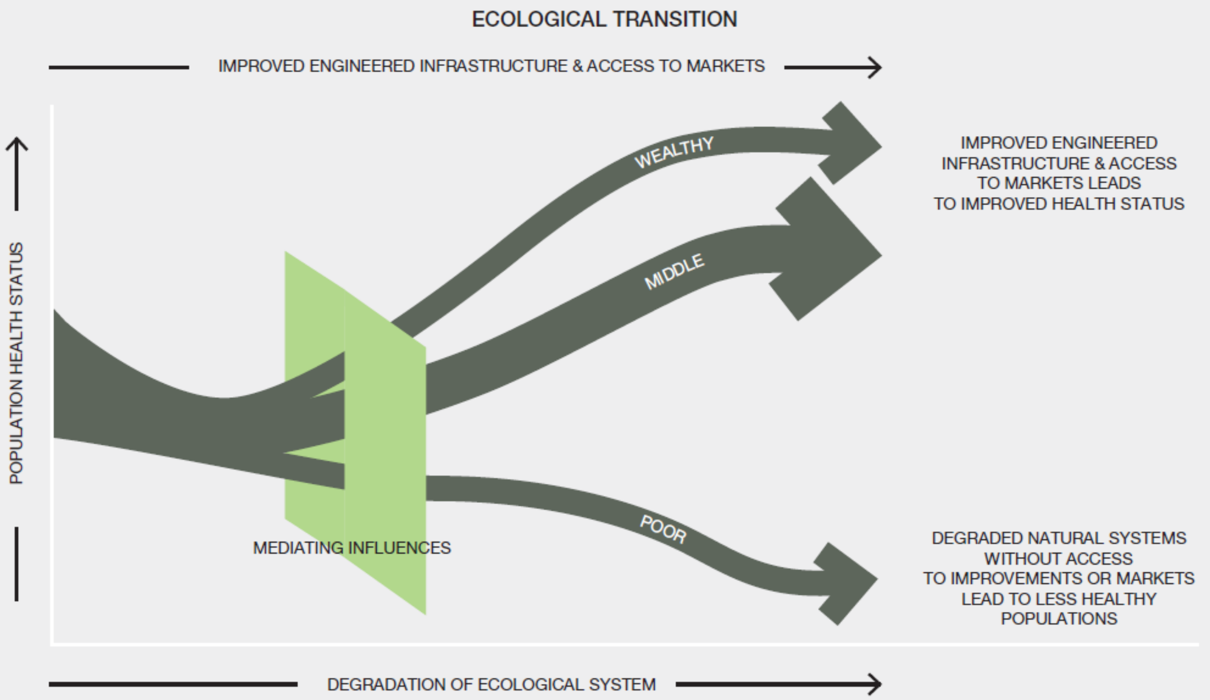
5.4 Health impacts of land degradation

Human health is one of the most readily (and objectively) measurable dimensions of quality of life, and pervasive improvements in the control and treatment of diseases, infectious or otherwise, have significantly advanced human quality of life over the past few centuries. Increases in life expectancy, decreases in child mortality, and falling prevalence of many diseases suggest a positive trend in human health globally over the past two centuries. Additionally, many infectious diseases have become better understood, leading to better management and substantial reductions in disease associated morbidity and mortality (termed “the disease burden”). Some of the most significant pathogens have been eradicated (notably smallpox and rinderpest) or are near eradication pending the successful continuation of public health initiatives (e.g., polio or Guinea worm). These advances not only directly benefit human health, but reduce the indirect cost of disease on livelihoods, and bolster food security (and thus health) by reducing the burden of disease in livestock.

Some of the most significant infectious diseases driving global mortality have been eradicated or substantially reduced as part of changing land-use patterns in developing countries. Efforts to reduce disease risk through land conversion have had substantial impacts on global disease burden, as in the reduction or eradication of malaria in many temperate zones via the in-filling of lakes and wetlands or via severe alterations like dredging or the construction of “mosquito ditches” (Hambright & Zohary, 1998; Rozsa, 1995; Willott, 2004). Access to medical care, especially antibiotics and vaccines, act in concert with these efforts, to vastly reduce the disease burden in developed nations. But in the developing world, underlying disparities due to poverty and social inequality complicate disease control, and often produce idiosyncratic interactions with land-use changes and environmental degradation (see Figure 5.6 from Myers *et al.* 2013). Diseases that are comparatively treatable or eradicated in developed countries can be particularly unmanageable in degraded ecosystems, especially where humans live in close proximity to waterways, forests, or other landscape features that increase pathogen exposure from vectors or reservoirs. Furthermore, land degradation often drives short-term declines in health by disturbing the environment and releasing pathogens, in the process of advancing infrastructure that benefits human health in the long term through economic development, food security, and greater mobility and access to healthcare. In this way, the relationship between human health and the environment can have complicated trade-offs at different scales, including through the immediate relationship of any given human with their surroundings, and in the broader feedback between environmental quality and the development and maintenance of technology, infrastructure, and other anthropogenic assets.

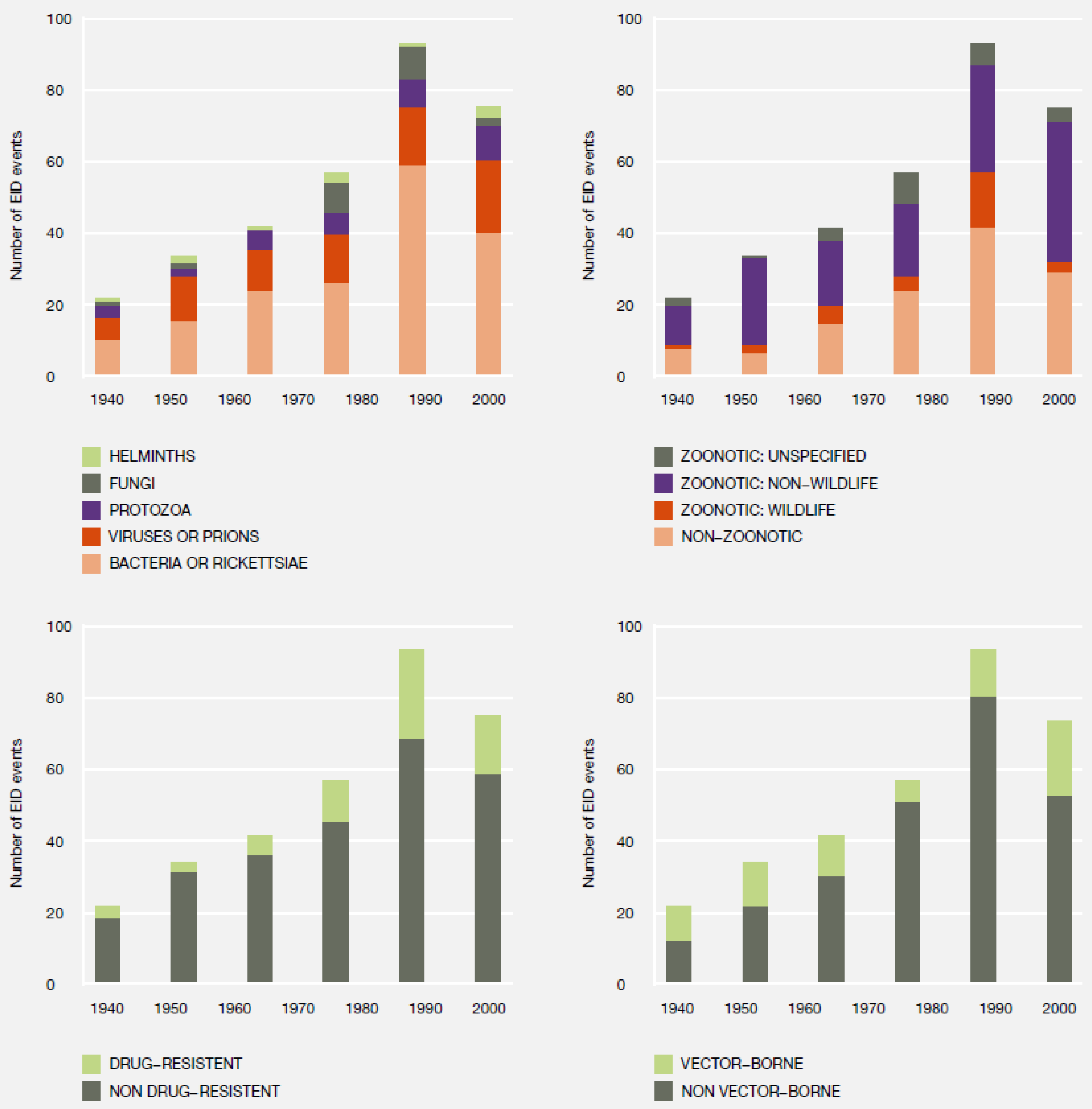
Figure 5 6 **Conceptual diagram illustrating an ecological transition from a natural ecosystem to a degraded one.**

Such a transition may be associated with increases in quality of life for those groups of people who manage to successfully benefit from the transition (possibly via better access to a market economy or increased production of certain goods). However, other groups that do not manage to benefit from the ecological transition may find themselves worse off than before when the safety net provided by natural ecosystems is degraded. Source: Myers *et al*. (2013).



One of the most difficult elements of land degradation impacts on human health is the role biodiversity loss plays in disease emergence, a process that, by definition, includes both entirely new pathogens and those with sudden increases in prevalence. The emergence of infectious diseases is an ecological process as well as a social one; the majority of emerging pathogens (roughly 75%) are zoonotic (originate in animals, termed reservoirs) and of those, the majority originate in wildlife (Jones *et al.*, 2008). While many pathogens are transmitted to humans by insect vectors like mosquitoes, others are spread from wildlife reservoirs into humans through a process called spillover, which can occur directly, or indirectly propagated by livestock or domesticated animals (Johnson *et al.*, 2015). Because of the diverse strategies that emerging pathogens can exploit, patterns of land use, agriculture, biodiversity, human-wildlife contact and human health infrastructure can interact to produce complex and often unpredictable disease dynamics (Wilcox & Colwell, 2005). On a global scale, the rate of emergence and re-emergence of infectious diseases has accelerated substantially since the industrial revolution, and continues to do so (Cohen, 2000), most likely as a consequence of global changes in climate and land use (Figure 5.7). The relationship between land-use driven changes in ecosystem diversity and disease emergence and re-emergence are complex (Daszak *et al.*, 2000). Biodiversity in undisturbed ecosystems may dilute the prevalence of disease in ecosystems in ways that ultimately benefit humans; and consequently, declines in biodiversity may increase the frequency of outbreaks in wildlife (termed epizootics) that originate human outbreaks (epidemics). However, unexpected patterns can also emerge. Higher biodiversity ecosystems can also have a greater overall richness of new pathogens that can eventually enter human populations. Biodiversity loss may therefore decrease the total richness of pathogens that humans encounter. We detail these complexities in Sections 5.4.1 and 5.4.2 below.

Figure 5 7 Change through time in pathways of disease emergence. Source: Jones *et al.* (2008).



Below, we explore that interplay deeper for three main case studies: (i) vector-borne diseases; (ii) rare episodic spillover zoonoses that originate in wildlife; and (iii) pathogens that reach human populations via livestock or agriculturally-related impacts. We further describe the relationship between land degradation and non-infectious diseases, in particular, noting that land degradation almost universally reduces water quality and exacerbates human exposure to pollutants, toxins, and pathogens. We conclude with an assessment of the potential impacts of land degradation and biodiversity loss on two key indirect components of clinical outcomes: the discovery of new pharmaceuticals in nature, and the role mental health plays in overall human health outcomes.

5.4.1 Vector-borne disease burden and land-cover change

As a general pattern, land degradation and land-use intensification increases the short-term burden of vector-borne diseases, often producing a trade-off with other dimensions of development and quality of life over longer timespans. Mosquito-borne diseases are particularly challenging in this regard, as development projects can increase human exposure to natural mosquito habitat (especially at the times of day *Anopheles* mosquitoes are most active) and produce more suitable habitat like forest edges and associated microclimates (de Castro *et al.*, 2006). Conversion of forests into agricultural or mining land especially facilitates accumulation of standing water that exacerbates *Anopheles* and *Aedes* mosquito-borne diseases (Patz *et al.*, 2004; Silbergeld *et al.*, 2002). Land-use changes associated with that conversion, like road building, are strongly linked in South America to workers presenting with "frontier malaria" and leishmaniasis, and in Africa to trypanosomiasis (Myers & Patz, 2009; Patz *et al.*, 2004). However, the effects of deforestation on malaria especially are regionally variable (and likely better understood than for any other vector-borne disease), and highly dependent on local vector ecology; for example, it is likely that malaria is more strongly associated with deforestation in Africa and South America than in Asia, due to a greater richness of *Anopheles* species especially in southeast Asia, only some of which are ecologically specialized in such a way that they benefit from deforestation (Myers *et al.*, 2009).

Deforestation is not the only land-use change with substantial, direct links to vector-borne disease. Development projects like dam building and irrigation, which produce substantial gross benefits through employment, and energy and food security, usually produce hydrological impacts that consistently exacerbate local risk for several pathogens, especially malaria, schistosomiasis (vectored by snails), and onchocerciasis (vectored by black flies) (Morse, 2001; Patz *et al.*, 2004). In cases like these, the downstream benefits of these projects often reach different populations than the local communities that face near-immediate increases in overall health burdens. Further development of rural land into urban or peri-urban environments may decrease direct human contact with nature and can increase access to medical care for environmentally-mediated diseases for some people; but pre-existing health disparities, such as poor diet or access to healthcare, can severely exacerbate morbidity and mortality from urban outbreaks (Redman & Jones, 2005). Urbanization, however, also increases the risk of other vector-borne pathogens like dengue fever where water collects and *Aedes* mosquitoes thrive (Gubler, 2011). Other vector-borne diseases like plague or leptospirosis, which utilize rats as amplification hosts, can pose a severe risk in urban settings (Costa *et al.*, 2014).

For vector-borne diseases, land use can indirectly affect disease burden through the diversion of vector bites that would infect humans to livestock or wildlife hosts – a documented phenomenon termed zooprophylaxis that can substantially improve health outcomes. Some evidence has suggested that cattle ownership can act as a sort of passive prophylaxis that decreases the burden of diseases like malaria, but case studies suggest that this phenomenon is inconsistent (Tirados *et al.*, 2011), and that greater numbers of available hosts can actually increase malaria transmission risk (Bouma & Rowland, 1995). Consequently, agricultural conversion may offer a limited buffer for human health. For some pathogens, such as Japanese encephalitis or Rift Valley fever, humans living in close proximity to livestock populations actually likely increases outbreaks (Jones *et al.*, 2013).

The relationship between biodiversity and zooprophylaxis is poorly understood, but current theory indicates that land degradation-driven loss of biodiversity could substantially increase disease prevalence in wildlife and humans. This is termed the “biodiversity dilution effect”, in which species richness of (usually mammal or bird) host communities corresponds to a decrease in the disease risk of pathogens. Dilution effects have been suggested as a potential factor in the outbreaks of a number of different pathogens, including Hanta virus, Lyme disease, West Nile virus and possibly Chagas disease. Whereas zooprophylaxis has little relationship to diversity, the dilution effect is conditional on high species richness and on a community structure in which additional hosts are less competent than common ones. The most competent hosts are often assumed to be the most generalist and resilient to ecological change (such as rodent pest species) and thus most resistant to biodiversity loss (and land degradation), potentially linking loss of wildlife species to increasing disease transmission risk. In the absence of data about the drivers of specific outbreaks, current scientific paradigms often recommend the maintenance of biodiversity as a buffer against disease (Civitello *et al.*, 2015; Keesing *et al.*, 2010; McCallum, 2015), as well as the restoration of diverse communities in degraded ecosystems. However, the biodiversity dilution effect is still a topic of significant controversy and has been not been observed in some cases (Salkeld *et al.*, 2013). Consequently, some studies have concluded that arguments based on other benefits of land restoration and ecosystem health are more convincing than disease dilution, as dilution may depend more on the species in a community than total richness (Randolph & Dobson, 2012).

5.4.2 Land degradation, human-wildlife contact, and zoonotic spillover

Although less closely documented, deforestation is similarly one of the biggest drivers of increased burden from directly-transmitted zoonotic diseases with rare and episodic spillover from wildlife. For example, studies have found that deforestation and land degradation facilitate rodent reservoirs of zoonoses in Southeast Asia (Morand *et al.*, 2015). Similarly, evidence indicates that the sudden increase in emerging infectious disease spillover events originating in bats in West Africa and Southeast Asia is likely a product of deforestation, for agricultural purposes especially, that pushes bats into human-occupied landscapes (Jones *et al.*, 2013; Schmidt *et al.*, 2017; Wallace *et al.*, 2014). Land degradation also effects social changes that can change patterns of human-wildlife contact and thereby indirectly change patterns of zoonotic spillover. Land-use transitions have distanced many populations from sources of infectious disease, such as bushmeat (a common reservoir for viral spillover). Viruses that spill over from wildlife hunting or contact (i.e., viruses capable of making the cross-species jump), are especially likely to be directly transmissible within human populations (Johnson *et al.*, 2015). Increasing the distance, in particular between human dwellings and livestock or wildlife, substantially reduces the direct risk of zoonotic spillover. However, land-use change can also increase the force of infection of some spillover diseases. For example, deforestation is believed to be linked to an increase in bushmeat consumption in many regions, which provides a pathway for spillover of viruses like Ebola and Marburg fever (Foley *et al.*, 2005) (Box 5.7).

While the negative impacts of land degradation on biodiversity are widely undesirable, biodiversity loss may not always be a driver of zoonotic emergence. Compared to vector-borne diseases, directly transmitted zoonoses lack a theoretically-established mechanism for a biodiversity dilution effect that would be reduced by land degradation. Moreover, strong evidence suggests higher biodiversity ecosystems have a higher overall diversity of pathogens in their zoonotic pool (Han *et al.*, 2016), though this may not directly correspond to patterns of infectious disease emergence (Jones *et al.*, 2008). For instance, maintaining diverse ecosystems on shared grazing lands, if responsible for increasing wildlife-livestock contact, could increase spillover and spillback of diseases like anthrax, brucellosis and bovine tuberculosis (Kruse & Handeland, 2004). This could ultimately increase spillover of human disease through livestock, such as the Nipah virus spread from bats, via pigs, to humans (Pulliam *et al.*, 2011). However, biodiversity may sometimes act as a buffer to the invasion of introduced species, and therefore the pathogens they vector or carry. Biological invasions or introductions have facilitated the majority of some classes of disease emergence (Anderson *et al.*, 2004), and as a consequence of climate change and similar drivers, pests and the vectors and hosts of diseases are globally experiencing range shifts and expansion (Léger *et al.*, 2013; Ostfeld & Brunner, 2015). These shifts have the clear potential to drive disease emergence in new ecosystem, especially in naïve host populations without immunity to new pathogens. The restoration of degraded ecosystems and maintenance of biodiversity hotspots is likely to slow the spread of invading facilitators, a special case of the more general idea that ecosystem diversity contributes greatly to the maintenance of natural enemies in cultured systems (Landis *et al.*, 2000). Consequently, from the perspective of biodiversity-disease relationships, a strong case exists for the restoration of degraded ecosystems.

Box 5.7 Deforestation, bushmeat, and virus emergence in the Congo basin

The spillover of pathogens from zoonotic reservoirs into human populations via bushmeat hunting and trade is one of the most complex avenues of infectious disease emergence, and highlights the challenging interplay of land use and development with patterns of emerging disease. Deforestation and associated development practices, such as road building and increased forest edge settlement, have the potential to significantly increase human-wildlife contact. The degree of extraction, however, also sets ecological changes in motion that in some contexts can amplify or reduce disease prevalence in reservoirs and vectors (Wolfe *et al.*, 2005). Regional variability in demand for bushmeat, and different food preparation practices, further contribute to exposure levels.

The Congo basin and surrounding region is characterized by high local dependency on bushmeat, as well as high rates of deforestation – a combination of factors that predisposes the region to a particularly severe burden of zoonotic diseases. The region has also been identified as the point of origin for a number of significant viral zoonoses including Ebola and Marburg viruses, monkeypox, and HIV/AIDS, all of which were likely first transmitted into human populations via bushmeat. HIV has become endemic in human populations, while spillover of Ebola still represents an unpredictable and enigmatic problem for local public health institutions. Even though the reservoir of Ebola (and many other elements of its basic biology) is controversial, recent work shows that Ebola spillover events are highly associated with hotspots of forest fragmentation due to deforestation (Rulli *et al.*, 2017). In response to the 2014 outbreak of Ebola, some have called for an end to the bushmeat trade in West Africa as a net benefit to both human health and primate conservation, and as the simplest solution to the continued threat of disease spillover in the region. Others have criticized that approach by conservation groups as potentially "tone-deaf" (Pooley *et al.*, 2015), particularly given that bushmeat is most significantly consumed by poor populations who depend on it for nutrition (Wolfe *et al.*, 2005). Land conversion can open up new pathways for more sustainable meat production, potentially lessening financial disparities and decreasing food insecurity for poor local populations (and cutting the Gordian knot of bushmeat and Ebola). However, while deforestation and land conversion may yield a short-term benefit to bushmeat availability (at a cost to long-term availability as wildlife populations decline), that has been shown in Congo to primarily benefit non-local, non-indigenous populations (Poulsen *et al.*, 2009).

5.4.3 Agriculture, livestock disease, and land-use change

Agricultural health is intimately tied to human health through three pathways: (i) increased human encroachment on natural areas in the process of agricultural land conversion; (ii) the direct sharing of pathogens through caretaking or consumption of livestock (which offers a stepping stone for pathogens to spread from the environment into human populations); and (iii) the indirect cost of livestock disease on food security and nutrition. The latter two pathways are often correlated, and outbreaks of livestock disease can have particularly negative human health impacts on local communities by depriving them of nutrition simultaneous to disease outbreaks. Because agricultural intensification is usually related to land conversion (often deforestation), increased contact between agriculture and disturbed land often introduces zoonoses and other pathogens into human populations.

Deforestation is especially common as a driver of agriculturally-linked outbreaks. Encroachment on forests alone is a particularly common disease driver for pathogens like leishmaniasis, malaria, and others; for example, farmers in deforested areas were the first to present with the rare Kyasanur forest disease (a tick borne viral disease) in India (Jones *et al.*, 2013). More directly, livestock can act as an intermediate host through which viruses enter human populations, such as in the transmission of Nipah virus from bats to humans via pigs. Agricultural intensification, especially at fragmented ecosystem edges, can especially amplify this process. In some cases, overcrowded livestock populations offer an environment for pathogen evolution that allows otherwise-impossible spillover events, as in the possible spread of new strains of highly pathogenic avian influenza via poultry into humans.

Especially in cases related to deforestation, agricultural intensification is liable to come at a cost to water quality, providing another entry point into human populations for disease. Agriculture-related irrigation amplifies several classes of pathogens, especially vector-borne diseases. For example, outbreaks of Japanese encephalitis virus (a mosquito-borne illness) are driven by the interaction of irrigation and pig farming, as pigs are an amplification host that intensify human outbreaks; and irrigation has similarly been linked to outbreaks of Rift Valley fever and human fascioliasis (Jones *et al.*, 2013). Protozoan diseases, especially cryptosporidiosis, are spread from livestock to humans when contaminated runoff enters waterways, sometimes capable of producing outbreaks in the hundreds of thousands of cases from a single storm event (Myers *et al.*, 2009). Water contamination also poses a significant problem for the spread of drug-resistant pathogen strains at the wildlife-human-interface. Macroparasitic diseases, like parasitic worms, may be particularly favoured by “environmental nutrient enrichment” from agricultural runoff (Jones *et al.*, 2013). Overuse of antibiotics in agriculture have produced one of the most significant modern crises in public health, driving the emergence of antibiotic-resistant bacteria in livestock that ultimately spill over into human populations (Witte, 1998), and a similar problem exists for the use of antibiotics in fisheries (Cabello, 2006). Drug-resistant strains often originate in sewage, as pharmaceutical compounds and their derivatives enter waterways through pollution and runoff, circulate in degraded ecosystems, and re-enter human populations via livestock. This can pose a severe threat to human populations; for example, Tamiflu-resistant influenza has originated in wild waterfowl and could re-enter poultry stock in the future (Järhult, 2015).

Climate-driven land changes are likely to change the disease dynamics of the human-livestock interface in complex ways. For example, aridification is likely to increase the burden of currently neglected diseases like anthrax that are tightly associated with desert environments. The relationship between anthrax, a soil-transmitted bacterium, and different types of soil degradation is poorly understood, and livestock outbreaks with human impacts could become more common over time (though little data has been collected). However, for other classes of pathogen, especially vector-borne diseases, evidence suggests the net impact of climate change may be comparatively less than the impact of land-use change. For example, land conversion is predicted to make a far more substantial impact on the overall burden of African trypanosomiasis (a disease of both cattle and humans) than climate change (Thornton *et al.*, 2009).

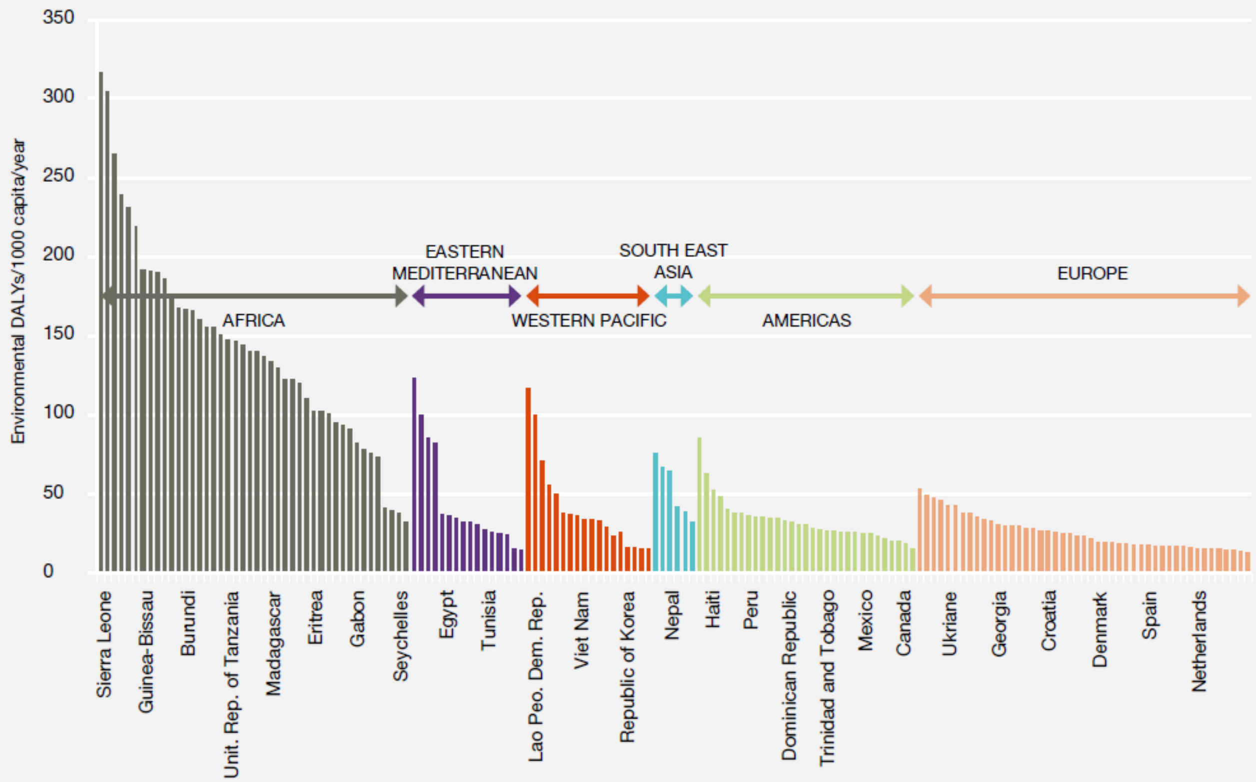
5.4.4 Water contamination and value lost from bioremediation

The degradation of natural ecosystems, and the intensifying human use of landscapes, is almost invariably associated with soil and water contamination (Nriagu & Pacyna, 1988). The focus of this section is primarily on water contamination and loss of regulatory bioremediation services. Neverthess, soil pollution from industrial process, while long known, is now being recognized as a key area where human land degradation is impacting human health (Brevik & Sauer, 2015). For example, in Europe it is estimated that there are 250,000 sites out of a total of 3 million that are in urgent need of remediation for heavy metal or oil pollution. The negative health impact of these sites included increased risk of cancer, kidney and bones diseases as well as neurological damage (Science Communication Unit, 2013). There is a growing body of literature exploring the impacts and options for restoration (Brevik & Sauer, 2015; Su *et al.*, 2014).

Turning to water contamination, destruction of wetlands and other ecosystems, that transform and accumulate nutrients (especially nitrogen and phosphorous) and toxins, often releases those directly back into waterways, to the detriment of human health. More intense land degradation programs like open-pit mining produce toxic runoff especially in the form of heavy metals (Dudka & Adriano, 1997), while destruction of Amazonian rainforests has been linked to the release of high levels of mercury into the soil (Mainville *et al.*, 2006). Polluted soils also significantly decrease agricultural yields, and in downstream impacts, heavy metal toxicity in humans (especially from arsenic, lead, cadmium, or mercury) can lead to both acute illness, and long-term neurological damage. Urbanization consistently increases pollutant load, both water- and airborne, while decreasing or eliminating natural ecosystems that filter those toxins, leading to human health threats like atmospheric brown clouds (Myers *et al.*, 2009). Similarly, urban and peri-urban slums with poor sanitation face a particularly severe risk from cholera outbreaks and from diarrhea and the responsible bacteria. The negative health impacts from water contamination and air pollution, as with many impacts of land and environmental degradation, are distributed unequally: the countries faring the worst in terms of those health outcomes see losses of years of healthy lives (DALYs) more than 20 times greater than the countries faring the best (Figure 5.8) (Prüss-Ustün *et al.*, 2008).

Figure 5 8 **Disability-adjusted life years (DALYs) lost per 1000 people per year as a result of a degraded environment.**

Exposes very large geographic variation–most of the health burden falls to a small proportion of countries. The data included is relatively narrow in its definition of environmental hazards, including only unsafe water and sanitation and indoor and outdoor air pollution. A broader definition of land degradation would shift these numbers, but it is likely that the general pattern of extremely unequal distribution would remain. Source: Prüss-Ustün *et al.* (2008).



One of the most significant human health costs of land degradation and biodiversity loss is the elimination of ecosystem functions and services like bioremediation, the potential for naturally occurring plants and microbes to remove toxins and pathogens from waterways and soil. The bioremediation services that most pristine ecosystems provide generate a significant net benefit to human health globally. Particularly well studied are the services that wetlands provide via water filtration and forests provide via air filtration, both through the removal of inorganic pollutants. Uninterrupted riverine systems that rapidly transport water without obstacles may make little impact; whereas landscape features like vegetation can slow down flow, maintain microclimates that optimize microbial processes relevant to bioremediation, and even increase flow overall, thereby diluting pollutants (Brauman *et al.*, 2007). This can be particularly important at the interface of natural ecosystems and degraded land; for example, studies show that up to a third of nitrate pollution from agriculture can be removed by every meter of streamside vegetation (Brauman *et al.*, 2007). Recent evidence shows that seagrass meadows act in a similar bioremediation capacity to filter out bacterial pathogens and reduce disease risk (Lamb *et al.*, 2017). These processes are poorly studied and poorly understood, but undoubtedly play a significant role in human health outcomes during terrestrial or marine land-use change.

Land restoration projects like the restoration of wetlands have the potential to recreate some lost ecosystem services (Horwitz *et al.*, 2012). Evidence suggests that wetlands and forests can seldom be restored with comparable diversity and resilience to their pristine state; but created or restored ecosystems can have greater targeted success in bioremediation, as some introduced species can have greater success removing heavy metals than native ones (Weis & Weis, 2004). One study suggests that restoring wetlands in a tenth of the Mississippi Basin would "reduce 10% to 40% of the nitrogen currently creating the hypoxic zone in the Gulf of Mexico" (Mitsch *et al.*, 2001). There is concern that restored wetlands may reintroduce mosquito populations that can be potential disease vectors, however the evidence for this is inconclusive and there are a range of options for managing mosquitos (e.g., Dale & Knight, 2008). In recognizing potential trade-offs for decision makers working to optimize the public health benefits of ecosystem restoration (Willott, 2004), care must be taken to ensure gains in biodiversity and bioremediation functions can proceed without increasing risks from disease vectors.

5.4.5 Clinical value of biodiversity

Adverse direct impacts of land degradation and biodiversity loss on healthcare in clinical settings are likely significant but, compared with direct impacts through disease, they are comparatively hard to quantify. One of the most important clinical benefits of natural systems is the availability of medicinal plants and resources, an important part of traditional ecological knowledge that not only benefits local health, but can provide key income to communities, especially to women (Mogotsi *et al.*, 2006). The same resources also provide a broader global health benefit through the potential for discovery of new medically-relevant compounds. Some of the most commonly used drugs, like aspirin, artemisin are derived from botanical compounds and the widely prescribed ACE inhibitors used to treat high blood pressure were first discovered in the Brazilian pitviper (*Bothrops jararaca*) (Vonk *et al.*, 2011).

The destruction of plant biodiversity hotspots like the Amazon, and associated species loss, could potentially lead to the loss of future pharmaceutical discoveries. However, the distribution and value of those discoveries is nearly impossible to forecast. Ecosystem valuation based on pharmaceutical discoveries is often controversial, making it difficult to assess the value lost because of land degradation. Some work suggests that at the per-species scale, these benefits might be negligible. A theoretical model developed by Simpson *et al.* (1996) shows that the marginal value of a given species for bioprospecting declines rapidly as total diversity increases, especially in a scenario where all species could equally merit investigation. Further, even if the loss of bioprospecting opportunity represents a cost of land degradation, it is almost universally one disconnected from the local communities directly affected by (and potentially benefitting from) those changes; and other benefits of conservation like carbon storage have been shown to make a far more significant difference in local cost-benefit analysis of conservation opportunity costs (Naidoo & Ricketts, 2006).

5.4.6 Psychological well-being and health improvements from interactions with natural landscapes

The impact of experiences with natural landscapes on human well-being has been emphasized for its significant and synergistic roles in improving physical and psychological well-being (Bowler *et al.*, 2010; Bratman *et al.*, 2012; Irvine & Warber, 2002; Lee & Maheswaran, 2010; Russell *et al.*, 2013; Sandifer *et al.*, 2015; Strife & Downey, 2009). Two bodies of work strongly support the linkage between physical and psychological health and natural landscapes. The first shows how interactions with natural landscapes improve health. The second demonstrates how loss, disconnection or degradation of natural landscapes negatively impacts health. For example, the stress level of people living near degraded forest was higher in Côte d'Ivoire (van Haaften & van de Vijver, 1996) and in the Sahel (van Haaften & van de Vijver, 1999). However, due to complex relationships between physical health, mental health and human well-being, a causal relationship between mental health and interaction or exposure to natural landscapes are often difficult to confirm (Lee *et al.*, 2010).

Urban greenspaces have been shown to have a positive impact on the physical health of residents, particularly with respect to cardiovascular conditions. This has been demonstrated through a spatial association between tree cover and self-reported health (Kardan *et al.*, 2015) as well as with a natural experiment where a sudden reduction in tree cover reduced health scores in affected areas (Donovan *et al.*, 2013). In addition, in a hospital setting, the reduction in stress associated with a view of natural landscapes from patient rooms has been found to have a surprisingly pronounced effect on surgery success, clinical improvement, and later health problems (Maller *et al.*, 2006). Finally, a recent prospective cohort study found that just living near greenness reduced non-accidental mortality by 12% (James *et al.*, 2016).

Looking beyond physical health effects, natural landscapes have been shown to improve psychological health in various ways. These include prevention and reduction in mental illness as well as “relaxation from stress”, “positive emotions”, “attention capacity” and “cognitive capacity” (Tzoulas *et al.*, 2007). People can get obtain these benefits by having contract with natural landscapes in different ways including: knowing, perceiving, interacting and living (Russell *et al.*, 2013).

Simply viewing natural landscapes has itself been shown to benefit mental health (Kaplan, 2001; Maller *et al.*, 2006) with the stress level of people exposed to natural landscapes reduced and recovery from stress faster in instances where people viewed natural landscapes as compared to urban landscapes (Ulrich, 1979, 1981, 1984). In addition, the recovery rate from psycho-physiological stress and mental illness was higher in people exposed to natural landscapes as opposed to those in urban areas (Berto, 2014). Green spaces like gardens in hospitals have also been shown to reduce stress and pain for both patients and visitors (Sherman *et al.*, 2005).

Direct interactions with natural landscapes can also be highly beneficial; walking or running through green parks and green university areas has been shown significantly reduced anxiety and rumination (an indicator of depression), while also increasing self-esteem and working memories (Alcock *et al.*, 2014; Barton *et al.*, 2010; Bowler *et al.*, 2010; Bratman *et al.*, 2015a). A Stanford University study that used brain imaging on healthy patients also showed that rumination, a psychological term to describe a state of the mind that sometimes leads to depression, is significantly reduced when walking 90 minutes in a natural landscape as compared to an urban landscape (Bratman *et al.*, 2015b).

The positive effects of exposure to natural landscapes is especially important for those who are more vulnerable, such as children (Strife *et al.*, 2009; Taylor & Kuo, 2006). There is a growing body of literature indicating that the children who experience more outdoor recreation in nature have not only improved physical conditions, but also better psychological well-being by reducing stress level, and accomplished higher academic achievement (Kellert, 2005; Strife *et al.*, 2009; Taylor & Kuo, 2006; Wells & Evans, 2003). This is confirmed by studies in the US that found as the opportunity for children to experience natural areas has decreased in recent years there has been an accompanying increase in the level of depression in children (Louv, 2005; Strife *et al.*, 2009; Wells & Lekies, 2006).

Looking towards future research on this important but understudied linkage between natural landscapes and human health, most research to date has bene conducted in developed countries such as the USA (e.g., Bratman *et al.*, 2015a), the U.K (e.g., Alcock *et al.*, 2014; Seresinhe *et al.*, 2015), and Sweden (e.g., Sundquist *et al.*, 2004). However, the most rapid urbanization is currently occurring in developing parts of Africa and Asia (UN, 2014). In these areas, there are already documented effects on loss of recreation in natural landscapes due to urbanization. For example, in Shenzhen, one of the fast-growing cities in China, residents experienced 10% less benefit from recreation in 2004 compared to the value of similar benefits in 1996, in part due to decreasing areas of woodland, wetland and water bodies (Tianhong *et al.*, 2008; Zhou *et al.*, 2011). In Baguio city in the Philippines, the provision of recreation benefits also decreased due to loss of forest cover from expansion of urban areas (Estoque & Murayama, 2012). Future research should thus focus on the effect of the rapid urbanization on mental health through loss of greenspace in these parts of the world as well as the possible benefits of restoration (Shanahan *et al.*, 2015).

5.5 Disasters, hazards, and extreme events

There is a growing evidence indicating that changes in land cover affects rate of occurrence and severity of natural hazards (MA, 2005; Nel *et al.*, 2014). Natural hazards may be defined as physical phenomena caused by rapid or slow onset events, which can be geophysical (earthquakes, landslides, tsunamis and volcanic activity), hydrological (avalanches and floods), climatological (extreme temperatures, drought and wildfires), and meteorological (cyclones and storms/wave surges). A disaster is the complex set of effects of hazards on human populations. The United Nations International Strategy for Disaster Risk Reduction (UNISDR) defines disasters as “a serious disruption of the functioning of a community or a society at any scale due to hazardous events interacting with conditions of exposure, vulnerability and capacity, leading to one or more of the following: human, material, economic and environmental losses and impacts” (UNISDR, 2015).

Natural disasters determine the interface between extreme physical elements and vulnerable human population (O´Kefee *et al.*, 1976; Sidle *et al.*, 2004). According to Sidle *et al.* (2004), a society experiences natural disaster, when there is such a great environmental disruption that surpass societal coping capacity. Natural disasters affect human societies, often destroying natural and physical capital and economic assets (Dilley *et al.*, 2005; Ibarrarán *et al.*, 2009).

The interconnections and feedbacks between land degradation, environmental management and disaster risk are complex and multifaceted. Poor watershed-scale, urban or regional planning might exacerbate the risk and reach of the so called “natural disasters” (Dolcemascolo, 2004). Land and environmental degradation also exacerbates the impact of natural disasters, by affecting natural processes, altering humanity’s resource base and increasing vulnerability. The degree to which environment can absorb impacts, increase overall resilience and provide effective and economical solutions to reduce disaster risks is therefore jeopardized by land and environmental degradation (UNISDR, 2004). This increases the economic, social and political burdens and costs for mitigation and recovery or restoration. Human populations are often disproportionally affected by disasters. Women, children, poorer communities and indigenous peoples are among the most vulnerable social groups (UN, 2015).

Trends in socio-economic factors, such as urbanization, expansion of economic activities, and population increase, will lead to greater vulnerabilities of people and economic assets to natural hazards (Adger & Brooks, 2003). Moreover, in the near future, current increasing trends in natural disaster frequency and associated economic damages are expected to continue (Adger & Brooks, 2003). In general, at global scale, current economic losses caused by natural disasters (earthquakes, tsunamis, cyclones and flooding) on average accounts annually for $250 billion to $300 billion, however future annual losses are expected to rise to $314 billion, when accounting only for built environment (UNIDSR, 2015).

The Sendai Framework for Disaster Risk Reduction for the 2015-2030 period is the main policy-oriented international instrument to guide disaster risk reduction by member governments. An important goal of the framework is to prevent new and reduce existing disaster risk through the implementation of integrated and inclusive economic, structural, legal, social, health, cultural, educational, environmental, technological, political and institutional measures that prevent and reduce hazard exposure and vulnerability to disaster, increase preparedness for response and recovery, and thus strengthen resilience (Aitsi-Selmi *et al.*, 2015; UN, 2015).

5.5.1 Costs and benefits of hazard avoidance

The regulating ecosystem services of hazard and disaster mitigation and regulation include storm protection, flood control, drought recovery, fire prevention, and coastal protection, which is investigated in the context of land degradation.

Ecosystems deliver a broad variety of ecosystem goods and services that contribute to human well-being. De Groot *et al.* (2012) provided global estimates of the value of ecosystem services of ten main biomes based on 665 value estimates out of approximately 320 publications that were standardized and stored in the Ecosystem Service Valuation Database (ESVD). ESVD value estimates were based on published individual case studies estimates that were converted into the standardized unit (US$/ha/year, 2007 price levels). The results show that most of the values are outside the market; many of the positive ecosystem externalities are lost or significantly decreased after land-use conversion. In case of disturbance moderation (Hazard and Disaster regulation), coral reefs estimation showed the highest mean value (16,991 US$/ha/year, 2007 price levels), followed by coastal wetlands (5,351 US$/ha/year), inland wetlands (2,986 US$/ha/year) and tropical forests (66 US$/ha/year).

Coral reefs provide significant disturbance moderation by reducing the wave energy, which would otherwise impact the coastal areas. Meta-analysis of Ferrario *et al.* (2014) based on 27 publications across the Atlantic, Pacific and Indian Oceans revealed that the whole coral reefs reduce wave energy by 97%. Moreover, by absorbing the storm energy coastal wetlands contribute to hurricane protection of coastal communities (Costanza *et al.*, 2008). However, there has been only limited number of studies (e.g., Farber, 1987; Costanza *et al.*, 1989; Barbier *et al.*, 2013) focusing on estimating the value for hurricane protection. Using spatially explicit data (on hurricane tracks, wetland area, storm damages and GDP), Costanza *et al.* (2008) analysed 34 major USA hurricanes since 1980 and estimated the annual value of coastal wetlands for hurricane protection. In the USA, the coastal wetlands were estimated to provide storm protection services annually accounting for $23.2 billion. The study also accounted for coastal wetlands changes. For instance, in Louisiana, 480,000 ha of coastal wetlands were lost before hurricane Katrina in 2005 and 20,000 ha during Katrina (Costanza *et al.*, 2006). The value of lost storm protection in Louisiana accounted approximately for 1,700 US$/ha/yr, and when multiplied by the area of lost wetlands, approximately $816 million yr^{-1} prior to Katrina and $34 million yr^{-1} during Katrina of wetland storm protection services were lost (Costanza *et al.*, 2008). More recently, during Hurricane Sandy, which hit the densely populated east coast of the USA in 2012 and was the second costliest hurricane in USA history, coastal wetlands saved an estimated $625 million in avoided flood damages. Following the storm, where wetlands were present, property damages were lower than in locations where wetlands were absent, with a 29% reduction in damages in areas near Washington DC (Narayan *et al.*, 2016).

Floodplains and wetlands contribute to flood mitigation, particularly by storing and decreasing peak water flows (MA, 2005; Bullock & Acreman, 2003). Watson *et al.* (2016) estimated the flood mitigation value of Otter Creek (Middlebury, Vermont, USA) floodplains and wetlands for the tropical storm Irene and nine other floods. The avoided damage costs (to inundated structures) were calculated based on flood extents scenarios (with and without floodplains and wetlands) for ten flood events. The study shows that floodplains and wetlands can provide important flood mitigation service, with damage reductions of 84-95% for tropical storm Irene and average damage reduction of 54% to 78% among all ten events. Moreover, mean annual value of flood mitigation service provided by Otter Creek floodplains and wetlands to Middlebury range from $126,000 based on no-wetlands low scenario, to $459,000 based on high scenario.

5.5.2 Degradation and hazards on land and freshwater systems

Deforestation and loss of native vegetation may increase or worsen the number of flood-related disasters (Bradshaw *et al.*, 2007; Tan-Soo *et al.*, 2016). The global study of Bradshaw *et al.* (2007), based on empirical data collected from 1990 to 2000 representing 56 developing countries, demonstrated that loss of natural forest area led to increase in flood frequency and flood lasted longer.

However, the effect of deforestation on flooding at a national scale is not robust (Ferreira & Ghimire, 2012; van Dijk *et al.* 2009; Meyfroidt & Lambin, 2011). Ferreira and Ghimire (2012) and van Dijk *et al.* (2009) conclude that the relationship between natural forest cover and large flood events in developing countries is debatable, and might be better explained by socio-economic aspects (e.g., population density, and urban population growth, flood management, corruption).

On the other hand, a national scale study conducted in Malaysia for 1984-2000 using disaggregated data on land-use types, provided robust evidence that deforestation and conversion of inland tropical forests to oil palm and rubber plantations can lead to increase in number of days flooded during heavy rainfall periods (Tan-Soo *et al.,* 2016). Furthermore, de la Paix *et al.* (2013) mapped deforestation in Rwanda leading to increase in floods that cause loss of human life. Between 1997 and 2008, 1,682 hectares of forest were destructed annually to be used mainly as fuelwood to secure local livelihood and industries (e.g., tea factories). Globally, flooding is the most prevalent natural disaster, causing more life losses compared to any other natural disaster (Doccy *et al.*, 2013)

In the last two centuries, the most significant societal interactions with natural hazards in Austral-Asian region has been indisputably generated by extensive land-cover changes, particularly due to deforestation, converting forest to farm and grazing land (Sidle *et al.*, 1985; Froehlich *et al.*, 1990; Garrity & Agustin, 1995; Harwood, 1996; Thapa, 2001).

Causes of increasing risk of natural hazards are often attributed to climate change and human-induced changes in land-cover management. Nel *et al.* (2014) quantified the impact of climate change and land-cover change on four natural hazards (e.g., floods, drought, wildfires and storm-waves) using scenario-based modelling in Eden district, South Africa. The findings showed that human-induced changes in land management are likely to increase the risk of natural hazards. For instance, changes in plantation forestry pushed the flood events from 1:100-year flood event to a 1:80-year return period for the extreme scenario. The finding suggests that appropriate land-use management (e.g., clearing invasive alien trees, re-vegetating clear-felled forests, and restoring coastal foredunes) substantially contribute to reduction of natural hazards impacts.

Climate change effects forests through disturbances, by changes in intensity, frequency and duration of fire, drought, introduced species, pathogens, hurricanes, windstorms, ice storms or landslides (Dale *et al.*, 2000, 2001). The combination of man-made technological hazards with climate change phenomena adds a great level of uncertainty regarding the frequency and magnitude of higher temperatures, drought and flood damages to both terrestrial and freshwater ecosystems. In Brazil, a country highly dependent of hydropower for electricity production, the extent to which climate change-related droughts and floods will impact the performance, security and reliability of hydroelectric dams has a high level of uncertainty, which makes long-term planning and decision-making challenging (Fearnside, 2017; Pittock, 2010; Prado *et al.*, 2016).

5.5.3 Degradation and coastal hazards

Around 10% of the world's population is currently living in coastal zones less than 10 meters above the mean sea level. By 2050, the global population living in the low elevated coastal zones is expected to substantially increase, to more than one billion (Merkens *et al.*, 2016). Furthermore, almost one quarter (23%) of world´s population live within 100 km distance from the coast (Small & Nicholls, 2003) and by 2030 it is expected to be half (50%) of the world´s population (Adger *et al.*, 2005). The economic losses and human risks associated with coastal habitat destruction are substantial (see also Section 5.5.1).

The resilience of coastal communities is more tightly connected to global processes, such as economic linkages (Adger *et al.*, 2005), globalization of commodity and ecosystem goods and services trade (Adger & Brooks, 2003; O'Brien *et al.*, 2004). In coastal regions, global tourism, as an ecosystem service, increases vulnerabilities of previously undeveloped coastal areas through land and environmental degradation (Davenport & Davenport, 2006).

The literature also shows substantial evidence that estuarine and coastal ecosystems (e.g., salt marshes, coastal wetlands, mangroves, sand beaches and dunes) provide services in terms of storm protection, protection against hurricanes, coastal floods and wave attenuation (Barbier, 2007, 2015; Costanza *et al.*, 2008; Barbier *et al.*, 2011; Gedan *et al.*, 2011; Shepard *et al.*, 2011; Spalding *et al.*, 2014). The loss or degradation of these ecosystems reduces their ability to provide protection from extreme events (Barbier *et al.*, 2007; Granek & Ruttenberg, 2007; Barbier *et al.*, 2011).

For instance, mangrove and salt marshes provide hazard and disaster regulation to local communities, by protection from erosion, storm surge and possibly small tsunami waves that is context-dependent (Gedan *et al.*, 2011). Even narrow bands of mangrove forest along a coastline can provide a meaningful amount of protection. Mangroves (coastal forests, located in tropical and sub-tropical regions) can reduce storm surge by 5 to 50 centimetres decrease in water level and reduce surface wind by more than 75% over one kilometre of mangrove width (McIvor *et al.*, 2012).

Removal of mangroves (often due to deforestation for intensive shrimp farming) diminishes the coastal protection in terms of storm protection to catastrophic events (e.g. hurricanes, tsunamis) as well as to more frequent low-energy events (e.g., tropical storms) (Barbier *et al.*, 2011; Godoy & De Lacerda, 2015; Granek & Ruttenberg, 2007; Lee *et al.*, 2014). Southeast Asia is the largest mangrove-holding region, at the same time a region with the highest mangrove deforestation rates between 3.58% and 8.08% per year (Hamilton & Casey, 2016).

Historically, mangroves have provided goods and services to the local communities (López-Angarita *et al.*, 2016). Barbier (2007) assessed mangroves ecosystem services of storm protection in Thailand. The net present value of mangroves as natural "coastal storm barriers" reached between 8,966 and 10,821 US$/ha. The valuation approach used was based on expected damage function (EDF), which estimates the value of how mangrove storm protection mitigates damage costs, taking into account changes in mangroves losses. When including annual mangrove deforestation estimate of 18 km^2 over 1996-2004, the annual welfare loss in mangrove storm protection for Thailand was estimated to be around $3.4 million ($2.3 to $5.8 million with 95% confidence).

IPCC (2014) states with very high confidence that combined effect of the drivers, such as sediment reduction, relative sea level rise, and land-use changes in estuarine ecosystems resulted in widespread degradation of deltas and deltaic coasts. The transformation of estuarine and coastal ecosystems substantially accelerated over the last centuries, and anthropogenic drivers pushed these ecosystems far from the historical baseline (Lotze *et al.*, 2006). Climate change factors, such as sea level rise, increase in storm events, changes in precipitation patterns and temperature increase, will likely have a significant impact on mangrove ecosystems (Ellison, 2015; Godoy *et al.*, 2015; Ward *et al.*, 2016).

Restoration of mangroves is often considered a way to provide hazards and disaster protection as well as additional ecosystem services to local communities (Iftekhar & Takama, 2008; Moberg & Rönnbäck, 2003). For instance, local communities in south-central estuarine island Nijhum Dwip Island in Bangladesh perceived major ecosystem services provided by mangroves as supply of raw materials (57% respondents), prevention against natural disasters (13% of respondents), climate regulation (13% respondents) and soil retention (12% of respondents) (Iftekhar *et al.*, 2008).

5.5.4 Role of ILK and IPLCs on disaster risk reduction and restoration

Indigenous peoples and local communities (IPLCs) around the world have retained relevant knowledge and coping strategies to face natural disasters, despite being among the most vulnerable and marginalized groups both in terms of disasters' preparedness, and in the aftermath restoration process (Lambert, 2014; Mercer *et al.*, 2007, 2010). IPLCs hold memories, stories, and experiential knowledge of social-ecological processes affected by disasters, which are being rapidly eroded with the loss of indigenous languages; formal westernized educational systems that do not recognize or articulate local and indigenous knowledge; lack of valorisation and respect for local traditions; urbanization of indigenous communities; problems of inter-generational knowledge transmission; and other interrelated factors affecting indigenous and local knowledge (ILK) erosion and persistence (Athayde *et al.*, 2017; Lambert, 2014; McCarter *et al.*, 2014; Mercer *et al.*, 2009; Walshe & Nunn, 2012).

In the international arena, policy instruments such as the UN's Hyogo framework (2005-2015), and more recently, the Sendai framework (2015-2030), have drawn attention to governments to the necessity of building resilience of nations and communities to disasters, recognizing the important role of indigenous communities in local, national and global disaster risk reduction practices (UN, 2015). Notwithstanding the recognition of the relevance of ILK and IPLCs for disaster risk reduction in international policy instruments, their implementation at local and national scales remains problematic in many high disaster-risk countries such as New Zealand, Nigeria, Vanuatu Islands and the Philippines (de Leon & Pittock, 2016; Lambert, 2014; Mercer *et al.*, 2007, 2010; Omeje, 2005).

In New Zealand, examining the aftermath of the 2011 Christchurch Earthquake, Lambert (2014) found that Mãori cultural practices of hosting and reciprocity (named *manaakitanga*) and kinship bonds (*whãnaungatanga*) were identified as contributing to community resilience. Mãori traditional communal meeting and learning places known as *Marae* have played an important role in New Zealand's disasters, providing spaces for indigenous and non-indigenous individuals and families. Even though most participants mentioned "being Māori" an important aspect of how and why they managed to cope with the earthquakes, the scale and severity of the overall disaster has meant serious impacts to Māori individuals and communities. The author concludes that disaster management policies and practices need to be more inclusive through meaningful collaboration with indigenous communities. In the case of New Zealand, it will require formal engagement with Maori communities and institutions, who need to be allowed to participate in disaster risk reduction (DRR) plans. In the Southern Pentecost Island of Vanuatu, Walshe and Nunn (2012) report the importance of local knowledge systems, referred to as *kastom* (a Bislama adaptation from the English "custom") to inform understanding and coping strategies toward earthquakes and tsunamis, which have historically impacted this region. Some *kastom* stories shared by interviewees demonstrate a common belief that human magic and spiritual beings have control over nature elements, and natural hazards such as tsunamis can be used to punish evil. One of the stories also offers guidance how to survive those waves by running uphill and avoid establishing residences in the low-lying areas. The authors argue that indigenous and local residents hold important stories and memories of recent tsunamis, which should be recalled, maintained, and applied towards future disaster risk reduction strategies and communication plans, along with science-based knowledge and technical procedures.

In an online video-documentary, "Dialogues between Indigenous Knowledge and Disaster Risk Reduction" (Amazon Dams Network, 2016), Maskoke activist and scholar Marcus Briggs-Cloud highlights the importance of conserving indigenous languages to maintain indigenous knowledge and communication pathways between indigenous peoples, spiritual leaders and disasters. He explains that natural elements and phenomena such as tornadoes, hurricanes, and rain carry ancestral spirits with whom spiritual leaders are able to connect and communicate through indigenous languages. Indigenous languages and lifeways also offer concepts, practices and experiences of disasters that might help science and society to understand, prevent, mitigate and manage their effects (Mercer *et al.*, 2010). Chief Herbert Jim, Seminole spiritual leader, tells the story of the spirit of a young man who was twisting and came to earth as a hurricane. People tried to control and kill him, cutting one of his arms, but they were not supposed to kill a nature being. Since then, Hurricanes have been coming back and punishing people. For New Zealand Māori groups, earthquakes and volcano activity are controlled by the God *Rūaumoko*, who rules geothermal activity (McSaveney, 2011). In parts of Bangladesh, cyclones are traditionally seen by Muslim groups as a punishment from Allah (Schmuk, 2000). In the island of Manpura, Bangladesh, preparedness attitudes are often guided by religious leaders, some of whom advocate prayer as the only appropriate measure (Howell, 2003).

Indigenous and local strategies, social-ecological indicators and weather forecasts, might inform different stages of disaster prevention and management (Figure 5.9). Local early warning systems and traditional weather forecasts are a critical component of preparedness among IPLCs. They involve using local environmental indicators and the reliance on informal personal networks to assist with interpreting the message and decision-making (Dekens, 2007). Indigenous and local weather forecasts are products of multigenerational observations of changes in the surrounding environment, and includes knowledge about the movements of the sun, the moon, and the stars. They might be developed over time, through need, trial, and error; or emerge, using pre-existing networks, and are simply disseminated among local communities, without requiring any special equipment or technology (Howell, 2003). Indigenous farmers in Peru and Bolivia use the appearance of the Pleiades to forecast the timing and quantity of precipitation for the rainy season, months later. They moderate the effect of reduced rainfall by adjusting the planting dates of potatoes, their most important crop. Orlove *et al.* (2002) uncovered the scientific basis of this knowledge, by articulating ethnographic information collected across twelve villages in the Andes, with climatological and atmospheric data assembled for the region. For the authors, the study of indigenous forecasts integrates a growing network that connects climate researchers, policymakers, administrators and citizens. The forecasts show that local social groups seek information that they can use to adapt to climate variations and or disasters. Combining traditional and scientific prediction techniques and data could be quite effective, helping meteorologists prepare useful projections, as well as improve communication between the producers and consumers of modern scientific forecasts (Dekens, 2007; Orlove *et al.*, 2002). According to Parker and Handmer (1998), one important disadvantage of local early warning systems relates to the fact that they are limited by the personal experience of member of the relevant network, and can not always be extended to other social groups beyond them.

Rural communities perceive animal behaviour and other natural phenomena as local early warning indicators of cyclones (Howell, 2003). These include, for instance, noticing dogs and domestic animals' behaviour and the turning leaves of the Mandar (cotton tree) in rural India; observing bird nesting sites along the trees; perceptions of water colour, quality and quantity; moon and stars positioning and phenomena (e.g., rings around the moon could be a sign of hurricanes); and many other context-specific indicators learned from practical experience and handed-on through generations by oral traditions, rituals and other customary practices (Athayde *et al.*, 2015; Galacgac & Balisacan, 2009; Mercer *et al.*, 2007; Molina & Neef, 2016; Sardali, 2013; Turner & Clifton, 2009).

Indigenous strategies such as placement or settlement in safer areas, are related to geographical (territorial) and seasonal memory and knowledge. This place-based knowledge might inform regional development plans, evacuation routes, or resettlement programs. Naomi Sherwood, an indigenous activist of the Oneida Nation of the Thames in the USA, mentions the importance of protecting forests and tress as buffers against wind and hurricane impacts and damages. Indigenous architecture also has helped communities to cope with hurricanes in Florida. According to Chief Herbert Jim, the wood and structure of indigenous houses helped to absorb the wind, protecting against hurricane impacts. Strategies for coping with disasters based on indigenous socio-cultural practices include reciprocity networks, spiritual guidance, and community-based food production and provisioning (Athayde *et al.*, 2015; also see http://indigenousknowledgenetwork.org/).

Stories, symbols, songs and rituals are also part of the oral lore of the indigenous peoples and local communities, used for communication and knowledge transmission across generations. The 'smong' song in Simeulue Island, west of Aceh, which was composed after the 1907 tsunami, helped locals to interpret signals of the 2004 tsunami (McAdoo *et al.*, 2006). According to Dekens (2007), based on work developed by the author with Nepalese ethnic groups, songs and proverbs may work as repositories of past flood events, contributing to the transmission of flood-coping strategies, creating common knowledge, and sharing a common understanding of environmental change events. These oral traditions can also help to build a sense of community and solidarity within the village and/or within the different groups affected.

Figure 5 9 The potential role of ILK throughout the stages of disaster management cycle.

The dashed lines indicate connection between the different ILK components across disaster prevention, mitigation, preparedness, response, rehabilitation and reconstruction. Source: Figure adapted from Bahadur *et al.* (2016).

Based on research carried out in different parts of the world, there is solid evidence to support the claim that indigenous peoples and local communities hold important knowledge and experiences to contribute to disaster risk reduction and management. Nevertheless, in the majority of the cases, communities and peoples have not participated, or being properly considered, in the development of communication, preparedness, mitigation, resettlement and reconstruction plans. A challenge remains for countries, regions, and cities, municipalities or villages, to develop appropriate policies and actions directed to broaden the inclusion of indigenous and local groups in DRR policies and practices. In addition, relevant institutions and actors need to coordinate efforts to address the socio-economic vulnerability of these groups to the impacts of disasters on land degradation and human well-being.

5.6 Human Security

Human security is defined as a condition where human lives are safeguarded and where people can live freely and to their full potential (Adger *et al.*, 2014). Land degradation has the potential to negatively impact human security, especially in countries and regions with high poverty rates and weak institutions. Land degradation can act as a threat multiplier for violent conflict, especially in countries and regions where weak institutions reduce the capacity to peacefully resolve disputes over limited resources (Bernauer *et al.*, 2012; van Schaik & Dinnissen, 2014). It may also force unwanted migration by limiting agricultural and rangeland productivity in areas of livelihood insecurity. The relationships between land degradation and both conflict and involuntary migration are not deterministic – they are mediated by social institutions and by societies' capacity for adaptation.

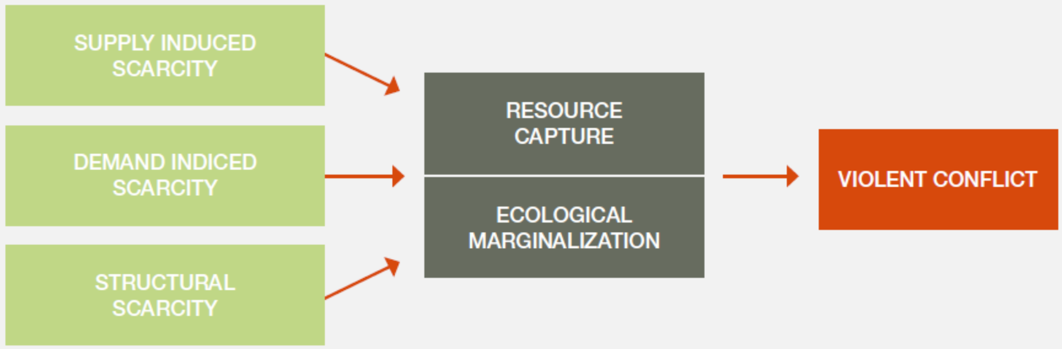
5.6.1 Conflict

Land and environmental degradation has the potential to play an increasingly important role in violent conflicts around the globe (also see Chapter 2, Section 2.2.2.3). The international community's recognition of this was highlighted when the 2004 and 2007 Nobel Peace Prizes were awarded to an individual and an organization – Wangari Maathai in 2004 and the IPCC in 2007 – for work safeguarding the environment. Resource scarcity has long been a central driver of conflict in human societies. Inter-state and intra-state violence has often had as its goal territorial acquisition or control over mineral, fossil fuel, or other resources. In recent decades, concern has arisen that land and environmental degradation may increase the risk of violent conflict by reducing access to natural resources and thereby increasing stress on individual livelihoods and on social systems. Land degradation may lead to decreased agricultural production on croplands, reduced water quality and storage, and constrained access to quality grazing land for livestock. These changes may lead directly to conflict as individuals and groups are forced to compete for the remaining resources. It can also lead to conflict indirectly by increasing levels of poverty and grievance or by decreasing the resilience of social and political structures.

5.6.1.1 Models of the degradation – conflict link

One line of reasoning supporting a link between land and environmental degradation and violent conflict stems from an updated version of the 18th century theories of Thomas Malthus. As land and environmental degradation reduces the rate at which resources can regenerate, it creates a resource shortage. When this decrease in the supply of resources is coupled with population increase and rising resource demand, resource scarcity rises. Neo-Malthusians describe this resource scarcity as leading to conflict via two primary mechanisms: resource capture and ecological marginalization (Bernauer *et al.*, 2012; Homer-Dixon, 1999). Resource capture operates when powerful elites manipulate social and political structures to their benefit to ensure access to resources. As resource availability declines, this problem is exacerbated, leading to a reduction in support for governance structures, an increase in grievances, and a rise in instability and potentially violence (Bernauer *et al.*, 2012). Ecological marginalization is the process where groups threatened with limited access to resources are forced to migrate into new areas that may already be environmentally stressed. This may result in conflict between migrants and new arrivals. The neo-Malthusian framework for the environment-conflict link is shown in Figure 5.10.

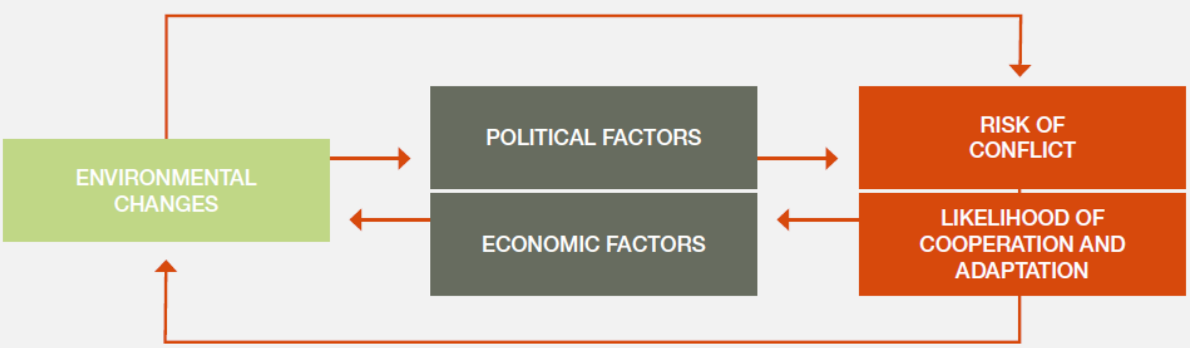
Figure 5 10 A neo-Malthusian framework for the link between land and environmental degradation and conflict. Source: Bernauer *et al.* (2012).



The neo-Malthusian description of resource scarcity leading to conflict is criticized by some as being overly deterministic in its outlook (Buhaug *et al.*, 2014; Burrows & Kinney, 2016; Gleditsch, 1998; Salehyan *et al.*, 2008). Social capital, institutions, and adaptation can all reduce or eliminate the risk of conflict resulting from degradation and climate change. Although many case studies exist where land degradation was linked to conflict events (e.g., Box 5.9, Box 5.10) (cases reviewed in Bernauer *et al.*, 2012), there are similarly many cases where land degradation occurred with no apparent increase in conflict or conflict risk (Salehyan, 2014). The probable cause for this discrepancy is that conflict is a complex socio-political process that cannot be explained by environmental processes in isolation (Buhaug *et al.*, 2014). Authors who dispute the neo-Malthusian conception of the environment-conflict link as overly-simplistic suggest that any link between degradation and conflict is more likely to act via indirect effects (Figure 5.11). In particular, the risk of conflict may increase because of the negative effect of reduced agricultural productivity on rural livelihoods and by weakening political institutions (Bernauer *et al.*, 2012).

In addition to the impact on livelihoods and institutions, land degradation may increase out-migration of affected areas (see Section 5.6.2) that may itself lead to conflict between new arrivals and longer-term residents (Reuveny, 2007; Watts, 2012; Box 5.9).

Figure 5 11 An expansion of the neo-Malthusian conception of the environment-conflict link shows the central role of indirect effects, particularly those mediated by local political and economic factors. Source: Bernauer *et al.* (2012).

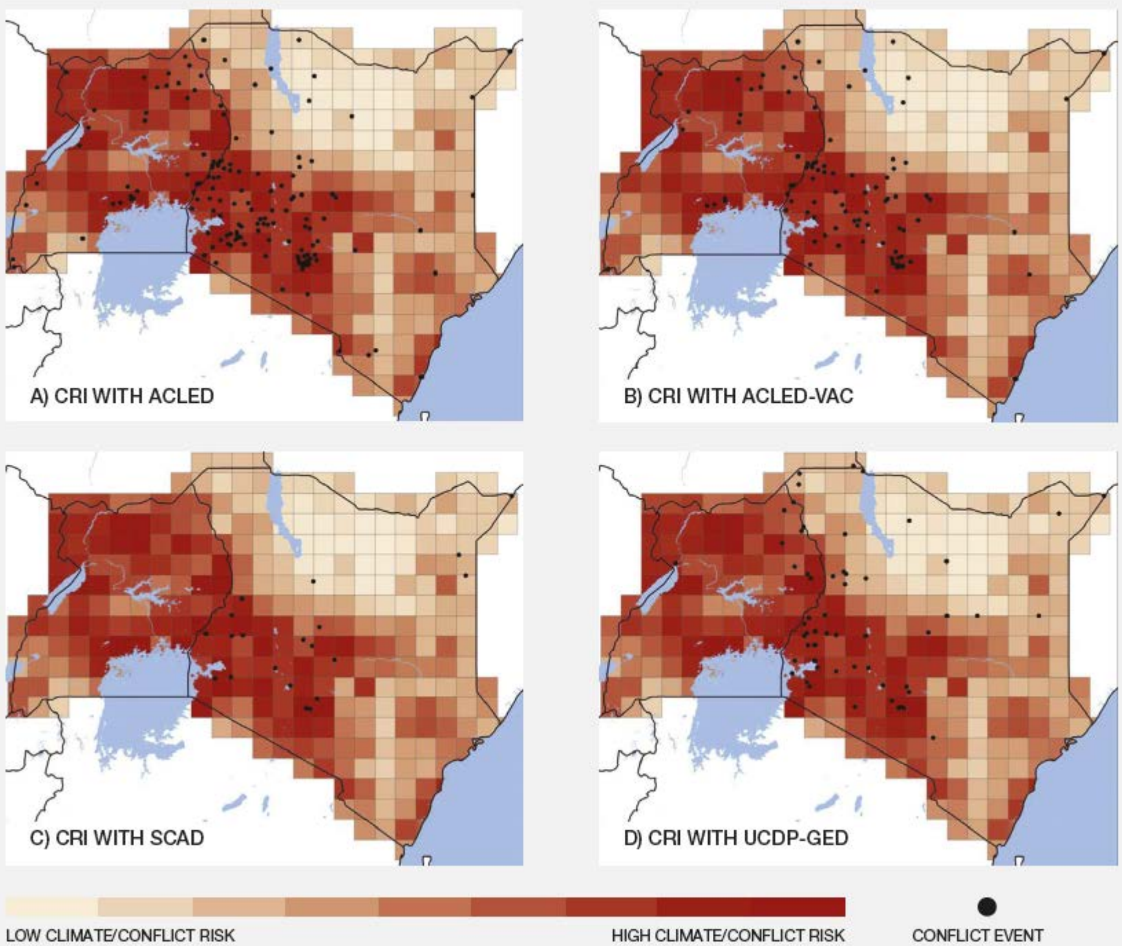


Box 5.8 Climate stress on rangelands and violence in pastoralist communities

Several studies of the link between natural resource stress and conflict have looked at the empirical relationship between short-term climate changes and the associated impacts on rangeland productivity and increases in inter-group violent conflict among pastoralists in East Africa (Ayana *et al.*, 2016; Ide *et al.*, 2014; Maystadt *et al.*, 2015). A climate-land-conflict is more likely in this context than in most others for several reasons: (i) a high level of livelihood dependence on the annual productivity of rangelands; (ii) high levels of poverty and limited adaptation capacity through outside economic opportunities; and (iii) a history of intergroup violence and livestock raiding in the region that predates concerns around climate change. While these climate and land attributes do predict conflict to some degree, an important message from the data is that climate and land processes are only one part of the picture: there are many conflicts that are better explained by an examination of social and political factors (Figure 5.12). Climate and land may magnify the risk of conflict, but they rarely, if ever, act as the sole cause of a conflict.

Fig. 5 12 **Risk map for conflict in Uganda and Kenya.**

Map includes land quality metrics such as soil degradation, exposure to temperature increases and precipitation reduction, and social-political indicators such as population density and level of democracy. Results indicate a general concentration of inter-group conflict in areas where climate and land metrics predicted high conflict risk. However, there were many exceptions, i.e. a lack of conflict in areas that predicted high risk and conflict occurring in areas where it would not have been predicted. These observations align with other studies (Ayana *et al.*, 2016; Jean François Maystadt *et al.*, 2015) that found slight increases in conflict with climate changes that would reduce land productivity, but more robust patterns with indicators of poverty and social marginalization.



5.6.1.2 Link between land degradation, economic growth, and conflict

There is a robust relationship between poverty, economic growth, and levels of conflict: countries and regions with lower levels of poverty and higher levels of economic growth consistently experience lower levels of violent conflict (Detges, 2014; Hendrix & Glaser, 2007; Theisen, 2008; Theisen *et al.*, 2013). One study that covered 41 African countries between 1981 and 1999 found that a 5% decline in GDP was associated with a 12% increase in violent conflict – and GDP declines are often associated with declines in agricultural production that may result from land degradation (Miguel *et al.*, 2004). Land degradation, by reducing agricultural productivity in rural landscapes, can indirectly increase conflict risk by increasing levels of poverty. One of the mechanisms whereby poverty increases conflict risk is that it has been shown to increase rates of recruitment to inter-group violence (Fjelde & von Uexkull, 2012). It is argued that individuals with less potential for economic advancement have less to lose by joining violence (i.e., have a lower opportunity cost). In addition, an increase in inequality (i.e., relative poverty) results in an increase in grievance against other groups and against institutions which further encourages recruitment.

Box 5.9 Somali refugees in Kenya

Drought, migration, and conflict are tightly intertwined in the case of Somali refugees in Kenya. Somalia, having been a "failed state" for most of the last two decades, has been witness to frequent and widespread conflict. Some work has shown that in Somalia, a country with high levels of poverty and livelihood insecurity, the intensity of the civil war increased in drought years when grazing productivity was lowest (Maystadt & Ecker, 2014). As a result of the continuing conflict, and likely encouraged further by the drought, large numbers of Somali refugees migrated to neighbouring Kenya.

In Kenya, many of the same resource constraints exist as in Somalia. Grazing land, water, and firewood are all in short supply in the areas of Somali refugee camps in Kenya. Competition over these resources has resulted in fractured relations between residents and refugees and allegations of violence (Kumssa & Jones, 2014; Martin, 2005). This case shows the potential both for drought and a reduction in the productivity of land-based livelihoods to increase conflict within a region (in Somalia), but also the potential for migration in response to change to cause conflict in the destination area (in Kenya) as new migrants compete with longer-term residents for already scarce resources.

5.6.1.3 Empirical evidence of link between land degradation and conflict

The general conclusion of the literature is that there is evidence for an empirical link between land and environmental degradation and increasing levels of conflict; however, the link is not as robust as the link between other factors, particularly poverty, and conflict (Buhaug *et al.*, 2014; Theisen *et al.*, 2013). It is for this reason that some authors have called for more empirical focus on the indirect pathways of causality between land degradation and conflict, particularly effects on livelihoods and on institutions (Bernauer *et al.*, 2012). It may in fact be the case that it is the role of land degradation in exacerbating poverty (see Section 5.2) that has the most significant (albeit indirect) effect on conflict risk. A recent report confirmed serious knowledge gaps to link land degradation and food security; however, it did conclude that the current literature on the influence of environmental factors on insecurity provides evidence for a very likely indirect relationship between land degradation and conflict (van Schaik *et al.*, 2014).

Some analyses at global scale have found a systematic link between soil degradation and levels of conflict (Hauge & Ellingsen, 1998; Melander & Sundberg, 2011; Raleigh & Urdal, 2007). However, this relationship is not deterministic, as is illustrated by a similar collection of studies that have found no association between conflict events and soil degradation or between conflict and other measures of environmental scarcity (de Soysa, 2002; Hendrix & Glaser, 2007; Urdal, 2005). The incongruity of results may partly result from data limitations and inconsistent definitions of conflict events that are used among different studies (van Schaik *et al.*, 2014). What is clear is that there is no direct, deterministic relationship between degradation and conflict that arises in all situations. However, there is sufficient evidence to consider degradation as a threat amplifier in certain contexts.

Results from some individual case studies are more convincing than are global analyses. For example, Wischnath and Buhaug (2014) use a state-level analysis in India to demonstrate an increase in violent conflict in years following a reduction in food production. They attribute this to a combination of grievance resulting from deprivation, as well as an increase in the ease of recruitment as farmer opportunity costs decline: they have less to lose from engaging in conflict (Fjelde *et al.*, 2012). This case demonstrates the dynamic (Figure 5.10) where land and environmental degradation can affect conflict via its impact on resource scarcity, poverty, and livelihood insecurity. Years of poor rainfall and livelihood stress have also been associated with increased conflict among East African pastoralists (Ayana *et al.*, 2016; Ide *et al.*, 2014; Maystadt *et al.*, 2015; Box 5.9), as well as between pastoralists and farmers in Mali (Watts, 2012). Data from across sub-Saharan Africa between 1990 and 2008 demonstrated extreme (90^{th} percentile) low rainfall anomalies being associated with up to 45% increases in rates of communal conflict (Fjelde & von Uexkull, 2012).

5.6.1.4 Environmental change and international conflict

There is little evidence to date that environmental change has led to a higher risk of violent conflict between countries. The most frequently-discussed scenario where the environment could lead to an international conflict is the situation of a cross-border dispute over water (Homer-Dixon, 1999; Wolf, 2007). In this case, the argument is compelling: freshwater is an essential resource with almost no possibility for substitution (the exception being desalination processes that are cost-prohibitive for the majority of the world's developing countries). Several of the world's more contentious geo-political relationships also sit in areas of water stress and/or areas where essential waterways are shared by two or more countries. For example, the water of the Indus River is shared by India and Pakistan, which has at times complicated the relationship between those two countries. Similarly, the Jordan River has been a target during times of conflict between Israel and Syria (Katz, 2011).

However, despite highly-publicized cases where water has been a factor in international disputes, it has rarely been, if ever, a central cause of these disputes. The evidence in fact shows that in cases where countries share a water supply in a water-stressed region, the result is generally an increase in between-country cooperation rather than an increase in conflict (Wolf, 2007). In the case of non-water resources, there are no case studies, as of yet, where land and environmental degradation itself has increased the risk of an international conflict. Although it is possible that context may change in the future, at present the strongest evidence for a link between environmental change and conflict lies in sub-national analyses (Hsiang *et al.*, 2013; Katz, 2011).

5.6.2 Migration

5.6.2.1 Land degradation increasing unwanted migration

Migration can be both a move towards greater economic opportunity, or a forced response to a negative change. In the case of land degradation, however, it is the latter category that is likely to see an increase: degradation generally tends to encourage out-migration from areas that are seeing reduced economic activity due to declines in soil fertility, drought, reduction in agricultural yields (Adger *et al.*, 2014; Black *et al.*, 2011). Land degradation in rural areas is often associated with higher rates of out-migration from those areas as individuals perceive limited economic opportunity in the given area. Some estimates have suggested that by 2050, the combined effect of land degradation and climate change will have resulted in 50 to 700 million people having migrated (Warner *et al.*, 2009). The IPCC's chapter on human security implications of climate change (Adger *et al.*, 2014) documents ten cases, the majority in Africa, where rates of out-migration increased due to environmental change, which resulted in decreased agricultural or grazing productivity. In two of these cases, the migration was international, but the other eight cases discussed within-country migration, either in search of new land in rural areas or else in search of wage labour opportunities. Migration to frontier areas where migrants bring new land into agricultural production may itself perpetuate and accelerate land degradation (Box 5.11)

One constraint on migration as a strategy for adaptation is poverty itself. In the most vulnerable communities, a lack of economic resources often means that migration is not an option. This situation is referred to as a "poverty trap" where households are faced with a declining quality of livelihoods resulting from the land and environmental degradation but have no ability to move in search of new opportunities (Adger *et al.*, 2014; Dasgupta *et al.*, 2005; Findley, 1994; Gray, 2011). In areas where poverty or social marginalization prevents individuals from being able to migrate, the result will tend to be a concentration of degradation and a tight coupling between the poverty of local population and the declining productive potential of land (Dasgupta *et al.*, 2005). In the absence of poverty or other constraints on migration, landholders can respond to declining productivity by moving to other locations, thus spreading the effects of land degradation more broadly and reducing the concentration of the burden of degradation on the poorest groups.

Box 5.10 Migration and land degradation in tropical forests

Deforestation in the tropical forest areas has at times been described as a process that is driven forward by poor smallholders as they respond to degradation and decreased productivity on established croplands. Smallholders clear forest in order to establish crops and claim tenure over land and then abandon the land after it becomes degraded before moving further into the frontier to start the process again (Fearnside, 2001; Maller *et al.*, 2006; Southgate *et al.*, 1990). This story, however, has been shown to be an over-simplification; migration has a more complex relationship with rates of deforestation and land degradation.

Figure 5 13 Frontier forest clearing and young coffee plantation, San Martin, Peru. Photo: courtesy of Timothy Holland.

Soil fertility declines have been shown to be an incentive for farmers to move further to the frontier and to clear additional forest (Arrow *et al.*, 1995; Carr & McCusker, 2009). However social factors may play a larger role in out-migration from agricultural lands in the Amazon than does land degradation per se. Households migrate for many reasons, among them the opportunity for wage labour in towns and cities (Bates & Rudel, 2004), seeking out social services that are not available in remote areas (Parry *et al.*, 2010), or because of household life cycles as adult children leave the area to seek economic opportunity elsewhere (Barbieri *et al.*, 2006; Caviglia-Harris *et al.*, 2012). Progressive declines in agricultural productivity resulting from soil infertility may less affect the rate of out-migration from frontier areas than it affects that type of migration that happens and the livelihood outcomes for individuals who do migrate (Caviglia-Harris *et al.*, 2012). When migration is forced by productivity declines, migrants are more likely to experience continued livelihood insecurity.

5.6.2.2 Climate change and land degradation acting in tandem

Climate change is projected to lead to large-scale migration, with much of the migration resulting from climate-induced land degradation (Adger *et al.*, 2014 and references therein). In some cases, migration in response to climate will be temporary and cyclical as drought-affected populations may temporarily move to urban areas for employment (e.g., Panda, 2010); however, long-term land impacts may lead to permanent population displacement. Climate change has resulted in and will continue to result in desertification, in coastal erosion, and in flooding – all of which have been documented as increasing outmigration from affected areas. In Bangladesh alone, it has been projected that there may be 3-10 million internal migrants (Hassani-Mahmooei & Parris, 2012). Large-scale outmigration is also expected from small island states as sea level rise progresses (Ballu *et al.*, 2011). As described above, desertification in the Sahel has resulted in pastoralists in northern Mali migrating southwards in the country, in some cases leading to conflict (Watts, 2012). Climate-induced reductions in agricultural productivity may also indirectly lead to outmigration as economic opportunities are reduced in the affected areas. In the Sahel, this has already been observed as climate change has increased the rate of land loss desertification and in turn the rate of outmigration (Scheffran *et al.*, 2012).

5.7 Energy

5.7.1 How does access to energy affect quality of life?

There is good evidence that access to energy improves human well-being and quality of life, but the type of energy (fossil-fuel, biofuel or other renewables), and mode of access (grid connection, local biomass collection, or small-scale gas or electricity grid) determines the type and intensity of land degradation, and the subsequent impacts on quality of life. With the passing of the Sustainable Development Goals in 2015 at the UN, global policy formally recognized the link between access to clean, reliable energy, economic growth, environmental sustainability and improved human well-being.

We give a short overview of the nature of the evidence for the links between energy and well-being, but we do not attempt a comprehensive assessment of these links. In the next section, we focus on those energy aspects that have a more direct link to land degradation issues – the use of traditional biomass and biofuel policies that trigger land-use change across national boundaries.

The extraction and use of fossil fuels were one of the key drivers that supported technological, cultural and social advances for humankind over the last 250 years. The development of centrally generated electricity allowed for more productive societies, through lighting enabling longer working hours, various appliances for specific tasks (i.e., refrigeration, water pumps, electrical motors), cleaner cooking and heating, industrial production and communications (Markandya & Wilkinson, 2007). However, the rate and intensity with which this transformation has taken place varies widely in time and space, while significant inequalities exist both in the use of and access to energy as well as in the subsequent impacts on quality of life. These inequalities are exacerbated by the use of different types of energy: centralized, large scale fossil fuel-based energy generation typically have intense local impacts in terms of the physical footprint of the installation, but also large regional and global impacts through pollution and greenhouse gas emissions. There are 2.4 billion people worldwide without access to such grid-tied electricity, and the traditional biomass-based energy sources that are used as an alternative lead to significant indoor pollution and associated health problems, negatively affecting well-being (Wilkinson *et al.*, 2007). In addition to these local effects, they are also exposed to the regional and global effects of fossil-fuel based energy for developed nations.

There is a correlation between per capita energy, and electricity consumption with a variety of measures of human well-being. The correlation shows a marked difference for industrial nations versus developing countries (Mazur, 2011), and there has been a number of studies that explored aspects of development trajectories, well-being and carbon emissions (Jorgenson, 2014; Steinberger *et al.*, 2012, 2013; Steinberger & Roberts, 2010). It is clear that the relationship between carbon and energy and human development is non-linear, so for relatively small increases in energy consumption and associated carbon emissions, the poorest countries show large benefits for a number of proxy measures of human development (Steinberger *et al.*, 2010). At a certain threshold, the benefits of energy access for households taper off, but there is still an underlying increase in the Human Development Index, probably due to increases in efficiency, resulting in a gradual decoupling of quality of life from the type of material support required. (Steinberger *et al.*, 2010). There remains some uncertainty whether the rate of increases in efficiencies will be sufficient to cope with increasing population pressure, and/or the transition of poor people into a middle-class lifestyle that is typically more energy intensive in developing countries (see Gertler *et al.*, 2016 for an example from Mexico). Subsequent work has shown that: higher life expectancies can be compatible with lower carbon emissions but not higher income (Steinberger *et al.*, 2012); future economic growth will probably improve human well-being throughout the world but at the cost of increasing carbon emissions (Jorgenson, 2014); and finally, an integrated assessment modelling study finds that the decreases in energy consumption required to meet mitigation targets without a structural economic change, will place sustainable development objectives at risk (Steckel *et al.*, 2013). Focusing on the African continent, economic model scenarios show that in the absence of climate policy, fossil-fuel energy demand grow over time to meet development needs (Calvin *et al.*, 2016) as the need for traditional biomass energy declines. This latter decline is associated with increasing affluence, but in the absence of external factors that increase per capita income, traditional biomass may remain an important energy source for longer. There is an opportunity for climate policies to be an important enabler of capacity growth in renewable energy sources (Calvin *et al.*, 2016).

5.7.2 How does energy extraction and generation affect land degradation and quality of life?

5.7.2.1 Traditional biomass energy sources

Biomass-based energy services include the provision of traditional biomass energy sources such as fuelwood, agricultural residues and animal dung (Karekezi *et al.*, 2004) and modern biomass energy sources such as processed wood briquettes and pellets, biogas and biofuels (bio-ethanol and bio-diesel) (Goldemberg & Coelho, 2004). Traditional biomass energy sources are usually available locally for domestic or subsistence use, at little or no cost and can be burnt directly for use, without need for specific technologies (Karekezi *et al.*, 2004). In comparison, modern biomass energy sources require processing before use and are often produced primarily for commercial ventures, for example biofuels are often produced from energy crops as agricultural enterprises (Gissi *et al.*, 2014) This distinction is reflected in the effect that each type of biomass-based energy has on land degradation and perceptions of well-being and quality of life.

Currently 2.7 billion people (38 % of global population) living mostly in the developing countries of Africa, Asia and Latin America depend on traditional biomass energy (mostly firewood and charcoal) to meet their basic household needs such as cooking and heating (International Energy Agency, 2016). The percentage value varies from region to region with the greatest dependence observed in Sub-Saharan Africa (80%) (IEA, 2016) and Developing Asia, particularly India (67%) (IEA, 2016).

At the micro-level, increasing human population pressures and other socio-economic factors, selective harvesting for the preferred tree/branch size and species will gradually result in a loss of biodiversity (Du Plessis, 1995) and changes in ecosystem structure and function (Luoga *et al.*, 2004; Shackleton, 1993), all of which has impacts on the delivery of other vital ecosystem services (Gissi *et al.*, 2014). Earlier work on global deforestation rates (Geist & Lambin, 2002) did not consider the gradual degradation of forests from such household level selective harvesting as a major driver of widespread deforestation. However, more recent work shows that woodland conversion from household level biomass extraction may lead to regional deforestation (Mwampamba, 2007). There is also a measureable health effect of harvesting fuelwood from degraded and recovering forested areas; the lower quality fuelwood from such areas increased respiratory ailments in a case study from Uganda (Jagger & Shively, 2014). Finally, at this micro-level, where household-level energy decisions determine landscape-level degradation, degraded landscapes have less fuelwood available, and then typically of a lower quality. The additional effort to remain energy secure in the face of these quality and quantity constraint, fall disproportionately on female household members in rural areas (Dovie *et al.*, 2004; Matsika *et al.*, 2013a, 2013b).

At the macro-level, there is a direct relationship between the extent of household use of biomass energy to meet domestic energy needs and the degree of impoverishment of a country. Generally, the poorer the nation, the higher the dependence of its populace on biomass energy to meet its primary domestic needs. This relationship extends to a feedback loop between poverty, (lack of) access to energy and environmental sustainability. The concerns about unsustainable woody biomass harvesting practices leading to land degradation and a negative feedback in the decline of human well-being are still valid today (Biggs *et al.*, 2004; Kaschula *et al.*, 2005; Twine *et al.*, 2003a). In Uganda, high rates of woodland loss are not only driven by local and urban charcoal needs, but also by livestock ranching, settlements expansion (see Coetzer *et al.*, 2010, for an example from a different area) and shifting crop cultivation (Kalema *et al.*, 2015). This is a positive feedback loop for the acceleration of deforestation-related degradation for biomass-based energy: less land is available for a resource that is extracted unsustainably, and no longer just for subsistence needs, but also to supply an almost unlimited demand in the nearby capital, Kampala (Kalema *et al.*, 2015). These examples highlight the direct link between biomass-based energy and deforestation. There is also some evidence for the changes in river flow due to deforestation, affecting hydro-power generation: increased sediment and vegetation debris stop hydro-power generation, and subsequently, urban power access (Wiyo *et al.*, 2015).

If access to alternative energy sources improve local energy security, there is a possibility that it may slow deforestation-related land degradation, but only if local economics, customs and culture support the energy transition. In addition, local intervention governed by regional policies should consider all local energy generation options (Heltberg *et al.*, 2000), and do so with all actors to form more effective energy institutions (Brew-Hammond, 2010). However, due to limited financial resources, most rural households are unable to make the transition to electricity as they cannot afford it or the appliances needed to fully utilize them (Williams & Shackleton, 2002). These societies remain dependent on the free forests and woodlands around them as a source of biomass energy (Biggs *et al.*, 2004; Twine *et al.*, 2003b) and this highlights the value of biomass energy provision as a safety net against the effects of widespread poverty (Shackleton & Shackleton, 2004).

5.7.2.2 Biofuel policies and indirect land-use change

Mitigation opportunities offered through biofuel production have sparked a lot of research on the trade-offs between potential for GHG emissions reductions from the use of biofuels, the impacts on food production, food prices and the other environmental impacts (Blanco-Canqui, 2010; Cotula *et al.*, 2008; Gasparatos *et al.*, 2011, 2012; Popp *et al.*, 2014). The direct impacts of cultivating biofuels on water, soil, biodiversity and associated ecosystem services, are very similar to the direct impacts of large agricultural fields, and is addressed in Section 5.3 and in Chapter 3.

Underlying the interaction between agricultural commodity prices and biofuels, is the concept of indirect land-use effects (Meyfroidt *et al.*, 2013; Persson, 2015; Villoria & Hertel, 2011). Changes in market prices for biofuel feedstock mediate land-use change, competing with food crops. The net result is that biofuel feedstock prices that respond to mitigation opportunities, becomes an exogenous driver of land use for food crops. This is more than just a displacement of land use spatially, as has been shown for collection site switching in response to depletion of communal fuelwood resources (Sonter *et al.*, 2017). The trade in, and price of, two different commodities (food and biofuel) are therefore connected in an important, but poorly understood manner.

Earlier work (Searchinger *et al.*, 2008) found that farmers expanding corn-based ethanol biofuel production into forest and rangeland in response to higher biofuel prices, resulted in almost a doubling of GHG emissions. Subsequent work that used the same model, focused on the effects of model assumptions and showed that this estimate may be too high by as much as two-thirds (Dumortier *et al.*, 2011). A more complicated indirect land-use effects example shows the effect of corn-based ethanol resulting in an increase of 14-43% of USA corn prices for the period 2000-2008 (Persson, 2015). In both cases, model structure and parameter uncertainties untested by sensitivity analyses (Meyfroidt *et al.*, 2013), casts doubt on the generalisability of these findings. A systematic review of 121 studies on the effect of biofuel demand on agricultural commodity prices show that there is unequivocal evidence that increased demand for biofuels lead to higher agricultural commodity prices, but there remains large uncertainty on the exact magnitude of the effect due to data limitations and modelling assumptions (Ahlgren & Di Lucia, 2014; Persson, 2015). In addition, even if the correct policy contexts are captured in these models, the extent and effectiveness of policy enforcement remains a significant source of variation. The EU biofuel demand contributed to one third of the price increase in vegetable oils in the EU between 2000-2008 (Persson, 2015). Biofuel energy penetration in conventional energy markets remain low, but given the opportunities for smaller, distributed biofuel plants in developing countries and the associated benefits to human well-being, there is an urgency to improve indirect land-use effects in economic models (Persson, 2015).

Indirect land-use effects may offer land restoration opportunities, where abandoned agricultural land is restored for GHG mitigation. Evans *et al.* (2015) shows that forest recovery provides better GHG sequestration than low yield biofuels such as oil palm and corn, but for high yield fuels such as sugarcane, biofuels may provide better GHG mitigation than forest succession. Earlier work on the same topic had less comprehensive answers on the relative merits of different land-use options, but did emphasize that the only long-term solutions are carbon-free fuel technologies (Righelato & Spracklen, 2007).

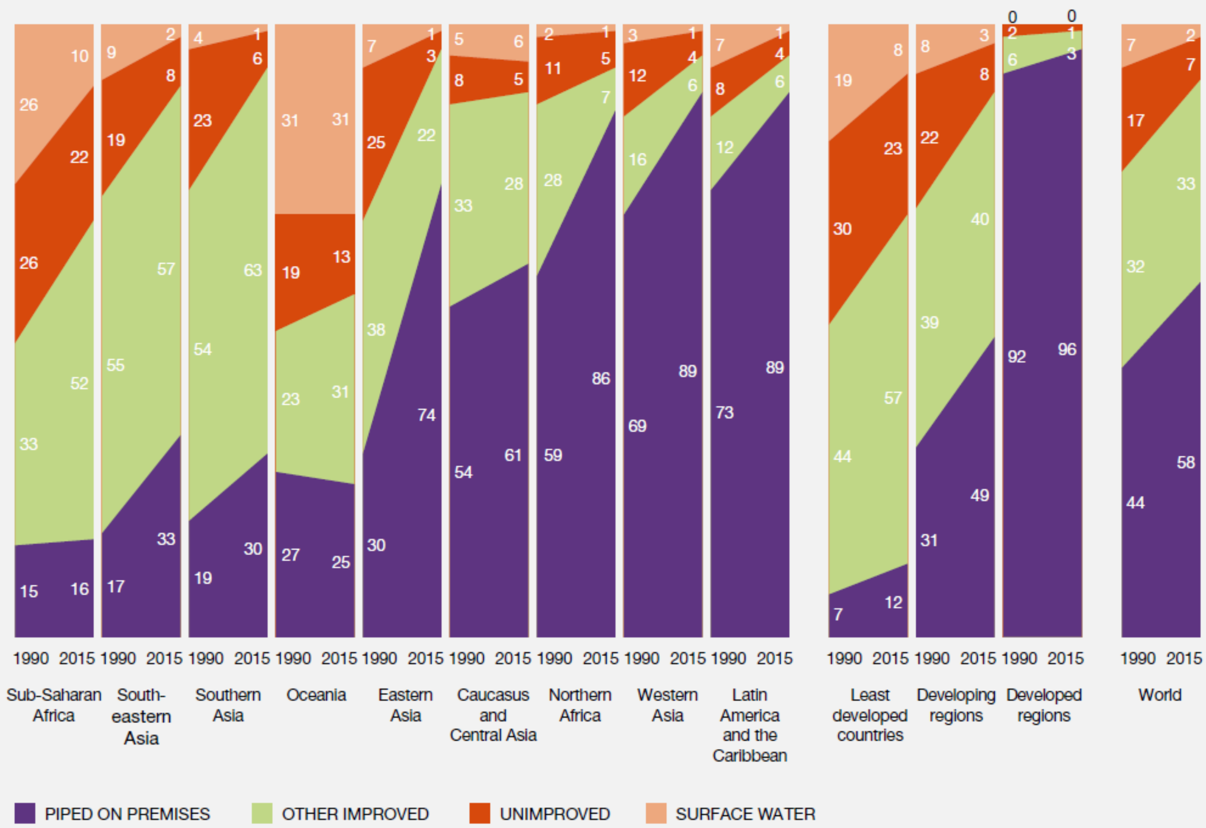
In conclusion, there is high agreement that biofuels increase agricultural commodity prices, but there is limited robust evidence due to the complex nature of the models, policy enforcement uncertainty, the lack of data on supply and demand elasticity in developing countries, and the lack of data on land markers and their drivers (Persson, 2015).

5.8 Water security

Water is directly linked to human health, food security, energy, and disaster risk due to extreme events such as floods, and droughts. The benefits related to an adequate supply of high quality water provided by freshwater ecosystems (e.g., rivers, streams, lakes and wetlands), are among the most central to the survival and well-being of human populations, with high economic and social values (Gleick, 2014; Postel & Thompson, 2005). As a result of the tight linkages between land management and the water cycle, practices that degrade lands reduce water supplies, both in terms of quantity and quality. In essence, land management is water management (Bossio *et al.*, 2010).

Water security describes the quantity and quality of water needed to sustain health, livelihoods, economic development, and ecosystems, and protect against water-borne pollution and water-related disasters (Grey & Sadoff, 2007; UN-WATER, 2013). Water and sanitation are a cornerstone of sustainable development, providing the societal benefits of adequate freshwater supplies for drinking water, waste disposal, irrigation, food production and supplies (crops, fisheries), cooling water for energy generation, and cultural and spiritual services (Gain *et al.*, 2016; Gleick, 1998, 2014). Development places increasing demands on water use, quality and availability, impacting its use and governance, leaving less freshwater available to meet the environmental flows needed to support the biodiversity and services that ecosystems provide. The UN Sustainable Development Goal 6 aim to ensure the availability and sustainable management of clean water and sanitation by 2030 (UN, 2016). Considerable progress has been made in providing access to drinking water, rising from 76% of the global population with access to an improved drinking water source in 1990, to 91% in 2015 (Figure 5.14). However, geographically widespread inequalities remain; 783 million people are using unimproved sources (UN-WATER, 2013; WHO & UNICEF, 2014) and 1.8 billion people are exposed to drinking water contaminated with faeces. The use of improved sanitation facilities increased from 59% globally in 2000 to 68% in 2015, leaving 2.4 billion people without improved sanitation (UN, 2016).

Figure 5 14 Percent of population with access to improved water sources, 1990-2015. Source: UNICEF / WHO (2015).



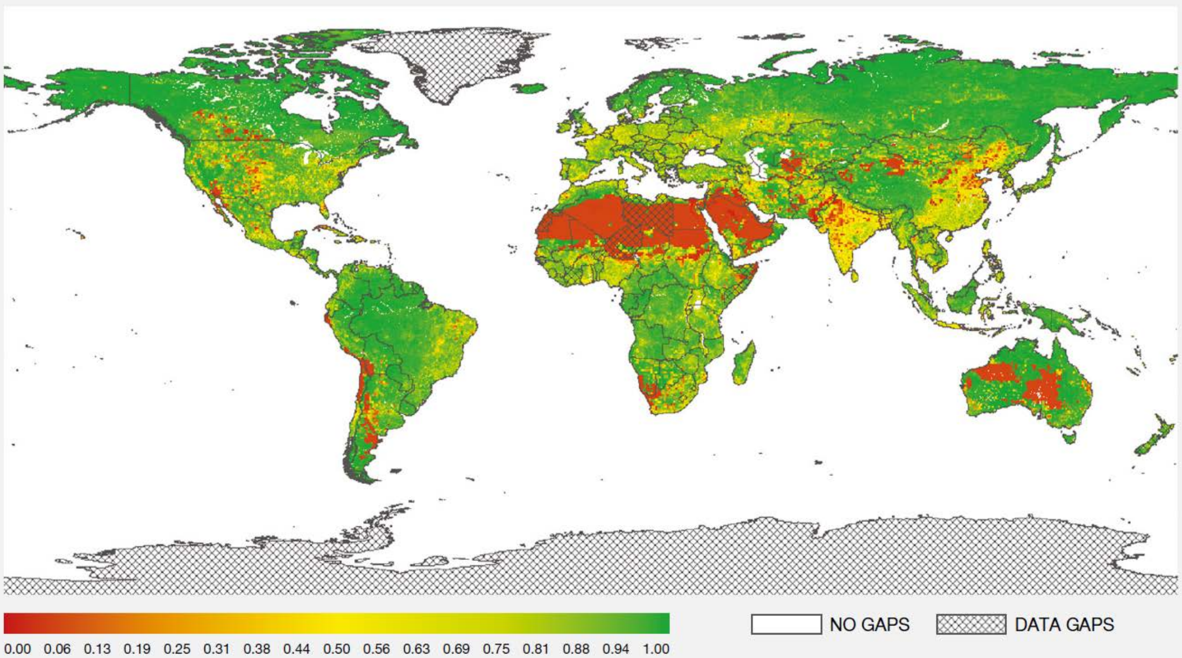
5.8.1 Status and trends in water security

Water security includes measures of water quality and governance, or the social and economic factors related to water planning, management, and the delivery of water services (Vörösmarty *et al.*, 2015). By contrast, water scarcity is a function of supply that is either demand driven (the ratio of demand to availability) or population driven (per capita availability of renewable freshwater). Scarcity is defined as less than 1000 m^3 per person annually (Gain *et al.*, 2016). Alterations to the water cycle are related to the drivers that limit water availability and use, including the loss of natural vegetation cover and/or vegetative biodiversity. This leads to erosion and soil loss, a reduction in natural filtration processes, and the loss of soil organic matter (that can prevent soil crusting and compaction), reducing infiltration and soil water storage capacity (Bossio *et al.* 2010). Expanding agricultural and urban land uses, climate change, population growth, and salinization and chemical contamination are drivers that impact the availability of adequate clean water; these are exacerbated by economic disparity and poor governance (Gain *et al.*, 2016; Vörösmarty *et al.*, 2015).

Global water security index scores, based on indicators derived from Sustainable Development Goal 6, including measures of water availability, accessibility to services, safety, quality, and management, indicate low water security index scores for large regions of Africa, South Asia, and the Middle East (Figure 5.15). In some areas of water scarcity (e.g., portions of Australia, the southwestern USA and Mexico, and Southern Europe), water security index scores are higher than predicted due to the mitigating effects of active water management and use of water technology. Engineering solutions to replace the ecosystem services that maintain water supplies are effective but expensive, and often rely on the input of fossil fuels (Cech, 2010; Palmer, 2010). Reliance on technology to overcome water issues does not address the underlying stressors, but may produce both false security in industrialized countries and chronic water issues (water insecurity) in developing regions. This calls for prudent water management policies with a focus on effectively valuing water and boosting efficiency to achieve the outcomes of universal access to safe drinking water, adequate sanitation and hygiene, improved water quality, enhanced adaption to climate change and improved ecosystem protection.

Figure 5 15 **An aggregated water security index based on measures of water availability, accessibility, safety and quantity, and management.**

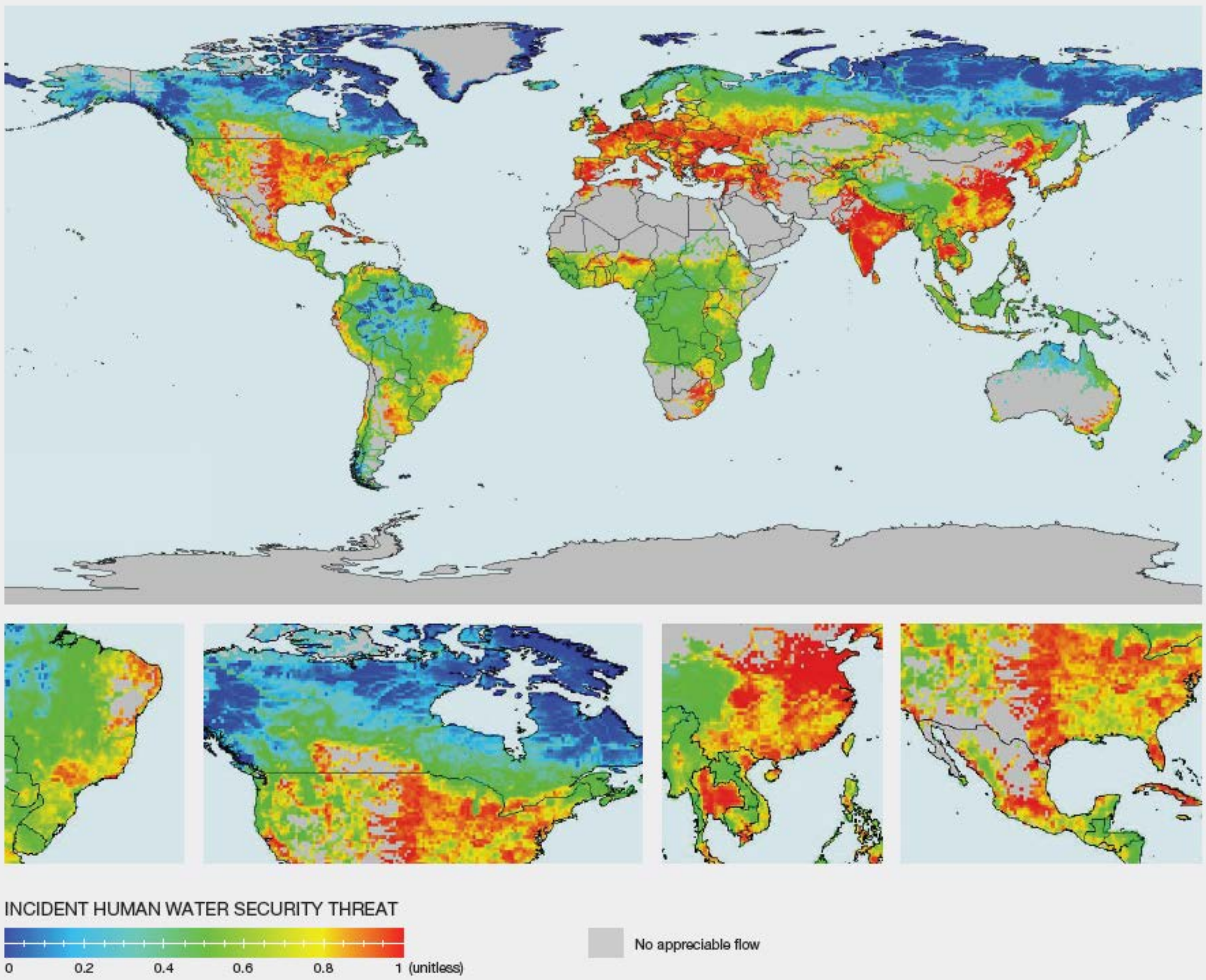
Scores are between 0-1, representing a continuum of low to high security. Shaded areas are data gaps. Source: Gain *et al.* (2016).



Flowing waters represent the best source of renewable freshwater, but ecosystem degradation, human activities, and growing populations have caused systematic degradation of water with 80% of the world's population living where human impacts pose an incident threat to human water security and biodiversity (Figure 5.16). Two-thirds of the global population face severe water scarcity at least 1 month per year, half of whom live in Asia (Mekonnen & Hoekstra, 2016). The impacts to human health are enormous, with an estimated 1.6 million deaths per year due to a lack of safe drinking water and poor sanitation and hygiene (many due to diarrhea) (Tarrass & Benjelloun, 2012). Even in remote areas of the Amazon basin, threats to water quality exist, primarily due to trans-boundary deposition of atmospheric pollution (Vörösmarty *et al.*, 2010). Globally, demands for water are increasing: withdrawals are predicted to increase by 50% by 2025 in developing, and 18% in developed countries (UN-WATER, 2013). This, coupled with increasing demands for food, energy, and other materials, are expected to increase demands to unprecedented levels (UN-WATER, 2013).

Figure 5 16 Global distribution showing incident threat to water security based primarily on bio-physical indicators.

Regional maps show areas of water security threat are that are coincident with intensive agriculture and high population density. Source: Reprinted by permission from Macmillan Publishers Ltd: [global threats to human water security and river biodiversity] (C. J. Vörösmarty, P. B. McIntyre, M. O. Gessner, D. Dudgeon, A. Prusevich, P. Green), copyright (2010).

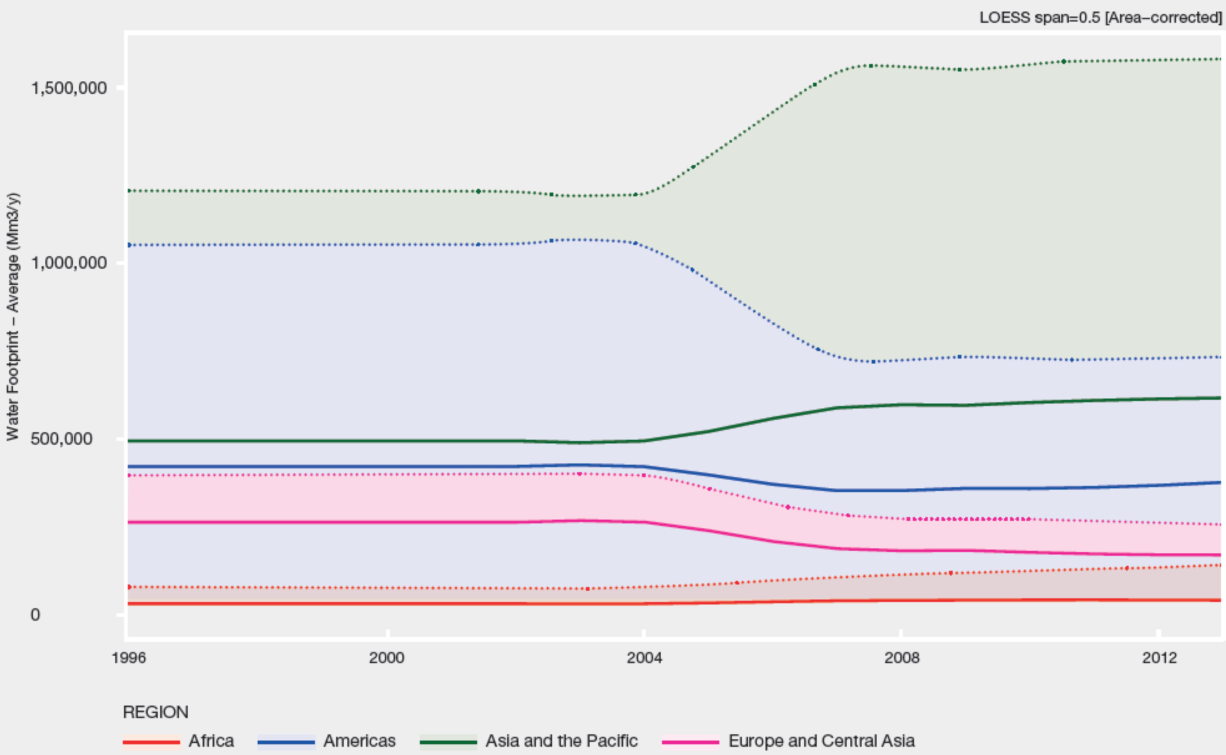


5.8.1.1 Water footprint

The water footprint is a measure of the human appropriation of freshwater (water volume consumed), a measure comprised of blue water (consumption of surface and ground water), green water (rainwater consumed in crop production) and grey water (freshwater required to assimilate pollutants using existing water quality standards) (Figure 5.17). Agricultural production contributes an estimated 92% to the total footprint, a substantial portion (~20%) of which supports production for export to other countries, or virtual water flow (the water flow embodied in food and other commodities). This allows water poor regions to support larger human populations by importing water intensive crops, preserving local water resources. Regions in the Americas tend to be major water exporters, in particular the USA, Argentina, and Canada (Hoekstra & Mekonnen 2012; Mekonnen *et al.*, 2015) (Figure 5.18), with consumption of cereals contributing the most to the water footprint of the average consumer (Hoekstra & Mekonnen, 2012).

Figure 5 17 Trends in water footprint by region, 1996-2013.

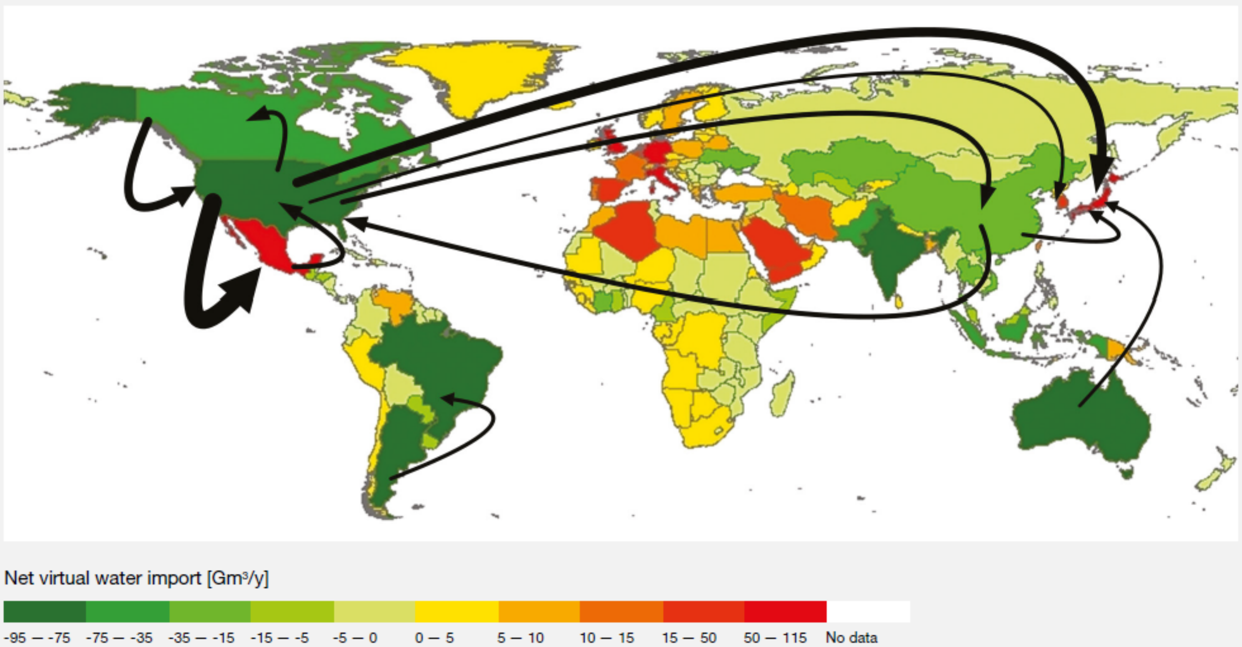
The figure prepared by Task Group on Indicators and Knowledge and Data Technical Support Unit. Indicator data source: Water Footprint Network.



While water withdrawals vary greatly by region, demands on water supplies in all sectors is increasing, with agriculture accounting for nearly 70% of global water withdrawals, and over 85% of consumptive water use (Doll & Siebert, 2002; Foley *et al.*, 2005; Gleick, 2014). Ultimately, water security is a prerequisite for food security, and the water requirements for increasing production to feed an estimated 9 billion people by 2050 will increasingly stress supplies. Irrigation has increased crop yields: irrigated cereal yields, for example, are 60% higher than non-irrigated yields (Rosegrant *et al.*, 2009). Changing dietary preferences also have an impact; the growing preferences for meat in many countries means increased water demands (one kg of meat requires 4,000-15,000 litres to produce, compared to 1,000-2,000 litres per kg grain) (Renault & Wallender, 2000).

Figure 5 18 Virtual water balance per country showing gross virtual water flows due to trade in agriculture and industrial products between 1996-2005.

Countries shown in green are net virtual water exporters; those in yellow and red import virtual water. The biggest net exporters are the USA, Canada Brazil, and Argentina (only flows > 15 billion m³ per year are used in the assessment; the wider the arrow the larger the flow). Source: Hoekstra & Mekonnen (2012).



5.8.2 Impacts of land degradation on freshwater ecosystem services

5.8.2.1 Land-use changes

Water resources (groundwater, wetland, lakes and reservoirs, streams and rivers) are embedded in watersheds and effective water management depends on effective management of land (Jordan & Benson, 2015; Bossio *et al.*, 2010; Wetzel, 2001). Land-cover change and degradation, especially deforestation and wetland drainage, have a direct impact on the availability of freshwater supplies. Natural forests, wetlands, floodplains and riparian zones play a key role in maintaining supplies, providing an estimated 75% of the worlds freshwater (Castello & Macedo, 2016; FAO, 2016; Meybeck, 2003; Singh & Mishra, 2014; Vörösmarty *et al.*, 2005). Globally, cropland and pastureland have increased by 460% and 560% respectively, over the past 300 years (Goldewijk, 2001). Conversion of wetlands to agricultural and urban land uses has been extensive, with losses ranging between 54-57% (but regionally as high as 87-90%, for e.g., in portions of Midwestern USA and Europe) since 1700, with most of those losses in the past 100 years (Carpenter *et al.*, 2011; Davidson, 2014; Mitsch & Gosselink, 2015). A consequence is degraded water quality, flood damage, diminished biodiversity including food and fisheries, and radically altered regional hydrology (Hey *et al.*, 2005; Prince, 1997). Agricultural drainage eliminates wetlands and riparian zones, reducing regional surface water stores and diminishing water yields, with impacts to flows need to sustain ecosystems and large, negative effects on downstream water quality (Schilling *et al.*, 2008; Wang *et al.*, 2010). Because hydrology determines the location, and structural and functional properties of wetlands, agricultural expansion has caused enormous losses of wetland area, and diminished the ecosystem services provided by those that remain, such as reduction in flood peaks and carbon sequestration and storage (Zedler, 2003).

5.8.2.2 Water quality

There is clear evidence that the land degradation and the resulting decline in freshwater quality limits human development and threatens freshwater biodiversity and ecosystem services (Gleick, 2014; Scholes & Scholes, 2013; Vörösmarty *et al.*, 2010; Zedler, 2003). Pollutants and poor water quality reduce the utility of drinking water, reduce the provision of fish and other food supplies, and when water becomes salinized, limits irrigation. Globally, hydrological modifications (dam building, river and stream channelization, drainage creation of impervious services, and the conversion of wetlands) have caused some of the largest declines in the water quality, biodiversity, and the contributions to human quality of life that freshwater systems provide (Palmer, 2010). Nutrient and sediment runoff associated with agricultural use leads to eutrophication of inland and coastal waters, harmful algal blooms and coastal hypoxic or "dead zones," which impacts to fisheries, recreational services, and so regional economies (Bennett *et al.*, 2001; Gleick, 2003). This can limit the use of water for human consumption due to algal blooms, including blooms of cyanobacteria that release microcystin, a potent liver toxin (Brooks *et al.*, 2016; Paerl *et al.*, 2016). Extreme events result in beach closings and pose threats to potable water supplies, such as the drinking water ban for half a million people in Toledo, Ohio (USA) during the summer of 2011, with similar outbreaks in Lake Taihu in China (Michalak *et al.*, 2013; Paerl & Huisman, 2008). Intensive agriculture can also lead to high concentrations of compounds such as nitrate, the cause of methemoglobinemia in infants, and some pesticides that are considered estrogen mimics that may cause developmental issues in humans and other species.

5.8.2.3 Industry mining

Rapid industrial growth has increased mining operations in some regions including the South American countries of Chile, Bolivia, and Peru. Mercury used to extract and consolidate gold mined from river systems is causing widespread mercury pollution, particularly in Peru where artisanal mining in the headwaters of the Amazon basin leads to severe mercury pollution of surface waters (Buytaert & Breuer, 2013). Mining leads to a complex set of processes that degrade land, involving deforestation, river bank destruction, water pollution, and human health effects, all driven by increasing gold prices. A link has also been found between the occurrence of malaria and past exposure to mercury in gold mining, either through water or in fish consumption (Buytaert & Breuer, 2013; Crompton *et al.*, 2002).

5.8.2.4 Urban environments

The loss of ecosystem services through urbanization is extensive. Cities occupy about 2% of the Earth's surface, yet support approximately 55% of the earth's population (~4 billion), use 75% of the world's natural resources, and produce 70% of the total waste produced globally (ICLEI, 2011). Urbanization intensifies demands for water and sanitation, with the result that there are currently 150 million people living where water supplies do not meet demands. Climate change is expected to increase water shortages for an estimated additional 100 million urban dwellers by 2050 (Mcdonald *et al.*, 2011). Diversion of water from agriculture to urban areas can decrease agricultural productivity and the need to bring water from distant sources can create conflicts between users (Reisner, 1993). High population densities, large areas with impervious surfaces, and soil degradation result in the loss of natural water purification processes and reduced water quality. Problems with water infrastructure and waste disposal within cities, particularly in slums, can also cause problems of contamination. This has caused outbreaks of diseases such as cholera or typhoid in some large cities in the tropics (Eisenstein, 2014). The recent emergence of the Zika virus and its outbreak in Brazil is linked to its introduction to urban centres. Work is on-going in Brazil to improve drinking water supplies to protect the health of the urban poor (Eisenstein, 2014).

Degraded and polluted water supplies are more likely to contain pathogens that can lead to diseases such as cholera, typhoid, and dysentery. While contamination by faecal and organic pollution has decreased in many regions due to infrastructural improvements, it has increased in severity in many developing countries, particularly in areas where urbanization is rapid. In industrialized countries health issues related to pharmaceuticals and personal care products in water supplies are seen as an emerging threat to human health (Evgenidou *et al.*, 2015).

There is strong evidence that investments in water security are directly correlated with jobs and economic growth. It is estimated that the jobs of half of the global workforce rely on eight water-dependent industries: agriculture, forestry, fisheries, energy, manufacturing, recycling, building and transportation (UN-WATER, 2016). Freshwater wetlands, rivers, and lakes have long been essential sources of food production such as fish, rice and waterfowl. Human water demands associated with land-cover change, water withdrawals, diversions, drainage, and increasingly climate change, contribute to the decline in the ecosystem goods and services, limiting food security and overall economic development (Horwitz *et al.*, 2012; Ramsar, 2015).

5.8.3 Impacts of restoration of degraded land on water security

The restoration of degraded lands along with improvements in the efficiency of water use (e.g., for agriculture) can reverse many of the trends associated with impacts to freshwaters and the services they provide (Bossio *et al.*, 2010; Postel, 2000). Restoration approaches vary with the stressors and types of degradation that freshwaters have sustained. In agricultural lands, wetlands and riparian zones can be strategically replaced in the landscape (Mitsch *et al.*, 2001) and the soils of many agricultural lands can be "re-carbonized," by increasing organic matter content, which allows water to soak in and be held in the soil (Gnacadja, 2013). Reforestation of watersheds can restore important provisioning services (e.g., clean water and food supplies) by reducing surface water runoff, and decreasing soil erosion and sedimentation (Jordan & Benson, 2015; Lele, 2009). While there is strong evidence for the effects of deforestation on waters, much of the understanding about anticipated improvements that might result from restoration are inferred by the cost of land degradation. For example, maintaining the cover of temperate forests in South America provided water with an economic value of $5.8 to 15.4 per household, depending on the season. In Mumbai, India it is estimated that for every one percent decrease in forest cover, turbidity increases by 8.4%, increasing the costs of drinking water treatment by 1.6% (Singh *et al.*, 2014). Reforestation doesn't necessarily increase water supply, but regulates seasonal flows and minimizes soil erosion, with the ancillary services of carbon sequestration and timber production (Simonit & Perrings, 2013). Payment for ecosystems services can incentivize landowners to undertake reforestation and promote water security (Lamb *et al.*, 2005; Sengalama & Quillérou, 2016).

Wetlands serve an important role in nutrient management and flow regulation at the landscape scale, and their restoration can mitigate downstream flooding and improve water quality by capturing and processing diffuse runoff (Fennessy & Craft, 2011). Prioritizing wetland restoration in agricultural watersheds to reduce the runoff of agricultural chemicals can benefit downstream waters (Comin *et al.*, 2014). For instance, restoring wetlands to cover 10 percent of the Mississippi River Watershed could reduce nitrogen loads to the Gulf of Mexico by an estimated 40 percent, improving hypoxia in the Gulf and protecting fisheries (Mitsch *et al.*, 2001). At the site-scale riparian zones and floodplains, even narrow strips of land (e.g., 10 m) adjacent to streams, ditches, or rivers can remove up to 90% of nitrogen and 50% of phosphorus (Lowrance, 1998; Zedler & Kercher, 2005). Forest cover also regulates stream temperatures and provides much of the leaf material used by instream biota, protecting fishery sustainability. Overall, success in restoring the structure and functions of lost wetlands is mixed. A global meta-analysis of 621 sites indicated that, even 100 years post-restoration, biodiversity and biogeochemical functions (related to soil carbon storage) were 26% and 23% lower than in unimpacted natural wetlands (Moreno-Mateos *et al.*, 2012); an analysis by Rey Benyas *et al.* (2009) showed similar results.

5.8.3.1 Restoration in urban environments

Urban environments provide novel ecological conditions, yet provide a wide array of ecosystem services (Pickett *et al.*, 2008). Although urban areas are not candidates for restoration to some historical, pre-disturbance reference condition, multiple strategies have developed to increase quality of life. Green infrastructure forms a network of protected land and structures to create a high-quality living environment, which includes “blue space” in the form of ponds, river banks, wetlands and coasts (Niemelä *et al.*, 2010; WHO Europe, 2016). This “green/blue” space is made up of natural and human modified structures such as green walls and roofs, eco-bridges and corridors, and constructed wetlands, or features such as porous pavements that increase water infiltration and decrease stormwater runoff. Wetlands are increasingly preserved and restored in urban and periurban areas to mitigate flood and climate risks, support food production and provide for recreation (McInnes, 2013). China has created a series of wetland parks through the restoration of degraded rivers and ditches to capture storm runoff and remove pollutants, support biodiversity and provide a place to experience nature (Li *et al.*, 2009), although this can lead to over use and degradation of the resource. “Sponge cities” in the USA and China use technologies to drain, store, and recycle storm water and, while expensive to install, will aid in reducing flood losses that are, for example, predicted to rise in Philadelphia alone from $89 million to $1 billion in 2050 (Gains, 2016). The creation of green infrastructure has important direct effects to human well-being, although it is often in short supply (Bertram & Rehdanz, 2015). Additional benefits in the form of improved mental health, reduced cardiovascular morbidity and mortality, obesity and risk of type 2 diabetes, and improved pregnancy outcomes have been attributed to the positive impacts of psychological relaxation and stress alleviation, increased physical activity, reduced exposure to air pollutants, noise and excess heat (WHO Europe, 2016).

5.9 Spiritual and cultural values

In the preceding sections, we have discussed in detail how land degradation and restoration contributes to human quality of life in many universal and quantifiable ways. In this section, we turn to the non-material aspects of human well-being that are impacted by land degradation and restoration. These nature-linked aspects of well-being are less tangible; however, they enable individuals to feel more fulfilled and allow cultures to thrive with a connection to place. They originate and flow from individuals’ spiritual, social, and/or philosophical beliefs about humanity’s relationship to nature, as well as from cultural traditions as they have developed in reference to particular aspects of nature (Chan *et al.*, 2012; Laband, 2013; Russell *et al.*, 2013; Winthrop, 2014).

To guide our assessment of these non-material impacts, we use the concept of "sense of place" as a unifying theme. This concept refers to the emotional bond between a person and location that has been shown to form the basis for cultural connections to land and place, particularly in traditional societies (Windsor & McVey, 2005). Below, we begin by looking at the connection between nature and individuals before turning to a broader assessment of the importance of nature in creating cultural identity, especially for traditional societies.

Non-material connections to nature help to shape, define, and give meaning to human existence. To assess them requires acknowledging and evaluating ways in which ecosystem services contribute to a good quality of life that may not be numerically measured. Thus, in our discussion below we strive to take into account the different ways people conceive of their relationship with nature, while also discussing the challenges that come with attempting to quantify the non-material contributions of nature to humans.

What emerges from our assessment below is that: (i) ongoing land degradation is having as significant or more significant of an impact on cultural diversity as ongoing anthropogenic climate change (Adger *et al.*, 2013; Crate, 2011); and (ii) many of the most pronounced cultural changes are occurring in the most ecologically diverse areas on Earth as there is a strong co-occurrence between linguistic diversity and biological diversity (Gorenflo *et al.*, 2012).

5.9.1 Sense of place and the individual

A rich, diverse and substantive literature exists supporting the importance of place and sense of place in maintaining human well-being (see Windsor *et al.*, 2005 and references therein). The concept of place has a long history and may be simply defined as the emotional tie between an individual and location. In contrast, defining the concept of sense of place is more difficult and has been referred over the years by various fields as "place attachment", "settlement identity", "homelands", or "landscape of home", for example (Windsor & McVey, 2005). Regardless of the definition, it is clear from the literature that sense of place provides a "sense of security to individuals and groups" as well as "sense of control over their own fate" (Steele, 1981 via Windsor & McVey, 2005).

To a large extent, the focus of much of the work looking at how land degradation effects can create loss of place has focused on urbanization in areas inhabited by people of European descent (Hewitt, 1983; Kunstler, 1994; Miller, 2005; Read, 1998; Relph, 2008; Rowley & Wood, 1985). In addition, there has been some work looking at how restoration of nature areas nearby urban environments can help residents reconnect with nature (Miller, 2006).

With regards to loss of sense of place due to land degradation, a smaller yet still significant literature exists documenting the pronounced loss of place that land degradation drives (Windsor & McVey, 2005). Well-documented examples of the land degradation-driven loss of sense of space include impacts of post-World War II resettlement in Algeria (Sutton, 1977), mercury pollution induced resettlement of an Ojibwa community in Ontario, Canada (Shkilnyk, 1985), and impact of a private hydroelectric dam construction on the Cheslatta T'En Canadian First Nations people (Windsor & McVey, 2005). Overall, the literature is clear that land degradation has a long-lasting, pronounced, and substantial negative impact on the well-being of individuals living in these landscapes through loss of sense of place.

In addition to the emotional ties to nature that derive from a sense of place, many individuals have reported perceptions of belonging, spiritual fulfilment, or a sense of something greater when experiencing nature (Calvet-Mir *et al.*, 2012; Vorkinn & Riese, 2001). There is strong evidence that a spiritual element of recreational experience with nature exists independent of an individual's particular belief system, with individuals expressing greater concern for the wellness of natural places after they partake in nature-based activities (Heintzman, 2003, 2012). This relationship is likely to exist for many cultures but has yet to be well-documented outside of a limited number of relatively wealthier countries. Importantly, degradation of nature has been shown to lead to emotional or spiritual harm to individuals with their feeling of attachment to nature places disrupted by degradation of natural ecosystems (Willox *et al.*, 2012).

Box 5.11 The relationship between religion and environmental stewardship

The spiritual aspect of nature connection that is common in many cultures suggests that concern for the environmental is an integral part of religious beliefs. Indeed, during the 21st session of the Conference of the Parties (COP21) of the United Nations Framework Convention on Climate Change (UNFCCC) in Paris in 2015, representatives of several world religious groups issued statements in support of global action on climate change. An interfaith group composed of leaders of more than 100 congregations from different religions issued a statement that stated the following:

"*Our religious convictions and cosmological narratives tell us that this earth and the whole universe are gifts that we have received from the spring of life, from God. It is our obligation to respect, protect and sustain these gifts by all means"* (Statement of Faith and Spiritual Leaders, 2015).

These broad statements on the relationship between the environment and world religions is further supported, albeit in a complex manner, by survey results from the World Values Survey (WVS, 2015), which began in 1981 and covers more than 90% of the world population in almost 100 countries. The survey polls people across the globe on their beliefs, values, and motivations on a diversity of topics, including religion and the environment. Results of the survey show that more religious people are more likely to say that looking after the environment is something important to them. However, when the question was phrased as a trade-off between the economy and the environment, less religious people put a slightly higher priority on environmental protection as compared to economic growth.

5.9.1.1 Existence value

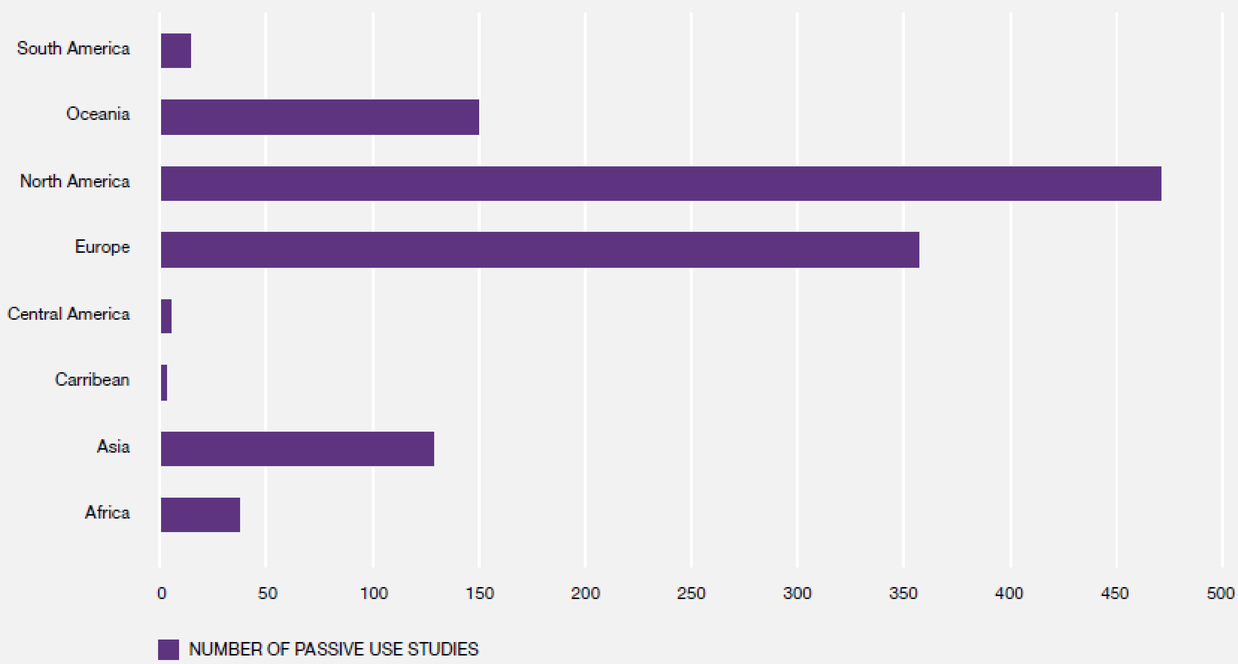
The concept of sense of place and spiritual connections to nature may be part of what leads people to identify an existence value of nature. Diaz *et al.* (2015) defined existence value to be "the satisfaction obtained from knowing that nature endures". It has been suggested that there is a human need for self-transcendence, which means to perceive personal identity as including objects or causes that are beyond the person (Davidson, 2013). In the case of nature, this desire to self-transcend may partly explain the sense people report in nature of feeling connected to something greater than themselves (Mayer *et al.*, 2009). That same desire to self-transcend may be at the root of people placing an existence value on natural areas even without a physical connection to those areas: it may be that when people include a sense of these places as part of their own personal identity, they feel personally affected by the fate of those natural areas (Davidson, 2013).

Many of the contributions of nature to human well-being discussed in this section are not easily quantified; however, there exists a large and ever-expanding literature on economic valuation of the existence value of nature. Two good resources are the Environmental Valuation Reference Inventory (EVRI, 2016) which consist of over 4000 studies on the stated valuation of nature and the Ecosystem Service Valuation Database (ESVD 2016; de Groot *et al.*, 2012) which consists of over 1300 carefully screened studies detailing the economic valuations of nature.

We assessed the studies in the Environmental Valuation Reference Inventory for their global coverage (Figure 5.19) as well as the types of driver of loss of existence value that the studies identified (Figure 5.20). The great majority of work on economic valuation of existence values of nature has been done in North America, Europe, and Oceania, although there is also a relatively large number of studies from Asia. Latin America and Africa, however, are greatly underrepresented. Across the studies, the biggest drivers of loss of existence value were habitat loss/degradation, resources extraction, and environmental pollution (Figure 5.20).

Figure 5 19 **Global coverage of passive use studies in EVRI database.**

The Environmental Valuation Reference Inventory (EVRI) consist of over 4000 studies on the stated valuation of nature. For studies looking at existence values of nature most of the work to date has been conducted in North America, Europe, and Oceania. Studies from Latin America and Africa are greatly underrepresented. Source: EVRI (2016).

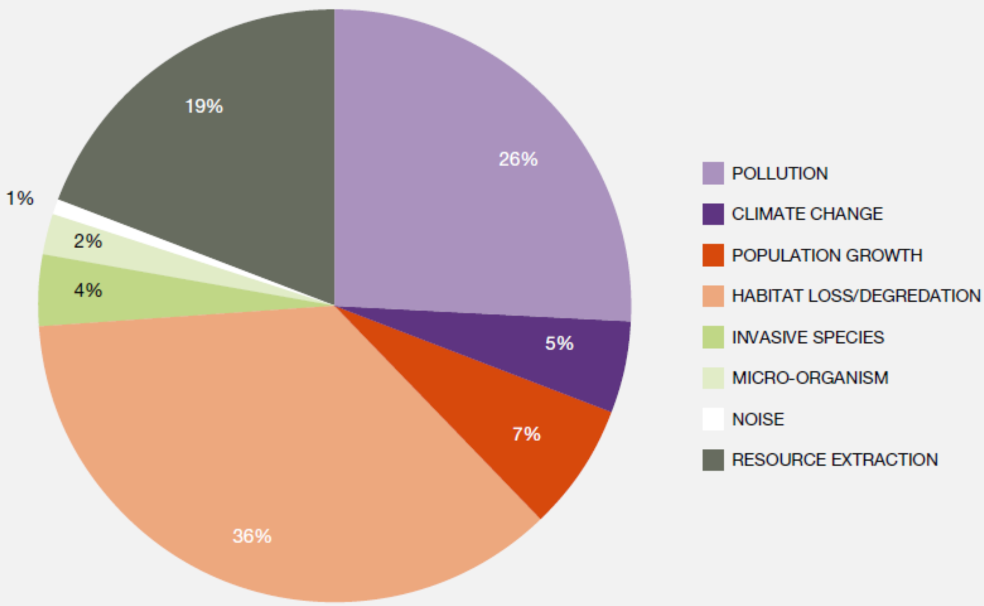


For the scale of reported economic valuation, we focus on the Ecosystem Service Valuation Database data, as the studies included in this database have been carefully organized, standardized and contextualized. The database shows the following ranges for the stated valuations given by respondents to specific cultural aspects of natural ecosystems (US dollars per hectare per year at 2007 price levels): aesthetics: 167-1292; recreation: 7-2211; inspiration: 0-700; spiritual experience: 21; and cognitive development: 1-22 (de Groot *et al.*, 2012). It should be noted that studies looking at cultural services were relatively sparse in the ESVD indicating a lack of standardized studies looking at the economic valuation of the cultural value of nature.

Furthermore, another more recent global survey of valuation studies restricted to forests noted that recreation values ranged between $2-279 per ha (2010 Purchasing Power Parity; Ninan & Inoue, 2013). More broadly, the large range of values points to the fact that the cultural services of nature are very site-specific and depend on the social-economic context for the group from which the values were estimated (Nunes & van den Bergh, 2001).

Figure 5 20 Number of studies by in EVRI database by type of stress.

The Environmental Valuation Reference Inventory (EVRI) consists of over 4000 studies on the stated valuation of nature. An analysis of the studies to date shows that the biggest drivers of loss of existence value have been habitat loss/degradation, resource extraction, and environmental pollution. Source: EVRI (2016).



5.9.2. Cultural significance of nature

Moving beyond the individuals and turning to societies and cultures, engagement with and situation within a particular natural environment is very often a cornerstone of cultural identity itself. These connections can take the form of both specific sacred sites as well as entire landscapes. Regardless of the connection, ecological degradation, even if it has limited impact on ecosystem function, has been shown to create a cultural or spiritual loss that contributes to the impoverishment of cultures.

5.9.2.1 Sacred sites

Sacred natural sites have been documented on every inhabited continent (Dudley *et al.*, 2009). Examples that are also within protected areas include: Kata Tjuta National Park (Australia), Laguna de la Cocha (Columbia), Sagarmatha National Park (Nepal), Sacred Groves of Oshogbo (Nigeria), Laponian area (Sweden), and Coconino National Forest (USA). Please see Dudley *et al.* (2005a) for the detailed descriptions and photo documentation of 100 sacred sites. Sacred natural sites can be the focal point for many communities with spiritual ceremonies performed there often involving key elements from the ecosystems as features (Jeeva *et al.*, 2006; Ormsby & Ismail, 2015). It has been shown that knowledge of the continued existence and integrity of sacred natural sites are linked directly to an individual's sense of well-being, and degradation can constitute a cultural or a spiritual loss (Russell *et al.*, 2013). In certain cultures, particular sites or species are a central underpinning of the culture itself, and their loss would result in a deep change to the society's identity (Garibaldi & Turner, 2004; Jeeva *et al.*, 2006; Vitebsky, 2015). Examples of land degradation leading to a loss of or a reduction in the use of sacred sites can be found in Canada, Zanzibar, India, and Kenya (Bagine, 1998; Khumbongmayum *et al.*, 2004; Madewaya *et al.*, 2004). Two good examples come from First Nations Peoples of Canada. Reduced harvest of the cultural keystone species edible red laver seedweed (*Porphyra abbottiae*) by Coast Tsimshian, Haida, Heiltsuk, and Kwakwaka'wakw people of British Columbia have reduced communal opportunities to learn and teach stories, song, and language, while the replacement of Indian swamp potato (*Sagittaria latifolia*) by the potato (*Solanum tuberosum*) has eliminated an important trade item and altered the family structure of the Katzie and other Sto:lo peoples, also in British Columbia (Garibaldi & Turner, 2004). Additionally, see Chapter 2.2.2.1 for the impact of gold mining on the Yanomami peoples of Amazonia. Even in cases where the loss of species or site would not result in a large change in ecosystem function, the cultural loss of such a change can be great (Russell *et al.*, 2013).

There is a growing body of evidence that suggests sacred sites and sacred species play an important role in conservation (Dudley *et al.*, 2009). Sacred sites often receive a degree of protection from local communities that is greater than that received by non-sacred ecosystems (Jeeva *et al.*, 2006). A study of sacred forests in Ethiopia found that, while all of the non-sacred forest patches declined in size over the study period, only two thirds of the sacred groves did (Daye & Healey, 2015). However, this cultural protection from degradation was to some extent counteracted by the fact that sacred groves tended to be smaller than non-sacred forest patches, and thus were more susceptible to edge effects. There is also a concern that as they do degrade, their cultural value to local communities may decline, which may result in decreased protection and further accelerating degradation.

5.9.2.2 Nature and cultural identity

Moving beyond specific sacred sites, there is compelling evidence from cultures around the world of the importance of natural landscapes and ecosystems for maintaining culture and identity. Individual species may have particular cultural importance: the various salmon species for many coastal peoples of the Pacific Northwest of the USA and of British Colombia, Canada (Winthrop, 2014), western red-cedar, again for peoples of the PNW and BC (Garibaldi & Turner, 2004), seal for the Inuit of Baffin Island, Canada, (Harder & Wenzel, 2012), caribou for the Saami of Siberia (Vitebsky, 2015), or black bucks for the Bishonoi cult of India (Kala & Sharma, 2010). The significance of these species goes beyond their contribution to consumption and livelihoods. The acts of harvest and consumption can themselves become central parts of identity, and can form the basis for demonstrations of communal reciprocity and structures of sharing that contribute to building a collective identity (Harder *et al.*, 2012; Kelty & Kelty, 2011; Smith & Bird, 2000).

A connection with nature can go to the very heart of a culture. Members of the Nez Perce Tribe in Idaho have described the connection to the land as the very essence of the culture itself, without which the culture itself would not survive (Kawamura, 2004). For the Gimi people of Papua New Guinea, simply articulating the notion of nature as separate from culture makes little sense as they view nature as a manifestation of their ancestors (Russell *et al.*, 2013). Kazakh communities in Western Mongolia often define Kazakhness itself in terms based in the ecology of the mountains (Post, 2007). For the Inuit peoples of the Arctic, without going to the land, "Inuit would not be Inuit any more…" (Dorais, 1995). For the Yanomami people of Amazonia the spirituality of the rainforest infuses the ethical principles that guide daily life (Kopenawa, 2013). Finally, Ecuador and Bolivia have written into their national constitutions the concept of "Buen vivir" or "Vivir bien" (used respectively in each country). This concept is based on ancient and traditional Andean knowledge and recognizes that individuals depend on nature (Acosta, 2008; Walsh, 2010). These examples and many others speak to an immense value placed on the natural world that truly cannot be measured in terms of economic benefit or other quantifiable aspects of quality of life.

5.9.2.3 Cultural loss, land degradation and land restoration

Given the close connection between many cultures and the natural environments upon which their traditions depend, it is not surprising that land degradation can in some cases lead to significant loss of cultural traditions. In one particularly pronounced examples, Reyez-Garcia and colleagues (Reyes-García *et al.*, 2013) found that among the Tsimane' people of Bolivia who inhabit an area experiencing rapid ecological change, there was a reported loss of knowledge of traditional plant uses that ranged from 9% to 26% (depending on the subset of individuals) over the course of only nine years (2000-2009). This loss of knowledge of traditional uses was associated with an increasing feeling of detachment from traditional culture, both at the individual level and at the community level (Reyes-García *et al.*, 2014).

There is some potential for land restoration activities to play a role in mitigating cultural loss. Research in Indonesia found that indigenous Batin Sembilan people were able to maintain their harvest of traditional forest products in a restored forest landscape at a similar level to what it was in natural forest areas (Widianingsih *et al.*, 2016). This suggests that there is potential for restored landscapes to indeed capture some of the cultural value of the same landscapes pre-degradation. However, in some cases, there may be a particularity of place that is difficult or impossible for restoration efforts to replicate. As an example, among the Yakama people of Washington State, USA, edible roots of several local plants are culturally significant; members of the Yakama also perceive roots from a given area to have distinct spiritual properties. When a pipeline company developed part of the Yakama's territory, the company suggested a restoration project that involved re-planting individual plants from nursery stock. This re-planting strategy was unacceptable to some members of the tribe who viewed this transplantation of plants of the same species from a different location as failing to replace what had previously been lost because the connection between plant and place had been broken (Winthrop, 2014). This example illustrates the particular relationship between restoration and the cultural values of ecosystems. Restoration may be effective in many contexts, but to effectively respect cultural values, any project must closely engage with local populations.

5.9.3 Ecosystem services under diverse approaches to valuation

It is clear from the preceding sections that land degradation impacts quality of life in significant ways. These impacts are generated both through the direct effects of natural systems on human lives as well as through the complex interactions of natural systems with anthropogenic assets, governance, institutions, and varied worldviews. For many of the measures of human quality of life – for example poverty, food security, health, energy and water security – there is conceptual and practical agreement in viewing nature's services in a similar way to economic commodities that may be managed to optimize quality, quantity, and distribution in such a way as to achieve the welfare goals of individuals and societies. However, this economic conceptualization of nature's contributions to people does have shortcomings; in particular, in its limited ability to address different worldviews and conceptions of the value of nature, as well as in its implicitly anthropocentric and utilitarian framing of the value of nature.

As humanity moves through the 21st century on a hotter, more crowded, and less natural planet, this approach to treating nature as a commodity may not be enough to preserve the species and ecosystems that humanity has benefited from and interacted with for millennia. As McCauley (2006) points out, the logic of conserving nature based on its services to humans relies on three implicit, and questionable, assumptions: first, that our motivation to conserve nature should be based on nature's benevolent provision of services to us and on the protection it affords us from malevolent abiotic forces. This provides us little guidance for what to do in those situations where some aspect of the natural world in fact provides a disservice to humans. Second, by allowing nature's value to be defined by markets or human preferences, the ecosystem service framing implies that the value of nature itself changes when the strength and direction of market forces or social preferences inevitably change – a proposition that is problematic for long-term conservation. Lastly, ecosystem service valuation implies that the value of nature declines any time a new technology is created that can replace a service provided by nature. In an era of rapid technological change – where human ingenuity is constantly finding new ways to imitate or replace components of ecosystem function – this results in a valuation framework where the value of nature is constantly falling, creating a clear problem for motivating conservation efforts.

Even if these issues can be overcome, it may never possible to overcome serious errors in valuation that an ecosystem services paradigm (Daily *et al.*, 1997; MA, 2005) imposes on non-material and cultural contributions that nature makes to human well-being. These errors in valuation include: (i) a conflation of ethical beliefs and economic beliefs; (ii) a framing of "nature as service provider" that is alien to some societies; (iii) a failure to recognize that in some societies, the idea of ownership of nature is deeply problematic; (iv) a limited accounting of the fact that some values are incommensurate and are not amenable to trade-offs; and (v) a reliance on the economic concept of an independent and rational self-actor that is often an inadequate basis for evaluating the highly social nature of many environmental practices (Winthrop, 2014).

Deciding what nature humanity wants requires elevating and strengthening conversations on the non-material and perhaps unquantifiable values of nature to humanity. There is a great need to strengthen, deepen and broaden research across disciplines on the cultural values of nature, how they are affected by land degradation, and to what extent they can be enriched by restoration. However, given the rate of global changes and the rapid loss of indigenous cultures, even emphasizing the non-material and cultural contributions of nature to human well-being may not be enough to preserve it for future generations of humanity. It may require moving beyond the instrumental as well as existence value of nature to humans to a conception of intrinsic value. Intrinsic value is separate from existence value, where the latter is the valuation that humans put on the simple fact of the existence of nature. Intrinsic value is one step further removed: it is the value of nature that exists completely apart from any assessment or valuation by humans (Davidson, 2013). This ethical perspective – that nature may have a value apart from any utilitarian value people may place on it – is seldom recognized in the ecosystem services literature, even when that literature does include discussion of existence values and of non-material services. Recognizing the intrinsic value of nature entails a reframing of nature and its components to see them as ends in themselves and not simply as means to the satisfaction of human ends (Batavia & Nelson, 2017).

5.10 References

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Chapter 6

Responses to halt land degradation and to restore degraded land

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Executive summary

The most cost-effective approach to reduce land degradation in the long run is to follow the adage "prevention is better than cure" *(well established)* {6.3.1, 6.3.2, 6.4.2}. The economic consequences of land degradation are significant. For example, a study of fourteen Latin American countries estimated annual losses due to desertification at 8-14% of agricultural gross domestic products (AGDP), while another study estimated the global cost of desertification at 1-10% of annual AGDP. Across all biomes, estimates of the ecosystem service values lost due to land degradation and conversion range from $4.3 to $20.2 trillion per year. In a global study that considered values of forests for wood, non-wood products, carbon sequestration, recreation and passive uses, it was estimated that the projected degradation and land-use change would reduce the value of these forest ecosystem services by $1,180 trillion over a 50-year period, between 2000 to 2050 {6.4.2.3}. However, a broad range of sustainable land management, soil and water conservation practices, and nature-based solutions, have been effective in avoiding land degradation in many parts of the world *(well established)* {6.3.1, 6.3.2}. For example, agroecology, conservation agriculture, agroforestry and sustainable forest management can successfully avoid land degradation, while enhancing the provision of a range of ecosystem services *(well established)* {6.3.1.1, 6.3.2.3}. Many of these same techniques and measures can also be used to restore degraded lands, but may be more costly than their use for avoiding land degradation *(well established)* {6.3.1, 6.3.2}.

There are no "one-size-fits-all" biophysical and technical responses for avoiding and reducing land degradation, nor for restoring degraded lands *(well established)* {6.3.1, 6.3.2, 6.4.2}. Actions to avoid or reverse land degradation (of croplands, forests, rangeland, urban land, wetlands) – or to deal with the adverse impacts of invasive species, mineral extraction activities, deterioration of soil health and water quality and climate change – are more effective when they are designed to fit local environmental, social, cultural and economic conditions *(well established)* {6.3.1}. Key considerations for response actions include: the types and severity of degradation drivers and processes affecting the land {6.3.2}; past and present land uses and their socio-economic contexts; and institutional, policy and governance environments {6.4.2} (*well established*). Further, the effectiveness of these actions is often enhanced by the integration of indigenous and local knowledge and practices (*well established*) {6.4.2.2, 6.4.2.4}.

Direct biophysical and technical responses, and their effectiveness to address land degradation drivers and processes, depend on the nature and severity of drivers and the prevailing enabling environment *(well established)* {6.3.2}. Responses to land degradation due to invasive species include identifying and monitoring invasion pathways and adopting quarantine and eradication (mechanical, cultural, biological and chemical) measures *(well established)* {6.3.2.1}. Responses to land degradation from mineral resource extraction include: on-site management of mining wastes (soils and water); reclamation of mine site topography; conservation and early replacement of topsoil; and passive and active restoration measures to recreate functioning grassland, forest and wetland ecosystems *(well established)* {6.3.2.2}. The responses to invasive species and mineral extraction-related degradation are successful where restoration plans are fully implemented and monitored following an adaptive management approach.

Conservation agriculture, agroecology, agroforestry and traditional practices are effective ways to use and manage soil and land resources sustainably *(well established)* {6.3.1.1}. These management practices can be effective in reducing soil loss and improving soil quality, as well as other biogeochemical functions and processes in soils including: biological productivity; hydrological processes; filtering; buffering and nutrient cycling; and habitat quality for soil and above-ground organisms and communities {6.3.1.1, 6.3.2.4}. A strong commitment to continuously monitor the quality of soil resources is needed to improve management decisions that consider not only short-term economic gains, but also off-site and long-term consequences.

Effective responses to rangeland degradation include land capability and condition assessment and monitoring, grazing pressure management, pasture and forage crop improvement, silvopastoral management, and weed and pest management *(well established)* {6.3.1.3}. These biophysical responses are generally effective in halting rangeland degradation, but the effectiveness can be enhanced by aligning these responses with social and economic instruments (*well established*) {6.3.1.3}. For example, historic nomadic pastoral grazing practiced on the Egypt-Israel border has been found to be more effective for maintaining rangeland resources than year-round livestock husbandry in pastoral farm and village settings. Shepherd communities of the Jordan Valley have avoided the degradation of pasture land through restrictions on their herds' mobility, with the establishment of new national boundaries throughout the 20th century. The ability of the stationary pastoral rural communities to maintain systematic or semi-systematic grazing and rangeland development regimes also improve their resilience to climate change {6.3.1.3}.

The effectiveness of responses to wetland degradation and water quality degradation depend on the adoption of integrated soil and water management techniques and their implementation *(well established)* {6.3.1.5, 6.3.2.4}. The effective responses to avoid or reverse wetland degradation include controlling point and non-point pollution sources by adopting integrated land and water management strategies and restoring wetland hydrology, biodiversity and ecosystem functions through passive and active restoration measures such as constructed wetlands *(well established)* {6.3.1.5}. Similarly, effective responses to improve water quality include soil and water conservation practices, controlling pollution sources and desalination of wastewater *(established but incomplete)* {6.3.2.4}.

Responses to halt urban land degradation and to improve the liveability in urban areas include improved planning, green infrastructure development, amelioration of contaminated soils and sealed soils, sewage and wastewater treatment, and river channel restoration *(well established)* {6.3.1.4}. The effectiveness of these responses to minimize urban land degradation depends on the context as well as effective implementation. In developed countries, where large urban populations are concentrated, catchment-level natural capital and/or ecosystem service approaches have been proven to be effective in reducing flood risk and improving water quality through the restoration of biodiversity and use of sustainable land management techniques *(established but incomplete)* {6.4.2.3, 6.4.2.4}.

Enabling and instrumental responses address indirect drivers of land degradation and create conditions to enhance effectiveness of direct biophysical and technical responses (*well established*) {6.4.1, 6.4.2, 6.4.5}. A range of enabling and instrumental responses are available to avoid, reduce and reverse land degradation, and address its indirect drivers (e.g., economic and socio-political). These include a variety of legal and regulatory, rights-based, economic and financial, and social and cultural policy instruments such as: customary norms and support for indigenous and local knowledge; strengthening of anthropogenic assets such as research and technology development, skills and knowledge development; and institutional reform (*well established*) {6.4.2}. For example, the application of appropriate legal and regulatory instruments - and the establishment of appropriate governance structures and the devolution of power - have enabled successful restoration or rehabilitation of degraded forest lands and watersheds, in many parts of the world {6.4.2.1, 6.4.2.4, 6.4.5}.

The benefits of taking action (restoring degraded land) are higher than the costs of inaction (continuing degradation) *(well established)* {6.4.2.3}. For example, a study of large-scale landscape restoration in Mali found that adapting agroforestry is economically beneficial, providing direct local benefits to farmers of $5.2-5.9 for every dollar invested over a time horizon of 25 years. Investments in restoration can also stimulate job creation and economic growth. In the USA for example, the average number of jobs created per $1 million invested in restoration programmes has been estimated to be 6.8 for local-level wetland restoration, 33.3 for invasive species removal, and 39.7 for national-level forest, land and watershed restoration. The direct employment of 126,000 workers in restoration projects in the USA generates $9.5 billion in economic output annually - which indirectly creates an additional 95,000 jobs and $15 billion in annual economic output. The employment multiplier for restoration activities in the USA ranges from 1.5 to 2.9, comparable to that of other sectors, including the oil and gas industry (3.0), agriculture (2.3), livestock (3.3) and outdoor recreation (2.0) {6.4.2.3}.

More inclusive analyses of the short-, medium- and long-term costs and benefits of avoiding and reversing land degradation can support sound decision-making by landowners, communities, governments and private investors *(established but incomplete)* {6.4.2.3}. Economic analyses that consider only financial or private benefits and utilize high discount rates favour less investment in sustainable land uses and management practices, while undervaluing biodiversity, ecosystem services, public values and intergenerational benefits. The incorporation of a broader set of non-marketed values in cost-benefit calculations - such as the provision of wildlife habitat, climate change mitigation and other ecosystem services - would encourage greater public and private investment in restoration projects *(established but incomplete)* {6.4.2.3}. Fulfilling land degradation neutrality objectives and large-scale restoration goals requires creating (economic) incentives that encourage landowners, land managers and investors to recognize and capture the public value of restoring degraded land, particularly in severely degraded landscapes.

The effectiveness of policy instruments depends on the local context, as well as the institutional and governance systems in place *(well established)* {6.2.2, 6.4.2}. A variety of instruments have been used to promote the adoption of sustainable land management practices and these have been generally effective {6.4.2}. Establishment of protected areas, as a legal/regulatory response, has been instrumental in avoiding land degradation across the world *(established but incomplete)*, but their effectiveness varies with context *(established but incomplete)* {6.4.2.5}. The area of production forestry under forest certification (eco-labelling) schemes such as the Forest Stewardship Council (FSC) and the Programme for the Endorsement of Forest Certification (PEFC) standards has increased in recent years {6.4.2.4}. Customary norms (local and indigenous practices) adopted by local communities have avoided land degradation and contributed to sustainable land management, for centuries {6.4.2.2}. While such practices are generally heterogenous and context specific, they are nearly always based on long-term experience and innovation, and in tune with local needs {6.4.2.4}.

The economic and financial instruments to avoid land degradation and to restore degraded land in order to provide ecosystem services and goods include: policy-induced price changes (i.e., taxes, subsidies); payments for ecosystem services; biodiversity offsets; improved land tenure security (establishing property rights); and the adoption of natural capital accounting to reflect the flow and stock value of natural assets in national accounts *(established but incomplete)* {6.4.2.3}. Tax measures which restrict land degrading behaviour and subsidies to promote land restoration activities have been mostly successful *(well established)* {6.4.2.3}. Effectiveness of emerging incentive schemes such as payments for ecosystem services (e.g., REDD+)

and biodiversity offsets are context dependent and hence are also sometimes in conflict with local norms and land management practices - requiring more evidence before upscaling these approaches *(established but incomplete)* {6.4.2.3}. Secure property rights are an essential and effective way to avoid land degradation in situations where these rights are not well defined *(well established*) {6.4.2.3}. Natural capital accounting as a response to land degradation is in its infancy, but is a promising tool for avoiding land (flow and stock) degradation by bringing the true value of land - including non-monetary societal values - into land management decision-making *(unresolved)* {6.4.2.3}.

Integrated landscape planning to address land degradation problems that involves both the private and public sector can successfully create synergies across relevant sectoral development policies while minimizing trade-offs *(established but incomplete)* {6.4.3}. This would typically involve: (i) the promotion of sustainable land management practices (arable and urban lands); (ii) community-based management and decision-making - including traditional and local practices; (iii) climate change adaptation planning; and (iv) enhancing effective corporate social responsibility approaches from private sectors in an integrated way (i.e., aligning with other sectoral development priorities) (*established but incomplete*) {6.4.2.4, 6.4.2.6, 6.4.3}.

Anthropogenic assets required to address land degradation and restoration needs (knowledge, capacities and resources) are unevenly distributed within, and especially between, countries and regions *(established but incomplete)* {6.4.4}. Gaps or inadequacies in knowledge and skills, capacity and resources among countries need to be addressed to halt land degradation and restore degraded lands {6.5}. Particularly, there is a need for capacity-building in sustainable land management, including efficient land information systems in many developing countries that are prone to and affected by land degradation {6.4.4}. However, while labour-intensive restoration approaches may be more feasible in countries with lower labour costs (such as in Asia and the Pacific), their application may be limited by the training or extension gaps required by local communities to implement such practices.

Institutional reform that enables community-based natural resource management and the utilization of both Western scientific and indigenous and local knowledge or practices have been proven effective for conserving forests, soils, wildlife (biodiversity) and water quality in developing countries *(well established)* {6.3.1.1, 6.3.1.2, 6.4.2.4, 6.4.5}. In Nepal, for example, the establishment of local Community Forest Users Groups have been highly successful in avoiding deforestation and forest degradations as well as restoring previously degraded forest landscapes {6.4.5}. In other countries and contexts, legal instruments and compliance mechanisms adopted by local authorities have been the preferred approach to avoid land degradation and to restore degraded lands, as for example in the case of the restoration of degraded watersheds in China’s Loess Plateau region {6.3.1.1}.

6.1 Introduction

The design and application of effective, preventive as well as mitigation responses to land degradation requires a thorough understanding of its drivers (Chapter 3), processes (Chapter 4) and impacts on human well-being (Chapter 5). Human responses to land degradation and restoration can be broadly grouped into enabling and instrumental responses (i.e., legislation, policy, institutions and governance systems) and direct biophysical and technical responses (i.e., on the ground actions). Because of complexity and site-specificity of land degradation and restoration responses, any type of human action must be based on the best available knowledge from all sources (i.e., natural and social science, indigenous and local knowledge) (Reed *et al.*, 2011; SRC, 2016a; SRC, 2016b). For responses to be effective in bringing desirable changes, they must be technically and environmentally sound, economically viable, socially acceptable and politically feasible (Hessel *et al.*, 2014).

Typical direct responses often include a wide range of conservation measures and land management practices that have been used to avoid or reduce land degradation (Liniger & Critchley, 2007). The effectiveness of these direct responses often depends on enabling and instrumental initiatives and policy instruments designed to halt land degradation and promote restoration (Geist & Lambin, 2002; Hessel *et al.*, 2014; Reed *et al.*, 2011). Those policy instruments include: (i) legal and regulatory rules; (ii) right-based instruments and customary norms; (iii) economic and financial incentives (e.g., taxes, subsidies, grants, or creation of new markets such as payments for ecosystem services); and (iv) social and cultural programmes (e.g., eco-labelling, education/training, corporate social responsibility and voluntary agreements).

Historically, various types of enabling, instrumental and direct responses have been applied to address land degradation drivers and processes under different situations. As stated by Lal *et al.* (2012), these mitigation or restoration responses have been applied individually, or in combination, at micro (e.g., farmer adoption of zero tillage practices) and macro scales (e.g., striving f5or a “land degradation neutral world” by the global community). Despite a growing knowledge base regarding drivers, processes and their interactions on both ecosystem services and quality of human life (i.e., food, feed, fibre, fuel supplies and social stability), progress towards effectively responding to land degradation remains a formidable challenge (Winslow *et al.*, 2011).

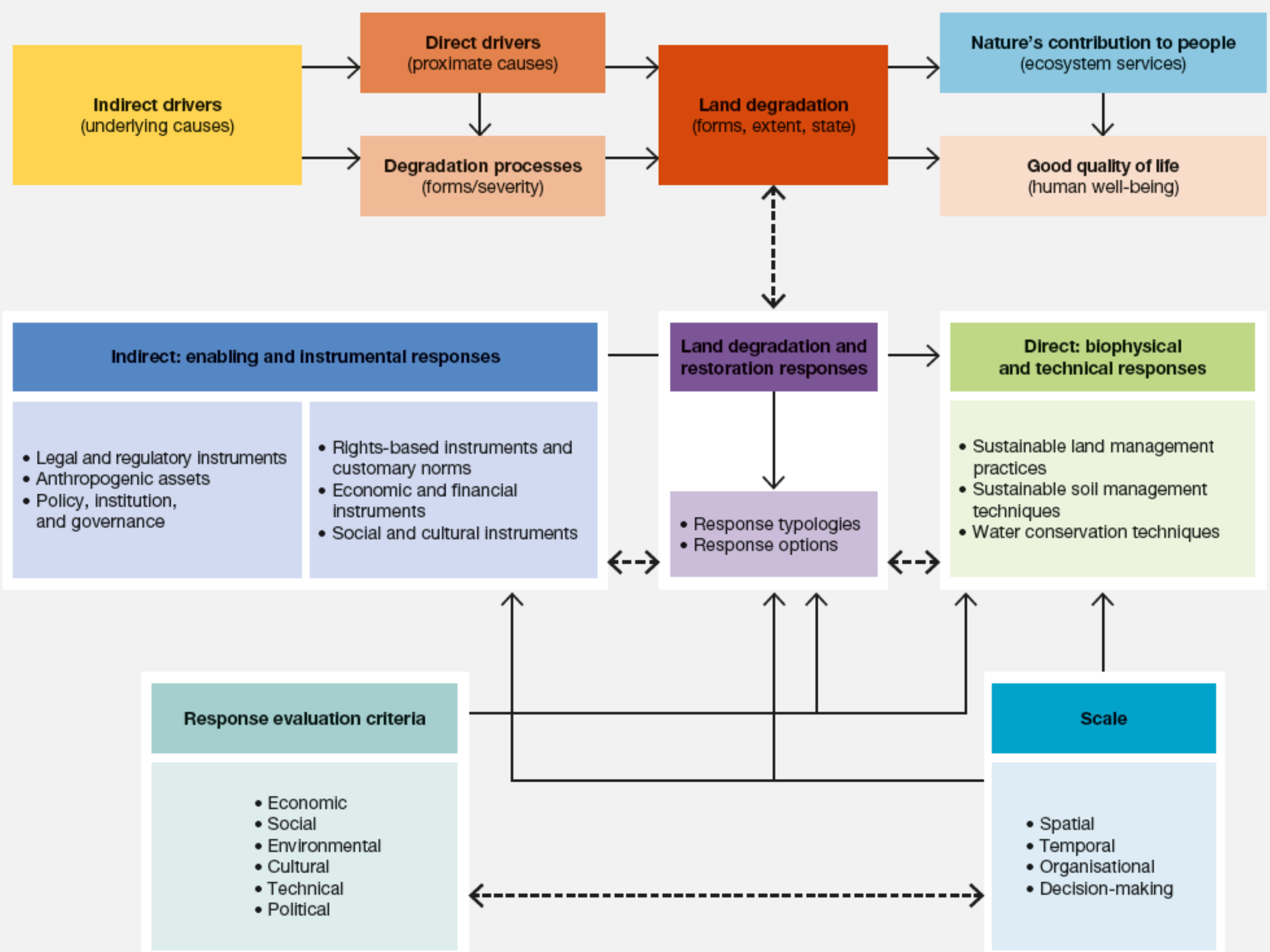
Consistent with the IPBES framework (Díaz *et al.*, 2015), this chapter focuses on critical evaluations of current response strategies; both their effectiveness for avoiding or mitigating land degradation and for restoring previously degraded lands are examined. More specifically, this chapter:

- Develops a chapter-specific framework to assess the effectiveness of existing interventions designed to avoid and reduce land degradation processes and to rehabilitate or restore various types of degraded lands (e.g., croplands, rangelands, forest lands, urban lands and wetlands) through the recovery of biodiversity, ecosystem structure and services. The ultimate goal is to enable the land to provide the essential functions needed to sustain human societies;
- Assesses how responses to land degradation and restoration vary according to site-specific characteristics, including the type and severity of degradation, underlying direct and indirect drivers, and effects on ecosystem services and quality of life;

- Evaluates the effectiveness of various response options to direct drivers (e.g., better land management techniques, access to training) and indirect drivers (e.g., institutions, governance systems) of land degradation;
- Examines the relative success of different institutional, governance and management response options to avoid, reduce and reverse land degradation across a range of economic, social, environmental, cultural, technical and political scenarios; and
- Assesses different institutional, policy and governance responses to research and technology development.

Recognizing that land degradation and restoration responses operate at different temporal, spatial, organizational and decision-making scales, we developed a chapter-specific conceptual framework (Figure 6.1) to evaluate the effectiveness of various response options based on the conceptual frameworks of IPBES (Díaz *et al.*, 2015) and the Economics of Land Degradation (Mirzabaev *et al.*, 2015).

Figure 6 1 Framework to evaluate effectiveness of land degradation and restoration responses, including prevention, mitigation and rehabilitation.



The dashed or two headed arrows in Figure 6.1 represent interdependencies between framework components, while the response criteria per se include: economic (feasibility, efficiency, effectiveness - on-/off- site, direct/indirect, present/future), social (equity - procedural/distributional, inclusivity, participatory, adoption potential), environmental (ecosystem function, ecosystem services, biodiversity, sustainability), cultural (compatibility with customary practice, local norms and values, indigenous and local knowledge and practices), technical (scientific skills and knowledge, technology), and political (acceptability, feasibility, policy, legal provisions and institutional support) considerations.

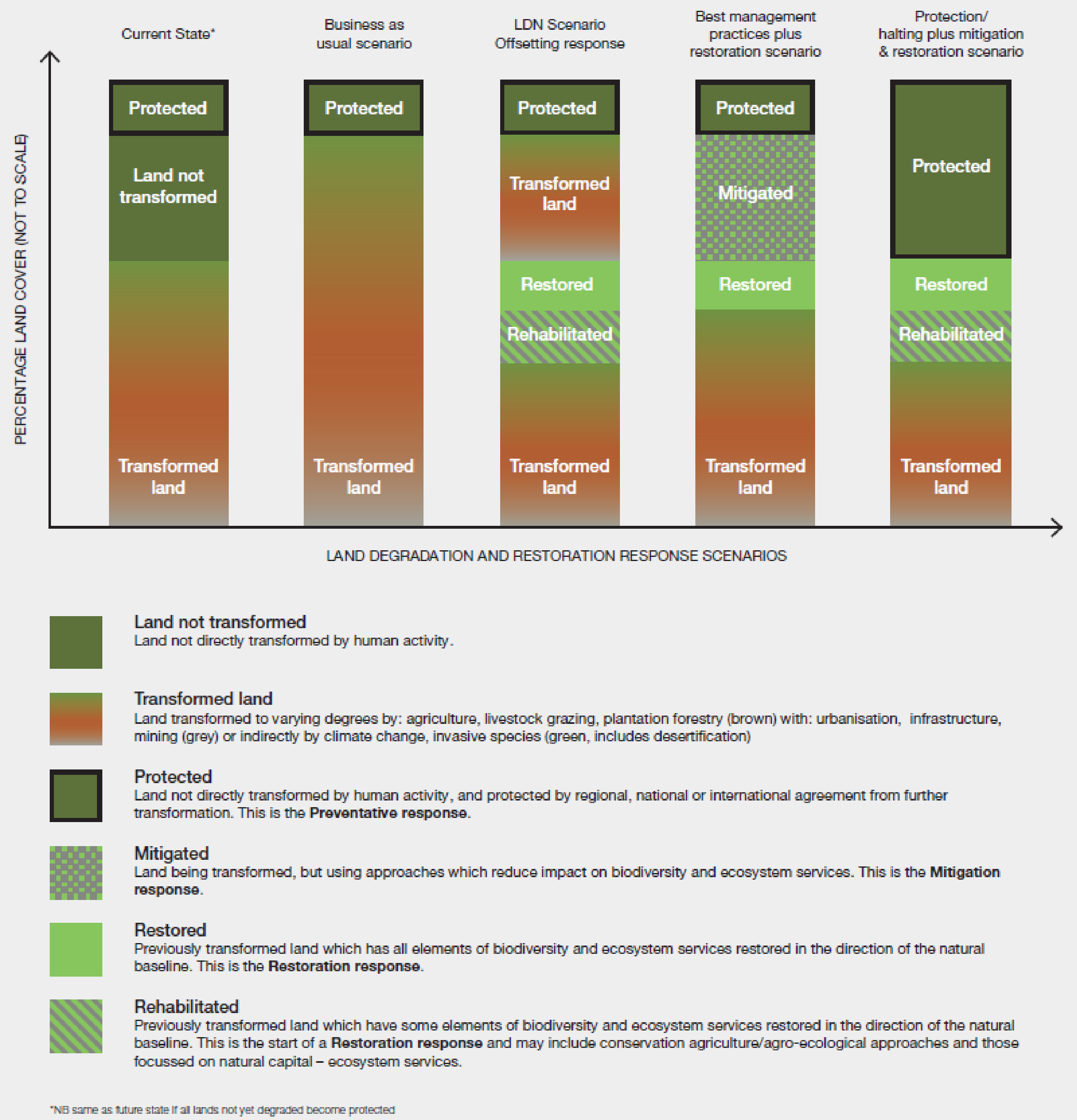
6.2 Response typology, options and evaluation framework

6.2.1 Response typology and options

To achieve land degradation neutrality, as stated in Target 15.3 of the Sustainable Development Goals, any response framework - which addresses biodiversity and ecosystem service impacts of land degradation - must consider the entire response hierarchy (i.e., prevention, mitigation, restoration and offsets). Furthermore, depending on the stage and severity of land degradation, the various drivers, processes and impacts will determine which enabling and instrumental and/or direct responses will be most effective for achieving land degradation neutrality and better scenarios (Figure 6.2, columns 3, 4 and 5).

Land degradation and restoration responses can be grouped into different typologies based on assessment needs. Response typologies can be developed based on: degradation drivers that need to be controlled; degradation processes that need to be halted or reversed; institutions that initiate the responses; types of responses that are applied to the drivers and processes (both direct and indirect); land-use categories that are affected by land degradation and need response actions; and the scale of responses - temporal (past, present), spatial and organizational (local, national, regional, global/international), and decision-making (household, community, private sector, public sector) levels.

Figure 6 2 Land cover type (not to scale) under different land degradation and restoration response scenarios.



Direct responses may seek to either avoid or reduce land degradation. Avoidance or preventive responses refer to conservation measures that maintain land and its environmental and productive functions, whereas reducing or mitigating responses are interventions intended to reduce or halt ongoing degradation and start improving the land and its functions. Reversing or restoration responses focus on the recovery of an ecosystem that has been degraded, damaged or destroyed (SERI, 2004). Offset refers to activities that compensate for residual degradation of biodiversity and ecosystem services, resulting in no-net loss in the ecological value of the impacted land (ten Kate *et al.*, 2004). In the cases where degraded land cannot be fully restored, offsetting becomes essential. Figure 6.2 shows plausible land degradation and restoration scenarios, based on the range of responses outlined in the legend. Each column in the Figure represents a unique scenario, ranging from the current state (column 1, which is same as the future state if all lands not yet degraded are prevented from becoming so) to a scenario that includes all forms of responses (column 5). The land degradation neutrality scenario with offsets is illustrated in column 3.

This chapter evaluates the effectiveness of various responses to halt land degradation and restore degraded land. Specific emphasis is given to land-use types (biomes) or complex degradation drivers and/or processes in assessing the responses. The responses are broadly grouped into two categories: enabling and instrumental, and biophysical and technical (MA, 2005; UK NEA, 2014). The enabling and instrumental responses include: legal and regulatory instruments; policy, institution and governance mechanisms; economic and financial instruments; social and cultural instruments; and rights-based instruments and customary norms. These responses seek to change or encourage human behaviour by creating a conducive environment for landholders, or other stakeholders, to operationalize biophysical and technical responses (i.e., land management practices).

Each response category has a range of appropriate response strategies depending on the form, severity and extent of degradation. Response options must be sensitive to both socio-economic and biophysical aspects of degradation and restoration strategies. Therefore, numerous options are available between enabling and instrumental responses as well as biophysical and technical responses (Liniger *et al.*, 2002; Liniger & Critchley, 2007). In practice, to achieve desired outcomes, land degradation responses need to be implemented simultaneously and in a coordinated fashion (Thomas, 2008) - using interdisciplinary and transdisciplinary perspectives which, in turn, help to fully evaluate the effectiveness of such responses (Reed & Stringer, 2015; STK4SD, 2015). Examples of synergistic response types include:

- Corrective methods (land rehabilitation and ecosystem restoration) that aim to halt and remedy degradation through, for example, conservation of soil and water, protection of vegetation, ecological engineering, and the re-establishment of functional ecosystems.
- Techniques to improve land use and management such as agroecology, agroforestry, conservation agriculture and other sustainable agricultural practices.
- Development of models and integrated natural resource management systems between local and national organizations.
- Implementation of favourable institutional, economic and political mechanisms. These may include: access to markets and sale of products from dry zones; diversification of rural economies; payment for ecosystem services; land ownership rights; access to credit; training for farmers; and insurance systems.
- Cooperation and knowledge exchange between land management, research and policy communities, as well as participatory approaches in research and development.

A detailed catalogue of sustainable land management approaches and technologies is available on the World Overview of Conservation Approaches and Technologies (WOCAT) website: https://qcat.wocat.net/en/wocat/ and in WOCAT publications (e.g., Liniger & Critchley, 2007). In Table 6.1, we present a set of land management strategies or response options illustrating the approaches and technologies outlined above.

Table 6 1 Biophysical and technical (direct) and enabling and instrumental responses to land degradation and restoration.

RESPONSE CATEGORY	MANAGEMENT STRATEGIES AND POLICY OPTIONS
DIRECT BIOPHYSICAL AND TECHNICAL RESPONSES	
Cropland degradation	Landscape approach; conservation agriculture; integrated crop, livestock and forestry systems; enhanced plant genetics; agroforestry; agroecology
Forest land degradation	Protected areas; restrictions on forest conversion; promotion of sustainable forest management practices; fire management; passive and active restoration
Rangeland degradation	Land capability and condition assessment and monitoring; grazing pressure management; pasture and forage crop improvement; silvopastoral management; weed and pest management
Urban land degradation	Improved planning; green infrastructure development; amelioration of contaminated soils and sealed soils; sewage and wastewater treatment; river channel restoration
Wetland degradation	Protected areas; control of point and non-point pollution sources; passive and active measures to restore hydrology, biodiversity and ecosystem function; constructed wetlands
Invasive species	Identification and monitoring of invasion pathways; quarantine measures; eradication measures; mechanical, cultural, biological, and chemical control
Mineral extraction	On-site management of mining wastes (soils and water); reclamation of mine site topography; conservation and early replacement of topsoil; passive and active restoration measures to recreate functioning grassland, forest and wetland ecosystems
Soil quality change	Improved agronomic practices; reduced tillage; increase diversity and vegetative cover in production systems; integrated crop, livestock and forestry systems; improved fertilizer and agrochemical use efficiency; improved irrigation and water use efficiency; reduce deposition of atmospheric pollutants
Water quality change	Integrated land and water management; rainwater harvesting; soil and water conservation practices; desalination wastewater treatment; constructed wetlands
ENABLING AND INSTRUMENTAL RESPONSES	
Responses to the adverse effects of globalisation, demographic change, migration	Trade and consumption; linking trade and environmental protection; voluntary product certification; population policies that interact with land such as resettlement, fertility rate, rural urban-migration
Legal and regulatory instruments	Land-use planning (national, regional, local); social and environmental impact assessments; incentives for sustainable land-use practices; establishment of protected areas
Rights-based instruments and customary norms	Improved land tenure security; clarification of natural resource-use rights; support for ILK-based traditional use practices
Economic and financial instruments	Policy-induced price changes; payments for ecosystem services; biodiversity offsets; improved land tenure security; clarification of natural resource-use rights; natural capital accounting
Social and cultural instruments	Participatory natural resource management and governance; support for ILK-based traditional use practices; eco-certification; promotion of corporate social responsibility;
Protected areas	Legal protection; private and community-based conservation; promotion of ILK-based traditional use
Climate change adaptation planning	Conservation of natural areas with high carbon stores (e.g., peatlands, old-growth forests, mangroves); land-use specific measures to reduce net greenhouse gas emissions; land-use specific adaptation measures
Integrated landscape planning	Sustainable land management; integrated planning and management; zoning
Anthropogenic assets	Capacity-building including: skills and knowledge development; research and technological development; extension; human resource development; infrastructure and facilities
Institutional and policy reform	Establishment of new institutions; strengthening existing institutions; mainstreaming Indigenous and Local Knowledge and Practices (ILKP); improving multi-level governance mechanisms

6.2.2 Response evaluation framework

Here, effectiveness is understood as a measure of the extent to which an activity accomplishes its objectives. Motivations of human behaviour and resilience capacity of natural systems are fundamental considerations when evaluating the effectiveness of land degradation and restoration responses. Based on the chapter-specific conceptual framework (Figure 6.1), a response evaluation framework is outlined in Table 6.2 for direct response options. The response evaluation framework considers a set of assessment criteria to evaluate the effectiveness of individual response options. Such assessment criteria include a range of economic, social, environmental, cultural, technical and political measures (Table 6.2). For example, from an environmental sustainability perspective, a response would be evaluated for its suitability to improve ecosystem functions, generate ancillary benefits (positive externalities) and its potential to address wider sustainability objectives. Similarly, from a technical feasibility perspective, a response would be evaluated on the basis of skill and knowledge requirements as well as the technological sophistication involved. For direct responses, the concept of response hierarchy is also used to evaluate response options - for instance whether a given strategy belongs to avoiding (prevention) or reducing (mitigation) land degradation or reversing (restoration) degraded land, or a combination of them. The effectiveness of response options can also be viewed on the basis of their speed and ease of implementation, time frame, acceptance by local stakeholders, endorsement by experts, institutional capacity, scale of benefits or number of beneficiaries (USAID, 2008).

Table 6 2 **Template for assessment of the effectiveness of various response options by land-use types and degradation drivers.**

LAND USE OR DEGRADATION DRIVER	RESPONSE OPTIONS	NATURE OF RESPONSE Avoid (Av), Reduce (Rd), Reverse (Rv)	RESPONSE EVALUATION CRITERIA AND EFFECTIVENESS RANKING [High effectiveness (H), Moderate effectiveness (M), Low effectiveness (L), or any combinations: L to M, M to H, L to H]					
			Economic [feasibility, efficiency, effectiveness (on/off-site, direct/ indirect, present/ future), equity -process, distribution, spill-over effect]	Social [equity, inclusivity, participatory, potential to adopt]	Environmental [potential to address environmental sustainability concerns - water security, climate change, biodiversity conservation, ecosystem service provisions]	Cultural [customary practice, local norms and values, ILK]	Technical [skills/ knowledge, technology, sophistication]	Political [legal provisions, institutional structure, political acceptability/ feasibility]
CROPLAND MANAGEMENT	1. 2.	Av/Rd/Rv	H/M/L or L-M/M-H/ L-H	H/M/L or L-M/M-H/ L-H	H/M/L or L-M/M-H/ L-H	H/M/L or L-M/M-H/ L-H	H/M/L or L-M/M-H/ L-H	H/M/L or L-M/M-H/ L-H
FOREST LAND MANAGEMENT	1. 2.	......	......	......	......	......	......	......
......	......	......	......	......	......	......	......	......

6.3 Direct biophysical and technical responses to land degradation and restoration

Land degradation and restoration responses are inherently context specific and such responses vary depending on the extent and severity of the drivers and processes, as well as specific biophysical characteristics of the place or system. In addition, on-the-ground restoration responses may depend on economic, social, cultural and technical factors. Use of case-specific analyses based on major land-use types (see Section 6.3.1) and selected drivers and processes (see Section 6.3.2) to provide an overview of the effectiveness of past and current responses to land degradation and restoration. To evaluate specific responses to the many land-use degradation drivers and/or processes, the following discussion will:

i. Identify specific land and soil management actions, based on both Western science and indigenous and local knowledge and practice (ILKP) that can halt land degradation;
ii. Specify which responses are preventive (i.e., capable of avoiding land degradation) and which are specific to mitigation (i.e., focused on reducing land degradation and reversing, rehabilitating and/or restoring degraded lands);
iii. Examine how well those responses are working and where (i.e., under what geographic, socio-economic and cultural settings);
iv. Provide examples of their effectiveness; and
v. Discuss what messages should be given to key stakeholders regarding the effectiveness of these responses.

6.3.1 Assessment of land-use specific responses

6.3.1.1 Responses to cropland degradation

Cropland soil degradation is very site specific and can occur physically, chemically and/or biologically. Potential responses to degradation include using: (i) a landscape approach; (ii) conservation agriculture; (iii) integrated crop, livestock and forestry systems; (iv) agroforestry; (v) enhanced plant genetics; and (vi) integrated watershed management.

Landscape approach

A landscape approach examines how soil resources, cropping systems, weather patterns, management practices, market development, community preferences and other factors affect ecosystem processes (Kosmas & Kelly, 2012). Indigenous peoples instinctively adopt a landscape approach as their connections to the land incorporate interactions across the landscape and understandings of the connections of all living things (Walsh *et al.*, 2013). The critical point for this response is that there is no single solution, because interactions of all these factors ultimately modify the entire landscape.

The Atlantic Forest Restoration Pact (Melo *et al.*, 2013) in Brazil provides an excellent example of the landscape approach (see Box 6.3). It demonstrated that continuous technology improvement, on-going teaching and community outreach, capacity-building, incorporation of local knowledge, a clear and transparent legal environment and effective economic instruments and incentives were all crucial for success. Other studies (e.g., Baker *et al.*, 2014; Norgaard, 2010) warn against blindly focusing on ecosystem services in lieu of ecological, economic and political complexities encountered when responding to land degradation.

Conservation agriculture

Conservation agriculture, as defined by the FAO, is characterized by three specific actions including: (i) continuous minimum mechanical soil disturbance; (ii) permanent organic soil cover; and (iii) diversification of crop species grown in sequences and/or associations. In general, conservation agriculture principles are universally applicable to all agricultural landscapes and land uses, because they emphasize the use of locally-adapted practices (based on ILKP), biodiversity and natural biological processes above and below ground (Forest People Program & Program, 2010). Interventions such as mechanical soil disturbance, and agrochemical or plant nutrient applications, are optimized so they do not interfere with or disrupt biological soil processes.

Global adoption of conservation agriculture has been increasing steadily (Friedrich *et al.*, 2012; Jat *et al.*, 2014; Reicosky, 2015) as documented by an FAO database that shows approximately 125 million hectares (8.8% of arable cropland) are now being managed using conservation agriculture. However, the FAO (2015) estimates a global growth of almost 32 million ha (26%) within the last five years. The primary limitations for the implementation of conservation agriculture include market pressure for monocrop production, climatic factors, access to conservation agriculture technology, appropriately scaled incentives and information regarding adoption (Jat *et al.*, 2014).

Two perceived conservation agriculture concerns are the high dependence on glyphosates and genetically modified plants. Regarding glyphosate, current safety evaluations have generally not indicated serious risks for human or environmental health (Williams *et al.*, 2000), although concerns persist among some public health researchers (Vandenberg *et al.*, 2017) as well as the International Agency for Research on Cancer (IARC), the specialized cancer agency of the World Health Organization, which classified glyphosate as “probably carcinogenic” to humans in 2015 (International Agency for Research on Cancer, 2015). Nonetheless, Health Canada recently determined that when used according to label directions, products containing glyphosate are not a concern to human health or the environment (Pest Management Regulatory Agency, 2017). Also, implementing conservation agriculture practices does not require the use of genetically modified plants, but rather minimum mechanical soil disturbance, permanent organic soil cover and diversity in crops grown.

The impact of conservation agriculture is illustrated in Table 6.3 which shows several countries with at least 14% of their arable cropland being managed using conservation agriculture practices. Argentina currently has the highest rate of adoption at 74%, and 90% of the 32 million ha increase during the last 5 years is accounted for by data from six countries (Table 6.4). Furthermore, data for India - which was not previously reported (Jat *et al.*, 2014) - accounted for a 1500 ha increase in conservation agriculture. We concur that adoption of conservation agriculture can be an effective preventive and mitigation strategy for addressing global cropland degradation.

Table 6.3 Countries with at least 10% of arable cropland within conservation agriculture. Source: (FAO, 2016).

Country	Conservation Agriculture (1000 ha)	Percent of Arable Cropland	Data Year
Argentina	29,181	74	2013
Paraguay	3,000	63	2013
Uruguay	1,072	44	2013
Brazil	31,811	44	2012
Canada	18,313	40	2013
Australia	17,695	38	2014
New Zealand	162	32	2008
United States of America	35,613	23	2009
Chile	180	14	2008

Table 6.4 Countries with largest recent increases in conservation agriculture. Calculated from values presented by Jat *et al.* (2014) and FAO (2015)

Country	Conservation Area Change (1000 ha)	Data Years
United States of America	+9113	2009, 2007
Brazil	+6309	2012, 2006
Canada	+4832	2013, 2006
Argentina	+3628	2013, 2009
China	+3570	2013, 2011
India	+1500	2013, none previous
Australia	+695	2014, 2008
Paraguay	+600	2013, 2008
Uruguay	+417	2013, 2008
Kazakhstan	+400	2013, 2011

Integrated crop, livestock and forestry systems

Another strategy for restoring degraded cropland (sometimes referred to as sustainable intensification) is to incorporate perennials and cattle into traditional row-crop production systems. In Brazil, sustainable intensification began slowly during the 1970s, as cattle production on native grass and bush lands within tropical savannahs became more extensive. Adaptation of new cattle breeds (mostly Nellore) and grasses such as *brachiaria* led to the development of integrated crop and livestock and integrated crop, livestock and forestry systems. These systems not only increased food and feed production at farm and regional levels, but also improved many ecosystem services (Carvalho *et al.*, 2017; Salton *et al.*, 2014; Sato & Lindenmayer, 2017).

Integrated crop and livestock has been used to restore degraded croplands in North America, Western Europe, Brazil, Uruguay and Argentina (Franzluebbers *et al.*, 2014; Peyraud *et al.*, 2014). Integrated crop and livestock - and integrated crop, livestock and forestry - have increased the amount of cultivated pasture in Brazil to nearly 101 million ha as compared to 57 million ha of native pasture. Although this is impressive, it accounts for only 32-34% of the estimated 274 -293 million animal units that could be produced in Brazil (Strassburg *et al.*, 2014). Striving for full adoption would not only result in substantial restoration of degraded croplands, but also enable Brazil to readily meet human demand for meat, crops, wood products and biofuel feedstocks until at least 2040, without any additional conversion of natural ecosystems (Strassburg *et al.*, 2014).

Agroecology

Agroecological practices encompass a broad array of agricultural technologies that take advantage of natural processes and beneficial on-farm interactions in order to reduce off-farm input use and to improve the productivity and efficiency of farming systems, enhance food security by diversifying crop production and managing environmental and economic risks, and avoid agricultural land degradation (Altieri, 2002; Gliessman, 2014; Pretty *et al.*, 2003) (see also Chapter 2, Section 2.2.4.3 and Box 2.4). Such systems, based largely on indigenous and local knowledge, have been developed and used worldwide by farmers. They typically involve management practices such as cover crops, green manures, intercropping, agroforestry and crop-livestock mixtures that promote organic matter accumulation and nutrient cycling, soil biological activity, natural control mechanisms (disease suppression, biocontrol of insects, weed interference), resource conservation and regeneration (soil, water, germplasm), and general enhancements of agrobiodiversity and synergisms between components (Altieri, 2002; Gliessman, 2014). Agroecological initiatives in many countries in Africa, Asia and Latin America - often promoted by NGOs - have had a demonstrably positive impact on farmers' livelihoods (Altieri *et al.*, 2012; Altieri & Toledo, 2011; Pretty *et al.*, 2003; Pretty *et al.*, 2011) (see also Chapter 5, Section 5.3.3.1 and Box 5.5). Success of such initiatives has been found to depend on human capital enhancement and community empowerment - through training and participatory methods as well as access to markets, credit and income generating activities, and supportive government policies (Markwei *et al.*, 2008; Pretty *et al.*, 2003; Pretty *et al.*, 2011)

Agroforestry

Agroforestry can reduce or reverse land degradation by: (i) maintaining soil fertility through increased carbon inputs, nitrogen fixation and nutrient cycling; (ii) reducing erosion; and (iii) conserving water (quantity and quality) through increased infiltration and reduced surface runoff. It can also conserve biodiversity, improve air quality, reduce reliance on fossil fuels and native forests for fuelwood, help adapt to climate change, and provide economic, social, cultural and aesthetic benefits (Murthy *et al.*, 2016). Agroforestry practices are for the most part rooted in ILK and emphasize the preservation of knowledge, local crop varieties and animal breeds, as well as native socio-cultural organizations (Lemenih, 2004; SRC, 2016b, 2016c). Innovative agroecosystem designs have been modelled on successful ILK-based practices (Altieri & Toledo, 2011; Brondízio, 2008) and it is estimated that, worldwide, as many as 500 million people practice some form of agroforestry (Nair *et al.*, 2009; Zomer *et al.*, 2014).

Box 6.1 Agroforestry responses to cropland degradation (adapted from Nair, 1993)

Agroforestry systems are typically classified on the basis of their structure (i.e., the nature and spatial and/or temporal arrangement of tree and non-tree components). They include:

- *Agrisilvicultural* - encompasses a diverse array of practices involving cultivation and management of trees and/or shrubs for food and/or non-food uses. Generally, in combination with agricultural crops, these subsystems include improved fallow (in shifting cultivation and rotational cropping), multilayer tree gardens and alley cropping. They also include different plantation crop combinations that are used not only for timber and fuelwood, but also as fruit trees within home gardens;
- *Agrosilvopastoral* - which uses domesticated animals, multipurpose woody hedgerows, apiculture, aqua-forestry and multipurpose woodlots in combinations with home gardens and fish ponds; and
- *Silvopastoral* - systems which include plantation crops, animals grazing pasture or rangeland and protein banks which produce concentrated, protein-rich tree fodder outside standard grazing areas.

Agroforestry systems are globally diverse and are widely practiced in:

- Humid and sub-humid tropical lowland regions, where they can help reduce deforestation and forest degradation. In these areas, they overcome productivity constraints of soil degradation caused by unsustainable forest management, poorly managed shifting cultivation, overgrazing, soil acidity, low soil fertility and high rates of soil erosion;
- Tropical and sub-tropical highlands, humid and sub-humid regions in the Himalayans, parts of southern India and Southeast Asia, highlands of east and central Africa, Central America, the Caribbean, and the Andes, where productivity and food security is often constrained by soil erosion, insufficient fallow periods, overgrazing, deforestation and forest degradation, as people seek fodder and fuelwood; and
- Semi-arid and arid regions where lack of precipitation, climatic change and increasing populations exceed the capacity of native forests and pastures.

A wide range of ILK-based agroforestry approaches have been used successfully in many parts of the world (Lahmar *et al.*, 2012; McLean, 2010; Parrotta & Trosper, 2012; Suárez *et al.*, 2012; Uprety *et al.*, 2012; Vieira *et al.*, 2009). In the Sahel, degraded lands have been restored using ILK techniques developed and applied by innovative farmers seeking to reverse desertification and preserve their agropastoral livelihoods (Behnke & Mortimore, 2016) (see also Chapter 4, Section 4.2.6.2). In Burkina Faso, 200 to 300 thousand ha of severely degraded farmland have been rehabilitated by combining ILK soil conservation measures and protecting on-farm trees (Botoni & Reij, 2009; Reij *et al.*, 2005; Reij *et al.*, 2009; Tougiani *et al.*, 2009). Similarly, in southern Niger, traditional agroforestry parklands have increased significantly across nearly 5 million ha through farmer-managed natural regeneration of a variety of native tree species (Reij *et al.*, 2009).

Agroforestry can be very important for mitigating and adapting to climate change in regions facing both land degradation and food security challenges (Mbow *et al.*, 2014; Parrotta & Agnoletti, 2012; Verchot *et al.*, 2007), because it provides poor farmers with alternative pathways to increase productivity and food security (Lasco *et al.*, 2014; Mbow *et al.*, 2014). It also has considerable potential for carbon sequestration (Albrecht &

Kandji, 2003), because the above- and below-ground carbon density of typical tropical agroforestry systems is estimated at 12 to 228 Mg ha^{-1}, with a median value of 95 Mg ha^{-1} (Albrecht & Kandji, 2003). For smallholders, potential carbon sequestration rates generally range from 1.5 to 3.5 Mg C ha^{-1} yr^{-1} (Montagnini & Nair, 2004). The potential of agroforestry to serve as a carbon sink, however, depends on the climatic zone conditions and silvicultural practices including planting density, species choice and length of rotation (Nair *et al.*, 2010).

In summary, agroforestry-based land restoration initiatives are relevant for the planning and/or monitoring of national and international policy objectives related to landscape restoration and biodiversity conservation, due to their potential for: (i) recognising and incorporating indigenous and local knowledge; (ii) combining social development and ecological conservation and restoration objectives; and (iii) fostering cross-sectoral collaboration between local communities, governmental agencies, NGOs, universities and research institutions (Altieri, 2004; Altieri & Toledo, 2011; Chirwa & Mala, 2016; Nair, 2007; Norton, 1998; Ouédraogo *et al.*, 2014; Parrotta *et al.*, 2015; Powell *et al.*, 2013; Walker & Macdonald, 1995)

Use of Enhanced Plant Genetics

The use of drought-resistant crop varieties by smallholder farmers to adapt to climate change and soil degradation in several African countries has been quite successful (Fisher *et al.*, 2010; Tschakert, 2007). By including pulses in mixed cropping systems, water-use efficiency and nutrient cycling were improved (Valentin *et al.*, 2008). Implementation of such practices could reduce global anthropogenic CO_2 emissions by 6 to 17% (Van Der Werf *et al.*, 2010); confirming that good agricultural management can increase productivity and carbon sequestration, while also reducing carbon emissions (West & Marland, 2003). Therefore, combining improved plant genetics with decreased tillage and efficient use of fertilizer and irrigation water can not only increase soil organic carbon, but contribute to climate change mitigation (Lal, 2002).

Integrated Watershed Management

Integrated watershed management provides another strategy to meet global demands of more than 9 billion people by the middle of the twenty-first century. Decreasing tillage frequency and intensity coupled with restoring or increasing soil organic carbon are two mitigation/restoration strategies that have been successfully demonstrated at the watershed scale (Box 6.2).

Box 6.2 Restoration of Degraded Watersheds: an example from China's Loess Plateau. Source: Liu & Hiller (2016); World Bank (2007).

The Loess Plateau in Northwest China occupies approximately 640,000 km^2 and is the dominant geological feature in the middle reaches of the Yellow River basin. The plateau has been inhabited for more than 8,000 years (Peng & Coster, 2007; Wang *et al.*, 2006). The forces that have driven landscape, vegetation and hydrological changes in the Plateau include the dual effects of human land use and climate change (Ren & Zhu, 1994; Saito *et al.*, 2001; Shi, 2002). The plateau's forest cover dropped down to 7–10%, from historical estimates of 50% (Cai, 2002; Liu & Ni, 2002) and 70% of the plateau is affected by soil erosion, 58% of which is extremely severe (Chen *et al.*, 2007) - with soil erosion rates among the highest in the world (Fu, 1989). In addition to downstream sedimentation and eutrophication problems (Wang *et al.*, 2006), dust storms (Luo *et al.*, 2003) and landslides (Zhou *et al.*, 2002) have also been problematic.

From 1994 to 2005, two Loess Plateau Watershed Rehabilitation Projects were implemented in 48 counties in the Shanxi, Shaanxi and Gansu provinces, and the autonomous region of Inner Mongolia. Rehabilitations of physical activities were performed over 35,000 Km^2 and with a total investment of $550 million.

A key factor leading to success in the Grain for Green Program was the integrated watershed management that created effective water harvesting structures. They were crucial for continuous vegetative cover in the large-scale reforestation, grassland regeneration and agroforestry activities (EEMP, 2013). Another, was the significant financial investment that included direct Chinese government expenditures and World Bank loans. This financing provided subsidies for farmers enabling them to restore degraded farmland by planting trees and other vegetation. The subsidies included $122/hectare for seeds and seedlings as well as annual payments for ecosystem services of $49/hectare for two to eight years (Buckingham & Hanson 2013). Specific actions that contributed to the project's success included:

Pre-rehabilitation actions

Project planning - which spanned over 3 years, integrating economic and social well-being of the people with the ecological health of the environment.

Land-use mapping - to optimize selection of cropland versus land left to regenerate naturally.

Adoption of new policies - including bans on planting steep slopes, cutting trees and allowing free range grazing (all to enable re-establishment of local vegetation).

Community participation - emphasizing local input into rehabilitation programmes.

Responses during rehabilitation

Technical - including hard and soft engineering for sustainable water management, terracing and dam construction in deep valleys for erosion and sediment control. Dam construction was continued until the entire gully bottom consisted of flat fields and rich productive croplands that increased farmer income, quality of life and discouraged them from planting on steep slopes.

Greening activities - which stabilized dunes using straw and plantings of grasses, bushes, trees and perennial cash crops.

Post-rehabilitation Responses

Buckingham & Hanson (2013) summarized several positive benefits including:

- Increased per capita grain output from 365 to 591 kg ha^{-1} yr^{-1}
- A 95% conversion of sloping land to improved land uses
- A 159% increase in community income
- New infrastructure and development opportunities
- Terracing of ~86,600 ha of new farmland
- A decrease in farming of unstable sloped lands from 451,000 to 278,000 ha
- A 99% decrease in sediment (~300 million tons yr^{-1}) deposited into the Yellow River
- Establishment of ~290,000 ha of shrub and economically valuable trees

Additional benefits of the Grain for Green Program have been reported by Cheng *et al.* (2016); Deng *et al.* (2014); Liang *et al.* (2012); Tsunekawa *et al.* (2014); and Wang *et al.* (2016).

Community development

The Grain for Green Program has resulted in profound lifestyle changes and has benefited many benefits for local people, in a variety of ways. Local communities now enjoy better facilities, infrastructure and amenities, including roads, clean water, electricity, schools, hospitals, new housing and township developments.

Figure 6 3 The Ho Family Gully on the China Loess Plateau before [A late August 1995] and after [B late August 2009] the "Grain for Green" conservation program. Photo Credits: Liu & Hiller (2016).

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6.3.1.2 Responses to forest land degradation

Responses to deforestation and forest degradation include preventive measures, the integration of production with conservation objectives (through agroforestry, natural and planted forest management) and restoration. Countries with low or negative deforestation rates have either managed their forests sustainably or restored degraded lands based on one or more of these strategies.

Avoiding deforestation, forest fragmentation and forest degradation

Avoiding deforestation and reducing forest fragmentation is particularly important for forest ecosystems that are still largely intact. It is both more cost-effective and conserves more biodiversity than is possible through restoration, at least in the medium term (Benayas *et al.*, 2009). While the establishment of protected areas has frequently been the only mean to conserve large intact forest areas, other landscape-planning strategies that have been effective in avoiding deforestation, including restrictions of agricultural expansion in ecologically-fragile areas and biodiversity hotspots, and intensification of agriculture in fertile and geomorphologically stable areas (Chazdon *et al.*, 2009; Lambin & Meyfroidt, 2011).

Deforestation can be avoided with controls over domestic and international markets for agricultural products where the supply chain for these products contributes to forest loss and degradation (Macedo *et al.*, 2012). For example, the Soy Moratorium in Brazil, in which traders agreed not to purchase soy from lands deforested after July 2006 in the Brazilian Amazon, resulted in a decrease in annual soy expansion into forested areas from 30% to 1% after 2006 - although expansion of soy cultivation into pastures and cleared land increased (Gibbs *et al.*, 2015), and potential leakage effects of the Soy Moratorium on the Brazilian savannahs and other countries have yet to be assessed.

Many intact (formally or informally protected) forest areas are embedded within human-modified landscapes (Melo *et al.*, 2013), where agriculture and urbanization have significantly modified landscape structure. This is often accompanied with declines in biodiversity due to dis-connectivity among remaining forest patches (Rappaport *et al.*, 2015) and with limited potential to avoid further species loss (Fahrig, 2003). Effective measures to address the negative biodiversity impacts of forest fragmentation require evaluation of the condition and attributes of remaining forest remnants (i.e., their size, shape, degree of isolation, and habitat quality and heterogeneity) and the land-use matrix in which they are embedded (Collinge, 1996).

Landscape planning (discussed further in Section 6.4.3) is an important tool for developing effective actions to avoid further deforestation and/or ameliorate forest fragmentation impacts and through conservation and restoration measures (Banks-Leite *et al.*, 2014; Tambosi *et al.*, 2014). Effective and widely-used measures to increase connectivity, conserve biodiversity and enhance delivery of ecosystem services within fragmented forest landscapes include: maintenance of vegetation corridors in riparian vegetation (Naiman *et al.*, 1993); establishing new fragments or expanding the size of existing ones through restoration (Brancalion *et al.*, 2013); and promoting agricultural practices such as agroforestry in areas surrounding intact forests (Chazdon *et al.*, 2009; Cullen *et al.*, 2001).

Payments for ecosystem services (see Section 6.4.2.3) can also promote sustainable forest management practices, particularly through the REDD+ mechanism (Reducing Emissions from Deforestation and forest Degradation), which has generated innumerable programmes worldwide - involving donors, consultants, experts, policymakers, researchers and communities (Corbera & Schroeder, 2011; Lund *et al.*, 2017). However, the effective implementation of REDD+ and other PES programmes hinges on the resolution of a number of issues related to: local conflicts among stakeholders regarding trade-offs between carbon sequestration and many of the other environmental, economic, social and cultural services provided by forests; community rights; independence from funding; and finding market funds to pay for the ecosystem services (Cadman *et al.*, 2016; Lund *et al.*, 2017; Parrotta *et al.*, 2012).

Firewood and charcoal for cooking and heating represents 55% of global wood harvest, which supplies 2.8 billion people (Bailis *et al.*, 2015) and 11.3% of the global energy demand (Guo *et al.*, 2015). Excessive firewood harvest is a significant driver of forest degradation in many countries (also see Chapter 3, Section 3.4.4.2 and Chapter 4, Sections 4.3.4 and 4.3.5). That said, forests and woodlands can and often are managed sustainably, and firewood demand is in some cases met through the use of by-products from commercial timber harvests (Bailis *et al.*, 2015; Chidumayo & Gumbo, 2013).

Over the last 40 years, concerns over the role of firewood extraction in tropical deforestation and the wood fuel shortages have prompted policy and programme interventions in many developing countries to reduce wood fuel demand and/or increase supplies, or some combination of the two. For the most part, these policy and programme interventions have failed to effectively deal with the problem of charcoal-based deforestation and its associated environmental concerns (Chidumayo & Gumbo, 2013). Nonetheless, some governments - having recognized the importance of firewood and charcoal as a principal source of energy - have sought to regulate and stimulate its sustainable production, especially given that it utilizes a local (and potentially renewable) resource and can generate local income (Chidumayo & Gumbo, 2013).

In some regions, wood fuels are being replaced by cleaner and healthier energy sources, including lignocellulosic bioethanol and biogas (Guo *et al.*, 2015). The environmental, social and economic impacts of land-use changes associated with increased production and other biofuels are the subject of considerable debate (Dai *et al.* 2011; Fargione *et al.*, 2008; Hasenheit *et al.*, 2016; Lambin & Meyfroidt 2011; Saez de Bikuña *et al.*, 2017; Whalen *et al.*, 2017).

Conserving and managing secondary forests

Secondary forests are a major part of many rural landscapes (Aide *et al.*, 2013; Hurtt *et al.*, 2006) and are increasingly recognized as important contributors of goods and services (Bongers *et al.*, 2015; ITTO, 2002), as is the need to incorporate them into land-use planning to balance conservation, production and sustainable livelihood needs. Their high potential to sequester carbon needs to be considered in public policies (Chazdon *et al.*, 2016; Poorter *et al.*, 2016), as well as their ability to restore forests at smallest costs (Bongers *et al.*, 2015). Secondary forests are often managed under adaptive and multiple-use management, not only for timber to provide short-term economic benefits, but also for food and other non-timber products through enrichment plantings with early production species, such as annual crops, fruit trees, palms and bamboos (ITTO, 2002). Managing secondary forests as productive agroforestry systems can be used to conserve biodiversity, limiting modification of the native vegetation, integrating ecosystem services schemes with benefits to local livelihoods (Mukul & Saha, 2017). Such management practices, relying heavily on indigenous and local knowledge, can be found throughout the world (Parrotta *et al.*, 2015).

Sustainable logging

Many criteria and indicators have been developed to guide sustainable forest management (Mendoza & Prabhu, 2003; Pearce *et al.*, 2003), including a comprehensive guide for reduced impact logging and sustainable management of tropical forests (ITTO, 2009; ITTO, 2016). These criteria and indicators are also used in forest certification, a market-based initiative aimed at promoting sustainable forest management (see Section 6.4.2.4). However, in countries where they would be particularly useful, these tools have not been extensively applied because of low consumer demand for sustainably-produced timber. Globally certified forest areas represented 11% of the world's forest cover in 2016, but 87% of certified forests were in the Northern Hemisphere and only 1.2% were in Africa, 3.1% in Oceania and 1.9% in Latin America (UNECE/FAO, 2016). Ninety percent of internationally-verified certification is in the boreal and temperate climatic domains, whereas only 6% of permanent forests in the tropics have been certified up to 2014 (MacDicken *et al.*, 2015).

Commercial and non-commercial planted forests

Planted forests are seen as a degradation driver, particularly when they replace natural forests (Brockerhoff *et al.*, 2008) (also see Chapter 4, Section 4.3.4). However, with the growing demand for wood products, planted forests have become a complementary measure to conserve natural forests when established on degraded lands. In fact, planted forests have reduced harvesting from natural forests globally by 26% (Buongiorno & Zhu, 2014). They currently produce 5 to 40 times more timber yield than certified natural forests (Paquette & Messier, 2010) and supply a quarter of global industrial roundwood production, while occupying only 7% of the world's total forest area (Payn *et al.*, 2015). Reducing potential negative effects and/or enhancing positive effects of establishing planted forests requires rigorous impact assessments that consider the changes in biodiversity and ecosystem services, as well as design and management measures that help to protect biodiversity. Such measures include: setting aside natural habitats along watercourses and establishing biodiversity reserves within large-scale plantation areas; utilizing or further developing silvicultural knowledge to expand the use of native species in planted forests; and adjustments to silvicultural practices to favour local biodiversity in planted forest stands and avoid introducing invasive tree species and/or their pests and diseases (ITTO, 2009).

Forest restoration

Significant opportunities exist to restore forest cover, biodiversity and ecosystem services on formerly forested degraded lands and abandoned agricultural sites (Benayas *et al.*, 2009). According to an analysis conducted by the World Resources Institute and the Global Partnership on Forest Landscape Restoration, more than two billion hectares could potentially be restored worldwide - including 1.5 billion ha considered best-suited for mosaic restoration, in which forests and trees are combined with other land uses such as agroforestry, smallholder agriculture and settlements - and up to about half a billion hectares are suitable for wide-scale restoration of closed forests (Minnemeyer *et al.*, 2011).

A variety of effective reforestation and forest management techniques are used to varying extents to restore forests in degraded landscapes, depending on ecological circumstances and management objectives (Lamb *et al.*, 2005).

These include:

- Protection of natural regrowth from fire, grazing and other stressors inhibiting secondary forest development;
- Protection of natural regrowth and enrichment with commercially, socially or ecologically valuable tree species to improve the economic and social value of these forests;
- Restoration plantings (or direct seeding) using a small number of short-lived nurse trees to accelerate natural regrowth, applicable to sites and landscapes with nearby natural forests that may serve as seed sources;
- Restoration plantings using large number of species from later successional stages, useful for sites lacking nearby natural forest seed sources and/or to promote desired forest structure and species composition;
- Tree plantation mixtures of native species;
- Tree plantation used as a nurse crop with under-plantings of native species not otherwise able to establish at the site;
- Tree plantation monoculture of native tree species; and
- Tree plantation monoculture of non-invasive exotic species.

To optimize biodiversity conservation and enhance the provision of forest ecosystem services, restoration efforts should be planned at the landscape level (Maginnis & Jackson, 2003; McGuire, 2014).

Governments can effectively support forest ecosystem restoration by providing financial and policy support for development of planted forests on previously degraded lands. For example, the central government of the Republic of Korea worked in close collaboration with communities and succeeded in increasing the country's forest area from approximately 35% to 65% between 1955 and 1980. Their approach included a combination of economic incentives and policy coordination, particularly between the forestry and energy sectors to replace firewood with fossil fuels, a process assisted by rural-urban migration (Bae *et al.*, 2012; Park & Youn, 2017) (see also Section 6.4.1 on demographic changes and restoration). By enhancing the profitability of a forest-based economy - through commercialization of timber and non-timber forest products, shaded crops and ecotourism - some governments have contributed to forest conservation efforts while enhancing their benefits to people (Calvo-Alvarado *et al.*, 2009; Chazdon *et al.*, 2009). Livelihood improvements in rural areas that facilitate the transition from firewood to coal or electricity can reduce forest degradation, thereby contributing to land restoration (Dube *et al.*, 2014; Sugiyama & Yamada, 2015).

Responses to forest fire

Fire is most commonly viewed as a driver of forest degradation, but it is also used as a management tool in forest and grassland ecosystem management, particularly by local and indigenous communities (Parrotta & Trosper, 2012) (also see Chapter 3, Section 3.4.6 and Chapter 4, Section 4.2.6.5). For example, the utilization of traditional fire management practices in northern Australia have been shown to yield multiple benefits, not only for the environment to reduce degradation and assist restoration by making landscapes less prone to large wildfires, but also for traditional people (Legge *et al.*, 2011; Russell-Smith *et al.*, 2003; Vigilante *et al.*, 2004).

Two complementary approaches to fire management are commonly used, namely integrated fire management and community-based fire management (FAO, 2011). Integrated fire management focuses on addressing underlying causes for long-term and sustainable solutions, incorporating the five essential elements (research, risk reduction, readiness, response and recovery) and thus integrating all activities related to fire management (FAO, 2011).

Community-based fire management includes the integration of science and fire management approaches with socio-economic elements, at multiple levels, and provides a comprehensive approach to address fire issues that considers biological, environmental, cultural, social, economic and political interactions (Myers, 2006). It involves local-scale fire management, community and volunteer involvement in fire management across private and public lands (FAO, 2011).

While fire suppression is often cost effective for containing small-scale fires, such an approach can increase the future risk of much more damaging fires, especially in forests adapted to low to moderate intensity fire regimes (Stephens *et al.*, 2013). Managing forests for other values will be futile in the long term without managing forest for long-term fire risks and resilience (Jones *et al.*, 2016; Stephens *et al.*, 2013; Tempel *et al.*, 2015).

Box 6.3 Restoration of the Brazilian Atlantic Rain Forest

The Atlantic forest is among the top five global biodiversity hotspots (Laurance, 2009), providing a range of ecosystem services including drinking water for more than 60% of Brazil’s population. However, more than 88% of the original forest has disappeared, largely due to deforestation and agriculture (Pinto *et al.*, 2014), making it one of the highest priority regions for restoration in the world.

The Atlantic Forest Restoration Pact, initiated in 2009, is a regional, multi-stakeholder platform formed by NGOs, research institutions, the private sector and government agencies to coordinate efforts and objectives for restoration (Brancalion *et al.*, 2016; Melo *et al.*, 2013). It links key stakeholders for knowledge sharing and connects those offering or requesting sites for restoration, as well as inputs and technical assistance. The Pact aims to facilitate and implement restoration projects across 17 Brazilian states. It manages both public funds allocated by government budgets and ODA as well as private funds obtained through payments for ecosystem services, offset schemes for Brazilian infrastructure mitigation, water user fees, compensation payments for restoration, grants and microloans for establishing alternative sources of income (Sewell *et al.*, 2016).

The Pact aims to make ecosystem restoration an economic activity - generating opportunities for business, employment and income for local communities, especially in less developed areas. Under the Pact tens of thousands of hectares of forest areas have already been restored, with a long-term target of restoring 15 million ha out of the total Atlantic Forest area of 132 million ha. Restoration goals include: conserving forest biodiversity and enhancing delivery of ecosystem services; reconnecting isolated forest fragments; and re-establishing forests to promote sustainable harvest of timber and non-timber products. A variety of active and passive restoration approaches and methods are being used to conserve small- and medium- sized, privately-owned fragments and restore small areas around protected zones to improve the connectivity of landscapes (Holl, 2017; Pinto *et al.*, 2014; Rodrigues *et al.*, 2011).

6.3.1.3 Responses to rangeland degradation

An estimated 73% of the world's 3.4 billion ha of rangeland is affected by degradation of soils and vegetation (WOCAT, 2009) (see also Chapter 3, Section 3.3.1 and Chapter 4, Section 4.3.2). Rangeland degradation and species loss is mainly caused by overstocking of livestock combined with poor grazing management by nomadic pastoralists and smallholder farmers (e.g., Bestelmeyer *et al.*, 2011).

Strategies to improve grazing land management have been applied at different spatial scales, from global transboundary regional planning and implementation – through governmental control of stocking rates, livestock types and water allocation – to local approaches involving rotation of pastures, controlled burning, fencing and pasture development through replanting, intercropping and removal of woody plants (Latawiec *et al.*, 2017; Reid & Swiderska, 2008). In addition, several indigenous pastoral projects indicated that grazing management systems can also be achieved. Successful strategies include tribal and community coordination and cooperation, integrated and sustainable land use (Haregeweyn *et al.*, 2012; Kong *et al.*, 2014), and hunting to mitigate overgrazing by wild livestock (Gibson & Marks, 1995).

Developing and implementing grazing management plans is an efficient response to avoid and reduce rangeland degradation in particularly sensitive parts of the landscape (e.g., slopes, water points, riparian strips) and for soil and water conservation. Key considerations for effective rangeland management planning include:

- *Land condition* - rainfall and natural runoff pattern, soil fertility and health and pasture biodiversity (both feedstock and livestock) (Bartley *et al.*, 2010);
- *Anthropogenic community structure* - development level of agriculture and municipal infrastructures, level of governmental regulatory capabilities, indigenous and local practices, local stakeholders and land tenure rights (Undersander *et al.*, 2014);
- *Grazing level and distribution* - pasture utilization, stocking rate influence, grazing system and livestock type (Undersander *et al.*, 2014); and
- *Diet gateway* - conversion of pasture into animal product, through herbage quality, legume content and pasture species (Fisheries & Forestry, 2013).

Implementation of grazing land management strategies may involve a combination of existing tools appropriate for specific grazing and pasture management scenarios (Lambin *et al.*, 2014). Effective tools for different pasture types typically consists of:

- *Spatial information monitoring* - which can utilize national and regional governmental data archives and remote sensing resources to assess key features, such as property mapping, paddock size, land types, land use and more. Spatial monitoring is an effective tool for regions that are prone to soil erosion and rangeland degradation, due to overgrazing along slopes, particularly in drier regions (Bartley *et al.*, 2010). Utilization of such available databases, and temporal and spatial analyses, can indicate trends such as vegetation cover, desertification, land uses and other physical parameters essential for rangeland management (Prince, 2016).

- *Land capability and condition assessments* - through field surveys when databases are insufficient. These should include key features, such as specific land capability, land conditions, means of sustaining and improving land conditions, current carrying capacity, potential carrying capacity and more (see Chapter 4, Section 4.3.2).

- *Land resource and use characterization* - including grazing and pasture development parameters, namely land type, fencing, water points, frontages, wetland management, biodiversity conservation measures, legislative responsibilities, tree-grass balance management, wildfire prevention and fire control.

- *Grazing pressure management* - involving economic and regulatory means to control stocking rates, timing livestock growth, herd sizes, grazing management zones and maintain more uniform pasture pressure (Bartley *et al.*, 2010). Effective application of such tools is often difficult as it typically requires coordination and regulation among authorities and other key stakeholders (i.e., pastoralists and farmers) (Latawiec *et al.*, 2017).

- *Pasture and forage crop, enhancement* - through development and management of pasture and forage crops, silvopastoral practices, prevention of sown pasture degradation and development of monitoring tools. Although most pasture and forage crops are grown in cultivated areas, if grazing exhausts natural rangeland, replanting using rangeland vegetation enhancement techniques is needed to preserve their fertility (Undersander *et al.*, 2014).

- *Weed and pest management* - through monitoring, management and control of invasive plants, insects and other pests. The incorporation of indigenous peoples’ traditional knowledge and rangeland management practices provide additional approaches for effective weed and pest management (Ens *et al.*, 2015).

- *Evaluation of social and economic potential* - for the adoption of more sustainable pasture management practices, including land tenure types and cultivation systems (e.g., farms, nomadic, rural settlements), as well as cultural aspects such as cattle sanctity (India), the integration of the land uses in local traditions and evaluations of the magnitude and effectiveness of governmental actions (e.g., taxation, law-enforcement) for the relevant community (Latawiec *et al.*, 2017; Reed *et al.*, 2015).

Finally, the assessment of grazing land management strategies should consider effects of each strategy on financial and technological capabilities of local farmers and their economic benefits, the level of local authorities' regulatory management capabilities and, above all, effects of the strategy on physical parameters of the grazing land (Weber & Horst, 2011).

Box 6.4 Grazing control and desertification in arid zones (Egypt-Israel-Jordan)

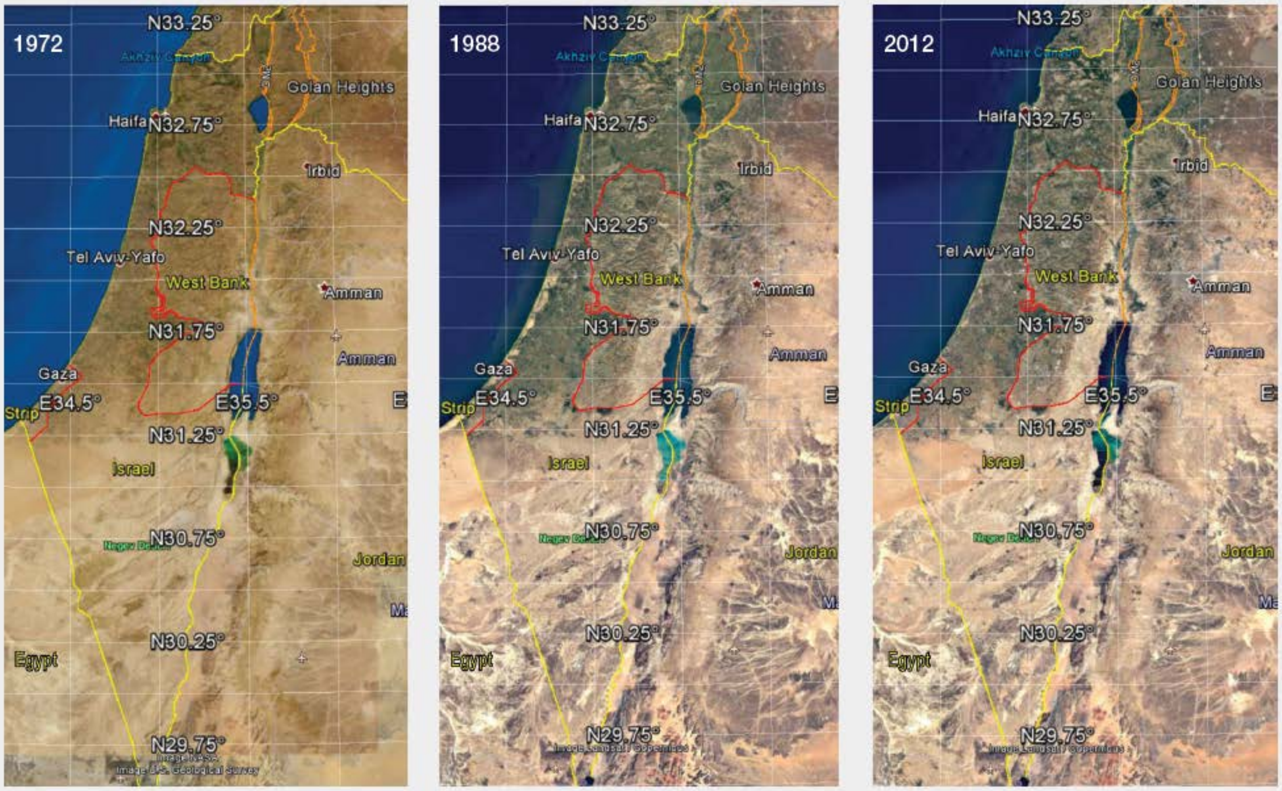
Throughout history, the cultivation of camels, sheep and goats played a major role in Eastern Mediterranean economies, through the sale of their meat, dairy or hair and wool products. During the last couple of centuries most herds were driven by tribes of pastoral nomads, known as Bedouin (Bienkowski & van der Steen, 2001). Until the 20th century, by permit of the Ottoman empire these nomads had access to transboundary traditional pastoral resources; but since the early 20th century - through a series of international treaties and the establishment of new States - several tribes were restricted to the North-Western Sinai Desert. This pasture land restriction gradually degraded the rangeland owing to chronic overgrazing (Meir & Tsoar, 1996), manifested in the albedo difference between both sides of the Egypt-Israel border (Figure 6.4).

Once natural pasture carrying capacity is exceeded by livestock demands (see also Chapter 4, Section 4.3.2), rangeland development actions are required. The dynamic nature of the process is well demonstrated by the temporal shift in vegetation density across the Egypt-Israel border (Warren, 2002). While vegetation density was similar during the years when the border was open (mainly during the 1970s) (Figure 6.4), since 1982 the closed border has been a barrier to grazing herds and, as a result, the vegetation density increased on the Israeli side compared to the Egyptian side of the border (Seifan, 2009). The desert dunes' stability, owing to the development of soil crusts, contributes to landscape resilience against natural phenomena such as large-scale dust storms (Figure 6.4) (Kidron *et al.*, 2017).

While along the Egyptian-Israeli border the disruption of grazing pastoral practice had led to deterioration of natural and human habitats, along the Israeli-Jordanian border (the Jordan Rift Valley) the Jordan River floodplain supplied sufficient rangeland resources, preventing the pasture over-burden. In addition, the Jordan Valley is one of the first locations with documented human settlements and probably the first evidence of livestock farming (Lu *et al.*, 2017; Martínez-Navarro *et al.*, 2012).

One of the differences between the nomadic pastoral grazing typical to the Egypt-Israel border, and the year-round livestock husbandry in pastoral farm and village setting, is better management of rangeland resources. The stationary nature of the Jordan valley shepherds community prevented the overgrazing of pasture land. The ability of the stationary pastoral rural communities to maintain systematic or semi-systematic grazing and rangeland development regimes improved their resilience to climate change and political issues.

Figure 6 4 Comparative satellite view (Google Earth) of the Egypt-Israel-Jordan borderlines in 1972, 1988 and 2012 respectively.



6.3.1.4 Responses to urban land degradation

Amongst the most severe forms of land transformation, urbanization results in land degradation both within and outside of urban areas - through its direct impacts on lands within established and expanding cities and suburban areas and the extension of their ecological footprints beyond their boundaries - leading to impacts on a wide range of ecosystems in surrounding landscapes.

Figure 6 5 Urban and suburban landscapes in Medellin, Colombia: the planned city A, the informal city B, and the quarries C. Source: Medellin Planning Department, 2006.

Responses to reduce these impacts include those that seek to: maintain or improve the health and sustainability of ecosystems within their zones of influence; the health, well-being and safety of urban dweller; and to improve the urban fabric.

Preventive responses to urban land degradation

Responses to urban land degradation fall into two categories, “grey” and “green” responses. Regarding “grey” responses, the New Urban Agenda (http://habitat3.org/the-new-urban-agenda/) incorporates sustainability as its third principle and 56 sustainable urban development commitments (Caprotti *et al.*, 2017; Watson, 2016). Out of these commitments, 3 contain responses to ecological-rural functionality; 3 to water management, mainly as an economic resource; 3 to the green public space, with emphasis on its social function and resilience factor; and 43 to technical and political responses to social and economic problems. Specific “grey” responses to achieve these commitments include urban planning and design instruments to support sustainable land-use management and natural resources by enhancing resource efficiency, urban resilience and environmental sustainability (amongst others).

Figure 6 6 **Aerial view of the rooftop garden of a multi-storey carpark in Singapore.**

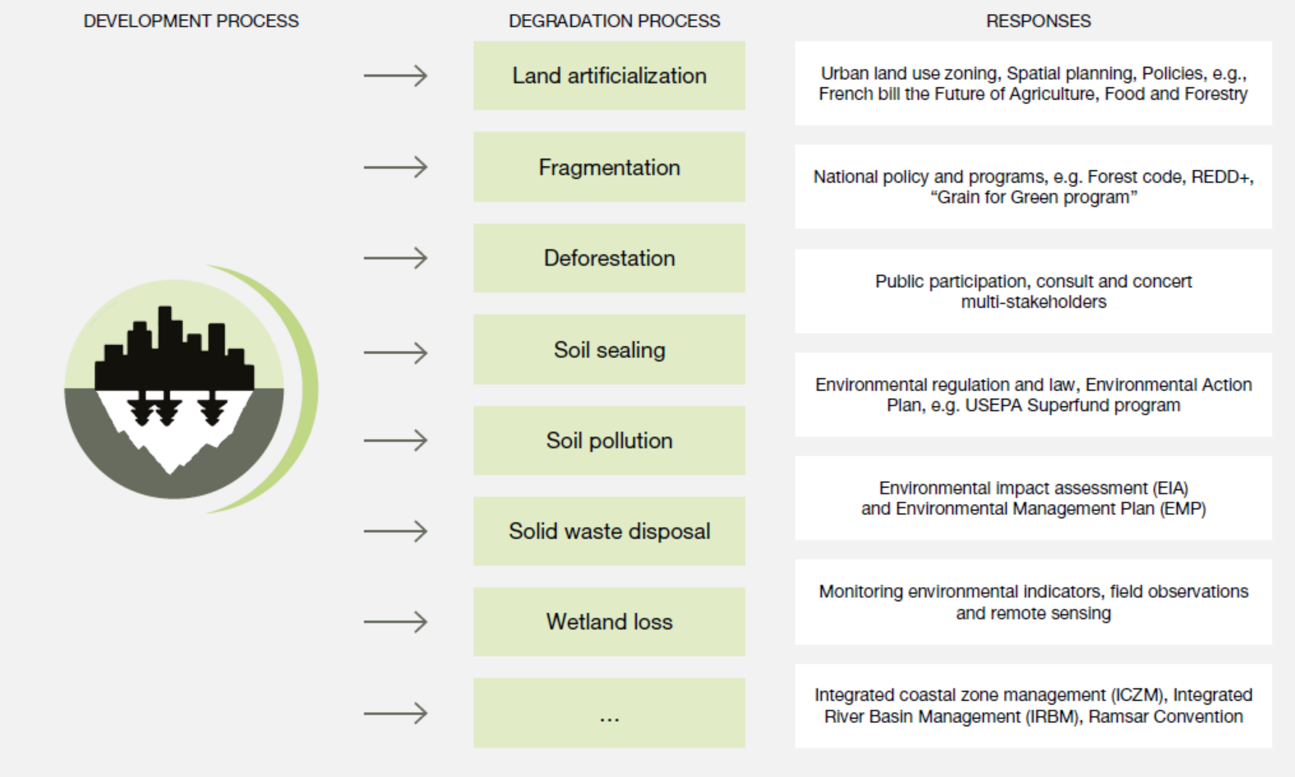
Among the many techniques used to create "green infrastructure" in urban areas, rooftop gardens are one. Photo: Jimmy Tan licensed under CC BY 2.0.

On green responses, the Cities and Biodiversity Outlook of the CBD (2012) highlights opportunities to reduce urban land degradation by utilizing the linkages between urbanization, biodiversity and ecosystem services. Response measures include developing and enhancing existing ecological infrastructure of cities (i.e., parks, gardens, open spaces, water catchment areas), and their ecosystems and biodiversity. It emphasizes the importance of valuation and explicit inclusion of urban biodiversity (also known as natural capital) as a determining factor in the planning and management of cities. Maintaining functioning urban ecosystems not only addresses the problems associated with urban land degradation, but can also significantly enhance human health and well-being as well as contribute to climate change mitigation and adaptation (CBD, 2012). Sustainable urban development includes managing and designing for biodiversity conservation (Aronson *et al.*, 2017; Müller & Kamada, 2011). "Green infrastructure" is widely proposed and, in some places, widely implemented (Hostetler *et al.*, 2011) - using techniques such as planting vegetation on roofs ("green roofs", Figure 6.6), rain gardens, paving with materials that allow infiltration of precipitation protected natural open space, planting native plant species and retaining corridors of non-developed land. These provide habitat for native plants, insects, animals and soil biota (McKinney, 2002).

Restoration practices in urban and built environments

Specific responses to urban land degradation depend on the main issues or processes that need to be addressed, such as soil contamination and soil instability, water contamination, invasive species impact, heat island effects and flooding risk from altered catchment hydrology (Figure 6.7).

Figure 6 7 Land degradation and restoration related policy challenges, goals, instruments and tools and methodologies.



In-built environments restoration practices are closely related to erosion and sediment control during the construction phase to prevent pollution of streams and rivers. Short-term erosion control practices are generally followed by establishment of vegetation for long-term erosion control. Bio-technical stabilization uses structural and biological elements to avoid severe erosion (Buchholz & Madary, 2016; Myers, 1993). These may include non-vegetated structures, such as retaining walls, or soil bio-engineering (the use of plants in bio-technical slope stabilization as the main structural component).

Soil contamination, a process by which the chemical properties of soils are changed, occurs mainly from industrial development in cities through factories releasing wastes that contain heavy metals, organic pollutants and other contaminants to surrounding areas (see also Chapter 4, Section 4.2.4.2). While soil contamination is rarely reversible (Siebielec *et al.*, 2010), it is sometimes possible to use brownfields to produce non-alimentary crops for energy or textiles. In this way, the past industrial soils recover new functions and their imperviousness is reduced (Huot *et al.*, 2015). However, the costs associated with remediation of past pollution in brownfields can be an obstacle to their re-use (EC, 2012). In such cases, financial compensation from the past polluters or the future developers is an approach to restore or improve the function of those soils.

Soil sealing is prevalent where materials such as asphalt, concrete and stone are used to construct buildings, roads, parking lots and other urban infrastructure (see also Chapter 3, Section 3.3.6). Sealing reduces or completely prevents natural soil functions and ecosystem services on the area concerned, including regulation of hydrology and temperature regimes in urban areas (EEA, 2011). Measures to compensate for soil sealing include: (i) re-use of topsoil excavated during building construction and infrastructure development in other urban locations; (ii) de-sealing of certain areas (soil recovery) to compensate for sealing elsewhere; (iii) use of eco-accounts and trading development certificates; and (iv) collection of fees on soil sealing activities, to be used for soil protection or other environmental purposes (EC, 2012). Some financial approaches can also help restore contaminated land, such as the “Superfund” programme of the US Federal government, which has funded decontamination of sites contaminated with hazardous substances and pollutants since 1980 (Acton, 1989; Daley & Layton, 2004).

Increasing urban populations and impervious surfaces intensify heat island effects in cities (also see Chapter 4, Section 4.3.10). Responses to reduce heat island effects include developing or maintaining “green infrastructure,” such as urban open spaces and urban forestry initiatives that include tree planting and management (Gill *et al.*, 2007; Miller *et al.*, 2015; Roy *et al.*, 2012). The importance of street trees, urban forests and their multiple benefits is increasingly recognized by urban planners, municipal governments and citizens worldwide (Pandit & Laband, 2010; Pandit *et al.*, 2014) and many cities have made urban greening a priority. Many urban greening tools have been developed, such as the Berlin Biotope Area Factor, the Malmo Green Factor (Hagen & Stiles, 2010), the Seattle Green Factor (Giordano *et al.*, 2017), the Poland Ratio of Biologically Vital Area (Szulczewska *et al.*, 2014) and a public open space planning tool (Bull *et al.*, 2013).

Water system degradation can threaten many cities. Filling rivers and lakes to develop real estate or infrastructure, for example, can alter flow regimes and increase flood risk. As this process is largely irreversible and often very costly, better land-use planning is essential (Hall *et al.*, 2014; Shen, 2015). In addition, water contamination and pollution from industrial wastewater or domestic sewage can have severe impacts on environmental quality and its related services. Water contamination can often be handled as part of brownfields projects - although law enforcement, filtration of wastewater before discharge and education are also effective ways to alleviate water pollution (Buchholz & Madary, 2016; Hall *et al.*, 2014; Kjellstrom *et al.*, 2006; Myers, 1993; Shen, 2015).

Methods to respond to altered catchment hydrology include river channel restoration and management of impervious surfaces through the reduction and adoption of technologies to improve infiltration in parking lots and transportation corridors, and installation of rain gardens. Urban forestry can also aid in hydrologic management through canopy interception. New soil media for cities can also be developed to create soil from waste and thus avoid agricultural soil consumption (Rokia *et al.*, 2014). Quantifying the economic value of green infrastructure can also promote restoration activities or maintenance of green infrastructure in urban areas. For example, Polyakov *et al.* (2017) report that restoration practices aiming to convert a “conventional drain” into a “living stream” in Perth simultaneously increased property price (private economic benefit) and the ecological outcomes such as better habitats for plants and animals (a public benefit), thus providing additional incentives for urban residents or the local authorities to restore degraded urban drains.

There are no panaceas for the urban land degradation issues and processes, and governments in different contexts must consider their financial, technological or political capacities to appropriate select restoration responses. Table 6.5 gives an overview of the effectiveness of different responses to halt or restore degraded urban land.

6.3.1.5 Responses to wetland degradation

Worldwide, the extent of wetlands is estimated to have declined by 64-71% in the 20th century (Davidson, 2014; Gardner *et al.*, 2015; Hu *et al.*, 2017). For several wetland types, such as tropical and subtropical mangroves, recent losses have been as high as 35% since 1980, with a current global area rate of loss of between 0.7 and 3% yr^{-1} (Pendleton *et al.*, 2012). The loss of these freshwater and coastal ecosystems have been estimated to result in more than $20 trillion in annual losses of ecosystem services (Costanza *et al.*, 2014). Consequently, the status of wetland-dependent species remains alarming. The Freshwater Living Plant Index has declined by 76% between 1970 and 2010 (Gardner *et al.*, 2015) (see also Chapter 4, Section 4.2.5.2).

The "wise use" approach of the Ramsar Convention is considered globally as a central tenet of wetland management (Maltby, 2009). Adopted by 169 countries, it builds on the premise that restricting wetland loss and degradation requires the incorporation of linkages between people and their surrounding wetlands (Finlayson *et al.*, 2011; Finlayson, 2012)). The removal of the stressors or pressures that limit the wise use of wetlands (or adversely affect their ecology) is considered the best practice response option for addressing wetland loss and degradation. The Convention has also developed a suite of guidance to support wetland restoration, including a specific resolution on avoiding, mitigating and compensating for wetland losses (Ramsar, 2012).

Ecological restoration of degraded wetlands is a global priority for addressing and reconciling conservation and sustainable development goals (Alexander & McInnes, 2012; Aronson & Alexander, 2013). Successful restoration of wetlands results in self-sustaining and resilient ecosystems dominated by native species (in characteristic assemblages and functional groups) that are part of a wider landscape in which the drivers of wetland degradation have been reduced or eliminated (SERI, 2004).

The most commonly-used responses to restore wetlands include recovering the hydrological dynamics, revegetating, removing invasive species and managing soil profiles. Restoring the hydrological dynamics usually involves either reconnecting the wetland to the tides or river flow (flow re-establishment), or reconstructing the wetlands topography (through surface modification). There has been considerable effort directed toward wetland restoration in some regions. Until 2014, the Wetland Reserve Program in the USA was a voluntary programme for landowners to protect, restore and enhance wetlands, resulting in nearly 1 million ha enrolled (USDA, 2014). In 2014, the first year of the Agricultural Conservation Easement Program, which replaced the Wetland Reserve Program, 168 wetland projects were supported covering about 15,000 ha (Smith *et al.*, 2015; USDA, 2014).

A recent meta-analysis of global wetland restoration (Moreno-Mateos *et al.*, 2012) - involving over 600 restored wetlands - found that those where either surface modification or flow re-establishment were used followed similar recovery trajectories, regardless of whether they were revegetated or not. It also found potential detrimental effects of revegetation measures on the recovery of the plant assemblage in cold climates and in wetlands restored in agricultural areas. This study also concluded that remediation efforts had failed to fully recover wetlands over the first 50 to 100 years (Moreno-Mateos *et al.*, 2012) with recovery of biodiversity and functions increasing to about 75% of the level in undisturbed reference wetlands after that time. Compared to degraded wetlands, however, restoration increased some ecosystem services and biodiversity, but the recovery was highly context dependent (Meli *et al.*, 2014). A study focused on recovery from eutrophication showed that lakes and coastal marine areas achieved a recovery of baseline conditions by an average of 34% and 24%, respectively, decades after the cessation or partial reduction of nutrients (McCrackin *et al.*, 2017)

These results indicate that there is an urgent need to understand how wetlands recover over the long term (20 years or longer) and what actions are most appropriate to restore them. As commonly used indicators of wetland recovery after restoration tend to be very simplistic (e.g., carbon storage), and do not encapsulate the complexity of ecosystems, there is a need to develop and use indicators to evaluate interactions among organisms and with the abiotic environment, for example, through measuring and recovering ecological networks (Anker *et al.*, 2013) with major roles in ecosystem functioning (e.g. decomposition, pollination, dispersal).

In recent decades, efforts to restore coastal wetlands (mangroves, tidal marshes and seagrass beds) have been made in many parts of the world to compensate or mitigate losses resulting from management activities (Hogarth, 2007; Lewis III, 2000; Orth *et al.*, 2012). Efforts have also been made to restore their capacity to provide ecosystem services such as buffering against extreme events (Marois & Mitsch, 2015). Methods for restoring such wetlands may include: active restoration measures (reshaping topography, channelling water flow, mangrove planting and control of invasive species); passive restoration approaches to enhance ecohydrological processes and improve hydrological connectivity; or in certain cases, the creation of wetlands (Zhao *et al.*, 2016). Complementary programmes in coastal planning (based on integrated coastal zone management approaches), marine spatial planning and marine protected areas have been established to address spatial issues. Recent research on economic efficiency of nature-based solutions has shown promising results. For example, maintenance of salt-marshes and mangroves have been observed to be two to five times cheaper than a submerged breakwater for wave heights up to half a metre and, within their limits, become more cost-effective at greater depths. Nature-based defence projects also report benefits ranging from reductions in storm damage to reductions in coastal structure costs (Narayan *et al.*, 2016).

Peatlands form a major proportion of total wetland area in the world and account for a major proportion of global soil carbon stores. Degradation of peatlands contributes significantly to global emissions of greenhouse gases (for example see Hooijer *et al.*, 2010). A range of measures for improving habitat conditions (e.g., regulating nutrient availability, base saturation, introduction of native species), peatland hydrology (e.g., increasing natural rewetting, damming and infilling of ditches, and reducing evapotranspiration) andcatchment management practices have been used in different parts of the world (Andersen *et al.*, 2017; Chimner *et al.*, 2017; Graham *et al.*, 2017).

Wetland creation and rewetting of drained soils are common activities in response to significant wetland loss and degradation on a global scale (Mitsch *et al.*, 1998). Wetland creation – where lands are artificially inundated and utilize natural processes to restore vegetation, soils and their associated microbial assemblages (Aber *et al.*, 2012) – is carried out for various purposes such as water-quality enhancement (treatment of wastewater, stormwater, acid mine drainage, agricultural runoff), flood minimization and habitat replacement (Mitsch *et al.*, 1998). Wetlands created for treating wastewater have been used with good results in many countries, including Cuba, China, USA and Thailand (IPCC, 2014; Land *et al.*, 2016; Vymazal, 2011). Recent advances in the design and operation of these wetlands have greatly increased contaminant removal efficiencies (Wu *et al.*, 2015). Wetlands may also be created unintentionally when the regulation of river flows (i.e., installation of large dams) results in periodic inundation of lands that previously did not experience inundation (Yang *et al.*, 2012).

Addressing the indirect drivers of change often requires policy-level changes, in the form of national policies on wetlands, or mainstreaming the full range of wetland ecosystem services and biodiversity values within sectoral policy and decision-making. Treating wetlands as natural water infrastructure can help meet a wide range of policy objectives such as water and food security and climate change adaptation (Pittock *et al.*, 2015; Russi *et al.*, 2013). Similar mainstreaming approaches, as wetlands as settings for human health (Horwitz & Finlayson, 2011), or wetlands restoration within nature-based approaches for disaster risk reduction (Monty *et al.*, 2016; Renaud *et al.*, 2016), are increasingly gaining traction in policy and decision-making. Considering their role in larger river basins and coastal zones, integrated land-use planning and management of wetlands can ensure that wetlands and their benefits are sustained in the long run (Maltby & Acreman, 2011; Ramsar, 2012). Enhanced understanding of multiple values of wetlands can greatly strengthen stakeholder engagement in mainstreaming wetland restoration agenda and actions (Kumar *et al.*, 2017; Russi *et al.*, 2013).

LAND USE OR DEGRADATION DRIVER	RESPONSE OPTIONS	NATURE OF RESPONSE	RESPONSE EVALUATION CRITERIA AND EFFECTIVENESS RANKING (COLOUR-CODED)					
		Avoid (Av), Reduce (Rd), Reverse (Rv)	Economic feasibility	Social accepta-bility	Environ-mental desirability	Cultural accepta-bility	Technical feasibility	Political accepta-bility
CROPLAND MANAGEMENT	Conservation agriculture	Av, Rd						
	Agroforestry	Av, Rd, Rv						
	Integrated crop, livestock and forestry systems	Av, Rd, Rv						
	Enhanced plant genetics	Rd						
	Agroecology	Av, Rd, Rv						
	Landscape approach	Av, Rd, Rv						
FOREST LAND MANAGEMENT	Agroforestry	Av, Rd, Rv						
	Protected areas	Av						
	Sustainable forest management	Av, Rd						
	Reduced impact logging	Rd						
	Landscape approach	Av, Rd, Rv						
	Restoration (active and passive)	Rv						
RANGELAND MANAGEMENT	Grazing management	Av, Rd, Rv						
	Pasture rotation	Av, Rd						
	Controlled burning	Av						
	Fencing	Av, Rd						
	Replanting	Rv						
	Intercropping	Av, Rd, Rv						
	Weed and pest control	Rd, Rv						
URBAN LAND MANAGEMENT	Green space management	Av, Rd						
	Street tree planting	Rv						
	Brownfield restoration	Rv						
	Removal of invasive species	Rv						
	Green infrastructure development	Av, Rd						
	Amelioration of contaminated soils and sealed soils	Rv						
	Sewage and wastewater treatment	Av, Rd						
	River channel/beach site restoration	Rv						

LAND USE OR DEGRADATION DRIVER	RESPONSE OPTIONS	NATURE OF RESPONSE	RESPONSE EVALUATION CRITERIA AND EFFECTIVENESS RANKING (COLOUR-CODED)					
		Avoid (Av), Reduce (Rd), Reverse (Rv)	Economic feasibility	Social accepta-bility	Environ-mental desirability	Cultural accepta-bility	Technical feasibility	Political accepta-bility
WETLAND MANAGEMENT	Protected areas	Av	Moderate to high effectiveness	High effectiveness	High effectiveness	High effectiveness	High effectiveness	Variable effectiveness (low to high)
	Control of point pollution sources	Av, Rd	High effectiveness	Moderate effectiveness	High effectiveness	Low to moderate effectiveness	Moderate to high effectiveness	Moderate effectiveness
	Control of non-point pollution sources	Av, Rd	Low effectiveness	Low effectiveness	High effectiveness	Low to moderate effectiveness	Low to moderate effectiveness	Low effectiveness
	Passive measures to allow natural recovery (e.g., control of human/ livestock pressures)	Rd, Rv	Low effectiveness	Moderate to high effectiveness	High effectiveness	High effectiveness	High effectiveness	High effectiveness
	Active restoration measures (e.g., reshaping topography and hydrology, revegetation, invasion control)	Rd, Rv	Moderate to high effectiveness	Variable effectiveness (low to high)	High effectiveness	High effectiveness	High effectiveness	Variable effectiveness (low to high)
	Constructed wetlands	Rv	Low effectiveness	Moderate effectiveness	Moderate to high effectiveness	Low to moderate effectiveness	Moderate effectiveness	Moderate effectiveness

EFFECTIVENESS RANKING OF RESPONSE OPTIONS

High effectiveness; Moderate to high effectiveness; Moderate effectiveness; Variable effectiveness (low to high); Low to moderate effectiveness; Low effectiveness

6.3.2 Assessment of responses to selected direct drivers and impacts

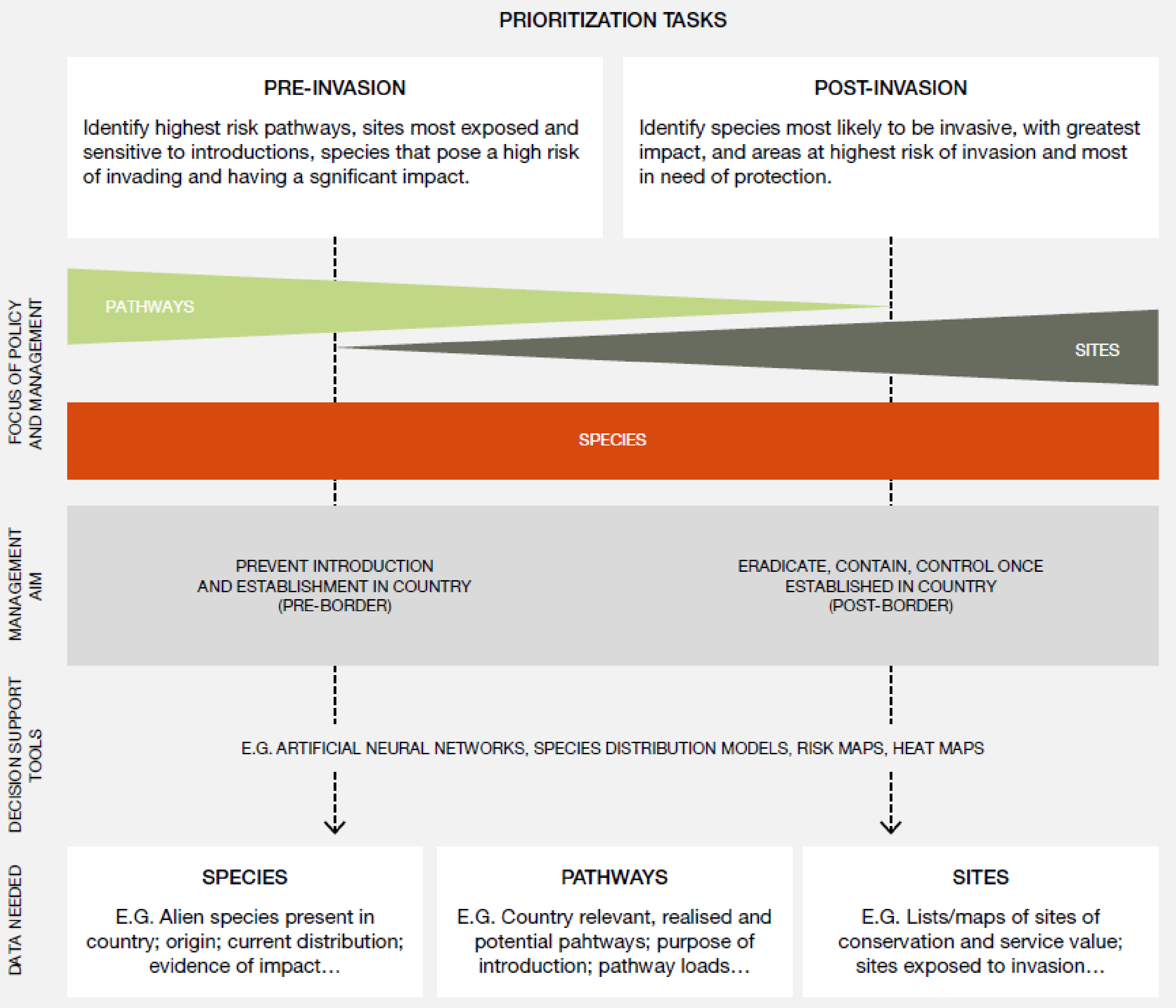
6.3.2.1 Responses to invasive species

Responses to invasion include institutional arrangements, policy and governance tools, as well as management strategies that interact in various ways based on spatial context. Managing invasive species is complex and challenging, primarily because of the dynamic nature of invasion processes, variable effects on different land-use systems (e.g., urban land versus agricultural land), and varying perceptions among stakeholders on ecosystem services or disservices generated by invasive species (Gaertner *et al.*, 2017). Typically, the costs of invasive alien species management strategies exceed available resources, particularly where socio-economic impacts of invasion disproportionately affect less advantaged social groups (Rai *et al.*, 2012; Shackleton *et al.*, 2011). Nevertheless, eradication or control of invasive species is often one of the aims of restoration (D'Antonio *et al.*, 2016).

Local communities in urban areas have detailed knowledge of the impacts of invasive species on biodiversity, their local environment and their values and perceptions of their local environment. To establish approaches to the management and restoration of invaded urban landscapes, engaging with local communities - along with experts in both restoration and invasion ecology, but led by local knowledge and those who continue to live in those landscape - provides innovative approaches and frameworks to manage and restore urban landscapes degraded by invasive species(Fisher, 2011; Fisher, 2016; Gaertner *et al.*, 2012). Local communities understand the importance of managing the landscape and the ecosystem as a whole. Invasive species management using a holistic ecosystem approach and driven by local communities, in differing urban landscapes - including coastal, woodlands, wetlands, rivers and estuaries - has proven to be highly successful in restoring functioning ecosystems. Long-term outcomes include restored urban environments resilient to changing climates with focus on the removal of all invasive species and their replacement with indigenous species (Fisher, 2011; Fisher, 2016; Gaertner *et al.*, 2012). Such an ecosystem approach to tackling invasive species has been adopted by the Sri Lankan Government at the national level and incorporated across policy, strategy, action planning, management and restoration (Fisher, 2015; Sri Lanka National Invasive Alien Species Committee, 2015).

The implementation of practical strategies usually occurs at local and national levels, and involves three successive steps - prevention, eradication and control (see Figure 6.8). In general, the most effective strategy is to prevent introductions of potentially invasive species before their establishment (Allendorf & Lundquist, 2003; Hulme, 2006; Leung *et al.*, 2002); due to the high cost of managing invasive species through eradication and control. *Preventive* measures focus on identifying and monitoring common biological invasion pathways (e.g., intentional and accidental introductions). Trade globalization and expanded transport networks have led to pathway risk assessments becoming the frontline in the prevention of invasions (Hulme, 2009). Pathway risk assessment relies heavily on spatial data, with risk maps that highlighting hotspots of invasion likelihood being a common product (Buckley, 2008). Linked to this is the identification of the invaders themselves and measuring their impacts (Blackburn *et al.*, 2014). This is where tools such as the Global Invasive Species Database (GISD) of the IUCN are useful. Many countries list prohibited species (e.g., categories of invasive alien species) and undertake awareness campaigns to educate the public about the threat to biodiversity posed by invasive alien species. The second component to prevention is interception (Boy & Witt, 2013), including the establishment of environmental biosecurity departments to carry out activities such as search and seizure procedures at first points of entry, as well as quarantine measures to block or restrict incursions. Examples of such bodies are the Australian Government's Department of Agriculture and Water Resources and the Animal and Plant Health Inspection Service in the USA. Such quarantine measures are, however, not necessarily feasible or effective in resource- and/or infrastructure-constrained settings.

Figure 6 8 Prioritization to support cost-effective allocation of resources is part of decision-making at nearly every stage of the invasion process, from preventing introduction of invasive alien species, to preventing their spread, to eradication or containment. Source: McGeoch *et al.* (2016).



Eradication is the next option in the practical response continuum and entails the systematic elimination of the invading species until it can be ascertained that no individuals, viable seeds or other propagules remain in an area (Boy & Witt, 2013). Eradication has been achieved, notably in island settings, with substantially more examples of successful eradication of vertebrate species than plant species (Genovesi, 2005; Glen *et al.*, 2013; Keitt *et al.*, 2011). Social acceptability of invasive animal eradication is controversial due to ethical issues (Cowan *et al.*, 2011; Rejmánek & Pitcairn, 2002; Simberloff, 2009). Early detection and decisive action are crucial for success (Pluess *et al.*, 2012; Rejmánek & Pitcairn, 2002; Simberloff, 2009) as early warning and rapid response systems enhance prompt detection of new incursions and correct taxonomic identification of invaders, assessing related risks and ensuring immediate reporting of relevant information to the competent authorities (EEA, 2011). In South Africa, for example, the National Department of Environmental Affairs has collaborated with the South African National Biodiversity Institute in the implementation of the Early

Detection and Rapid Response programme (Ntshotsho *et al.*, 2015a). Similarly, the European Commission has proposed a formalized early warning mechanism in the EU Regulation on invasive alien species which came into effect in January 2015.

Control of established invaders is the last line of defence, with the primary goal being the reduction of abundance and density in order to minimize adverse impacts. Successful control depends more on commitment and sustained diligence than on the efficacy of specific tools themselves, as well as the adoption of an ecosystem-wide strategy rather than a focus on individual invaders (Mack *et al.*, 2000). For invasive plant species, integrated weed management, which involves a combination of measures (Adkins & Shabbir, 2014), may be effective for long-term control in cases where invasive plants are able to survive individual measures. Generally, four types of control measures are in use for invasive plants: mechanical and/or manual, cultural, biological, and chemical; but “control by use” has also been considered as a control measure.

Mechanical and/or manual control of invasive plant species are often labour intensive, but in countries where communities manage land, and affordable labour is available, manual control is feasible (Rai *et al.*, 2012). Activities like hand-pulling and hoeing are site specific, can be effective in loose and moist soils, and to control small infestations (Sheley *et al.*, 1998). Mowing is most effective for annuals and some perennials (Benefield *et al.*, 1999), success depends on its timing and frequency (Benefield *et al.*, 1999; Rai *et al.*, 2012).

Cultural practices include controlled grazing, prescribed burning, and physical manipulation of habitat. There are several examples of such practices, for instance: controlled grazing to control *Parthenium hysterphorus* and *Centaurea solstitialis* (Adkins & Shabbir, 2014; DiTomaso, 2000); manipulating shading by overstorey to hinder the growth of *Lantana camara* (Duggin & Gentle, 1998); and prescribed burning to control invasion of annual broadleaf and grass species (DiTomaso *et al.*, 2006; Keeley, 2006). Indigenous practices for responding to invasive species provide important opportunities for effective responses and vary across the globe and the landscape (Ens *et al.*, 2016; Ens *et al.*, 2010). However, considering that invasive plants are likely to become established in disturbed habitats, cultural practices do pose a risk of promoting their proliferation (Fine, 2002; Moore, 2000).

Biological control (or biocontrol) is a means for controlling pests such as insects, mites, weeds and plant diseases using these organisms’ natural enemies to reduce their abundance, rather than eradicate them (Charudattan & Dinoor, 2000; Ghosheh, 2005). Its effective implementation - based on extensive testing and validation for host-specificity to predict risk and minimize adverse environmental impacts (Delfosse, 2005; Messing & Wright, 2006) - is considered to be a cost-effective, long-term and self-sustaining control measure (Schlaepfer *et al.*, 2005).

Chemical control (use of biocides) is probably the most widely-adopted measure to control invasive plant and insect species. It is also the least desirable due to unintended adverse impacts on other non-target species in the surrounding environment and human health impacts (Giesy *et al.*, 2000; Khan & Law, 2005; Williams *et al.*, 2000). It is financially feasible under certain conditions such as high-value crops, at roadsides, public parks or on small areas (Adkins & Shabbir, 2014). Of concern is the growing global incidence of herbicide resistance in agricultural weeds (Heap, 2014; Preston, 2004). Herbicide resistance threatens to undermine control efforts and, consequently, underscores the need for integrated management (Kohli *et al.*, 2006; Shabbir *et al.*, 2013).

In terms of the effectiveness for controlling invasion of *Prosopis* spp., invasive species with global reach, mechanical and chemical measures are costlier than biological and “control by use” measures (van Wilgen *et* *al.*, 2012). But these latter control measures have been found less effective to reduce the invasions (FAO, 2006; Shackleton *et al.*, 2014). In Kenya and Ethiopia, *prosopis* has also been managed through “control by use” method (e.g., firewood, producing electricity for local use), but without any noticeable impacts on invasions (Zimmermann *et al.*, 2006). Biological control to manage prosopis has been found more effective in Australia with the use of four biological control agents: *Algarobius bottimeri, A. prosopis, Evippe* species, and *Prosopidopsylla flava* than in South Africa where three seed-feeding beetles: *A. prosopis, A. bottimeri and Neltumius arizonensis* were used (van Klinken, 2012; van Klinken *et al.*, 2003).

Box 6.5 The South African Working for Water programme

South Africa has a long history of problems with invasive alien plant species and management of biological invasions (Marais & Wannenburgh, 2008; Ntshotsho *et al.*, 2015a; Richardson & van Wilgen, 2004; van Wilgen *et al.*, 2002). These invasions pose a threat to human well-being by negatively impacting the provision of ecosystem services such as water and grazing (van Wilgen *et al.*, 2001). For example, it was estimated that the 1.5 million ha of land dominated by invasive alien plants were responsible for a total reduction of 1.44 million m^3/yr in mean annual runoff (van Wilgen *et al.*, 2012; Versfeld *et al.*, 1998). For a water-scarce country this is a substantial impact.

The Working for Water programme , arguably South Africa’s largest nationwide conservation project, was initiated in 1995 with the primary aim to clear invasive plant species in order to increase water supply (Marais & Wannenburgh, 2008; van Wilgen *et al.*, 2002) while generating employment for marginalized people (Ntshotsho *et al.*, 2015a). Government funding to the programme increased from an initial f R25 million/yr (approx. $1.7 million/yr) in 1995, to R1.28 billion/yr (approx. $88 million/yr) in 2013/14 (WfW historical expenditure, http://sites.google.com/site/wfwplanning).

Figure 6 9 Images of the Upper Berg River Dam site in 2006 (left) and in 2015 (right). Source: ©2016 Cres/Spot Image & Image ©DigitalGlobe.

Figure 6.9 Images of the Upper Berg River Dam site in 2006 (left) and in 2015 (right)

The Working for Water programme has always adopted an integrated approach to invasive alien plant control, combining manual and chemical measures together with biocontrol. The programme is strongly supported by several pieces of legislation, primarily the Conservation of Agricultural Resources Act No. 43 of

1983 and the National Environmental Management: Biodiversity Act No. 10 of 2004, and their Regulations. Since its inception, the programme has maintained close links with the research community and has been influenced by scientific research (Ntshotsho *et al.*, 2015a). More than a million ha have been cleared since the beginning and employment opportunities are provided to approx. 20,000 individuals annually. Because of its positive societal and environmental impacts, the programme has grown and diversified into other programmes and, together, they now all fall under the Natural Resources Management umbrella programme.

At a local level, a recent assessment of one of the projects has demonstrated significant water gains (Ntshotsho *et al.*, 2015b). Modelling shows that clearing of the upper catchment of the Berg River Dam (Figure 6.9), which covers an area of approximately 12,000 ha, has resulted in estimated water gains of between 9.0 and 12.7 million m3/yr. This gain represents 7 to 10% of the capacity of the 126.4 million m3 dam. The dam is located within one of South Africa's 21 strategic water source areas (these are areas that occupy 8% of South Africa's land area and supply 50% of the country's surface water) (Nel *et al.*, 2013) and is the second most important source of water for Cape Town.

Improved water supply is not the only potential benefit of invasive alien plant eradication. Another project looking at the rangeland impacts of invasion has shown that *Acacia mearnsii* can reduce grazing capacity by 56% and 72% on lightly and densely invaded sites respectively, whereas clearing can reverse these losses by 66% within 5 years (Yapi, 2013). This translates to 2 to 8 hectares required to support one large livestock unit (ha/LSU) on uninvaded and densely invaded sites, respectively. Improved pasture condition has a direct positive impact on livestock condition and this can lead to improved human well-being at the household level (Ntshotsho *et al.*, 2015b). This has been demonstrated in yet another Working for Water project which looks beyond just the clearing of invasive alien plant species (*Acacia* spp.) and takes a land stewardship approach. Indigent communities in a rural part of South Africa were trained, guided and supported, through the programme to restore communal land. After two growing seasons post-clearing, there was discernible improvement in the physical condition of cattle. The cattle owners were then assisted to sell their stock to commercial butchers in the area in two auctions that generated revenue totalling just over R1.3 million (~$89 300) for the 63 households involved. The success of the Working for Water programme can be attributed to four interconnected factors at project level: commitment, passion, strategic planning and the consideration of context (Ntshotsho *et al.*, 2015b). In addition, political buy-in and long-term commitment of funds by government are equally important for the success of the programme at national level.

6.3.2.2 Responses to mineral extraction

The significant effects of mining surface lands include complete removal of ecosystems, hydrological disruption and degradation of soil resources during removal, storage and re-instatement (Harris *et al.*, 1996) (see also Chapter 3, Section 3.4.7.3). The use of heavy equipment and soil stockpiling during mining remains a major limitation to quickly re-establishing ecosystem structure and function (Harris *et al.*, 1989). Potential off-site impacts, particularly the generation of acid mine drainage, need to be minimized by on-site management.

Reclamation, rehabilitation and restoration of these sites to a variety of end-uses entails overcoming abiotic and biotic barriers or limitations to establishing functioning ecosystems (Hobbs & Harris, 2001). An overemphasis on idealized optimal conditions has often led to prescriptive targets for restoration, with the danger that this limits variability and spontaneity in the restored ecosystem (Brudvig *et al.*, 2017; Hiers *et al.*, 2016). Approaches include active intervention such as re-contouring, planting, soil amendment, inoculation, animal re-introduction and “spontaneous redevelopment” (Parrotta & Knowles, 2001; Prach *et al.*, 2013; Šebelíková *et al.*, 2016; Walker & Del Moral, 2009), with a variety of possible post-mining uses from natural systems to agriculture (Howieson *et al.*, 2017).

Sound waste management and rehabilitation plans are key elements in environmental restoration following the closure of mines (Adiansyah *et al.*, 2008). Topsoil management is of course critical, but only after a replacement of overburden and landscape reformation (Harris & Birch, 1989; Parrotta & Knowles, 2001). However, activities related to site rehabilitation yield no capital returns to mining operations and can have significant impacts on their operational costs and economic feasibility. Therefore, in less developed economies with weak mining governance, mitigation plans may be neglected.

On mined lands, active restoration is required to trigger natural processes of succession and to develop functioning soils (Gardner & Bell, 2007; Koch & Hobbs, 2007; Skirycz *et al.*, 2014; Tischew & Kirmer, 2007). The use of native species tolerant to heavy metals (metallophytes), and others capable of rapid soil development (e.g., nitrogen-fixing legumes), is a priority for restoration of contaminated mining sites (Ginocchio & Baker 2004; Whiting *et al.* 2010). However, this is not important when non-metalliferous materials have been extracted, especially coal, which covers a significant portion of the total area affected by surface mining, despite the fact that some sites suffer from an acidic pH, which is usually addressed by liming. A wide range of responses is available, ranging from “spontaneous regeneration”, through direct seeding and planting, to animal species reintroduction (see Stanturf *et al.* 2014 for a major review on this). Although significant research into physical management, organic and inorganic additions, plant reintroduction and fungal propagule inoculation has been carried out, the restoration of mined lands remains an intractable problem, with estimates of recovery varying from 10 to1000 years. Predicting time for ecosystem recovery is in practice difficult to determine, as different ecosystem characteristics recover at different rates, depending on degradation and disturbance type, site topology, on-site resources and off-site recruitment potential (Curran *et al.*, 2014; Jones & Schmitz, 2009; Spake *et al.*, 2015). Frouz *et al.* (2013) demonstrated that restoration to simple shortgrass prairies could be achieved faster than complex communities in tallgrass prairie and forest, on essentially the same post-mining substrates.

When only sub-soils and overburden materials are available for reclamation and/or restoration after mineral extraction, the addition of topsoil and composts can greatly aid establishment of vegetation (Spargo & Doley, 2016) and fauna (Cristescu *et al.*, 2013). Active intervention with fertilizers and soil amendments can enhance nutrient cycling and tree establishment (Howell *et al.*, 2016), and inoculating soil with appropriate mycobionts (especially mycorrhizal fungi) can aid tree establishment and survival (Asmelash *et al.*, 2016; Hoeksema *et al.*, 2010).

Soil ecology research has been used extensively to track the changes in sites subject to restoration programmes (Harris, 2003). Earthworm reintroduction has a positive effect on ecosystem service re-establishment (Boyer & Wratten, 2010), but only where they are natives. Mine site restoration in the Jarrah forest of Western Australia has been considered a largely successful case in terms of restoring vegetation (Grant & Koch, 2007) and fauna (Craig *et al.*, 2017). However, Banning *et al.* (2011) demonstrated that 26 years after mine restoration in these restored forests, microbial communities were not able to use the same range of carbon substrates than the reference sites. Nonetheless, progress towards a “reference” was more rapid than in less intensive programmes of restoration where fewer plant species and soil stockpiling were used; as opposed to the direct soil replacement and multiple tree species planting practices used in the Jarrah restoration programme.

Plant species additions, especially trees (Chodak *et al.*, 2015), can influence the eventual composition of the soil biota as well as chemical and carbon cycling (Harris, 2009; Józefowska *et al.*, 2017). Furthermore, by amending post-mining soils with “live” soils from a desired reference state site can enhance the rate at which ecosystem characteristics recover on drastically disturbed post-mined sites (van der Bij *et al.*, 2017) and these amendments can control the assembly of vegetation communities to reach the “desired” plant community configuration (Wubs *et al.*, 2016). Moving from stockpiling soils during mining operations, to “direct replacement” involving careful handling of soils during transfer, secures both better plant establishment and below-ground invertebrates, especially earthworms (Boyer *et al.*, 2011). Moreover, the re-use of stockpiled soil materials - combined with on-site waste mineral resources - can ensure a more complete and functionally-capable soil microbial community in post-mining sites (Kumaresan *et al.*, 2017).

“Spontaneous regeneration” is an approach which has been used extensively in Central and Eastern Europe, principally on post-coal opencast (strip) mines. Here, sites are re-contoured but not planted and can effectively regenerate. Šebelíková *et al.* (2016) demonstrated that while the species richness of such spontaneously regenerated sites were no different than that of sites reclaimed by active forest planting were after 20-35 years post-mining, they tended to be more diverse in terms of species of conservation interest (11 as opposed to 4 IUCN Red List species). Further, in many cases, woodland vegetation may become established on a successional trajectory through spontaneous regeneration after just 20 years on previously forested sites, but wetland sites are more variable in their progress (Prach *et al.*, 2013; Tropek *et al.*, 2010). Spontaneously regenerated sites provide better cover for establishing climax woody species than those sites which are deliberately planted (Frouz *et al.*, 2015). An essential caveat here is that without a readily available source of seeds and fungal spores that are able to reach these sites by natural means, such successional processes may take much longer.

6.3.2.3 Responses to soil quality changes

Healthy soils are a prerequisite for meeting global food, feed, fibre and energy needs (FAO, 2015). To meet those needs, while sustaining or improving soil health or soil quality, several soil and crop management response strategies have been developed - including various combinations of tillage, crop rotation, nutrient management, cover crops and other practices collectively referred to as “agronomic practices”. Other response strategies include agroecology, organic farming, ecological intensification, conservation agriculture, integrated crop livestock and integrated crop livestock forestry systems. All of these strategies have different energy intensities, effects on biodiversity and levels of reliance on agrichemicals. These must be balanced through site-specific decisions which also recognize inherent constraints including climate change, acidification and salinization.

To monitor the effects of any response strategy, several soil health and/or soil quality indicators have been identified: biomass growth, development and productivity (Ponisio *et al.*, 2015); increased soil biodiversity and function (Birkhofer *et al.*, 2008; Roger-Estrade *et al.*, 2010); and species richness across a continuum from the field, to the farm, to the landscape level (Egan & Mortensen, 2012). Ideally, producers voluntarily select the most appropriate combination of practices to meet economic, environmental and social goals, but science-based regulations may be imperative in some situations (Chasek *et al.*, 2015; Karlen & Rice, 2015).

Soil health and quality have become essential for evaluating profitability and, as a guideline, for avoiding and reducing land degradation or restoring degraded lands due to their influence on: water entry, retention and release to plants; nutrient cycling; crop emergence, growth and rooting patterns; and ultimately yield. One of the most important soil health and quality changes, associated with any response strategy, is an increase soil organic carbon, because it directly influences a multitude of soil properties and processes. For example, applying animal or green manures can improve soil health and quality by increasing soil porosity, enhancing soil structure (i.e., binding of sand-, silt-, and clay-size particles), decreasing compaction, increasing aggregation and decreasing wind and water erosion.

Tools for assessing the effects of various response strategies on soil health and quality - at level of the field, farm, catchment, or larger areas - include the Soil Management Assessment Framework (Andrews *et al.*, 2004; Cherubin *et al.*, 2016) and the Comprehensive Assessment of Soil Health protocol (Moebius-Clune *et al.*, 2016). The EU Thematic Strategy for Soil Protection addresses soil health and quality and land degradation by striving to ensure that soils can provide seven critical functions: (i) food and other biomass production; (ii) storing, filtering and transformation of materials; (iii) habitat and gene pool of living organisms; (iv) physical and cultural environment for humankind; (v) source of raw materials; (vi) acting as a carbon pool; and (vii) archive of geological and archaeological heritage. This has been done by integrating soil protection into several European Community Policies (Toth, 2010), since efforts to establish a universal "Soils Framework" were unsuccessful.

Soil health and/or quality responses to selected degradation drivers

A combination of high-yielding, water-efficient plant varieties, the adoption of reduced- or no-till farming practices, improved pest and pathogen management, and optimizing planting schedules and crop rotations can improve soil health and quality, while reducing production costs and helping to mitigate atmospheric greenhouse gas (GHG) emissions. Burney *et al.* (2010) concluded that appropriate, site-specific combinations of those practices reduced GHG emissions by 161 GtC between 1961 and 2005, while Canadell & Raupach (2008) concluded that reforestation of 231 million ha could lead to an increase in carbon sink capacity from 0.16 to 1.1 Pg C yr^{-1}, between now and 2100. Afforestation of unused, marginal and abandoned land, as well as harvesting forests more frequently, could further promote carbon sequestration (Bird & Boysen, 2007; Harris *et al.*, 2006; Liu & Hiller, 2016; Valatin & Price, 2014). For China, Canadell & Raupach (2008) estimated that 24,000 km^2 of new forest was planted - offsetting an estimated 21% of China's 2000 fossil fuel emissions. Better harvest management and prevention of forest fire or other disturbances can further increase forest carbon storage capacity (Liu *et al.*, 2016; Pilli *et al.*, 2016) and soil health.

Acidification

Cropland acidification (see Section 4.2.2.1) is caused by both natural and anthropogenic processes (Bhattacharya *et al.*, 2015; Günal *et al.*, 2015; Koch *et al.*, 2015) and has been calculated to reduce farm gate returns in Australia by $400 million per annum through lost production (Koch *et al.*, 2015). Response strategies include reducing atmospheric deposition and use of acidifying soil amendments such as anhydrous ammonia. Transitioning from long-term, high-rate nitrogen fertilizer applications and continuous cropping without organic inputs, in Africa, has been recommended to mitigate acidification (Tully *et al.*, 2015).

Acidification increases the mobility and leaching of exchangeable base cations (calcium, magnesium, potassium and sodium), decreases soil buffering capacity and increases concentrations of aluminium, magnesium and several heavy metals that are toxic to most plants. Therefore, the most direct approach to manage acidification is to apply lime (CaCO3) or other basic materials. This increases base saturation, decreases concentrations of aluminium, magnesium and other contaminants, improves the acid-base status of streams draining the area and stimulates recovery of biotic resources (Battles *et al.*, 2014; Johnson *et al.*, 2014). Unfortunately, liming is less effective for acidified subsoil, as time is required for lime to penetrate through topsoil before it can neutralize the acidity (Johnson *et al.*, 2014). Another response strategy is to change the amount and type of nitrogen fertilizer which Chen *et al.* (2008) reported influence soil acidity as follows: $(NH_4)_2SO_4$ > NH_4Cl > NH_4NO_3 > anhydrous NH_3 > urea. Acidification can also be reduced by decreasing atmospheric acid deposition. This has been occurring in Western Europe since 1980, because of increased air quality regulations (Virto *et al.*, 2015), but forest recovery remains limited because simply reducing acid input decreases aluminium and magnesium concentrations more rapidly than it increases base saturation.

Salinization

Salinization negatively affects soil health and quality by impairing productivity and several ecosystem functions. Globally, 23% of all irrigated land is classified as saline (FAO, 2014). Response strategies such as: (i) preventing excessive groundwater withdrawal and seawater intrusion, (ii) irrigating only where there is proper drainage, (iii) increasing aquifer recharge; and (iv) improving land and water management decisions, have been developed in response to an estimated $27.3 billion in lost crop production, alone (Qadir *et al.*, 2014).

In humid regions such as Canada, Northern Great Plains in the USA and Western Europe, a combination of geological conditions, climate patterns and cultural practices (tillage, crop selection, fallow lands and so on) have created saline seeps. The saline seeps form when soil water, not used by plants, moves below the root zone through salt-laden substrata to impermeable layers, and eventually flows to depressions where the water evaporates and leaves deposits enriched in sodium, calcium, magnesium, SO_4-S and NO_3-N which subsequently retard plant growth (Black *et al.*, 1981). This latter process is much more severe in arid and semi-arid regions (Anker *et al.* 2009). Response strategies include diverting surface drainage from recharge areas and intensifying cropping systems to fully utilize precipitation (MAFRI, 2008).

In Europe, most saline areas are located in areas with a Mediterranean climate (i.e., Spain, Greece and coastal parts of France and Portugal), often the result of improper irrigation (Virto *et al.* 2015). Suggested responses include: using high-quality (low electrical conductivity) irrigation water; applying sufficient irrigation water to leach soluble salts below the plant root zone; planting of salt tolerant cultivars; implementing phytoremediation with halophytes and subsequently harvesting them; adding calcium sulfate or strong acids; and increasing organic matter (FAO-ITPS, 2015). Another approach is to restrict the use of natural water resources to quantities that drain into terminal reservoirs as oceans, saline or dip aquifers (Schaible & Aillery, 2012). Growing salt-tolerant crops often have an added soil health and/or quality benefit, because they generally support the formation of stable soil aggregates that improve infiltration and resistance to wind erosion, while also decreasing surface crusting. Finally, there are several agro-hydro-salinity models such as SALTMOD, DRAINMOD-S or SAHYSMOD that can predict water distribution and salt balance, thus helping to reduce or even prevent salinization.

Soil management strategies to enhance soil health and/or quality and mitigate degradation

Tillage frequency and intensity, crop rotation, animal and/or green manure application, cover cropping, grazing intensity and agroforestry can improve soil health and/or quality (Wingeyer *et al.* 2015; Veum *et al.* 2015) and avoid, reduce or reverse land degradation by increasing biomass content and biodiversity. Tillage is especially important (Hammac *et al.*, 2016), because it affects surface cover and the size, composition and activity of the biological community below ground (Lehman *et al.*, 2015). Tillage also affects soil structure and stability, aeration, water balance and nutrient cycling - although response time when converting from high to low impact activities can take a decade. Soil health and quality changes - in response to fertilizer management, cover crops, animal or green manure applications, biochar and/or compost applications and site-specific management - also require time to be detectable. This temporal effect is therefore the basis for recommending soil health and quality monitoring to avoid, reduce or reverse land degradation. Finally, policy changes and especially national regulations, are currently very limited; relying instead on industry “best-practice” approaches to avoid further degradation and reductions in soil functional capacity (Chasek *et al.*, 2015).

Agroecological and ecological intensification approaches can enhance soil health and/or quality, reduce destruction or degradation of semi-natural ecosystems and homogenize landscape structure (Dumanski, 2015) (see also Chapter 5, Section 5.3.3.2). Ecological intensification involves actively managing farmland to increase natural processes that support production, including better biotic pest regulation, nutrient cycling and pollination (Bommarco *et al.*, 2013; Tittonell, 2014). Both ecological intensification and agroecology (see Section 6.3.1.1) emphasizes making smart use of ecosystem functions and services at field and landscape scales, to enhance agricultural productivity, reduce reliance on agrochemicals and thus avoid further land-use conversion. As a practice for preventing or mitigating cropland degradation and maintaining or improving soil health and quality, planting a green cover between crop rows has been suggested because it reduces soil erosion. However, the cover crop can use a considerable portion of the plant-available water. Hence good, data-driven and science-based management practices are essential for a win-win outcome in these practices.

Many have advocated “organic” farming practices to enhance carbon sequestration (Gattinger *et al.*, 2012), reduce cropland soil degradation and avoid unintended consequences such as impaired water quality and/or quantity associated with intensive agricultural practices (Cambardella *et al.*, 2015). Typical organic farming practices include the application of composted animal manure, use of forage legumes and green manures and extended crop rotations. National regulation and/or policy changes may help advance organic farming, but costs of production, tillage for weed control and possible yield reductions, are still often cited as being significant.

Conservation agriculture (see Section 6.3.1.1) encompasses many different practices that, in combination, can avoid, reduce and even reverse land degradation (Dumanski, 2015; Farooq & Siddique, 2015; Lal, 2015a, 2015b). Implementing conservation agriculture practices can improve soil health and quality by intensifying production, enhancing environmental benefits and protecting against water pollution. Conservation agriculture can also help increase soil organic carbon content, conserve soil structure and ensure or enhance soil microbial biomass.

By preventing excessive or uncontrolled livestock grazing, ensuring that crop residue removal is not excessive, decreasing wind and water erosion and avoiding depletion of soil organic matter, integrated crop, livestock and forestry practices provides a multitude of benefits for soil health and quality. Optimal response strategies will differ between arid or semi-arid ecosystems and humid areas, and success very much depends on the biome type. In some areas, national grazing regulations can influence whether land is managed sustainably or not (Nielsen & Adriansen, 2005). The practices can be optimized by implementing evaluation schemes focused on soil organic matter, because of the influence it has on several soil health and/or quality properties and processes. However, even though soil organic matter content is effective for assessing and monitoring effects of the land-use policies and optimizing crop, livestock and forestry integration (Toth, 2010), it is a poor surrogate for characterizing soil biodiversity.

In summary, several different management strategies can be used to avoid or mitigate soil health and/or quality changes and many can be implemented in developing countries. Regardless of the specific practice, the most important strategy may be to adopt policies that ensure efficient, economical and sustainable methods are being used to enhance soil health and quality and avoid further land degradation.

Use of indigenous and local knowledge (ILK) with scientific inputs can be an effective response to reduce or reverse soil degradation (see Box 6.6 for an example of highly effective ILK use to enhance soil health).

Box 6.6 Use of farmers' knowledge to enhance soil health in India

An extensive indigenous and local knowledge (ILK) base for natural resource conservation and management exists in most countries. In India, where traditional soil and water conservation practices are implemented under a variety of agroecological conditions, many agronomic practices including terracing, applying soil amendments, harvesting water, controlling seepage, recharging groundwater, optimizing tillage and using different land configurations, are influenced by ILK (Mishra, 2002).

One example focused on soil health is the use of mixed and diversified cropping systems. In rainfed areas, farmers use traditional practices to grow various annual crops (including millet) that exploit different growth habits and rooting patterns. Those differences enable the crops to use nutrients and soil water from different soil layers, thus increasing resource-use efficiencies. In turn, this results in more rapid canopy closure which reduces weed growth and competition with the annual crops, as well as the erosive impact of intensive (monsoon) rainfall when it does occur. Furthermore, the sequence of crops is selected in a manner that enables the above-ground crops to be harvested before the underground crops and to support grazing of crop residues by animals. The combination of residual root biomass, crop residue, animal excreta and farmyard manure helps sustain the soil organic matter content, which in turn improves soil health, crop nutritional status and economic returns to the farmers.

6.3.2.4 Responses to water quality changes

Land-based pollution and degradation of freshwater and coastal ecosystems have implications for both the health of aquatic, coastal and marine ecosystems (see Chapter 4, Sections 4.2.4 and 4.2.5), as well as food and water security, human health and exposure to flood risk (see Chapter 5, Sections 5.3.2, 5.8.1 and 5.8.2). Local responses to water resources pressures - exacerbated by climate change impacts in many regions - focus primarily on improved crop and soil management (see Sections 6.2.1.1 and 6.3.2.4) as well as ILK related to water conservation and management. They also include a variety of other water management approaches such as: construction of large or small dams, reservoirs and irrigation systems; wastewater treatment; river and stream rehabilitation; and development of advanced water management technologies (CGIAR, 2016).

Integrated land and water management is an effective response to ensure catchment-scale hydrological balance and to minimize the occurrence of extreme hydrological events (floods and drought) and their impacts on people. Other responses applied to agricultural land management (see Sections 6.3.1.1 and 6.3.2.4) include: improvements in rainfed agricultural productivity (through, for example, increased use of drought-resistant crop varieties); managing soil health and fertility; managing soil moisture in rainfed areas; increasing efficiency of irrigation systems and improving on-farm water productivity; and managing environmental risks associated with agricultural intensification (FAO, 2011). An example of a management programme that has had some success in improving water quality and ecosystem health is the Chesapeake Bay Program: a regional partnership established in 1983 that directs and conducts the restoration of the Chesapeake Bay in the mid-Atlantic region of the USA. This Program, and the 2014 Chesapeake Bay Watershed Agreement, coordinates efforts of various state, federal, academic and local watershed organizations. The aim is to build and adopt policies which support the goal of reducing the amount of pollutants and nutrients from upstream land-based sources - particularly nitrogen and phosphorus from agricultural runoff that have, since the 1950s, resulted in extensive eutrophication and hypoxia of the region's rivers, estuaries and marine ecosystems (Goesch, 2001; Hagy *et al.*, 2004; Kemp *et al.*, 2005).

Responses to hydrological regime changes include the use of soil and water conservation techniques, judicious land management practices and the provision of incentives to landholders and communities (Brunette & Germain, 2003). The use of mobile-based networks and apps allows for rapid, reliable decisions on monitoring, acquiring and processing real-time data on water level, rainfall, runoff, water quality and leakage detection. Such systems help farmers to optimize irrigation and obtain (cloud-based) information on soil data - allowing them to determine the amount of water necessary to produce the maximum yield in a given irrigation zone. Responding to a drought of historic severity, California started a pilot programme to install smart water meters that detect leaks and optimize water use at the household level. At the same time, they are using sensors for smart irrigation control to reduce water consumption by the State's large agricultural producers (IWA 2015).

The coordination of environmental, economic, trade and development policies can promote practices that improve natural resource-use efficiency, which is essential for countries with relative water shortages. New solutions for appropriate water balance have been devised, such as water trading, cloud stimulation and climate-smart technologies.

Water quality technologies such as desalination and wastewater treatment are energy intensive and may be expensive and/or produce effluents that must be disposed of. One prominent challenge in water reuse (particularly potable reuse) lies in community acceptance, because many people are inherently averse to drinking or using reclaimed water (Brown & Davies, 2007). Uses of non-potable reclaimed water that are more widely acceptable include agricultural irrigation, industrial processes, street washing, toilet flushing and landscaping. Greywater can also be used for irrigation but, like wastewater, it must undergo some treatment to remove oil, surfactants and other organic contaminants before it is applied to crops (Travis *et al.*, 2010). Reclaimed water also has potential uses in urban and suburban landscape maintenance and other non-agricultural spaces, thereby reducing the use of potable water for non-drinking purposes. Industrial processes that utilize reclaimed water include evaporative cooling, boiler feed, washing and mixing (Levidow *et al.*, 2016; Thoren, Atwater, & Berube, 2012).

Wastewater treatment using constructed wetlands (see Section 6.3.1.5) has been used effectively in both developed and developing countries (IWA, 2015; SIWI 2010). Making these systems more automated, low maintenance and user-friendly may help promote widespread implementation of small-scale systems, that together can save vast amounts of potable water (IWA 2015).

Effective water management solutions range in their cost, accessibility and energy efficiency. Most demand-based management strategies tend to be relatively low cost, and by reducing water consumption, they decrease pressure on water resources. Rainwater and runoff harvesting techniques are often energy neutral and include low-cost practices that can be used almost anywhere (Mekdaschi-Studer & Liniger, 2013).

Technologies for addressing water challenges are becoming more advanced and increasingly energy efficient (IWA 2016; UN Water 2015), but unfortunately many of the countries with the greatest need for more reliable water supplies lack the economic means to implement them. Some promising examples of alternative water management technologies being used in developing countries (IWA 2016) include:

- Small-scale rural greywater reuse systems in rural Madhya Pradesh in India, which was so effective in reducing water demand and improving sanitation that similar systems were later implemented to serve over 300 schools and 1,500 households, thus avoiding contamination of soils and water, and negative impacts on human health (Godfrey *et al.*, 2010);
- In the village of Cukhe, on the outskirts of Hanoi in Vietnam, rainwater harvesting systems (costing less than $400) that consisted of screens, settling tanks with calm inlets, UV filtration and first flush systems were installed. They eliminated the need for expensive bottled water to supply potable water and avoided groundwater contamination by arsenic and sewage runoff. Furthermore, by using previously less-trusted groundwater to meet outdoor and non-potable needs, the village was able to diversify its water supply and conserve rainwater (Nguyen *et al.*, 2013).

A comprehensive understanding of the water-energy nexus is therefore needed in decision-making about technological options and considerations for clean, renewable energy sources should be incorporated into projects as much as possible (IWA 2016). Because no solitary solution is globally applicable, water managers and relevant stakeholders must together find the solutions most appropriate to the social, economic, political, institutional and environmental conditions of a given area (IWA 2015). A nearly globally-standardized set of best available technologies or techniques aimed at optimizing systems of integrative pollution prevention and control have been developed, primarily for the industrial sector (Entec, 2009; Geldermann & Rentz, 2004; Karavanas *et al.*, 2009). Similarly, best practice guidelines for water harvesting, based on experiences from throughout the world, are also available (Mekdaschi-Studer & Liniger, 2013).

Box 6.7 Improving food security in Ethiopia through agrometeorological monitoring

Ethiopia, where one in three people currently live below the poverty line, has one of the world's largest populations dependent on the vagaries of annual rainfall (ECSA & WFP, 2014). When droughts occur, very large numbers of people can be adversely affected by crop production shortfalls. At times, as many as 7.6 million people may require emergency support. Since Ethiopia has many inaccessible regions, an objective, country-wide, geographic assessment of conditions called the Productive Safety Net Program has been developed (FAIS, 2012; GOE, 2015).

The Program uses a numerical model - the water resource satisfaction index - which can be related to crop yield using a linear yield-reduction function, specific to each crop. In this way, crop yield is modelled at the start and end, and for the entire season (Senay & Verdin, 2003). In addition to water, other factors that affect food security - such as poor roads and the cost of grain transport (Rancourt *et al.*, 2014) - are taken into account.

Since the water resource satisfaction index is a numerical index, it can be used for comparisons within and over multiple years; for example, the number of seasons when the crops failed completely between 1982 and 2011. Figure 6.10 shows that while mountainous highland areas experienced increases in rainfall during this period, the region in the rain shadow, in Tigray, became drier and less productive - with the area experiencing failed seasons in most years increasing to the east. The South-central and Southern Ethiopian regions, where most of the population is located, has experienced declines in rainfall over a thirty-year period (Funk *et al.*, 2005). This is due to both the changes in rainfall, as well as higher temperatures driving increased evapotranspiration. An advantage of the country-wide method is that it can show where rainfall anomalies are affecting crop yield, considering multiple drought-sensitive crops. Detecting and responding to changing rainfall, and consequent agricultural productivity, are key ways for Ethiopia to anticipate food security issues and respond early. In many countries at risk of food insecurity, similar schemes are used (e.g., Brown 2008, the Famine Early Warning System, FEWS; GEOGLAM Crop Monitor for Early Warning, https://cropmonitor.org).

Figure 6 10 The number of seasons that have a water requirement satisfaction index value of 50% or less for small grains between A 1982-1991, B 1992-2001 and C 2002-2011 in Ethiopia. The higher the number, the more failed seasons; D population density per square kilometer in 2020. Source: GPWv3 CIESIN (2005); Brown *et al.* (2017).

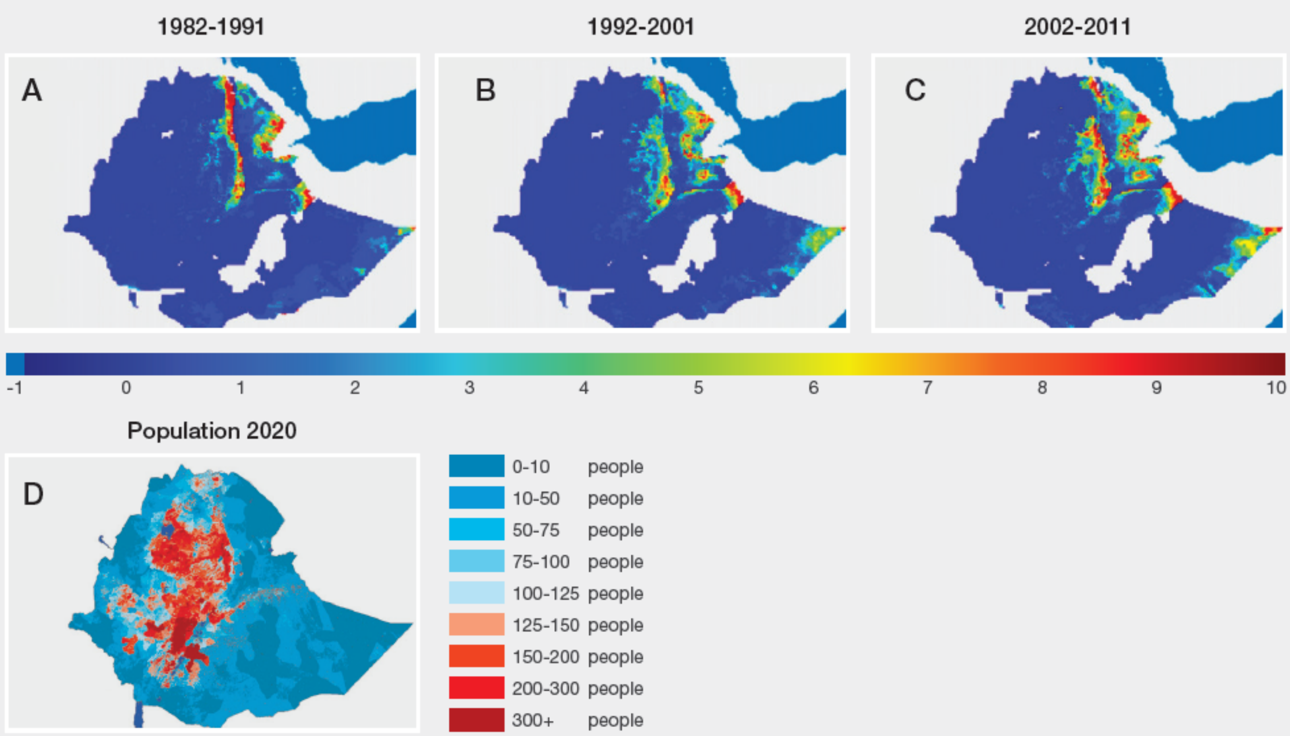


Table 6 6 **Summary of direct biophysical and technical responses, their nature and relative effectiveness in avoiding, reducing or reversing land degradation caused by invasive species, mineral extraction, soil quality change and water quality change.**

LAND USE OR DEGRADATION DRIVER	RESPONSE OPTIONS	NATURE OF RESPONSE	RESPONSE EVALUATION CRITERIA AND EFFECTIVENESS RANKING (COLOUR-CODED)					
		Avoid (Av), Reduce (Rd), Reverse (Rv)	Economic feasibility	Social accepta-bility	Environ-mental desirability	Cultural accepta-bility	Technical feasibility	Political accepta-bility
INVASIVE SPECIES MANAGEMENT	Identification and monitoring of invasion pathways	Av						
	Quarantine measures	Av						
	Mechanical control	Rd						
	Cultural control	Rd						
	Biological control	Rd						
	Chemical control	Rd						
MINE SITE MANAGEMENT	On-site management of mining wastes (soils and water)	Rd, Rv						
	Reclamation of mine site topog-raphy	Rv						
	Conservation and early replacement of topsoil	Av, Rd						
	Passive restoration measures to recreate functioning grassland, forest and wetland ecosystems	Rd, Rv						
	Active measures to restore natural hydrological dynamics, biodiversity and soil profiles	Rd, Rv						
SOIL QUALITY IMPROVEMENT	Managed or rotation grazing	Rd, Rv						
	Agroecological management	Av, Rd, Rv						
	Conservation Agriculture	Av, Rd						
	Organic farming	Av, Rd, Rv						
	Reduced tillage frequency and/or intensity	Av, Rd						
	Increased crop diversity and perennials	Av, Rd, Rv						
	Using cover crops	Av, Rd, Rv						
	Crop rotation	Av, Rd, Rv						
	Fertilizer management	Rd						
	Adding animal or green manure	Rd, Rv						
	Adding compost or biochar	Rd, Rv						
	Provide adequate drainage	Rd, Rv						
	Erosion control	Av, Rd, Rv						
	Phytoremediation	Rd, Rv						
	Repositioning eroded soil	Rd						

LAND USE OR DEGRADATION DRIVER	RESPONSE OPTIONS	NATURE OF RESPONSE	RESPONSE EVALUATION CRITERIA AND EFFECTIVENESS RANKING (COLOUR-CODED)					
		Avoid (Av), Reduce (Rd), Reverse (Rv)	Economic feasibility	Social accepta-bility	Environ-mental desirability	Cultural accepta-bility	Technical feasibility	Political accepta-bility
WATER QUALITY IMPROVEMENT	Rainwater harvesting	Rd, Rv	High effectiveness	High effectiveness	High effectiveness	High effectiveness	High effectiveness	Moderate effectiveness
	Wastewater treatment	Av, Rd	High effectiveness	High effectiveness	High effectiveness	High effectiveness	High effectiveness	High effectiveness
	Constructed wetlands	Rv	High effectiveness	High effectiveness	High effectiveness	High effectiveness	High effectiveness	High effectiveness
	Desalination	Rd	High effectiveness	High effectiveness	High effectiveness	Moderate effectiveness	High effectiveness	Moderate effectiveness
	Integrated land and water man-agement	Av, Rd, Av	High effectiveness	High effectiveness	High effectiveness	Moderate effectiveness	Moderate to high effectiveness	Moderate to high effectiveness
	Soil and water conservation practices	Av, Rd, Rv	High effectiveness	High effectiveness	High effectiveness	Moderate effectiveness	Moderate to high effectiveness	Moderate to high effectiveness
	Point source pollution control	Av, Rd	Moderate effectiveness	High effectiveness	Moderate effectiveness	High effectiveness	Low to moderate effectiveness	Moderate to high effectiveness
	Non-point source pollution control	Av, Rd	Moderate effectiveness	Low effectiveness	Low effectiveness	High effectiveness	Low to moderate effectiveness	Low to moderate effectiveness

EFFECTIVENESS RANKING OF RESPONSE OPTIONS

- High effectiveness
- Moderate to high effectiveness
- Moderate effectiveness
- Variable effectiveness (low to high)
- Low to moderate effectiveness
- Low effectiveness

6.4 Enabling and instrumental responses to land degradation and restoration

Enabling and instrumental responses are intended to address the direct and indirect causes of land degradation, thus avoiding further degradation and ultimately restoring or rehabilitating the land. The responses are broadly grouped into policy instruments, institutions, governance and anthropogenic assets (infrastructure, human resources, capacity, technology and indigenous or local knowledge-based practices) (MA, 2005a). This section complements Section 6.3 by briefly assessing potential responses to key indirect drivers and then assessing effectiveness of policy, governance and institutional responses to land degradation.

6.4.1 Responses to indirect drivers: globalization, demographic change and migration

Indirect drivers including pollution, migration, globalization, consumption patterns, energy demand, technology and culture can degrade land in many ways (see Chapter 3, Sections 3.6.3 and 3.6.4). The optimum response to those drivers will depend on which driver is most influential, how it interacts with other indirect drivers, the current institutional, policy and other governance factors (see Chapter 3, Section 3.6.2). As comprehensive evaluation of all indirect drivers is impractical (see Chapter 3, Section 3.6 for details), this section focuses on three: globalization, demographic change and migration. Although increased globalization and international trade can reduce economic growth barriers, they also bring environmental challenges, including land degradation. For example, increased demand for food and fuel in Asia and Europe led to rapid expansion of soybean production in the Amazon, Chaco and Cerrado biomes - pointing to how the shortening of supply chains, facilitated by information and transport technology, affects land-use decisions in distant places (Garrett *et al.*, 2013; Liu *et al.*, 2013). Responses to control the unintended consequences of globalization, international trade and consumption preferences in developed and developing countries involve raising public awareness, multi-sectoral and coordinated governance arrangements between private and public sectors, and the use of innovative policy instruments (Lambin *et al.*, 2014) (also see Section 6.4.2 and Chapter 8, Section 8.3).

Responses to land degradation caused by globalization and international trade of commodities include linking trade and environmental protection as a continuum from local to global levels (Lambin & Meyfroidt, 2011), with the use of policy instruments (e.g., tariffs). In conjunction, voluntary product certification schemes have been used to regulate land use, trade and consumption patterns, and have been environmentally effective for coffee (Lambin & Meyfroidt, 2011). The introduction of eco-certification of forest products in the early 1990s did not halt the decline of biodiversity in the tropics, as was intended, but it raised awareness and increased dissemination of knowledge on comprehensive sustainable forest management by embracing economic, environmental and social issues at a global level (Rametsteiner & Simula, 2003). Maintaining social and environmental standards for production, supply chain and consumption practices is imperative to minimize the ecological footprint of globalization and international trade.

Demographic change not only affects local land use and cover, but is also associated with land degradation and biodiversity loss at multiple spatial and temporal scales. Population density and other demographic factors (e.g., population structure, growth rate, migration dynamics and gender inequality) have complex relations with land degradation per se, and their impacts differ greatly (Waggoner &

Ausubel, 2002), often due to differences in affluence and behaviour. Responses to land degradation and restoration actions are more effective when aligned with high-level population policies that take into consideration specific population and land degradation interactions. Policy responses to address human-land interactions versus population change are not the same. The former may focus on reducing negative impacts of agricultural activities on biodiversity and land condition through sustainable intensification or other means (see Sections 6.3.1.1, 6.3.1.2 and 6.3.2.3), whereas the latter focuses on resettlement, fertility rate and rural-urban migration.

Forest ecosystem recovery through natural regeneration following rural-urban migration is well documented for many parts of Latin America (especially Patagonia, Northwest Argentina, Ecuador, Mexico, Honduras and the Dominican Republic) and for non-forested ecosystems (e.g., montane deserts and Andean tundra ecosystems of Bolivia, Argentina and Peru) (Aide & Grau, 2004). In Puerto Rico, forests have recovered from a low of less than 10% of the island's land area in the late 1940s to more than 40% in the 2000s, as a result of rural-urban migration (Grau *et al.*, 2003). In Misiones, Argentina, rural emigration "reduced" deforestation by 24% compared to a "no-migration" scenario. If future emigration rates increase, deforestation will be reduced by 26% in 2030 compared to the current trend (Izquierdo *et al.*, 2011). Within Latin America and the Caribbean, 362,430 km^2 of woody vegetation recovered between 2001 and 2010 because of outward migration and socio-economic changes (Aide *et al.*, 2013).

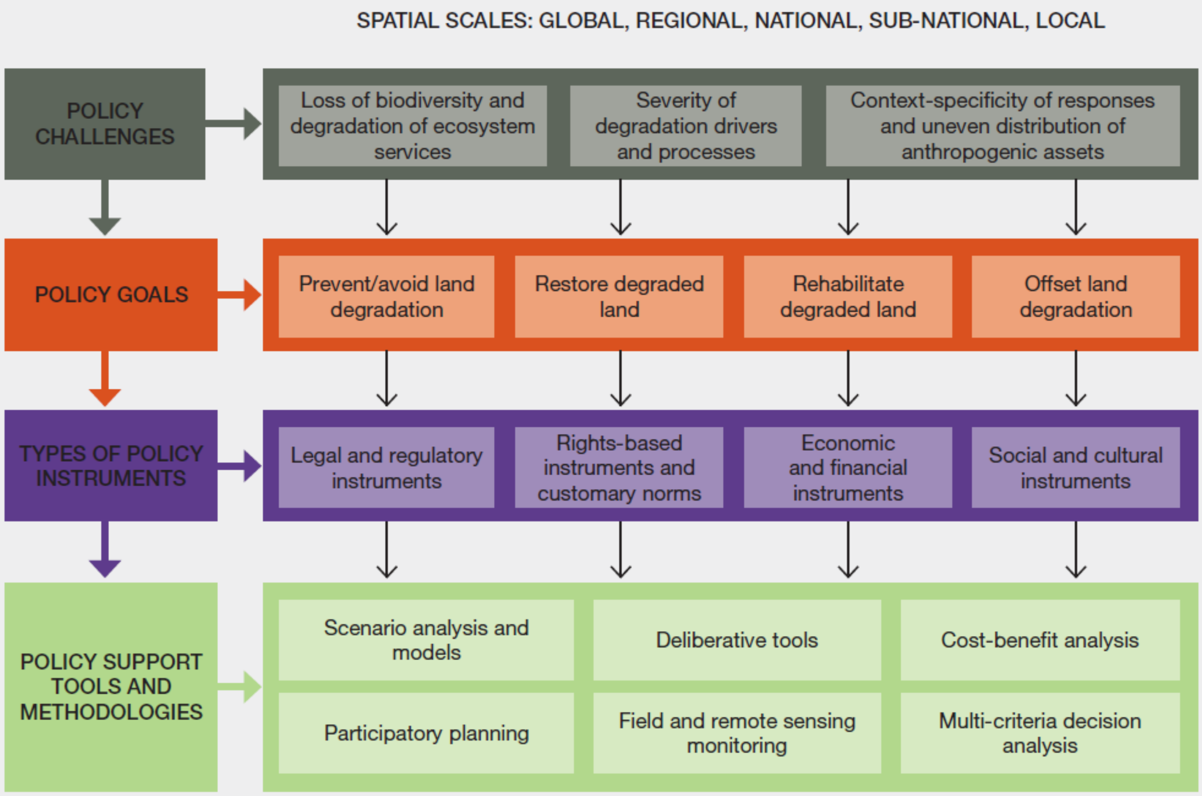
Migration-related land sparing and forest transitions have occurred historically in developed countries, but are now happening in many developing countries (Lambin & Meyfroidt, 2011). In China, ecological migration is a driver for resettlement policies and actions to promote ecosystem recovery (Wang, Song, & Hu, 2010). For example, the Chinese government has relocated millions of people from ecologically vulnerable areas, such as mountain areas of Guizhou and Shannxi province, to other rural or urban areas to facilitate land recovery (Chen *et al.*, 2014). From 2000 to 2012, about seven million farmers in Western China, alone, were relocated to areas within or outside their provinces (Tsunekawa *et al.*, 2014). However, this kind of relocation (for ecosystem recovery) requires careful assessment of its effectiveness and long-term impact. A study in a resettled area of north-western China found that water scarcity and its associated risks have not been alleviated due to land degradation (Fan *et al.*, 2015).

Voluntary rural-urban migration is a common adaptation response to land degradation. Household migration and depopulation of the countryside can lead to ecological restoration (Gao *et al.*, 2014; Wang *et al.*, 2010). In recent years, with the exception of North America, several parts of the world have experienced depopulation in mountain regions due to climate change and socio-economic conditions (Black *et al.*, 2011; Piguet, 2012). This trend has contributed to land restoration through natural processes in mountain regions.

6.4.2 Institutional, policy and governance responses

Institutional, policy and governance responses are designed to create, enable and implement actions on the ground to avoid, halt and reduce land degradation or reverse/restore degraded lands. The effectiveness of these responses is primarily associated with their design and implementation, including the type of policy instrument used and access to anthropogenic assets (e.g., research and technology development, institutional reform and capacity-building). This section focuses on types and effectiveness of policy instruments for guiding long-term decisions to avoid, halt and reduce land degradation and to restore degraded land at national and local levels (also see Section 8.3). Figure 6.11 illustrates several land degradation and restoration challenges and the associated policy goals, instruments, and support tools and methods to address them.

Figure 6 11 Land degradation- and restoration-related policy challenges, goals, instruments, tools and methodologies.



The appropriate policy instrument may depend on the spatial scale (i.e., local, regional, national or global) needed to achieve policy goals - although the same policy instrument can be applied at two different spatial levels for related policy goals. In Figure 6.11, the horizontal arrows expand the policy domain while the vertical arrows show relationships among policy support tools, methodologies and challenges. The vertical arrows thus represent many combinations that can contribute to one or more policy goals and challenges. Land-management policies and instruments are effective only when land managers are supported by those policies and have the means, commitments and control to restore, maintain or improve the quality of land (ELD, 2015). Furthermore, the appropriate policy instrument choice to promote sustainable land-management practices or landscape restoration depends on its environmental effectiveness, costs of implementation, monitoring, enforcement, distributional effects and conformity with other policies and political preferences (Low, 2013). This means that to be effective, policy instruments must be: economically and technically feasible; environmentally beneficial and desirable; and culturally, socially and politically acceptable (see Section 6.2.2).

6.4.2.1 Legal and regulatory instruments

Legal and regulatory instruments are used to encourage land managers to operate within the prescriptions of a given policy. The effectiveness of such instruments depends on specific policy settings (Alterman, 1997; Kairis *et al.*, 2014). For states that control land management, the first and most commonly-used legal and regulatory instrument - to avoid land degradation and to reduce or reverse adverse consequences of improper land use - is planning at national or regional (master plan) and local (zoning map) levels. The second set of instruments involves legal frameworks designed for industrial and agricultural activities based on national or regional standards.

Planning is a legal response according to the principle of subsidiarity and division of powers between public authorities (Dumanski, 2015; ESPON, 2012). This kind of legal response allows authorities to manage land use. Land planning and associated zoning enable the division of land for specific uses (e.g., natural, agricultural, or urban areas, limited housing density and/or urban growth areas, cluster zoning and/or obligation to build in continuity areas), and to establish legal or contractual conservation easements (Dissart, 2006; Hassan & Lee, 2015; Yucer *et al.*, 2016). In support of local planning, national and local authorities may also use other legislative and regulatory instruments, such as land-use or building permits, purchase of development rights, eminent domain (used in the most sensitive areas, e.g., coastal zones), or freezing the use of certain lands through land reserve funds. Territory control also allows the use of tax incentives, such as tax relief for non-waterproof or non-constructible lands, to maintain or relocate farming operations (Dissart, 2006).

International law can influence national policies related to soil protection and even compel states to adopt new legislation (Hannam & Boer, 2001; Leibfried *et al.*, 2015; Montanarella & Vargas, 2012). Local planning is thus subject to national and international law which can provide indirect protection for soils, safeguarding of wetlands and groundwater (e.g., Directive 2000/60/EC on Groundwater Protection of the Ramsar Convention; Dooley *et al.*, 2015; Kløve *et al.*, 2011), management of coastal land (eminent domain and/or easement), establishing targets for land degradation neutrality (Dooley *et al.*, 2015), management of public domain forests and conservation of biodiversity (e.g., UNCCD, CBD, Directive 2009/147/EC on the conservation of wild birds). International law can also improve national policies by converging policies within the same geographical territory across state boundaries (e.g., Cuypers & Randier, 2009; Directive 92/43/EC on the conservation of natural habitats and of wild fauna and flora**,** Alpine Convention).

Planning is also an instrument to avoid and reduce land degradation, commonly used in response to urban sprawl (Artmann, 2014), land encroachment (Gennaio *et al.*, 2009; McWilliam *et al.*, 2015), impermeability (Prokop *et al.*, 2011) and drought (Wilhite *et al.*, 2014). Indirectly, it works against the loss of organic matter and biodiversity, as well as flooding and soil compaction (DeFries *et al.*, 2010; Turbé *et al.*, 2010; Vu *et al.*, 2014).

The second most common set of legal and regulatory instruments used to avoid land degradation is based on legal frameworks designed to regulate economic activities known to be associated with land degradation (i.e., a similar approach to industrial regulation). Negative impacts on land and ecosystems from economic activities can also be mitigated through environmental impact assessments (Prieur, 2011) and provision of offsets for residual impacts of development activities. In addition to applying environmental standards on development activities, incentives such as eco-conditionality on financial assistance can also be adopted to minimize land degradation. Examples of such incentives include providing shares in favour of reducing the use of pesticides, enhancing crop diversification, converting to organic farming and organizing short distribution channels (Arnalds & Barkarson, 2003; Billet, 2008; Bodiguel, 2014; Pretty *et al.*, 2001; Singh, 2015, 2016). Incentives can also be used to reduce soil pollution or contamination, compaction or impermeability, and loss of organic matter or biodiversity. For example, EU farm policy promotes environmental protection with "agri-environment measures" that provide payments to farmers who participate in such measures (on a voluntary basis) to pursue a number of management practices. Such practices include: the management of low-intensity pasture systems; integrated farm management and organic agriculture; preservation of landscape and historical features such as hedgerows, ditches and woods; and conservation of high-value habitats and their associated biodiversity (Baylis *et al.*, 2008; Bodiguel, 2014; Bredemeier *et al.*, 2015; Bureau & Thoyer, 2014; Dal Ferro *et al.*, 2016; Huttunen & Peltomaa, 2016; Russi *et al.*, 2016).

The mechanism by which legal and regulatory instruments typically operate is based on the "polluter pays" principle, with an obligation to restore the site - failing of which requires an equivalent compensation to be paid for the damages suffered. To rehabilitate or compensate the residual effect of development (e.g., after a strategic environmental assessment or an environmental impact assessment) or contaminated sites, the project proponent is responsible for remediating impacted sites or contaminated soils when project activities end (Sirina *et al.*, 2013). Public authorities often assist in restoring sites (Lecomte, 2008; Steichen, 2010; Veenman, 2014). In the case of brownfields redevelopment/orphan site, restoration can be the direct responsibility of public authorities (Reinikainen *et al.*, 2016; Van Calster, 2005; Vanheusden, 2007).

For states that either do not control their land or have land management authority, contractual approaches are often used. These are characterized by the implementation of national plans (e.g., national plan against desertification or forest protection). Such plans establish a link between public authorities and indigenous or local communities, in the form of contracts, to adopt practices for soil conservation, choice of crops and farming practices, reduction or ban on clearing (Lavigne Delville, 2010; Mekouar, 2006; Plançon, 2009; Reij & Smaling, 2008; Sietz & Van Dijk, 2015)). The effectiveness of contractual arrangements as a response to land degradation varies depending on contract provisions. The contract holders can respond to reduce soil degradation, following a response hierarchy of prevention, mitigation and offsets (Adugna *et al.*, 2015).

Regulatory and legal responses to land degradation are in principle substantive and definitive, usually including specific preventive (fear of punishment) and curative (repair of environmental damage) measures. But how these measures have been operationalized in reality varies considerably, raising questions on their effectiveness (especially for the EU) (Paleari, 2017). The effectiveness of regulatory responses can depend on who is responsible for, who is impacted by, and the context of land degradation. For example, it was found that farmers in South-western Canada preferred voluntary policies (education, advice, grants) to reduce soil erosion and encourage soil conservation, even though they perceived regulatory approaches (penalties, cross-compliance, direct control) as being potentially more effective (Duff *et al.*, 1991).

In a study focused on the politics of land-use planning in Laos over the past three decades, Lestrelin *et al.* (2012) showed that land-use planning helped to reconcile different land uses, and interests among central and subnational governments, local actors, as well as national and foreign institutions. In another, multi-level analysis in Laos, Broegaard *et al.* (2017) found that cumulative effects of different legislations can reduce the potential positive impacts of legal reforms implemented to strengthen the rights of rural households (e.g., private property rights and planning processes). In a study of Wildlife Management Units in Mexico - with a focus on environmental policy instruments designed to promote ecosystem conservation and rural development via sustainable use of wildlife by local populations - Gómez-Aíza *et al.* (2017) highlighted the effectiveness of policy instruments as well as the importance of simultaneously adopting bottom-up and top-down management approaches. The protection of land depends on integrating the needs of local populations in policy instruments and understanding social vulnerabilities (McNeeley *et al.*, 2017).

Establishing protected areas to conserve biodiversity from human actions is a legal and regulatory response which often avoids land degradation. The management effectiveness of protected areas is discussed in Section 6.4.2.5.

6.4.2.2 Rights-based instruments and customary norms

A human rights-based approach in the fight against land degradation and desertification has been recognized as an important tool, because it brings together the legal strengths of international human rights and environmental law. This combination of laws can thus be used to combat land degradation and restore degraded lands at local to international levels.

The Global Mechanism of the UNCCD, for example, is supporting interested countries in the national land degradation neutrality target-setting programme, by helping to define national baselines, measures and targets to achieve land degradation neutrality (Orr *et al.*, 2017). Protecting human rights is one of the principles underpinning the vision of land degradation neutrality (Orr et al, 2017). The Voluntary Guidelines for Responsible Governance of Tenure of Land, Forests and Fisheries in the Context of National Food Security also applies existing governance standards, especially for human rights, to the management of land (Seufert, 2013; Windfuhr, 2016). Similarly, trade in agriculture and rights to food as human rights apply to land management (Cottier, 2006; Mechlem, 2006). What is unknown is whether and to what extent these human rights-based standards are taken into consideration as state parties take policy steps and make financial and human resource investments to achieve restoration of degraded lands.

Although the link between human rights and land degradation has been established in academic literature and soft law documents, it lacks legally-binding mechanisms at the international level, to operationalize the rights-based approach for restoration. In order to achieve Zero Net Land Degradation, legal and scientific literature has suggested the development of a global soil regime (Boer & Hannam, 2015; Lal *et al.*, 2012; UNCCD, 2012), that could take the form of a Protocol to the UNCCD and/or the Convention on Biological Diversity, or a separate convention focused on soil conservation.

A crucial element of a human rights-based approach to land degradation is the gender dimension (Lal 2000; UNCCD 2011). For example, in 2011 the UNCCD established an Advocacy Policy Framework on gender and “gender-sensitivity” - which is now seen as an important principle for achieving land degradation neutrality (Orr *et al.*, 2017). However, additional efforts (including financial support) will be needed to make sure that commitments on gender issues are actually implemented (Broeckhoven & Cliquet, 2015). The gender dimension of ecological restoration and benefits of mainstreaming it remain underexplored, but several recommendations have been made on how to improve it (Broeckhoven & Cliquet, 2015). They include using human rights instruments as a legal basis to push for greater involvement of women in restoration practices and for addressing underlying social and gender inequalities.

Empirical evidences from many developing countries suggest that halting resource (forest) degradation is possible and often effective when customary practices of local people and their rights to fulfil basic needs (e.g., fire wood, fodder) are incorporated in resource governance mechanisms (Agrawal & Ostrom 2001; Forest People Program & Program, 2010; Madrigal Cordero & Solis Rivera, 2012; Ostrom *et al.* 1999). States should ensure that policy, legal and organizational frameworks for tenure governance recognize and respect, in accordance with national laws, legitimate tenure rights (including those based on tenure) that are not currently protected by law (FAO, 2012).

It is important to recognize that customary practices (or local and/or indigenous practices) adopted by local people do have significance in halting land degradation and sustainable land management. Understanding the enabling socio-cultural factors – which could be defined on the basis of a rights-based approach, customary practices, and/or participatory processes – are instrumental to the success of land degradation or restoration responses. Thus, when designing responses to land degradation drivers or processes, local knowledge and customary practices should be given a high priority (Reed & Stringer, 2015).

6.4.2.3 Economic and financial instruments

Institutional, market and policy failures create differences in private and social costs, resulting in under-pricing of scarce resources (Panayotou, 1994) - including land and the associated goods and services it provides (Requier-Desjardins *et al.*, 2011). Externalities in land-use practices leads to socially sub-optimal, inefficient results (i.e., the costs of unsustainable land management practices are disproportionately borne by "off-site" parties who do not receive any compensation). Conversely, many sustainable land management practices benefit the public, whereas the costs of adopting them fall on the "on-site" actors (Low, 2013). Consequently, the actions taken by actors to avoid or reduce land degradation or to facilitate the adoption of sustainable land management practices would be less than socially desired due to such external effects (CBD, 2011).

Economic and financial instruments internalize such externalities from (un)sustainable land management practices into product price mainly through two types of incentive mechanisms: restrictive and supportive. Restrictive incentives for negative externalities (e.g., emission taxes, emission trading and quantity standards) are based on the polluter pays principle for negative externalities. Supportive incentives for positive externalities (e.g., subsidy and various types of payment for ecosystem services) are based on a beneficiary pays principle for positive externalities (Panayotou, 1994; Rode *et al.*, 2016).

The instruments to correct institutional, market and policy failures related to land degradation and restoration include the use of existing markets by inducing price changes (e.g., taxes, subsidy, bonds and so on) and/or the creation of new markets by providing new economic incentives (e.g., payment for ecosystem services, biodiversity offsets, conservation banking, natural capital accounting and so on.) (Initiative, 2015; Requier-Desjardins *et al.*, 2011; Sterner & Coria, 2012). The effectiveness of these instruments is highly context dependent, because of the interplay among broader socio-economic, institutional and policy environments - including the value systems and motivations of targeted actors (Beymer-Farris & Bassett, 2012; Kosoy & Corbera, 2010). In the following paragraphs, we synthesize empirical evidence on the use of these instruments and their effectiveness in avoiding, halting and reducing land degradation and restoring degraded lands.

Policy-induced price change

The effect of policy-induced price changes on halting land degradation or restoring degraded land depends on site-specific conditions. In some situations, higher agricultural commodity prices may encourage land management practices that accelerate degradation, especially when land tenure is insecure. In others, higher prices can provide scope for soil conservation measures that yield longer-term benefits. Examining the various interactions and trade-offs between agricultural development policy and land degradation, in the case of Sudan, Abdelgalil and Cohen (2001) found that four policies - namely price incentives, defined property rights, poverty reduction and enhanced human capital - were associated with reduced land degradation. While Zhao *et al.* (1991) found that commodity price distortions were associated with land degradation that negatively affected agricultural production in 28 developing countries, Pagiola (1996) found no simple relationship between price distortions and farmers' incentives to adopt soil conservation measures in developing countries. In Kenya, higher commodity prices incentivized farmers to adopt conservation measures on less productive steep slopes, but decreased investment on less steep slopes. In the Philippines, lower corn prices - after removing import tariffs - had the effect of conserving soil and reducing soil erosion in areas marginally suited to corn production (Briones, 2010). Similarly, European farm subsidies to meet good agricultural and environmental standards have been effective for erosion control, ground water management and increasing soil organic matter (Sklenicka *et al.*, 2015). These findings emphasize the importance of "getting prices right" and the need to adopt sustainable land and water management practices in agricultural production.

Payment for ecosystem services

Payment for ecosystem services, whereby services providers are financially rewarded by beneficiaries in return for otherwise "non-market" services, is a potentially economically-efficient way of achieving desired environmental and social outcomes. This instrument has been used in integrated conservation and development projects and can be effective in cases where proper institutional support is provided (Campos *et al.*, 2005; Engel *et al.*, 2008; Krause & Loft, 2013; Kroeger 2013; Wunder *et al.*, 2008; Zabel & Roe 2009). Allowing land managers to internalize some of the positive externalities created by sustainable land management - through payment for ecosystem services schemes - is seen as an important means to achieve land degradation neutrality (Mirzabaev *et al.*, 2015). In practice, these schemes have been financed by: (i) private beneficiaries of ecosystems services (i.e., individuals, organizations or companies), but are less common (Milder *et al.*, 2010; Sattler & Matzdorf, 2013; Tacconi, 2012); and (ii) governments or public agencies (e.g., agri-environmental programmes in the EU; Sattler & Matzdorf, 2013). The effectiveness of payment for ecosystem services schemes, however, varies considerably. The well-known Costa Rican programme is often considered as a successful case, because it had the effect of increasing forest cover and improving rural livelihoods (Porras *et al.*, 2014). The agri-environmental programmes in the EU are prone to adverse selection and moral hazards, reducing their effectiveness (Quillérou *et al.*, 2011; Quillérou & Fraser, 2010). The effectiveness of payment for ecosystem services schemes also depends on whether the payment is for temporary or permanent measures, with the latter generally being more effective.

Reducing emissions from deforestation and forest degradation in developing countries (REDD+) is a payment for ecosystem services scheme specifically focused on restoration of degraded forest land. Under REDD+ governments or multinational organizations compensate communities in developing countries for avoided deforestation and related climate-smart forest management. A recent review of the role of community-based forest management to achieve forest carbon benefits and social co-benefits suggests that REDD+ is likely to reduce forest degradation but not necessarily deforestation (Pelletier *et al.*, 2016). Some scholars argue that REDD+ is a cost-effective climate change mitigation policy (Komba & Muchapondwa, 2016), while others criticize REDD+ as a new conservation fad (Lund *et al.*, 2017; Redford *et al.*, 2013) that limits access to forests, compromises local people's customary rights (Poudel *et al.* 2014; West, 2012) and slows or reverses the promising trend of community-based forest management and governance in developing countries (Phelps *et al.*, 2010). The available evidence strongly suggests that the effectiveness of REDD+ to deliver climate change mitigation benefits - while reducing deforestation and forest degradation, biodiversity loss and providing social and economic "co-benefits" - depends on how its land management activities are implemented and the extent to which livelihood needs, governance, rights and social equity issues are addressed in REDD+ programme design, implementation and monitoring (Parrotta *et al.*, 2012).

Conservation tender or green auction among landholders, to act or manage the lands by adapting conservation practices, is considered an innovative payment for ecosystem services scheme (Latacz-Lohmann & der Hamsvoort, 1997; Latacz-Lohmann & Schilizzi, 2007). The oldest conservation tender programme is the Conservation Reserve Program in the USA which started in 1985 (USDA Farm Services Agency, 2011). Under the Conservation Reserve Program landowners' bids are ranked based on the Environmental Benefit Index: the ratio of ecological value of environmental benefits supplied and the value of the bid (Hanley *et al.*, 2012). In a review of the programme, Ferris and Siikamaki (2009) concluded that - even after about 25 years of implementation - it continues to be viewed positively by both conservation and agricultural communities. Farmers view that it is beneficial, because it is voluntary, does not transfer property rights, provides guaranteed income for the length of the contract and has the potential for supporting commodity prices by removing some land from production (Ferris & Siikamäki, 2009). Conservationists value the programme's conservation contributions such as habitat improvements, wildlife conservation and the provision of other ecosystem benefits (Ferris & Siikamäki, 2009). Like other OECD countries, Australia has also practiced conservation auction in the form of bush tender or eco-tender contracts (Eigenraam *et al.*, 2007; Stoneham *et al.*, 2003), landscape recovery auctions that include biodiversity and other environmental benefits (Hajkowicz *et al.*, 2007) and the Tasmanian Forest Conservation Fund (Binney and Zammit 2010). In a variety of land management and conservation contexts, scholars have found that bidding scheme for conservation contracts, to allocate government ecological funds, are practical, feasible and more cost-effective than fixed payment programmes (e.g., Connor *et al.*, 2008; Latacz-Lohmann & Schilizzi, 2007; Pannell *et al.*, 2001). They also claim efficiency gain on allocation of public funding through competitive bidding for ecological restoration.

However, payments for ecosystem services approaches may result in motivational "deadweight", providing unnecessary rewards for activities that would have occurred irrespective of payments (Beymer-Farris & Bassett, 2012; Kosoy & Corbera, 2010). For example, landholders who previously used sustainable land-use practices for various reasons would expect financial incentives under payment for ecosystem services schemes (Frey & Jegen, 2001; Reeson & Tisdell, 2008). To avoid such inefficiencies, engaging landholders in payment for ecosystem services programme design and the implementation of stewardship actions through cost-share programmes are considered by some to be more effective (Lukas, 2014; van Noordwijk & Leimona, 2010). Payments for ecosystem services approaches often promote economic values from a technocratic and economic perspectives and ignore indigenous and local knowledge and practices, human-nature relations and interactions, and social, cultural and spiritual values originated from such relations and interactions (Turnhout *et al.*, 2012, 2013), which need to be integrated in design and implementation of payment for ecosystem services schemes to enhance their effectiveness.

Biodiversity offsets

Biodiversity offset or ecological compensation has been introduced in many countries (OECD, 2016) to help balance economic development and environmental conservation goals. In principle, it is the last step in the mitigation (or response) hierarchy: avoid, minimize, restore and compensate (offset). One scenario of offsetting involves a developer - affecting land or habitat through activities such as mining, housing, industrial and infrastructural development (on the "impact site") - compensating for the resultant habitat loss by financing habitat restoration in a degraded land elsewhere (on the "offset site") of equivalent ecological value (Hahn *et al.*, 2015). From an economic perspective, offsetting is a combination of a cap (on habitat loss) and trade system in which the "spoiler" of habitats pays for restoration, possibly through a payment for ecosystem services scheme (Bull *et al.*, 2013; McKenney & Kiesecker, 2010; OECD, 2015). Offsets can be direct (on the ground actions) or indirect (e.g., funding for conservation programmes) and involve key concepts such as no net loss, additionality, permanence, timeframe, uncertainty, and monitoring and evaluation (BBOP, 2012; IUCN, 2014; Spash, 2015).

Biodiversity offsetting is common in the USA and Australia, while ecological compensation is common in the European Union where, for example, any loss of designated Natura 2000 sites must be compensated and this is done by government agencies on a case-by-case basis. The USA's wetland mitigation/banking, stream mitigation, and conservation banking programmes are among the world's largest offset programmes (OECD, 2016). Conservation banking involves legally-mandated biodiversity offsets, modelled after wetland banking (McKinney *et al.*, 2010). However, critics of the conservation banking system argue that the approach places too much focus on the compensation (offsetting) aspect and neglects earlier stages of the mitigation hierarchy (Hough & Robertson, 2009), resulting in a poor performance of the mechanism (Kihslinger, 2008; National Research Council, 2001). For example, an evaluation of 391 wetland offset projects in Massachusetts showed that 54% were not in compliance with the wetland regulations (Brown & Veneman, 2001). Similarly, Ambrose and Lee (2004) found that 46% of the 250 sites surveyed in California failed to replace key wetland ecosystem services. This could be due to the shortcomings of on-site and off-site compensatory mitigation provided directly by permittees, which has been substituted by wetland mitigation banking, a third party variation of off-site mitigation in recent years and also found to be more effective over the permittee-responsible mitigation (Briggs *et al.*, 2009; Orr *et al.*, 2017; Ruhl & Salzman, 2006). In Australia, biodiversity offsets have been widely used to compensate the residual impact of development, but the monitoring and verification of offset activities to achieve zero net loss remain inadequate (Martine Maron *et al.*, 2012; Office of the Auditor General Western Australia, 2017) and ecological compensation guidelines have often been neglected in practice (Briggs *et al.*, 2009; Coggan *et al.*, 2013). As a result, the effectiveness of offsets or compensation mechanisms to stop biodiversity loss remains debatable (Maron *et al.*, 2010, 2012, 2015). Similar to payments for ecosystem services approaches, biodiversity offsetting also promotes commodification of nature and economic values (Robertson, 2004; Turnhout *et al.*, 2013). For effective conservation and management of biodiversity through biodiversity offsetting, capturing and acting up on diverse forms of social values created and perpetuated through human-nature relations and interactions is essential (Turnhout *et al.*, 2012, 2013). Under the land degradation neutrality approach, the UNCCD's Science-Policy Interface recommends that ecological compensation should use land potential to ensure equivalence in exchange, and follow the response hierarchy of: avoid > reduce > reverse land degradation (Orr *et al.* 2017).

Property rights

Well-defined property rights on common property resources (e.g., forests and rangelands) and tenure security on agricultural lands are efficient ways to internalize externalities arising from these land uses (Panayotou, 1994). Halting forest and rangeland degradation through the adoption of community-based management - facilitated by common property regimes - has been successful in many places and contexts across the world (Agrawal & Ostrom, 2001; Ostrom, 1990, 1999). Establishing a land rental market for agricultural land could support sustainable farming (Sklenicka, 2016). For example, the emergence of land rental markets in central and eastern European countries, after 1990, helped to reduce land fragmentation and potential land degradation following the decommissioning of state farms (Sklenicka, 2016).

Although the costs of inaction in the face of global land degradation almost always outweigh the costs of actions (Giger *et al.*, 2015), a severe lack of investments on sustainable land management often persists, because appropriate effective incentive structures are virtually inexistent - especially for private landholders (Mirzabaev *et al.*, 2016). Box 6.8 presents various examples of the economics of land degradation and highlights the need for secure land tenure, information and market access, and appropriate incentive structure to halt or reverse land degradation.

Box 6.8 Case studies on economics of land degradation and improvement

In Sub-Saharan Africa, low livestock productivity was found to be a major cause of land degradation and conversion (rangeland to cropland) (Nkonya *et al.*, 2016). Results show that adoption of soil fertility enhancing practices, as a solution, requires improvement in market infrastructure (i.e., market access and advisory and extension services) along with the provision of appropriate incentive schemes (Nkonya *et al.*, 2016). As an incentive, conditional fertilizer subsidies were effective in promoting use of nitrogen-fixing trees in agroforestry systems.

In Central Asia, the key factors in promoting the adoption of sustainable land management practices include: better market access; access to extension; private land tenure; learning from other farmers; livestock ownership; lower household sizes; and lower dependency ratios (Alisher Mirzabaev *et al.*, 2016).

In an analysis of nationally-representative household surveys, Gebreselassie *et al.* (2016) found that access to agricultural extension services, secure land tenure and market access are important incentives for sustainable land management and its associated technologies. In addition, collective action to manage grazing lands and forests - fostered by local institutions - can successfully address land degradation.

In Niger, Moussa *et al.* (2016) found that enhancing government effectiveness - by giving communities a mandate to manage natural resources and incentivizing land users to benefit from their investment - played a key role in realizing simultaneous improvements in land management and human welfare.

In a total economic value-based study on the drivers of land degradation in India, Mythili and Goedecke, 2016 found that agricultural input subsidies and “decreasing land-man ratios” are two major determinants of land degradation at state levels - suggesting that reform of environmentally-harmful input subsidies is necessary. A similar study from Kenya, Tanzania and Malawi found that halting land degradation involves secured land tenure, improved market access and extension services on sustainable land management practices among agricultural households (Kirui, 2016; Mulinge *et al.*, 2016).

The Chinese national ecosystem assessment (2000-2010) reported that investment in restoration and preservation of natural capital has improved the provision of major ecosystem services at the national level, although with very little effect on habitat loss and environmental pollution (Ouyang *et al.*, 2016).

Natural Capital Accounting as a response to land and ecosystem degradation

Land degradation and loss of biodiversity are symptomatic of the failure to account fully for the value of natural capital in decisions made by individuals, businesses and governments (MA, 2005; Groot *et al.*, 2010). Natural capital accounting involves integrated physical and monetized accounts that show the type, quantities and qualities of the stocks of renewable and non-renewable natural assets, including land and biodiversity based assets - available and used, in a country or region - and the diversity of flows of services generated by them (ONS, 2017; TEEB, 2012). Examples include the UN’s System of Environmental-Economic Accounting (UN, 2014) and the World Bank’s Wealth Accounting and the Valuation of Ecosystem Services Partnership (WAVES, 2017). Natural capital accounting has also been used to design and justify business responses to environmental pressures and corporate responsibilities, including the management of land and biodiversity impacts (TEEB, 2012) (see Section 6.4.2.4 on corporate social responsibility).

To date, most progress in natural capital accounting has been made in the development of physical accounts of asset stocks and service flows as a basis for subsequent valuation (Guerry *et al.*, 2015; UNDESA, 2017), usually with a focus on land use and conversion (EEA, 2016; EU, 2013), land and soil degradation (EEA, 2016; EU, 2014; Graves *et al.*, 2015; Robinson *et al.*, 2014; Robinson *et al.*, 2017) and biodiversity loss (UNEP-WCMC, 2016a). For example, natural capital accounting supported actions in the

Uganda National Development Plan II to restore degraded ecosystems (UNEP-WCMC, 2016b) - focusing on spatially-specific land cover, ecosystem extent, non-timber forest products and iconic mammals. Losses of natural ecosystems were associated with land conversion to agriculture, particularly for forests (29% remaining) and moist savannahs (32% remaining). From a policy response perspective, the accounts show that protected area designations performed well by avoiding the loss of natural ecosystems and securing benefits of managed wildlife tourism. Large areas of potentially natural vegetation were identified for sustainable harvesting of non-timber forest products, simultaneously maintaining species richness (UNEP-WCMC, 2016b).

The potential of natural capital accounting rests on the integration of physical and economic assessments (Remme *et al.*, 2014, 2015) in order to inform policy choice. Using the case of Kalimantan, Indonesia, Sumarga *et al.* (2015), show how natural capital and ecosystem accounting supports land-use planning through improved understanding of trade-offs between agriculture, forestry, carbon sequestration, wildlife and recreation services - especially when there is pressure to convert land to plantations. In the context of Small Island States, natural capital values, for international tourism, informed the introduction of a Green Departure Tax on tourists to fund protection of coastal biodiversity – for example, in the Republic of Palau, Micronesia (Weatherdon *et al.*, 2015). Hein *et al.* (2016) use cases of natural capital accounting from Europe and North America to value existing and likely future capacity to supply ecosystem services associated with, for example, soil organic carbon, timber harvesting and scenic views. Ruckelshaus *et al.* (2015) review experience of moving from natural capital accounting's "promise to practice", including its use in over 30 payment for ecosystem services and investment planning projects in Latin America (Box 6.9).

Despite numerous natural capital accounting initiatives and pilot projects, and the awareness it raises (Guerry *et al.*, 2015), the use of natural capital accounting for actual policy decisions remains relatively low, especially in developing countries (Edens & Graveland, 2014). A survey of 42 respondents from 17 countries (Virto *et al.*, 2018) showed that data availability and institutional barriers - including lack of political support and leadership - have constrained progress in adoption of natural capital and ecosystem accounting. In a first instance, rather than attempting to devise comprehensive natural capital accounting assessments of land-based ecosystems (Bartelmus, 2015), a staged, interactive approach focused on key indicators of land and biodiversity condition, as well as the economic consequences of change, may be more effective (Ruckelshaus *et al.*, 2015).

While mainstreaming natural capital has its supporters (Daily *et al.*, 2011; Remme *et al.*, 2015; Ruckelshaus *et al.*, 2015), the capitalisation of land and biodiversity values can: marginalize other culturally-resonant evaluative criteria (Sullivan, 2014); be confined to the "the nature that capital can see" or measure (Robertson, 2006); and serve to reinforce established worldviews, entitlements and practices dominated by political and economic imperatives (Robbins, 2012). Nonetheless, natural capital accounting can serve as a monitoring response to assess changes in the physical state and value of natural capital (land, biodiversity and ecosystem services) and as an evaluation tool to support decisions by governments and businesses - provided that an inclusive and collaborative approach is used to incorporate cultural and social values.

Box 6.9 Natural Capital, Ecosystem Accounting and Watershed Management in Colombia (Source: Ruckelshaus *et al.*, 2015)

Natural capital accounting was used to guide investment priorities and payments for watershed services under the Water for Life and Sustainability programme in Cali, Colombia. The programme was funded by water users, including sugar growers and producers, The Nature Conservancy and local NGOs. Working with stakeholders and drawing on biophysical data and local knowledge, a combination of simple scenario modelling and ranking of options was used to explore preferred watershed outcomes. Investment portfolios were drawn up, including options for grazing control, silvopastoralism, reforestation and restoration of degraded land. Working with available data, biophysical models contained in the *INvest* model were used to explore the effect of land-use change on erosion, sediment loss and/or retention and water yield. Options were assessed on their relative cost effectiveness to deliver target outcomes and then selected up to the limit of available funds. This more ‘”data and resource intensive”, yet better targeted approach, gave an estimated threefold increase in return on investment for sediment retention compared with investments based on participants’ general willingness to fund. Lessons from this experience are being used to support initiatives on over 30 new watershed funds in Latin America (Guerry *et al.*, 2015; Ruckelshaus *et al.*, 2015).

Figure 6 12 Mixed land-use mosaic and forest restoration in the Cali River Watershed, Colombia. Photo: courtesy of James Anderson under CC BY-NC-SA 2.0

These economic valuation and incentive-based instruments provide governments, NGOs and the private sector additional avenues to assess and avoid degradation of land, biodiversity and ecosystem services. However, a careful assessment of the limitations and suitability of these instruments is needed before using them in given social and cultural contexts. In policy practice, a mix of policies and regulations are usually required to define minimum environmental standards and restrictions on practices known to result in unacceptable environmental risk. By harnessing market forces to achieve intended outcomes, economic instruments are often used to complement, rather than substitute, legal and regulatory instruments and locally evolved institutions for environmental governance (Barton *et al.*, 2013; Cashore &

Howlett 2007). The current enthusiasm for monetization and market-based mechanisms in natural resource management - such as natural capital accounting and payment for ecosystem services - has potential for mobilizing new sources of funding for land degradation remedies; despite uneven access and fairness of these market-based mechanisms (Andersson *et al.*, 2011).

Benefits and costs of ecological restoration

Landowners, communities, governments and private investors need to understand the immediate and long-term costs and benefits of restoration activities in order to make optimal restoration investment decisions (BenDor *et al.*, 2015). The literature on full cost-benefit analyses of restoration projects is scarce (Aronson *et al.*, 2010; Bullock *et al.*, 2011): either restoration costs are not fully accounted for or the benefits to society are not examined in detail (De Groot *et al.*, 2013). For example, out of over 20,000 restoration case studies examined by The Economics of Ecosystem and Biodiversity initiative, only 96 studies provided meaningful cost data, with significant variations in costing methods and breadth and quality of cost-related information (NeBhoever *et al.*, 2011). Nevertheless, it is clear that restoration costs vary with restoration aims, timescales considered, the degree of degradation, ecosystem type and restoration methods used (Aronson *et al.*, 2010; Bullock *et al.*, 2011; Daily, 1995; De Groot *et al.*, 2013; NeBhoever *et al.*, 2011; UNCCD, 2017; Verdone & Seidl, 2017). Similarly, on the benefits end, most available studies often only considered financial benefits or private benefits (Barbier, 2007; De Groot *et al.*, 2013). Failure to incorporate a broader set of non-marketed values of restoration - such as the provision of wildlife habitat, climate change mitigation and other ecosystem services (Barbier, 2007; De Groot *et al.*, 2013) - discourages public and private investment in restoration projects (Verdone & Seidl, 2017). In addition, the use and choice of discount rates to assess present value of future benefits, an unresolved issue in the literature, affects net estimated benefits of restoration (Farber *et al.*, 2006). Some ecosystem service values cannot be monetized (e.g., cultural services that reflect spiritual values) and hence require a different approach than monetary valuation to estimate their value. However, recent advances in valuing non-marketed benefits of ecological restoration, and subsequent incorporation of such values and a wider range of social discount rates in cost-benefit analyses of restoration projects, still point to restoration investments being economically beneficial (De Groot *et al.*, 2013; Verdone & Seidl, 2017).

A study of fourteen Latin American countries estimated annual losses from desertification at 8-14% of agricultural gross domestic products (Morales *et al.*, 2011), while another study estimated the annual global cost of desertification at 1-10% of agricultural gross domestic products (Low, 2013). Using the benefit transfer method, Costanza *et al.* (2014) found that, across all biomes, the ecosystem service values lost due to land degradation and conversion ranges from $4.3 to $20.2 trillion per year. In a study that specifically considered only the values of managed forests (for wood, non-wood and carbon sequestration) and natural forests (for recreational values, passive use values and carbon sequestration values), Chiabai *et al.* (2011) estimated that projected degradation and land-use change would cost $1,180 trillion in forest ecosystem services, over a 50-year period (2000-2050). While these studies provide useful indications of the magnitude of land degradation costs, the many challenges in estimating the cost of land degradation at local and national scales remains a challenge for quantifying costs at the global level.

Box 6.10 Cost-benefit analyses of restoration

In a meta-analysis of restoration projects in over 200 studies that considered costs (i.e., direct costs, capital costs and management costs of restoration process, but not the opportunity costs) and known benefits (ecosystem services, not other indirect benefits), De Groot *et al.* (2013) reported that only 94 estimates on costs and 225 estimates on benefits of ecological restoration were found across 9 major biomes, including coastal systems, coastal wetlands, inland wetlands, freshwater rivers and/or lakes, tropical forests, temperate forests, woodlands and grasslands. The mean total economic value (in 2007 $/ha/yr) of all ecosystem services from these biomes were estimated at $28,917, $193,845, $25,682, $4,267, $5264, $3013, $2588, $2871, respectively. Cost estimates included original restoration costs, 5% per year maintenance costs as the financial costs of capital from year 2 onwards and 2.5% for coastal and wetland systems - whilst the benefits included the sum of the monetary values of 22 ecosystem services in the form of total economic value estimates. The project costs vary between several hundreds to thousands of $/ha (for grasslands, rangelands and forests) to several tens of thousands (inland waters) (Neßhöver *et al.*, 2011). De Groot *et al.* (2013) considered 12 alternatives scenarios: 6 based on 100% maximum restoration costs under 3 benefit scenarios (75%, 60% and 30% of the mean benefit values) and 2 discount rate scenarios (-2% and 8%); and 6 based on 75% maximum restoration costs under 3 benefit scenarios (75%, 60% and 30% of the mean benefit values) and 2 discount rate scenarios. Under all possible scenarios, the benefit-cost ratios were greater than 1.0 for inland wetlands, tropical forests, temperate forest, woodlands and grassland biomes – with the highest (35) for grasslands under a best-case scenario (75% restoration costs, 75% benefits at -2% discount rate), and less than 1 for coastal systems, freshwater, and coastal wetlands under a worst-case scenario (100% restoration costs, 30% benefits and 8% discount rate). While considering a slightly modified benefits (100% and 60% of total economic value), costs (100% and 130% of typical restoration costs), discount rate (-2%, 2% and 5%), and two-time horizons (20 years and 30 years) scenarios for the same 9 biomes, Blignaut *et al.* (2014) reported that the average benefit-cost ratio varies between 0.4 (for coastal systems) and 110 (for coastal wetlands) with most of the biomes at about 10 on average.

A recent cost-benefit analysis of the Bonn Challenge - a global initiative initiated in 2011 with the aim to restore 350 million hectares of degraded forest and agricultural land by 2030 - provides new insights on the value of investing in restoration (Verdone & Seidl, 2017). In this analysis, the extent of degraded area was based on the Global Assessment of Soil Degradation (GLASOD), calibrated to determine areas of degraded, managed and natural forests in each forest biome and across 12 world regions (Verdone & Seidl, 2017). It considered different benefit types (private, public or both), land degradation types (light, moderate, extreme or severe), forest management types (natural or managed) and discount rates (4.3% following Nordhaus, 2014 and 1.3% following Stern, 2007). In this analysis, average costs of restoration ranged from $214-3790/ha (mean: $1276 ± $887/ha); based on comprehensive data from a World Bank project database and TEEB reports for four degradation levels: *light* (mean - one standard deviation); *moderate* (mean); *severe* (mean + one standard deviation); and *extreme* (mean + 2 standard deviations). As one would expect, the average restoration costs increased with the extent of degradation: $389, $1276, $2163, and $3051/ha in the light, moderate, severe and extreme degradation categories, respectively (Verdone & Seidl, 2017) (cf. http://www.worldbank.org/projects and teebweb.org for more information). Estimated benefits of forest restoration, in terms of wood products (including wood fuel), were derived following Chiabai *et al.* (2011) - with adjustments for expected productivity losses of wood products due to degradation (i.e., 10%, 25%, 50% and 100% for light, moderate, severe and extreme degradation levels) (Daily, 1995). Benefits for services - including recreation and passive use benefits - were derived from a meta-analysis of 59 and 27 studies, respectively, and carbon sequestration benefits from a study on social costs of carbon sequestration ($43.46/ton) (Nordhaus, 2014). The results of this analysis suggest that achieving the Bonn Challenge target of restoring 46% of the world's currently degraded (managed and natural) forests would costs $0.299 trillion - providing a net present value of benefits of $2.254 trillion (benefit-cost ratio of 7.54, considering both private and public benefits from forests at a 4.3% discount rate), $0.565 trillion (benefit-cost ratio 1.88, considering only private benefits at a 4.3% discount rate) and $9.245 trillion (benefit-cost ratio 30.92, considering both private and public benefits at a 1.3% discount rate) (Verdone & Seidl, 2017). In the case of a "private benefits only" scenario, only 197 million ha could be profitably restored, and to meet Bonn Challenge restoration target governments would have to provide landowners a total subsidy of approximately $139 billion or $911/ha (also see Chapter 5, Section 5.2.3.4).

Conventionally, restoration is viewed by countries as a cost to be paid, rather than an investment that has tangible, beneficial returns (Bullock *et al.*, 2011). However, the available evidence strongly supports the view that restoration of degraded lands is a worthwhile investment that brings multiple benefits and can outweigh costs (Blignaut *et al.*, 2014; Bullock *et al.*, 2011). For example, in a study of large-scale landscape restoration in Mali, Sidibé *et al.* (2014) found that adapting agroforestry is economically beneficial at the local and global levels; providing local benefits to farmers in the range of $5.2 to $5.9 for every dollar invested and with net present values ranging between $17.8 and $62/ha/yr when discounted at 2.5%, 5%, and 10% over a time horizon of 25 years. When carbon sequestration is integrated in the analysis, practicing agroforestry and reforestation options yield up to $13.6 of benefits for every dollar invested (at a discount rate of 5%), equivalent to a value of $428.8/ha/year.

Investments in restoration have also been found to create jobs. Using an input-output model to estimate the direct, indirect (business to business) and induced (household spending) impacts of restoration on the economy in the USA, BenDor *et al.* (2015) analyzed 45 restoration programmes with an average programme cost of $44.4 million. Their analysis indicated that the number of jobs created per $1 million invested in restoration programmes range from 6.8 wetland restoration at county level (Department of the Interior, 2012) to 39.7 on national level forest, land and watershed restoration (Pollin *et al.* 2008). Moreover, the number of direct, indirect and induced jobs supported by these projects ranged from 14.6 per $1 million invested for hydrologic reconnection, to 33.3 per $1 million invested for invasive species removal. In the State of Oregon, the number of jobs supported by restoration projects ranged from an estimated 14.7 jobs/$1million invested for in-stream restoration to 23.1 jobs per $1 million invested for riparian restoration (Nielsen-Pincus & Moseley 2010). The employment multiplier ranged from 2.7 to 3.8 and economic output multipliers ranged from 1.9 to 2.4 for all projects. In Massachusetts, ecological restoration investment supported about 9.9 jobs per $1 million for wetland restoration (with dam removal) to 12.9 jobs per $1million invested for tidal creek recreation (Industrial Economics Inc., 2012).

The employment multiplier for the restoration industry ranged from 1.48 (Edwards *et al.*, 2013) to 2.87 (Shropshire & Wagner, 2009) and corresponding output multipliers are 1.60 and 2.59, respectively. The employment multiplier of restoration projects is comparable to that of other industries, including the oil and gas industry (Price Waterhouse Coopers, 2011), agriculture, livestock and outdoor recreation industry - with employment multipliers of 3.0, 2.33, 3.34 and 1.97, respectively (BenDor *et al.*, 2015). In a national survey of businesses that participate in restoration work in the USA, BenDor *et al.* (2015) estimated that direct employment of 126,000 workers generates $9.5 billion in economic output (sales) annually. The indirect linkages and increased household spending - through restoration-related investment - accounts for 95,000 additional jobs and $15 billion in economic output (BenDor *et al.*, 2015).

Despite the increasing awareness of the importance of natural ecosystems and sustainably managed working lands, in conserving biodiversity and providing ecosystem services - as well the social, economic and ecological benefits to be derived from rehabilitating degraded lands - investments in restoration are hampered by the typically short time horizon of private investment and land-use decisions, including low discount rates applied in economic analyses. For example, when forest restoration is viewed from a financial accounting lens that ignores public values and the intergenerational nature of forest restoration, it discourages investment despite the long-term societal benefits. Fulfilling large-scale restoration goals requires creating economic incentives and schemes (e.g., payments for ecosystem services and REDD+) that encourage landowners to recognize and capture public values of restoring degraded land, particularly in severely degraded landscapes.

6.4.2.4 Social and cultural instruments

Social and cultural instruments used to halt land degradation and restore degraded lands include: community-based (participatory) approaches in natural resource management; the integration of indigenous local knowledge and practices in land restoration and reclamation; public engagement and awareness-raising (eco-labelling, certification, education and/or training); corporate social responsibility; and voluntary agreements, amongst others. The complex and dynamic nature of land degradation drivers and processes requires flexible approaches to halt land degradation – which embrace a diversity of social and cultural knowledge and values from public and private sectors (Scherr, 2000; Shiferaw *et al.*, 2009).

Participatory approach in resource management and governance

Community-based natural resource management is a participatory approach for natural resource management and governance prevalent in many countries. It allows devolution of authority to local users to exercise their rights to manage and govern these resources. Decentralized community-based approaches have been proven effective in restoring degraded forests and conserving soils and water in many parts of the world (Agrawal & Ostrom, 2001; Ostrom *et al.*, 1999); including Australia, where involving indigenous communities in such approaches has been effective (Hill *et al.*, 2013; Pert *et al.*, 2015). In Nepal, the development and practice of community forestry since the late 1970s has been a successful response to halt deforestation and reduce the severity of associated soil erosion and landslides, prevalent in 1960-70s (Eckholm, 1976; Pandit & Bevilacqua, 2011) . This has involved devolution of forest management and governance authority to local forest users organised into “community forest user groups” (Acharya, 2002). As a result, it is estimated that the forest area in Nepal has increased from 37.4% in 1985-86 to 40.4% in 2015 (DFRS, 2015) (see Box 6.13).

Despite anecdotal evidence on the successes of community-based resource management, a meta-analysis of 41 studies from 13 countries in Asia, Africa and Central America focusing on three types of outcomes (forest condition and land cover, resource extraction and livelihoods) found that community- based forest management was associated with improved forest condition (i.e., greater tree density and basal area), but not with other indicators of global environmental benefits (Bowler *et al.*, 2012). The effectiveness of community forestry varies greatly with specific contexts, rights and management rules (Robinson *et al.*, 2014), and the main factors affecting effectiveness include forest area per person, level of monitoring and clarity regarding property rights (Nagendra, 2007; Pagdee *et al.*, 2006) .

Stakeholder participation in resource management and governance – supported by institutional structures and policies – can effectively facilitate interventions designed to halt land degradation or restore degraded lands (Reed & Dougill, 2008). For instance, improved land tenure in the Philippines has been associated with effective soil conservation (Briones, 2010), which in turn help to maintain land productivity and provide a form of safety net for farmers. On the other hand, scholars also note that a rise in insecure land tenure, involving both family and communal land, has been a major cause of unsustainable land use (Agrawal, 2002; Ostrom, 1990). Within community-based forest management or restoration programmes, Geist and Galatowitsch (1999) found that knowledge transfers in these programmes enhance social learning and self-esteem of the participants.

Cultural considerations on land use and management

Cultural context influences the choices that people make regarding land-use practices, in both long and short time frames. The drivers of land degradation from a cultural perspective include: changing cultural context of land; loss of cultural identities; and loss of cultural relevance of place-based indigenous and traditional ecological knowledge (Agrawal, 2002; Berkes, Colding, & Folke, 2000; Hartmann *et al.*, 2014; Ostrom, 1990; Parrotta & Trosper, 2012) (see Chapter 2, Section 2.2.2 and Chapter 3, Section 3.3.2.1).

Effective cultural responses to land degradation and restoration include the maintenance of traditional land-use practices and support for traditional knowledge which commonly underpins these practices (also see Sections 6.3.1.1 on agricultural practices and 6.3.1.2 on forestry practices). There is considerable evidence that the disparagement of the epistemological values and perspectives of traditional (particularly indigenous) communities that view nature/land and culture/values as indivisible (Claus *et al.*, 2015; Hartmann *et al.*, 2014), has been a major factor behind both the commercial exploitation and degradation of lands, as well as conservation measures that exclude traditional uses (Hartmann *et al.*, 2014; Parrotta & Trosper, 2012). The preservation or revival of ILK – and associated local and indigenous land-use practices – have been key to cultural resurgence and improvements in land management practices to avoid degradation in many parts of the world (Berkes, 2017; Berkes *et al.*, 2001; Corntassel & Bryce, 2012; Dublin et al, 2014; Parrotta & Trosper, 2012; Trosper, 2017; Ramakrishnan, 2002). Long *et al.* (2003) describes how youth ecology camps – where tribal adults teach youths how to care for their land – is an effective way to promote: restoration in more subtle ways; the passing on of cultural traditions sustaining the collective action needed for successful restoration work by providing a vision for restoration; a sense of place and community; and guidance for decision-making. In successfully opposing mining and logging operations on their traditional lands, many indigenous groups have also reproduced and transformed their identities and worlds (Poirier, 2010) through innovative practices around their land-based resources (Haglund *et al.*, 2011).

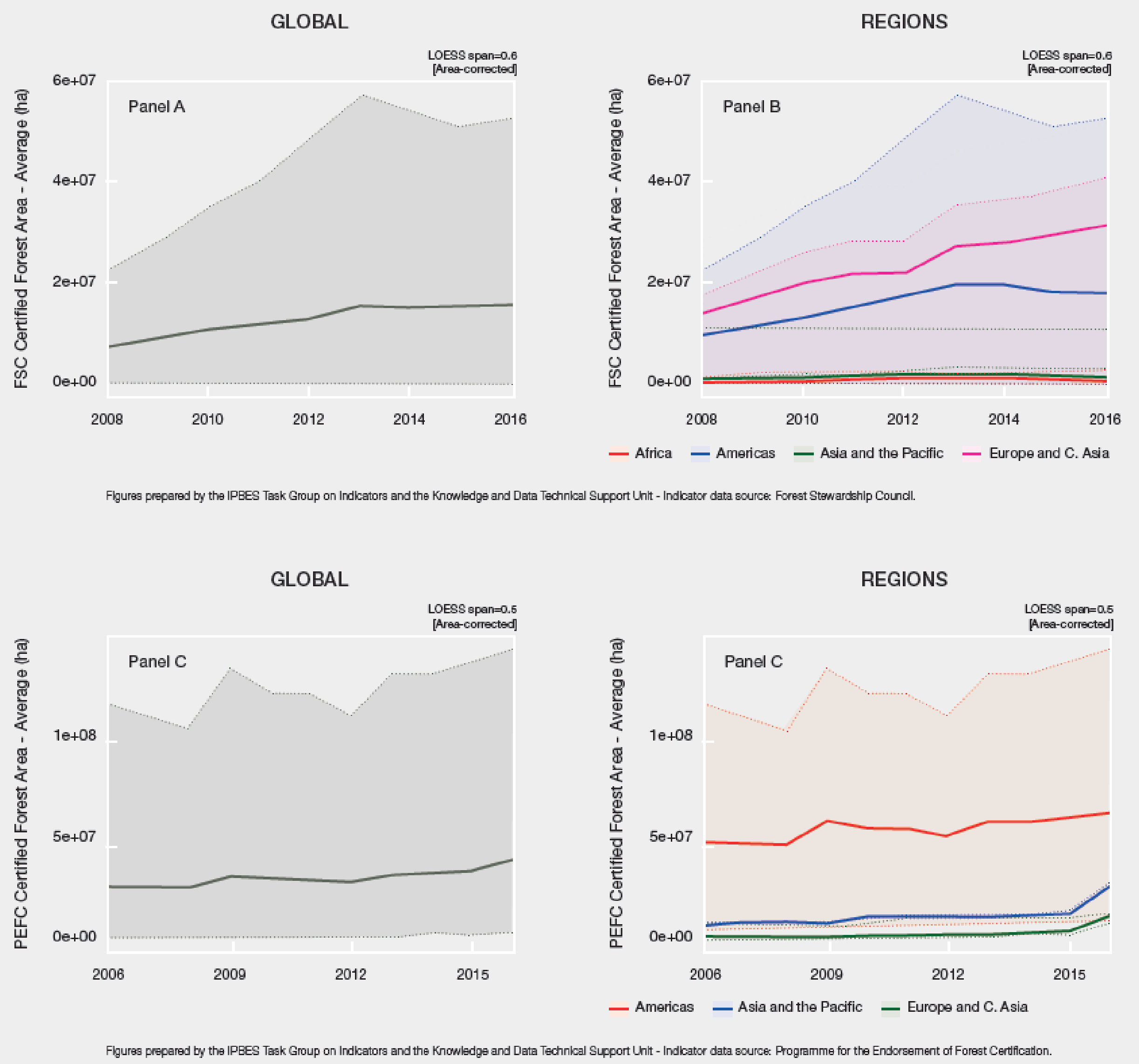
The adoption of soil conservation measures often faces cultural barriers when their implementation is perceived as a cost to local farmers, while benefits accrue at regional to global levels (Knowler & Bradshaw, 2007). Farmer decisions are strongly influenced by socio-economic factors (role of subsidies, quotas, cost savings) (Boardman *et al.*, 2003; Lahmar, 2010) and changing farmers' practices is a challenge for the adoption of voluntary soil conservation measures (Mbaga-Semgalawe & Folmer, 2000; Sattler & Nagel, 2010; Wauters *et al.*, 2010). In such contexts, participatory approaches have been found to be effective in promoting the adoption of soil conservation measures (Bewket, 2007), with economically- and environmentally-beneficial outcomes (Shiferaw & Holden, 2000).

A deliberate focus on otherwise "hidden" or "hard-to-value" cultural aspects such as the revitalization of ILK-based cultural practices (Hartmann *et al.*, 2014; Kittinger *et al.*, 2016) and faith-based beliefs (Cochrane, 2013) has been found to yield positive outcomes for halting and reversing land degradation. However, since ILK and associated natural resource management practices are influenced by history and contested locally, their representations within collaborative land restoration efforts can also trigger dissatisfaction amongst participants (Shepherd, 2010). For instance, the literature produced around the REDD+ programme has described how the matter of community tenure rights is also an extremely contentious issue given the inevitable vested interest of the dominant actors (e.g., government agencies, local elites) to maintain a dominance over land ownership (Ngendakumana & Bachange, 2013).

Certification

Eco-certification (or eco-labelling) is a voluntary instrument that has been applied to certain crops and forest products (e.g., coffee and timber). In principle, eco-certification enables consumers who prefer “green goods” to identify the good and purchase them in a price-differentiated market, which can address the environmental problems associated with production of goods by creating incentives for producers, otherwise difficult to handle with regulatory instruments alone (Lambin *et al.*, 2014). Studies examining the impacts of eco-certification schemes have found limited economic benefits of certification, but significant social and environmental benefits. In comparing certified and non-certified coffee growers and their land-use practices, certified coffee growers were found to be adopting environmental-friendly practices in Colombia (Rueda & Lambin, 2013) and they had a higher biodiverse coffee farms in Mexico (Mas & Dietsch, 2004). Eco-certification of forest products – through, for example, the Forest Stewardship Council (FSC) or the Program for the Endorsement of Forest Certification – provides some assurance that these products are from a responsibly managed forest (natural, semi-natural and plantations) with respect to: biodiversity conservation; the protection of critical ecosystem services; and the promotion of social, economic, cultural and ethical dimensions of sustainable forestry. While there is little evidence of positive environmental or socio-economic impacts of forest product certification, at the global level (Dauvergne & Lister, 2010), positive local impacts have been documented in Brazil, Malaysia and Indonesia (Durst *et al.*, 2006). In Indonesia, the effectiveness of FSC on social and environmental outcomes was evaluated using matching technique between FSC-certified timber concessions and non-certified logging concessions (Miteva *et al.*, 2015) . They estimated that between 2000 and 2008, FSC reduced aggregate deforestation by 5%. In addition, they note that FSC reduced firewood dependence by 33%, respiratory infections by 32%, and malnutrition by 1% on average across participating households (Miteva *et al.*, 2015). Figure 6.13 shows the area of certified forests under FSC and the Program for the Endorsement of Forest Certification schemes – indicating that certified forest area is on the rise at global and regional levels, with some regional differences. In 2016, Canada (>50 billion ha) and Finland (17 billion ha) had the greatest areas of certified forests, at the country level, under FSC and the Program for the Endorsement of Forest Certification schemes, respectively (IPBES, 2017).

Figure 6 13 Annual certified forest areas managed under Forest Stewardship Council (Panel A and B) and Endorsement of Forest Certification (Panel C and D) schemes at global and regional levels.



Corporate social responsibility

Among other forms of corporate social responsibility, natural capital accounting has also been used to design and justify business responses to environmental pressures and corporate responsibilities, including the management of land and biodiversity impacts (TEEB, 2012). Natural capital accounting broadly follows the accounting conventions of balance sheets and profit and loss accounts to reflect natural assets and service flows respectively, as well as exposure to natural capital risk (Trucost, 2013). Of particular interest is the Natural Capital Coalition (NCC, 2016), comprising over 250 collaborating organizations, which has produced The Natural Capital Protocol: a standardized framework supported by a toolkit to identify, measure and value impacts and dependencies of businesses on natural capital. The Coalition has assembled over 60 cases studies of natural capital accounting assessments and responses, half of which cover specific corporate applications and half covering topic- and location-specific cases (NCC, 2016). Many contain data and methods that may be applicable for use elsewhere. For example, Denkstatt (2016) used The Natural Capital Protocol to review water replenishment options for the Coca Cola Company showing, for example, that wetland restoration provided particularly high benefits beyond those linked to water conservation alone. Novartis, a multinational pharmaceutical company, used the Protocol to assess the monetized impact on natural capital for the Novartis Group and its supply chain (reported in NCC, 2016). For Novartis operations in Argentina, it was shown that alongside initiatives to improve energy and material use, contributions to forestry projects (prompted by the desire to offset the company's environmental footprint) generated net positive benefits through carbon sequestration, increased biodiversity and watershed protection. The approach has been integrated into the company's Financial Social and Environmental Accounting system and its Corporate Responsibility programme. In a similar vein, Hugo Boss used the natural capital accounting framework to assess the effects on ecosystems services of the supply chains for their cotton, wool and leather fashion goods (Zeller *et al.*, 2016). In their case, cotton cultivation and sheep farming accounted for large shares of monetized natural capital impacts for the clothing sector, while tanning processes dominated environmental costs for footwear. The assessment is being used to promote environmental provenance in the supply chain for their products, including the use of natural, less environmentally-burdensome substitute materials and processes. Despite these notable efforts, systematic reviews of the empirical evidence on direct correlation between corporate social responsibility and prevention of land degradation are scarce.

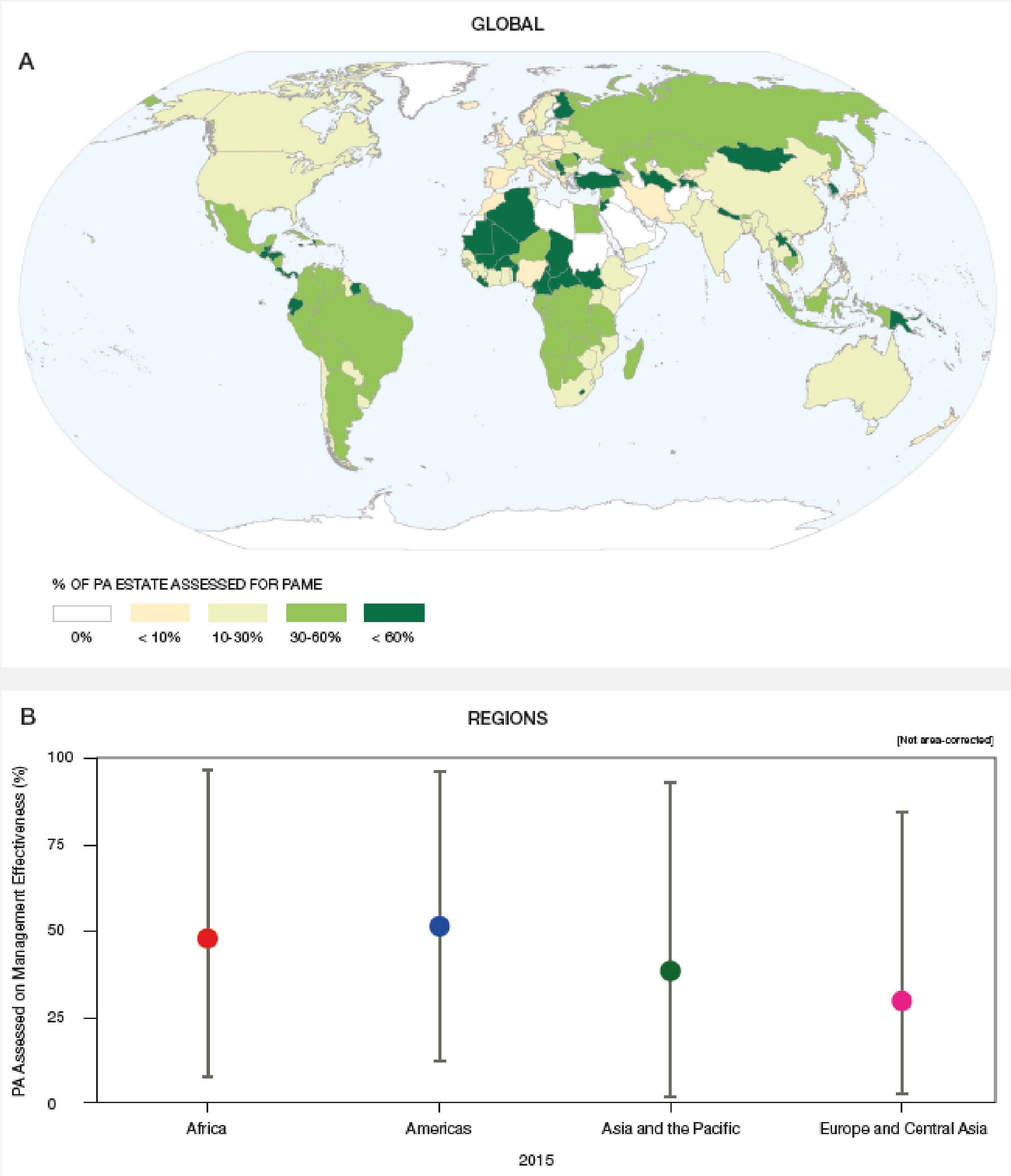
6.4.2.5 Protected areas

Protected areas are widely regarded as one of the most successful measures implemented for the conservation of biodiversity. The global community has committed to protect 17% of terrestrial areas by 2020, in line with Aichi Biodiversity Target 11 (Pringle, 2017; SCBD, 2014).

Since the mid-1990s, various methodologies have been developed for assessing protected area management effectiveness. Assessment data from all over the world have now been collated in the Global Database for Protected Area Management Effectiveness which contains records of almost 18000 assessments of protected area management effectiveness. The database includes information about the methodologies and indicators used, and records details of individual assessments. As of January 2015, nearly 18,000 of the assessments had been collated in the database, representing over 9000 protected areas, with 3,666 sites having multiple assessments. Some 17.5% of countries have already assessed the effectiveness of 60% of their protected areas. The differences in proportion of protected area assessed for effectiveness, by country and region, are given in Panel A and B in Figure 6.14.

Figure 6 14 Proportion of protected area assessed for management effectiveness by country (Panel A) and region (Panel B).

Source Panel A: Coad *et al.* (2015); Source Panel B: Figures prepared by Task Group on Indicators and Knowledge and Data Technical Support Unit - Indicator data source: UNEP-WCMC and IUCN (2016), Protected Planet: The Global Database on. Protected Areas Management Effectiveness (GD-PAME) [On-line], [Oct 2016], Cambridge, UK: UNEP-WCMC and IUCN. Available at: www.protectedplanet.net



Empirical evidence on protected area management effectiveness is mixed. A systematic literature review of impact evaluation papers that used a composite-single indicator for measuring effectiveness, (Coad *et al.*, 2015) found a positive correlation between overall management performance score and biodiversity outcomes for 5 of the 9 reviewed final studies (Henschel *et al.*, 2014; Zimsky *et al.*, 2010, 2012). It remains unclear whether this lack of correlation with the impact of protected areas in some studies is real, meaning either that protected area management has no impact on biodiversity outcomes or more plausibly that good management (as measured by protected area management effectiveness scores) is necessary but not sufficient to ensure effective conservation (Carranza *et al.*, 2014).

Protected area effectiveness correlates with basic management activities such as enforcement, boundary demarcation and direct compensation for local communities – suggesting that even modest increases in funding would directly increase the ability of designated parks to protect tropical biodiversity (Bruner *et al.*, 2001). Further evidence indicates that the rate of conversion of landscape is lower in protected areas. Examining the impact of protected areas globally – by matching analysis of protected and unprotected areas – Joppa and Pfaff (2011) found that legal protection had reduced landscape conversion in 75% of 147 countries. Using the same matching technique, Andam *et al.* (2008) evaluated the impact on deforestation of Costa Rica's renowned protected area system (between 1960 and 1997) and found that protection reduced deforestation. They argued that approximately 10% of the protected forests would have been deforested had they not been protected. Based on an assessment of the impacts of anthropogenic threats to 93 protected areas in 22 tropical countries, the parks were found to be an effective mean to protect tropical biodiversity by stopping land clearing, and to a lesser degree by mitigating logging, hunting, fire and grazing (Bruner *et al.*, 2001). In Dana Reserve, Jordan, degradation has been partially reversed by agreeing with local farmers and herders to reduce stocking density of goats by 50% and providing alternative livelihood options through ecotourism and craft development (Schneider & Burnett, 2000).

On the other hand, protected areas are not always effective in halting land degradation. Liu *et al.* (2001) examined remotely-sensed data before and after the establishment of the Wolong Nature Reserve (established in south-western China to protect pandas) and found that habitat loss and fragmentation inside the reserve had unexpectedly increased to levels that were similar to or higher than those outside the reserve. Watson *et al.* (2014) reviewed the history and effectiveness of protected areas and found that conservation would be effective by establishing protected areas that are large, connected, well-funded and well-managed. Focusing on understanding causes of land degradation and deforestation in the Wildlife Reserve of Bontioli (Burkina Faso), Dimobe *et al.* (2015) found that despite the classification of two protected areas, vegetation cover was reduced over a 29-year period due to conversion of woodland and wooded savannahs to agricultural lands. They concluded that this was due to the lack of long-term adaptive management and conservation strategies in the communal areas and recommended strengthening the scientific foundation for greater involvement of local populations and staff in conservation and management activities.

Indigenous protected areas as a response

Globally, 18% of land is formally recognized as either owned by, or designated for, indigenous peoples and local communities. Within the 18%, 10% is owned by indigenous peoples and local communities and 8% is designated for (or "controlled by") indigenous peoples and local communities (Rights and Resources Initiative, 2015). For example, Australia has included Indigenous Protected Areas as a key part of the National Reserve System, in recognition that indigenous Australians have managed their country for tens of thousands of years. There are 70 dedicated Indigenous Protected Areas across 65 million hectares – accounting for more than 40% of the area of the National Reserve System – which protect biodiversity and cultural heritage and provide employment, education and training opportunities for indigenous people (The Natural Resource Management Council, 2010).

6.4.2.6 Climate change adaptation planning

Even though climate change is a threat in itself as well as a threat multiplier (see also Chapter 3), adapting to climate change to avoid land degradation impacts is closely linked to land-based resource management (of croplands, forests, rangelands, urban lands, wetlands and so on). Specific responses to climate change mitigation and adaption based on land-use types have been discussed in earlier sections (such as cropland in Section 6.3.1.1 and forests in Section 6.3.1.2). The focus in this section is on climate change adaptation planning, noting however that assessing its effectiveness in terms of avoidance of future impacts is difficult partly due to high uncertainty around climate change itself (Füssel, 2007).

Given the pervasive influence of climate change on socio-ecological systems, climate change adaptation planning has important implications for land resource management and conservation (Lawler, 2009). Climate change adaption depends on a variety of factors including: land-use domains; adaptation purpose, timing and planned horizon; form and measures of adaptation (i.e., technical, institutional, legal, educational and/or behavioural); actors (people at different hierarchy levels from farmers to many public and private organizations); and general context (environmental, economic, political and cultural). Thus there is no single best approach for assessing, planning and implementing climate change adaptation measures (Füssel, 2007).

To design, plan and implement effective adaptation measures, certain pre-conditions should be fulfilled (Füssel, 2007) and adaption barriers need to be systematically identified (Moser & Ekstrom, 2010). Such pre-conditions for effective climate change adaptation planning include: awareness of the problem; availability of adaptation measures; information about the measures; availability of resources to implement the measures; cultural acceptability of the measures; and incentives for implementing these measures (Füssel, 2007). To enhance effectiveness of climate change adaptation plans and strategies, Moser and Ekstrom (2010) proposed a framework to diagnose the barriers, which is underpinned by four principles and consists of three components. The four principles underpinning the framework are: (i) socially-focused but ecologically-constrained; (ii) actor-centric but context-aware; (iii) process-focused but outcome and/or action-oriented; and (iv) iterative and messy, but linear for convenience (Moser & Ekstrom, 2010). Three components to identify adaptation barriers include:

i. *process of adaptation* – understanding the barriers, planning adaptation options and managing the implementation of adaptation options;
ii. *structural elements of adaptation* – the actors, larger context in which they act (governance and broader human-biophysical environment) and the system of concern (the object or system upon which they act); and
iii. *overcoming the barriers through interventions* – spatial and/or jurisdictional and temporal barriers (Moser & Ekstrom, 2010).

The uncertain and varying nature of climate change impacts in different places and land-use systems necessitates adaptive management, which has often been referred to as a critical adaptation strategy for resource management (Lawler, 2009). A broader spatial approach (e.g., landscape or regional approach) and temporal perspective (e.g., scenario-based planning) has been argued for climate change adaption planning to manage land and ecosystems (Lawler, 2009; Peterson *et al.*, 2003). For example, scenario planning allows managers and planners to evaluate multiple potential scenarios of change, for a given system, in order to develop alternative management goals and strategies (Peterson *et al.*, 2003) – which in turn enhance the effectiveness of an adaptive management approach (Lawler, 2009). In the context of climate change and managing forests in the future, Millar *et al.* (2007) suggest that management strategies should promote both resistance and resilience to climate change impacts in forest ecosystems. For example, restoring ecosystem functions of a degraded land through restoration would increase resilience of the system (Julius *et al.*, 2008). Similarly, Harris *et al.* (2006) argue that a focus on ecosystem structure in restoration planning – in the context of changing climate – is challenging and that a focus on process (ecosystem services) rather than structure (species composition) may be a preferred option.

Many industrialized countries have developed comprehensive national adaptation assessments (e.g., the USA and Canada) (Lemmen & Warren, 2004; Scheraga & Furlow, 2001), while adaptation assessments in developing countries have usually been conducted as a part of bilateral or multilateral assistance schemes (Leary *et al.*, 2013) or the National Adaptation Program of Action processes. In addition, adaptation to climate change has been increasingly considered in regional- and local-level planning (e.g., regional forest management plan of Western Australia; see Conservation Commission of Western Australia, 2013; and the City of Melbourne Climate change adaptation strategy and action plan; see Commonwealth of Australia, 2013). However, in a systematic review of climate change adaptation literature comprised of 39 studies from developed countries between 2006 and 2009, Ford *et al.* (2011) found limited evidence of adaptation actions, even in developed nations. Those adaptation interventions that are found in practice are localized (municipality level) and funded through higher-level government interventions mostly concentrated on transportation, infrastructure and utility sectors and based on non-structural adaptation responses (i.e., management strategies, plans, policies, regulations, guidelines or operating frameworks to guide planning) (Ford *et al.*, 2011). In addition, their review highlighted that stakeholder engagement in adaptation planning and implementation, and adaptation actions did not focus on vulnerable populations (Ford *et al.*, 2011).

Addressing land degradation through climate change adaptation planning requires a broad-base integrated and adaptive approach involving all affected stakeholders. The failure to mainstream cultural and economic considerations – relevant to land degradation into environmental or other sector policies – has led to policy failures in many countries, including several in Africa (Kiage *et al.*, 2007; Koning & Smaling, 2005). As countries are affected differently by climate change-induced land degradation, adaptation plans and their effectiveness will vary depending on the socio-economic context of the place or system in question. For example, in a survey of 127 agro-pastoralist households in Kenya, Speranza *et al.* (2010) found that poverty limited any responses related to markets, while lack of skills limited adaptation capacity to droughts and climate change. They conclude that building adaptive capacity through extension services, maintaining infrastructure and embedding indigenous knowledge in adaptation plans would be effective adaptation measures for agro-pastoral communities (Speranza *et al.*, 2010). Indigenous communities have adapted to change for centuries and their practices and knowledge provide effective responses in land management responses (Fisher, 2013).

6.4.3 Integrated landscape approach as a response

Three main approaches have been used to respond to land degradation and land restoration through land planning at different scales: (i) sustainable land management; (ii) zoning; and (iii) integrated landscape planning and management. Although they share general motivations and objectives, they have different specific reaches.

Sustainable land management

In order to achieve socio-economical goals, sectoral policies typically have particular objectives when it comes to land, for example: agriculture and grazing consider soil quality, water availability and connectivity to markets; mining projects analyse the territory in terms of mining demands and mining stocks; transportation and energy infrastructure sectors focus on efficiency in terms of technical feasibility and competitiveness; while the housing sector considers urban expansion and land availability. Consequently, each policy has its own “map”, with a biased and fragmented approach to land. This fractional approach to social and environmental issues can result in overlapping maps and in inequitable and unsustainable use and transformation of land.

To address these limitations, spatial management responses to land degradation at national, regional and local levels need to combine and complement sectoral planning in ways that improve the resilience of socio-ecological systems, while supporting social and economic development, by using scientific evidence-based land-use information and tools. This goal can be achieved by delineating and modelling changing scenarios, and through the promotion of coordinated and concerted actions involving governments, private sectors and civil society.

The land-use planning (zoning) approach

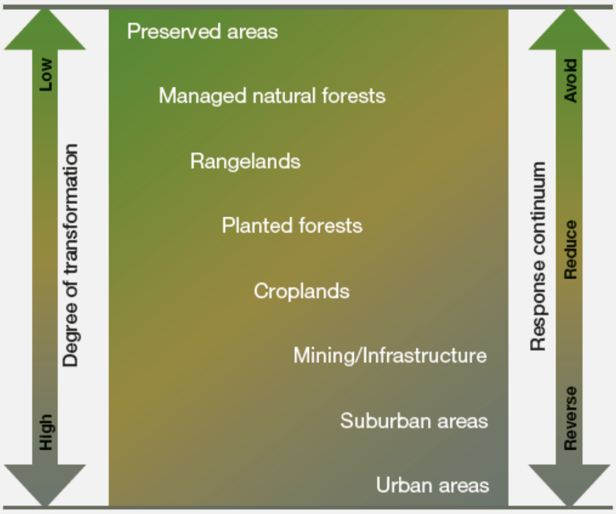
"Land-use planning is a systematic and iterative procedure carried out in order to create an enabling environment for sustainable development of land resources which meets people's needs and demands. It assesses the physical, socio-economic, institutional and legal potentials and constraints with respect to an optimal and sustainable use of land resources, and empowers people to make decisions about how to allocate those resources" (FAO & UNEP, 1995).

Land-use policies - which are often developed under spatial development frameworks at some administration level - involve spatial planning or zoning (i.e., allocation of the distribution, extent and intensity of land uses in a given landscape). Many jurisdictions have found that biodiversity conservation, sustainable resource management and the restoration of degraded habitats are best accomplished using a landscape-based approach. Pressures on the landscape and natural resources continue to grow due to increased population levels, urbanization and intensification of agriculture. An integrated, strategic landscape approach to biodiversity conservation is proving to be the most effective and efficient coordinate stewardship, resource management and planning activities.

Integrated landscape planning and management

An integrated landscape approach is a regulatory response to land-use planning and practice (see Section 6.4.2.1). It seeks to better understand the interactions between various land uses and stakeholders by integrating them in a joint management process (GLF, 2014) and is essential for development of sustainable land-use and livelihood strategies in rural areas (FAO, 2017). It allows for an encompassing consideration of a range of land uses in a given landscape – from pristine natural areas to highly transformed urban areas – into an integrated approach to make land-use decisions for multiple purposes and functions, as illustrated in Figure 6.15. Governments and organizations such as WWF, IUCN, and the World Bank argue that a landscape approach would bring environmental gains, enhance synergies and minimize trade-offs compared to sectoral approaches (e.g., agriculture, forestry, urban lands and so on) of managing lands within a resource-constrained context to reap more value from existing resources.

Figure 6 15 A schematic diagram showing the degree of land transformation (none or minimum in dark green colour to substantial transformation in dark grey colour) resembling land use types from preserved natural areas to urban areas with a response continuum (avoid, reduce and reverse).



Within the landscape approach for land conservation or restoration, scholars argue the merits of land sharing (i.e., wildlife-friendly farming) versus land sparing approaches (i.e., intensification of production to maximize agricultural yield) (Collas *et al.*, 2017; Law & Wilson, 2015; Mertz & Mertens, 2017; Phalan *et al.*, 2011). A landscape approach that embraces an integrated land-sharing philosophy has been increasingly promoted in science, and in practice, as an alternative to conventional, sectoral land-use planning, policy, governance and management. Sayer *et al.* (2013) have provided 10 principles for a landscape approach for reconciling agriculture, conservation and other competing land uses. They include: (i) continual learning and adaptive management; (ii) common concern entry point; (iii) multiple scales; (iv) multifunctionality; (v) multiple stakeholders; (vi) negotiated and transparent change logic; (vii) clarification of rights and responsibilities; (viii) participatory and user-friendly monitoring; (ix) resilience; and (x) strengthened stakeholder capacity (Sayer *et al.*, 2013).

Integrated landscape approaches may be effective for land resource management and governance for a number of reasons. They can correct the inability of sectoral approaches to: sufficiently address the interests of other sectors (such as nature protection versus livelihood needs of the poor); consider spatial spill-over effects of policies and decisions (i.e., decisions of a land use in one area is linked to environmental pollution, biodiversity loss, water shortage, erosion elsewhere within the landscape – downstream of a watershed, for example); or to better understand the linkages between humans and their surroundings (Arts *et al.*, 2017). For example, based on their analysis of the main environmental problems in mining areas, Lei and others (2016) recommend the utilization of a landscape strategy for planning and evaluating the ecological restoration and sustainable development of mining areas.

Role of the private sector

Businesses dependent on landscape resources have a central role to play in sustainable sourcing and collaborative actions to address water scarcity, biodiversity decline, deforestation and climate change (Goldstein *et al.*, 2012; Kissinger *et al.*, 2013; Natural Capital Declaration, 2015). There are notable examples of landscape-level restoration initiatives promoted by the private sector (WBCSD, 2016), such as the Landscapes for People, Food and Nature Initiative (http://peoplefoodandnature.org), and Commonland (http://www.commonland.com/en). However, out of 428 documented multi-stakeholder landscape partnerships, only a quarter involved private companies (Scherr *et al.*, 2017).

Nonetheless, experience indicates that initiatives for landscape restoration, sustainable farming, watershed management and natural capital accounting offer entry points for mutually beneficial cooperation, creating value, reducing risk and strengthening local relationships (Scherr *et al.*, 2017). Furthermore, natural capital accounting methods have facilitated multi-partner, private-public funding mechanisms for landscape initiatives (Shames *et al.*, 2014). For example, European supermarket chains, international development agencies and local non-government organizations came together to invest in enhancing natural capital through support for small farmers, soil and water conservation and wildlife protection in Kenya's Lake Naivasha Catchment (Shames *et al.*, 2014). Commonland brings together investors, companies, farmers and/or landholders for long-term, large-scale landscape restoration to create four types of returns from the land: inspiration, social capital, natural capital and financial capital. In a recent report of Community of Practice Financial Institutions and Natural Capital, formed by 15 financial organizations, van Leenders and Bor (2016) argue that although the project is in its early stages, financial institutions have been investing in natural capital to measure their impact and manage their risks while taking steps towards a green economy. Innovative financial instruments, such as green bonds and crowdfunding, can accelerate this transition (van Leenders & Bor, 2016).

Landscape governance

A key prerequisite for effective landscape governance – in view of halting or reversing land degradation – is the clarification of the spatial extent (territory) of the landscape to be conserved or restored and stakeholders involved (see Box 6.12). Several authors show that there has been a shift in considering the "territory" from a restricted involvement of only the actors who are technically supposed to conserve and/or restore the site, to a larger and more complex mosaic territory involving all the stakeholders concerned with the restoration site (Couix & Gonzalo-Turpin, 2015; Flores-Díaz *et al.*, 2014; Hobbs *et al.*, 2011; Petursdottir *et al.*, 2013; van Oosten *et al.*, 2014). This latter approach involves an appreciation of how people understand and value the place they live in (Flores-Díaz *et al.*, 2014), encourages citizens to reconnect to their place (van Oosten, 2013) and engages them in a process of "collective sense-making" (Couix & Gonzalo-Turpin, 2015).

Box 6.11 Restoration of Xingu watershed in the Amazon

The Xingu River is one of the Amazon's main tributaries. Its basin, in west-central Brazil, has 51 million hectares and is home to one of the largest conservation areas, the Xingu Indigenous Park, comprising of 24 indigenous groups (Schwartzman *et al.*, 2013). While the river channel is well protected within the Park, high deforestation rates have taken place in recent decades in the Xingu headwaters just outside the Park boundaries – mostly driven by cattle ranching and more recently by soybean production (Schwartzman *et al.*, 2013). Concerned about the degradation of water resources and the threat to the traditional ways of life within the Xingu basin, civil society organizations, indigenous organizations, state and municipal governments and farmers initiated the "Y Ikatu Xingu" campaign (YIX– "Save the Good Water of Xingu," in the Kamaiura language) (Schwartzman *et al.*, 2013).

The objectives of this forest restoration campaign included: conservation of water, fruit and wood production; carbon sequestration; and compliance with Brazilian environmental legislation (Durigan *et al.*, 2013). Forest restoration strategies were flexible and considered farmers' demands, motivations and farm facilities, as well as manpower, infrastructure and inputs. For forest restoration, direct seeding was deemed the appropriate method for tree establishment, and involved a mixture of green manure and seeds of forest species of different successional classes, applied and/or sown with the same tractors and implements used for crop and pasture cultivation (Campos-Filho *et al.*, 2013). This method of restoration was attractive to farmers, due to its low cost and familiarity of farmers and employees with the planting techniques and equipment. Also, since direct seeding requires large volumes of seeds (ca 400,000/ha), this approach stimulated the foundation of the Xingu Seed Network, formed by 420 indigenous and peasants collectors (Urzedo *et al.*, 2016). The Network produces 225 tree species and since 2007 has commercialized 137 tons of native seeds (www.sementesdoxingu.org.br). Five seed houses throughout the territory store seed lots and redistribute seeds to clients of the Y Ikatu Xingu restoration projects. Until now, the Y Ikatu Xingu Campaign has restored 900 ha using direct seeding, 300 ha by planting seedlings, and 1,500 ha by passive restoration (natural regeneration). The Y Ikatu Xingu Campaign is an example of a practical approach to large-scale restoration through law enforcement, shared governance and technological arrangements – ultimately leading to reductions in restoration costs, income generation and social mobilization.

Box 6.12 Landscape restoration and governance

Referring to landscape restoration, van Oosten *et al.* (2014) distinguish three modes of governance that steer decision-making:

- *Landscape governance as a management tool* – with a rather traditional hierarchical system of decision-making based on a central locus of authority, professional knowledge and binding regulation. Responsibilities can be shared among stakeholders, who can be considered co-managers of the system (generally in a well-defined system).

- *Landscape governance as a multi-stakeholder process* – in which attention is paid to new institutional interactions with increasing importance to private actors and soft law approaches, as well as local practices. It is most relevant in complex mosaic landscapes with delicate and politically-oriented decision-making. For example, between the forest and agricultural sector as it can enable better negotiation and conflict mediation.

- *Landscape governance as the creation of an institutional space* – in which actors from different sectors and scales create a new institutional space by creatively combining traditional and locally-embedded institutions, crafting hybrid institutions adapted to the specific socio-ecological characteristics. Such modes are most adapted to landscapes that stretch across administrative boundaries, scales and political entities.

6.4.4 Responses based on research and technology development

Global challenges associated with chronic land degradation – due to increasing populations, lack of fiscal or human resources, or inappropriate management decisions – have attracted numerous researchers from an array of disciplines to study the numerous underlying social, environmental and economic drivers and consequences (Bai *et al.*, 2008; Bojö, 1996; Conacher & Sala, 1998; Taddese, 2001). Most have concluded that appropriate land degradation responses can be developed and could be successful if research, improved local practices and appropriate institutional development activities become more widespread.

At a global level, UN organizations (e.g., UNCCD, UNEP, FAO), other multilateral agencies (e.g., WB, IFAD, WOCAT), research institutions (e.g., universities, and research centres) and government departments have all pursued research on how to avoid land degradation, restore degraded lands and develop human capital. These activities have resulted in numerous peer-reviewed and “grey” research reports and literature – providing excellent sources of information or knowledge on how to avoid and reduce further land degradation. Anthropogenic assets, including technology and infrastructure, are available for guiding improved land resource management (UNCCD, 2014). There has been significant progress towards the development of a conceptual framework for monitoring the progress of the UNCCD in addressing land degradation. For example, UNCCD decision 22/COP.11 has established a monitoring and evaluation approach consisting of: (i) progress indicators; (ii) a conceptual framework that allows the integration of indicators; and (iii) mechanisms for data sourcing and management at the national and/or local level (Low, 2013). Following this, India has developed a "desertification and land degradation atlas" by monitoring land use, processes of land degradation and severity levels between 2003-05 and 2011-13 (Space Applications Centre, 2016).

The spatial distribution of human capital (information, knowledge and skills) and technology have been influenced by socio-economic and technological factors – often leading to an uneven distribution among stakeholders (governments, communities and households). As a result, access to research knowledge and technology for sustainable land management or soil and water conservation and their adoption by land managers has been inconsistent. Therefore, in addition to research focused on soil degradation per se, the adaptive capacity of stakeholders also needs to be explored to determine what additional research and technology transfer investments are needed (UNEP, 2014). A recent assessment report on "unlocking the sustainable potential of land resources" concluded that improved land-use information systems and land-use planning and management are required to minimize the expansion of built-up land on fertile soils, and to invest in the restoration of degraded land (UNEP, 2016). This again points to integrated systems approaches, since efficient land management and major technological innovations (in agriculture) have potential to avoid a shortage of productive land while restoring degraded land (Lambin & Meyfroidt, 2011).

Advancements in technology and greater access to information are significantly increasing efforts to respond to land degradation problems more effectively. With appropriate data sources, new techniques based on land capability assessments can be used to monitor the extent and effects of both climate change and land degradation. Enhanced remote-sensing techniques have also made it possible to monitor the extent to which response options reduce or reverse degradation effects. Remote sensing has been used to monitor the provision of many ecosystem services including: provisioning, regulating, supporting and cultural services. However, determining specific degradation causes generally requires more detailed, field-level biophysical and socio-economic assessments, because of the wide range of factors that can cause any given change (Reed & Stringer, 2015). Furthermore, although several biophysical indicators can be monitored cost-effectively via remote sensing at broad spatial scales, field-based measurements are necessary to accurately interpret the data and establish cause and effect relationships (Reed & Stringer, 2015).

The combination of research, technology development and information transfer – initiated in the 1960s through the Green Revolution – has significantly contributed to increased production in food, feed and fibre for an ever-increasing global population (Khush, 1999). However, even though the revolution successfully enhanced productivity and income from farm-based communities, it unintentionally encouraged ecological destruction through unsustainable production practices – ultimately resulting in negative effects on the farm economy (Shiva, 1991). Therefore, to address sustainability issues while increasing per capita food production, combinations of technology with indigenous, traditional knowledge are needed (Conway & Barbier, 2013). One such example is the sloping agricultural land technology programme which has been very effective and popular in mountainous areas, such as the Loess Plateau of China and denuded uplands in Philippines, by conserving conserve soil and enhancing farm incomes (Sureshwaran *et al.*, 1996; Tacio, 1993; World Bank, 2007). Capacity-building of all stakeholders – from farmers to decision makers – is recognized as an effective means to combat land degradation and to achieve land degradation neutrality targets. This includes: the enhancement of scientific capacities to address key knowledge gaps; awareness-raising among decision makers and the general public; technology and knowledge transfer; and training. Perhaps the most significant need for capacity-building is in land resource management to deal with the complex issues of building efficient land information systems and sustainable institutional infrastructures, especially in developing countries and countries in transition (Enemark & Ahene, 2003). Given its pivotal role, several international organizations (such as FAO) and countless non-governmental organizations support capacity-building to combat land degradation worldwide. Among initiatives to support capacity-building to achieve land degradation neutrality, the Land Degradation Neutrality Target Setting Programme – conducted by the Global Mechanism of the UNCCD – currently supports 110 countries to set voluntary national targets (Orr et al, 2017) (see Chapter 8, Sections 8.2.1.1 and 8.4.3).

6.4.5 Responses based on institutional reforms

Land conservation and restoration policies have been implemented in a number of countries for several decades, leading to a growing body of assessments and comparative studies at different scales. Although many programmes derive from common international and national frameworks, several authors observe that similar legislation and policies can have very different outcomes depending on the existing local institutional arrangements (Hayes & Persha, 2010; He, 2014; Prager *et al.*, 2012; van Oosten *et al.*, 2014).

In recent years, the evolution of conservation or restoration policies beyond the traditional top-down state policies has led to a range of governance regimes and new institutional arrangements, with a transfer of responsibilities towards local governments and non-state actors (Agrawal *et al.*, 2008; Hayes & Persha, 2010). This decentralization can be more or less successful depending on the power transfer, accountability mechanisms and local participation involved (Ribot & Larson, 2005). Although effective stakeholder involvement is often cited as one of the main factors of success (France, 2016; Light, 2000), in practice, it is far from being systematic, often because of a lack of definition of who are the important stakeholders (Couix & Gonzalo-Turpin, 2015), and because formal institutions usually lack the flexibility and openness to cope with the more dynamic and innovative informal organizations. Furthermore, the history of community-based natural resource management suggests that simply understanding the value of local participation is complementary to reforming existing institutions or establishing new institution (e.g., community-based organizations, for example).

Governments, multilateral development banks, private sectors, and donor agencies have advanced various institutional models to engage local communities and others in reforestation, including partnerships with commercial plantations (Barr & Sayer, 2012). Such initiatives are supposed to generate benefits for rural communities, including employment, access to credit, low cost inputs (seeds, fertilizers and so on) and ready markets (Lamb, 2010). However, as many authors warn, diverging interests and power relations embedded in conservation or restoration are often overlooked in such arrangements (Baker *et al.*, 2014; Barr & Sayer, 2012; Bliss & Fischer, 2011; Hayes & Persha, 2010): Who really benefits from the resources? Who is actually able to make the rules? Who monitors and enforces the rules? The equitable distribution of burdens and benefits is probably the main challenge and the greatest obstacle to overcome in inter-institutional reform and decision-making processes.

Not all institutional arrangements for reforestation or restoration programmes are effective in generating greater benefits for local people. For example, reforestation programmes in the Asia Pacific, which are led by administration or corporate interests, have led to displacement of local communities, channelling international funding towards state elites, facilitated corruption or perverse incentives to convert secondary forests in plantations (Barr & Sayer, 2012). Local communities generally have little leverage in negotiating agreements with plantation companies or ensuring accountability (Barr & Sayer, 2012). Inequitable land-rental contracts and out-grower agreements, sometimes even forced onto the farmers, can have very detrimental effects on smallholders. People's involvement can be limited to handing over common lands and wage employment (Saxena, 1997) shaped by local power relations (Barr & Sayer, 2012).

One of the key aspects in institutional reform is guaranteeing tenure rights to local populations (Barr & Sayer 2012; Mansourian & Vallauri 2014; Williams & van Triest 2009). Although many programmes are put forward as community management, they are often limited by tenure uncertainty and non-participatory decision processes. For example, national forestry laws often recognize traditional tenure systems, but those rights are often subordinate to state claims over forest resources and few institutional mechanisms exist to resolve competing claims between state and customary systems (Vandergeest & Peluso, 2006). Conversely, in the Sloping Land Conversion Program in China, the institutional reform that secured long-term property rights over the restored land was found most effective compared to other incentives offered to engage locals in restoration (Grosjean & Kontoleon, 2009). However, formalization of private tenure can exclude the more marginalized populations, such as women or the "poorest of the poor" (Barr & Sayer, 2012). This points to the necessity of developing an approach to resolve competing claims between local communities managing land under customary tenure systems and state agencies relying on national codes, perhaps by at least committing to the principles of free, prior and informed consent of affected communities (Barr & Sayer, 2012).

Several studies show that innovative types of collaborative network governance are emerging that bring together natural resource users, NGOs, concerned citizens, private corporations and various branches of government. Such arrangement can accommodate, numerous initiatives within a large-scale framework (Adams *et al.*, 2016; France, 2016; Petursdottir *et al.*, 2013; Pinto *et al.*, 2014). These forums or advisory committees ensure the representation of the different interests at stake. However, as underlined by Baker *et al.* (2014), there are still limited studies in which these interests are articulated and negotiated. Too many programmes are still focused on end-products and not enough on the developmental process and social learning that such networks enable, to build true adaptive capacity (Pahl-Wostl, 2006; Zedler *et al.*, 2012).

Box 6.13 Community Forest User Group: Reformed institution to manage forests in the hills of Nepal

The practice of forest management in the hills of Nepal shows how institutional reform help to address deforestation and restore degraded forest lands. Until 1957, before forests were nationalized, forests in the hills of Nepal were protected and managed by nearby villagers for generations based on customary practices. Even though the forest nationalization in 1957 had good intention to cease large tracts of forests hold by ruling class, it sent a wrong signal to ordinary villagers in the hills resulting in policy failure and a trigger for rampant deforestation. During the 1960s, the Nepalese government adopted a "command and control" approach to halt deforestation, but failed due to inadequate institutional capacity – leading to continued deforestation and degradation of hill slopes with increased problems of landslides and soil erosion (Pandit & Bevilacqua, 2011). This phenomenon of forest degradation and soil erosion is famously described in the form of "Himalayan Degradation Hypothesis" by Eckholm (1976).

To address the deforestation, forest degradation and soil erosion problems in the hills, by 1978 the Nepalese government reformed forest policy and initiated a new institution to manage hill forests based on a bottom-up and participatory approach, now commonly referred to as "community forest user group". This approach transferred forest-use rights to "forest user groups" and reconnected them with their nearby forests – named as community forests – with a sense of ownership (HMG/N, 1993), allowing "forest user groups" to develop rules (i.e., constitution of community forest user group) to manage the forest based on a collective forest management plan and share the benefits amongst themselves (HMG/ADB/FINNIDA, 1988). With the inception of a new institution, and reformed forest policy in 1978, degraded hills were extensively planted with the mobilization of local users. Due to its success in the hills, community forestry became a nationwide programme since 1993. By 2015, a total of 1,798,733 ha of forests (approximately 30.85% of total forest area in Nepal) have been managed by 18,960 "community forest user groups", benefitting nearly 2,392,755 households (DoF, 2015). As shown in Figure 6.16, community forestry programmes have transformed many degraded hills into productive forests and have either halted or at least reduced deforestation, and associated land degradation. Forest statistics of Nepal indicate that forest cover decreased from about 38% of country's land mass (147,181 Km^2) in 1978/79 to about 37.4% in 1985/86, which then increased to about 38.3% in 1995, and 44.74% (covering 59,624.38 Km^2, of which 40.36% forests and 4.38% shrub lands) in 2015 (DFRS, 2015). Most of this gain in forest cover has been in the hills where community forestry programme has been in operation since 1978; initially as Panchayat, or Panchayat Protected forest, and later as community forestry.

Figure 6 16 Restored degraded hill forest in Nepal (right panel) through community forestry programme. The degraded site is showcased on the left panel. Site: Dandapakhar, Sindhupalchok district. Photo: Courtesy of Fritz Berger on behalf of Nepal Swiss Community Forestry Project (2011).

6.5 Knowledge gaps and research needs

There currently exists a deep and broad base of knowledge and experience to support sustainable land management and soil and water conservation, biodiversity conservation and restoration practices, as well as a rapidly developing understanding of the importance of policies, institutions and governance responses in providing an enabling environment for effective responses to land degradation and its drivers. There is enormous potential for applying this existing knowledge more widely, given adequate support by decision makers, land managers and the general public. Nonetheless, there remains a number of key areas where significantly enhanced effort - by the research and development communities, farmers and other land managers, planners and decision makers - is required to halt and reverse current land degradation trends.

Further work is needed to:

- Develop analytical methodologies and tools to better understand and quantify the full range of values (nature's contributions to people) people derive from land (and ecosystems), the short- medium- and long-range costs associated with biodiversity loss and degradation, as well as costs and benefits associated with avoiding, mitigating and reversing land degradation;

- Provide knowledge, tools and skills (by the scientific community) on land condition monitoring for land managers and planners - both conventional and ILK-based approaches, including citizen science;
- Bridge, among and within countries, current gaps in knowledge and skills, capacity and resources needed by landowners, communities and governmental land management agencies to effectively halt land degradation and restore degraded lands - through, for example, the development of easily accessible geospatial land information systems, and enhanced North-South, South-South and triangular knowledge-sharing, research and development activities;
- Better understand the conditions under which indigenous and local knowledge and practices, for sustainable land management and restoration, can be used more extensively, and how such knowledge and practice can better inform the development of strategies and specific technologies for sustainably managing croplands, rangelands, forests, wetlands and urban lands;
- Develop policies that encourage sustainable land use at the landscape level, in a coordinated and integrated fashion among development sectors; and
- Better understand which policy instruments, institutional and governance systems are most effective for avoiding, reducing and reversing land degradation under local environmental, social, cultural and economic conditions. Addressing land degradation issues at a local level, by aligning policies and instruments that could generate benefits on multiple scales, is fundamentally important for the success of restoration responses in conserving biodiversity, providing ecosystem services and supporting livelihoods.

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Chapter 7

Scenarios of land degradation and restoration

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Executive Summary

No scenarios have been found that collectively address and meet global goals *(well established*). The world is increasingly interconnected and needs cross-sectoral approaches to foster sustainable use of resources (*well established*). In a "business-as-usual" world, sectoral approaches to meeting individual global goals for food, water and energy security, while mitigating climate change and halting biodiversity loss may be successful, but will fail to meet these goals collectively because of the considerable trade-offs that currently exist between sectoral solutions (*established but incomplete*). Alternative scenarios with partially integrated modelling approaches provide a more complete depiction of potential outcomes (*well established*). There is a need to fully adopt integrated approaches to find sustainable solutions to the land degradation challenge, and take into account interactions across the supply of commodities, the environment as well as the increasing demand from a growing population and rising consumption levels {7.2, 7.3}.

Some regions will be disproportionately affected by land degradation *(well established*). In the coming decades, the occurrence of incidental and structural deficits in food, water and energy are likely to grow with local variations in type and extent. There is serious risk that these may lead to unmanageable societal and environmental problems in regions that combine features such as low productivity soils that are vulnerable to degradation (*well established*) {7.2.1}, climate change that amplifies extreme conditions (*established but incomplete*) {7.2.4, 7.2.5}, low reserves of productive land (*established but incomplete*) {7.3}, high population density or population growth (*well established*), high rates of poverty (*well established*) {7.3}, weak institutions and political systems, and absence of economic coping mechanisms (*well established*) {7.3}. There are signals that serious deficits of food, water or a liveable environment may lead to social and economic instability, conflict and mass migration, which may destabilise adjacent regions {7.2.3, 7.3}. Many of these features disproportionately impact arid, semi-arid and dry sub-humid areas.

Incremental changes do not suffice *(established but incomplete*). To address these concerns, major and transformative changes are required in three domains: consumption, demographic growth and technology transfer (*established but incomplete*) {7.2, 7.3}. Changes in each domain influence the extent and efficiency of land use proportionally (*well established*) {7.3}. Simultaneous action in all domains will have the highest impact on maintaining productive landscapes while mitigating climate change and halting biodiversity loss (*well established*) {7.3}. Economic scarcity can exaggerate biophysical scarcity (*well established*). Therefore, alleviation of poverty and building effective institutions for policy implementation require attention as well.

More efficient production systems are needed to halt land conversion *(unresolved*). Changes in the systems of food, timber and fibre/bioenergy production can either relieve or exacerbate pressures on land and related water resources and reduce degradation. Sustainable intensification of agricultural, livestock systems and forestry, where feasible, can prevent further loss in biodiversity, land-based carbon storage, and water holding capacity from land conversion {7.2, 7.3}. Sustainable intensification of agriculture, livestock systems and forestry is possible, particularly in regions where current yields per hectare are low and physical, institutional and technological constrains can be overcome. Achieving sustainable intensification requires multiple actions: technical assistance and appropriate technologies; reward systems for sustainable farming practices, especially in resource poor regions; access to markets and capital; institutional reform, and environmental conservation (*well established*) {7.2.1, 7.2.3, 7.2.6, 7.3}.

Responsible consumption is essential to halting land degradation *(well established*). Globally, future consumption patterns along current trajectories are likely to have growing negative impacts on land and thus biodiversity and ecosystem services {7.2, 7.3}. Adjusting future consumption and reducing waste would significantly reduce these impacts. Increased demand for food, fibre and bioenergy due to increasing population and consumption levels is likely to foster an expansion of agriculture, livestock systems and forestry into remaining natural land. Reduced meat consumption and the resulting switch from livestock to plant-based food systems can help to reduce or halt expansion and environmental burdens {7.3}. Actions to increase educational attainment, provide health care services and social security and to manage the distribution of urbanization and human settlement would reduce the loss of biodiversity and ecosystem services, and the risk of social instability - especially in areas with high growth levels and low availability of arable land per capita (*established but incomplete*) {7.2, 7.3}.

Bioenergy and bioenergy with carbon capture and storage (BECCS) have been increasingly framed as a key element of climate change mitigation scenarios that require negative emissions to meet radiative forcing targets *(well established*). Climate change mitigation scenarios that limit the global average temperature to 2°C above pre-industrial levels estimate that the large-scale production of fibre and timber for energy purposes is expected to grow to around 1.5 million km^2 in 2050 and 4 to 6 million km^2 in 2100 under 2°C emissions pathways (*established but incomplete*). Vastly expanded timber and energy biomass production for climate change mitigation and biofuels production purposes will exacerbate biodiversity loss, water scarcities and compete with food for land - potentially resulting in indirect land-use change and adverse impacts on food security (*established but incomplete*) {7.2.2, 7.2.5, 7.2.6}.

Increasing complexity requires evolving institutions (*established but incomplete*). The issue of land degradation is highly complex in its drivers and impacts and is evolving with an accelerating speed (also called “raplexity”), requiring sound insights and complex societal solutions, supported by new institutions and political arrangements. Land degradation concerns a kind of “wicked problem” where there is often incompatibility between the complexity of the challenges and the demand for simple and politically attractive solutions. Particularly in vulnerable regions, problems are likely to grow faster than institutions can cope, resulting in an “ingenuity” gap. Effective policies to address land degradation can only be acquired by constant interaction, discussion and negotiation with stakeholders including indigenous and local peoples, and addressing underlying equity, equality and gender issues.

Integrated models and scenarios are indispensable tools for unravelling complexity (*well established*). Integrated and spatially-explicit models are tools to better understand the complexity of land degradation processes. They enable a better understanding of the complex trade-offs, interdependencies and synergies between biodiversity and ecosystem services. Currently, no integrated future scenarios exist that simultaneously consider changes in land and soil properties, water, food, timber, fibre, bioenergy, climate change and biodiversity. Consequently, many mutual synergies, interdependencies and trade-offs are not considered, are uncertain or are simply unknown. Addressing this gap is a prerequisite for building integrated models that take into consideration the influence of a wider range of drivers, both biophysical and societal (*well established*) {7.2, 7.3}.

7.1 Introduction

This chapter assesses scenarios of future land degradation and restoration in terms of change in: (i) soil properties; (ii) biodiversity; and (iii) ecosystem services as a result of human activities up to 2050. The ecosystem services considered are provisioning services, such as production of food, bioenergy, fibre and timber as well as regulating services, such as water and climate regulation through carbon storage and sequestration. The effects of land degradation and restoration on cultural services are less explored in scenarios. This chapter outlines the different types and roles of scenarios, assesses global and regional scenario outcomes and recommends future scenario developments. Impacts are described at the global and regional scale.

Section 7.1 elaborates on the function and types of scenarios, criteria for selecting appropriate scenarios, their use in the policymaking process, and a brief stocktaking of regional scenarios. Section 7.2 describes the impacts on the above-mentioned individual components of land degradation, while Section 7.3 shows interdependencies according to integrated scenarios.

7.1.1 Why scenarios?

Scenarios are representations of possible futures for one or more components of a system, particularly, in this assessment, for drivers of change in nature and nature's contributions, including alternative policy or management options (IPBES, 2016). They can characterise and reduce uncertainties, link different fields and domains, and deal with complexity such as synergies and trade-offs. Moreover, scenarios outline possible future developments - which is important given the relationship between decisions that are made in the short-term and their long-term consequences (as a result of inertia in the natural and socio-economic systems). Given an increasingly connected, complex and rapidly changing world - with increasing stakes, declining resilience and increasing irreversibilities - scenarios become a key instrument to cope with "raplexity", rapid and complex change. Scenario building inspires people to think openly, to exchange views and knowledge, and to jointly explore the threats of land degradation and opportunities of restoration (Reed *et al.*, 2013).

Practically, scenarios can help to identify the effectiveness and efficiency of individual measures or measure combinations, and assess the cost of policy inaction such as in "business as usual" scenarios. In concert with models, they may reduce uncertainty and reveal road maps to achieve targets. Further, they are vital to project alternative socio-economic development pathways and create new opportunities for restoring ecosystems (IPBES, 2016).

7.1.2 Key concepts

7.1.2.1 Types and role of scenarios

While the semantics of scenario typologies are disputed, there is consensus regarding the underlying fundamental categories (e.g., IPBES, 2016). In this assessment, two categories are particularly relevant: (i) exploratory; and (ii) intervention scenarios – the latter encompassing target-seeking scenarios. Exploratory scenarios are largely used in the initial stages of policymaking to understand the extent of a problem and map its various potential futures by systematically varying key drivers. If policymakers determine that action is desirable, exploratory scenarios are frequently followed by intervention scenarios that explore the impacts of alternative targets and alternative pathways to achieving a target. In the last phase of the policy cycle these scenarios can be followed by a retrospective policy evaluation examining the success or failure of policy interventions to achieve the targets. Examples can be found in

Duncan & Dorrough (2009); Egoh *et al.*, (2014); Salvati & Zitti (2009); Schelhas *et al.*, (2012); Schwilch *et al.*, (2009); Suding *et al.*, (2004); Turner *et al.*, (2015); Zucca *et al.*, (2015). All these scenario types are applicable to land degradation and restoration issues. Given the relative initial stage of applying scenario analysis to land degradation and restoration, the vast majority of all land degradation and restoration scenario analyses are exploratory in nature. Some examples include Ceccarelli *et al.*, (2014); Rogier de Jong *et al.*, (2011); Van der Esch *et al.*, (2017); Märker *et al.*, (2008); Shrestha & Roy (2009)

7.1.2.2 Scenario properties and the effect of choice of metrics, baselines and scales

Scenarios and models, applied to scenarios, are characterized by many features: they can be dynamic or static over time; they can take interactions between variables into account; and they can have a sectoral (one issue) or integrated (multiple issues and/or domain) character. Moreover, scenarios can be qualitative or quantitative, and they can include both exogenous and endogenous drivers. Exogenous drivers are assumed to be unchangeable by policies ("a given development"). Endogenous drivers can be changed by policies, depending on the scale under consideration (MA, 2005a). Often, scenarios that include measures are compared with a so-called "baseline scenario" (also called "business as usual", "no new policies" or "do nothing" scenarios) in order to highlight the specific impact of one or a package of measures in the context of an ever-changing world. Baseline scenarios should not be confused with indicator baselines. The baseline of an indicator is a reference value to make the current or future state meaningful, for instance the current population numbers of orangutans compared to those in the natural state or the minimum number of a viable population. The baseline of an indicator should not be confused with a target. Targets are the result of balancing socioeconomic and ecological interests and objectives, and can hold a value between 0 and the baseline (Kotiaho *et al.*, 2016), or exceeding the baseline in case degradation is part of the baseline (such as in case a reference year has been used) or when the target exceeds the natural baseline, such as a food productivity target in intensive agriculture (for further discussion on baselines and targets, see Chapter 2, Section 2.2.1.1). In conjunction with scenarios, models are used to quantify future impacts of policies and/or uncertainties in the socioeconomic and biophysical field, and to explore potential alternative futures.

The choice of indicators or metrics highly determines the results. This also applies for how issues are framed, which aspects are looked at and which ignored, which temporal and spatial scales are considered, which baselines are used and which assessment principle has been applied (Basso *et al.*, 2012; Kairis *et al.*, 2013; Kirkby *et al.*, 2000; Kotiaho *et al.*, 2016; Stavi & Lal, 2014; UNEP, 2003) (for further discussion on baselines, see Chapter 2, Section 2.2.1.1 and Box 2.1). An assessment principle is the way a change in an indicator is valued. This arbitrary choice of the assessment principle is materialized in the baseline. The "natural state" as baseline has "naturalness" as assessment principle (a change towards the natural state is considered as positive and vice versa), while the "minimum viable population size" as baseline has "viability of a population" as assessment principle. These different assessment principles may lead to entirely different valuations of the same state. Applying a natural state as baseline shows human impact and a theoretical restoration potential, a critical level marks a point beyond which an impact becomes detrimental, and a policy target shows the gap between the current and the politically desired state. An analogy in economics is the consideration of unemployment rates as an absolute number of people, as relative to a previous year, or as relative to a policy target.

Which land transformations are perceived as improvements and which as degradation and why? How is the baseline determined? These questions are particularly salient due to the subjectivity inherent to land degradation and restoration (see also next section). A farmer's perception of whether a highly intensive farmland is degraded or not may contrast with that of a water manager, conservationist or a tourist (see also Chapter 2 for further discussion on perceptions). Indigenous peoples' assessments of 'degradation' or soil quality does not always coincide with conventional methods (Gray & Morant, 2003; Meyer, 1996). Changes in soil, biodiversity, land cover and ecosystem functions are inherent to the transformation of landscapes favouring one or a few functions, such as food and fibre production, at the cost - often unintentionally - of others, such as biodiversity, water and climate regulation. In essence, land degradation is about the assessment of these trade-offs.

Temporal and spatial scale can also highly influence the perception of outcomes. For example, a short-term trend can be negative but in the long term it may bend towards the positive. This scale dependency may also apply when looking at geographic extent (the 'scale paradox'). For example, the intensification of food production can be assessed as detrimental for farmland biodiversity, but can be assessed as positive when taking into account the natural area that is secured from conversion as a consequence (“external effects”).

7.1.3 Why are land degradation and restoration scenarios scarce?

Land degradation is an extremely elusive problem. Despite considerable efforts, the scientific community has not been able to provide a detailed global assessment addressing what kind, where and how much land has been degraded. Global assessments such as the Global Assessment of Soil Degradation (GLASOD) and the Global Assessment of Land Degradation and Improvement (GLADA) have not provided a comprehensive, quantitative and unambiguous picture of the current and future state and distribution (of the various components) of land degradation (Bai *et al.*, 2008a; Oldeman *et al.*, 1991). However, recent steps have been made (Van der Esch *et al.*, 2017) in the form of a contribution to the UNCCD's first edition of the Global Land Outlook (UNCCD, 2017).

One of the main reasons for the lack of conclusive quantitative data on land degradation is because land degradation is a global and multidimensional problem that occurs at multiple spatial scales and involves multiple factors and actors. While in the climate community the one dimensional cold-to-hot trajectory has a very intuitive meaning and atmospheric concentrations of carbon dioxide are an obvious target, measuring land degradation trajectories is more complex. Further, as indicated above, the definition and parameterization of land degradation is open to far greater scrutiny and to a wider variety of perceptions among various stakeholders, who may have conflicting interests at stake.

This complexity has been fuelling discussions on the definition of land degradation for decades. The lack of a clear definition has hindered the development of clear, broadly accepted and consistent indicators, baselines, thresholds, monitoring, calculation procedures and models (Caspari *et al.*, 2014; Kotiaho *et al.*, 2016). The persistent deficit of monitoring data, the shortfalls in current land cover mapping technology, and poor data harmonization and integration precludes the scientific community from providing a clear baseline from which we can measure change, in particular for soil characteristics. Given the multidimensional nature and subjective character of land degradation, this chapter focuses on a more neutral approach, including - where data allows - the changes in, and trade-offs between, biodiversity, soil properties and ecosystem services induced by human interventions. This provides a flexible approach that will appeal to a range of stakeholders and allows for a comparison over time and between regions, as well as aggregation from local to global scales. Consequently, the use of the word land degradation has been avoided. Following this logic, while land-use change is not considered synonymous with land degradation, various scenarios that often comprise land-use change (e.g., cropland expansion, deforestation) next to other drivers are exhaustively explored.

7.1.4 Use of scenarios related to policy targets

Climate change scenarios and their impacts have been widely used to influence decision-making. Each scenario implicitly or explicitly contains information about development, equity and sustainability (IPCC, 2014b). Sustainable development pathways can make a major contribution to climate goals and the IPCC Fourth Assessment Report examines sustainable development scenarios in relation to climate change mitigation (IPCC, 2007). In 2012 the PBL Netherlands Environmental Assessment Agency developed three "policy-rich" sustainable development scenarios and a "business as usual" scenario to describe changes in climate, biodiversity, agriculture, forestry, energy sources, water regulation and consumption with and without strengthened environmental policies, intending to meet global goals (PBL, 2012).

The process of establishing the Sustainable Development Goals (SDGs) has benefitted from a number of scenario exercises. A report on Sustainable Development and Planetary Boundaries for the Post-2015 Development Agenda examined three contrasting development scenarios against global environmental constraints combined with population and economic growth. These scenarios included: boundaries defining a safe global level of depleting non-renewable fossil resources; boundaries defining a safe global level of using the living biosphere, including exploitation of ecosystems, protection of biodiversity and consuming renewable resources, such as land use; and boundaries providing a safe global level of Earth's capacity to absorb and dissipate human waste flows (Rockström *et al.*, 2013). Scenarios have been used to examine outcomes of the 2010 target of the Convention on Biological Diversity and of the Aichi Biodiversity Targets of the Strategic Plan for Biodiversity 2011-2020 (CBD, 2014). The CBD has analysed the relationship between biodiversity actions and broader challenges facing human societies by comparing "business as usual" with plausible scenarios for simultaneously meeting biodiversity, climate and poverty reduction objectives.

Joshi *et al.* (2015) examined common governance transitions (on security, capacity-building and inclusion) to compare a "dynamics as usual" governance forecast to alternative scenarios and showed that progress towards the Sustainable Development Goals will be accelerated by stronger governance and better development policies. Socioeconomic and demographic drivers of land degradation were recently included in newly developed scenarios: Shared Socioeconomic Pathways (SSPs). The SSPs (see Section 7.1.5.1) consist of five narrative storylines structured according to socioeconomic challenges for climate change mitigation and adaptation (O'Neill *et al.*, 2014).

7.1.5 Selection of scenarios for this assessment

It is not easy to define precise criteria for the selection of land degradation and restoration scenario literature for the aim of this assessment. If the criteria are overly broad, any state or change of land perceived as degraded by a stakeholder could be part of this analysis. If the criteria are overly narrow, only the scenarios dealing with a total loss of biodiversity and ecosystem services are considered.

The following criteria for the selection of global scenarios were applied:

1. Terrestrial systems only;
2. Large-scale natural areas with significant loss of the original biodiversity, soil properties and/or a selection of key ecosystem services (i.e., food, water, climate regulation and timber/fibre/bioenergy);
3. Large-scale cultivated areas with significant loss of its traditionally accompanying biodiversity, soil properties and/or the above-mentioned ecosystem services;
4. Scientifically sound, quantitative and qualitative reports;
5. A balance between sectoral and integrated scenarios, scenario types, and passive and active

restoration.

For the assessment of regional scenarios (from sub-continental to local), we draw upon approximately 250 studies that were systematically searched as local scenarios of land degradation and restoration, as well as the related literature assessing these scenarios. The formal review carried out with Scopus utilised the following search queries: land degradation, global; land restoration, strategies; land restoration, scenarios; land degradation, scenarios; land degradation and restoration. The full literature search was then reviewed and reduced to those studies that presented scenarios of land degradation and restoration.

Regional scenarios were primarily exploratory scenarios. The primary direct driver considered was land use change. Scenarios were typically integrated, and most often included land use and climate change with feedbacks. The state and impact variables which were most often included – and therefore summarized in this assessment – were changes in extent of ecosystems, changes in biodiversity status (e.g., IUCN threatened species listing), carbon stored in vegetation, water discharge, freshwater availability, salinity, agricultural extent (as a proxy for increased food production), economic returns to agriculture (as a proxy for changes in food production) and per capita income. Due to the place-specific nature of cultural ecosystem services, these variables differed considerably across studies and were not amenable to summarizing. Regional-scale scenarios were typically at the local or catchment level and operated on shorter temporal scales than global scenarios (e.g., up to 2030 in a local scenario compared to 2050-timeframe in national and global scenarios). Responses included intensification of current agricultural production to reduce future land clearing, changes in agricultural and livestock management practices and increased conservation activity such as protecting land at risk of future clearing (both private and public protection). Restoration of degraded land was less often explored as a response.

Table 7.1 Regional statistics of land degradation and restoration scenarios

Drivers	Region	Percentage of studies
Land degradation (27%)	Africa	1%
	Asia - Pacific	7%
	Europe	14%
	Americas	5%
Land use change (73%)	Africa	3%
	Asia - Pacific	19%
	Europe	36%
	Americas	15%

Europe, Asia and North America represented 90% of the regions in the scenarios and were primarily at the regional and catchment scale (80%), reflecting the need to model processes at local scales, such as nutrient cycling. In all regions, coverage across countries was sparse, with several countries typically dominating the literature: namely, China in Asia; Australia, Indonesia and Japan from the broader Asia Pacific region studies; and Canada and the United States of America from the Americas group. Europe had a greater coverage of countries, but several countries (France, Germany, Switzerland and Italy) still represented half of the studies. Scenarios for South America and Africa were limited, with only a few countries represented in the literature.

Consistent with global scenarios, regional scenarios suggest that future loss of ecosystem extent is concentrated in Central and South America, sub-Saharan Africa and Asia due to the relative large amount of land suitable for production purposes in those regions (Alcamo *et al.*, 2011; Jetz *et al.*, 2007; PBL, 2010;

van Vuuren *et al.*, 2006; Verburg *et al.*, 2008; Visconti *et al.*, 2011). Other factors of this concentration are relatively low cost of land, low labour cost, growing demands for food, and the globalization of trade (PBL, 2010; CBD, 2007).

The spatial gaps in regional scenarios in Africa and South America represent a mismatch in the expected concentration of future biodiversity loss versus existing scenarios to provide insights and guide policy responses. The sparseness of coverage across regions and the diversity of contexts covered point to the difficulty of devising general trends from regional-scale scenarios.

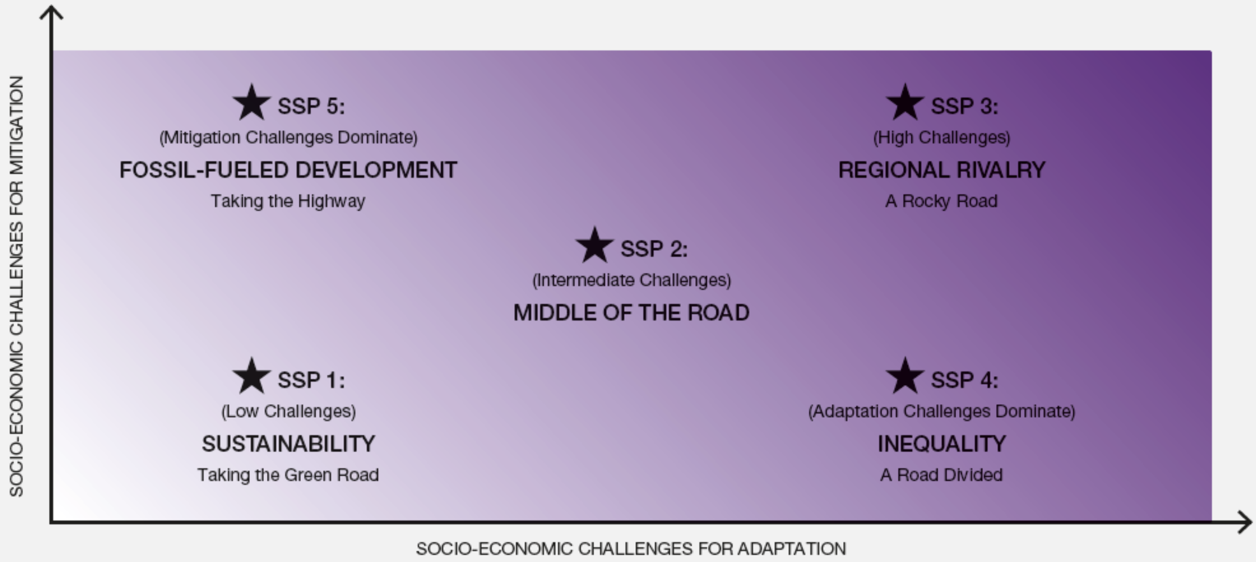
7.1.5.1 IPCC Special Report on Emissions Scenarios and Shared Socioeconomic Pathways

Global scenarios developed under the IPCC have the potential to become more relevant for land degradation and restoration scenarios in the near future. The Special Report on Emissions Scenarios (SRES) of the IPCC placed four scenarios of climate change on axes of societal preference (economic versus environmental) and governance (global versus regional) (Nakicenovic *et al.*, 2000). Although not addressing land degradation directly, SRES scenarios showed how a combination of climate change and socioeconomic drivers can increase degradation due to the stresses imposed by drought, floods, storms and rising temperatures on water resources and ecosystem services, such as pollination and bio-security due to invasive species and disease risk (IPCC, 2001).

More recently, in support of the IPCC's Fifth Assessment (IPCC, 2014b), the Shared Socioeconomic Pathways (SSPs) have been developed (Riahi *et al.*, 2017), comprising five scenarios structured in accordance with the socioeconomic challenges of climate change mitigation and adaptation (Figure 7.1). Scenario narratives are supported by key indicators and metrics, and describe trends in demographics (Box 7.1), human development, economy and lifestyle, policies and institutions, technology, environment and natural resources. The majority of existing scenario studies utilizing the SSPs have so far employed SSP1-SSP3. They were also used for the first edition of the Global Land Outlook (UNCCD, 2017).

Figure 7 1 SSP narrative matrix.

SSP1 Sustainability (Taking the Green Road) represents a world shifting toward sustainability, characterised by low population growth and economic development respecting environmental bounderies, reduced inequality, and lower consumption oriented toward low material growth and lower resource and energy intensity. **SSP2** Middle of the Road represents a business-as-usual world that does not deviate significantly from historical trends. Development and income growth proceeds unevenly. Environmental systems experience degradation. Global population growth is moderate and levels off in the second half of the century. **SSP3** Regional Rivalry (A Rocky Road) represents a fragmented world characterised by increased inequality and high population growth in developing countries, with high levels of environmental degradation. A resurgent nationalism, concerns about competitiveness and security, and regional conflicts push countries to increasingly focus on domestic or, at most, regional issues. Investments in education and technological development decline. Consumption is material-intensive. Economic development is slow. **SSP4** Inequality (A Road Divided) represents a world characterised by conflict and high levels of social and economic polarisation, with environmental protections established in high income regions and low income regions experiencing environmental degradation. **SSP5** Fossil-fueled Development (Taking the Highway) represents an integrated and technologically advanced world with low population growth, but the push for economic and social development is coupled with the exploitation of abundant fossil fuel resources and the adoption of resource and energy intensive lifestyles around the world. Source: O'Neill *et al.* (2014) and Riahi *et al.* (2017).

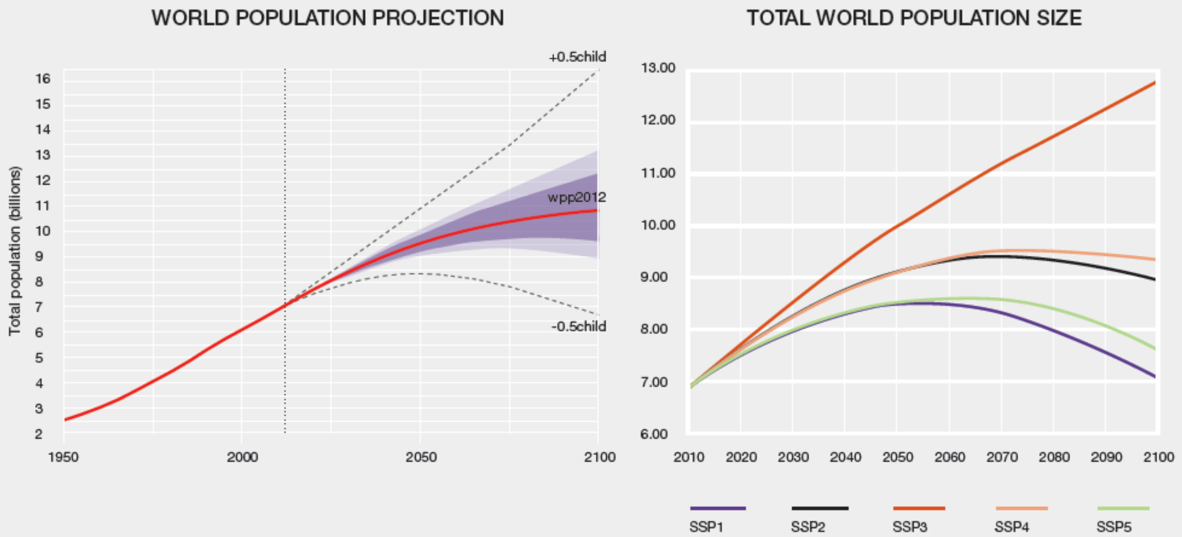


Box 7.1 Global demographic projections

The SSPs have been used to examine environmental and food security related indicators, with recent work indicating that policy scenarios exert a more significant impact on joint environmental and food security outcomes than the influence of population and economic growth scenarios stipulated within the SSPs (Obersteiner *et al.*, 2016). While population growth continues to be the primary driver of total consumption in the developing world, global consumption is currently dominated by the developed countries (Steffen *et al.*, 2015), with the environmental impacts of unsustainable consumption frequently displaced to the developing world (i.e., telecoupling) (Liu *et al.*, 2013). Recent studies have emphasized that reduced population growth is the most effective way to reduce carbon emissions and related impacts (Wynes & Nicholas, 2017). Thus, pathways that take into consideration alternative population trends (e.g., through investment in education and health services) may be considered within land degradation and restoration driver scenarios.

Figure 7 2 United Nations Population Division (left) and IIASA World Population Projections (right).

For more information regarding divergent methodologies, see Chapter 3 of IPBES deliverable 3c (Pichs-Madruga *et al.*, 2016). Source (left to right respectively): Gerland *et al.*, (2014); Samir & Lutz (2017).

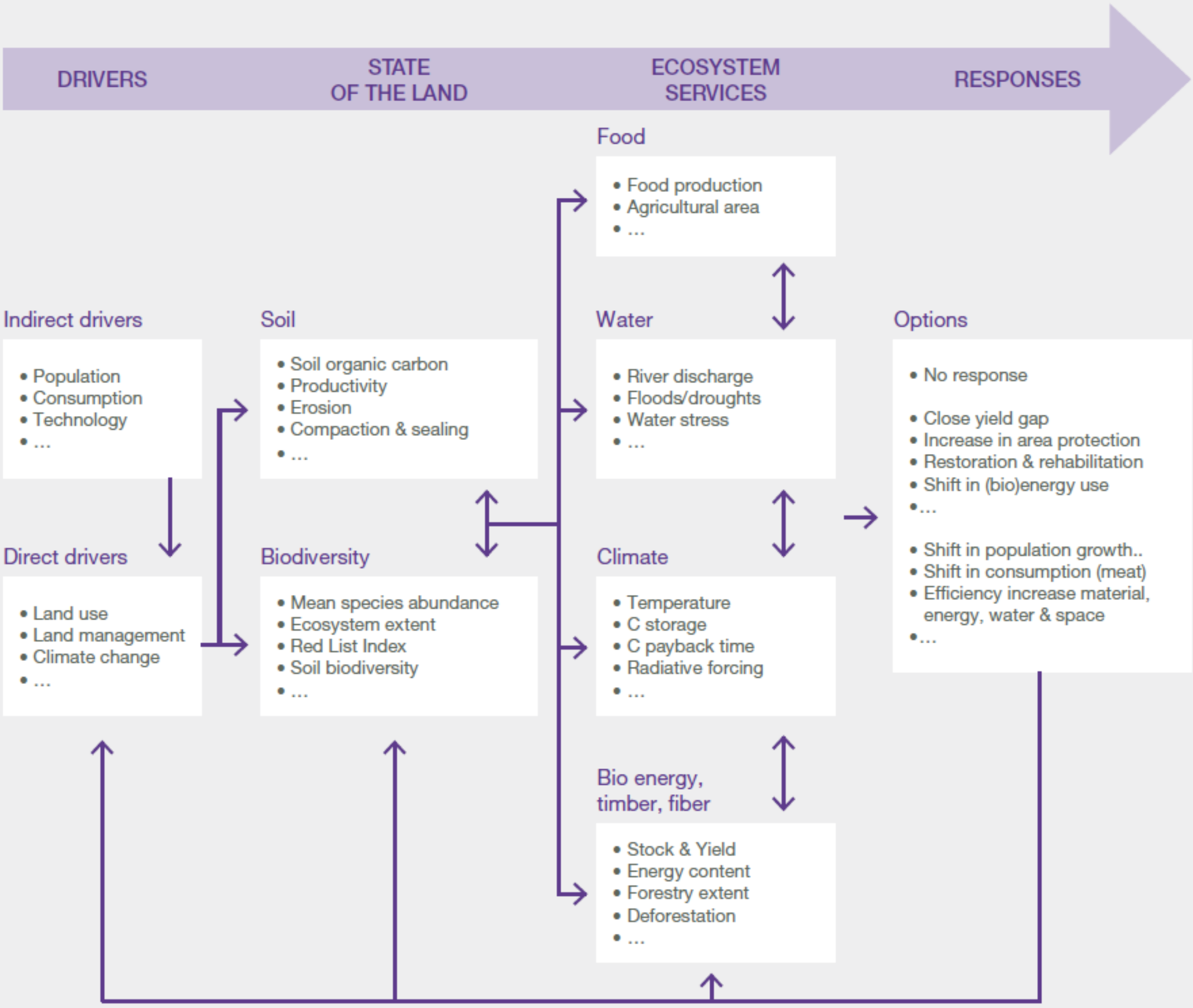


7.2 Scenario assessment by land degradation and restoration theme

This section reviews a range of global and regional scenarios that explore the change in six individual themes of land degradation. Figure 7.3 shows these themes within the frame of the causal effect chain. Soil and biodiversity are the abiotic and biotic components of land ("state"). Food, water, climate and bio energy/timber/ fibre are the ecosystem service components related to land ("impact"). The themes are structured in this order. The major (sub)components per theme are listed in theme-boxes. The elaboration of the themes varies, reflecting the different scientific stages and approaches for each theme. Section 7.3 focuses on integrated scenarios that take into account two or more interdependencies.

Figure 7 3 The six themes (boxes) and specific components of land degradation that are dealt with in this chapter (middle-left and middle-right column).

The major drivers and responses found in the scenarios are presented in the boxes in the left and right column. Climate change is dealt with as driver of land degradation as well as impact from land degradation. This scheme applies for all scales.



7.2.1 Soil

Key findings

- Historical estimates of soil organic carbon loss range between 50 to 176 Gt C (*established but incomplete*), of which the majority originates from the topsoil in croplands (*well established*).
- Future losses of soil organic carbon until 2050 are estimated at approximately 65 Gt C, of which around 15 Gt C originates from conversion of natural land, around 10 Gt C from decline in land cover and productivity from detrimental land management, around 10 Gt C from drainage and burning of peatlands, and around 30 Gt C from a 1°C warming primarily from high organic carbon soils in northern latitudes (*unresolved*). The impact of CO_2 fertilization on soil organic carbon is unknown (*well established*). These future losses from soils are still modest compared to the 10 Gt C annually from fossil fuels and cement (*well established*).
- Halting soil organic carbon loss from land conversion, poor soil management, and burning and drainage of peatlands would potentially reduce future contribution of soils to atmospheric greenhouse gas levels with around 35 Gt C (*unresolved*). This does not include the prevention of carbon loss from vegetation loss (around 45 Gt C in biomass) and from soil organic carbon from warming.
- Sustainable intensification on existing agricultural land has considerably less emissions from soil organic carbon than expansion of agricultural area (*well established*). The total carbon restoration potential of improved cropland management is between 2 and 12 Gt C over the period 2020-2050, depending on carbon pricing (*established but incomplete*). The total carbon storage potential in croplands would increase up to roughly 80 Gt C in innovative agricultural systems that combine high yields with close to natural soil organic carbon levels (*inconclusive*).
- Preventing future land-based emissions (around 35 Gt C, carbon from vegetation loss not included) and utilizing the carbon sequestration potential in agricultural land (around 80 Gt C) would be significant from a climate change mitigation perspective, given a remaining climate budget of 170-320 Gt C to keep global temperature change below 2°C (*inconclusive*).
- Arid, semi-arid soils and highly weathered soils of tropics and sub-tropical areas are especially vulnerable to soil degradation, in particular, due to soil erosion, soil organic carbon decline, nutrient imbalance and acidification (*well established*).

Indicators: soil organic carbon, productivity, soil erosion, nutrients, compaction, sealing, salinity, soil moisture

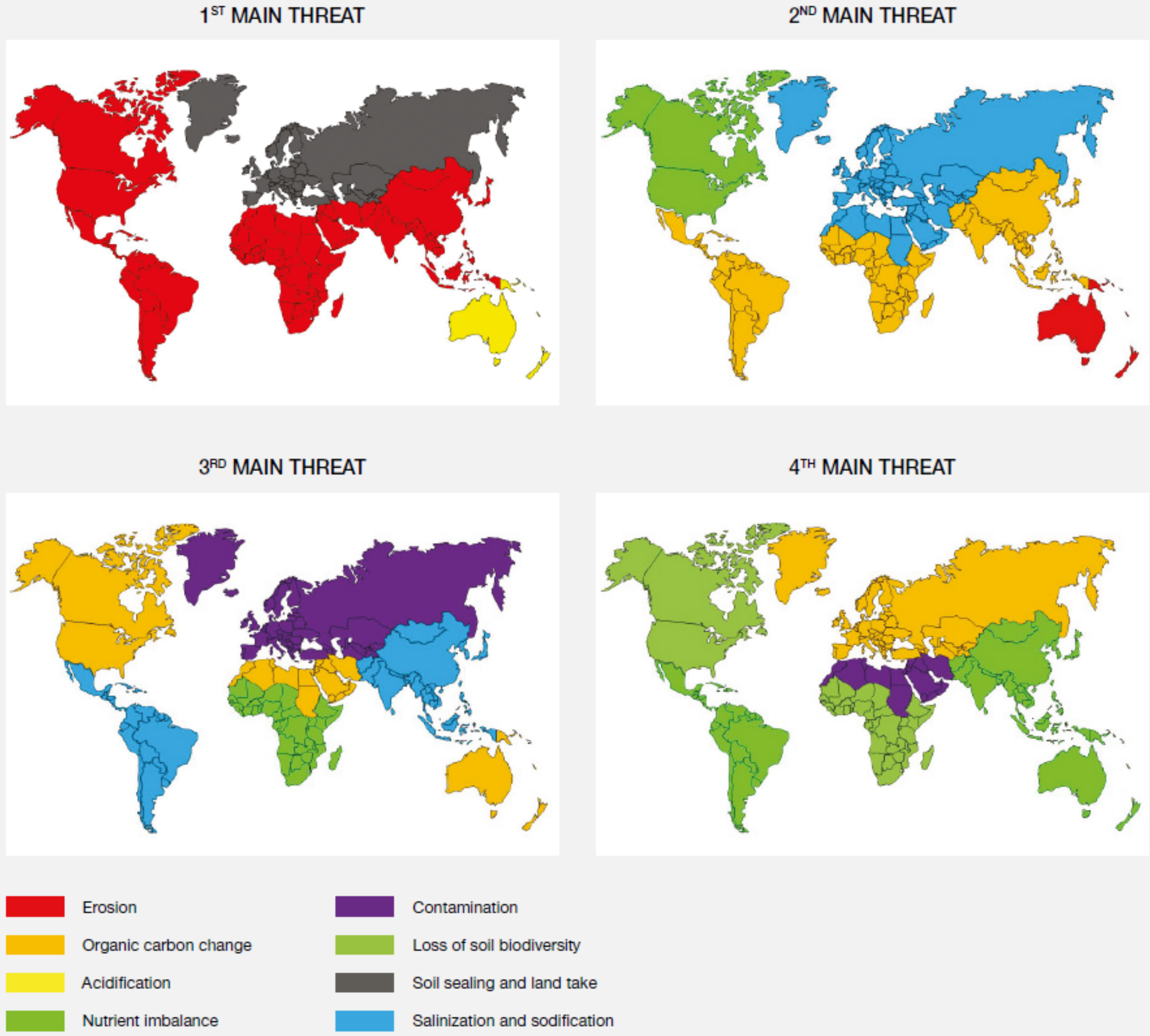
7.2.1.1 Scenarios for threats to soil functions

General

Based on the first State of the World's Soil Resources Report (FAO & ITPS, 2015), Montanarella *et al.* (2016) state that the most significant threats to soil function at the global scale are soil erosion, loss of soil organic carbon and nutrient imbalance from depletion in some agricultural regions and over-application in others (Figure 7.4). While these three overarching threats are global in scope, specific regions are at greater risk from other threats. In Europe for example, soil sealing by the expansion of urban areas is judged to be the greatest threat. In Australia and the South-West Pacific, soil acidification is the greatest concern. In arid and semi-arid parts of Asia the main issue is desertification. In the Middle East, North Africa and in drier sub-regions of Europe, soil salinization is of particular concern.

Figure 7 4 **Global assessment of the four main threats to soil by FAO regions.**

The first main threat is the most severe threat in a region, the second main threat is the second-most severe and so forth. Source: Montanarella *et al.* (2016).



Regional assessments in the State of the World’s Soil Resources Report (FAO & ITPS, 2015) also highlight the differences in the inherent susceptibility of different soil types to degradation. Soils that are low in organic matter often have limited capacities for storing water or nutrients. They are also more likely to experience problems related to the physical properties of soils, such as poor aggregation and hence lower resistance to erosion processes. Low organic matter levels are common in soils in drier regions and in soils formed in ancient, highly weathered landscapes of the humid tropics and subtropics. The inherent susceptibility of these soils to degradation coupled with the regionally specific issues - such as acidification and wind erosion - make them especially susceptible to degradation.

The lack of current global data on the extent of soil degradation, and even on basic soil properties such as soil organic carbon or soil depth, has been widely identified as the major impediment to authoritative simulations of soil degradation processes. For instance, the GLASOD results are still presented in recent major papers such as Amundson *et al.* (2015), even though the evaluations in GLASOD were based on soil conditions in the 1980s. The State of the World's Soil Resources Report (FAO & ITPS, 2015) identifies the need to address the lack of reliable data as one of the four most pressing priorities for the soil science community.

Soil organic carbon

Soil organic carbon (SOC) was identified in the State of the World's Soil Resources Report (FAO & ITPS, 2015) as the key property for land management (see also Chapter 4, Section 4.2.3). To date, the most advanced simulations are those of global soil organic carbon change due to climate and land-use change, although the results differ substantially depending, in part, on the range of soils included in the simulations. Gottschalk *et al.* (2012) use several SRES emission scenarios and gridded land-use data from the Integrated Model to Assess the Environment (IMAGE) to drive a multi-pool SOC model (Roth-C) for simulated soil organic carbon change between 1971 and 2100. Their study does not include changes in carbon stocks in organic soils (peatlands), which are not modelled in Roth-C. The simulations produced consistent increases in global mineral SOC in the first 30 cm of soil. From a baseline of 502 Gt of SOC in 1971, carbon increases between 26.4 to 81.2 Gt C by 2100. The increase in soil organic carbon occurs due to greater plant inputs and a negative trend in global climatic water balance which slows soil organic matter (SOM) decomposition. These two factors override the increase in soil organic matter decomposition rate caused by increasing temperatures. The two factors also counteract the negative impact of land-use change (as shown in meta-analyses, such as Guo & Gifford (2002)). Gottschalk *et al.* (2012) note, however, that the effects of land conversion on carbon levels might be underestimated by the model as only the effects of conversion on net primary productivity (NPP) and litter quality are assessed. The simulations indicate very wide differences between regions. These differences, and their sensitivity to small changes in drivers, lead the authors to suggest that the focus should be on those land management practices that can be implemented to protect and enhance soil organic carbon stocks rather than on attempts to further refine global-scale simulations.

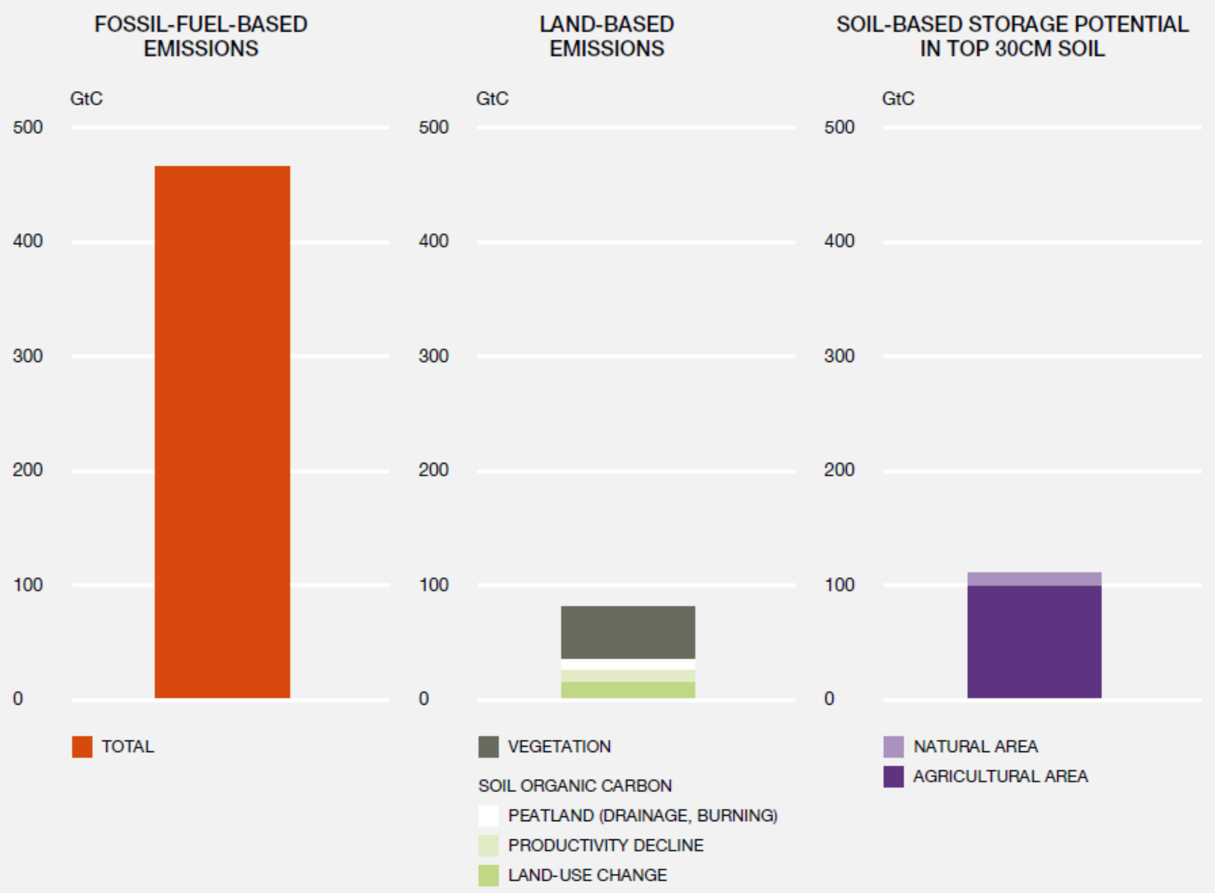
Simulations carried out by PBL Netherlands Environmental Assessment Agency and collaborating institutes (Van der Esch *et al.*, 2017), as part of the UNCCD Global Land Outlook, estimated a total current soil organic carbon pool for the upper 1.2 m of soil at 2013 Gt C. They estimated that 176 Gt C (or approximately 8%) has been lost due to historical changes in land use and land management, and that about 100 Gt C of the total loss comes from topsoil (first 30 cm) in agricultural soils (derived from Stoorvogel *et al.*, 2017a,b) (Figure 7.5). They used the Shared Socioeconomic Pathway 2 (SSP2) scenario to estimate a future loss of about 27 Gt C up to 2050, mainly from southern regions and in particular from Sub-Saharan Africa. These future losses come from the combined effects of future expansion of agricultural land (16 Gt C from land conversion) and continued decline in land cover and productivity related to detrimental land management and hence decline of carbon inputs into soils (11 Gt C). In addition, continued drainage of peat soils and peat fires are estimated to contribute cumulatively about 9 Gt C (±2) by 2050. This amount is based on projections of emissions in Southeast Asia (Hooijer *et al.*, 2010) and extrapolations of current emissions from Europe, including European Russia (Byrne *et al.*, 2004). Combining the impact from future land-use change (conversion of natural land), detrimental land management and loss in peat soils by fire and drainage result in a total loss of soil organic carbon of around 36 Gt C. This does not include the prevention of future carbon loss from vegetation loss estimated at around 45 Gt C (Figure 7.5) (Van der Esch *et al.*, 2017).

Although these cumulative future soil-based emissions are relatively small compared to annual emissions from fossil fuels and cement, 9.9 Gt C/y (Olivier *et al.*, 2015), reducing land-based emissions and utilising the carbon sequestration potential in agricultural land is still significant for climate change mitigation.

Scenarios with a likely probability of keeping global temperature change below 2°C assume future cumulative CO_2 emissions to be limited at 170-320 Gt C (IPCC, 2014a; Rogelj *et al.*, 2016).

Figure 7 5 **Cumulative carbon emissions from fossil fuels from the energy and industry sector (left). Cumulative land-based emissions from soils and vegetation (middle). Carbon sequestration potential in the top soil (< 30 cm) in agriculture and natural land (right).**

All according to the SSP2-productivity decline scenario over the period 2010-2050. Source: PBL Netherlands Environmental Assessment Agency (Van der Esch *et al.*, 2017).



Crowther *et al.* (2016) found that warming-induced carbon losses from high organic content soils – such as those found in boreal forest and tundra – overwhelms minor additions of SOC in mid- and low-latitudes. Their study was based on the extrapolation of measured carbon losses from the upper 10 cm of soils, using results from 49 soil warming experiments. Losses were greatest for soils with initial carbon stocks greater than 7 kg C m^{-2} in the first 10 cm of soil. They extrapolated these losses for 35 years into the future and found that losses under a 1°C warming would likely produce losses of 30±30 Gt C and a 2°C warming could lead to losses of 55±50 Gt C. Their study is not a complete simulation insofar as land use effects are not considered, but land-use change is less likely to be a major factor in high latitude soils such as those most affected under warming scenarios. Changes from organic soils were not considered in the Gottschalk *et al.* (2012) study, and losses from these regions, due to temperature change, were not included in the modelling carried out by PBL for the Global Land Outlook.

The projected carbon changes can be contextualised by comparing them to historical data. Smith *et al.* (2016) use three Dynamic Global Vegetation Models to assess soil organic carbon change due to land use for the period from 1860 to 2010 and report a mean loss (across the three models) of 50.7 Gt C, with a range of 29.7 to 61.9 Gt C. Hence the climate-induced soil organic carbon loss projected by Crowther *et al.* (2016) for a 1°C warming is approximately 60% of the estimated total loss caused by land-use change in the past 150 years (Smith *et al.*, 2016) and approximately 30% of the historical losses estimated by PBL Netherlands Environmental Assessment Agency (Van der Esch *et al.*, 2017).

The management options for increasing soil organic carbon in Europe have been examined in considerable detail, beginning with the widely cited study by Smith *et al.* (1997). They examined five land management scenarios in Europe and found that while the addition of animal manure, sewage sludge or straw show had only limited potential to increase soil carbon stocks over the next century, land sparing through agricultural intensification and afforestation of surplus arable land showed much greater potential. The research shows that although efforts in temperate agriculture can contribute to global carbon mitigation, the potential is small compared to halting tropical and sub-tropical deforestation or reducing fossil fuel burning. More recently, Yigini and Panagos (2016) used four General Circulation Model (GCMs) and Land Use Modelling Platform (LUMP) models, as well as soil and terrain data, to calculate projected changes in soil organic carbon for Europe up to 2050. Their simulations included management-induced land cover changes and projected an increase in soil organic carbon stocks in the top 20 cm for all scenarios - leading them to stipulate an increase between 7 and 13 Gt by 2050. This regional increase has to be placed in the context of the greater losses from high-latitude soils discussed previously.

Productivity

GLADA (Bai *et al.*, 2008b) mapped changes in productivity over the period 1982-2008 derived from NDVI (greenness). It shows areas of high losses, but cannot be extrapolated to the future. The primary reason for this limitation is that broadband indices (such as NDVI) aggregate many different factors into a single metric and the absence of a link to individual drivers makes extrapolation into the future impossible (Jong *et al.*, 2013). Van der Esch *et al.* (2017) estimated a global reduction of net primary production of 5% compared to the natural state, similar to that reported by Smith *et al.* (2016). They extrapolated negative NDVI trends over the period 1982-2010 and up to 2050 – after correcting for the effect of climate change over the same period – and reported an additional cropland expansion of 5% to compensate for future loss over the period 2010-2050. The relation between change in NDVI and land degradation in terms of production loss has been contested (Mbow *et al.*, 2013), and changes in climate, land use, land management and fire may affect NDVI as well. Other regional studies in Europe reach different conclusions regarding how land use in Western Europe will respond to climate change. While some studies suggest that agricultural production demand will be met by using only 30% to 50% of the current agricultural area (allowing remaining land to be reallocated), other studies estimate that current agricultural land area will need to be maintained in order to enable low external input agriculture, as well as to respond to significant growth in global demand, especially from Asia (Bouma *et al.*, 1998; Pizano & García, 2014).

Soil erosion

While global simulations of soil erosion under different management scenarios are not available, projections of future trends based on aggregated data from the past have been devised. A synthesis of meta-analyses of soil erosion plot data in the Status of the World's Soil Resource Report (FAO & ITPS, 2015) produced an average global mean rate of erosion on arable land of between 12 and 15 t ha yr^{-1}, which is approximately equivalent to a loss of 0.9 mm yr^{-1}. The effect of this loss of productivity causes a global median loss of 0.3 % of annual crop yield. Extrapolated to 2050, this would be equivalent to the removal of 1.5 M km^2 from crop production (current global cropland is around 15 M km^2) or around 4.5 M ha yr^{-1}. Please note that this extrapolation is a median global value and does not correspond to the actual loss of specific land areas.

The regional scenarios produced for Europe provide useful examples of policy-relevant information. For example, Panagos *et al.* (2015) developed an estimate of the effects of implementing Good Agricultural and Environmental Conditions (GAEC) on water erosion in Europe (from 2003-2010), using a modified version of the Revised Universal Soil Loss Equation (RUSLE2015). They found that the total annual soil loss in the EU amounted to 970 Mt. The highest mean annual soil loss rate at country level was found in Italy (8.46 t/ha), followed by Slovenia (7.43 t/ha) and Austria (7.19 t/ha). The lowest mean annual soil loss rates were found in Finland (0.06 t/ha), Estonia (0.21 t/ha) and the Netherlands (0.27 t/ha) (Panagos *et al.*, 2015). Over the period 2003-2010 they estimate a reduction in water erosion on arable land from 3.35 t ha^{-1} yr^{-1} to 2.67 t ha^{-1} yr^{-1} due to GAEC implementation. They also combined their model with the HadGEM2 climate scenario and the pan-European Land Use Modelling Platform (LUMP) to estimate water erosion in 2050. The simulation predicts a decrease in agricultural land uses in Europe over that period and an overall reduction of soil loss by 5.8% by 2050. According to the simulations, runoff and erosion decrease by about 10% if conservation measures are applied to the present land use, while the predicted decrease for the alternative land uses (with much more woodland/scrubland) is between 40% and 60%. Similarly, land management related scenarios of soil erosion have been examined in Europe to inform the application of prevention and control measures. Examples include studies examining the impact of land abandonment and related gully erosion and run-off in Spain (Lesschen *et al.*, 2007) and the impact of soil erosion by water in southern Italy (Terranova *et al.*, 2009).

Nutrients

Estimates for nutrient imbalance at the global scale (Foley *et al.*, 2011; Steffen *et al.*, 2015) clearly show a disparity between oversupply of nutrients in some regions and chronic undersupply in others, especially Sub-Saharan Africa. These estimates can be coupled with fertilizer use projections to simulate near-future conditions. Steffen *et al.* (2015) used data on global nitrogen and phosphorous inputs to assess their role in the degradation of soil, water and air, and found that anthropogenic nitrogen inputs are likely to be already beyond the boundary at which significant planetary harm occurs.

Besides carbon, scenarios have been used to examine changes in other chemical properties of soil, including storage and loss of nitrogen and phosphorous in national and sub-national studies. For example, in the upper Mississippi River Basin of the United States, scenarios have examined the impact of land management practices on nitrogen and phosphorus. Expanding continuous corn cultivation throughout the basin resulted in increased nitrogen pollution, while adopting no-till, was the most environmentally effective practice able to sustain production at almost the same levels (Panagopoulos *et al.*, 2014).

Soil compaction and sealing

Soil compaction is an issue for soil management throughout the world (see also Chapter 4, Section 4.2). It is a long-standing phenomenon not only associated with agricultural management, but also with forest harvesting, amenity land use, pipeline installation, land restoration and wildlife trampling. Soil compaction is principally caused by the compressive forces of wheels, tillage machinery and from the trampling of animals. Compaction alters many soil properties and adverse effects are mostly linked to a reduction in permeability to air, water and roots. Topsoil compaction in sloping landscapes enhances runoff and may induce erosion particularly along wheel tracks. Indirect effects of compaction include denitrification which is likely to lead to nitrogen deficiency in crops (Batey, 2009). The loss of soil functions due to surface sealing can be estimated from projections for global urban land expansion. For example, Seto *et al.* (2011) provide an estimate of increasing soil compaction of approximately 1.5 M km^2 by 2030.

Salinization

Salinization is the major degradation threat to irrigated soils globally, affecting over 10 % of irrigated land (see also Chapter 4, Section 4.2.2.2). Groundwater irrigation, the main source of human-induced salinity, is predominant in agricultural areas of the arid and semi-arid climatic zones (FAO, 2011b). The GLASOD study on global soil degradation (Oldeman *et al.*, 1991) revealed that human-induced salinization affects up to 0.76 Mkm^2 globally and 0.53 Mkm^2 in Asia alone. Salinization is present in all continents and more than 0.34 Mkm^2 of irrigated land is already severely salinized (Montanarella, 2007). No quantified projection is available on the possible future extent of global saline areas. However, climate change can induce salinization, as subsoil salts can be propagated to the productive topsoil on new areas of negative atmospheric water balance - either through natural processes or through irrigation and degradation of cultivated land and potentially arable land. Expanding irrigated agriculture to land on saline groundwater results in fast salinization. This is ongoing in many regions, mainly in southwestern Asia and Africa. In some cases, water percolation and influx of upwelling salt from deeper layers can produce soil salinization.

Local salinization scenarios suggest an increase of land under threat of excessive salt accumulation up to 1.5 Mkm^2 in the next decades, degrading both cultivated land and potentially arable land. All continents will experience an increase, which in Asia may be up to 0.60 Mkm^2. Central Asia and the Middle East are among the most threatened, with a possible increase of 0.48 Mkm^2. The current 0.06 M km^2 of land with high salinization potential in Australia may reach 0.17 Mkm^2 by 2050, of which 80% is on agricultural land (NHT, 2001). The extent of salt affected soils might double in Europe, adding another 0.21 Mkm^2 (Szabolcs & Fink, 1974).

Soil moisture and other components of soil degradation

For soil moisture, there is scarce information related to past trends and future scenarios across ecosystems. Some studies have focused on the variability and trends in soil moisture and drought characteristics. Globally and regionally, over the second half of the 20th century, results show an overall increase in global soil moisture. This overall increase was most pronounced over the western hemisphere and especially in North America, while West Africa has experienced significant drying. Europe appears to have not experienced significant changes in soil moisture, a trait shared by Southeast and southern Asia (Hamlet *et al.*, 2007; Sheffield & Wood, 2008). In South America land cover change and extensive use of soil are producing new desert areas where there was previously natural cover, such as dry forests (Pizano & García, 2014). Global estimates for other components of soil degradation such as soil physical deterioration, contamination, or waterlogging have not been attempted since the estimates presented in GLASOD.

7.2.1.2 Prevention and restoration options

Restoration of degraded soil can be a challenging task. For adequate soil restoration to take place, soil needs to be considered as a complex ecosystem that requires the manipulation of physical, chemical and biological components. Single-factor manipulations may in fact produce cascading effects on several ecosystem attributes and can result in unintended recovery trajectories. When complex outcomes are desired, intentional and holistic integration of all aspects of soil knowledge is necessary (Heneghan *et al.*, 2008).

There are three basic strategies that can help in restoring soil conditions: (i) minimizing losses from the pedosphere; (ii) creating a positive soil carbon budget, while enhancing biodiversity; and (iii) strengthening water and elemental cycling (Lal, 2015). Some successful measures in soil restoration – as a response of the degradation process – involve the recovery of nitrogen and carbon, where organic soil amendments like composted material are increasingly applied (Domene *et al.*, 2009). A study conducted in the sub-tropical humid grasslands in South Africa indicated that the decline in grass (vegetative) cover from 100% to 0%-5% reduced the soil organic carbon pool by 1.25 kg/m^2 and the soil organic nitrogen pool by 0.074 kg/m^2 (Lal, 2015). A cost-benefit analysis showed that the implementation of anti-erosion measures, such as terracing, stone walls, grass margins, contour farming, reduced tillage, cover crops and plant residues, in severely erosion-prone agricultural areas could have economic benefits, on- and off-site, of 1.35 billion Euros (Panagos *et al.*, 2015). Good examples of reducing soil erosion using bio-engineering techniques with local communities can be found in Latin America (Petrone & Preti, 2013).

The soil management options that reduce threats and restore soil functions are generally well established and have been codified at an intergovernmental level in the Voluntary Guidelines for Sustainable Soil Management (FAO, 2016). These guidelines draw upon the extensive review contained in the Status of the World's Soil Resource report (FAO & ITPS, 2015). In most cases the barriers to adoption are socioeconomic and cultural, rather than the absence of suitable soil management options. However, there are exceptions. For instance, the issue of nutrient imbalance (simply stated as oversupply of nutrients to soils in developed countries and undersupply in other areas) of Sub-Saharan Africa and South Asia is particularly challenging (Foley *et al.*, 2011; Mueller *et al.*, 2014). Mueller *et al.* (2014) use the approach of trade-off frontiers and suggest that crop production levels in 2000 could be achieved by using approximately 50% less nitrogen fertilizer, and that reallocation from areas of oversupply could allow moderate increases to occur in areas of undersupply. This would also involve decreases of yield in some regions and the authors acknowledge that such reduction could be politically and economically undesirable.

Initiatives to restore soil organic carbon in degraded agricultural soils (e.g., Box 7.2) are gaining prominence in many regions, both as a possible climate change mitigation measure (by removing carbon from the atmosphere) and as a measure to improve soil functions. The "4 per mile" initiative - which aims to increase soil organic carbon levels in global soils by 0.4% per year - has broad support (Minasny *et al.*, 2017). Such a target would be difficult to achieve, as many soils are not actively managed, although various studies have suggested that the sequestration of carbon by increasing soil organic carbon by 0.2-0.5 tonnes per hectare per year could be possible (Minasny *et al.*, 2017). The overall contribution for agricultural-based greenhouse gas mitigation is considerable. Paustian *et al.* (2016) estimate the total GHG removal or reduction potential of improved cropland management at 0.3 to 1.5 Gt CO_2(eq) yr^{-1}, and the potential for restoration of degraded land between 0.1 and 0.7 Gt CO_2(eq) yr^{-1}. The range indicates the effects of different levels of carbon pricing on adoption. Concerns have been raised (e.g., Kirkby *et al.*, 2016; van Groenigen *et al.*, 2017), however, about the concomitant lock-up of nitrogen along with carbon in soil organic matter, and the environmental costs of this lock-up of nitrogen need to be fully explored.

Box 7.2 Biochar

Biochar, or the production of charcoal from biomass via pyrolysis (Ronsse *et al.*, 2013), has been proffered as a potentially valuable transition technology for its climate mitigation and agronomic benefits (Jeffery *et al.*, 2011; Liu *et al.*, 2013). While most biochar studies reporting agronomic benefits have been conducted in the tropics, biochar utilized in temperate soils has the potential to sequester carbon as well as restore soils through greater water retention (Atkinson *et al.*, 2010). An investigation of composted biochar applied to loamy and sandy soil substrates did reveal greater plant yields proportional to the amount of added biochar (Schulz *et al.*, 2013). In addition to variability by soil type, the benefits of biochar application are also contingent on production method, additives and application method (Barrow, 2012), as well as the interaction between soil biota and different biochar types

(Lehmann *et al.*, 2011). Taking into consideration such heterogeneity and the BECCS alternative, exacting the greatest potential may involve employing biochar where the agronomic benefits are most needed (in regions with infertile soils) while utilizing BECCS elsewhere (Woolf *et al.*, 2010). However, the ecological role or impact of biochar, once it has eroded from soil or moved through a soil profile into watercourses, must be assessed (Biederman & Harpole, 2013; Rumpel *et al.*, 2006), taking into account the potential risks associated with the contaminants it may contain (Kuppusamy *et al.*, 2016). Finally, many studies assume that there will be no land clearance for biomass feedstock, indicating potential negative effects (e.g., indirect land clearance) of large-scale biochar implementation (Woolf *et al.*, 2010).

The benefits of increased soil organic carbon in degraded soils could be lost if the need for greater food production for the expanding global population causes expansion of agriculture into forests or grasslands (Rockström *et al.*, 2017). Expansion of agriculture causes a considerable release of carbon into the atmosphere, and many studies (e.g., Rockström *et al.*, 2017) argue that sustainable intensification of agriculture on existing land is necessary (see also in paragraph on soil organic carbon above). The benefits of sustainable intensification are substantial: Burney *et al.* (2010) estimate that the net effect of higher yields (rather than expansion) could reach 161 Gt C of avoided emissions in the period 1961-2010. Hence, combining measures to increase soil organic carbon on degraded land and to sustainably increase crop yields on existing agricultural land is required. A potential carbon storage in the topsoil of croplands of roughly 80 Gt C could be created if innovative agricultural systems are developed that combine high yields with close-to-natural soil organic carbon levels.

7.2.2 Biodiversity

Key findings

- By 2010, the average population sizes of species had declined by around 34% compared to the natural state, and the decline is projected to reach 38-46% by 2050, depending on the scenario (*well established*). Species extinction rates have rapidly increased, and suggest that we are on the edge of a sixth mass extinction event, and extinction risks are increasing for most taxonomic groups, although at widely differing rates (*well established*). In the second half of the 21st century, the long-term target of halting biodiversity loss can not be achieved in any of the business as usual or prevention-oriented scenarios. Between 2050 and 2100, an additional 9% is projected to be lost under the SSP2 scenario (*inconclusive*). The remaining extent of wilderness is expected to decline by 10-13 million km^2 by 2050. Declines are expected to continue in all world regions, but the greatest losses will most likely be in Central and South America, sub-Saharan Africa and Asia, since those are the locations of the remaining land suitable for production purposes (*well established*). By 2050, much of the remaining biodiversity will be situated in areas unsuitable for settlement or production – in particular, deserts, mountainous areas, tundra and polar systems (*well established*).

- The major causes of future terrestrial biodiversity loss are expansions in land for crops, pasture and forestry, climate change, infrastructure development, nitrogen deposition, invasion by alien species and urban spread (*well established*). The impact of climate change on biodiversity will accelerate in all scenarios (*established but incomplete*). Drivers of biodiversity loss vary regionally and across biomes (*well established*).

- Alternative, prevention-oriented scenarios illustrate the importance of trade-offs between siloed, sectoral policies, and the importance of integrated approaches, utilizing co-benefits between different policy objectives. Measures for reducing biodiversity loss which favour multiple goals such as climate change mitigation, food production, water and energy security, are most effective. Examples of such measures include the moderation of total consumption of proteins, halting or even reversing the conversion of natural land, ecologically-efficient measures to increase the yields per hectare for all commodities, halting or reversing the loss of land-based carbon, reducing carbon emissions from fossil fuels, and expanding the global protected area network in strategic locations (*well established*). In contrast to the projected impact of these prevention measures, the potential of ecosystem restoration is unknown (*well established*).

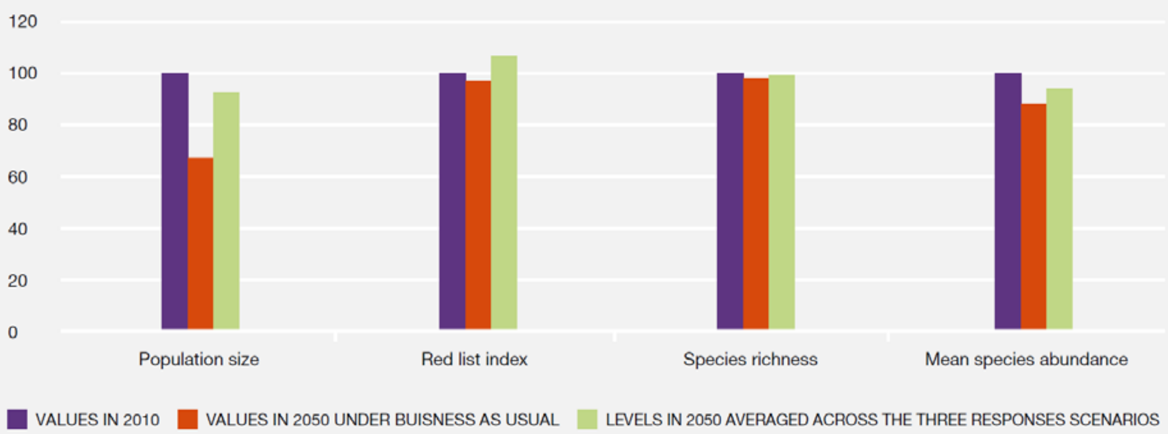
Indicators: mean species abundance, wilderness, Red List Index, population size, species richness, extinction rate, ecosystem extent.

7.2.2.1. Baseline projections for biodiversity loss

The process of biodiversity loss is generally characterized by the decrease in population size of many original species and the increase in the population size of a few other species (often common and human-favoured ones) as a result of human activities. This process of replacement of many original species by a few common species is called "homogenization". Indeed, ecosystems are becoming increasingly similar (Lockwood & McKinney, 2001; CBD, 2007). To study this process, biodiversity loss is generally expressed by several different types of indicators. Decreasing population size of a species may eventually lead to local and global extinction (Red List Index). The full extinction of species is just the last step in this process. The overall picture of homogenization is provided by the mean species abundance (MSA). The MSA of a particular ecosystem is the average decline in the population size of the original species compared to their population size in the undisturbed state (Alkemade *et al.*, 2009). Another indicator is species richness: the species count in a specific site of study. Change in global species richness is not the same as a global average of change in local species richness. Local species richness may even temporarily increase due to the introduction of new, common species from human activities, gradually replacing the original species (Lockwood & McKinney, 2001). Figure 7.6 shows the global loss in biodiversity using the aforementioned indicators. Wilderness describes the extent of natural area that is still close to its natural state. Ecosystem extent describes the remaining extent of natural ecosystem types independent of its remaining quality. As a group, these indicators provide complementary information on different aspects of the state of biodiversity.

Figure 7 6 **Four indicators of terrestrial species response to socioeconomic scenarios following the business-as-usual or Rio+20 scenarios for 2050.**

Population size and IUCN Red List status are for carnivores and ungulates; species richness and mean species abundance (MSA) are for a wide range of species groups. See the 4th Global Biodiversity Outlook technical report 78 for details. Source: Leadley *et al.* (2014).



Various global scenario analyses have been published in scientific journals (Jetz *et al.* 2007; Visconti *et al.* 2011; Rondinini & Visconti 2015) and reports of international bodies (MA, 2005a; OECD, 2012; PBL, 2010, 2012; CBD, 2001, 2006, 2010, 2014; UNEP, 2007, 2012). The business as usual scenarios found in these studies all show major loss in biodiversity up to 2050 for all indicators, both globally and in all ecosystem types and continents.

According to these studies, in a business as usual scenario, MSA values will drop around 10% from about 70% to 60%[2], between 2000 and 2050, with the majority of loss expected to occur before 2030 (OECD, 2012; PBL, 2010). The recent SSP scenarios show a similar trend: biodiversity loss is projected to increase from 34% in 2010 to 38%, 43% and 46% under SSP1, SSP2 and SSP3 by 2050, respectively. The SSP2 productivity-decline scenario (a SSP2 variant that assumes continuing productivity decline caused by land management and the ensuing impacts on land condition) projects an additional biodiversity loss of about 1% by 2050, equivalent to a complete loss of the original biodiversity (entire populations of all original species) of an area about 2.4 times the size of continental France. The major share of the 1% is caused by agricultural expansion to compensate for the loss in productivity (Van der Esch *et al.*, 2017).

In line with this baseline projection, between 2050 and 2100, an estimated additional 9% of MSA is projected to be lost under the SSP2 scenario, with land and other policies likely unable to completely compensate for accelerating climate change after 2050 (Van der Esch *et al.*, 2017). On a regional level, reflecting on future trends in land use, major declines in future freshwater biodiversity (in MSA) are expected in Africa, Asia, and Latin America while modest improvements in the USA and Europe are expected under baseline conditions (Janse *et al.*, 2015).

Extinction risks are increasing for most taxonomic groups, although at widely differing rates. Based on historical trends, observed from the IUCN Red List Index, increasing numbers of studies support the idea that we are on the edge of a sixth mass extinction event: last few centuries showed exceptionally high and increasing rates of species loss (Ceballos *et al.*, 2015; WWF, 2016; IUCN, 2014). Also, extinction rates are expected to accelerate further as a result of global future temperatures, threatening up to one in six species under current policies, especially in South America, Australia and New Zealand (Urban, 2015).

Newbold *et al.* (2015) estimate that, globally, land conversion and related changes had reduced average within-sample species richness by 13.6% by 2005, compared to the natural baseline, and predict another 3.4% losses under a business as usual land use scenario by 2095 – with losses concentrated in biodiverse but economically poor countries. They indicate that more than 20% loss of local species richness could substantially impair the contribution of biodiversity to ecosystem function and services and thus to human well-being. Globally, they estimate that reductions in average plot-level species richness exceeding this 20% reduction level will increase to 41.5% in 2095. At a regional scale, studies found similar spatial trends in expected losses in bird and mammal species (in species richness) and identified that Mexico, Congo (DRC), Tanzania, Brazil and USA rank highly for both national and global species losses (Jetz *et al.*, 2007; Visconti *et al.*, 2011).

Under the business as usual scenarios, the wilderness indicators are expected to decline by 10-13 million km^2 (PBL, 2012). By 2050, about 50 million km^2 of all land (~132 million km^2) is estimated to remain close to its natural state (wilderness), mostly because it is too poorly suited for widespread human activity. These wilderness areas will generally be found in deserts, mountainous, boreal and/or sub-polar, arid and semi-arid zones.

Major drivers of biodiversity loss

[2] On this biodiversity section, all % should be read as per cent points.

The literature provides quite a consistent picture of the major drivers and their relative share to future biodiversity loss in their baseline scenarios (MA, 2005a; OECD, 2012; PBL, 2010; WWF, 2016): loss of habitat (resulting from land use changes, encroachment, fragmentation and infrastructure), overexploitation, invasive species, climate change and pollution (i.e., nitrogen deposition). The order of impact that these drivers have had in the past (and will have in the future) is expected to shift, with climate change playing an increasingly big role (Van der Esch *et al.*, 2017).

Drivers of biodiversity loss may differ between biomes. For instance, climate change is a dominant driver in tundra, boreal and cool conifer forests, savannah, and deserts; nitrogen deposition is particularly important in warm mixed forests and temperate deciduous forests which are sensitive to this driver (MA, 2005a, b) (see also Chapter 3 for discussion on drivers). Conversion of natural forests and grasslands was dominant in developed countries until recently, but now has shifted to developing countries where it will continue to be a dominant driver in the next decades (Van der Esch *et al.*, 2017; Leadley *et al.*, 2014; PBL, 2010).

For regional scenarios on biodiversity loss, the extent of ecosystems is the only indicator uniformly reported across regions and hence comparable across local scenario studies. The absolute values and spatial pattern of losses predicted under regional scenarios are similar to the global ones, which is to be expected given that many regional scenarios relied upon downscaling of global scenarios to derive more accurate spatial predictions of change (e.g., Carpenter *et al.*, 2015; Sohl *et al.*, 2014; Verburg *et al.*, 2013).

These continued changes in ecosystem extent results in significant loss of biodiversity, by 2050, across all regions (Alcamo *et al.*, 2011; Radeloff *et al.*, 2012; Sohl *et al.*, 2014; van Vuuren *et al.*, 2006; Verburg *et al.*, 2008). However, the spatial distribution of land-use changes is highly variable, driven by local land suitability, population and economic growth and climate (Rondinini & Visconti, 2015; Verburg *et al.*, 2008, 2013). For example, focusing on the expansion of agricultural land as a driver of loss of natural areas – based on EURURALIS scenarios – agricultural land is predicted to remain stable across OECD countries (-2% to +5% by 2030 relative to 2000). Significant increases are expected in Latin America (32% to 78% by 2030 relative to 2000), Sub-Saharan Africa (38% to 52% by 2030 relative to 2000) and Asia (13% to 16% by 2030 relative to 2000). This indicates that natural areas in Latin America and Sub-Saharan Africa will significantly decline from expanding agricultural lands, at least up to 2030 (Verburg *et al.*, 2008).

7.2.2.2 Alternative pathways

Policy options, trade-offs and synergies

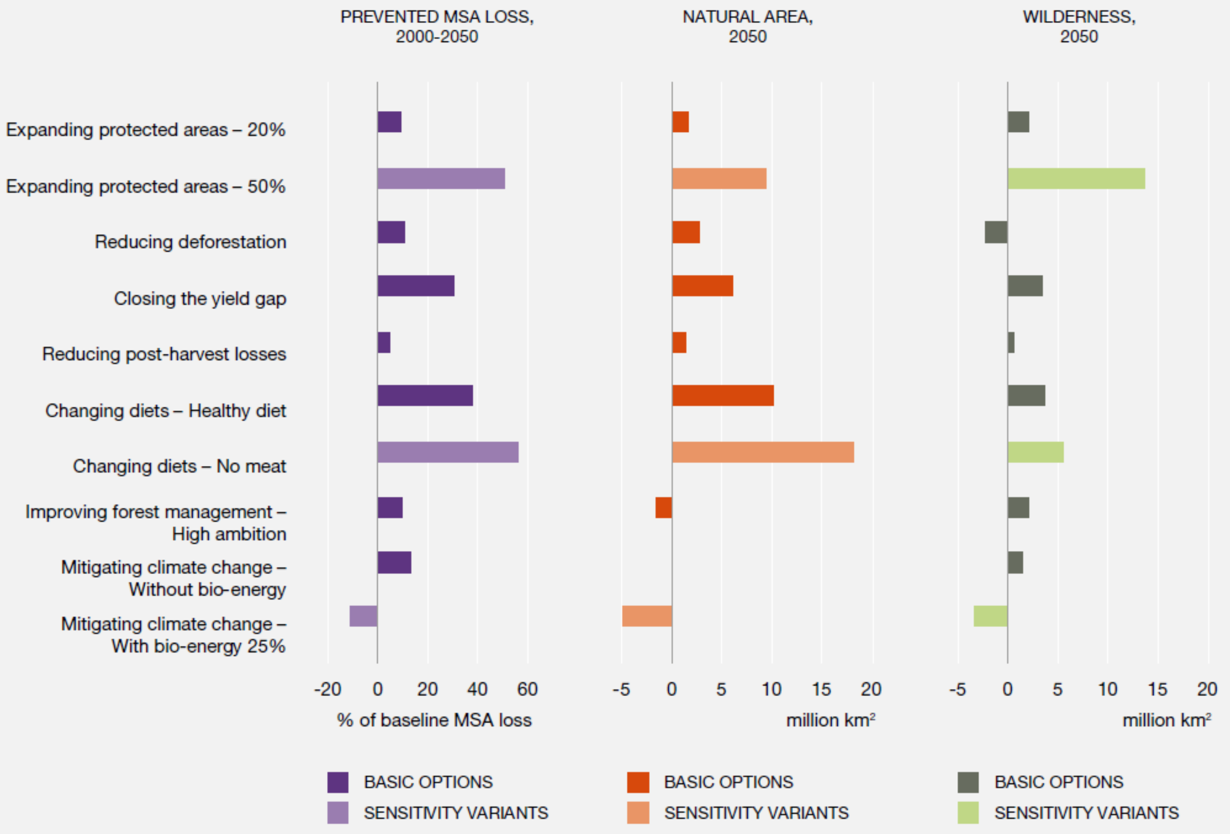
The increased demands for food, fibre, biofuels, water, infrastructure and settlements result in growing competing claims for land, and as a consequence of that, trade-offs between biodiversity and ecosystem services. Coordination of targets, strategies and instruments across different policy fields is essential to reap co-benefits and synergies to avoid any unnecessary or unexpected negative side effects (PBL, 2010). This highlights the need for integrated policies in order to meet these demands while preventing or reducing loss of biodiversity and ecosystem services (MA, 2005a; Obersteiner & Walsh, 2016; OECD, 2012; PBL, 2010; CBD, 2010).

Alternative global scenarios on the prevention of biodiversity loss have been analysed by PBL (2010), of which less meat consumption[3] and closing the yield gap in agriculture[4] were found to be most effective (Figure 7.7), although neither would be sufficient to fully halt future biodiversity loss. Forest plantations

[3] Worldwide, consumption patterns slowly converge to 50% above the consumption level suggested by a ‘Healthy Diet’ (daily per capita intake of 10 g beef, 10 g pork, and 46.6 g of chicken meat and eggs), the so-called Willet-diet (Stehfest *et al.*, 2009).
[4] Baseline yield improvement are increased by 50% in OECD countries to a maximum increase of 1.5% per year.

and mitigation of climate change take more time to come into effect, but their impact will increase up to 2100. Bio-energy has a substantive net negative impact on biodiversity considering the loss of natural land, on the one hand, and biodiversity gains from climate mitigation, on the other hand (see also Section 7.3.5).

Figure 7 7 Prevented global biodiversity loss per option compared to the baseline scenario. Source: PBL Netherlands Environmental Assessment Agency (2010).



Analyses done by Strassburg *et al.* (2012) indicate that an adequately-funded and widely implemented REDD mechanism could prevent many species extinctions, on top of its climate mitigation potential. However, it should be considered that second order effects of REDD might have negative effects on biodiversity, for instance through the expansion of species-poor tree plantations in situations where forests are managed purely from a carbon perspective.

Expansion of the global protected area network is another key measure to prevent biodiversity loss. Expanding protected areas to 17% of land in 2020 (Aichi Biodiversity Target 11) would protect up to 21% of the threatened species (by protecting the cheapest land), although these areas might just displace the issue of land conversion. A more strategic location of protected areas (at 1.5 times the cost of the cheapest option) would protect up a range of 75% of species, but would result in a mere 1% decline of global production land (Venter *et al.*, 2014). Projecting such strategic protected areas (with the 17% target) to 65 major ecoregions, indicates that the largest efforts would have to come from OECD Europe (10%) and BRICS countries, especially Russia (14%) and India (10%) (OECD, 2012). However, in order to achieve the targets cost-efficiently for all countries, ecoregions, important sites and species, about 7 million km^2 would need to be added to the existing 19.7 million km^2 terrestrial protected area network. Here, poorer countries have the largest relative shortfalls (Butchart *et al.*, 2015). Local scenarios demonstrate that strategic land-use planning that considers multiple objectives (e.g. food production and biodiversity conservation) can reduce trade-offs (Bryan *et al.*, 2011; Adams *et al.*, 2016).

According to PBL (2010), protected areas are becoming increasingly more effective at larger percentages of area protected (Figure 7.7). At larger percentages of protection, agricultural expansion into unprotected natural areas becomes less attractive due to scarcity of available land. Consequently, yield increase per ha within existing production systems becomes a more profitable alternative.

Measures preventing biodiversity loss can sometimes have unexpected rebound effects and trade-offs when implemented in combination with each other. For example, reducing post-harvest food loss has a downward impact on food prices, discouraging (costly) investments in yield-increase per ha and resulting in more agricultural expansion. Globally, closing the yield gap in agriculture will reduce agricultural expansion into forests, lowering the one-off yield of timber. This would lead to an increase in forestry plantation area (and related biodiversity loss) to meet global demand in the coming decades (PBL, 2010).

Prevention and restoration scenarios

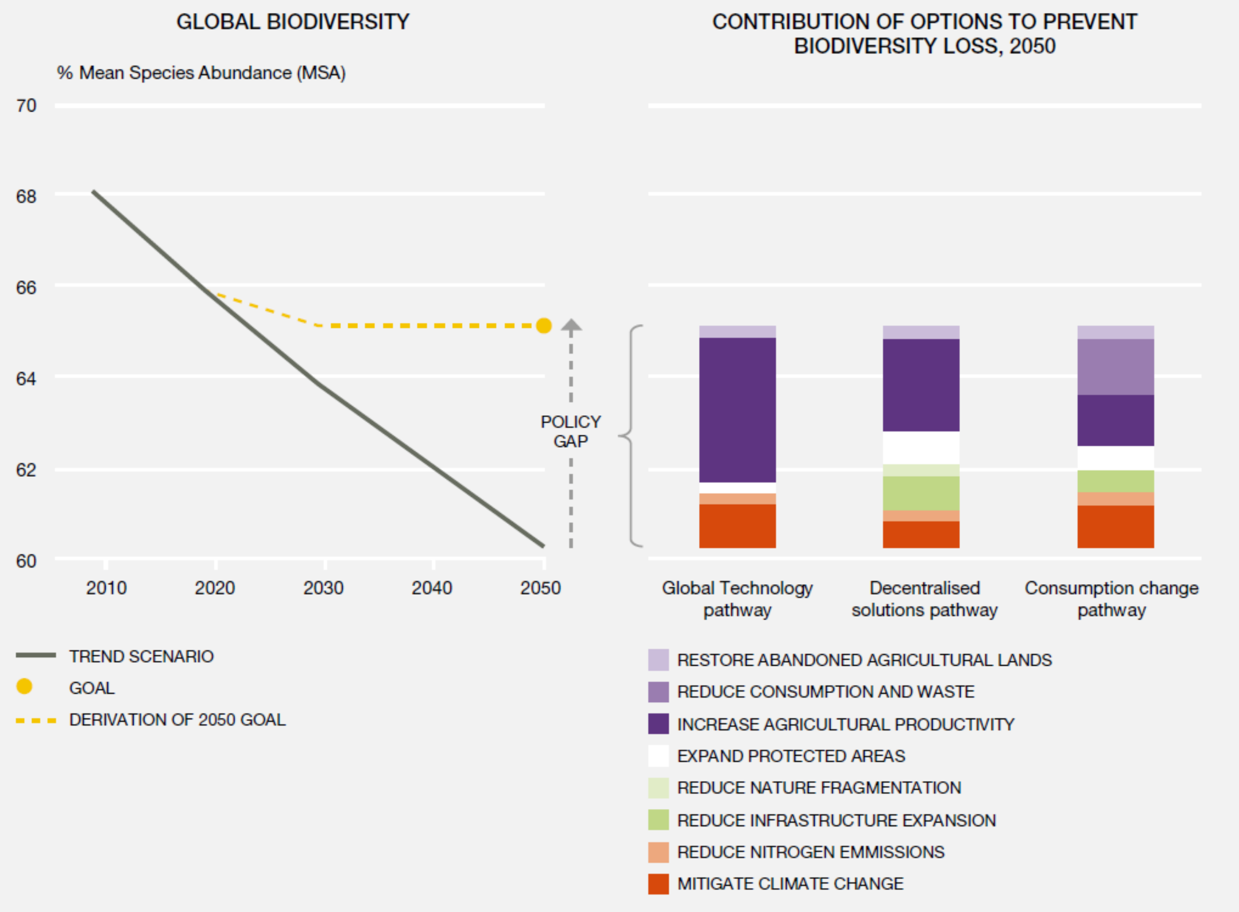
In an ambitious yet feasible scenario, combining measures in agriculture, forestry, climate mitigation, consumption change, food waste and nature protection (Figure 7.6), PBL (2010) found that the loss of biodiversity could be halved by 2050 (5% MSA) compared to the baseline scenario - although the sum impact of individual measures could theoretically halt most of the biodiversity loss (around 10% MSA). This serious reduction in effectiveness is a by-product of trade-offs between measures.

The fourth Global Biodiversity Outlook (GBO4) presented three alternative pathways to achieve the Aichi Biodiversity Targets, taking into account multiple targets from international agreements, including eradicating global hunger, avoiding biodiversity loss, limiting global mean temperature increase to 2°C, universal access to safe drinking water, basic sanitation and modern energy sources, and reducing urban air pollution and fertilizer use (Leadley *et al.*, 2010; PBL, 2014). These three alternative pathways are:

1. Technology improvement such as intensive agriculture and a high level of international coordination;
2. Decentralized solutions, such as extensive agriculture, natural corridors and national policies for equitable access to food; and
3. Changing human production and consumption patterns, including reduced meat intake, food and water waste.

Figure 7.8 shows the contribution of different measures to prevent biodiversity loss under these three pathways. In the search for scenarios that achieve these targets simultaneously, major and transformative change in all relevant sectors were assumed. Although the analysis showed a major reduction of the rate of loss by 2050, the study did not address the question of whether the rate of biodiversity loss gets halted by 2050 and beyond. Rondinini and Visconti (2015) found that suitable habitats for mammals decline with 10% across Europe under both the “Trend” and “Consumption Change” scenarios in GBO4 (see Figure 7.8), the latter partly due to rebound effects. According to Newbold *et al.* (2015), “concerted action and the right societal choices” can deliver positive biodiversity changes. In one scenario, they found an up to 1.9% increase in average within-sample species richness by 2095. This scenario assumed carbon pricing, crop yield increase, improvement of agricultural efficiency, dietary shifts and small increase in biofuel plantations. Urban area was kept constant and non-land use related impacts such as climate change and nitrogen deposition were not included.

Figure 7 8 Alternative pathways to reduce biodiversity loss up to 2050.
Source: PBL Netherlands Environmental Assessment Agency (2012).



Another example of a target-seeking scenario is the analysis by Visconti *et al.* (2016) on how to eradicate hunger, ensure universal health and access to modern energy, while not being mutually exclusive with short-term biodiversity goals. They found that meeting these objectives would require a scenario with aggressive policies for rapid and widespread implementation of sustainable production practices and changes in consumption (less energy and meat consumption, waste reduction), progressive environmental legislation (carbon tax) and strategic placement of protected areas. This analysis is roughly in line with the findings of PBL and the GBO4.

Many local scenarios take advantage of optimization approaches to land restoration scenarios that meet multiple objectives for nature's contributions to people while minimizing trade-offs (e.g., Bryan *et al.*, 2011; Law *et al.*, 2015). The use of optimization techniques to cost-effectively achieve restoration targets may significantly reduce costs (by half in Finland compared to business as usual, Kotiaho *et al.* (2016)) and increase economic returns up to four-fold in Australia (Bryan *et al.*, 2011). In contrast to global scenarios, some scenarios at the regional and local level show an increase in local biodiversity after a mix of prevention (protection) and restoration actions (Bryan *et al.*, 2011; Petz *et al.*, 2014a). However, the external impacts of these actions outside the study area are often not explicitly taken into account, in particular, restrictions on local production functions leading to lower yields. The need for more land to compensate for the production loss should be taken into account to assess net biodiversity gain or loss. In a catchment scale analysis in South Africa, Petz *et al.* (2014b) found a slight (1%) decrease in MSA after an increase of production. Under a compromise (balances agriculture, restoration and conservation) or conservation scenario (extended conservation and restoration), they found a 2-7% increase in MSA, albeit at the cost of production levels.

In conclusion, even under alternative pathways, biodiversity losses appear unavoidable given the accelerating growing demand for food, energy, fibre, timber, housing, infrastructure and water, a quadrupled global economy and accelerating impacts from climate change over the first half of 21st century. Virtually all global and most regional restoration and conservation scenarios show a continued decline in biodiversity, although a few regional or local, more extreme intervention scenarios resulted in improved components of biodiversity. Significantly reducing the speed of biodiversity loss has not been achieved yet and goals to halt the loss by 2050 appear hard to meet.

7.2.3 Food

Key findings

- Current trends in drivers of land degradation suggest that in a "business as usual" scenario land degradation and its impacts on food security are likely to increase, especially in developing regions with high and increasing demographic pressure, scarce land and water resources and weak governance structures, leading to increased risks of hunger, conflict and mass migration (*established but incomplete*).

- The effects of land degradation on food security are not taken into account in any global scenario study. The reason for this is a lack of comprehensive cross-disciplinary analysis of the interlinkages between the different types of drivers and mechanisms of land degradation, and their effects at various spatiotemporal scales. Nevertheless, today, the tools and knowledge exist to start such an endeavour (*established but incomplete*).

- Such a study should also take explicit account of the limits to which the effects of land degradation on food production can be compensated or concealed by increasing inputs, as commonly occurs in developed regions (*established but incomplete*).

Indicators: total production, productivity, crop area, abandonment.

7.2.3.1 Global scale

Land degradation affects agricultural production, and thus food supply, in several direct and indirect ways (Box 7.3). Current trends in most drivers of land degradation mentioned in Chapters 3 and 4 suggest that land degradation is expected to increase in many regions. This is especially the case in developing regions where population growth is high, people lack resources for long-term investments in land, there are few alternatives to agriculture and existing weak governance structures to secure land rights, let alone reward non-marketable public goods. In such regions, the impacts of these combined pressures – aggravated by land degradation and rapid growth in food demand – can imperil local food security. Climate change and land degradation are often blamed for contributing to conflict and large-scale migration in vulnerable regions (Burke *et al.*, 2009; Kelley *et al.*, 2015; Van Schaik & Dinnissen, 2014). Studies investigating causal relations have yet to come to a consensus on the extent of such attribution in specific cases. This is mainly due to the complex nature of such societal phenomena, with multiple causes and easily confounded cause-effect relations (e.g., Black *et al.*, 2011; Gleditsch, 2012; Hermans-Neumann *et al.*, 2017; Ide, 2017). A comprehensive study from Clingendael Netherlands Institute for International Relations (Van Schaik & Dinnissen, 2014) found that land degradation functions as a potential 'threat amplifier' for conflict, with food and water scarcities exacerbating existing socioeconomic problems that are further compounded by rapid demographic growth. Further, impacts from climate change and land degradation will disproportionately affect those least developed countries which are biophysically vulnerable to land degradation impacts (e.g. semi-arid regions) (Van der Esch *et al.*, 2017). Comprehensive scenario analyses can help to assess knock-on effects, to raise awareness for risks and to identify promising "levers of intervention" to curb negative trends or to prioritize policy action.

Regarding the current state of the art, despite the fact that the impacts of land degradation on crop production are fairly well understood at the individual plot level, global scenarios regarding the impacts of land degradation on food are practically restricted to assessments of projected losses of productive agricultural land or crop yields. Moreover, such estimates are quite variable and contrast with actual yield records at national, regional and global levels (which are, for the most part, still rising), and are affected by the lack of evidence for agricultural abandonment of vast areas of land due to land degradation (Gisladottir & Stocking, 2005; Lambin & Meyfroidt, 2011).

The main reason for such divergences is that the effects of land degradation on crop yields can be compensated (or masked) by shifting production to newly explored areas and increasing inputs of fertilizers, irrigation, chemical pest control, plant breeding and deep ploughing. In low-input systems, nutrients collected from large semi-natural areas by cattle can be applied as manure to small crop land areas (effectively mining the surrounding areas) and partially degraded arable areas can be temporarily set aside. In the short- and medium-term, the net effect of such measures on crop yields can be positive. In general, however, these measures can only be effective to a certain extent and for a certain amount of time. While they help to overcome site-specific negative effects immediately, in the long run, as land use intensifies, they contribute to land degradation in a less visible, broader sense and in a wider area.

Box 7.3 Summary of mechanisms of how land degradation affects agricultural production

- The effective volume of soil that can be explored by roots is reduced in physically degraded soils, due to reduced soil depth and difficulty for roots to penetrate dense layers - thus limiting the availability of soil water and nutrients to plants.
- Physically degraded soils typically have poorer aeration and water retention characteristics than non-degraded soils.
- Gully erosion in extremely degraded land makes the land unfit for agricultural practices.
- Chemically depleted soils lose their stock of essential plant nutrients.
- Soil salinization directly affects yields and can make the land totally unfit for agriculture.
- Loss of soil organic carbon from top soils leads to the partial loss of the ability of soils to store nutrients and water, and a loss in structural soil stability - thus making the soil more prone to physical degradation.
- Increased risks to irrigated agriculture due to irregular water supply (in rivers) and depleted groundwater resources of adequate quality.
- Loss of supporting ecosystem services due to biodiversity loss in field margins and adjacent areas (e.g. pollination, resilience against pests and diseases).
- Soil sealing and expansion of other land uses (e.g. associated with urbanization reducing agricultural area).

Source: (Bindraban *et al.*, 2012; FAO, 2011b; UNCCD, 2017)

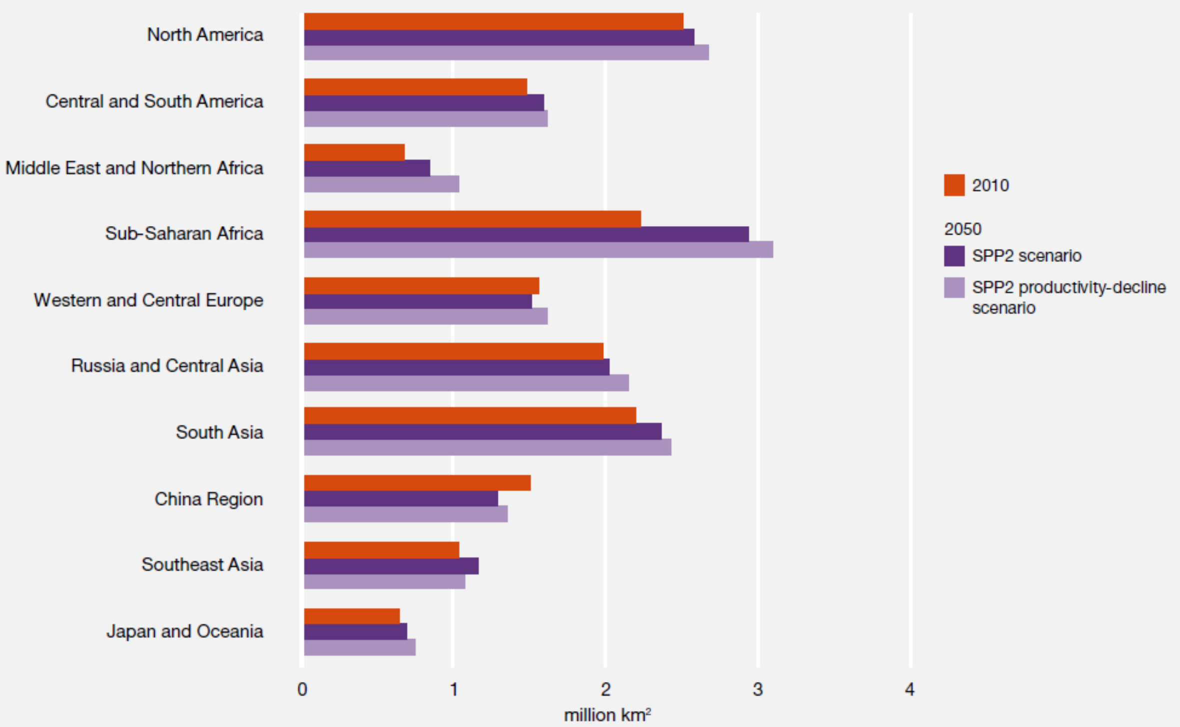
The challenge is therefore to model, not only the current impacts of land degradation on food supply, but also the underlying processes and the proximity of possible tipping points (Bindraban *et al.*, 2012; Montanarella *et al.*, 2016).

However, a crucial weak link within current global integrated assessment modelling frameworks, such as IMAGE (Stehfest *et al.*, 2014) and the GLOBIOM-EPIC-G4M-POLES Integrated Model Cluster (IIASA, 2014), is the deficiency of feedback from environmental impacts (calculated by these models) as changes in soil properties, freshwater availability for irrigation and so on. As a consequence, climate permitting, yields will always tend to increase due to assumed technological developments. Hence, the challenge is to combine such integrated assessment models with models or mechanisms which account for the most important interactions between land use, land management, land qualities, water relations and soil properties, and which are well calibrated and tested at the global level.

A first attempt in this direction – without explicitly modelling such feedback mechanisms – was made in a recent study as a support to the first Global Land Outlook (Van der Esch *et al.*, 2017; UNCCD, 2017). In the SSP2-productivity decline scenario, the business as usual scenario SSP2 was constrained with continuing decline in productivity in those areas that show negative trends over the period 1982-2010, as inferred from time series of satellite images (Schut *et al.*, 2015). To approach the change in land condition from detrimental land management, the impact of climate change on productivity, over the period 1982-2010, was subtracted before trend extrapolation to 2050. The impact is expressed in terms of the additional cropland area required to compensate for the productivity loss. Furthermore, it is assumed that for each of the 26 world regions considered in the IMAGE model, total crop production is the same as in the SSP2 scenario. Nevertheless, the increase in cropland area is not a simple inverse proportion of declines in yield. Because the model was designed to convert the potentially more productive areas first, subsequent expansion will tend to take place in areas with decreasing productivity. The resulting developments in cropland area, at a global level, are shown in Figure 7.9. Compared to 2010, at the global level, the cropland area in 2050 increases by about 12% in the SSP2 productivity-decline scenario; 8% originates from the increase in food demand and about 5% from compensation for productivity decline. Over the considered 40-year period, this corresponds to an average of about 2 Mha per year, which is comparable to the 1-2.9 Mha per year estimated "loss of land and productive potential due to land degradation equivalent" by Lambin and Meyfroidt (2011). This is however considerably less than the 10 Mha per year quoted by David Pimentel (Pimentel, 2006), based on studies on soil erosion in the nineties.

Under these assumptions, the regions that show the biggest expansion associated with productivity decline are the Middle East and Northern Africa, Sub-Saharan Africa, Russia and Central Asia. Particularly in Africa – where a strong impact of land-related productivity loss is combined with a steep increase in food demand – arable land use increases by 57% from 2010 to 2050 in the SSP2 productivity-decline scenario, of which 15% is attributed to productivity decline. In contrast the net change in arable land area in Western and Central Europe, and China-Mongolia region, is mostly negligible or even negative. In these regions, the negative impacts of productivity loss due to changes in land condition are mainly reflected by smaller agricultural areas that returned to nature, rather than arable land expansion.

Figure 7 9 Cropland area in 2010, and in 2050 according to the SSP2 and SSP2 productivity-decline scenarios. Source: Van der Esch *et al.* (2017).



Overall, the results suggest that continued land productivity loss at current rates – as reflected in the SSP2-productivity decline scenario – will most strongly affect developing regions where population pressure is increasing and/or land is already scarce. In a more refined analysis, the results clearly show the mounting pressure on land in the forest and savannah areas of Sub-Saharan Africa, the very marginal lands of northern Africa and parts of the Middle East, and in regions where much of the land is already under intensive agriculture, such as in India and parts of Southeast Asia.

Beyond the biophysical and macro-economic drivers considered in the PBL study, socioeconomic and demographic drivers should also be accounted for within land degradation specific scenario analyses. Model-assisted scenario analyses could examine how land degradation affects the competition for land and water resources in more or less developed regions, and depending on the strength of governance structures, demographic pressure, and the scarcity of land and other resources. Subsequent analyses may examine impacts on social cohesion and eventually the risk of large-scale migration and/or conflict. Given the complex, multi-scale and multi-disciplinary nature of the processes involved, this requires advanced, spatially explicit, integrated assessment modelling tools complemented with expert-based analysis and participatory approaches. Moreover, the explorative study by PBL did not include economic analysis nor a scenario for restoration (Van der Esch *et al.*, 2017). Other weaknesses in the analysis include the underlying assumptions to translate 30-year trends in remotely sensed indicators to yield projections and the assumption that agricultural production at the IMAGE region level, in a given year, is equal in all scenarios (Van der Esch *et al.*, 2017). Therefore, the results presented here as cropland increase to satisfy future food demand, could perhaps be best interpreted in terms of "pressure on or demand for land" rather than km^2. The most important possible flaw in the analysis, however, is the lack of process-based analysis and thus the inability to pinpoint potential productivity loss that is hitherto masked due to increasing use of inputs, but which might become visible in the future as yields are levelling off or declining. Taking account of such processes requires a combination of remote-sensing and field data, and process-based modelling, not yet available.

7.2.3.2 Regional scale

Similar to global scenarios, regional scenarios with specific attention to land degradation and its interrelations with food production and availability are scarce. Ye and Van Ranst (2009) simulated the effect of soil degradation on long-term food security in China. For the estimation of relative yield penalties due to soil degradation, they used the qualitative soil degradation classes of ISRIC's ASSOD map (van Lynden & Oldeman, 1997), which distinguishes five impact classes and five types of soil degradation: water erosion, wind erosion, physical deterioration, fertility decline and salinization. Scores were attributed to the impact classes, making a distinction between three levels of input management. An iterative least square procedure was used to link the overall scores, at the level of geographic subdivisions, to observations of yield penalties. Three soil degradation scenarios were designed: (i) a zero-degradation scenario, without further soil degradation; (ii) a business as usual (BAU) scenario, with soil degradation continuing at current intensity (i.e., relative yield losses in the next 15 years would be the same as in the past 15 years); and (iii) a double-degradation (2SD) scenario. Combining these results with trends in cropland area, cropping intensity, population growth and food consumption habits, they calculated that, at the national level and in per capita terms, the relationship between food supply and demand will turn from an 18% surplus in 2005 to deficits of 3-5%, 14-18% and 22-32% by 2030-2050 under the zero-degradation, business as usual and double-degradation scenarios, respectively. Based on these results, they discuss possible technical counter-measures and policy interventions to avoid food insecurity.

According to Bindraban *et al.* (2012), this analysis provides an indication of the likely impact of degradation on productivity, yet the quantitative nature is arbitrary and extrapolations based on statistical likelihood are questionable, as they do not account for underlying processes which, in turn, hamper the identification of potential intervention measures.

More recently, an interesting example was provided by scenario studies conducted at the continental and subcontinental levels, in the framework of the CGIAR Research Program on Climate Change, Agriculture and Food Security (CCAFS) and in collaboration with the International Food Policy Research Institute and the International Institute for Applied Systems Analysis (IIASA, 2014). In these studies, comprehensive scenario narratives, and associated assumptions, are built into participation with regional stakeholders, considering aspects of plausible futures (in terms of factors such as global context, regional and local governance, GDP, technology impacts on yields, and environmental degradation and its impacts on yields); and linked to the global SSPs, for each region. The generated scenario assumptions were then used as input for the IMPACT, GLOBIOM and LANDSHIFT models for quantitative analysis. Available kilocalories per capita, from crops and livestock products, were used as a proxy for food security. These scenario results were adapted, reinterpreted and used to guide a number of strategic policy plans in the fields of agriculture, climate change and socioeconomic development. Subsequent studies focus on smaller regions, and build on and refine the work constructed in the regional analyses (e.g., van Soesbergen *et al.*, 2017; Vervoort *et al.*, 2013). Similarly, in these subsequent studies, the impacts of land degradation on yields were estimated as a lump factor, without considering the underlying processes.

7.2.4 Water

Key findings

- Managing land involves managing water. Although the science projecting future impacts of land degradation on freshwater is limited at global scales, land degradation is currently having a significant negative impact on freshwater and this impact is expected to intensify by 2050 (*established but incomplete*).

- Nearly half of the global population will live in water scarce areas in 2050, with the highest proportion in Asia. Agricultural water demand and increasing loads of sediment and pollutants, due to intensified agriculture, are among the primary drivers of water scarcity in this region (*well established*).

- Three policy solutions are: (i) protecting natural landscapes to reduce water stress and conflict; (ii) enhancing nutrient-use efficiency including higher nutrient-use efficiency in crop production to substantially reduce accumulation of contaminants; and (iii) implementing no-till or reduced tillage and other conservation measures – such as terraces, soil or stone bunds, or buffer strips along water bodies – to dramatically reduce soil erosion, protect water bodies and the water holding capacity of the land (*well established*).

- Even under restoration scenarios that address land use change or land degradation, regional population increases may result in increased water extractions and increased sewage emissions – leading to a net negative impact on water quality and quantity (*established but incomplete*).

Indicators: river discharge, run off, water stress, irrigated area, floods, droughts, water holding capacity, ground water depletion.

7.2.4.1 Global Scale: the drivers of future changes in hydrological cycles linked to land use

Hydrological cycles and precipitation patterns will be impacted by both anthropogenic and biophysical drivers. A rapidly growing population and increasing demands for food, water and energy are all expected to significantly increase pressure on lands (Conacher, 2009). The associated land-use change required to meet future demands is likely to have negative impacts on freshwater resources unless more efficient management schemes are adopted (e.g., Beddington, 2010). In particular, expansion of agriculture and increasing irrigation reduce water quantity and result in major declines in global freshwater biodiversity over the period 2010 to 2050; 80% of species composition lost in standing water bodies, 70% in running waters (Fekete *et al.*, 2010; Janse *et al.*, 2015). The future stresses on water quality and quantity, such as floods and droughts, in a global population of over 9 billion people will exacerbate competition for water, especially between urban, industrial and agricultural water use and environmental water needs (Alcamo *et al.*, 2007). Climate change-driven increases in the frequency and intensity of floods and precipitation will exacerbate both sheet and rill erosion (Nearing *et al.*, 2004). Equally, changes in land cover and irrigation affect water balances at different scales – from local to global – including non-point source pollution, water scarcity and increased floods and droughts (Eshleman, 2004; Haddeland *et al.*, 2007; Nearing *et al.*, 2004; Scanlon *et al.*, 2007).

Numerous scientific studies examine historical and current land degradation impacts on freshwater at local or regional scales. Global studies examining the future impacts of freshwater, however, tend to focus almost solely on climate change impacts and do not specifically examine the impacts of land degradation.

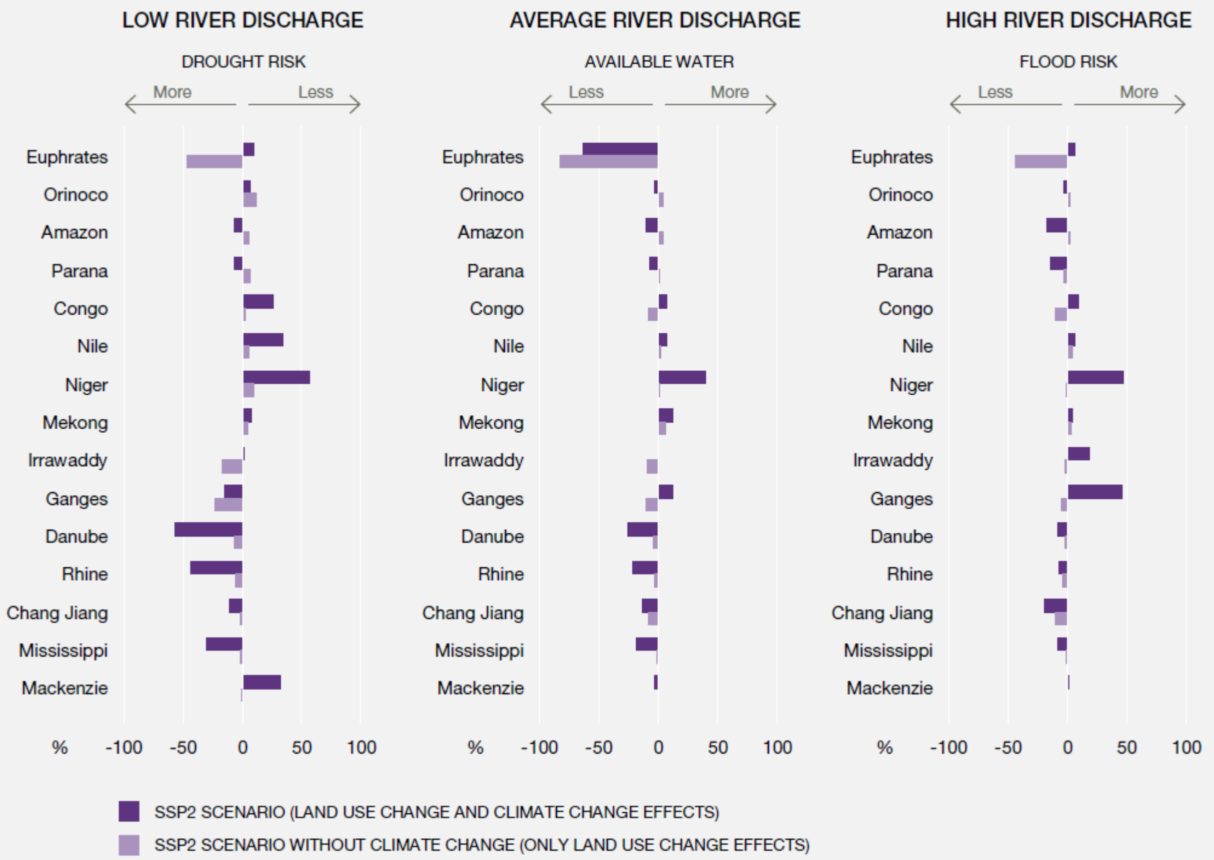
The relative lack of literature examining land-use change at a global scale, and its impact on water resources, may be due to regional variations and lack of comprehensive data to predict the multi-scale impacts.

The newly released Shared Socioeconomic Pathways (SSPs) – Representative Concentration Pathways (RCPs) – have been extended to include water futures under a range of socioeconomic and climate change scenarios (Burek *et al.*, 2016). The results from this global analysis indicate that 43% to 47% of the global population will live in water stressed regions, 91% to 96% of which will live in Asia. Demand for irrigated land to feed growing populations outpaces climate change-induced demand increases (warming and precipitation change) through 2050, with global irrigation requirements increasing by 32.6% and a projected 13.8% increase in irrigated area relative to 5.6% climate change-induced increase by 2050, under the SSP2 scenario (Burek *et al.*, 2016).

Changes in climate, land cover and soils alter the probability of floods and droughts. Figure 7.10 shows the change in discharge for major river basins between 2010-2050 from land cover change, water demand and climate change under the SSP2 scenario (Van der Esch *et al.*, 2017). Highest increase in runoff is projected in arid climate zones, where a little intensification in land use may cause a strong change in runoff. The effects on people are particularly amplified in drylands where populations are projected to increase by 43%, from 2.7 billion in 2010 to 4.0 billion in 2050. Regrettably the impact of changes in soil were not yet been calculated in this study.

With the loss of soil organic matter, the ability of soils to hold water declines. Water holding capacity is especially relevant for rain fed agricultural production in drylands, where rainfall can be erratic and the buffering function of soils to store water is used by plants to bridge longer dry spells. Land surface subsidence, sea-level rise, stream flow depletion, ecological damage, loss of topsoil and seawater intrusion all impact land and water due to groundwater depletion. Groundwater is used to supply half of the world's irrigation and two billion people rely on it as their main source of water. Globally groundwater has been depleted by approximately 4,500 km^3 from 1900-2008 with depletion rising to an average of 145 km^3 / year from 2000-2008 (Famiglietti, 2014; Wada *et al.*, 2010).

Figure 7 10 Change in discharge for major river basins, 2010-2050. Source: Van der Esch *et al.* (2017).



In terms of the water effects from land degradation, globally, floods and droughts are expected to increase in occurrence over much of the planet. Areas that are expected to be especially impacted include central and southern Africa, India, the Middle East, China and Southeast Asia and North and South America (Hirabayashi *et al.*, 2008). A warmer climate, with its increased climate variability along with land degradation, including reduced soil depth and de-vegetation, will increase the risk of both floods and droughts (IPCC, 2014b). In turn, increased floods and droughts will cause landslides, soil degradation (especially erosion), and changes in land use and land cover. In floodplain agriculture, flooding often refreshes mineral soils, but in upland agriculture flooding can significantly thin and degrade soil. When combined with poorly managed agriculture thin and degraded soils may, in turn, increase floods and droughts.

Loss of soil organic matter affects the water holding capacity of soils, reducing local water productivity. Van der Esch *et al.* (2017) estimates that by 2050, many large river basins which are expected to be impacted by higher participation levels will also have larger than expected increases in runoff, with changes in land cover and soil losses appearing to reduce the ability of ecosystems to buffer water flows. River basins in arid climate zones are expected to be most heavily affected.

Consistent with global scenarios, regional scenarios show that continued land-use change along with climate change could result in significant impacts on both water quality and quantity by 2050. Common indicators related to water quantity were typically measured as discharge (m^3/s) or runoff (mm/yr). In order to model hydrological dynamics, regional scenarios of water impacts are predominantly at a catchment scale (more than 90% of studies). To investigate regional-scale patterns of change, studies typically model impacts on water at a catchment scale and then aggregate results to broader spatial scales (Sattar *et al.*, 2014). The absolute value and spatial pattern of losses predicted under regional scenarios is similar to global scenarios - which is to be expected given the similarity in underlying data and the use of a common set of models such as global land-use change models (e.g., Fekete *et al.*, 2010; Qu & Kroeze, 2010; Sattar *et al.*, 2014).

Approximately 40% of the future population of Asia will live in severely water scarce areas (Wiberg *et al.*, 2017). Although the proportion of agricultural demand relative to industrial and domestic demand will fall at the global level, within Asia agricultural water demand under business as usual assumptions (SSP2) in 2050 will amount to 2044 km^3/year compared to 734 and 532 km^3/year for industrial and domestic water demand, respectively. Business as usual scenarios widely report decreased discharge by 2050, due to future irrigation and reserves – although some regions, such as Southeast Asia, will have a slight increase due to climate change under MA scenarios, primarily related to increased rainfall (Ercin & Hoekstra, 2012; Fekete *et al.*, 2010).

In addition to the potential water scarcity engendered by the over-extraction of water resources, other predicted changes in water quantity include: increases in the reported number of flood events; increases in the frequency and intensity of droughts; and increases in water temperature in rivers and lakes. The directions of these changes are, however, specific to individual regions. For example, river flows are predicted to decrease in Southern and Eastern Europe (particularly in the summer) and to increase in other regions (particularly in the winter). Moreover, there are also projected increases in the reported number of flood events and an increase in the frequency and intensity of droughts (particularly in southern Europe) (European Environment Agency, 2015). In baseline or business as usual scenarios, negative impacts on freshwater biodiversity, from hydrological disturbances and overexploitation, are projected to occur across all regions in 2050 (Janse *et al.*, 2015).

The expansion of agricultural land combined with increased irrigation will result in reduced runoff and losses in freshwater biodiversity (Fekete *et al.*, 2010; Janse *et al.*, 2015). Following these trends, major declines in freshwater biodiversity are expected in Africa, Asia and Latin America, while modest improvements in the USA and Europe are expected under baseline conditions (Janse *et al.*, 2015). Similarly, in regions with high population density and projected increases in irrigation (i.e., India, China, Southeast Asia and Japan), runoff alteration is primarily attributed to human disturbance rather than climate change (Fekete *et al.*, 2010).

7.2.4.2 Protection, restoration and prevention

Protection and restoration interventions include putting caps on water consumption by river basin, sustainable land management and water conservation such as terracing, increasing water-use efficiencies and better sharing of the limited freshwater resources (Ercin & Hoekstra, 2012; Mekonnen & Hoekstra, 2016). Restoration scenarios may result in improvements to water quality. However, even under restoration scenarios that address land-use change or land degradation, regional population increases may result in increased water extractions and increased sewage emissions, resulting in a net negative impact on water quality and quantity.

Major Water Policy Options

Policy and financing vehicles to address long term goals, programs that provide the necessary frameworks to invest in soils and land, and vegetation restoration are essential to sustain and increase water availability (also see Chapters 6 and 8). Ecosystem-based adaptation and restoration-targeted policies for soils are needed to improve soil health and reduce loss of natural land while at the same time promoting land reform. While land management is critical for water management, heavy industry such as energy production and utilities, also have a significant impact on land and water. A lack of environmental liability requirements surrounding the development and operation of these industries, for example, is a significant cause of soil and water contamination worldwide and appears to be increasing in the future with rapid industrial growth especially in developing countries (Su, 2014).

Comprehensive land-use planning can mitigate some effects of agricultural expansion and its impacts on water quality (Tong & Chen, 2002). This can be done through: planning the pattern and location of agricultural development to preserve biodiversity hotspots; minimizing fragmentation; maximizing the range of ecosystem types preserved; and preserving wetlands and riparian zones that protect surface waters from inputs of nutrients, pesticides, eroded soil and pathogens. Increasing crop yields, for example, through closing the yield gap can save significant water resources and help conserve ecosystems and remaining forest areas. Based on our historical understanding, we are able to recommend the following three policy options:

1. **Protect natural landscapes.** Minimizing large-scale clearing of vegetation or land conversion, such as conversion of natural land to urban land and agriculture, allows for the retention of soil organic matter and natural vegetation - which significantly improves the water holding capacity of soils and landscapes. Increased soil organic matter facilitates water infiltration in the root zone, helping to reduce soil evaporation and augmenting soil moisture storage capacities. When the water holding capacity of soils is improved, it helps to reduce land degradation and erosion and minimize the impact of drought. Evaporation and transpiration are affected by changes in the vegetative cover with a decrease in evapotranspiration leading to predicted increases in runoff (UNCCD, 2017).

2. **Enhance and protect water quality**. If water is significantly polluted, it becomes either unusable or expensive to treat. Enhancing nutrient-use efficiency, including in crop production, can substantially reduce accumulation of contaminants. Other strategies in this area include enhanced fertilizer management, with practices such as: deep placement of urea; increased use of precision agriculture methods, such as yield monitors, to apply fertilizers and pesticides where they are needed most or where they generate the highest yield impacts; and replacement of furrow irrigation with drip, which allows direct fertilizer application to the crops and their root systems (Zhang *et al.*, 2002).

3. **Embrace no-till or reduced tillage and other conservation measures.** Measures such as terraces, soil or stone bunds, or buffer strips along water bodies have been shown to dramatically reduce soil erosion and protect water bodies from the adverse effects of nitrogen and phosphorus runoff (Derpsch & Friedrich, 2010). Crop rotations with nitrogen-fixing (cover) crops are also an important conservation measure to reduce the need for fertilizers thereby increasing soil health and reducing fertilizer-related pollution into water bodies (see also Chapter 6).

7.2.5 Climate

Key findings

- Climate and land degradation are linked by multiple connections such as the greenhouse gas (GHG) emissions associated with land conversion, biofuel policies, the impact of climate change on ecosystems, erosion and crop yields. Large uncertainties exist with respect to most of these connections. Policies will need to take these linkages into account and monitor possible impacts (*well established*).

- Land-use policies for protection and restoration of ecosystems can form an important contribution to mitigating and adapting to climate change. Policies aimed at avoiding deforestation and protecting and restoring soils are particularly effective in reducing further carbon losses and thereby mitigating climate change (*well established*).

Indicators: Greenhouse gas (GHG) emissions, stored carbon, carbon payback time, radiative forcing, ^{o}C.

7.2.5.1 Baseline projections

The most recent set of scenarios are the Representative Concentration Pathways (RCPs) (van Vuuren *et al.*, 2011a) and Shared Socioeconomic Pathways (SSPs) (Riahi *et al.*, 2017), which can be used together to explore different futures with respect to environmental challenges (van Vuuren *et al.*, 2014) (see Box 7.4). A middle-of-the-road scenario (SSP2), with respect to population and income growth as well as emissions and land use could - without additional climate policies - lead to an increase in radiative forcing (i.e., the additional energy trapped by greenhouse gases) of 6-7 W/m^2 (900-1000 ppm CO_2e) compared to pre-industrial levels, corresponding to about 4^{o}C of warming (for a medium-climate sensitivity) (Riahi *et al.*, 2017). Accounting also for uncertainty in socioeconomic development and the climate system, typical values for the increase of global mean temperature in 2100 range from 3 to 7^{o}C (compared to pre-industrial) (IPCC, 2014b) in the absence of stringent climate policy. Climate policies could still avoid part of this increase. For instance, scenarios aiming at stabilizing radiative forcing at 2.6 W/m^2 (450 ppm CO_2e) would most likely lead to about 1.5-2^{o}C warming (IPCC, 2014b). The relationship between land degradation and climate policy, however, is complex. Several climate policies rely, for instance, on land-use changes which could cause ecosystem degradation as well.

Box 7.4 The RCP/SSP scenarios

Scenarios form an important component of climate research and assessment to explore long-term consequences of current trends and policies and to connect information of different research communities (mitigation, climate science and impacts) and across different scales (van Vuuren *et al.*, 2014). Climate impacts and mitigation challenges depend on: (i) the level of (desired) climate change and (ii) socioeconomic circumstances. The Representative Concentration Pathways (RCPs) scenarios (van Vuuren *et al.*, 2011a) have been developed to mostly explore different levels of climate change and have been analysed in terms of emission trajectories, costs, climate change and impacts. The Shared Socioeconomic Pathways (SSPs) have been developed more recently and describe how different socioeconomic development may result in very different environmental policy challenges for this world (Riahi *et al.*, 2017). The combination of RCPs and SSPs can also be used to look into different biodiversity and soil consequences.

Figure 7 11 Development of CO_2 emissions, forest area and radiative forcing of the SSP-RCP scenarios over time. Source: Riahi *et al.* (2017).

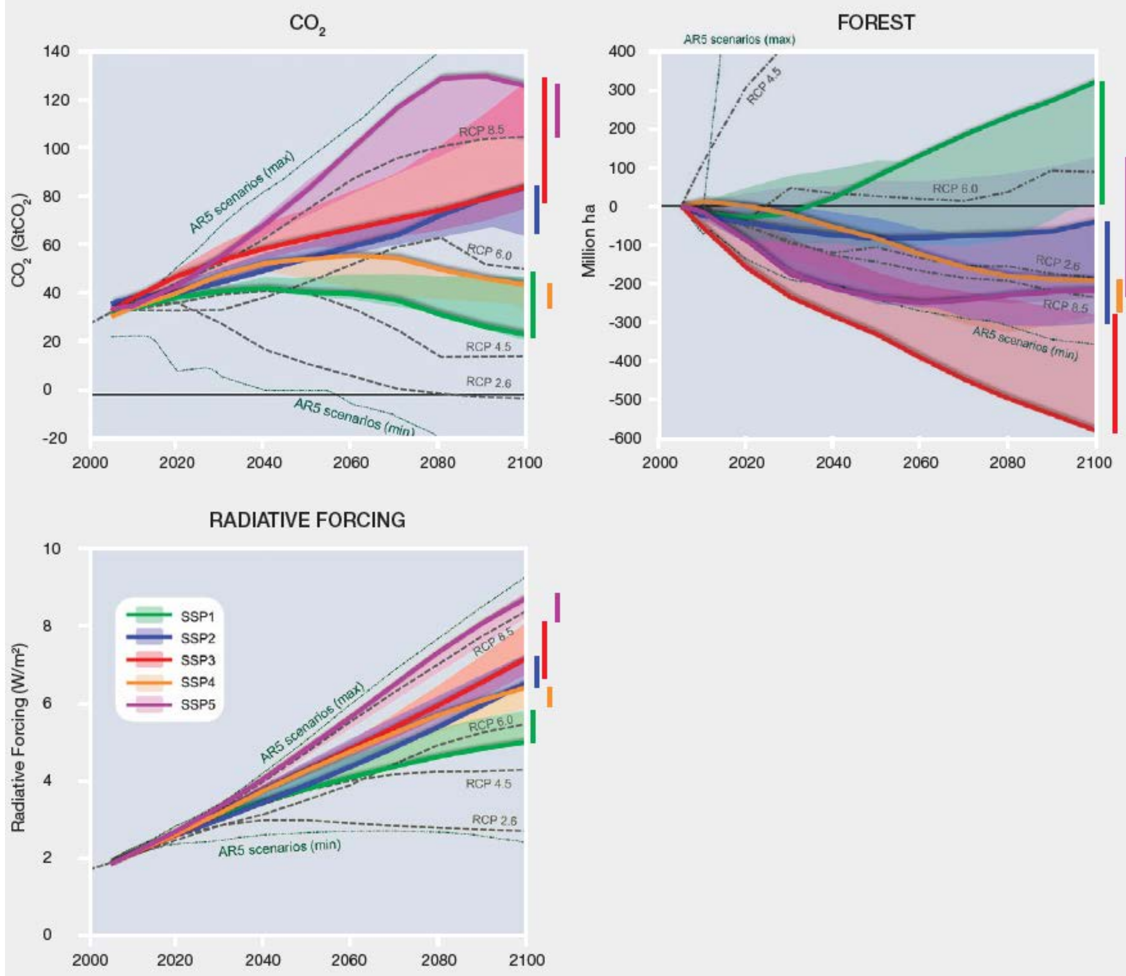
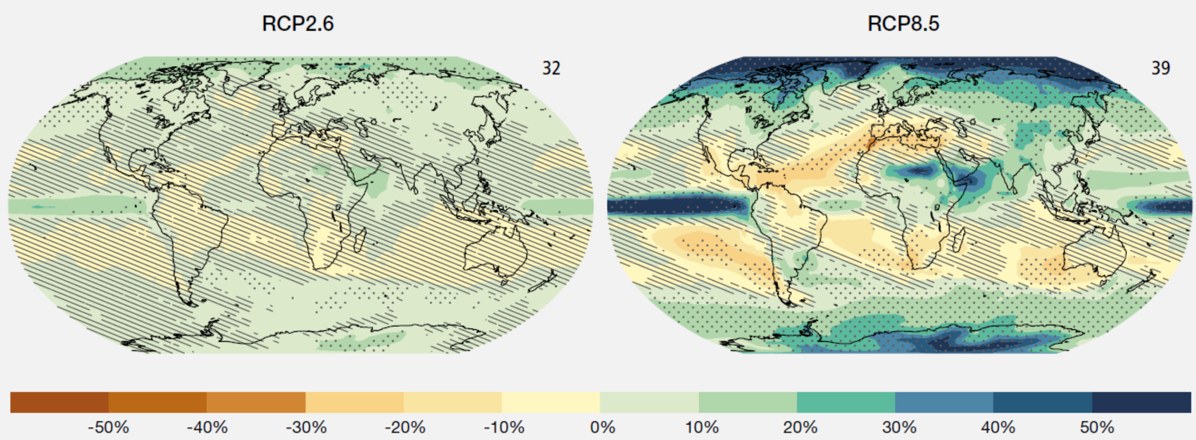


Figure 7 12 Maps of the CMIP5 multi-model mean results for the scenarios RCP2.6 and RCP8.5 in 2081-2100 of average percent change in annual mean precipitation.

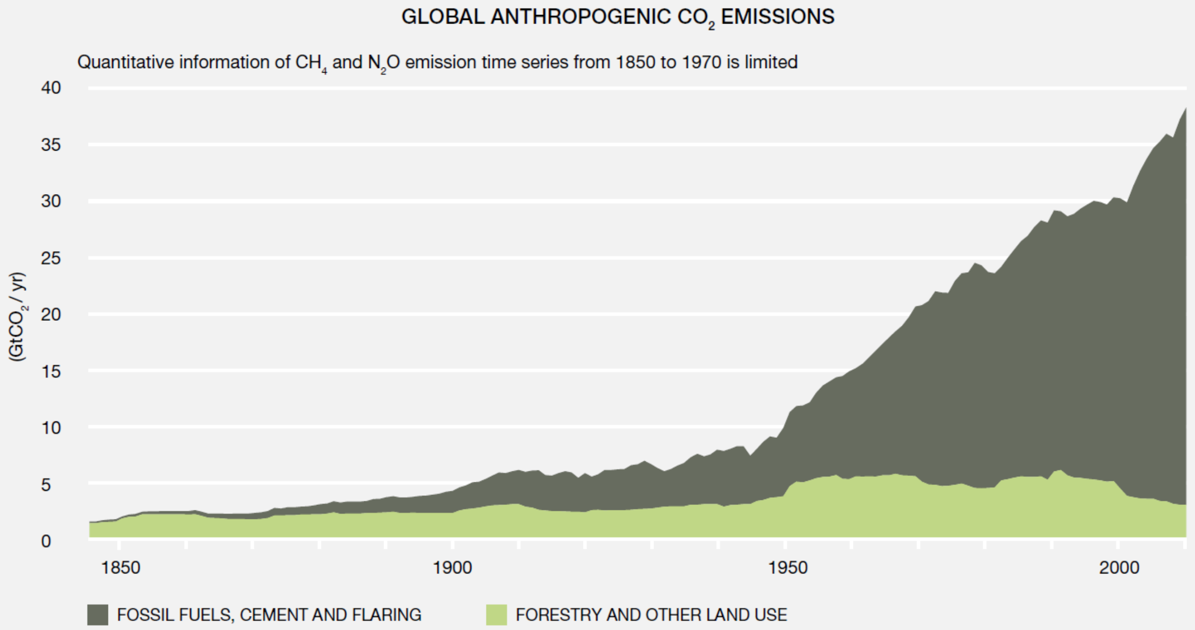
Hatching indicates regions where the multi-model mean is small compared to natural internal variability (i.e., less than one standard deviation of natural internal variability in 20-year means). Stippling indicates regions where the multi-model mean is large compared to natural internal variability (i.e., greater than two standard deviations of natural internal variability in 20-year means) and where at least 90% of models agree on the sign of change. Source: IPCC (2014b).



Land degradation as a cause of climate change

While fossil fuel combustion currently remains the most important contributor to climate change, land-use change and ecosystem degradation are not far behind (Figure 7.13). With regard to net contribution, however, the CO_2 uptake of natural vegetation also needs to be accounted for. The Global Carbon Project estimated that the global terrestrial vegetation (i.e., the net emissions resulting from land-use change and CO_2 uptake of natural vegetation) forms a net sink in the 2005-2015 period (Le Quéré *et al.*, 2015). The numbers are nevertheless beset with uncertainty originating from unknown carbon content of forests, peatlands and soil degradation. SSPs and other scenarios project a substantial and continued increase in CO_2 emissions from fossil fuel combustion in the foreseeable future. In contrast, emissions from land-use change are projected to decline in most scenarios consistent with declining deforestation rates (IPCC, 2014b; Popp *et al.*, 2017). Factors contributing to this include lower population growth (e.g. in the UN medium and SSP2 scenario) and further increases in yields per unit of land. Some scenarios even show abandonment of farmland areas leading to regrowth of vegetation (in SSP1), while in other scenarios further expansion of farmland is expected (e.g. SSP2 and SSP3). The reduction of deforestation rates in SSP1, in turn, is driven by an expected stabilization of the global population (and thus food demand) from 2050 onwards, combined with further yield improvements.

Figure 7 13 Contribution of fossil fuel and land-use change to global CO_2 emissions over time. Source: IPCC (2014b).



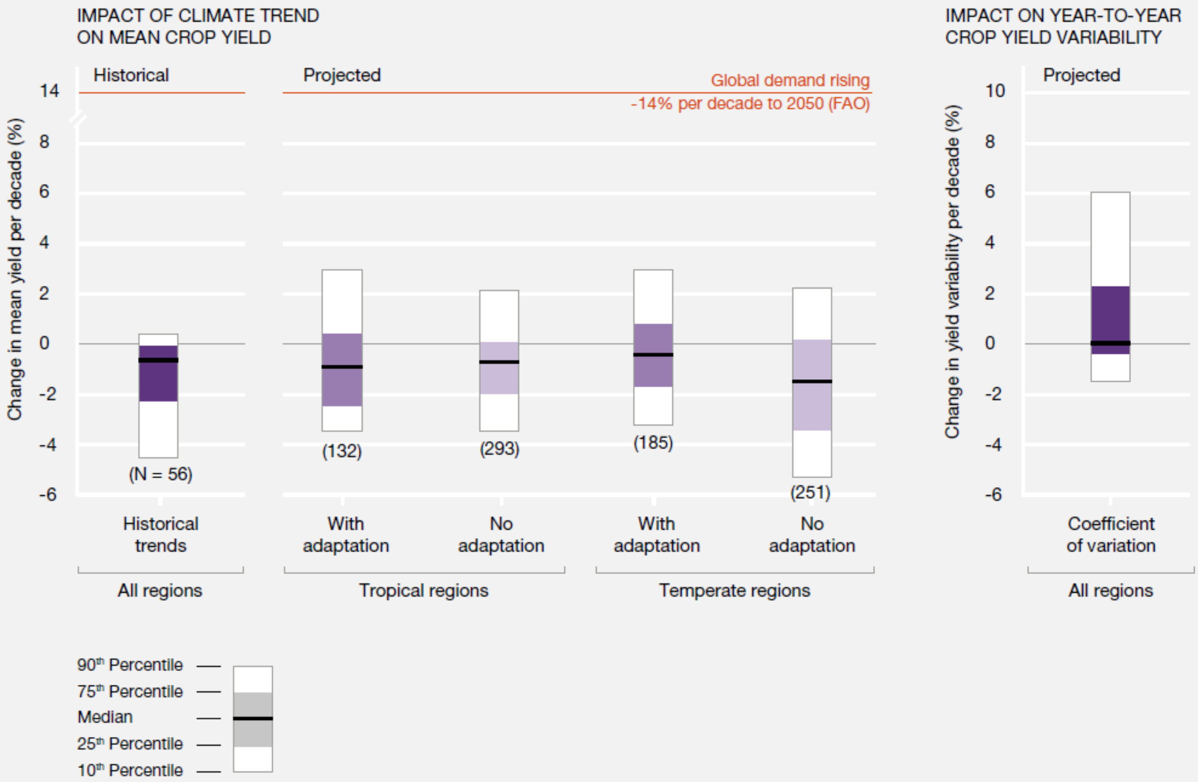
Currently, the largest CO_2 emissions from land-use change occur in South America, South East Asia and Africa (Haberl *et al.*, 2007; Hurtt *et al.*, 2011; Krausmann *et al.*, 2013; Popp *et al.*, 2017), which is also expected to continue into the future. In addition to deforestation, the conversion of peatland or high organic soils to cropland or pasture areas, also leads to large CO_2 emissions (WWF, 2015). For the Brazilian Amazon area, Oliveira *et al.* (2013) expect that, without new policies, there will be a severe decrease of carbon storage in above-ground biomass (to less than a third of the current amount) from logging and land conversion to cropland or pastures. Baseline scenarios for Southeast Asia also expect further losses of above-ground and soil carbon due to deforestation and land conversion at a regional scale, again with peatlands playing an important role (Koh & Ghazoul, 2010a; Mulia *et al.*, 2013; Van Noordwijk *et al.*, 2008). In other regions, there might be a small net uptake resulting from land-use change. For instance, in North America, Sleeter *et al.* (2012) predict a strong increase in agricultural land. In the case of Europe, most scenarios show a decline of agricultural land, leading to an increase of above-ground and soil carbon stocks (Lugato *et al.*, 2014; Rounsevell *et al.*, 2006). Such reforestation trends might also result from more deliberate policies to increase forest area. For example, studies on Europe, North America and parts of Asia show increases of forest and bioenergy plantation areas (Lee *et al.*, 2013; Rounsevell *et al.*, 2006; Sleeter *et al.*, 2012) – leading to enhanced carbon sequestration and increased above-ground and soil carbon stocks (Lee *et al.*, 2013; Lugato *et al.*, 2014).

The impact of climate change on land degradation

Climate change could have significant impacts on land degradation in different parts of the world. For instance, studies have emphasized the possible severe negative impacts of climate change on tropical ecosystems, dry areas in Africa, Mediterranean ecosystems and tundra – see for instance Thomas *et al.*, (2010). Globally, it is expected that CO_2 fertilisation leads to increases of carbon in natural ecosystems. The impact of climate, however, would most likely work in a different direction (Friedlingstein *et al.*, 2010). Gumpenberger (2010) showed, using LPJ DGVM vegetation model and 5 different climate projections, that even if deforestation stopped, climate change may lead to loss of carbon stocks in some tropical regions like Brazil and Southeast Asia. If drainage of peat soils is not reversed and restored, soil carbon losses continue (Dommain *et al.*, 2016). Land-use change may also lead to local climate alterations, provoking yield loss and thereby self-enhancing feedback loops of further land conversions (Oliveira *et al.*, 2013; Paeth *et al.*, 2009). Climate change could also indirectly lead to land degradation by impacting future food production (IPCC, 2014a) and water availability, thus leading to further land conversion to meet food security demands. Most studies expect that such impacts are modest in stringent mitigation scenarios (2°C world) if adaptation and CO_2 fertilisation effects are accounted for. However, without adequate climate policies, such impacts may lead to decreasing yields (in particular, as a result of precipitation and evaporation trends) and hence to a further expansion of agricultural land, which in turn, leads to more land-related carbon emissions.

Figure 7 14 Impact of climate and CO_2 changes on crop yield, including historical and projected impacts, mean and range of yields changes, for crops in temperate and tropical regions.

All impacts are expressed as average impact per decade (a 10% total impact from a 50-year period of climate change would be represented as 2% per decade). The underlying studies use very different methods and may include the impact of extreme events or only focus on changes in average climate. Source: IPCC (2014c).



7.2.5.2 Policy options for climate, prevention and restoration

Climate policy

Climate policy could also be an important factor for future trends in land use. Climate policy would be needed to prevent the negative impacts of climate change on biodiversity, but several climate policies can also have direct positive impacts on biodiversity. For instance, one option is to reduce deforestation rates and to increase forest restoration in lands that were formerly forested (e.g. Popp *et al.*, 2011; Reilly *et al.*, 2012; Wise *et al.*, 2009). Another option is the use of more bioenergy to replace fossil fuels (see Box 7.6). The literature on the impact of bioenergy use on land use is far from univocal: there are widely different results and large uncertainty ranges. In the energy system, bioenergy can be quite important to reduce emissions – as very few other options exist to reduce emission from various transport modes (such as air traffic). Bioenergy and Carbon Capture Storage (BECCS) could also be an attractive option to create negative emissions. The large-scale production of bioenergy, however, could require large amounts of land, leading to competition with other land uses and possible reduction of natural area through forest loss (Melillo *et al.*, 2009; Reilly *et al.*, 2012; Searchinger *et al.*, 2008; Wise *et al.*, 2009) (see also Section 7.2.6.1). On the other hand, bioenergy, can be produced with considerably less land; for instance, from agricultural and forest residues, or grown in specific areas with high yielding crops. In some cases, bioenergy could even lead to soil restoration. Such a diversity of results is also found at the regional scale. While some studies report the positive impacts of climate policy on deforestation rates in the Amazon region, Oliveira *et al.* (2013) expect deforestation and land conversion in Brazil to continue, despite the implementation of carbon protection policies. Kho & Ghazoul (2010b) and Fuller *et al.* (2011) showed that, for Indonesia, strict forest conservation may lead to severe carbon losses due to leakage effects and the expansion of palm oil plantations on other peat soils. Only under an explicit carbon protection scenario do carbon stocks grow. More careful strategic planning is needed to avoid potential trade-offs between objectives.

Economic valuation of carbon protection measures – such as changes in GDP per capita or mitigation costs – show that opportunity costs may vary widely across regions and are dependent on carbon pricing (Bonn *et al.*, 2014; Bryan *et al.*, 2014; Koh & Ghazoul, 2010b; Mulia *et al.*, 2013; Overmars *et al.*, 2014; Van Noordwijk *et al.*, 2008). Such regional studies also emphasize that the impacts of bioenergy on land use could be very different for different crops and in different regions (e.g., Albanito *et al.*, 2016). Searchinger *et al.* (2015) show that carbon payback time (i.e., the time it takes for CO_2 emissions from land conversion for bioenergy plantations to be compensated by GHG savings thanks to biofuel use and fossil fuel savings) may vary significantly and may take up to 50-100 years, as exemplified by African wet savannahs. In addition, trade-offs with biodiversity and food production need to be considered.

Policy options for protection and restoration

Most policies which are aimed at prevention of soil degradation and restoration of natural ecosystems will lead to an increase of natural storage of CO_2 and avoided carbon losses, thus contributing to climate change mitigation (Bonn *et al.*, 2016) (see also peatland case study Box 7.5). The IPCC estimated the potential of afforestation measures, for instance, to be 0-4 GtC/yr at prices of around 100$/ton CO_2. In addition, soil restoration would be an important measure, especially at higher carbon prices. Scenarios on the protection or restoration of carbon stocks through avoided wood harvest (omitting shifting cultivation) and on the conversion for bioenergy are mostly of the ex-ante type. Policies focusing on protection and restoration of natural ecosystems lead to a protection of carbon stocks and an increase of carbon sequestration. The greatest potentials for concomitant ecosystem protection and carbon sequestration are in Central and South America, Sub-Saharan Africa and Southeast Asia (Hurtt *et al.*, 2011; Overmars *et al.*, 2014; Reilly *et al.*, 2012) due to the extent of available land area for protection measures, the relative low price of land and given the size of current and ongoing carbon losses (e.g. in Southeast Asian peatlands). Reilly (2012) as well as Hurtt (2011) also identify Eastern Europe and Central Asia as important areas for carbon storage.

Box 7.5 Restoration of Lowland Peatland in England and Impacts on Agriculture and Environment.

Of the 325,000 ha of lowland peatlands in England, 240,000 ha (74%) are used for farming and food production. Concerns about the continued loss of peatland habitats, the degradation of agricultural peatlands and the associated release of soil carbon, have led to calls for large-scale restoration of peat-forming vegetation, to provide a range of environmental benefits such as nature conservation, water resource protection, carbon storage and recreation. Taking English peatlands out of agricultural production could, however, affect national food security.
Focusing on 66,000 ha of the “Target Areas” of Natural England’s aspirational “Wetland Vision”, the effect of alternative land-use scenarios on a range of outcomes were considered, namely: (i) agricultural output and food security; (ii) farm incomes and profitability; and (iii) environmental costs and benefits, with particular reference to carbon emissions and landscape benefits. Four target locations were considered and estimates of agricultural and environmental benefits and costs (£/ha/year) were derived for each of the following scenarios (assuming full operation):

- **Baseline Agricultural Production** - existing land use in 2010 generates relatively high agricultural profitability which is offset by high environmental costs associated with carbon loss, resulting in overall economic loss for the assumptions made.
- **Continued Agricultural Production** - results in the severe degradation of peat soils and a change to less intensive, less profitable land use, albeit with reduced ongoing environmental costs. In many areas peats will be “farmed-out” within 30 years.
- **Peatland Conservation** - involves extensive wetland grazing with potential to generate relatively high environmental benefits linked to landscape services and carbon storage. Low farm incomes could be supplemented by payments for ecosystem services.
- **Peatland Restoration** - assumes peat-forming conditions with permanently high-water levels. Livestock are retained for habitat management only. This scenario generates relatively high net benefits due to carbon sequestration and the “cultural” benefits of landscapes, wildlife and recreation.

Restoration of these 66,000 ha of peatland is unlikely to significantly impact national food supplies. However, restoring all 240,000 ha of agriculturally-managed peatlands – accounting for about 2% of total lowland agricultural land in England, over 3% of total value, and probably 5-8% of the area of some specialist crops – could affect national supply if relocation elsewhere in the UK was not possible.
There is a strong economic argument for actions to avoid or minimise the degradation of lowland peatlands under agricultural use. In some cases, taking peat soils out of intensive farming could result in overall economic gain, with losses in agricultural output offset by a wide range of environmental benefits, including carbon storage. Future food security could be enhanced by farming peatlands extensively so that they could be returned to more intensive agricultural production should the need arise. Land managers would need to be rewarded or compensated under new land management regimes.

Source: Morris *et al.* (2010)

Figure 7 15 **Pictures of the study area.**

On the left: onions, as example of the current arable land use in the intensively farmed areas. On the right, the cattle show adjacent degraded peatland recently restored to wet grassland under The Great Fen Project, a Wetland Restoration programme.
Photo credit: Joe Morris.

7.2.6 Bioenergy, timber, and fibre

Key findings

- The majority of climate change mitigation scenarios that limit temperature to 2° C above pre-industrial levels assume large amounts of bioenergy and bioenergy with carbon capture and storage (*well established*). The commercial, large-scale production of fibre and timber, for energy purposes, is expected to grow to approximately 1.5 million km^2 in 2050 and 4-6 million km^2 in 2100 under 2° C emissions pathways, assuming a "middle of the road" socioeconomic trajectory. There is a lack of consensus on the capacity for marginal and degraded lands to meet the biomass production levels proposed in climate change mitigation scenarios. No significant negative influence on timber production is expected from land degradation, although vastly expanded energy wood production for climate change mitigation purposes will likely exacerbate water scarcities and compete with food for land, potentially resulting in indirect land-use change and adverse impacts on food security.

- The majority of future forestry scenarios show that greater tracts of natural forests will come under management to meet energy and material demands (*established but incomplete*). They also indicate that 2 to 3 million km^2 of natural forests will be under various degrees of management to meet timber and fibre demand by 2050. This projected expansion will result in negative impacts on biodiversity and soil organic carbon, the latter being contingent on the type of conversion underway and the time horizon under consideration. Increased future fuelwood and industrial roundwood demand is projected to grow in most scenarios, with particularly large growth in scenarios with increased bioenergy demand. Growing demand under all scenarios will disproportionately impact forests in Asia, South America, Africa and to a lesser extent North America.

Key Indicators: stock or yield in billion m^3 fibre, volume in Gm^3, energy content in exajoules (EJ), land use in million km^2, Carbon storage in Gt C.

7.2.6.1 Bioenergy

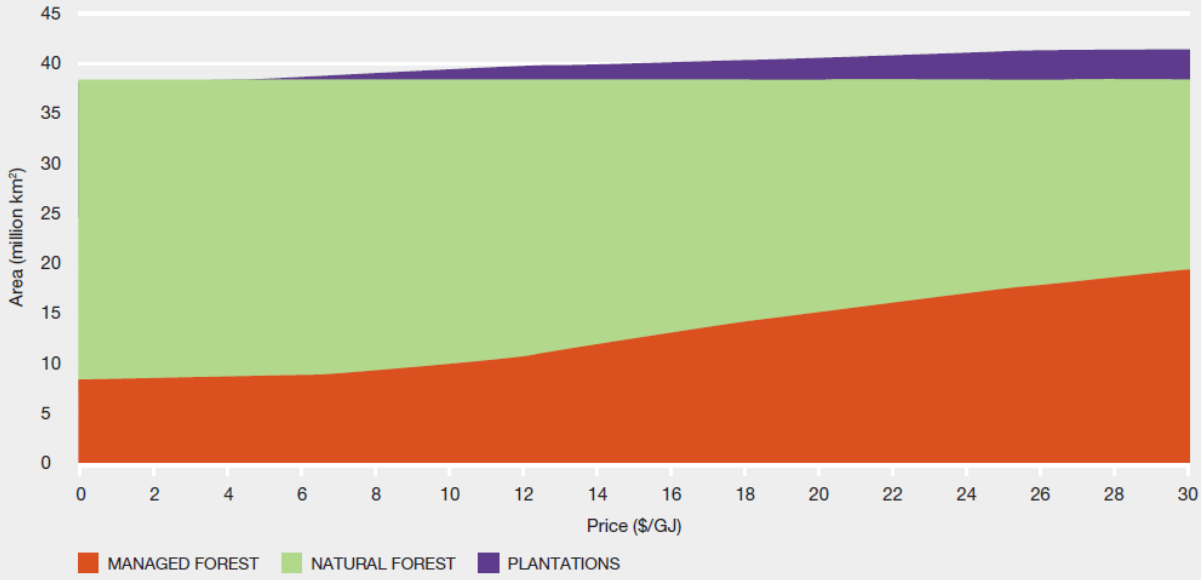
Bioenergy expansion is a critical component of most climate mitigation scenarios. In the absence of immediate and drastic global emissions reductions (an unlikely scenario) or a breakthrough in CO_2 removal technology, limiting warming to 2° C will not be possible without negative emissions from Bioenergy with Carbon Capture and Storage (BECCS) (van Vuuren *et al.*, 2011). There are significant concerns regarding the degree to which more ambitious bioenergy production scenarios contribute to indirect land-use change and the subsequent implications for food security in the less developed world (Lambin & Meyfroidt, 2011). The degree of land-use change emissions from bioenergy is highly contingent on the type of crop and the type of land under conversion: for instance, peatland has a higher emission factor (Valin *et al.*, 2015). Meeting climate change targets with BECCS would thus necessitate establishing bioenergy production on land not currently in use or no longer needed for food and feed production, as well as degraded lands (Dauber *et al.*, 2012; Nijsen *et al.*, 2012). There is however a lack of consensus on the potential for bioenergy on marginal lands (Cai *et al.*, 2011; Campbell *et al.*, 2008; Nijsen *et al.*, 2012), although there seems to be some potential for well-adapted agroforestry systems to improve soil conditions on land with little agricultural value (Gruenewald *et al.*, 2007).

Projections from the Living Forest Model of woody biomass for energy purposes in 2050 range from 6 billion m^3 in a “Do Nothing” scenario to 8 billion m^3 in a “Bioenergy Plus” scenario (WWF, 2012). Bioenergy scenario variants range from business as usual projections extrapolated from historical trends, scenarios assuming varying levels of bioenergy demand and bioenergy demand coupled with deforestation and biodiversity constraints (Kraxner *et al.*, 2013).

Figure 7 16 **Projected global timber and bioenergy use with corresponding land cover implications for 2050 under different energy wood-pricing scenarios.**

Table and figure adapted from Lauri *et al.* (2014). Note: Managed forest refers to forests where harvests take place while natural means no harvesting.

		Energy wood price	10 $/GJ	30 $/GJ
Volume (Gm³)	Energy Wood	Roundwood	1.99	8.62
		Residues	2.83	5.91
		Plantations/energy crops	2.85	8.42
		Total	7.67	22.95
	Fuelwood & Materials	Roundwood	2.79	2.61
		Residues	0.59	0.52
		Fuelwood	1.97	1.86
		Total	5.35	4.99



Timber and forests play a vital role in bioenergy scenarios as first-generation biofuels are phased out (with the exception of sugarcane) in favour of second generation biofuels made from lignocellulosic biomass. This nexus between agriculture, bioenergy and forestry is illustrative of the various cross-sectoral feedbacks with land degradation and restoration implications through water, fertilizer and land cover change (see Box 7.6). Assuming a shift from first to second generations biofuels around 2030-2040, Kraxner *et al.* (2013) find potential 2050 bioenergy production ranges from about 60 to 75 EJ/year. Previous studies have exposed a considerable disparity in estimates, due in large part to yield projections and subsequent land availability, with medium agreement among experts at 100-300 EJ/year in 2050 (Creutzig *et al.*, 2015).

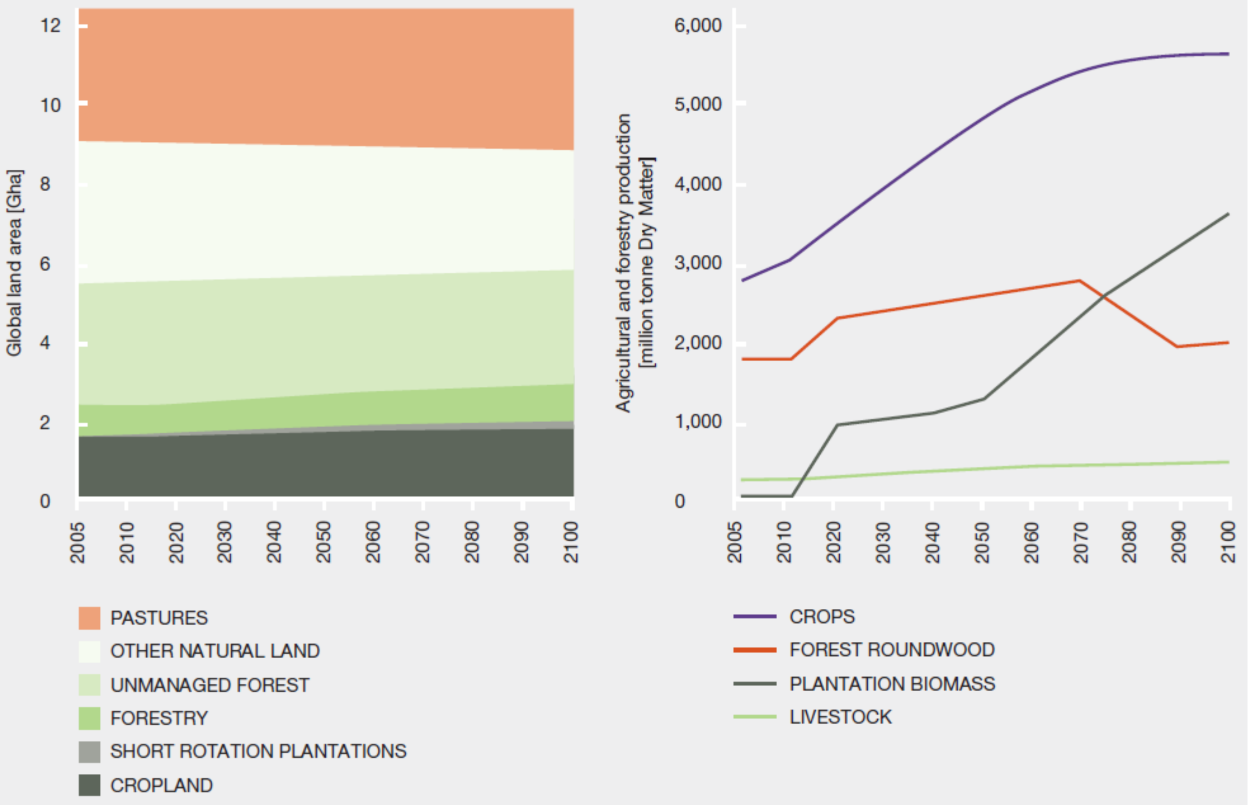
Box 7.6 Bioenergy with Carbon Capture and Storage: meeting the 2° C climate target

In the context of growing global food demand and consequent land-use pressures, negative emissions enabled by bioenergy coupled with carbon capture and storage (BECCS) is now an essential mitigation technology in a majority of 2° C scenarios employed by the IAM community (Anderson & Peters, 2016). This is despite significant biophysical, technological and societal barriers to employment at a level consistent for limiting the global mean temperature to 2°C above pre-industrial levels (Fuss *et al.*, 2014; Smith *et al.*, 2016).

Further, the massive increase in biomass production assumed under many 2.6 W/m^2 radiative forcing scenarios is very likely to intensify competition for land as energy crops would need to be planted on anywhere from 4.3 to 5.8 million km^2 - one-third the total arable land on the planet (Williamson, 2016). Meeting BECCS land requirements, while concurrently feeding a growing population, would drastically increase pressures on natural systems, particularly when plantations displace existing cropland due to water abstraction implications, indirect land-use change due to increased land competition and impact on commodity prices (Havlík *et al.*, 2011). This development comes at the cost of natural forests and pastures, with potentially negative consequences for biodiversity, water availability and ultimately long-term carbon sequestration (see Figure 7.17a-b below).

Figure 7 17 **Land-use development in the marker SSP2 scenario in line with a 2.6 W/m^2 climate target.**

Left panel: evolution of global land area over time. Right panel: agricultural and forestry production over time in units of million tonnes of dry matter. Source: Fricko *et al.* (2017).

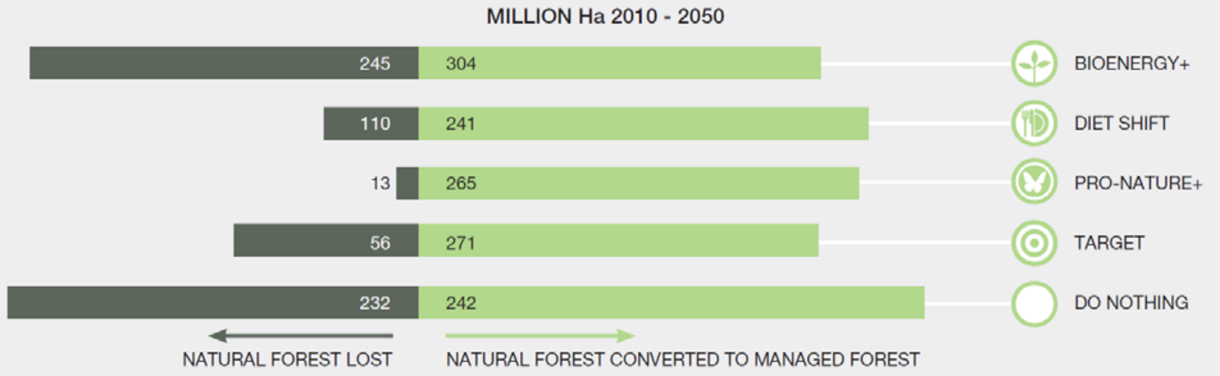


7.2.6.2 Timber and fibre

Tree harvests represent a substantial pathway of forest degradation. Over half of all forest degradation in developing countries is due to timber extraction and logging (Hosonuma *et al.*, 2012). Reducing forest degradation will require preventing “unnecessary” forest loss due to weak institutional capacity, poor governance, illegal deforestation and inefficient agricultural practices (WWF, 2011b).

Scenarios of natural forest loss are typically part of overall land-use change scenarios, as agricultural expansion is the major driver of deforestation. However, the second largest driver is forestry. Even under pessimistic scenarios, the forestry sector is projected to increase production, with 2030 estimates ranging from 3 to 3.5 million km^2 from a 2005 level of 2.6 million km^2 (Carle & Holmgren, 2008). Forest scenarios to 2050 project that under business as usual 2.3 million km^2 are lost to agriculture and another 2.4 million km^2 of currently intact forests will be converted to forests actively managed for timber products. Under a strong bioenergy scenario, another 650,000 km^2 of forest will be used for timber exploitation (WWF, 2011a). The impacts of large amounts of forests entering formal management are still poorly understood.

Figure 7 18 Projected land cover change under 5 scenarios showing the degree of natural forest conversion to managed forests and deforestation. Source: WWF (2011a).



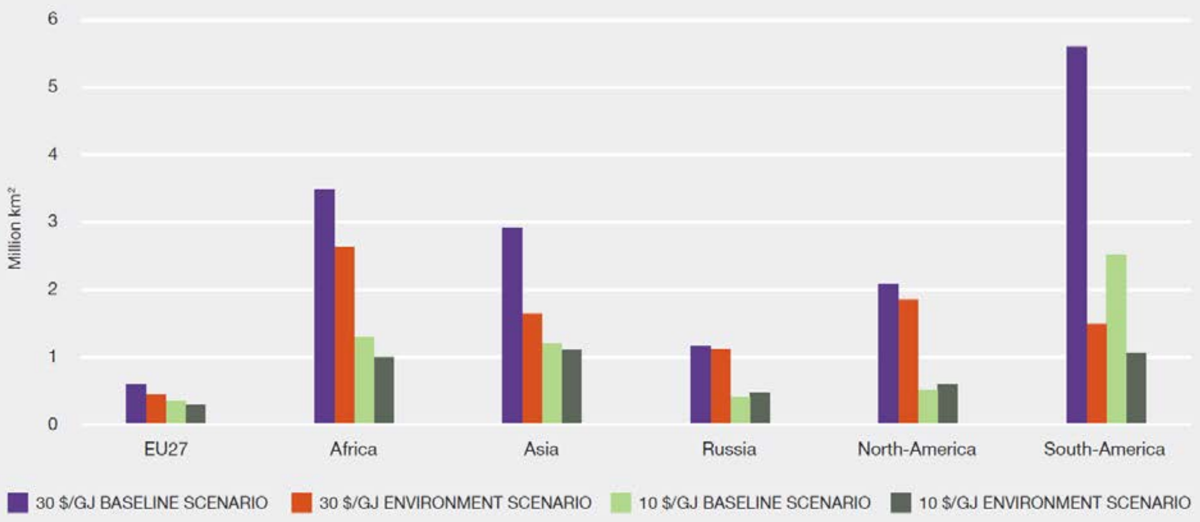
Projections of forest cover and forest management heavily depend on assumptions of land-use policies and forest policies in particular. The reduction of emissions from deforestation and forest degradation (REDD) has been formally recognized as a climate change mitigation option and REDD scenarios are abundant in the literature (also see Chapter 6). It has been found that the implementation of a REDD mechanism might have important co-benefits for biodiversity conservation. Strassburg *et al.* (2012) found that the continuation of historical deforestation rates is likely to result in large numbers of species extinctions, but that an adequately funded REDD program could substantially reduce these losses.

Consistent with global scenarios, local and regional scenarios found that continued land-use change, along with socio-political conditions such as armed conflict, ineffective policies and poor governance, resulted in significant loss of forest cover. The Amazon Basin, Congo Basin and Southeast Asia (see section below) represent nearly half of the global forest growing stock (FAO, 2011a). Scenarios included changes in land management practices – such as the period and intensity of wood harvests and the use of fire as a management tool – and socio-political interventions, such as changes to governance and law enforcement.

Lauri *et al.* (2014) conducted an extensive examination of timber for industrial and bioenergy purposes for 2050, noting that future timber demand is likely to be dominated by wood for energy purposes and the regional focus will likely shift into tropical regions. The WWF supports this analysis, estimating losses (by 2050) of 1.12, .82, and .38 million km^2 in Africa, Latin America and Asia, respectively, under a “do nothing” scenario (WWF, 2011a). Additional studies have confirmed that increased biofuel demand - consistent with climate change mitigation targets - will result in increased demand for fuelwood and eventually industrial roundwood (Buongiorno *et al.*, 2012), with particularly large increases in fuelwood demand in South America, Oceania and Asia.

Figure 7 19 **Projected cumulative unmanaged forest loss for 2050 under different energy wood pricing scenarios.**

Environment scenario sets aside primary forests for protection. Source: Figure adapted from Lauri *et al.* (2014).



Amazon Basin

In the Brazilian Amazon, total tree density can be recuperated following moderate intensity logging after a period of 6 years, while higher intensity logging results in greater mortality in burns due to increased canopy openings and fire susceptible woody mass (Gerwing, 2002). The use of fire as a land management tool is also likely to be more prevalent when there is a high chance that stakeholder investments will be lost to fire, resulting in a tipping point between swidden agriculture and more sustainable land-use practices (Nepstad *et al.*, 2008). Over half of the closed-canopy forests of the Amazon Basin could be severely degraded by 2050 under current trends, largely due to the expansion of agriculture (Soares-Filho *et al.*, 2006). The Amazon Basin is the most studied region, with most analyses focusing on the prediction of spatial patterns of deforestation (e.g., Soares-Filho *et al.*, 2002), while essentially only 1-2 models predict the total amount of deforestation given different policy assumptions (Aguiar *et al.*, 2016).

Congo Basin

Of the world's largest rainforest basins, the Congo Basin has experienced the least amount of deforestation (FAO, 2011a). Whereas logging activities in the Amazon Basin are frequently a precursor to land-use change and the expansion of agriculture, logging within the Congo Basin is highly selective and does not usually result in changes in land use (Megevand *et al.*, 2013). Deforestation in the Congo Basin has been modelled using a tailored version of the GLOBIOM model, pointing to an average deforested area of 4,000 km^2 per year from 2020-2030, under baseline projections, and 13,000 km^2 per year under transportation infrastructure improvement scenarios, which allow access to denser yet unexploited forests, as well as facilitating the development and expansion of agricultural markets (Mosnier *et al.*, 2014).

Southeast Asia

Relative to the Amazon and Congo Basins, population densities are relatively high in Southeast Asia, with higher proportions of the region's forests in production and under a management plan (FAO, 2011a). Combined with high levels of corruption and low per capita GDP, forests in Southeast Asia have experienced far greater degradation than other tropical forests, and three quarters of original forests could potentially be lost by 2100 (Sodhi *et al.*, 2004). With half of Asian tropical countries having experienced deforestation levels at about 70%, there is some doubt whether a forest transition to more sustainable management and conservation will occur before near total forest depletion (Laurance, 2007). The primary driver of deforestation in this region is the conversion of natural forests into commercial and subsistence crops particularly within Indonesia where palm oil production is expected to double from 2010-2020 (Kissinger *et al.*, 2012; Koh & Ghazoul, 2010a). There are nonetheless ample opportunities for restoration within Southeast Asia. For instance, according to the FAO, an estimated 400,000 km^2 of low productivity grasslands could be easily restored (FAO, 2011a).

7.2.6.3 Restoration

Afforestation and Forest Management

There are ample opportunities for mosaic and wide-scale restoration of degraded forest landscapes. Ecological restoration entails an effort to restore native ecological system with regeneration or planting of native species. Restoration can also involve the establishment of plantation forests for harvest as well as agroforestry or mosaic forests. WRI finds that over 20 million km^2 of degraded forests would be suitable for restoration worldwide, mostly in tropical and temperate forests (Laestadius *et al.*, 2011). The Bonn Challenge sets a global restoration target of 3.5 million km^2, by 2030, and is supported by Initiative 20x20 with a target of 200,000 km^2 in Latin America and the Caribbean, and AFR100 with a target of 1 million km^2 in Africa. A set of REDD scenarios exploring pathways to zero net deforestation and degradation (ZNDD) show that gross deforestation rates could be reduced to a few million of hectares per annum and could be compensated by active afforestation and forest restoration, elsewhere. 2030 ZNDD targets have been adopted by many private and public-sector actors aiming at sustainable supply chains for forest commodities (e.g. EU, CGF).

Afforestation of degraded land with plantation forests has been put forth as a cost-effective manner of restoring biological productivity, as well as meeting biomass feedstock demand without increasing competition for land. The concept of degraded land may however be misleading, as the majority of such land is currently fulfilling either a societal, climate or biodiversity function (Searchinger *et al.*, 2015). Further, afforestation of grassy biomes can have significant negative effects on ecosystem services (Veldman *et al.*, 2015). Restoration of sodic land has been accomplished through reforestation, with restorative effects including greater water holding capacity and increase in soil organic carbon (Tripathi & Singh, 2005). In Argentina, soil degradation has been found to affect seed viability – and consequently reforestation efforts – through potential biophysical as well as genetic impacts on seeds produced in degraded woodlands (Renison *et al.*, 2005; Renison *et al.*, 2004). The impact of afforestation on water yield and precipitation is heavily dependent on tree type and the scale of afforestation, among other factors (van Dijk & Keenan, 2007). We are not currently aware of any studies examining the impact of land degradation on timber or bioenergy production outside of the potentially ameliorating effect of establishing agroforestry systems on marginal land.

Both selective logging and clear-cutting have impacts on forest health, precipitating biodiversity loss and negative carbon sequestration implications – although it is more difficult to detect selective logging, particularly under lower intensities (Asner *et al.*, 2005). However, green-tree retention cutting has been shown to better preserve biodiversity with the selection of retention tree species, tree density and spatial arrangement as the most important factors (Rosenvald & Lõhmus, 2008). Selective logging intensity is highly correlated with forest degradation, even where deforestation does not occur (Mon *et al.*, 2012). At higher intensities, selective logging can lead to considerable forest fragmentation, canopy openings and edge effects (Broadbent *et al.*, 2008). Further, soil compaction by heavy machinery is pervasive and long-lasting, with a number of adverse impacts (Batey, 2009). In some cases, government policies encouraging the establishment of plantations are not only unnecessary for plantation development, they are often exploited by developers to gain access to extant timber (Kartodihardjo & Supriono, 2000). Within boreal forests, it has been found however that intensively managed forests frequently lack the deadwood necessary to support a variety of species essential to nutrient cycling (Spence, 2001). There are only a few specialised deforestation scenarios which project both "necessary" and "unnecessary" deforestation (WWF, 2011a). These unnecessarily converted forests are squandered because of land-use inefficiencies due to social and political barriers, which include lack of knowledge, poor governance, conflict, perverse incentives, shortage of capital and poverty (WWF, 2011b). Projections of improved governance suggest that unnecessary deforestation can be reduced to about one-fifth, by 2030, following the example of Brazil's forest law enforcement policy.

Forest Governance

Restoration of landscapes by indigenous communities has been taking place for hundreds and thousands of years (Bhakta *et al.*, 2016). In the last few decades, there has been increasing international attention for landscape restoration, heavily driven by non-state actors such as the Landscapes for People, Food and Nature Initiative (LPFN), the Global Partnership on Forest and Landscape Restoration (GPFLR), and the Global Restoration Initiative. The Bonn Challenge was launched in 2011 by the GPFLR, an International Cooperative Initiative (ICI), with the aim of realizing existing international climate change, biodiversity and land degradation commitments through restoration. It is a global effort to restore 150 million hectares of the world's deforested and degraded land by 2020 (later extended to 350 million hectares by 2030).

International targets and objectives (e.g. REDD+, Aichi, SDGs), as well as country level commitments and the Bonn Challenge pledges, show evidence of the increased political will for restoration. However, much uncertainty remains on the extent to which these commitments will be implemented and what the actual impact of these efforts will be (Wentink, 2015).

The UN Declaration on the Rights of Indigenous People (UNDRIP) is an example of a rights-based approach for involving the usually-neglected stakeholders (indigenous people) in various matters and specifically in the conservation and rehabilitation of degraded resources (Cittadino, 2012; Wright *et al.*, 2014).

International targets, the Bonn Challenge pledges and national plans, however, offer no guaranteed implementation of forest and landscape restoration programmes, due to for instance: weak governance systems; poor downscaling or upscaling of initiatives; limitations in financial, legal or social capital. Enabling conditions crucial for more successful implementation of such programmes can be found from case studies in Australia, Brazil, China, Costa Rica, Ethiopia, Indonesia, the Sahel, the US and Vietnam (Bennett *et al.*, 2014; Brancalion *et al.*, 2014; Brown *et al.*, 2014; Buckingham & Laestadius, 2014; Burger, 2002; Calvo-Alvarado *et al.*, 2009; Cao *et al.*, 2009; Chokkalingam *et al.*, 2001; De Jong, 2010; Hartshorn *et al.*, 2005; Reuben & Buckingham, 2015; Robins, 2004; UNCCD, 2015). From looking at these cases and some additional studies, the following preconditions appear to be prominent:

i. **Political momentum as an enabling condition for landscape restoration.** Framing a common agenda is crucial in getting actors involved. Current political momentum for restoration is partly linked to domestic issues and the added value of restoration for meeting international targets. A leading role for local institutions is imperative for local actors to take ownership of and see value in restoration efforts. Informal, flexible systems with low barriers and participation costs help bring together such stakeholders (Wentink, 2015).

ii. **Safeguard restoration quality**. Landscape restoration strategies should take into account the local, natural ecosystem as well as set explicit spatial and temporal goals (Manning & Lindenmayer, 2009).

iii. **Trade-offs are acknowledged and addressed.** Restoration efforts often involve a trade-off between goals and ecosystem services. It is key to acknowledge that such trade-offs exist and consider these early in the design and implementation process (Caspari *et al.*, 2014).

iv. **Stakeholder involvement on different levels.** Landscape and forest restoration efforts have been proven more successful when multiple stakeholders became active participants and rural development objectives were incorporated in program design. For instance, land users with leadership skills and knowledge of climate change and degradation issues can trigger the involvement of their peers in restoration activities (Curran *et al.*, 2012; De Jong, 2010).

v. **Multi-sector involvement.** The 2008 economic crisis increased risk regulations for private financing and dried up much public funding. In addition, growing public awareness for the environment made companies worry about their reputations. As a result, restoration has increasingly become a business practice. Public-private partnerships (PPP) offer a strategy to include multiple actors, receive funding from multiple sectors, and aid in knowledge sharing.

vi. **Supporting regulations and legislation.** Legislation can support intrinsically motivated actors if it fits current knowledge and practices. This works if restoration policies and legislation do not conflict with other policies, or they will undermine each other (perverse incentives).

vii. **Financial incentives**. Financial incentives are necessary to cover investment, maintenance, monitoring and opportunity costs, international funding, public and private sector investments are critical. However, investment without guarantee of project longevity and returns is risky. Local business cases help to decrease transaction costs and risks, while improving the likelihood of returns (Sewell *et al.*, 2016).

viii. **Available and accessible information.** Systems to disseminate information on monitoring and implementation provide a way to share learnings and enhance political momentum.

7.3 Insights from integrated scenario analysis, analysis of the use and effectiveness of scenarios and current gaps

This section provides insights from integrated scenarios drawing on the assessments of the individual themes covered in Section 7.2 and on selected methodological, practical and decision support issues.

7.3.1 Exploratory scenarios

As this chapter has shown, scenarios have systematically been used to explore plausible futures and the effect of a diverse range of drivers on environmental and ecosystem change with consequences for people and the natural world. Typically, changes in land use and land management act as major pathways for environmental change with implications for long-term sustainability and well-being. Scenarios have been constructed to reflect different social and economic motivations, scales of decision-making and degrees of interconnectedness, institutional frameworks, technological options and a range of responses evident in policy and behaviour. Broadly, there is consensus about the identity if not the magnitude of: key indirect and direct drivers; the effects of drivers on the state of natural resources and living systems; and the potential impacts on the wellbeing and prosperity of people. Whereas scenario analysis at the global scale has mainly considered (components of) land degradation as a product of land-use change, scenario analysis at the regional scale has been more closely attuned to defining plausible futures for land use under different context-specific policies and response regimes, including target-setting for sustainability.

Operating at the global scale “The Limits to Growth” (Meadows *et al.*, 1972) was an early example of exploratory scenario analysis of environmental change. Assessing the effect of unconstrained population growth and consumption habits on the depletion of natural resources, it reached a pessimistic forecast of environmental degradation, economic collapse and social disruption unless preventative actions were taken to ease pressures on natural resources. Twenty years later, the Millennium Ecosystem Assessment (MA, 2005a) revisited the link between people and natural resources, using four scenarios that vary in terms of scale of decision-making and the relative importance attached to economic, social and environmental imperatives. Although different scenarios generate different development outcomes measured in terms of quantitative indicators of sustainability, none manage to meet all conditions for sustainability. In particular, all scenarios lead to continuing biodiversity loss, mainly associated with the conversion of land to agriculture, forestry, bioenergy and urbanisation - generating further uncertainties about the extent of future climate change. Moreover, due to data, knowledge and model limitations, these scenarios fail to adequately incorporate key issues such as changes in soils characteristics, productivity, biomass loss, many types of pollutants, water holding capacity, floods and droughts, land abandonment, and their interactions with socioeconomic factors.

Most exploratory scenarios confirm the pivotal role of agriculture in the relationship between people and the natural environment (FAO, 2011b; IAASTD, 2009; OECD, 2015). According to IAASTD (2009), for example, human populations are expected to increase to between 8 and 11 billion people by 2050, possibly requiring an expansion of the global agricultural crop and grassland areas by 50%, including substantial increases in crop irrigation and grassland stocking rates. Long term food prices are expected to rise, driven either by increased consumer demand, resource constraints or measures to protect natural resources. In some scenarios, environmental pressures are further increased by the effects of meat-based diets and food waste, alongside competing demands for land. In others, changes in population growth, diets and consumption patterns enable a move to less intensive farming systems – although this may require extending the area under cultivation.

Scenarios have unanimously identified future challenges associated with increased environmental burdens generated by the intensification and/or expansion of agriculture to meet increased demand for food and fibre from a growing, wealthier human population. Business as usual scenarios show commercial agriculture being increasingly dependent on relatively high-cost technological solutions that partly substitute for degraded ecosystem functions, to support crop and livestock production, including irrigation, artificial fertilisation, disease control, pollination, water holding capacity and genetics. Simultaneously, scenarios reveal how pressures on natural resources increase the vulnerability of those rural communities, dependent on traditional agriculture for their livelihoods.

Most exploratory global scenarios of land use identify the need to improve the sustainable productivity of farming while simultaneously protecting biodiversity and ecosystem services. They also identify the need to harmonise policies that interact with land use, such as climate change, food security, biodiversity and the rural economy, and to promote modern and traditional technologies to enhance agricultural output as well mitigate potential risks (IAASTD, 2009). In this respect, alternative scenarios are devised to explore how different drivers and responses, including measures to alleviate environmental pressure, result in different outcomes. Examples include the Millennium Assessment's heavily engineered Techno Garden Scenario that modifies rather than eradicates the processes of degradation, mainly by substituting natural capital with man-made capital (MA, 2005a). The concept of 'sustainable intensive agriculture' (Foresight, 2011; The Royal Society, 2009) – whereby advanced technologies facilitate ecologically benign yet highly productive farming – has emerged as a plausible scenario for feeding a growing global population within finite environmental limits (Rockström *et al.*, 2013; Roehrl, 2012; Smith *et al.*, 2010).

A recent integrated scenario analysis for UNCCD's first Global Land Outlook (UNCCD, 2017) assessed the impact of projected changes in soil properties, including water holding capacity, food production and cropland expansion, water discharge of major rivers, climate mitigation and biodiversity (see Section 7.1.5.1 on SSP scenarios). The SSP2 productivity-decline scenario showed that, if current trends continue, land-based carbon emissions will generate about 80 Gt Carbon over the 2010-2050 period, equivalent to about 8 years of current fossil fuel emissions. Of these 80 Gt C, about 16 Gt C originates from land conversion due to agricultural expansion, 11 Gt C from detrimental land management, 9 Gt C from drainage and burning of peatlands, and around 40 Gt C from vegetation loss. The analysis shows that changes in land condition and climate have major implications including serious hydrological consequences in major river-basins – generally amplifying variability in water discharge and increasing the probability of droughts and floods. Furthermore, projected productivity loss in croplands is equivalent to a 5% additional expansion of the global cropland area by 2050. Biodiversity loss is projected to continue by 4 to 12% in percent points of mean species abundance, depending on the scenario, and will continue after 2050. From a climate mitigation perspective, preventing land-based carbon emissions would significantly contribute to achieving the global carbon budget of 170-320 Gt C (that is the amount of carbon that can be emitted without jeopardizing the 2°C target), especially in combination with the restoration of global soil organic carbon levels. This carbon storage potential is mostly situated in agricultural land (around 80 Gt C), requiring the development of production systems that combine high yields with close-to-natural soil organic carbon levels.

While the above estimates from Global Land Outlook apply at the global scale, the assessment also identified considerable differences in the components of land degradation between the world's regions. Taking a regional approach in a wider socioeconomic and environmental perspective, the assessment concludes that three regions – sub-Saharan Africa, the Middle East and North Africa, and South Asia – will experience the greatest pressure on land resources, particularly associated with population growth and corresponding increase in total consumption. Here, the most productive agricultural lands are already in use and expansion will increasingly take place on marginal lands with lower and declining yields, thus requiring yet more land. Several regions have limited remaining stocks of uncultivated land suitable for agriculture. What is left is of low agricultural quality. Furthermore, these regions are challenged by relative low GDP/capita, low food security, high under nourishment, low and in some cases declining crop yields, growing water stress, high exposure to climate change and high biodiversity losses (Van der Esch *et al.*, 2017). It is this combination of socioeconomic and ecological challenges that is of great future concern.

Operating at the regional scale, and more attuned to the needs of decision-makers, a number of recent scenario exercises and their supporting programmes explicitly consider aspects of land degradation, biodiversity loss (e.g., Sukhdev, 2008; UKNEA, 2010) and climate change (Audsley *et al.*, 2015; Dubrovsky *et al.*, 2015). These applications, designed to inform management actions and policies at the local-, catchment-, national- or regional-scale, combine qualitative and quantitative methods. They usually engage key stakeholders to help formulate plausible scenarios (Malinga *et al.*, 2013), using locally relevant indicators (Jackson *et al.*, 2014; Weber *et al.*, 2012). Scenarios here have considered the implications of land degradation through the effects on soil properties (and associated degradation processes), water stress, declining agricultural yields, reduced air quality, increased GHG emissions and biodiversity loss, developing suitable metrics to measure degradation processes and outcomes. They can also explicitly consider the distribution of impacts amongst different social groups, such as subsistence and commercial farmers (Adams *et al.*, 2014; Stoeckl *et al.*, 2013).

Scenario analysis at the local scale has been enhanced by improvements in scientific understanding of ecosystem processes and interactions, as well a better appreciation of the importance of traditional knowledge. Furthermore, an ecosystem services approach, supported by new metrics to value diverse flows of services, can provide more complete framework for scenario analysis that has greater resonance with decision-makers (UKNEA, 2010). However, compared with global scenarios, local scenarios often fail to consider the total consequences of changes at the local scale, such as when measures to protect biodiversity that reduce local agricultural production lead to the virtual 'export' of land degradation by sourcing agricultural products from elsewhere.

7.3.2 Visionary and target-setting scenarios

Until recently, much scenario analysis has adopted a positivist (the future 'is') viewpoint, exploring what is plausible and mainly based on an extrapolation of observed trends and divergence from a conventional, business as usual pathway. Although scenarios have included futures which are "more sustainable", they have been characterised by continued failure to achieve sustainable outcomes in all aspects and, in the case of global scenarios, for all regions. This has prompted calls for a paradigm shift in decision-making to firmly commit not only to the principles of sustainability, but also to its achievement. For this reason, recent scenario work, mainly promoted by conservation and development agencies, has adopted a more normative visionary approach (the future 'should be').

Working with stakeholders, visionary scenarios include statements of an intent driven by a shared commitment to the setting and achievement of sustainability targets (Joshi *et al.*, 2015; Rockström *et al.*, 2013; UNEP, 2012), defined in terms of key indicators of human well-being and prosperity (in its widest sense), and the protection and enhancement of the natural world (UN, 2015). Supported by a growing understanding of the links between natural capital and human well-being, these visions incorporate social and economic systems of governance and behaviour that operate within environmental limits, where the benefits of ecosystem services are duly recognized and built into decision-making.

Visions of sustainable futures have been developed at the global, regional, local and industrial sector scale. Sustainable intensive agriculture, whether based on organic farming or sophisticated "precision technologies", is often part of this visionary pathway. Furthermore, the visionary approach demands much greater integration across policy domains, strengthening the connections between food security, energy production, protection of biodiversity and natural resources, and climate change mitigation and adaptation – all of which have implications for the control of land degradation.

Scenarios have been used to examine outcomes of the 2010 Targets of the Convention on Biological Diversity (CBD) and of the Aichi Biodiversity Targets (CBD, 2014). The CBD has analysed the relationship between biodiversity actions and broader challenges facing human societies by comparing business as usual with plausible scenarios for simultaneously meeting biodiversity, climate and poverty reduction objectives. Here scenarios explore the potential synergies and trade-offs between different development goals, such as food security, bioenergy production and conserving biodiversity. These scenarios consider trends in species extinctions and projected habitat loss as well as the implications of degradation for ecosystem services, associated with, for example, loss of wetlands and forests. Scenarios also include possible abandonment of agricultural land and rewilding in the European Union and the resultant reduction of agricultural land in some situations. CBD scenarios have focussed on four major challenges for the period up to 2050, namely: (1) climate change as a major driver of biodiversity loss and ecosystem change; (2) substantial increases of demand for fertile land for food production; (3) the collapse of many wild fisheries and aquaculture as dominant fish production system; and (4) increased water scarcity in many regions. Combinations of drivers could push some systems beyond irreversible tipping points. While these scenarios point to improvements in resource-use efficiency, productivity gains are typically insufficient to offset the effects of population growth and affluence-driven consumption on the depletion of natural resources and loss of biodiversity.

Focusing on land degradation, the ELD Initiative (ELD, 2015) explores economically-viable sustainable land management scenarios, guiding actions and investments for improving land management and achieving "degradation neutrality". The analysis uses spatially explicit representations of the productive capacity of land resources and of land degradation, at global and national scales. Land degradation is estimated to be associated with reductions of between 9% and 15% in the annual monetary value of global ecosystem services, with considerable variation between world regions and nations. The ELD Initiative uses four scenarios, broadly drawing on some of the aforementioned studies, which consider plausible futures with implications for land and water use and management. The value of land-based ecosystem services was estimated globally and for selected countries and regions, including Kenya, France, Australia, China, United States and Uruguay. Of particular interest here, the Initiative includes a visionary Great Transition Scenario that addresses the sustainability challenge through new socioeconomic arrangements and a fundamental change in values. This gives primacy to the preservation and restoration of natural systems and to equitably distributed human welfare. Under the Great Transition scenario, the continued losses in ecosystem services that are otherwise associated with land degradation under market driven or protectionist scenarios are reversed, with an annual difference in value of about US$75 trillion globally by 2050. There are greater relative differences for the most vulnerable country cases where restoration of critical services is important for poverty alleviation. While the visionary Great Transition restoration scenario shows the plausibility of reversing land degradation trends, it confirms that sustainable futures, rather than resting solely on technological solutions, are mainly predicated on changes in social, economic and institutional arrangements.

7.3.3 Lessons from scenario analysis

The lessons from scenario analysis as it applies to land degradation and restoration are as follows.

On the root causes of land degradation: Scenario analysis emphasises the central role played by agriculture in the determination of sustainable land use futures. The key challenge is to satisfy growing demands for food, fibre and energy from rural land, while protecting biodiversity and avoiding degradation of natural capital and loss of ecosystem services. Global scenarios point to particular pressures in regions where growth in agricultural production, whether for domestic consumption or export, exceeds capacity due to low geographical potential, low water availability and soils vulnerable to degradation. As a result, there is an increased likelihood of land degradation and a persistent poverty trap for many people (see also Chapter 5, Section 5.2.2.2).

"Decoupling" agricultural production from land and ecosystem degradation, whether by technological or institutional means, is seen as the most important pathway for sustainability. Technological solutions include enhancing and remediating technologies as part of "sustainable intensive farming", including the use of indigenous and local knowledge (Foresight, 2011; The Royal Society, 2009). Institutional solutions refer to the governance of land-based resources in the broadest sense, including the development of formal and informal social networks for sustainable land management (Foresight, 2011).

Scenario analysis also points to the uncertainties regarding the future of bioenergy and forestry as major land users. While climate change mitigation scenarios identify an expanding role for bioenergy production, there is considerable uncertainty about the effect on land degradation and water stress, especially on marginal lands and on the displacement of land for food production. Most future forestry scenarios consider that natural forests will come under increasing pressure from agricultural conversion, with remaining stocks being increasingly subjected to management agreements for timber and bioenergy, with implications for biodiversity loss.

Scenarios consistently point to the connectedness of land degradation to high level drivers and other outcomes. Population growth and patterns of consumption, whether induced by wealth or poverty, exert major yet indirect influences on land use. Strong interactions with other policy domains means that land degradation must not be seen in isolation.

On the quality of scenario: Much depends on the robustness and reliability of the underlying modelling processes and whether they are perceived by end-users to be relevant and fit for purpose. Scenario analysis is subject to limitations associated with: (i) the ability to incorporate the wide range of potential drivers and pressures; (ii) the adequacy of current knowledge, data and models to represent ecosystem functioning and dynamics; and (iii) the considerable uncertainty associated with the appraisal of social and political responses, including behavioural change and the efficacy of policy interventions and governance regimes (Smith *et al.*, 2010). As in other fields characterised by considerable complexity, development of integrated and spatially-explicit models will be necessary to better understand land degradation processes (Pichs-Madruga *et al.*, 2016). In spite of considerable progress, scenarios currently have limited ability to integrate diverse fields of knowledge – not only regarding the complex dynamics of ecological processes, but also the workings of parallel social systems of governance – especially as they detect and respond to tipping points and critical thresholds.

On key trade-offs and synergies: Scenario analysis helps to assess the synergies and trade-offs between social, economic and environmental objectives, and the causes and effects of land degradation and biodiversity loss. Scenarios can show how best to reconcile competing and growing demands on finite natural resources for food and water, bioenergy, rural livelihoods, urban development, as well space for nature itself. Scenario analysis can also help to identify opportunities for simultaneously achieving multiple objectives while protecting natural resources, especially at the local and regional scale. These include the use of natural pest control and pollination to support food production, the protection of soil carbon and biota to support farming and to mitigate climate change. Furthermore, scenarios can consider how forestry may provide a range of flood risk management, carbon sequestration, timber, soil formation and biodiversity services. In the urban space, local scenarios can help to assess the considerable potential for multi-purpose greenspace (EC, 2012). The extent to which scenarios have explicitly addressed the interactions between development objectives, land degradation and biodiversity loss, however, remains limited.

On scenarios for decision support: While global scenarios can support a general discourse and confirm the need for high-level concerted action, regional and local scenarios are more attuned to the needs of policymakers and practitioners. The relevance of scenarios for decision support increases at regional and national scales because context and drivers are better defined, processes and outcomes are more tangible and responses are potentially more actionable. Visionary scenarios, in particular, help to focus on key interactions and expose the gaps and fragmentation in current knowledge and skills. They also help to define the interface between policy domains, social and economic values, motivation and behaviour, as well as systems of governance and reward. They can focus on vulnerable land systems, on land degradation processes and locally specific impacts on ecosystem services and people. The ultimate test for decision-making is whether scenarios can help to determine best response options, especially those that are likely to be effective under a range of possible futures.

7.3.4 New approaches: visioning for sustainable land futures

Scenario analysis applied to land degradation and restoration have, to date, revealed gaps in knowledge and sources of uncertainty that must be filled if progress is to be made along the science-policy-practice continuum in pursuit of sustainable land management. These gaps not only apply to fundamental understanding of the land degradation-ecosystems-people nexus, but also of methods to support both the visioning process and how best to implement its enactment. Visioning, in particular, requires greater clarity about the objectives to be achieved, articulated in terms of sustainable development goals (Griggs *et al.*, 2014). This objective-oriented perspective places land degradation and restoration in the broader context of how the capacity of land systems can best be managed to meet societal needs, alongside those of other living organisms. In this context, there is a need to strengthen future treatment of land degradation and restoration scenario analysis with respect to the following:

Build knowledge of land degradation and restoration **processes, non-linear dynamics, critical thresholds and consequences for ecosystem services,** including the likely efficacy of response options to halt or minimize loss. This requires the amalgamation of often disparate and fragmented data sets and modelling capabilities into an integrated framework, developing context-specific metrics and indicators to assess land degradation and restoration processes and impacts (Turner *et al.*, 2015). Setting land degradation and restoration in a broader social and political context widens the scope of data and methods required.

Develop new data and methods to support a visioning approach, especially facilitating the integration of different knowledge systems: scientific and traditional. There is a particular need to strengthen methods for the social and economic valuation of ecosystem services, the evaluation of systems of governance and adaptive management, and the processes that affect motivation and behavioural change. The visioning approach requires a refinement of response options to address land degradation and determine context specific transformation pathways towards the vision of sustainable land use.

Construct new land degradation and restoration **scenarios that focus on vulnerable land systems, ecosystem services and peoples,** with explicit treatment of major threats, sources of risk and uncertainty, and synergies and trade-offs. These should not only include "exploratory" extrapolations of baseline scenarios to assess the possible risks of land degradation and the effectiveness of response options, but also "visionary", solution-oriented scenarios of sustainable land use and ecosystem futures that are supported by appropriate systems of governance. New scenarios are needed to explore land degradation and restoration and ecosystem loss in broad economic, social and political settings – allowing for possible changes in population growth, economic development and social preferences. There is also a need to explicitly consider the effects of land degradation and biodiversity loss on context-specific cultural services associated, for example, with the benefits of connectedness to nature and a sense of place and identity (see Chapter 2). Furthermore, new scenarios should explicitly consider the interactions between land degradation and restoration and major policy domains such as food security, poverty alleviation, energy, urban and industrial development, water resource management, education, science and technology, and international trade – exploring the potential synergies and conflicts that might arise.

Better represent stakeholder perspectives and institutional dynamics. Of particular importance for visioning, a better understanding is required of how diverse stakeholder interests and influences affect the distribution of flows of ecosystems and services amongst society members. There is also a need to explore how the prospect of tipping points in ecosystems simultaneously act as points of “leverage” in parallel systems of governance – with a range of possible outcomes, some more sustainable than others (Crona & Hubacek, 2010). A participatory social and ecological approach to scenario-building – possibly involving novel simulation, role play and negotiation methods to address “real world” challenges (Reed *et al.*, 2009; Robson, 2013) – is needed to achieve the stakeholder “buy-in” for legitimacy and commitment to change policy and practice.

Develop valuation tools to support the appraisal of the diversity of scenario outcomes and of response options, especially regarding the valuation of non-market ecosystem services. Scenario analysis will be strengthened by applying methods of economic cost-benefit analysis and social preferences for the assessment of possible land degradation and restoration outcomes, with clear support for decision making. Again, this broadens the data sets and methods required to support scenario analysis.

Undertake experiments and demonstration projects to further develop scenarios of sustainable land and biodiversity management and governance that act as showcases and learning opportunities to underpin a visionary approach, especially in areas that have experienced land degradation and ecosystem loss with major consequences for people and the natural world.

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Chapter 8

Decision support to address land degradation and support restoration of degraded land

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Table of Contents

Executive Summary

Decision-making on land degradation avoidance and restoration strategies requires an analysis and accessible information. Such an analysis can allow comparison between relative long-term and short-term merits of plausible options for a particular socio-ecological system (*well established*). Decisions on feasible options are more likely to reach their goal when guided by scientific scrutiny of the risks, costs and benefits, social and environmental fulfillment associated with each of the available options and climate change scenarios {8.2.1, 8.2.2}. Degradation mitigation and restoration responses are, however, constrained by availability of resources, technologies, knowledge of the system and institutional competencies {8.2.2, 8.3}.

Although conceptual frameworks for combatting land degradation and enabling restoration exist, current knowledge, information and tools cannot seamlessly support the complete process of evidence-based decision-making *(well established).* The use of tools and the associated data require close cross-disciplinary collaboration and enabling conditions. Monitoring strategies, verification systems, adequate baseline information and data are needed to measure, understand, design, implement and adapt decisions on land degradation avoidance and restoration. Currently, most decision support tools are mainly focused on assessing the biophysical state of the land; more-integrated tools that combine socio-economic and biophysical variables are needed to capture social-ecological interactions and impacts and are being developed {8.2.1, 8.2.2, 8.2.3, 8.3.5}.

Institutional competencies and policies are key drivers of land degradation and restoration (e*stablished but incomplete*). Building an adequate set of institutional competencies is a crucial first step to design, implement and combine efficient policy instruments {8.3.1, 8.3.2, 8.3.3, 8.3.4, 8.3.5, 8.3.6}. Robust science to evaluate the impact and efficiency of different institutional competencies and strategies in mitigating land degradation and developing restoration is still in its infancy {8.3}.

Institutions able to apply and align diverse policy instruments are more likely to mitigate land degradation and promote land restoration (e*stablished but incomplete*). To design, implement, select and align policy instruments (including legal, regulatory, financial, cultural and technical measures), different institutional competences are required {8.3}. Economic instruments like payments for ecosystem services and biodiversity offsets are efficient in theory, but require a set of institutional capacities to deliver expected outcomes {8.3.1, 8.3.3, 8.3.6, 8.4.3}.

Evidence shows that customary practices and indigenous and local knowledge are used within local, tribal or indigenous communities for sustainable land management (*well established*). Formalizing customary practices requires the adaptation of policies based on multi-stakeholder participatory approaches towards restoration of degraded lands. The use and development of community protocols can play an important role in advancing the respect of customary norms in formal decision-making {8.3.2.3}. Participatory and stakeholder engagement approaches can lead to co-development of restoration responses and jointly agreed prioritizations, making it easier to identify opportunities for collaborative responses that harness synergy {8.2.2, 8.3.4}.

To address multiple environmental and social challenges as well as harnessing synergies, restoration decisions and strategies to combat land degradation must be well aligned to ensure impact within other decision-making areas (w*ell established).* For example, national-level decisions seeking to ensure availability of adequate food - through the reduction of land degradation - need also to consider the impacts of the selected strategies on the achievement of policy goals targeting (e.g., water, energy and shelter for the growing population at other scales). Tools and approaches are available to assess coherence between policy areas. Reducing trade-offs, enhancing alignment and harnessing synergies among decision-making areas requires institutional coordination, multi-stakeholder engagement and the development of governance structures that bridge different ministries, types of knowledge, sectors and stakeholder groups {8.4.2, 8.4.3}.

Effective responses to land degradation can simultaneously contribute towards multilateral environmental agreements and goals including the Aichi Biodiversity Targets, the Sustainable Development Goals, the Ramsar Convention and climate change-related agreements such as the Paris Agreement and REDD+ (w*ell established*). Taking a multi-level approach towards preventing and reducing land degradation, and restoring degraded areas offers the potential to deliver benefits at various spatial and/or institutional levels, as well as working across a number of policy areas and stakeholder groups {8.4.1}. While these policies seek to ensure good quality of life and that national growth is supported, they sometimes fuel land degradation, which over time reduces productivity – leading to higher demand for more land and can increase deforestation with negative impacts on climate {8.4.1, 8.4.2}.

8.1 Introduction

In this chapter we consider how decisions are made to halt land degradation and restore the degraded lands, including actions to prevent, reduce and/or mitigate the processes of land degradation and to rehabilitate or restore degraded land. Decision makers operate across spatial levels ranging from local to international level, and can be part of different entities like international agencies, regional consortiums, national or local governments or even a farm. The decisions they make require knowledge and information about the resource and the tools available to address land degradation, institutional competencies to implement the decision, and an enabling environment. In light of the above, it should be noted that decisions to halt land degradation and restore degraded lands do not operate in isolation. They interact with other policy areas at regional, national and international level.

Decision making is a process not a single act in time and it does not follow strict sequential steps (Mintzberg *et al.*, 1976). In a decision making process, problems and objectives are normatively described and agreed upon, appropriate actions are explored, and actions are put in place and evaluated (Cowling *et al.*, 2008; Reed & Dougill, 2010a; Simon, 1986). At all stages, information, knowledge and/or tools are used by the decision maker. Decision support tools and methods particularly support the normative understanding and evaluation of trade-offs throughout the decision-making process, be it for an individual or groups of decision makers. Decision support tools are approaches and techniques based on science and other knowledge systems that can inform, assist and enhance decision-making and policymaking (IPBES, 2016a). A decision support tool aims to capture the trade-offs (Ackoff, 1981) between often nested, chained and poorly structured decision problems that can be wicked in nature (Rittel & Webber, 1973). In this chapter, we do not synthesize various theories of planning, decision and policy processes. We provide guidance in choosing and using decision support tools.

Decision makers can opt to use one or more policy instruments to achieve the decided upon goals for land degradation and restoration strategies. These include legal, financial, and cultural instruments (see Chapter 6). To design, select, and implement a policy instrument, institutional competencies are needed. Institutional competencies are the set of abilities which a given institution can use to achieve policy goals. Institutions encompass formal and informal social interactions and structures that determine how decisions are taken and implemented, and how responsibilities are distributed (IPBES, 2015a).

Land degradation and restoration is a cross-cutting issue. It influences the delivery of various ecosystem services that are essential for human well-being and a good quality of life (see Chapter 5). Various policy areas influence land degradation or enhance possibilities to address land degradation and develop restoration actions. These include climate change adaptation, biodiversity and ecosystem conservation and use, pollution, invasive alien species and disease management, infrastructure development, and flood risk and water resource management. Efforts to avoid and reverse land degradation will require the identification of synergy or trade-offs of multiple policy areas, and evaluating the possible outcome of a decided action.

As such, decision making strategies and policies to avoid land degradation and restore degraded land will depend on: (i) available information; (ii) institutional competencies to design and implement policy instruments; and (iii) influences of other policy areas.

Therefore, in this chapter, we consolidate information and tools necessary to support evidence-based decision-making for policy makers and practitioners responsible for selecting and implementing strategies to halt and reverse land degradation. We also assess institutional competencies necessary in the detection and analysis of land degradation problems, and the design, implementation, management and monitoring of response strategies. In the final section of this chapter, we place land degradation problems and potential restoration solutions within the wider policy context and describe other indirect drivers which can also be root drivers of both land degradation and land restoration. We consider interactions between land degradation, restoration and other major policy areas addressing agriculture, water, climate, infrastructure, and biodiversity. Where possible, we endeavor to separate information related to decision making levels and entities.

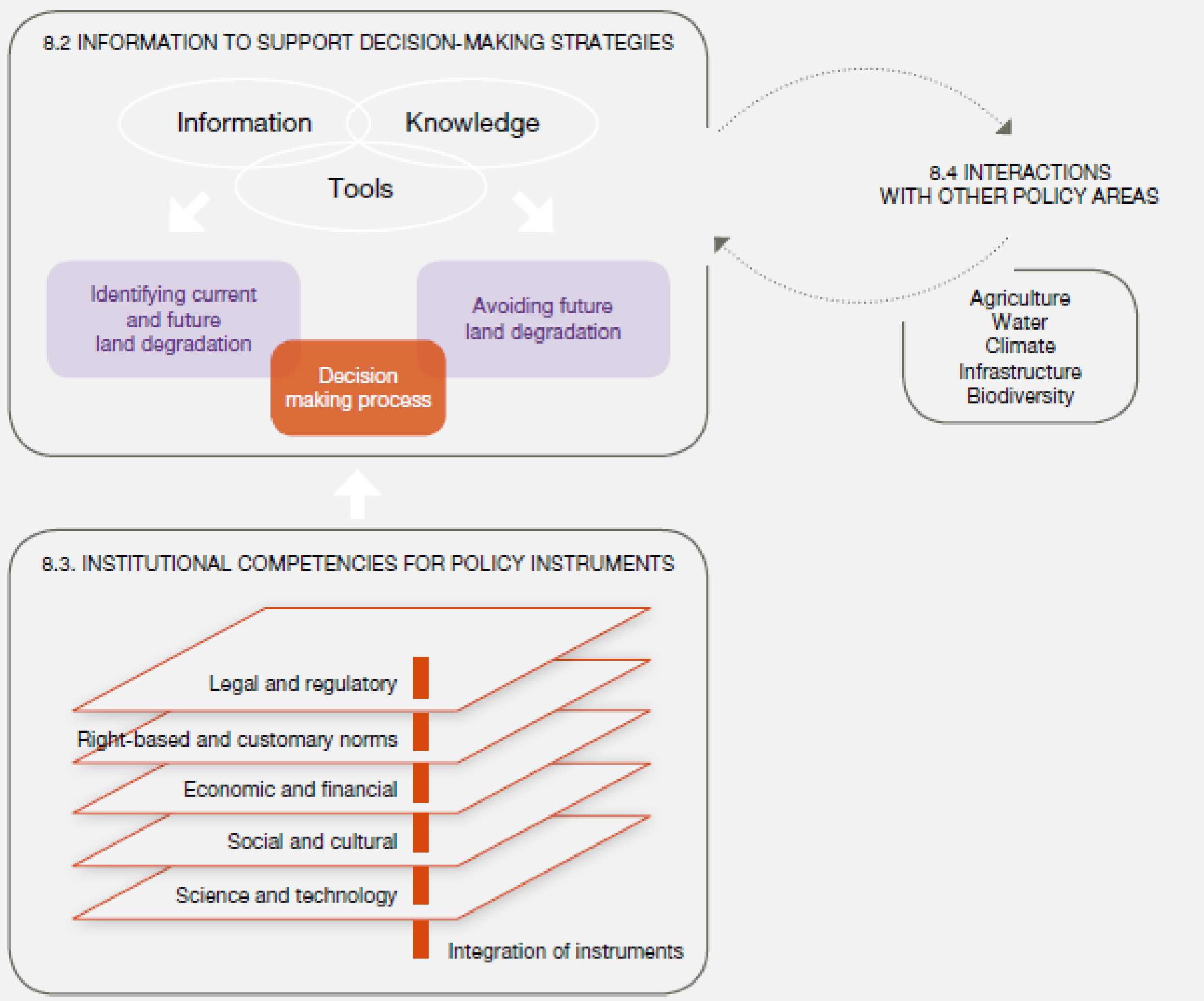
This chapter is structured in three main sections (Figure 8.1), which include an assessment of evidence on:

i. **Information, knowledge and tools** decision makers need to develop strategies on land degradation and restoration (8.2)

ii. **Institutional competencies** to design and implement strategies on land degradation and restoration, with a specific focus on national level actions and abilities (8.3)

iii. **Interactions between policies to halt land degradation and restore degraded lands, and other major policy areas** (8.4)

Figure 8 1 Restoration decision-making addressed in the three sections of Chapter 8.



8.2 Information to support decision-making strategies on land degradation and restoration

In this section, we focus on decision-making needs regarding information, knowledge and tools to identify land degradation problems (see Section 8.2.1), restorations solutions (see Section 8.2.2) and requirements for seamless-use of information, knowledge and tools throughout the different phases of the decision-making process (see Section 8.2.3). We address decision-making as a process over time as opposed to a single, discreet moment in time. Throughout the process, different questions need to be addressed which require insight into both biophysical and social systems.

8.2.1 Information, knowledge, and decision support tools available to identify land degradation problems

Empowering decision- and policy-makers with the spatial and temporal knowledge on the extent and severity of land degradation (see also Chapter 4) is essential to choose and implement adequate response actions. Effective decision support tools are of paramount importance to address land degradation problems. Decision support tools are approaches and techniques based on science and other knowledge systems that can inform, assist and enhance decision-making and policymaking (IPBES, 2016a). Decision support tools can provide insight into the extent and severity of land degradation and possible future alarming scenarios influencing decision makers to initiate conservation or restoration initiatives. A response process to halt or reverse land degradation is more effective when the problem assessment is carried out in a participatory way (Borrini-Feyerabend *et al.*, 2000; Bousquet *et al.*, 2007; de Vente *et al.*, 2016). Specifically, stakeholder participation can increase the likelihood that environmental decisions are perceived to be holistic and fair, accounting for a diversity of values and needs and recognizing the complexity of human-environmental interactions (Richards *et al.*, 2004). It may also promote social learning (Blackstock *et al.*, 2007; Reed *et al.*, 2008). Multi-scale approaches – making use of common indicators and a variety of information sources including scientific data and local knowledge through participatory methods – allow cross-scale analyses and there is established and documented evidence based on local experiments for decision-makers at various levels (Schwilch *et al.*, 2011). A study from southern Africa shows that local land managers participate in the collection and reporting of data, especially when tangible benefits come out of this process (Reed *et al.*, 2011). In this section, we describe decision support tools and their related information and knowledge sources which can support decisions makers in identifying and mapping current and future land degradation problems.

8.2.1.2 Identifying and mapping current land degradation

A range of decision support tools are available for assessing land degradation elements, such as: accelerated soil erosion; landslides; deforestation; problems of water logging, salinity and alkalinity; sea water encroachment; wind erosion; forest fire; declining soil fertility and crop yield; water scarcity; soil compaction and crusting; increases in wasteland; overgrazing; invasion of alien weeds; chronically drought- and flood-prone areas. Common technologies used in decision support are databases and look-up tables, geographical information systems (GIS), remote sensing, computer-based simulation models, knowledge-based or expert systems and hybrid systems. The methods behind these decision support tools employ qualitative or quantitative measures to assess the severity of land degradation and enumerate degradation footprints. Here we describe the functionality of the most commonly used qualitative and quantitative land degradation assessment tools per spatial level (and see Table 8.1). This

Section does not cover all available decision support tools, as they are compiled on the online IPBES Policy Support Tools and Methodologies catalogue (https://www.ipbes.net/policy-support).

At the global level, and to some extent at regional levels, tools like GLASOD (Bridges & Oldeman, 1999; Jones *et al.*, 2003; Oldeman *et al.*, 1990), GLADA (Bai *et al.*, 2008; Bai & Dent, 2006), LADA (Koohafkan *et al.*, 2003), are available to describe the distribution and intensity of degradation and to identify where degradation has been halted or reversed (see Table 8.1, all full names of the tools are listed there). GLASOD provides expert judgement on land degradation and can be used to raise the awareness of policymakers and governments for the continuing need for soil conservation (Bridges & Oldeman, 1999). ASSOD (Van Lynden *et al.*, 1997) is a more detailed tool, but has a strong regional affiliation to South and Southeast Asia. The NFPA (Borucke *et al.*, 2013; Weinzettel *et al.*, 2014) is a tool based on the concept of "bio-capacity". The tool calculates the amount of biologically-productive land and sea area available to provide the resources for a given population and absorb its wastes - with its current state of technology and management practices. Countries differ in the productivity of their ecosystems and this is reflected in the accounts. IMAGE (Hootsmans *et al.*, 2001; Stehfest *et al.*, 2014) is an integrated ecological-environmental model framework that simulates the environmental consequences of human activities at spatial levels (global or national level). IMAGE represents interactions between society, the biosphere and the climate system to assess sustainability issues such as climate change, biodiversity and human well-being. The objective of the IMAGE model is to explore the long-term dynamics and impacts of global changes that result from interacting socio-economic and environmental factors, and are therefore data intensive. One of its components assesses the loss in soil productivity as a result of human-induced land degradation, its effect on the carbon cycle, nutrient balance and crop productivity. The global IUCN Red List (IUCN, 2017) presents the extinction risk of thousands of species and subspecies. The Red List aims to: (i) provide scientifically-based information on the status of species and subspecies at a global level; (ii) draw attention to the magnitude and importance of threatened biodiversity; (iii) influence national and international policy- and decision-making; and (iv) provide information to guide actions to conserve biological diversity.

For regional or national levels, a range of tools is available to assess land degradation though soil related measures (see Table 8.1). These include PESERA (Kirkby *et al.*, 2004), SWAT (Arnold *et al.*, 1993), Geo-WEPP (Arnold *et al.*, 1993; Flanagan & Nearing, 1995; Renschler & Harbor, 2002), CORINE (Dengiz and Akgul, 2004) and the USLE/ RUSLE/ MUSLE models (Nearing *et al.*, 1989; Wischmeier & Smith, 1978). Soil organic matter is influenced by land management. The soil organic matter turnover can indirectly indicate the state of degradation and can be assessed using various models such as Roth C (Coleman & Jenkinson, 1996), CENTURY (Parton *et al.*, 1992), DNDC (Li *et al.*, 1992) to a considerable degree of confidence. These models are point-scale models and can be extrapolated to large spatial extents (for global or regional level applicability) using remote sensing and GIS approaches. Though these models are widely used, the erosion and hydrological flux associated soil organic matter movement requires coupling to multiple hydrological and erosion models. These process-based models are very accurate owing to their capabilities to simulate and describe the spatial distribution of degradation, but are heavily dependent on local and spatial input databases on land-use, soil and weather information. Lack of field validation and uncertainty in model parameters are major barriers in their applicability to areas where local databases are very scarce. Remote sensing-based information sources to assess land degradation – including high resolution Digital Elevation Models (DEM) by Shuttle Radar Topography Mission (SRTM) or Advanced Space borne Thermal Emission and Reflection Radiometer (ASTER) – provide morphometric and hypsometric characteristics of the land mass and are used as an indicator of degradation activities (Farhan *et al.* 2015; Prasannakumar *et al.*, 2011).

Other tools mostly applied at regional or national levels focus on land degradation from a biological perspective (see Table 8.1). SPLASH (Davis *et al.*, 2017) uses bioclimatic indices to assess ecosystem function, species distribution and vegetation dynamics under changing climate scenarios, for which direct observations on surface fluxes are sparse. The MODIS-NPP/GPP product (Zhao *et al.*, 2005) provides a remote sensing-based solution to quantify the primary production of vegetation as an indicator of land degradation, and is used in tools like LNS (Prince, 2004; Prince *et al.*, 2009). Biota (http://viceroy.eeb.uconn.edu/biota) offers a robust database with spatially-referenced, taxonomically-classified biodiversity inventories ranging from one-hectare vegetation plots, to regional or protected-area biotic inventories, to continental-level specimen databases. The database updates help to provide degradation status of biodiversity. Complimenting IMAGE derived outputs, GLOBIO (Janse *et al.*, 2015) assesses impacts of human-induced environmental drivers on land biodiversity in terrestrial ecosystems and freshwater systems in the past, present and future. Impacts on biodiversity are captured in terms of the biodiversity indicators Mean Species Abundance (MSA) and ecosystem extent. They can be considered applications of the Convention on Biological Diversity (CBD) indicators (i.e., “trends in abundance and distribution of selected species” and “trends in extent of selected biomes, ecosystems and habitats”, respectively).

Land degradation assessments at global or regional levels can provide a coarse resolution assessment to identify large areas and patterns or types of areas likely to have degradation problems. But due to the coarse resolution of these assessments, the management units related to the exact degradation becomes difficult to locate. As halting and reversing land degradation requires location-specific solutions and multi-sectoral collaboration, global and/or regional decision support tools do not provide any prescriptive solutions to combat the degradation problem.

At farm and landscape levels, the FALLOW (Forest, Agro-forest, Low-value Lands Or Waste) model provides prospective information on the impact of a particular strategies (Suyamto *et al.*, 2009; van Noordwijk, 2002). The model simulates land-use and/or land-cover change dynamics with various feedback loops and assesses the consequences of the resulting land-use mosaics on economical utilities and ecosystem services. Model results identify trade-offs between ecological and economical values. Process-based models such as SWAT, Geo-WEPP/WEPP are also capable of accurately map land degradation in quantitative terms at a fine spatial resolution.

Land degradation can be described using different methods. What constitutes an appropriate method depends on applicability and adaptability to a condition or form of land degradation. Table 8.1 provides an overview of the popular and mostly freely available land degradation assessment tools. In Box 8.1 we present examples of applications of decision support tools to assess land degradation at different spatial levels.

Table 8.1 Popular land degradation assessment tools.

Tools	Description	Spatial application level	Application outcome
Global Assessment of Human-induced Soil Degradation (GLASOD) method http://www.isric.org/projects/global-assessment-human-induced-soil-degradation-glasod	• Provides basic data on the world distribution and intensity of erosion, chemical and physical types of degradation	• Global	• Maps distribution and intensity of degradation
Global Assessment of Land Degradation and Improvement (GLADA) http://www.isric.org/projects/global-assessment-land-degradation-and-improvement-glada	• Involves a sequence of analyses to identify land degradation hotspots using remotely-sensed data and global ISRIC datasets	• Global • National • Local	• Identifies degradation hotspots and restoration bright spots
Assessment of the Status of Human-Induced Soil Degradation (ASSOD) https://esdac.jrc.ec.europa.eu/content/assod-status-human-induced-soil-degradation-south-and-southeast-asia-dominant-degradation	• Follow-up study of GLASOD in South and South-East Asia • Provides data for 17 countries and includes data on water and wind erosion, chemical deterioration	• Regional	• Identifies areas with severe erosion risk • Provides more spatially explicit and detailed information on land degradation
Land Degradation Assessment in Dry lands (LADA) http://www.fao.org/nr/lada/	• A Global Land Degradation Information System (GLADIS) database	• Global • National • Local	• Maps pressure and threat indicators at global level • Allows access to information at national, land use and pixel levels
Coordination of Information on the Environment (CORINE)	• Erosion and land quality database • Preparation of erosion maps and classification accordingly	• Regional • National	• Provides spatial and temporal soil erosion status maps (severity, impact)

https://www.eea.europa.eu/publications/COR0-landcover			
Pan-European Soil Erosion Risk Assessment (PESERA) http://www.isric.org/projects/pan-european-soil-erosion-risk-assessment-pesera	• Spatially-distributed model • Quantitative analysis of soil erosion by water	• National • Regional	• Provides spatial and temporal soil erosion status maps (severity, impact)
Universal Soil Loss Equation model (USLE)/ Revised Universal Loss Equation (RUSLE) http://milford.nserl.purdue.edu/weppdocs/overview/usle.html http://www.iwr.msu.edu/rusle/	• Empirical model • Quantitative data on spatial distribution of soil erosion • Requires data on annual average rainfall, soil, land use, management practices and terrain	• Local • Watershed • Regional • National	• Provides maps of soil erosion severity • Provides long-term annual soil loss due to the rill- and inter-rill erosion by water from the agricultural lands
Water Erosion Prediction Project model (WEPP/ Geo-WEPP) http://geowepp.geog.buffalo.edu/ http://milford.nserl.purdue.edu/weppdocs/overview/wepp.html	• Process-based erosion model • Quantitative estimate of soil erosion • Requires data on soil, DEM, daily climate, land use	• Hill slope • Landscape • Watershed	• Provides quantified estimates of severity of erosion on a hill slope
Soil and Water Assessment Tool (SWAT) http://swat.tamu.edu/	• Process based hydro-ecological model • Quantitative estimate of water yield, sediments, pollutants • Requires database on soil, daily weather data, land use, DEM	• Watershed • Sub-basin • River basin	• Provides spatial and temporal distribution and magnitude of soil erosion, water yield, pollutant load • Applied to quantify the impact of land management practices in large and complex watersheds

Integrated Model to Assess the Global Environment (IMAGE) http://themasites.pbl.nl/models/image/index.php/Welcome_to_IMAGE_3.0_Documentation	• Process-based ecological-environmental model framework • Quantitatively simulates the environmental consequences of human activities • Requires global database on precipitation, temperature, aridity index, biomass, land cover, Net Primary Production	• Global	• Identifies socio-economic pathways and projects the implications for energy, land, water and other natural resources
GLOBIO http://www.globio.info/	• Empirical/statistical model • Quantitative assessment of past, present and future human impact on biodiversity	• Global • Regional • National	• Provides a single measure of the intactness of ecological communities and the average abundance of all species
DNDC/ RothC/ CENTURY https://soil-modeling.org/resources-links/model-portal	• Process based modeling using data on long- and short-term climate, land management history, organic carbon status	• Local • Regional • National • Global	• Provides carbon turn over in soil from land management practice with plant input • Provides information on organic carbon status as an indicator of soil degradation
FALLOW (Forest, Agro-forest, Low-value Lands Or Waste) https://www.worldagroforestry.org/publication/forest-agroforest-low-value-landscape-or-wasteland-fallow-model	• GIS-based spatially explicit model • Quantitative analysis of land use change • Operates at spatial resolution of 1 ha, temporal resolution of 1 year and socio-economical resolution of 1 community,	• Local • Regional	• Applied for rural agro-forested landscapes • Provides simulated land- use and/or land-cover dynamics due to local responses on external drivers biodiversity
Simple process-led algorithms for simulating habitats (SPLASH)	• Process-based species distribution model • Requires bio-climatic variables derived from climate database • Uses global climate data	• Global • National • Regional	• Provides species distribution as an indicator of habitat loss or gain • Applied as a surrogate indicator of degradation
National Foot Print Accounts (NFPA) http://www.footprintnetwork.org/resources/data	• Quantitative database • Based on approximately 15,000 data points per country per year • The accounts calculate the Footprints of more than 200 countries, territories, and regions from 1961 to the present	Global National	• Provide time series of both Ecological Footprint and bio-capacity • A surrogate for indirect estimation of bio-capacity degradation

Local Net Primary Production scaling (LNS) method https://earthobservatory.nasa.gov/GlobalMaps/view.php?d1=MOD17A2_M_PSN	• Spatial manipulation model with vegetation index values derived from satellite imagery (MODIS)	• National • Regional	• Estimates potential production inhomogeneous land capability classes and models the actual productivity using remotely-sensed observations. • The difference between the potential and actual productivities provides a map of the location and severity of degradation

To ensure effective dissemination of land degradation-related information to those stakeholders who are at the level where they can influence decision-making, the assessment levels should be scalable from global to local level commensurable to the implementation level. The information exchange between the stakeholders and the science-driven knowledge should live up to five principles comprising: (i) the knowledge exchange goals; (ii) adjustability to changing user needs and priorities; (iii) long-term trusting exchangeability; (iv) having deliverables tangible in nature; and (v) sustaining a knowledge legacy (Reed *et al.*, 2014).

Box 8.1 Examples of Application of decision support tools at various assessment levels.

Global/ National: China (Bai *et al.* 2005)

The study on the status and trends of land degradation and identification of hotspots (using the GLADA method) was carried out in North China using the 22-year NOAA-AVHRR GIMMS dataset of normalized difference vegetation index data and ancillary information. The results indicate that overall green biomass increased over the 22-year period with an insignificant correlation with rainfall. A delayed response of declined biomass production was observed with diminished rainfall. Rain-use efficiency was found to follow an inverse trend with improvement in land conditions. Normalized difference vegetation index attenuation took place quite long before the growing season climax. Declining green biomass production, a surrogate indicator of land degradation, is highly localized. Authors opined that various indicators developed - with direct and indirect reference to land degradation such as soil erosion, infiltration, water storage and soil organic matter - could be used as input for an early warning system for land degradation. These facts were corroborated through field validation.

Regional: Australia (Jackson & Prince 2016)

This study employed the local NPP scaling (LNS) approach to identify patterns of anthropogenic degradation of NPP in the Burdekin Dry Tropics region of Queensland, Australia, from 2000 to 2013. This region (7.45×10^6 km^2) was investigated at a spatial resolution of 250 m. The average annual reduction in NPP due to anthropogenic land degradation in the Burdekin Dry Tropics region was estimated at 2.14 MgCm^{-2}yr^{-1}, or 17% of the non-degraded potential, and the total reduction was 214 MgCyr^{-1}. Extreme average annual losses of 524.8 gCm^{-2} yr^{-1} were detected. Approximately 20% of the region was classified as “degraded”. Varying severities and rates of degradation were found among the river basins. Inter-annual, negative trends in reductions of NPP occurred in 7% of the entire region, indicating ongoing degradation. There was evidence of areas that were permanently degraded.

Local: China (Zheng & Hong 2012)

The spatial pattern of soil erosion and deposition on a catchment scale were estimated with the Geo-WEPP model in a small catchment of the Sichuan Hilly Basin. The estimated sediment delivery per unit area and sediment delivery ratio was estimated to be 2760 Mg $km^2 yr^{-1}$ and 0.485, respectively. Compared with the results derived by the second soil erosion survey based on remote sensing, the results by the Geo-WEPP model were validated through field observation. Post-validation of the scenario analysis was carried out to establish spatial pattern sediment delivery. It was found that the woodland has better soil and water conservation benefits than cultivated slopes. Geo-WEPP was found to be a useful tool to establish effective policy.

8.2.1.3 Identifying future land degradation

Decisions addressing land degradation problems are not only based on an assessment of the current land degradation, but also on the expected future state of the land. Scenarios can be used to assess the dimensions of future land degradation (IPBES, 2016b) (see also Chapter 7). Scenarios employ climate conditions, anthropogenic and natural drivers, and institutional and governance drivers in a future time frame. These could be linked to process-based land degradation models with GIS integration like SWAT or Geo-WEPP (Table 8.1). Assessing land degradation drivers and future degradation is key for deciding the urgency, societal relevance and stakeholder's preparedness for land degradation responses.

Land seldom remains in a state of equilibrium and often exhibits multiple ecological and social states. Underlying socio-economic processes can move systems slowly towards thresholds, and once reached, the bio-physical integrity of the system can rapidly be interrupted. This process is also known as non-linear regime shifts and can be extremely difficult and costly to reverse. To understand land degradation and prioritize action, there is a need to identify and manage for the small set of "slow changing" variables (e.g., loss of soil nutrients) that drive the "fast changing" ecological variables (e.g., reduction in crop yield) which matter at any given scale, in the context of multiple system thresholds. These thresholds need to be evaluated and the cost of recovery quantified in order to seek ways of managing the thresholds to increase resilience (Reynolds *et al.*, 2007).

A new dryland development paradigm (Stringer *et al.*, 2017) which builds upon the work by Reynolds *et al.* (2007) identified three integrative principles: (i) to identify linkages and feedbacks among multiple actors involved in decision-making by "unpacking" relationships and interactions between socio-ecological systems, livelihood portfolios and value chains; (ii) research needs incorporating multiple knowledges "traversing" across spatial and temporal scales and comprising of "fast" and "slow" variables – the reason being that degradation is mediated by interactions between multiple drivers of change, socio-technical innovation and investment options across sectors and scales; and (iii) "sharing" knowledge across multiple decision-making stakeholders to co-produce contemplative output for communities – at broader spatial and social scales - through social learning, including empowering disadvantage groups to participate in research and development process.

The identification of a unifying concept or explanation for land degradation processes is still a challenge. Such complexity can be tackled referring to the concept of "syndromes" (Ceccarelli *et al.*, 2014). "Syndromes" of land degradation can be evaluated in the past constructing land-use and/or land-cover change trajectories using prediction rules and scenarios developed for the future, using external drivers such as climate change and anthropogenic interferences. This can serve as information baselines for sustainable land management strategies and interventions. Still, challenges exist to develop an effective scenario pathway to develop the future land degradation trajectories. There is a need, through proactive science and policy dialogue to: (i) embrace a long-term scenario strategy that has the potential to significantly improve the relevance of future assessments on biodiversity and ecosystem services; and (ii) adopt a participatory, multi-scale scenario approach that captures the diversity of local social-ecological dynamics and builds understanding of interactions between global and local processes intertwined in generating ecosystem services and human well-being (Kok *et al.*, 2017).

8.2.2 Information, knowledge and decision support tools to identify land degradation prevention and restoration options

8.2.2.1 Quantitative and comparative analysis of land degradation avoidance solutions and restoration options

Land degradation response actions include land degradation prevention and restoration. While prevention lies in proactive policy decisions on conservation and the sustainable use of resources, restoration is a forward-looking process that seeks to initiate or accelerate the recovery of an ecosystem from a degraded state. The decision on a restoration option needs to be goal oriented, specific to a certain ecosystem, at various scales taking into account the recovery potential of the system as well as the needs of the society (see Chapter 1 and 6). Hence, defining clear restoration goals requires not only the identification of plausible options that are available for the particular ecosystem, but also considerations of the diverse interests of stakeholders. Besides, restoration and degradation mitigation responses are constrained by variables such as available resources (e.g., budget, community support), technologies, knowledge of the system and choice of options. Given the heterogeneity of such variables across systems and scales, a context-specific restoration or degradation avoidance solution is more likely to be effective than generic prescriptions (Gärtner *et al.*, 2008; Hobbs & Harris, 2001).Therefore, a comprehensive assessment of the biophysical, socio-economic and governance/institutional variables is essential to make informed decisions on restoration.

Decision support for restoration or degradation avoidance solutions aim to assist in making informed decision on available option – one that is optimal and feasible in terms of technology, cost and stakeholder satisfaction. Decision support tools can help to maximize the cost-effectiveness of restoration by identifying areas with different capacities for natural regeneration (Príncipe *et al.*, 2014; Guzmán-Álvarez & Navarro-Cerrillo, 2008). These tools require data and information from scientific studies of risks, cost-benefit analysis and qualitative assessment of stakeholders' views. The tool can be either written guidelines or software-based guidance. Some of the commonly applied decision support tools include Multi-Criteria Analysis (MCA), Life Cycle Analysis (LCA), Cost-Benefit Analysis (CBA) and Cost-Effectiveness Analysis (CEA), as described in Onwubuya *et al.* (2009) (see Box 8.2).

Box 8.2 Description of common decision support tools to select land degradation response actions. Based on Onwubuya *et al.* (2009).

Multi-Criteria Analysis (MCA): identifies the preferred option, ranks and distinguishes acceptable from non-acceptable alternatives. MCA is largely driven by expert judgment and a degree of bias in the outcome is unavoidable. MCA can be applied in combination with monetary and non-monetary values in the decision-making process, which is also called Multi-Criteria Decision Analysis (MCDA). MCDA is applied to analyze complex problems that are characterized by any mixture of monetary and non-monetary objectives. The tool can be used to synthesize data and information on identified problems and organize a set of decision criteria for each category of problems, so as to enable decision makers to choose the appropriate solutions.

Life Cycle Analysis (LCA): compiles and evaluates the inputs, outputs and the potential environmental impacts of a product system throughout its life cycle. The technique is often used to analyze for example the “cradle to grave” of products. Though LCA is popularly applied in the manufacturing industry, it has also become an important environmental decision support tool in managing and selecting technological options for degraded land restoration and remediation of contaminated lands. LCA enables comparisons between impacts of effectiveness of actions on land restoration. It also allows the selection of technological options by taking into account of stakeholder interests and views.

Cost-Benefit Analysis (CBA): assesses all costs and benefits involved in the different available options. Costs can be considered not only as monetary context, but also anything that can reduce human well-being - while benefits are anything that can enhance human and environmental well-being. Application of CBA may require expert knowledge, and sometimes difficulties associated with the monetization of ecosystems and the evaluations of the social acceptability of a certain option can be barriers to implementation.

Cost Effectiveness Analysis (CEA): provides a framework for making decisions on the least costly option to deliver the required standard outcomes. It is a relatively simple balance of the costs of a measure against its effectiveness and whether it meets given restoration objectives.

In the UK, the Environment Agency and Defra have developed a written guidance document entitled “Model Procedures for the Management of Contaminated Land" or simply referred to as "Contaminated Land Report 11 (CLR11)". This document outlines procedural guidance for the whole life cycle of the management of contaminated sites. Another example is from Germany, which has detailed written guidance documentation used for decision-making in contaminated land management - providing procedural step-by-step guidance for each and every activity. Similarly, the Swedish Environmental Protection Agency (SEPA) provides a broad national (written) guidance on remediation of contaminated sites, extending from inventory estimation to implementation of remediation projects. The guidance is given in the form of guidelines and manuals that are used (as decision support tools) by local authorities and practitioners taking responsibility for the investigation, remediation and after-care of contaminated sites. In addition to the written guidelines, software-based tools are also developed for remediation of contaminated sites in Europe (Onwubuya *et al.*, 2009). Examples of such models are given in Box 8.3.

Box 8.3 Examples of decision support tools for remediation of contaminated sites in Europe.
Based on Onwubuya *et al.* (2009).

PhytoDSS: applies phytoremediation technology to restore contaminated or polluted sites with the use of targeted plant species. The technique restores degraded sites through uptake of selected contaminants by specifically-selected plants (a process called phytoextraction) and through immobilization of contaminants through re-vegetation of sites with target species of plants and through the addition of other chemical inputs to immobilize the pollutants (mainly metals and metalloids), which is a process of phytostabilization. *PhytoDSS* uses the REC model (described below) for its implementation (http://www.eugris.info/displayProject.asp?ProjectID).

REC (Risk reduction, Environmental merits and Cost): combines risk reduction, environmental merits and cost, which in earlier times had been studied individually and integrated into decision-making to manage contaminated land.

ABC (Assessment, Benefit, Cost)**:** it is similar to REC, but improved in many respects. The tool assesses the feasibility of different options and utilizes LCA to assess the advantages and disadvantages of each option and evaluates the cost of each of the remediation technical options.

There is a large variety of ecosystem-based management tools that can be applied for selecting a solution for land degradation (Table 8.2). The tools can be found in the "Ecosystem-Based Management Tools Database, 2012" (https://ebmtoolsdatabase.org/). Despite the wide range of tools provided in the database, few are directly relevant and applied for multiple ecosystem services analysis (Bagstad *et al.*, 2013). For instance, ESR is a simple spreadsheet-based process decision support tool developed by the World Resource Institute (WRI) to qualitatively assess the impact of corporate businesses on the ecosystem services, so as to identify mitigation options at multiple scales, both to benefit the business and society at large (Hanson *et al.*, 2012).

Amongst the spatially-explicit ecosystem-based tools, MIMES can incorporate inputs from stakeholders and biophysical data sets for ecosystem valuation and decision-making. MIMES simulates human and natural systems interactions and provides estimates of near-term and long-term effects at different spatial levels. At the landscape or watershed levels, InVEST helps decision-making based on quantitative assessment of trade-offs in alternative management options (Kareiva *et al.*, 2011; Tallis *et al.*, 2013). Similarly, the ARIES model is a watershed-scale model that quantitatively maps natural capital, natural processes, the human beneficiaries and ecosystem service flows in an understandable way to manage ecosystems (Villa *et al.*, 2011).

There are also the GIS-based spatial analysis tools such as the SolVES and LUCI tools, which are applied at landscape and watershed scales (Jackson *et al.*, 2013). SolVES incorporates quantified social values and perceived non-market values that the public ascribes – such as cultural services, aesthetic and recreational services – into the ecosystem services assessments for different stakeholder groups (Sherrouse & Semmens, 2014). LUCI uses Multi-Criteria Analysis to explore the impacts of decisions on land-use and management changes. Among the web-based tools, Co$ting Nature is a model that aims to facilitate decisions on conservation priorities and to assess impacts of development activities such as agricultural production, mining, industrial developments on ecosystem services, as a result of human pressure on biodiversity and ecosystem services. The WOCAT tools collect and share standardized local knowledge on sustainable land management. Table 8.2, below, provides descriptions on the applications of some of the common and freely-available ecosystem-based decision support tools.

Table 8.2 Tools for finding restoration solutions.

Tools	Description	Spatial application level	Application
Integrated Valuation of Ecosystem Services and Tradeoffs (InVEST) http://www. naturalcapitalproject.org	• Software-based spatially- explicit model (GIS-based). • Quantitative analysis of spatial changes on ecosystem services for different management options • Requires parameterization of qualitative variables	• Landscape • Watershed	• Quantitative spatial output (ecosystem services mapping and valuation) • Flexibility to assess alternative management options by measuring the trade-offs • Operation involves expert rules and outputs may involve some degree of bias
Multi-scale Integrated Models of Ecosystem Services (MIMES) http://www. afordablefutures.com	• Set of software-based integrated dynamic models developed through web-based participatory process • Qualitative and quantitative analyses of changes in ecosystem services • Serves as a training tool, allowing simulation of policy options before making decisions • Interactive and participatory analysis of ecosystem services based on different policy scenario	Multi-scale: • Global • National • Regional • Local	• Spatially explicit quantitative output on ecosystem services • Spatial and temporal changes on the values of ecosystem services • Through a simulation iterative process, it allows decision makers to understand ecosystem dynamics, the link to human well-being and how the values change under different management scenarios.
Ecosystem Services Review (ESR) http://www.wri.org/	• Simple spreadsheet-based model • Qualitative analyses of impacts on ecosystems and society	• Landscape • Watershed	• Qualitative output • Direct and indirect negative impacts of development and corporate business that are linked to ecosystem services • Output is used to make decisions on mitigation and management options. • Applied for environmental auditing

			• Improves reputability of corporate businesses
Artificial Intelligence for Ecosystem Services (ARIES) http://aries.integrated modelling.org/	• Agent-based software modelling tool • Quantitative analysis of ecosystem services (e.g., carbon sequestration, using Bayesian networks and Monte Carlo simulation) • Monetary valuation of ecosystem services	• Landscape • Watershed	• Quantitative output • Spatially explicit ecosystem service flows (maps) and the trade-offs, including uncertainty maps • To make decision on efficient and cost-effective actions that improve biodiversity and ecosystem services
Land Utilization & Capability Indicator (LUCI) http://www.lucitools.org/	• Process-based Spatial software • Quantitative analyses of spatial information on ecosystem services	Multi-scale: • National • Regional • Watershed • Landscape • Local • Land unit/ site	• Spatially explicit ecosystem service tradeoff maps • Potential trade-offs and synergies among multiple ecosystem services • Quantitative output on potential gain or loss of ecosystem services different management scenarios. • Map outputs with ecosystem services • Quantitative output (data on ecosystem services) • Explores the capability of a landscape to provide ecosystem services.
Co$ting Nature http://www.policysupport.org/costingnature	• Web-based spatial model • Quantitative analyses of ecosystem services under future climate change scenarios • Simulates human actions to identify intended and unintended consequences • Helps to understand effectiveness of policies before implementation.	• Landscape	• Spatially-explicit quantitative output • Baseline indicators • Provides index for analyzing changes on ecosystem services (e.g., carbon stock, clean water availability, hazard mitigation) • Applied for Natural Capital Accounting and analyzing the ecosystem services supply

	• Freely available for non-commercial use (open access)		• Used for conservation prioritization and analysis of co-benefits
Social Values for Ecosystem Services (SolVES) http://solves.cr.usgs.gov	• GIS software-based spatial model • Quantitative analysis of social values for ecosystem services • Freely available.	• Landscape	• Transforms non-monetary social values of ecosystem services as perceived by different social groups • Provides scaled index of quantified non-market values of ecosystem services
World Overview of Conservation Approaches and Technologies (WOCAT) https://www.wocat.net	• Sustainable Land Management Database of good practices • Quantitative data on local knowledge, tested technology and practices	Multi-scale: • Local, • National • Regional • Global	• Identifies suitable SLM technologies and approaches • Helps to determine priority areas for interventions

Some of the above-mentioned tools have been applied in a variety of ecosystems (Box 8.4) and delivered encouraging results for restoration decision making. Spatial modelling and decision support tools can provide decision makers with information on optimal options in restoring degraded ecosystems (Goldstein *et al.*, 2012) by quantifying natures' contribution to people under different scenarios of management decisions.

Box 8.4 Application of SoLVES and InVEST in Taiwan for conservation priority decision-making.

Lin *et al.* (2017) applied SolVES and InVEST models to prioritize ecosystem services in systematic conservation planning in the Datuan Watershed of Northern Taiwan. The study was aimed at making a comparative spatial analysis of biophysical service areas with social value areas. High priority areas of biophysical ecosystem services were identified and mapped based on location-specific data, which were generated using the InVEST model. The social ecosystem services (high priority social value) areas were identified using SolVES based on data generated from questionnaire surveys. Land-use suitability maps, which ultimately dictate future land-use change, were calculated based on both land-use allocation maps and direct drivers of environmental variables. The systematic conservation planning zonation then generated spatial-prioritization scenarios based on different inputs. The zonation results were then compared in multiple objective programming via social-ecological matrix analysis. The findings showed that while the biophysical services were distributed with high spatial variability, the social values had high spatial overlap. About 6% of the watershed area showed both high biophysical and social services, while about 24.5 % of the areas were identified either high in biophysical services or vise-versa. Urban development scenarios affected the conservation area selection drastically. The results indicate trade-offs and potential synergies between development, social values and biophysical services. The results can be used for finding solutions to social-ecological planning complexities that serve multiple stakeholders. The results of the comparison can also inform decision makers and prompt further discussion about conflicting priorities.

8.2.2.2 Spatial prioritization of land degradation avoidance solutions and restoration options

Different approaches to prioritize locations and spatially plan for land degradation avoidance and restoration actions exist.

Spatial conservation prioritization (SCP) addresses resource allocation and ecologically based land-use planning. It is a quantitative analytical step that is often utilized within a broader operational framework for the implementation of conservation, such as systematic conservation planning (Kukkala & Moilanen, 2013). SCP analyses are often carried out using special software, originally designed for solving reserve selection problems - such as Marxan, Marxan with Zones (Watts *et al.*, 2009), or Zonation (see Pouzols *et al.* (2014) for references). The strength of SCP analyses is that they can integrate a large number of spatial data layers relevant for ecologically-based land-use planning. Most common analyses are based on data about the distributions of species and habitat types, but additional information about costs, threats (including land degradation), connectivity or ecosystem services is sometimes used depending on analysis needs and data availability.

The original form of the conservation area selection problem is a target-based formulation: which set of sites satisfies targets given for biodiversity features (often species) with minimum cost (see Moilanen *et al.* 2009 for review)? This type of problem is frequently solved with the Marxan or Marxan with Zones software (Watts *et al.*, 2009). A second form of analysis is balanced spatial priority ranking, which allows versatile analysis – also from the perspective of impact avoidance and accounting for land degradation. Spatial priority ranking is often done using the Zonation approach and software (see application examples in Box 8.5). Linking land degradation to spatial conservation prioritization can help answer the following types of questions: (i) How much biodiversity has been lost due to land degradation compared to the reference state? (ii) Where are optimal expansion areas for reserve networks given that parts of the landscape have become reduced in quality? (iii) Where would it be most important to avoid further land degradation? (iv) Where are areas where further land degradation is least harmful for biodiversity?

Box 8.5 Examples of spatial prioritization applications.
Based on Lehtomäki & Moilanen (2013); Pouzols *et al.* (2014).

Typical uses of spatial priority ranking include:

i. Traditional reserve selection, which is the identification of the highest-ranked part of the landscape (~reserve network) that produces high return on investment and balanced outcome across all biodiversity features.
ii. Reserve network expansion. Here, an optimal balanced expansion of an existing reserve network is identified, optionally accounting (e.g., connectivity or costs).
iii. Evaluation of an existing or proposed conservation area network. This is implemented as a comparison between how good it is and how good it could have been.
iv. Spatial ecological impact avoidance (e.g., Kareksela *et al.* 2013). Here, the objective is to identify areas where economic development leads to limited ecological losses.
v. Balancing of alternative land uses. A balance between many biodiversity features and the needs of several alternative land uses is achieved by entering alternative uses (~opportunity costs) as negatively weighted features into the analysis - which helps to resolve conflicts between conservation and resource utilization (Kareksela *et al.*, 2013).
vi. Target-based planning. This addresses the requirement for identification ways to meet the targets with least cost or to maximize the number of targets met (achieve highest output) with a given resource (Moilanen, 2007).

vii. Biodiversity offsetting. Find areas that best compensate for ecological damage: how to expand the existing reserve network in a balanced manner to compensate for specific losses. This requires land degradation and offsetting gains to be developed into spatial layers for input.
viii. Planning under climate change. These analyses use both present and future distributions of biodiversity features, as well as connectivity between the present and future distributions to identify current and future areas of relevance.
ix. Targeting of habitat restoration or habitat management. This requires modelling of the feature-specific "difference made" by management or restoration, leading to a comparatively complicated and data demanding analysis.

The **Restoration Opportunities Assessment Methodology (ROAM)** offers a framework for countries to identify and assess potential for forest landscape restoration and to locate specific areas for restoration at the national or sub-national level (IUCN, 2014). ROAM is used to support planning of national restoration programmes, based on collaborative engagement with stakeholders. The methodology is meant to be quick and non-technical, allowing broad stakeholder engagement in the process.

In implementing restoration programmes, decision makers need to prioritise which landscapes they will be working in, taking into account the multiple uses of areas and considering diverse social and ecological needs (Vogler *et al.*, 2015). A **production possibility frontier (PPF)** framework can be used to graphically illustrate trade-offs between two inputs in pursuit of a particular output level. For example, to understand how distributions of forest stressors and ecosystem services shape restoration options across the landscape (Vogler *et al.*, 2015).

Another example is the **Ecosystem Management Decision Support (EMDS)** tool (Reynolds & Hessburg, 2005). This tool is based on an integrated approach to evaluate the system, which answers the question of "what is the state of the system?" and planning of response options which answer the question of "what are the optimum solutions to address the problem?". This tool can be applied in a single watershed or sub-watershed in a landscape.

Bayesian Network for catchment restoration (Stewart-Koster *et al.*, 2010) facilitates the development of conceptual models of likely cause and effect relationships between flow regime, land-use and river conditions and provides an interactive tool to explore the relative benefits of various restoration options. When combined with information on the costs and expected benefits of intervention, one can derive recommendations about the best restoration option to adopt - given the network structure and the associated cost and utility functions.

Another tool that can be used for prioritization of land degradation response options is the use of **a scorecard** (see ELD Initiative (2015); CATIE & The Global Mechanism (2011)). Scorecards can be developed to assess - based on stakeholder knowledge - how feasible different options are and can also include considerations of trade-offs and synergies in identifying preferred options to halt, prevent and reverse degradation. Scorecards have been used to prioritize incentive- and market-based mechanisms in countries such as Zambia, Panama and Cambodia. The use of these can facilitate a ranking of options through the use of numerical scoring. However, scorecards need to be used as part of a suite of tools that allow overall evaluation of the implications of decision-making.

Dynamic systems modelling has been used to develop options for prioritization in environments as diverse as Botswana's Kalahari (Dougill *et al.*, 2010), Brazil's tropical forests (Vitel *et al.*, 2013) and in watershed planning in Quebec, Canada (Adamowski and Halbe, 2011). **Scenario modelling** is another useful approach as it can highlight possible plausible futures and therefore land degradation response priority locations (Costanza *et al.* 2015) (see also Chapter 7).

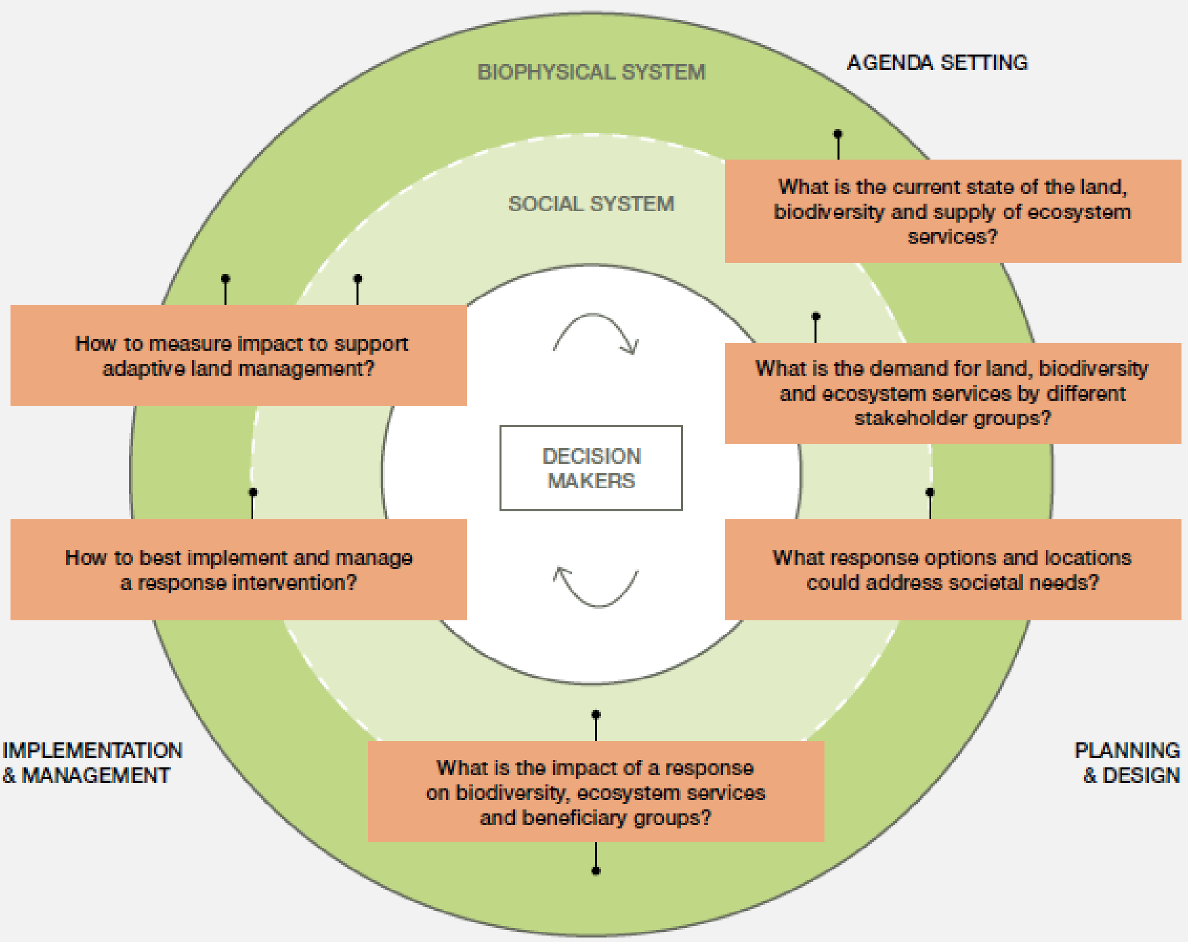
8.2.3 Linking decision support tools to the whole land restoration decision-making process

Different divisions and labels are proposed to describe such decision-making processes in land management (e.g., Cowling *et al.* 2008; Hessel *et al.* 2014; OECD 2016; Reed & Dougill 2010; Scherr *et al.* 2014). We describe the process with the Agenda setting, Planning and Design and Implementation and Management phases, followed by a review of progress towards meeting the objectives as set in the Agenda-setting phase (IPBES, 2016b). This iterative cycle of improving management policies and practices by learning from the outcomes of previously employed policies and practices can be referred to as adaptive management (Cowling *et al.*, 2008; Lal *et al.*, 2002; Sayer *et al.*, 2013). Figure 8.2 depicts such an adaptive cycle. Throughout the different phases the focus of decision makers changes from understanding to exploring, to planning, to revisiting and revising. The strict sequential occurrence of these phases (as shown in Figure 8.2) is, in practice, not always observed (van Stigt *et al.*, 2015). However, these phases do provide a useful architecture for grouping and linking activities and information needs in a decision-making process.

Both land degradation and restoration emerge from the interplay of social (including economic) and biophysical processes (Benayas *et al.*, 2009) (see also Chapter 4, 5 and 6). To support decision-making regarding land degradation response strategies, information and knowledge on social as well as biophysical characteristics are needed. Figure 8.2 shows examples of questions decision makers address when identifying and resolving land degradation problems. These questions relate to the social and biophysical sphere, or their specific interlinkage. As there is no single decision support tool that is able to deal with the full suite and complexity of decision-making questions on land degradation and restoration responses, multiple tools and approaches are required throughout the decision-making process (Turner *et al.*, 2016). Tools that are used to address initial questions in the Agenda-setting phase should generate information and knowledge to feed into Planning and Management phases. Therefore, decision-making support is shaped by the compatibility of different tools and actor collaborations. By discussing decision-making support as an interlinked pathway rather than in terms of single tools, we can assess what information is needed to support the subsequent step and indicate the different actors that need to be involved in each stage of the policy cycle. In this Section, we describe the use of information, knowledge and tools to move from Agenda Setting to Planning & Design, to Implementation & Management phases in the policy cycle.

To describe the linkages, we selected six example questions which also relate to the different chapters of this IPBES assessment (See Figure 8.2 and Table 8.3). Policy support tools depend on information and knowledge, but also generate crucial new information and knowledge as input to subsequent phases of the decision process. Here, we assess what types of tools, information and knowledge are required to smoothly move through the different decision-making phases, eventually leading to informed decision-making. For example, to guide the selection of policy support tools from online repositories such as NEAT (http://neat.ecosystemsknowledge.net/tools.html) or the IPBES online tool catalogue (https://www.ipbes.net/policy-support).

Figure 8 2 Lining up evidence-based tools to address questions throughout the decision-making cycle. Source: After Willemen *et al.* (2014).



From Agenda Setting to Planning & Design

During the Agenda-setting phase tools are needed to specify land degradation problems in order to plan and design adequate responses. This phase motivates and sets the direction for policy design and implementation (IPBES, 2016b). To identify solutions for land degradation, information on land degradation – together with social demands and values – need to be linked to plan and design viable options to mitigate land degradation and restore land. A wide range of tools are available to identify and describe land degradation (see Section 8.2.1 and the IPBES online tool catalogue), with a varying applicability for different spatial extents. Key outputs of these tools for decision-making includes knowledge and information on location, type, severity, temporal aspects of land degradation. These are preferably described with measurable indicators, adequate for the location, livelihood system and land degradation processes (see Table 8.1 and Chapter 4) (also see Convertino *et al.*, 2013; Geijzendorffer *et al.*, 2015; Kairis *et al.*, 2014). The selected and measured indicators, in this phase, must be measurable over time to play a role in monitoring land degradation trends and impact assessment of response actions (Heenan *et al.*, 2016; Reed *et al.*, 2010b). The scope of the land degradation problem is set by the demand, expectations, values and perceptions of stakeholders regarding land availability and ecological functioning (also see Chapter 2) (Couix & Gonzalo-Turpin, 2015) - and other stakeholder objectives. A plurality of values can lead to different demands for land and ecosystem services, and different perceptions of the severity of degradation and impact, among different stakeholders. An important step, here, is agreeing upon the land degradation problems and related mitigation objectives; facilitation and negotiation and consensus-building tools can play a role in this (Van Noordwijk *et al.*, 2013). A review by Turner *et al.* (2016) shows that limited decision support tools and methods are available to assess stakeholder values and the social context of land degradation and response strategies.

An additional function of the assessments in the Agenda-setting phase is creating an understanding of the social and biophysical system contributing to land degradation and restoration. This dual outcome (i.e., knowledge on land degradation problems and response objectives) feeds into the next phase (Table 8.3).

From Planning & Design to Implementation & Management

To identify possible land degradation response strategies, tools that incorporate knowledge and information on social and ecological processes are needed. In the Planning and Design phase, response options are selected based on an assessment of financial and social capital. Financial capital includes total costs, return on investment (Goldstein *et al.*, 2008) and options for financing mechanisms (Jack *et al.*, 2008). Integrated land management is based on the idea that coordinated planning and action can be more effective than disparate, uncoordinated actions of individual land managers in delivering the full complement of benefits expected from strategies to halt or reverse land degradation. Integrated land management therefore requires strong stakeholder collaboration and engagement, which makes an assessment of social capital necessary (Brondizio *et al.*, 2009). Ex-ante impact assessment tools have a function in highlighting synergies and trade-offs between different locations, ecosystem services supply and stakeholder interests (Rosa & Sánchez, 2016). Impacts of land degradation response actions can affect different groups in society in different ways, and insights on these potential impacts contribute to reduced human conflicts and improved benefit-sharing (Daw *et al.*, 2011). Formalized ex-ante impact assessments to support decision-making include environmental impact assessments (EIA) and strategic environmental assessments (SEA). These two tools have overlapping conceptual foundations (Bina, 2007). In many countries these assessments are part of legislation, but are conducted within a wide range of quality levels (Pope *et al.* , 2013). In impact assessments of urban development the impact on land consumption is rarely taken into account, yet it is highlighted as a serious pressure on landscapes worldwide (Nuissl *et al.*, 2009). At the moment the suite of available generic decision support tools to support the selection of a land degradation response strategy mostly focus on biophysical impact, however participatory scenario planning is an increasingly popular tool in place-based environmental research for evaluating alternative futures of social-ecological systems (Oteros-Rozas *et al.*, 2015). An assessment of trade-offs and synergies among ecosystem services beneficiaries and stakeholders (to select socially-feasible solutions) is currently based on case-specific surveys (Karrasch *et al.*, 2014), economic valuations (ELD Initiative, 2015), or not included at all in impact assessments.

From Implementation & Management to Evaluation & Adaptation of the Agenda

To support good long-term governance and technical management of the implemented restoration and mitigation actions, information, knowledge and competencies are needed (Table 8.3). Due to the multiple dimensions of land degradation and response actions, a multi-sector, urban-rural, multi-level, adaptive governance system is most effective (Brondizio *et al.*, 2009; Gómez-Baggethun *et al.*, 2013; Kenward *et al.*, 2011; OECD, 2016). Based on early defined indicators (in the Agenda-setting phase) a monitoring strategy is put in place to allow for evaluation and adaptation of the response strategies. Monitoring begins before the implementation of response actions. Systematic monitoring of the implemented response activities is vital for designing new (or adjusting) activities and policies. Monitoring information of land degradation response interventions is scarcely available, due to a lack of standardized monitoring strategies and adequate baseline information. Some long-term monitoring initiatives exist. These include: the Millennium Villages (Chapman *et al.*, 2016); the Long-Term Ecosystem Research (LTER) network (Stoll *et al.*, 2015); the GLORIA network (Mark *et al.* (2006)) (see also Box 8.6); and Group on Earth Observations Biodiversity Observation Network (GEOBON) (Proença *et al.*, 2017); and the UN Convention to Combat Desertification (UNCCD) which has – with the Land Degradation Neutrality Targets – defined and provided indicators to establish a baseline to allow for tracking land degradation over time. Local stakeholder-based monitoring approaches include ground-based photo monitoring (Lassoie *et al.*, 2014) and participatory monitoring programs (Kusters *et al.*, 2017; Singh *et al.*, 2014).

Besides monitoring, retrospective assessments of restoration interventions are carried out to evaluate restoration actions which were implemented without monitoring schemes. For example, a meta-analysis of 70 experimental studies (Meli *et al.*, 2014) showed that in wetlands restoration, effects on biodiversity recovery and ecosystem services recovery depended on the following factors, listed in order of decreasing importance: main cause of degradation; restoration action; experimental design; and ecosystem type. Restoration age did not significantly affect restoration outcomes in their meta-analysis.

Decision support tools to guide the evaluation and adaptation of decisions on halting and reversing land degradation are lacking in the scientific literature. This relates to the often-unknown thresholds leading to sudden non-linear ecosystem regime shifts (see also Chapter 4). Preventive and rapid actions are often required before the undesired and irreversible regime shift occurs. Long-term monitoring and adaptive decision-making can be jeopardized by the much shorter political life cycles of elected representatives in democratic regimes (see also Chapter 2).

Table 8 3 Illustrative decision maker questions in relation to information and knowledge outputs the support the next decision phase.

PHASE	DECISION MAKER QUESTIONS	ADDRESSED IN CHAPTER	AVAILABLE TOOLS	INFORMATION AND KNOWLEDGE OUTPUT NEEDS PER PHASE
Agenda setting	What is the current state of the land, biodiversity and ecosystemt services?	1,3,4	• Degradation assessment tools, 8.2.1	• Quantified land, biodiversity, ecosystemt services indicators • Knowledge built
	What is the demand for land, biodiversity and ecosystemt by different stakeholder groups?	1,2	• Surveys, negotiation, facilitation tools	• Quantified and agreed demand indictors • Knowledge built
Outcomes		Land degradation problems identified, located and understood Land degradation responses objectives set		
Planning and Design	What interventions options and locations could address social needs?	1,6,7	• Option and location screening, 8.2.2	• Response options • Response design • Ex-ante assessment social needs
	What is the impact of the interventions on land, biodiversity and ecosystemt services?	1,5,7	• Impact assessments, 8.2.2 • Policy inter actions, Section, 8.4	• Ex-ante assessment land, biodiversity, ecosystemt services
Outcome		Selected land degradation response strategy		
Implementation and Management	How to best implement and manage and intervention?	1,6	• Instruments and competencies, 8.3	• Technical planning • Roles and responsibility governance actors
	How to the measure impact to support adaptive management?	6	• Instruments and competencies, 8.3	• Monitoring land, biodiversity, ecosystemt services indicators • Evaluating objectives • Communication
Outcome		Adaptive management to halt and reverse land degradation		

Seamless use of information, knowledge and tools

Conceptual frameworks on integrated environmental decision-making, including land degradation responses, exist (Cowling *et al.*, 2008; Hessel *et al.*, 2014; Reed *et al.*, 2010b; Scherr *et al.*, 2014) and the FAO is one of the institutions to have set up general guidelines to guide their implementation (FAO, 2015a). However, to apply these to the geographic, cultural, political, economic and historical contexts in different countries and regions, location-specific tools are needed. The current knowledge, information and tools base cannot seamlessly provide evidence-based decision support throughout the decision-making process. With seamless use, we mean a technical, conceptual and operation linkage between outputs and inputs of decision support tools for each decision-making step. This does not mean that successful decision-making on land degradation responses do not exist (see examples in Chapter 1, and Boxes in this Chapter).

To improve information, knowledge and tool use throughout the policy cycle, knowledge and information outputs (Table 8.3) need to be adequately generated. This could be done by cross-disciplinary and multi-actor collaboration, in order to tune research efforts and cross-sector harmonization. Also this could be achieved by encouraging scientists and leaders in government, businesses and civil society to work more closely together to develop the knowledge, tools and practices necessary to integrate social-ecological interactions into decision-making (Guerry *et al.*, 2015). However, there is limited evidence on when scientific tools are used in decision-making, as many factors influence actual uptake, including, but not limited to, relevance for policy objectives, time and cost cost-effectiveness, usefulness in case of missing data (Gibson *et al.*, 2017; McIntosh *et al.*, 2011; Zasada *et al.*, 2017). Decision-making is about more than having access to and using information, knowledge and tools. A range of institutional competencies are needed to support land degradation and restoration decision-making. These are addressed in detail in Section 8.3.

Box 8 6 An example of governance halting malpractice and land degradation but with no immediate solutions for restoration. A case study from the rangelands of New Zealand.

Lying relatively isolated in the temperate region of the Southwest Pacific, New Zealand was first settled by Polynesian Maori. Their main influence was through fire, which increased dramatically from the rare natural fires, and was a major factor in increasing the extent of grassland. By the time Europeans settled in the 1840s, forest cover had decreased from ~75% to ~50%, largely at the expense of tussock (bunch) grassland cover that increased to ~31% (82,436 km^2; Mark & McLennan 2005). Of this, some 54% still remains, mostly in the uplands. All of the remaining indigenous grasslands have been modified to varying extents through the effects of pastoral farming (burning and domestic grazing) on the more accessible rangelands and feral herbivores on the remainder, mainly deer introduced for hunting. Erosion was probably a feature of upland landscapes ahead of pastoral farming (Whitehouse, 1984), but degradation increased as a result and ranged from a drastic reduction in above-ground biomass through replacement of the tall tussock cover by a mixed short turf or herb field of grazing-tolerant grasses and forbs, and greatly increased bare soil and consequent erosion. In the more remote, non-rangeland regions, displacement of tussock cover was less serious. The extent of degradation was also related to the basement rock, as well as to variation in the topographic factors of elevation, aspect and slope.

Official responses to rangeland degradation.

The government took legislative action to address the situation in 1948 with an amendment to much earlier legislation, to provide much greater security for the pastoral use of the government-leasehold high-country tussock lands. Previously, the leases were reviewed at 11-year intervals with no right of renewal; such insecurity clearly encouraged unrestrained resource exploitation. The amendment carried some discretionary management constraints, but significantly provided a formal right of lease renewal at 33-year intervals, with “the same conditions and provisions as the original lease”— clearly offering absolute security of tenure. Despite the amendment, with continued deterioration in the rangelands condition and carrying capacity, the government established regional catchment authorities in the early 1950s. There was also provision of central government subsidies for improving both land management and access. Seasonal or intermittent spelling from grazing, although generally considered desirable, was only rarely practiced - even as a component of post-fire management. Retirement of land deemed to be unsuitable for sustained pastoralism, was usually a condition for subsidized assistance though relinquishment of the lease on this land, also a condition, but was rarely enforced.

Many of the discretionary constraints exercised by the local authorities on management and development activities were equivocal - there was a predominance of farmers among the elected members of the catchment authorities – and the continued degradation has been recorded in a long series of scientific papers from the 1860s onward and throughout the 20th century (see Allen *et al.* 1994; Buchanan 1865; Mark 1994; O’Connor 1982, 1984). During this time, a special committee on senior government ecologists was established to report to central government on the ecological basis for degradation of the upland snow tussock grasslands of the South Island pastoral lands. This committee recommended that research be carried out on both the systematics and ecology of the dominant tussock species and their communities, including the roles of introduced plants and animals. Several such studies were initiated, including those leading to separation of the effects of rangeland burning from those of grazing. Spelling for one (and preferably two) seasons following a management fire has therefore been recommended and now generally adopted, at least for the first post-fire season. Combined with these studies were the first measurements of upland water yield. Maximum yields among a wide range of cover types, including bare soil, came from the tall tussock grassland: 63–80% of measured precipitation (1300–1400 mm p.a.), varying mainly with frequency and intensity of fog. Yields from all other cover types were significantly less, but those from burned and clipped tussocks increased as they recovered (Holdsworth and Mark, 1990; Mark and Dickinson, 2008). Subsequent controversy over the contribution of fog interception by the tall fine tussock foliage was largely resolved with a stable isotope study (Ingraham *et al.*, 2008; Ingraham and Mark, 2000).

Concerns with the degraded state of the South Island rangelands continued and resulted in the establishment of a Ministerial appointed High Country Review Committee in 1994, to which the New Zealand Ecological Society and New Zealand Society of Soil Science made a comprehensive joint submission (Allen *et al.*, 1994). The Committee concluded (Party, 1994) that “a decline in soil condition is very likely on the unimproved lands. These lands comprise approximately 80% of the land area of the pastoral high country and receive no inputs. In the long term, the pastoral use of extensive areas of the South Island high country is unlikely to be sustainable.” Largely in response to this report, the government initiated a review of the tenure of these high country rangeland leases in the mid-1990s, under special legislation (Republic of New Zealand, 1998). This provided for lessees to negotiate freehold title for the more productive, generally lower elevation lands in exchange for relinquishing their tenure on the more vulnerable, degraded and generally higher elevation lands with significant ecosystem services (Mark *et al.* 2013) and heritage values – as well as indigenous ecosystems and biodiversity, soil conservation, landscape, recreation, eco-tourism and water production values - for management in the public interest by the Department of Conservation. This is an ongoing process, described up to April 2012 by Mark (2012) and now (January, 2017) there remain the same ten conservation parks totaling 581,032 ha and five whole-property purchases (128,792ha), with 119 of the original 303 leases now reviewed, totaling 623,413ha: 53% allocated to freehold and 47% to conservation and with 48 leases at various stages of the process.

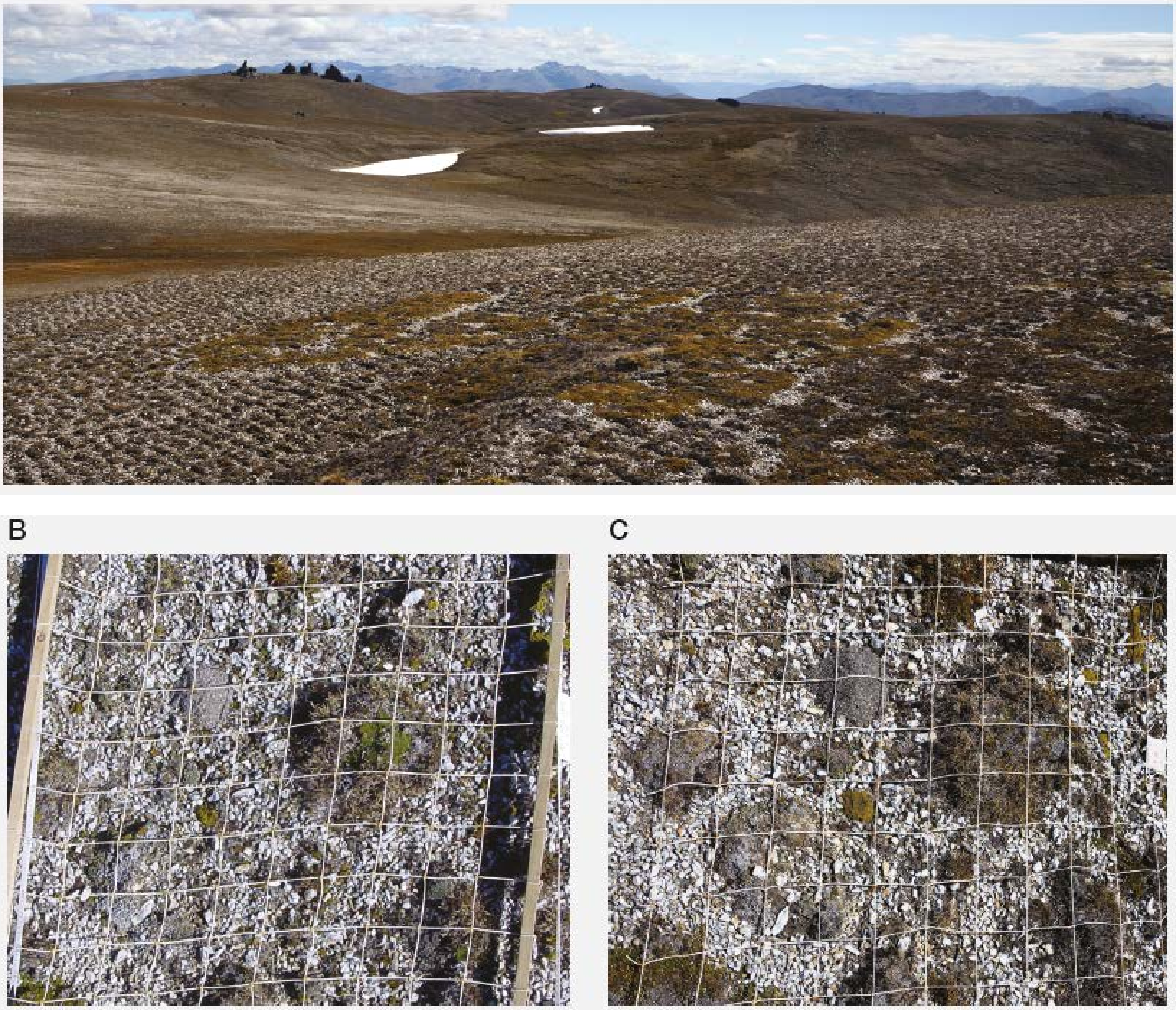
The long-term monitoring within the GLORIA network (http://www.gloria.ac.at/network_gloria_longterm.html, and Mark *et al.* (2006) revealed very slow recovery rates of degraded upland ecosystems. For example, the crest of the Pisa Range was used for extensive merino sheep summer grazing, in combination with intermittent burning, until 2012 when this detrimental land-use practice was brought to a halt through tenure review. Allen *et al.* (1994) quote an average soil organic matter recovery rate at 35 years, but for heavily-degraded areas such as the Pisa Range, the rate is even slower than that with the recovery rate of cushion plants (and subsequent soil formation) being no more than 5% over a decade. The photos below show hardly any vegetation recovery, between 200 and 2014.

There is, therefore, no “easy fix” as to restoration methods of the upland grasslands, besides just conserving areas and facilitating their return to the original conditions at their own pace. Recovery of these degraded ecosystems was a very slow process because of the prevailing environmental conditions.

Figure 8 3 **Severely degraded alpine vegetation on the crest of the Pisa Range, south-central South Island, New Zealand.**

A Overview of the Pisa Range crest, B and C Close-ups of a permanently marked 1 m² plot on the Pisa crest GLORIA site, in late summers (February) of 2003 B and 2014 C, respectively, visualizing the very slow rate of vegetation recovery. Grey cushions are *Anisotome imbricata* (Apiaceae), dark green ones are *Dracophyllum muscoides* (Ericaceae), and more bright green ones being *Colobanthus buchananii* (Caryophyllaceae). Photos by Ulf Molau (A & C) and Katharine Dickinson (B).

A



8.3 Building institutional competencies

Institutional competencies are a cornerstone for developing and implementing policy strategies, yet are often overlooked as drivers for ecosystem restoration or degradation. Well thought through decision-making processes regarding environmental issues – which forms part of environmental governance – and resulting environmental policies are paramount to halt and reverse land degradation. Institutional environmental policies define how and when to mitigate ecosystem degradation or regulate the use of natural resources (Ostrom, 2009). Institutional competencies are the set of abilities which a given institution can use to achieve those policy goals. Examples include the ability to collaborate with local communities, support the design of scientifically-sound restoration interventions, or foresee possible undesired secondary effects of policies. Besides organizational mandate and informal organizational practice, institutional competencies are an inherent part of the design and implementation of policy instruments and may even act as important drivers of land degradation or restoration (Ferraro & Hanauer, 2014; Guariguata & Brancalion, 2014; Primmer, 2017).

Institutional competencies support the successful design and implementation of policy instruments. Land degradation and restoration issues are typically driven by a complex interaction of drivers operating at different spatial and temporal scales. While rapid variables such as contamination events can cause sudden ecosystem degradation, slow variables are underlying structural processes - such as law implementation - that sets the scene for land degradation and restoration strategies (Reynolds *et al.*, 2007). Institutional competencies are a slow variable that indirectly triggers environmental degradation and/or the implementation of sound restoration strategies. Building the right institutional competencies is therefore as important as for instance providing financial incentives to promote land restoration. Robust science to evaluate the impact and efficiency of different institutional competencies in relation to land degradation and restoration decision making is still in its infancy stages (Guerry *et al.*, 2015; Polasky *et al.*, 2011) and is often hampered by significant time lags between policy implementation and measurable outcomes. For now, decision makers can learn from examples of how instruments are used in diverse contexts, such as the impact of institutional polices on biodiversity conservation or the provision of crucial ecosystem services such as carbon sequestration. For example, Prager *et al.* (2012) provide a framework describing key considerations for effective institutional and governance response that include the type of policy measure, specific institutions and governance structure. In this section, we aim to assess institutional competencies to design and implement: (i) legal and regulatory; (ii) rights-based and customary; (iii) economic and financial; (iv) social and cultural; (v) science-based instruments; and (vi) their selection and combination of instruments to support decision-making to avoid and reverse land degradation. We use examples from diverse decision-making contexts, specifically examples that relate to Sustainable Development Goals' targets, to illustrate these competencies.

8.3.1 Competencies for legal and regulatory instruments

8.3.1.1 Strengthen the implementation of legal and regulatory instruments

Institutional competencies create favourable conditions for the implementation of the law and regulations. Goal 16 (Peace, Justice and Strong institutions) of the Sustainable Development Goals specifies the need to develop conditions for access to justice and the law, as well as to encourage representative and participatory decision-making.

The development of conditions for access to justice and law

SDG Target 16.3 states: "promote the rule of law at the national and international levels and ensure equal access to justice for all." Access to justice can contribute to combating land degradation and developing restoration solutions. Sustainable land use could be safeguarded by access to justice to secure land-use rights, especially those of indigenous peoples. It has been recognized that the major part of the natural resources coveted for commercial purposes such as minerals, forests and oil are in territories used or occupied by indigenous peoples (UN General Assembly, 2013). At the same time, it is recognized (CIPTA & WCS, 2013; Jonas, 2017; Nelson & Chomitz, 2011; Porter-Bolland *et al.*, 2012; Schabus, 2017) that, under certain conditions (e.g., Ostrom 1990, 2000), "areas and resources under the governance and/or management of indigenous peoples and local communities are more effective than strictly protected areas at preventing deforestation, maintaining forest health and ecosystem connectivity, and conserving biodiversity and natural resources" (Jonas, 2017). Actions leading to the legal protection of the traditional land management methods can be favourable to address land degradation problems and develop restoration solutions. However, in this case, the question of how to secure land-use rights arises (see Sub-section 8.3.2.1) and more specifically the recognition of the rights of indigenous peoples and local communities (Schabus, 2017) (see also Chapter 5). Also, as Oliver de Schutter noted: "there is no reason not to extend the recognition of communal rights beyond indigenous or traditional communities particularly where the management of common pool resources at the local level proves effective" (De Schutter, 2010).

However, access to justice is inseparable from access to law, as pointed out by SDG Target 16.10: "ensure public access to information and protect fundamental freedoms, in accordance with national legislation and international agreements". In other words, actors must first know their rights in order to initiate legal proceedings. This requires improving the transparency of institutions at different levels of decision-making, as targeted for in SDG 16.

The other side of this access to justice and law is to know legal obligations and prohibitions. These obligations and prohibitions must be accompanied by dissuasive sanctions in the event of non-performance, but also by strong institutions capable of enforcing them. This involves police and judicial institutions (SDG Targets 16.4 and 16.5). In addition to the fact that this corruption impedes the implementation of legal instruments designed to protect the environment, it also has a cost: "corruption, bribery, theft and tax evasion cost some $1.26 trillion for developing countries per year" (Kar & Freitas, 2011: i). Law enforcement problems are generally related to a lack of political willingness or high levels of corruption. Corruption, in particular, reduces the financial resources for land degradation response actions. It also hampers adequate project evaluation as reported results could inflate successes and omit failures (Langseth, 1999). Independent and sufficiently paid project coordinators and evaluators could reduce this risk.

Law enforcement and regulation are key in successfully halting and reversing land degradation. Irreversible ecosystem regime shifts towards non-desired states can be avoided by regulating or prohibiting activities such as the over-extraction of natural resources, uncontrolled pollution and/or land-use changes. While top-down law enforcement may be efficient in countries with strong and well-equipped institutions, regions with reduced social, physical and financial capital may suffer high levels of impunity. In the latter case, local surveillance brigades, regulation mechanisms and justice systems can contribute to monitoring systems and reduce the prioritizing of individual benefits over the conservation or restoration of the ecosystem (Karp, 2011). Local fire brigades (Beringer, 2000) and poaching surveillance teams are good examples of local (and often informal) initiatives to reduce ecosystem damage.

However, while it is important to provide legal access to justice, other dispute-resolution processes that also contribute to social peace should not be excluded (Frison-Roche, 2012). In other words, local social dispute-settlement mechanisms should not be overlooked. More generally, the role of local actors in the decision-making process should not be overlooked at all decision-making scales. This is discussed in the following Sections.

Representative and participatory decision-making

SDG Target 16.7 states: "ensure responsive, inclusive, participatory and representative decision-making at all levels". Consequently, different types of actors need to be involved in international and national decision-making, including companies, NGOs, indigenous peoples and local communities.

At the international level, the need for inclusive decision-making can be illustrated with intergovernmental decision-making to combat climate change. This top-down approach is sometimes considered a failure, because of the weak administrative capacity to monitor and enforce global greenhouse gas emission standards, and the strong debates between nations about the responsibility of those who benefited from climate-destructive practices (Orts, 2011). Also, the relocation of high-emission activities to less regulated places is a challenge for global regulation. The solution to intractable problems like climate change and land degradation can be found in new modes of global environmental governance (Orts, 2011). This has resulted in the opening up to other governance bodies such as cities, businesses, NGOs and universities, a phenomenon also known as "global assemblages" (Sassen, 2006). International law and international environmental governance gradually opens up to transnational law and not only intergovernmental law (Maljean-Dubois, 2003). These new modes of global environmental governance involve ensuring and strengthening the representation and participation of key players in international decision-making bodies. For this purpose, UNEP ensures that the voices of indigenous peoples are heard and taken into consideration in the development of programs at the local, regional, national and international levels. This consideration is formalized within the framework of the UN Declaration on the Rights of Indigenous Peoples of 13th September 2007. Indigenous groups operate at an international level through the UN Permanent Forum on Indigenous Issues (UNPFII), the Convention on Biological Diversity International Indigenous Forum on Biodiversity (IIFB) and the United Nations International Indigenous Forum on Climate Change (UNIFCC),

Nation-level participatory decision-making is also promoted. Concretely, this implies opening up spaces for participation at the local level (Box 8.7), but also ensuring the effective participation of local actors in the decision-making process.

Box 8.7 Competencies for including local participation in legal conservation actions.

Sacred groves are relict forests conserved through reverence. They are found in many parts of the world across Asia, Africa, Europe and North America. In India the sacred groves are predominant in the Eastern-Ghats, Western-Ghats and Northeast region. In order to protect and conserve relict and keystone species of ecological importance many countries in the Asia Pacific region have been conserving sacred groves through traditional practices. Many keystone species are conserved by rural and tribal communities, who believe that these trees are abodes of their deity or spirits of their forefathers. The sacred groves have helped in many places to maintain the water table. The sacred groves have been conserved through traditional mechanisms and responsibility of protection and maintenance of the area of sacred grove is passed from one generation to the next through traditional means. Due to the dilution of traditional beliefs and increasing disinterestedness within the tribal youth, these sacred groves are gradually losing their importance and are shrinking in area. The Indian Biodiversity Act of 2002 aims to address this issue. A mechanism under this Act facilitates the recognition and formalization of a sacred grove as a Biodiversity Heritage Site (BHS) which are maintained by a legal body, namely the Biodiversity Management Committee (BMC); responsible for conserving bio-resources in a Panchayat. There is a need for further policy initiatives in the Asia Pacific region to recognize and conserve these sacred groves. The Indian mechanism could be replicated in the countries of Asia Pacific region that are signatories to the CBD, for example (Chandrashekara & Sankar, 1998; Ramakrishnan *et al.*, 1998).

Ensuring the effective participation of local actors in the decision-making process (Tamang, 2004) could offer some win-win solutions (De vente *et al.*, 2016), such as increases in local support for biodiversity conservation laws and policies and enhancements in the capacity to adapt to shifting environmental conditions (Thériault, 2011). Inclusive decision-making can also lead to legal reforms. For example, in Canada, the "free-entry mining" principle (on which most mining regimes are based) finds its direct origins in the practices and customs established by miners in the context of the 19th century's gold rushes (Barton, 1993). This principle entails a prioritization of mining activities over other uses of the land and may be fathomable in its historical context. However, there is a growing consensus that it has become at odds with competing priorities and values regarding environmental protection, social acceptability and respect for Aboriginal rights (Bankes, 2004; Campbell, 2004). Yet, Aboriginal peoples' constitutional rights have influenced recent mining governance reforms in Quebec and Ontario, in particular with the duty to consult and accommodate Aboriginal people (Thériault, 2013). On a global level, the International Council on Mining has developed a good practice guide for working with indigenous people (ICMM, 2015).

However, strengthening institutional competencies to ensure the implementation of legal policy instruments requires not only that the instruments exist, but also that the design of these instruments is relevant.

8.3.1.2 Design or improve legal and regulatory instruments

The improvement or design of legal tools applicable to land degradation and restoration decision-making relates to the gradual evolution of environmental law. In order to develop strong environmental law, institutional competencies are needed to help to strengthen existing legal tools. This is particularly the case for environmental impact assessment, a critical tool for making informed decisions. As for the design of legal instruments, it is important that it be guided by legal principles that favour a better consideration of the environment.

Improving existing tools: the key role of environmental impact assessment

In many countries, environmental impact assessments (EIA) are carried out before a project, plan or programme is implemented and constitutes an essential legal decision-making tool that typically leads to the authorization of the plan, the project or the programme in question (Wood, 2003). The "avoid-reduce-compensate" sequence helps to reinforce the consideration of the environment by specifying the nature and sequence of the data to be included in an impact study. This focuses chiefly on assessing the environment and, as such, omits the impacts of proposed developments on the culture, societies and ways of living of indigenous peoples and local communities. That explains the role of the Akwé: Kon Guidelines (adopted by the Convention on Biological Diversity) to conduct cultural, environmental and social impact assessments regarding developments on lands and waters traditionally occupied or used by indigenous and local communities (Schabus, 2017).

The "avoid-reduce-compensate" sequence can be improved in several ways to address land degradation challenges. The first possible improvement is to give a better visibility to the combat against land degradation and the development of solutions for its restoration by avoiding, reducing and offsetting land (UNCCD, 2013) (see also Chapter 2 for examples).

Secondly, another possible improvement is to clarify the "avoid-reduce-compensate" sequence as one of the elements of the environmental assessment, in particular regarding offsets. Indeed, the use of compensation in the land degradation decision-making must be properly understood. Offsetting can give the illusion of effective prevention of land degradation. More broadly, there is a risk of giving the illusion that economic development can nearly always be reconciled with environmental protection. Indeed, compensation can be an ecological solution since the degradation of one land will be compensated by the restoration of another, despite the fact that, in any event, land will ultimately be degraded. This is what conditions compensation. Moreover, it is based on a short-term logic since it assumes that there will always be a quantity of land available to compensate for past degradation. However, efforts can be made to prioritize the different steps of the avoid-reduce-compensate sequence. This sequence has been clarified, for example in the 2016 French Biodiversity Law by clearly hierarchizing the different stages, so that the compensation appears at the end of the sequence after the measures of avoidance and reduction of environmental damage (Article L. 110-1 French environmental code). The hierarchy of the different stages of the avoid-reduce-compensate sequence is an important evolution, but could be considered insufficient. In French legislation, maintaining and integrating the avoid-reduce-compensate sequence as a whole, into the principle of prevention, offers a paradoxical message: it suggests that compensation is a matter of prevention while the damage has in most cases already occurred (Martin, 2016).

Environmental impact assessment is an application of the principle of prevention defined in Principle 17 of the UN 1992 Declaration on Environment and Development. However, different environmental principles can be instruments of environmental law to guide states in the development of new policy instruments relevant to land degradation and restoration decision-making.

The design of new environmental tools: the role of environmental principles in order to take better account of the environment

Environmental principles are specific legal instruments insofar as they govern environmental action at the international, but also national level. These environmental principles can be integrated into the national law of states through the application of international law, but also through a reciprocal influence of the national laws. Environmental principles guide the writing of environmental law by contributing to coherence between environmental interests and other interests, but also by participating in the structuring of environmental law.

Many principles have already been developed and recognized internationally in the context of various international conventions and declarations (for example, Declaration of the UN Conference on the Human Environment, 1972 or the Rio Declaration on Environment and Development, 1992). Examples include the polluter-pays principle and prevention principle. More recently, the precautionary principle or the principle of public information and participation in decision-making has also been applied. In general, these environmental principles are now recognized as common references for the international community. Indeed, by virtue of being enshrined in international texts (even non-binding ones) these principles have acquired legal value. These same principles can also be incorporated into binding international conventions. In this case, the principles become binding to the country Parties of these conventions. For example, the Convention on Biological Diversity (CBD) includes the precautionary principle. Several principles also guide the parties to the UN Convention to Combat Desertification (UNCCD) "to achieve the objectives" of the convention and "to implement its provisions" (Article 3). These principles include, for example, the participation of local populations in the design and implementation of programmes to combat desertification and/or mitigate the effects of drought (Article 3a). It includes establishing "cooperation among all levels of government, communities, non-governmental organizations and landholders to establish a better understanding of the nature and value of land and scarce water resources in affected areas and to work towards their sustainable use" (Article 3c) (Chasek *et al.*, 2011; Stringer *et al.*, 2007).

So, to improve their state legislation, state institutions must have the necessary institutional competencies to participate in the elaboration of these international texts and above all to be able to respect these international conventions (Box 8.8). UNEP is helping countries to meet commitments under multilateral environmental agreements and to integrate environmental concerns into national plans and strategies.

Box 8.8 Local environmental regulation in Pará, Brazil

In order to ensure the effectiveness of environmental regulation, the establishment of international standards is considered an important step. However, it is also established that the ultimate responsibility for the implementation of the regulations rests with the actors on site. These actors include state institutions which, in the name of national sovereignty (Hashmi, 1997; Willmore, 2017), may or may not adhere to international conventions for the protection of the environment and/or draw-up national environmental legislation. Other actors include citizens, NGOs and local public officials (Arnaud, 2014; Giddens, 2009).

Here the example is the case of halting deforestation in the Brazilian Amazon. In 2008, the offices and cars of the Brazilian Institute for Environment and Renewable Natural Resources (IBAMA) were burned down, because the institute's work had led to the closure of ten sawmills in Paragominas, State of Pará. The extraction of timber was the source of income for half of the inhabitants of this town. A pact between public administration, citizens, rural producers and businesses helped to relax these tensions. In 2011, the city of Paragominas adopted a municipal environmental code and created a council composed of representatives of the public administration and civil society to develop the project of living in a green municipality, where forest products would benefit from environmental certification. This project also appealed to companies. In a second stage, the municipality, with the help of NGOs, also instituted monitoring mechanisms to ensure compliance with the municipal environment code, in particular the ban on transforming indigenous trees into coal.

There are principles that do not fall within the scope of an international reference system, that is to say an internationally-shared vision by all states. In that case, a principle can be defended at the national level by a state. An example of this is the principle of ecological solidarity; an unprecedented principle adopted by French legislation since 2016 (see Chapter 2, Section 2.2.1.2). Looking at other national laws can be an important source of inspiration for other state legislations, but also for regional or even international conventions.

There are several ways of developing and integrating environmental principles that consider human relationship with land into environmental law. The adoption of these principles can guide decision-makers in the choice of relevant policy instruments to be developed. In particular, they contribute to the establishment of a coherence between environmental interests and other interests, but also among environmental interests (e.g., as called for in SDG 17). In addition, these principles structure environmental law and contribute to its progression while leaving room for decision makers to manoeuvre.

Firstly, environmental principles contribute to the development of environmental law by bringing environmental interests in line with the objectives of halting or reversing land degradation, together with sustainable development. This is particularly the purpose of the principle of integration. According to Principle 4 of the Rio Declaration on Environment and Development (1992), “in order to achieve sustainable development, environmental protection shall constitute an integral part of the development process and cannot be considered in isolation from it". Land-use planning and management is a practical way to achieve it (see also Chapter 6). The principle of integration can also be reflected in the promotion of integrated management, such as the “ecosystem approach” (Chapter 1). The ecosystem approach also concerns the principle of participation, as it underscores the need to understand and factor in societal choices, the rights and interests of indigenous peoples and local communities, and have inclusive decision-making (Morgera, 2017). In addition, the ecosystem approach concerns the precautionary principle, because of the lack of our knowledge and uncertainties in ecosystem functioning, further highlighting the need for adaptive management (Armitage *et al.*, 2009; Morgera, 2017). In this case, adaptive management is understood as a "new legal paradigm" (Tarlock, 2007).

Coherence is necessary within environmental objectives, themselves, and not just between the objectives of the various pillars of sustainable development. Further emphasis is placed on land in SDG Target 15.3: "by 2030, combat desertification, restore degraded land and soil, including land affected by desertification, drought and floods, and strive to achieve a land degradation-neutral world". Goal 15 and its targets must therefore consider many different environmental goals. While legal solutions can be sought to identify synergies, they often also involve combating the fragmentation of the law regularly accompanied by institutional fragmentation (Maljean-Dubois, 2003). This requires water laws to interact with land and climate laws (Reed & Stringer, 2016). Dialogue must also take place between each corresponding institution (Stringer *et al.*, 2009). Indeed, institutional compartmentalisation can lead to inconsistencies or conflicts in policy proposals, between the authorities and in the missions of the institutions.

Secondly, environmental principles are tools to structure environmental law and the development of policy instruments without imposing a given instrument. In other words, they help to establish a strategy for environmental law in which land degradation and restoration decision-making must be implemented without pre-defining or pre-establishing any particular legal instrument. Depending on the place-specific needs, it can be a set of legal instruments adapted to the situation at a given moment.

8.3.2 Competencies for rights-based instruments and customary norms

This sub-section provides examples of institutional competencies that contribute to the effectiveness of a human rights-based approach to strategies to address land degradation problems and develop restoration solutions. In this context, it focuses particularly on competencies that have proven to be useful in processes and procedures aiming at securing land rights (see Section 8.3.2.1), in those aiming at advancing the enjoyment of a clean and healthy environment (see Section 8.3.2.2) and in those aiming at fostering the respect of customary norms (see Section 8.3.2.3).

8.3.2.1 Securing land rights

Secure rights to land of rural communities (indigenous and non-indigenous) and of their members are considered as an essential contribution to the realization of human rights such as the rights to adequate food, water, health and housing; even though a human right to land has not yet been recognized in international human rights law (UN High Commissioner for Human Rights, 2014; UN Human Rights Office of the High Commissioner, 2015). Secure land rights are also inextricably linked to land degradation and restoration issues (UNCCD CSO Panel, 2017). At the same time, the human rights principle of participation in decision-making plays an essential role for securing land rights and in the responsible governance of land and natural resources (FAO, 2012; UN Human Rights Office of the High Commissioner, 2015). Institutional competencies for the development of effective participatory processes are hence a core element of land tenure security and policy responses regarding land degradation, as shown in the following examples in Box 8.9.

Box 8.9 Securing land rights in Colombia and Costa Rica.

In Colombia, Article 58 of the Constitution recognizes that property, as a social function that implies obligations, has an inherent ecological function. In this context, a participatory approach has been adopted in the procedure for expanding the territory of indigenous reserves (“*resguardos indígenas*”). As part of this procedure, compliance with the ecological function of property in an indigenous reserve is verified and certified by national authorities, as a legal requirement for delivering the authorization for the reserve’s expansion. Acknowledging its intrinsic relation with the physical and cultural survival of those peoples, the ecological function of property in indigenous reserves has been seen as an opportunity for crossing different views over a territory and so, as a tool for facilitating dialogue between different disciplines and worldviews. As a means for enhancing institutional competencies for this facilitation purpose, it has been proposed that agencies in charge of the procedure put together multidisciplinary, multiethnic and interinstitutional teams (Londoño Toro *et al.*, 2004).

In Costa Rica, a participatory process initiated in 2005 has enabled taking important steps both for securing land rights of members of the non-indigenous communities that live inside the Ostional Wilderness Refuge and for perpetuating the management and conservation of sea turtles programme developed by the Ostional community. This process was formalized in 2008 with the creation of the Interinstitutional Advisory Council of the Ostional Wilderness Refuge (CIMACO). The enactment, in 2016, of the Ostional Wilderness Refuge Act, which authorized granting 25 years renewable concessions to members of the communities inside the Refuge, has been to a great extent possible thanks to this participatory process that succeeded in ending conflicts between different stakeholders in the Refuge through dialogue. Institutional competencies have been very important for driving this process: first by gathering all the interested parties in the Refuge around the discussion table; second by breaking the deadlock at a moment where no consensus could be reached, hiring the support of specialists in alternative dispute resolution, facilitation and social mediation; and third by building up a solid scientific base for decision-making support (Brenes Chaves & Cedeño, 2017).

8.3.2.2 Advancing the enjoyment of a clean and healthy environment

After recognizing that no global agreement explicitly establishes a right to a healthy (and other related adjectives) environment, the Independent Expert of the United Nations on human rights related to the enjoyment of a safe, clean, healthy and sustainable environment, identified two aspects of the relationship between human rights and the environment as “firmly established” in a first report to the United Nations Human Rights Council (United Nations Human Rights Council, 2012). First, that “environmental degradation can and does adversely affect the enjoyment of a broad range of human rights, including rights to life, health, food and water” (see also UNEP, 2015). Second, that “the exercise of certain rights can and does benefit environmental policymaking, resulting in better environmental protection and, as a consequence, greater protection of human rights that may be threatened by environmental degradation”. He was referring specifically to the following procedural rights: rights of expression and association, the rights of information and participation, and the rights to remedy. In 1992, Charles Alexandre Kiss (a French environmental law pioneer) stressed the need to ensure that everyone and all human groups have adequate procedures to protect "their" environment, which is often shared with others. Kiss described this right to the environment as a procedural right of an individual, or an individual right to protect the environment. Concretely, Kiss explains that the constitutional provisions that impose on the State to protect the environment are generally formulated, whereas procedural law obliges the public authorities to intervene in concrete situations, on individual complaints (Kiss, 1992, 1993, 2004).

In a second report to the Human Rights Council (United Nations Human Rights Council, 2013), the Independent Expert identified three procedural obligations that human rights law imposes on States in relation to environmental protection: duties to assess environmental impacts and make information public, duties to facilitate public participation in environmental decision-making and the duty to provide access to legal remedies. The Independent Expert also identified the following three substantive obligations: (i) the obligation to adopt and implement legal and institutional frameworks; (ii) obligations to protect against environmental harm from private actors; and (iii) obligations relating to transboundary environmental harm. According to a 2012 survey, there are at least 92 countries that have granted constitutional status to this right and 177 countries recognize the right through their constitutions, environmental legislation, court decisions or ratifications of international agreements (Boyd, 2012). In 2017, the Brazilian High Court's Justice Antonio Herman Benjamin made similar conditions to guarantee the right to a healthy environment. He was invited to answer the following question within the framework of UNEP: "more than 100 constitutions recognize the human right to a healthy environment. So why do environmental degradation and natural resource exploitation continue to plague the planet?". His reply stressed the need for effective access to legal proceedings: "just because a right is laid out in a treaty doesn't mean that it's implementable. That's why we need courts". He added that "the best system for implementing environmental law is a system where all stakeholders participate in a transparent manner. This means that we need a transparent executive, an effective legislature, an efficient administration and a strong civil society to help hold the system to account" (UNEP, 2017).

Finally, in a third report to the Human Rights Council (United Nations Human Rights Council, 2015), the Independent Expert presented a compilation of good practices of governments, international organizations, civil society organizations, corporations and others in the use of human rights obligations relating to the environment. The following are examples of those good practices related to the enhancement of institutional competencies for designing and implementing policy instruments that are useful for fulfilling the above-mentioned obligations:

- Since 2010, the Asian Development Bank hosted a series of judicial symposiums on environmental decision-making, the rule of law and environmental justice, with the aim of building relevant expertise of judges.
- Certain states have committed to support the implementation of the Guiding Principles on Business and Human Rights, endorsed by the Human Rights Council in Resolution 17/4. The United Kingdom and Northern Ireland have done so by "ensuring that agreements facilitating overseas investment do not undermine the host country's ability to impose the same environmental and social regulations on foreign investors as does on domestic firms".
- Some states have promoted informed participation by those most affected by environmental harms. In Finland, the Action Programme on eServices and eDemocracy, implemented in 2009, was designed to develop new tools on citizen participation in land-use planning. In Finland too, the city of Tampere created public advisory groups. Since 2007, and for the time of the Independent Expert's Report, they had participated in more than 350 planning-related decisions.

Box 8.10 provides a recent example of a case in which human rights are linked to the right of a healthy environment.

Box 8.10 Appeal to the Philippine Commission on Human Rights for violation of the right to a healthy environment as human right.

On 22 September 2015, Greenpeace Southeast Asia and the Philippine Rural Reconstruction Movement – alongside persons surviving typhoons or cyclones – appealed to the Philippine Commission on Human Rights to identify the responsibility of 47 companies for climate change and their violation of fundamental human rights, such as the right to a healthy environment. In particular, this appeal asks the Philippine Government to take appropriate measures to reduce these effects, that is to adopt legislation which imposes environmental obligations on these companies and which would enable victims to seek redress in the courts. On 8 December 2016, the Philippine Commission on Human Rights decided to grant the request from civil society and to initiate investigations against companies accused of participating in climate change. On 27 July 2016, the Commission on Human Rights sent the complaints lodged by the applicants to the CEOs of these companies. On 11 December 2017, the Commission held a preliminary conference to which all parties had been invited with the aim of considering the following: simplification of issues, stipulation or admission of facts and of documents, witnesses to be presented, marking of documents and such other matters as may aid in the prompt resolution of the petition. The first formal inquiry hearing is expected to be conducted at the end of the first quarter of 2018.

This is a legally-unprecedented situation, since it is the first time that a complaint of this nature has been relayed (pointing to the risk of climate change for society and highlighting the involvement of private actors in the violation of human rights) by a Commission on Human Rights. This confers a clear legitimacy on the complaint.

8.3.2.3 Fostering the respect for customary norms

Adequate institutional competencies of indigenous and local communities to develop and use biocultural community protocols -- simply referred to as “community protocols” in the Nagoya Protocol to the Convention on Biological Diversity -- can play an important role in advancing the respect of customary norms and in this way contributing to strategies to reduce land degradation and restore degraded land (Box 8.11). One of the main characteristics of these protocols is that they are developed through a community-led and endogenous participatory process (Jonas *et al.*, 2010). However, for reasons as simple as the fact that this concept is completely new for communities, they will usually need external assistance to enhance competencies both to initiate and to execute the process (LPP & LIFE Network, 2010). In this regard, NGO Natural Justice’s toolkit for facilitating the development and use of biocultural community protocols suggests the facilitation should be done by members of the concerned community or from supporting organizations with whom they have long-standing and positive relationships (Shrumm & Jonas, 2012). For the same reason, in the context of Article 12 of the Nagoya Protocol to the Convention on Biological Diversity, commentators have considered that assistance from Parties in developing community protocols is not always appropriate when the concerned indigenous or local community has the capacity and the will to handle it by itself (Greiber *et al.*, 2012). It has also been acknowledged that the facilitating entity and mediators should only provide assistance if and when required by the community, that they should act with professionalism and dedication, that they should not rush or bias the process, that they should conduct background research before the process is started and that assistance could take the form of training on aspects such as documentation, data collection, legal empowerment and facilitating meetings with government (LPP & LIFE Network, 2010).

There are different actions required for developing a biocultural community protocol, such as reflecting about the interconnectedness of various aspects of the community’s ways of life, including their customary norms; learning about the national and international legal regimes that regulate those aspects;

articulating all this and other information in the protocol; and enhancing the community's capacity to engage with other stakeholders like government agencies, researchers and project proponents (Jonas & Bavikatte, 2009). This includes the ability to better advocate in favour of the effective implementation of and compliance with Articles 8(j) and 10(c) of the Convention on Biological Diversity, in particular the Parties' obligations to "respect, preserve and maintain knowledge, innovations and practices of indigenous and local communities embodying traditional lifestyles relevant for the conservation and sustainable use of biological diversity" and to "protect and encourage customary use of biological resources in accordance with traditional cultural practices that are compatible with conservation or sustainable use requirements".

Box 8.11 Biocultural community protocols and their importance for advancing the respect for customary norms and contributing to strategies to reduce land degradation and restore degraded land.

A biocultural community protocol can be defined as "a document that is developed after a community undertakes a consultative process to outline their core cultural and spiritual values and customary laws relating to their traditional knowledge and resources"; but it can also been considered as being at the same time a process and a product (LPP & LIFE Network, 2010). A detailed description of community protocols and their contribution to the respect of customary law has recently been provided in the Mo'Otz Kuxtal Voluntary Guidelines adopted by Decision XIII/18 of the Conference of the Parties to the Convention on Biological Diversity. By documenting and describing aspects such as the way of life of the community, its customary laws, cultural and spiritual values, governance and decision-making structures, as well as the relevant national and international law (Bavikatte, 2011), indigenous and local communities provide clarity to external agencies and stakeholders – facilitating recognition of all the rights that are needed to secure community stewardship over their lands and waters (Bavikatte & Bennet, 2015). Biocultural community protocols can help communities gain recognition for, among other things, their territorial sovereignty, community-based natural resource management systems and community conserved areas, sui generis laws, sacred natural sites and globally-important agricultural heritage systems (Jonas *et al.*, 2010).

8.3.3 Competencies for economic and financial instruments

A range of public and private economic and financial instruments are available to steer strategies to halt or reverse land degradation. These include, among others, payment schemes for ecosystem services, voluntary payments, subsidies, insurance schemes, taxes, tradable rights, offsets, microfinancing, eco-labeling, auctions and efforts leverage corporate social responsibility mechanism in production sectors. Chapter 6 provides an overview of these instruments and their effectiveness (see Section 6.3.2.3). Adequate institutional competencies support the design and implementation of these economic and financial instruments. Here we will discuss competencies for two instruments related to ecosystem service markets in detail: payments for environmental services and offsets (see Section 8.3.3.1) and the need for standardized national-level information on ecosystems and their contribution to economic development (see Section 8.3.3.2).

8.3.3.1 Payments for ecosystem services and biodiversity offsets

For both payments for ecosystem services and biodiversity offsets, the devil is in the detail: small changes in the institutional design may have large consequences for the performance. Market offset programmes are often believed to be efficient due to the low transaction costs involved, compared to the bureaucratic case-by-case compensation procedure of Natura 2000 sites in the EU. However, if neither the seller nor the buyer have incentives to assure quality of the traded object (because this is largely a public good), and the object is extremely complex, then robust monitoring and enforcement are needed to assure intended outcomes. Hence, the institutional capacity and transaction costs associated with markets for ecosystem services (biodiversity offsets and voluntary payments for ecosystem services) are often high compared to taxes, subsidies and regulations, which suggests that a priori assumptions about cost-effectiveness of various policy tools should be avoided (Gómez-Baggethun & Muradian, 2015; Hahn *et al.*, 2015). Briggs *et al.* (2009) warn that "without careful regulation, habitat banks could offer low-cost compensation as a result of cutting corners on conservation, and the market would reward poorly managed banks and thus harm conservation efforts" (p. 117). The dichotomy of government regulations versus markets is therefore false (Vatn, 2015); the more we use markets to finance restoration of complex ecosystems, the more institutional capacity and regulations are needed to safeguard the intended outcomes (Glicksman & Kaime, 2013; Hahn *et al.*, 2015; Koh *et al.*, 2017). The institutional capacity needed to design and monitor market-type biodiversity offsets, where conservation credits are traded on market conditions, seems to be too high even for advanced market economies. For example, the German compensation scheme, in which the "trading" is conducted by municipal or private agencies appointed by the state, require less institutional capacity to create and enforce market-like trade (Eftec, 2010).

Institutional capacity includes both general governability (as opposed to incapability and corruption) and specific ability to craft regulations as well as resources (e.g., government funding to environmental agencies) to undertake monitoring and enforcement. In that sense, there is no surprise that Pigouvian-type payment for ecosystem services -- financed by environmental taxes and paid by the government as in Costa Rica – are the most common and most successful. Coasean-type payment for ecosystem services relying on voluntary private payments require advanced market institutions. As emphasised by the Millennium Ecosystem Assessment (MA, 2005), effective policy tools are those that realize synergies and minimize trade-offs. When designing payments for ecosystem services to support the restoration of degraded land, it is therefore important to evaluate the potential effects on equity, tenure rights, biodiversity and ecosystem services. An important element to consider when predicting or assessing the effectiveness of economic incentive-based tools, is their interplay with the normative systems and motivations of targeted actors. The critics of ecological compensation are concerned that such schemes may create the false impression that any impact can be compensated for, whereas ecosystems' link to livelihood opportunities and psycho-cultural well-being (Brown *et al.*, 2013; Ryan *et al.*, 2010; Weimann *et al.*, 2015) are locally specific and therefore not fully replaceable (Escobar, 2008; Forest Peoples Programme, 2011; Quétier & Lavorel, 2011).

Economic instruments provide governments and civil society with an important tool for tackling biodiversity and ecosystems services loss. They have been developed towards improved ecological targeting and improved economic incentives. When combined with a careful verification and monitoring system, they will help to improve habitats and ecosystem services. Naturally, decision makers must carefully assess their limitations and suitability within diverse social and cultural contexts.

8.3.3.2 Ecosystem accounting

The design and implementation of national policies aiming at reversing ecosystem degradation is constrained by a lack of national-level information on ecosystems and their contribution to economic development (Hein *et al.*, 2015). In all countries, the gross domestic product (GDP) and related indicators are compiled based on the System of National Accounts. The need to integrate ecosystem change into statistical frameworks is recognized (Boyd & Banzhaf, 2007; Obst & Vardon, 2014). Under auspices of the UN Statistical Commission, the System of Environmental-Economic Accounting -Experimental Ecosystem Accounting (SEEA-EEA) (UN, 2014) (in short "ecosystem accounting") has been developed as an experimental approach to systematically integrate ecosystems and ecosystem services into national accounts. Although ecosystem accounting is not yet recognized as an international standard, it complements the internationally-recognized approach described in the System of Environmental-Economic Accounting-Central Framework (SEEA-CF) (UN, 2014). Ecosystem accounting includes and provides guidance on the measurement of ecosystems in terms of condition, spatial extent, the capacity of ecosystems to supply ecosystem services and the benefits they generate (Hein *et al.*, 2016; Vargas, Hein, & Remme, 2017). Developing ecosystem accounts requires significant resources for the collection and integration of spatial, survey and statistical data, and skills needed to carried out the required spatial modelling and for the valuation of ecosystem assets and services (Hein *et al.*, 2015).

8.3.4 Competencies for social and cultural instruments

Institutional competencies and social-cultural instruments can set the scene for several key features for strategies to avoid and reverse land degradation at different levels, ranging from increased awareness of resource users to efficient national councils that implement broad-scale restoration strategies. Socio-cultural bottlenecks for successful conservation or restoration projects can often be reduced by strengthening competencies and promoting political willingness that address the following processes: poor collaboration between stakeholders; lack of well-trained local people; and single focus on short-term economic development. Competencies to address these bottlenecks are discussed in this sub-Section.

Land degradation affects many stakeholders and hence requires multi-objective strategies (see Section 8.2). Polycentric networks (Folke *et al.*, 2011) with active participation at multiple organizational levels are essential to oversee all interests (Dyer *et al.*, 2013). The ability to set up **multi-stakeholder partnerships** is considered crucial to simultaneously tackle different aspects of land degradation (Berkes, 2007; Folke *et al.*, 2011; Stringer *et al.*, 2012). Institutional mechanisms that facilitate transparent joint decision-making processes regarding environmental issues increase the efficiency of land degradation response strategies and its local adoption (see also Chapter 6, Section 6.4.2). Multi-institutional teams with the ability to foresee the possible trade-offs**,** offsets and/or synergies between different interests or institutions may create more win-win situations across environmental and other policies (Goldstein *et al.*, 2012), or reduce unexpected negative impacts on halting and reversing land degradation, such as unplanned land-use changes in areas with relatively healthy ecosystems. Local participation is best planned as partnerships or multilevel deliberation (Berkes, 2007) – a process where as many as possible involved parties collectively discuss land degradation and restoration issues and reflect on root problems, desired outcomes and strategies to get there. Especially indigenous and local knowledge can be of value to downscale existing broad-scale restoration strategies and adapt to local contexts (Rist *et al.*, 2010; Uprety *et al.*, 2012). Local resource users are often the first persons to detect ecosystem changes and the impacts of land degradation (Berkes, 2007), so monitoring programs and the design of restoration management plans can benefit from including local ecosystem experts (see Armitage *et al.*, 2007; Berkes, 2009; Cundill & Fabricius, 2010; Folke *et al.*, 2011; Gunderson & Light, 2006; Gunningham, 2009; more examples Schultz *et al.*, 2015).

Sufficiently-trained local people are paramount for many degradation-related processes including the design of locally-adapted restoration strategies, monitoring advances and ecosystem evolution and cost accounting. Important areas for biodiversity conservation and restoration are often remote rural areas with little access to high-quality education. The presence of a few local leaders with advanced education can create snow-ball effects and increase local awareness of nature’s contributions to people and the importance of restoring degraded lands (Schmiedel *et al.*, 2016). Capacity-building goes further than knowledge and technology transfer; it also includes exchange of failures and successful experiences, training and awareness training. In addition, the competency to continuously auto-evaluate and adapt decision-making processes - and the resulting policies - creates the necessary flexibility to adjust land degradation and restoration strategies to changing realities. Ecosystems are in constant movement and can suddenly shift between different coexistent states or regimes (Folke *et al.*, 2004). Due to this spatial and temporal ecological heterogeneity, as well as changing socio-cultural contexts of large-scale projects, steady-state resource managements that aim to prevent change and reduce variability are likely to fail at some point. Instead, policies that embrace change and direct changes to desired outcomes for society and nature may yield better results (Chapin *et al.*, 2009). This can be more easily achieved with adaptive governance (Allen & Garmestani, 2015). Such a strategy combines several policies and is sufficiently flexible to adapt its goals to meet changing needs detected by reiterative monitoring (Gavin *et al.*, 2015; Guerry *et al.*, 2015; Levin *et al.*, 2013; Schultz *et al.*, 2015)

Finally, natural resources are often exploited with a short-term vision dominated by market-oriented forces (see also Chapter 2, Section 2.1). An attitude shift towards **environmental stewardship** is much needed for reducing indirect drivers of land degradation (Chapin *et al.*, 2009; Messier *et al.*, 2015). Economic drivers of land degradation, when put in context by inclusive wealth cost-benefit analyses, captures better nature's economic, social and cultural contributions to people. More and more scientists point to the shortcomings of traditional economic indicators such as countries Gross National Product and call for including mid- and long-term costs and benefits that come with exploiting or restoring natural resources (Folke *et al.*, 2011; Guerry *et al.*, 2015; Ouyang *et al.*, 2015; UNU-IHDP & UNEP, 2014). Institutional reforms may be required to better align private short-term and public long-term goals. Such approaches should be sufficiently communicated to the broader public to stress the human dependence on healthy ecosystems and direct natural resource management strategies towards community benefits, rather than self-interest of more powerful players. In this context, being able to learn from indigenous worldviews may be of particular interest. Natural elements and humans are often equally-valued parts of the indigenous environment and natural resources and services are cared for instead of exploited (Roué & Molnár, 2016) (see also Chapter 2, Section 2.2). Nature and culture are often so closely interwoven that a reduction (degradation) or increase (restoration) in one is directly reflected in the other based on the principle "what we do the land we do to ourselves". The term "reciprocal restoration" has therefore been proposed to reflect this deep sense of stewardship among indigenous people (Kimmerer, 2011).

8.3.5 Competencies for science and technological instruments

Integrated environmental governance is an emerging scientific discipline. Several new and relatively easy-to-use modelling and support decision tools, which combine social and biophysical information, are rapidly being developed and are freely accessible online (Astier *et al.*, 2011; Bagstad *et al.*, 2013; Peh *et al.*, 2013) (see Section 8.2 and the online IPBES catalogue of policy support tools and methodologies).

On the other hand, understanding and managing land degradation, restoration and ecosystem functioning is challenged by: highly-heterogeneous contexts; complex cross-disciplinary processes with social, economic, cultural and ecological dimensions; poorly understood non-linear relations; trade-offs and amplifying or stabilizing feedbacks - often with effects and origins in different locations (Reynolds *et al.*, 2007; Simonsen *et al.*, 2014) (see Chapter 3, Section 3.2 for examples). Current important knowledge gaps include: (i) understanding environmental governance and the impact of environmental policies on land degradation and restoration in different contexts; (ii) measuring ecosystem services and natural capital as well as their changes during restoration; and (iii) understanding the links between altering ecosystem services and human well-being (Guerry *et al.*, 2015; Miteva *et al.*, 2012; Ruckelshaus *et al.*, 2013). Although commonly used simplified proxies – such as forest cover, carbon uptake rates or biological diversity – can reveal ecosystem changes (Belnap, 1998; Pereira *et al.*, 2013), they often result in misleading or partially-valid conclusions regarding ecosystem recuperation (Ferraro *et al.*, 2015). For example, indicators often reflect rapidly changing processes, while the underlying mechanisms may be evolving much more slowly and hence are more difficult to detect and monitor (Simonsen *et al.*, 2014). For these reasons, some scholars suggest that strategies to avoid and reverse land degradation require new integrative data, collected with innovative methods, to create comprehensible frameworks to guide decision-making processes (Miteva *et al.*, 2012). A current challenge is to account for spatial and temporal gaps between action and response. Therefore, continuously evaluating through time (for instance by not interrupting long-term monitoring programs when new governments are elected) and space (international or inter-regional) will create better insights into the various dynamics and changes in land degradation and restoration success.

New technologies continue to be developed for reducing implementation and monitoring costs, such as climate-smart agriculture or resource-conserving agriculture (see also Chapter 5), the use of drones for large-scale tree planting and remote monitoring (Zahawi *et al.*, 2015; http://www.biocarbonengineering.com/) or digital models to infer patterns of status, trends and detect causal mechanisms of biodiversity change (Cheung *et al.*, 2011; Franklin, 2009; Gill *et al.*, 2011; Guisan & Thuiller, 2005). However, these digital, computer-based, models can only approximate nature and human judgement and their use should not replace actual field monitoring programs, which are needed to ground-truth and calibrate the models.

The following three institutional competencies are key to develop and use sound scientific and technological instruments:

1. **Cross-institutional and interdisciplinary collaboration**. Restoration programme success requires strong, strategic and coordinated leadership among prominent government, scientific, citizen or private industry organizations – as well as sources of stable funding and adequate staff. Efforts should be participatory and cross-disciplinary (e.g., combining biophysical, social, economic and political data; Chazdon *et al.*, 2009; Ferraro *et al.*, 2015; Sassen *et al.*, 2013). The participation of the community and local land users and/or managers is paramount for collecting fine-scale local ground data and guaranteeing sufficient local labour. Local volunteers, citizen scientists and para-ecologists can implement assessment and monitoring activities (Couvet *et al.*, 2008; DeVries *et al.*, 2016; Sassen *et al.*, 2013) (Figure 8.4). Formal and recurring training, tailored for the biodiversity and conservation community, is needed to build capacity within local communities and to promote the emergence of a new generation of scientists and land managers able to carry out integrated, multi-disciplinary work. Cultural and socio-political backgrounds influence levels of participation by community members and different recruitment strategies are needed for the retention of volunteers, para-ecologists, and communities (Bell *et al.*, 2008; Schmeller *et al.*, 2009; Schmiedel *et al.*, 2016; Vandzinskaite *et al.*, 2010). Participatory monitoring is most efficient when users: benefit directly from the resource; participate in conservation/management decision-making; socialize with other participants; and get rewards for their commitment and effective monitoring (Singh *et al.*, 2014). However, although citizen science and broader participation of informally-trained scientists will yield more field data, some types of monitoring require more technical expertise (e.g., assessing chemical or radioactive contaminants). Furthermore, there needs to be well-trained staff to coordinate and oversee the data collection to ensure quality control and correct data archiving - otherwise the data are likely to not be comparable among data collectors and hence of limited utility.

Figure 8 4 Knowledge sharing on field techniques to classify soil types between scientists and local land owners to improve savannah rangeland management in Kenya. Photo Credit: Jayne Belnap.

2. **High-quality information collection and sharing**. Monitoring networks with coordinated and standardized nomenclatures, concept definitions, monitoring questions and/or goals and assessment and/or monitoring protocols (data collection, analyses and dissemination) allow for more complete assessment and monitoring programmes across larger regions (Herrick *et al.*, 2016; Schmeller *et al.*, 2014)). Communicating and defining common goals with other institutions or councils can be eased when common units and metrics are used. The ability to quantify ecosystems and its services as natural capital (next to financial, manufactured, social and human capital; Aronson *et al.*, 2007; Daily *et al.*, 2009; Wu *et al.*, 2011) can pave the road for interdisciplinary collaboration. Where monetary valuation is difficult to realize or is highly contested, natural capital can be quantified in biophysical terms or in impacts on human livelihoods (Myers *et al.*, 2013). New technologies, such as mobile phones and associated apps (Box 8.12) (e.g., EpiCollect; see also Aanensen *et al.*, 2009; Herrick *et al.*, 2016) can be used to upload bottom-up, fine-scale data to central Internet databases, and their interface with broader-scale regional and global data. This improves accessibility to high-quality data analyses across larger spatial and temporal scales and increases knowledge sharing (Guerry *et al.*, 2015; Olson *et al.*, 2013). Social media, using natural language processing, is very promising (Lin *et al.*, 2015), as is crowd sourcing for analyzing large datasets. Online repositories with free access to results from monitoring programmes are essential, starting with baseline assessments (see "deriving baselines" in Chapter 1) and change detection through time. User-friendly, intuitive and centralized data portals enhance communication and exchange of data among scientists, policymakers and the public. Information on existing policy and conservation strategies, as well as research findings, help to ensure that conservation and environmental policy strategies are up-to-date and compatible.

3. **Holistic understanding**. Ecosystems should be seen and studied as coupled human and natural systems or socio-ecological systems and hence land restoration requires integrative approaches where political, socio-economic, ecological, cultural, legal and technical actors and processes interplay (Berkes, 2007; Ferraro *et al.*, 2015; Folke *et al.*, 2011; Liu *et al.*, 2015). The ability to use integrated social and ecological information creates a more holistic understanding of land degradation problems and can help to design restoration strategies that tackle the underlying causes of environmental degradation. Integrative cost-benefit assessments of land restoration or land degradation processes

include societal impacts (Daily *et al.*, 2009). Ecological damages or benefits often interact in two directions with social, economic and cultural changes, such as in health-water or energy-food-water networks (Liu *et al.*, 2015). Integrated studies of coupled human and natural systems are needed as such cross-disciplinary processes are often poorly understood when studied by social or natural scientists, in isolation (Liu *et al.*, 2007). Both successful stories and failures are important to extract lessons learnt and common pitfalls.

An additional challenge is to guarantee the inclusion of newly generated knowledge into the decision-making process. Continuous collaboration between different stakeholders, particularly scientific leaders and high-level decision makers, during the design, implementation and monitoring is crucial to further develop and refine scientific frameworks and technical tools.

Funding institutions and international organizations including the Society of Ecological Restoration (SER), International Union for Conservation of Nature (IUCN), World Resources Institute (WRI), The Nature Conservancy, Future Earth, Global Land Project, The Global Partnership on Forest and Landscape Restoration, IPBES, and the Natural Capital Project can play a crucial coordination role in developing the above-mentioned competencies. These institutions can set research agendas through funding priorities and promote interdisciplinary investigation. More efficient global and local collaboration can be achieved by online platforms and forums, data and experience repositories and face-to-face meetings. In addition, research funding that matches for long-term processes involved in halting and reversing land degradation is called for (Stringer & Dougill, 2013).

8.3.6 Competencies for the selection and integration of policy instruments

A combination of instruments (policy basket) is typically used to govern policy interventions as it can strengthen impact. For example, for land degradation responses legal instruments have been used in combination with market-based tools to compensate land owners for their sustainable land management practices and have benefited areas and people beyond the field. Policy instruments can also negatively impact each other, for example when market-based and social-cultural instruments produce contrasting incentives. Moreover, the effects of the resulting policy interventions can have unanticipated consequences which may be positive (co-benefits), negative (trade-offs), or even perverse (the opposite of what was intended) (Bryan & Crossman, 2013) (see Section 8.4).

To account for interactions among instruments and their impact, the selection of policy instruments to halt or reverse land degradation needs to be based on an evaluation of the current institutional framework (Barton *et al.*, 2014; Ostrom, 2005). Primmer (2017) flags that: (i) new policy instruments need to match higher-level regulations and the law; (ii) need to match the mandate and competencies of the implementing organization; and (iii) should not be constrained by rigid organizational practice. This analysis of institutional constraints should pay attention to the mandates, competencies and practices at the different levels of administration. This was exemplified by a study showing the institutional constraints of the design and operationalization of a conservation auction, as an innovative mechanism for nature conservation, in Finland (Primmer, 2017). Box 8.12 shows how legal constraints define the search for economic incentives for nature conservation.

Box 8.12 Payments for environmental services: additionality to a legal standard

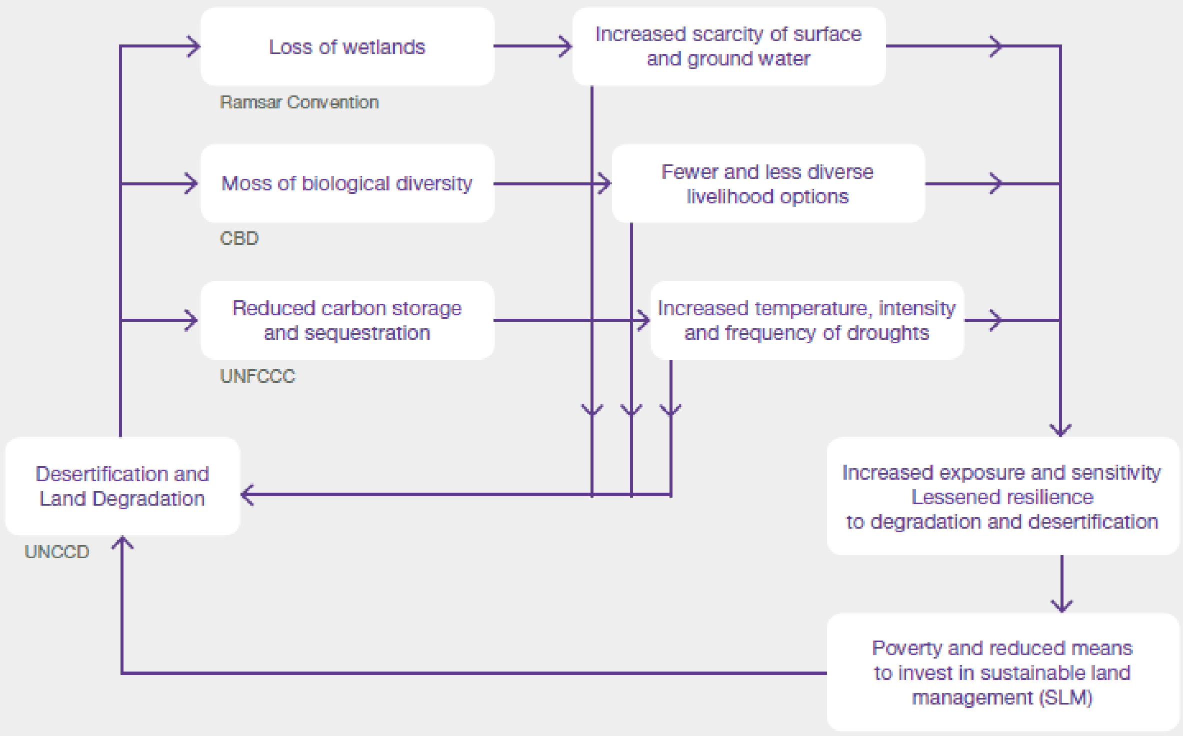
Payments for environmental services are generally defined as a transaction between agents with the aim of using land for maintenance or restoration of certain ecological functionalities. This contractual financing mode is considered an innovative law (Bennet & Carroll, 2014), but also asks for a degree of law required to receive a payment in return for an environmental service rendered. In several countries, environmental policy is based on the polluter-pays principle and not on the protective-pay principle, a logic conveyed by the payments for environmental services. In other words, in the name of the polluter-pays principle, a minimum of environmental obligations may be required (Defra, 2013; Langlais, 2013; Leonardi, 2014). However, in some countries, some actors are paid to comply with the law and stop illegal practices (Pirard & Sembres, 2010). The NGO GRET, states in this respect that "compliance with standards may be out of the reach of communities when their livelihoods are at stake" and adds that payments for environmental services may accompany "the transition to practices, permitting compliance with the law, the time they become effective" (http://www.gret.org/2016/04/paiements-services-environnementaux-pse-de-theorie-a-pratique/)

In reality, the difficulty for law is that under the same title, it is not quite the same instrument. In the Global North, payments for environmental services, a new concept, are perceived as a potential tool for their implementation. On the other hand, in many countries in the Global South, payments for environmental services are not a novelty. They remain "classic" funding tools for conservation (Langlais, 2017a; Le Coq *et al.*, 2016; Pesche *et al.*, 2013). Moreover, it should be emphasized that there is a significant gap between the payment for environmental services theory as presented by Wunder (2005), the actual promotion of this instrument on an international scale (Langlais, 2017b; Méral, 2012) and practice.For more in the different legal aspects of payments for environmental services, see: (Greiber, 2009, www.katoombagroup.org)) .

8.4 Interactions among land degradation, restoration and other policy areas

The linkages between land degradation and other global environmental challenges are increasingly recognized (Ding *et al.*, 2017; IUCN, 2015; Kumar & Das, 2014). Description of the land degradation-other global environmental challenges linkage is provided in Chapter 3 (see Sections 3.4.1 and 3.4.4), Chapter 4 (see Sections 4.2.7 and 4.2.9) and Chapter 7 (see Section 7.2.2). Degradation reduces the productivity of the land base, which in turn negatively impacts the provision of ecosystem services (e.g., food, fuel, fibre, freshwater, air and water purification and climate regulation) (MA, 2005). Land conversion and degradation are estimated to account for 4.4 Gt of CO_2e emissions each year (Matthews & Noordwijk, 2014). With each additional degraded piece of land, biodiversity loss is also exacerbated. The converse applies: addressing land degradation, for example, through restoration and prevention of degradation (action relevant to the UNCCD), can reduce greenhouse gas emissions (outcome relevant to the UNFCCC), contribute to conservation of biodiversity (outcome relevant to the CBD) (Figure 8.5), provide ecosystem services and enhance land productivity (outcome relevant to the SDGs).

Figure 8 5 Links between land degradation and desertification, biodiversity loss and climate change and their respective multilateral environmental agreements.



This Section explores how various policy areas influence degradation or enhance possibilities to address land degradation and develop restoration. It also explores ways of identifying trade-offs in order to improve coherence and synergies between land and other policy areas. Other policy areas that are explored include agriculture, water, climate change and biodiversity conservation.

8.4.1 Existing multilateral agreements to harness synergy and co-benefits for land

The United Nations Convention to Combat Desertification (UNCCD), United Nations Framework Convention on Climate Change (UNFCCC), Convention on Biological Diversity (CBD) and Sustainable Development Goals (SDGs) all aim to halt or mitigate the deterioration of the ecological processes on which life depends. Effective responses to land degradation can simultaneously contribute towards the goals of the three Rio Conventions (Cowie *et al.*, 2011). They also support other multi-lateral environmental agreements such as the Ramsar Convention – the Strategic Plan of which has goals and targets addressing wetland loss, degradation and restoration. Effective responses also contribute to the achievement of the SDGs. Each of these international agreements and global goals operates at multiple levels. Taking a multi-level approach towards preventing and reducing land degradation and restoring degraded areas offers the potential to deliver benefits at various spatial and/or institutional levels and work across a number of policy areas and stakeholder groups (Hurni, 1998, 1997). This sub-section focuses specifically on the Rio Conventions and the SDGs. It reviews progress towards achieving the Aichi Biodiversity Targets and the SDG Targets, considering the implications for the linkages between land degradation, restoration, biodiversity and climate change.

Evaluation of relevant Aichi Biodiversity Targets indicates that progress is being made on the restoration of degraded lands and increase of forest land under sustainable forest management principles (FSC International, 2017) (Table 8.4).

Table 8.4 Land degradation and restoration relevant Aichi Biodiversity Targets and examples of progress to date.

Aichi Target	Target description	Examples of progress made
Target 3	By 2020, at the latest, incentives, including subsidies, harmful to biodiversity are eliminated, phased out or reformed in order to minimize or avoid negative impacts, and positive incentives for the conservation and sustainable use of biodiversity are developed and applied, consistent and in harmony with the Convention and other relevant international obligations, taking into account national socio-economic conditions.	Transparent and comprehensive subsidy inventories and inventories of possible positive incentive measures were established by 2012 by all OECD countries, and an assessment of their effectiveness against stated objectives, of their cost-efficiency, and of their impacts on biodiversity, is underway.
Target 9	By 2020, invasive alien species and pathways are identified and prioritized, priority species are controlled or eradicated and measures are in place to manage pathways to prevent their introduction and establishment.	Policy responses to deal with the invasive species problem have increased since the 1970s, and also the number of successful eradications; but the management implementation statistics are patchy and progress in this area less apparent (McGeoch *et al.*, 2015). Progress towards this target globally remains rather uncertain, nevertheless the importance of managing invasive alien species if land productivity is to be retained is well established (see e.g., Obiri, 2011). For example, over 560 alien species (most of them invasive) of various taxa were identified in the Southern Ocean Islands.
Target 15	By 2020, ecosystem resilience and the contribution of biodiversity to carbon stocks have been enhanced, through conservation and restoration, including restoration of at least 15% of degraded ecosystems, thereby contributing to climate change mitigation and adaptation and to combating desertification.	Forest and wetland restoration programmes involving positive incentive schemes are underway worldwide with signs of improvement evident in the state of forests and wetlands in many parts of the world. Such improvements can help to reduce flood risks and improve water management while also increasing carbon stocks (Locatelli *et al.*, 2015). The UN REDD programme launched in 2008 supports REDD+ activities in over 64 countries to mitigate climate change through reducing deforestation and forest degradation along with sustainable management of forests ("UN-REDD Programme," 2016). Carbon mitigation initiatives (like REDD+) also deliver substantial biodiversity benefits (Venter *et al.*, 2009). Sustainable

		management of forests following Forest Stewardship Council's principles and criteria has increased over the years from ca. 149 million ha in 80 countries in 2012 to over 195 million ha in 83 countries in 2016 (FSC International,2017).

The 13th Conference of the Parties (COP 13) of the CBD, in 2016, also agreed upon a range of measures expected to accelerate the implementation of the Aichi Biodiversity Targets by 2020, as well as stimulate the expansion of protected areas, ecosystem restoration and sustainable wildlife management, which can contribute positively towards reducing land degradation as well as towards other policy areas like public health (FAO, 2015b). Countries further agreed on actions to integrate biodiversity in forestry, fisheries, agriculture and tourism sectors, and to work towards achieving the 2030 Agenda on Sustainable Development. CBD COP 13 also included a decision to encourage Parties to consider biodiversity as they undertake climate change mitigation and adaptation actions (under the Paris Agreement 2015), and disaster risk reduction measures.

In addition to the Aichi Biodiversity Targets, halting, reducing and reversing land degradation and restoring degraded land are directly relevant to SDG 15 (sustainably manage forests, combat desertification, halt and reverse land degradation, halt biodiversity loss) and its targets (Table 8.5) (see also Akhtar-Schuster *et al.* 2017).

Table 8.5 Land degradation and restoration relevant SDGs and examples of progress to date (Information synthesized from Akhtar-Schuster *et al.,* 2017).

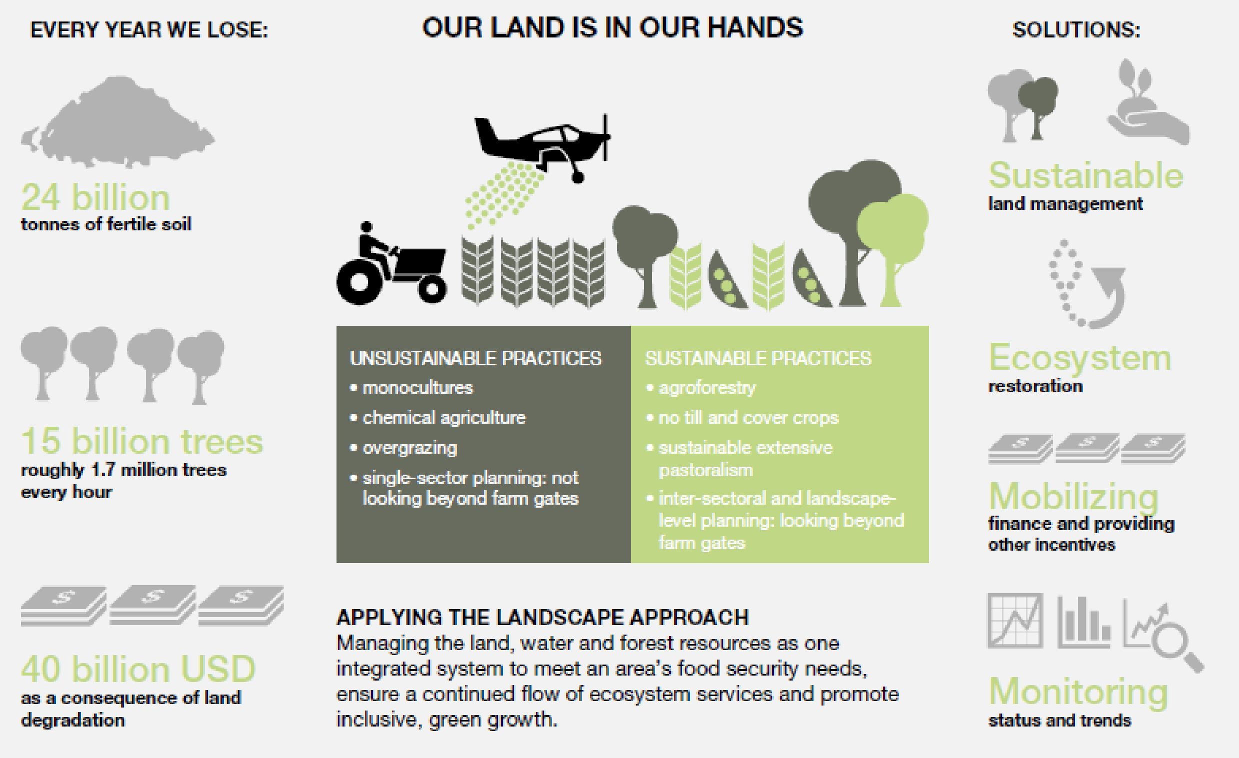
SDG Target	Target description	Examples of progress
15.1	By 2020, ensure the conservation, restoration and sustainable use of terrestrial and inland freshwater ecosystems and their services, in particular forests, wetlands, mountains and drylands, in line with obligations under international agreements.	By 2014, 15.2% of the world's terrestrial and freshwater environments were covered by protected areas. The percentage of terrestrial key biodiversity areas covered by protected areas increased from 16.5% in 2000 to 19.3% in 2016. Over the same period, the share of freshwater key biodiversity areas that are protected increased from 13.8% to 16.6% and the share of mountain key biodiversity areas under protection grew from 18.1% to 20.1% (UN, 2016).
15.2	By 2020, promote the implementation of sustainable management of all types of forests, halt deforestation, restore degraded forests and substantially increase afforestation and reforestation globally.	Between 1990 and 2015, the world's forest area decreased from 31.7 % of the world's total land mass to 30.7 % (FRA, 2015). During the same period, other areas were reforested through planting, landscape restoration activities or the natural expansion of forest. As a result, the net annual global loss of forest area declined from 7.3 million ha in the 1990s to 3.3 million ha per year during the period from 2010 to 2015 (FAO, 2015).
15.3	By 2030, combat desertification, restore degraded land and soil,	Striving towards Land Degradation Neutrality, emerged from the UN Conference on Sustainable Development (Rio+20) in 2012, and responds to the immediate challenge of how to

	including land affected by desertification, drought and floods, and strive to achieve a land degradation-neutral world.	sustainably intensify the production of food, fuel and fiber to meet future demand without the further degradation of the finite land resource base (UNCCD, 2015b). Its objective is to maintain or even improve the extent of healthy and productive land resources over time and in line with national sustainable development priorities, through efforts such as the landscape approach (Figure 8.7). Three global indicators are being used to monitor progress towards the Land Degradation Neutrality target: change in land cover; change in land productivity (net primary production) and change in soil organic carbon stocks. Although important steps forward have been made in operationalizing the concept of Land Degradation Neutrality by the UNCCD's SPI in their Land Degradation Neutrality framework (Orr *et al.*, 2017), data on progress is currently lacking despite promising indications (Akthar-Schuster *et al.*, 2017).

Although there have been conflicting reports about the success of sustainable agriculture practices (e.g., Garbach *et al.*, 2017), other studies (e.g., Enderton, 2014; Pretty *et al.*, 2005) suggest that sustainable agriculture practices and organic farming increase farm productivity through higher yields, higher water-use efficiency, and lower input costs compared to conventional practices (Figure 8.6). Such low(-er) input approaches often involve less use of herbicides, also helping to maintain pollinator populations by having a positive effect on the abundance and diversity of the flowering plants that provide their food source (IPBES, 2016c). Sustainable land management, therefore, has potential to simultaneously address Targets 15.1, 15.2 and 15.3, and improve livelihoods while also contributing to other Sustainable Development Goals and Targets (e.g., Goal 2 Zero Hunger).

Figure 8 6 Comparison of sustainable and unsustainable agriculture practice activities.

The figure highlights the actions required to achieve land degradation neutrality. Source: UNEP/UNCCD (2016).



Climate change and land degradation and restoration are closely interlinked and have impacts on a range of ecosystems and ecosystem processes, which in turn influence the provision of nature's contributions to people (Reed & Stringer, 2016). Interaction between climate change and land degradation will be felt very differently around the world: some areas will become drier while others become wetter (Business @ Biodiversity, 2010; UNCCD, 2015a) with knock-on implications for the people living there and the ecosystems that support them. The IPCC is currently preparing a special report on climate change, desertification, land degradation, sustainable land management, food security and greenhouse gas fluxes in terrestrial ecosystems - due to be completed in September 2019. The report is expected to synthesize knowledge on the links between climate change and land issues, offering possible ways forward to harness synergy between efforts that address the two issues.

Although it is too early to evaluate the impacts of the Paris Agreement (2015) and its effect on mitigating climate change and halting land degradation, earlier evidence suggests that land-based carbon sequestration and storage objectives create strong potential synergies between the UNCCD and UNFCCC and can improve resilience and capacity to adapt to the anticipated impact of climate change (Cowie *et al.*, 2011). The UNFCCC also tackles land issues in other ways. Afforestation and reforestation programs, which are part of the Clean Development Mechanism, increase carbon storage in soils and vegetation. Soil carbon management is further considered as one of the most cost-effective mitigation options under the Kyoto protocol (Al-Juaied & Whitmore, 2009; McKinsey & Company, 2009). Sustainable land management practices, which also build soil carbon, include conservation agriculture and agroforestry practices.

Reducing Emissions from Deforestation and forest Degradation (REDD+) is another initiative at the forefront of climate change mitigation efforts. REDD+ as an approach to halt deforestation and forest land degradation is simultaneously reducing carbon emissions and enhancing biodiversity conservation, while providing financial incentives to local communities and governments in developing countries (UNCCD, 2013) as well as Sustainable Development Goals on improving livelihoods. It is too early to make overall conclusions on the effectiveness of REDD+ as there are opposing views about it in the literature (e.g., Pasgaard *et al.*, 2016). With respect to REDD+ project implementation, socio-economic assessments may be useful not only as a means to evaluate impacts on livelihoods, but also to help understand the root causes of land degradation and deforestation at the community level. In practice, not considering or addressing the social dynamics (e.g., land degradation due to poverty) can lead to leakage, conflicts and the volatility of projects (Benessaiah, 2012; Parrotta *et al.*, 2012).

Other notable initiatives that complement the Aichi Biodiversity Targets on restoration and SDG 15 span multiple levels and include the Bonn Challenge and Global Restoration Initiative – efforts aimed at restoring 150 million hectares of degraded lands by 2020 and 350 million hectares by 2030 (Box 8.13). Some country and regional initiatives such as Initiatives 20x20 (to restore 20 million ha of land in Latin America and Caribbean by 2020) and AFR100 (to restore 100 million ha of land in Africa by 2030) are also evolving to complement the Bonn Challenge (GRI: http://www.wri.org/our-work/project/global-restoration-initiative).

To date, it is estimated that $1.25 billion has been committed to finance projects on the ground linked to the Bonn Challenge (Vergara *et al.*, 2016), which include improved agricultural production, enhanced food security, carbon storage, ecotourism and wood-forest and non-wood forest products production.

Box 8.13 Examples of restoration at multiple levels stemming from various initiatives.

- In Tanzania, the rebirth of the traditional Ngitili management system led to the restoration of approximately 500,000 hectares of woodland between 1986 and 2001. The integration of sustainable land management and restoration activities benefited over 800 villages, providing an economic value of $14 per month per person – almost double the average level of rural consumption in Tanzania (http://sapiens.revues.org/1542).
- The internationally-funded Sustainable Land Management Programme has helped Ethiopia to make 180,000 hectares of degraded land productively usable through practices, such as terracing, crop rotation systems, improvement of pastureland and permanent green cover. These measures have benefited more than 194,000 households and contribute to increased productivity in the affected areas. They also enhance the resilience of small-scale agriculture to the impacts of climate change and related stressors (https://www.giz.de/en/worldwide/18912.html).

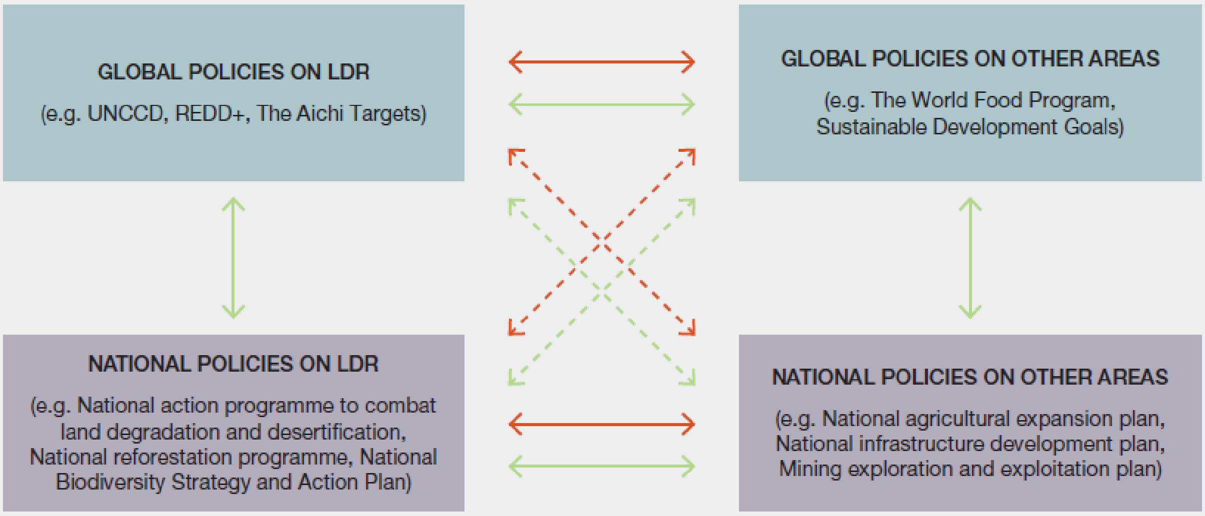
In summary, sustainable land management practices that conserve moisture, reduce or reverse soil degradation, maintain or enhance species diversity - simultaneously and synergistically - contribute to the objectives of the three Conventions (Cowie *et al.*, 2007). However, there are trade-offs as well, as optimization for one objective can reduce outcomes for others. For example, monoculture of exotic species may produce greatest carbon sequestration benefits, but reduce biodiversity values. At the same time, certain land-use and land-management practices are widely recognized as threats to biodiversity (CBD, 2008), including land clearing for agriculture (Losos & Schluter, 2000), overgrazing of rangelands (Tasker & Bradstock, 2006) and unsustainable harvesting of wild plant and animal species (De Roos & Persson, 2002). These kinds of challenges are explored in the next sub-Section.

8.4.2 Policy interactions across sectors

Policies to combat land degradation do not operate in isolation (Figure 8.7), even though sometimes they are treated in a siloed way. Policy makers have already started integrating ecosystem health concerns into some sectoral policies with a focus on harnessing synergies between biodiversity conservation and sustainable production. However, there are other policies operating at multiple levels and over several scales that govern the drivers and impacts of land degradation and the types and distributions of benefits emerging from restoration. As the growing population places pressure on finite land resources, policies aim to ensure adequate supply of food, water, energy and shelter, and to support a country's growth. This has resulted in increased pressure on land from agriculture, forestry, livestock grazing, energy production and urbanization. Indeed, urban and industrial development that consumes land is a growing driver of changes in land use and land cover, requiring proactive management to ensure that detrimental effects on land, soil and ecosystem services do not ensue (Cerreta & De Toro, 2012) (see also Chapter 3).

Figure 8 7 Relationships between decisions to combat land degradation and support restoration and other policy areas at various levels.

Green and red arrows represent positive and negative relationships respectively, while solid and dashed arrows respectively depict direct and indirect links.



While policies seek to ensure that these needs are met, they sometimes fuel land degradation, which over time reduces productivity - leading to higher demand for more land and can increase deforestation with negative impacts on climate. Identification of such interactions within policies from different sectors is key to combating land degradation and ensuring land restoration through sustainable land management.

For instance, land degradation over the next 25 years may reduce global food production by up to 12% if the land degradation trend remains unchecked (International Food Policy Research Institute, 2013). Targeted plans to increase food production often neglect taking into account the negative factors that may arise and the contribution this can make to exacerbating overall human vulnerability (Stringer, 2009). For example, projections of a required 50% increase in food production by 2050 do not take into account environmental degradation and a changing climate, which could reduce agricultural yields by 13 to 45% (UNCCD, 2012). Climate change, water scarcity, invasive pests and land degradation could cause up to 25% reduction of the world food production (Nellemann *et al.*, 2009). Additionally, even if all forests in developing countries were protected under the REDD+ policy initiative, agricultural expansion into other natural lands could lead to 50% reduction of mitigation from forest production because of emissions from "deflected" expansion into non-forested land (Terrestrial Carbon Group, 2010). These examples show the importance of having a comprehensive view of policy interactions. To efficiently balance trade-offs and link social and economic development with environmental and climatic protection and enhancement, all land uses should be examined in an integrated manner - especially because land, including freshwater and coastal systems (IPBES, 2015b), is the bond that keeps together the interdependent loop of food, water, energy and environmental health.

Assessing policy impacts across sectors often requires the use of indicators, such as impacts on productivity of the land, the extent to which the land resource is able to provide the expected ecosystem services and the availability and quality of raw materials extracted from the land (Stolte *et al.*, 2016). In some cases, shared indicators can be used across multiple sectors to provide useful information on complementary policy areas. Ecosystem services nevertheless present more complex interactions between ecosystem components which are often non-linear. Soil characteristics (e.g., soil fertility, water holding capacity, soil organic carbon) are some of the best ecosystem performance measures, because they are sensitive and specific to numerous stressors, are ubiquitous and simple to sample, and they integrate various ecological processes (Davis *et al.*, 2012; Siebielec *et al.*, 2010). To understand ecosystem changes over large areas, however, often requires enormous time and financial investment, especially if on-the-ground data monitoring and evaluation data are to be collected.

Policies regarding water, waste, chemicals, industrial pollution prevention, nature protection, pesticides, agriculture often affect and are affected by soil protection measures. The trade-offs this can create can be faced through approaches such as integrated programmes and approaches for land-use, spatial planning and land-management practices that include the implementation of renewable energy targets, forest and agricultural land use, green infrastructure, land re-use and more general holistic land resource management (EEA, 2010). It can also require some degree of policy analysis to assess the coherence of proposed actions before those actions are implemented, allowing decision makers to reduce any unintended negative effects. Policy analysis approaches can offer important insights into where different sectors are undermining or supporting one another horizontally, as well as showing where they are aligned vertically (e.g., with international treaties; Chandra & Idrisova, 2011). Conversely, policy approaches that are coherent can help to deliver greater overall effectiveness and efficiency and reduce competition between sectors for finite financial and other resources (Akhtar-Schuster *et al.*, 2011).

One approach to policy analysis is Qualitative Document Analysis (QDA) (e.g., Altheide *et al.*, 2008). It uses subjective scoring followed by validation through expert interviews and generally follows five main steps: (i) set the criteria for selection of the documents to be analyzed; (ii) obtain the selected documents; (iii) analyze the documents and undertake the scoring; (iv) validate the initial findings; and (v) finalize (Altheide *et al.*, 2008). An example of the type of scoring criteria is shown below in Table 8.6, in relation to an assessment of water, agriculture, national development plans, climate change strategies, national adaptation plans and Intended Nationally Determined Contributions in Malawi, Tanzania and Zambia (England *et al.*, 2017). The literature also provides methods of doing this in relation to climate change and coherence between the Intended Nationally Determined Contributions and the SDGs in member states of the Economic Community of West African States (Antwi-Agyei *et al.*, 2017).

Table 8.6 Example of criteria to assess coherence (Adapted from Le Gouais & Wach, 2013; England *et al.*, 2017) .

Type of coherence	Description of coherence	Score
High	The policy aligns strongly across water, agriculture and climate change statements. Type of coherence and description of coherence Score High. Policy devotes specific attention to both water and agriculture inter-sector alignment and relation to climate change adaptation. It includes numerous and detailed complementary activities (including projects) for achieving that.	3
Partial	Although the policy supports both water and agriculture inter sector alignment and, in relation to climate change, adaptation (particular in the form of general statements), it is less clear and distinct on how it could be achieved. Relatively fewer details and activities are included within the policy.	2
Limited	The policy supports water and agriculture inter-sector alignment and/or in relation to climate change adaptation. Lack of relative details in terms of activities and plans.	1

None	There is no evidence in the policy to suggest that sectoral statements are coordinated and/or aligned.	0

The literature is nevertheless lacking in terms of detailed multi-sector policy coherence analyses. Coherence and trade-offs between strategies to reduce land degradation and promote restoration and environment policies, water management policies, energy and climate policies and transport policies have been explored by Stolte *et al.* (2016), while Stringer *et al.* (2009) examine policy relationships both horizontally and vertically, assessing the extent to which international- and national-level policy supports local adaptations in Botswana, Malawi and Swaziland. A detailed analysis of the impacts of various instruments is underway to identify potential incoherence, contradictions and synergies of existing national and EU policies in the RECARE project (http://www.recare-hub.eu/recare-project). Other analyses (see England *et al.*, 2017; Frelih-Larsen *et al.*, 2016) have already taken stock of existing soil protection policies and measures in the EU and its member states, helping to identify gaps with respect to selected soil threats and functions. This has led to an inventory of existing and future policy instruments at the EU level and an analysis that assess the coverage of soil threats and functions in EU policies and their strengths, weaknesses, opportunities and threats. Such assessments could usefully be provided across sectors at the national level and in other locations around the world.

Although policies to restore degraded lands may be in place, decisions made by the local communities will determine the level of implementation of the policies. For example, while policies in Vietnam guiding forest land allocation, sedentarization and reforestation programmes specifically targeted uplands management, they did not translate into action until individuals in the targeted communities abandoned fields due to low crop yields; which eventually led to the breakdown of the informal collective arrangements for farmland protection and forced others to abandon theirs due to increased cost of field protection. Only then was tree planting, a government subsidized activity, taken up as a "least bad solution" (Clement, 2006). Policy formulation is, therefore, only part of the story. Implementation and moving policy into action is needed to effect on-the-ground change.

8.4.3 Reducing trade-offs and enhancing coherence in policy

Reducing trade-offs and enhancing synergies to address land degradation and/or develop restoration includes measures such as: institutional and capacity-building, policy instruments, research and development. It is nevertheless impossible to provide an accurate and appropriate general prioritization of responses under each of these categories. Communities, countries and regions experience different political, economic, social, historical and environmental contexts (see Warren, 2002) as well as having to select from responses that consider different scales (both temporal and spatial). What is appropriate and should be prioritized in one point in time (and in one location) to tackle degradation or advance restoration, may be entirely inappropriate in another. Recognizing this diversity, this sub-Section outlines ways in which response prioritization can be agreed amongst different groups and individuals involved in decision-making, with a view of reducing trade-offs and enhancing synergies.

Reducing trade-offs and enhancing synergies can be viewed as different sides of the same coin to some degree. From a networked and polycentric governance perspective, trade-offs can occur across time, space, sectors and different stakeholder groups. Similarly, synergies can be across different horizontal and vertical governance levels through a focus on synergy in processes, as well as focusing on outcomes that are synergistic. Reducing trade-offs and enhancing synergies is very much a governance issue. It, therefore, requires institutional coordination, multi-stakeholder engagement and the development of committees and governance structures that bridge different ministries, types of knowledge, sectors and stakeholder groups (see examples in Akhtar-Schuster & Thomas, 2011; Chasek *et al.*, 2011; Stringer *et al.*, 2012). By bringing together the necessary mixture of expertise and policymakers, it can help, for example, to ensure that restoration of degraded forests uses appropriate species that do not negatively affect surrounding land uses and livelihoods, or that rehabilitation of degraded mangroves do not cause changes to sedimentation that negatively impacts upon fisherfolks' river access. It also allows the inclusion of local knowledge in decision-making (Stringer & Reed, 2007).

Improved institutional coordination and multi-stakeholder involvement can also help to mitigate and diffuse conflict between different groups. This is especially so if they create a space for social learning to take place and to build the capacity of those involved, so that they can better understand the perspectives and needs of different stakeholders (Reed *et al.*, 2010b). Participatory and stakeholder engagement approaches can also lead to the co-development of restoration responses and jointly agreed prioritizations (see Section 8.3.4).

From an ecosystem services perspective, trade-offs can occur as a result of decisions and policies that aim to enhance delivery of some (often provisioning) ecosystem services, at the expense of others (particularly regulating, supporting and cultural services), undermining the quality of the land. This can lead to degradation as well as biodiversity loss. Often trade-offs occur and synergies are missed because decision-making and selection of options occurs at different scales by different groups.

Van der Biest *et al.* (2014) observe three distinct degrees of trade-offs between ecosystem services:

- **First level trade-offs** are linked to the land's biophysical potential (e.g., soils with high levels of organic matter have higher water holding capacities than low organic soils). Land capability assessments can play a useful role in determining land uses in such a way that degradation is minimized and can help decision makers to prioritize options.
- **Second-level trade-offs** relate to the actual delivery of potential services within a defined system, taking into account biophysical potential trade-offs as well as land-use and management based trade-offs (e.g., decisions to drain peatlands for forestry or palm oil plantations, as seen respectively in locations as diverse as Belarus and Indonesia, determines which potential services are delivered to a greater or lesser degree). Recent research in Botswana that combined quantitative and qualitative data in a Multi-Criteria Decision Analysis, showed that rangeland areas under communal tenure delivered a wider range of ecosystem services than land under private ownership in which cattle production is prioritized as a result of privatization and trade and subsidies (Favretto *et al.*, 2014, 2016). This shows the importance of policy and economic instruments (including the incentives and disincentives they create) in shaping whether land degradation occurs, and in determining where restoration is required or may be needed in future. It also highlights the utility of Multi-Criteria Decision Analysis as a tool in helping diverse groups of decision makers to prioritize options. Multi-Criteria Decision Analysis can also help to identify which groups in society will benefit and lose out from particular options.
- **Third-level trade-offs** concern the final nature's contributions to people, depending on factors such as demand, accessibility and ecosystem service flows. For example, whether provisioning services such as food are actually sold (often requiring policies to support the development of particular markets) or whether forests are accessed for recreation (requiring particular property rights that permit access). Prioritizing options at this level demands consideration of human and environmental (including climatic) processes at multiple geographical scales and multiple levels of governance. Interactions across scales and levels must also be considered if synergies are to be harnessed. Often, prioritization of decision-making options is driven by dominant political or economic agendas, even if it is known (e.g., through scenario analysis and modelling, or cost-benefit analyses) that particular

choices will worsen degradation over the longer term and result in greater costs in developing restoration strategies (ELD Initiative, 2015), or increase the vulnerability of the poor.

While a growing body of literature illustrates case examples of the factors and opportunities that can promote synergy between policies and policy processes at the international and national level (e.g., Chasek *et al.*, 2011; Cowie *et al.*, 2007; Cowie *et al.*, 2011; Gomar *et al.*, 2014; Gomar 2016), concrete examples and empirical evidence of synergistic outcomes are still lacking, and in many cases are in need of further research. Nevertheless, responses to land degradation that manage the interactions between different types of ecosystem services have been noted to produce better outcomes for society (MA, 2005) and can enhance synergy in outcomes. For example, sustainable land management in the form of conservation agriculture is one approach that takes a more holistic view of ecosystem services. Conservation agriculture practices have been widely used in countries including Zimbabwe, Zambia and Malawi, and include reduced soil tillage, permanent coverage of the soil with organic matter and crop rotation and/or intercropping, all of which are reported to yield multiple benefits (Whitfield *et al.*, 2015). These benefits include enhanced crop yields (provisioning services), enhanced soil carbon storage (regulating services), reduced soil erosion and improved soil water retention (aiding both provisioning and regulating services) (Thierfelder & Wall, 2009). Similarly, Altieri and Toledo (2011) report the use of new multi-stakeholder approaches and technologies that combine agroecological science and indigenous knowledge systems in Latin America. Outcomes from these approaches are delivering enhanced food security while conserving natural resources, and empowering peasant organizations and movements at a range of different scales. Examples in the literature complement those presented in Chapter 1, which showed how land conservation and restoration measures have helped to deliver improvements in livelihoods, reduce poverty and strengthen long-term sustainability of land use and the extraction of natural resources.

We now have at our disposal a greater range of approaches, tools and actions to understand and act upon land degradation than at any other time in human history. These are supported by lessons learned from a wide variety of different contexts, indigenous and local knowledge and practices that sustain the environment, and experiences gained in the restoration and rehabilitation of degraded areas. As we proceed further into the Anthropocene, it is clear that conceptualizing humans as an integral part of nature is vital (Warren, 2002) if we are to prevent, reduce and reverse degradation – furthering the shift away from outdated views of people as external to ecosystems. Harnessing the potential of the available tools, policies and instruments to make informed decisions and responses that minimize trade-offs and harness synergy – to deliver more efficient, sustainable, effective and equitable outcomes – necessitates consideration of the needs of stakeholders within local production systems, as well as the expectations that they (and society at large) place upon the land.

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Glossary

Abundance (ecological)	The size of a population of a particular life form in a given area.
Acceptance	Acceptance of IPBES outputs at a session of its Plenary signifies that the material has not been subjected to line-by-line discussion and agreement, but nevertheless presents a comprehensive and balanced view of the subject matter.
Acidification	Ongoing decrease in pH away from neutral value of 7. Often used in reference to oceans, freshwater or soils, as a result of uptake of carbon dioxide from the atmosphere.
Acid deposition (acid rain)	Precipitation with a low pH (acid) caused by atmospheric pollutants.
Acid sulfate soils	Common name for soils that contain metal sulphides.
Active restoration	See “restoration”.
Adaptive capacity	The general ability of institutions, systems and individuals to adjust to potential damage, to take advantage of opportunities, or to cope with the consequences.
Adaptive management	A systematic process for continually improving management policies and practices by learning from the outcomes of previously employed policies and practices.
Aeorobic	A condition in which molecular oxygen is freely available.
Afforestation	Converting grasslands or shrublands into tree plantations. Afforestation is sometimes suggested as a tool to sequester carbon, but it can have negative impacts on biodiversity and ecosystem function, for example by reducing runoff and so decreasing water production.
Agenda setting	One of four phases in the policy cycle. Agenda setting motivates and sets the direction for policy design and implementation.
Agribusiness	Denotes the collective business activities that are performed from farm to table. It covers agricultural input suppliers, producers, agroprocessors, distributors, traders, exporters, retailers and consumers. Agro-industry refers to the establishment of linkages between enterprises and supply chains for developing, transforming and distributing specific inputs and products in the agriculture sector. Consequently, agro-industries are a subset of the agribusiness sector. Agribusiness and agro-industry both involve commercialization and value addition of agricultural and post-production enterprises, and the building of linkages among agricultural enterprises. The terms agribusiness and agro-industries are often associated with large-scale farming enterprises or enterprises involved in large-scale food production, processing, distribution and quality control of agricultural products.
Agricultural commodity	A primary agricultural product that can be bought and sold.
Agricultural extensification	The process (or trend) of developing a more extensive production system, i.e., one which utilizes large areas of land, but with minimal inputs and expenditures of capital and labour.

Agricultural Intensification	An increase in agricultural production per unit of inputs (which may be labour, land, time, fertilizer, seed, feed or cash).
Agricultural orientation index (AOI)	The Agriculture Orientation Index (AOI) for Government Expenditures is defined as the Agriculture Share of Government Expenditures, divided by the Agriculture Share of Gross Domestic Product (GDP), where Agriculture refers to the agriculture, forestry, fishing and hunting sector.
Agrisilvicultural systems	A land-use system in which growing of trees and agriculture crops occur together in same lands.
Agrisilvipastoral systems	A land-use system, implying the combination or deliberate association of a woody component (trees or shrubs) with cattle in the same site.
Agrobiodiversity	Agrobiodiversity or agricultural biodiversity is a broad term that includes all components of biological diversity of relevance to food and agriculture, and all components of biological diversity that constitute the agricultural ecosystems, also named agro-ecosystems: the variety and variability of animals, plants and micro-organisms, at the genetic, species and ecosystem levels, which are necessary to sustain key functions of the agro-ecosystem, its structure and processes (CBD COP decision V/5, appendix). Agricultural biodiversity is the outcome of the interactions among genetic resources, the environment and the management systems and practices used by farmers, in some cases over millennia.
Agrochemical	Any substance used to help manage an agricultural ecosystem, or the community of organisms in a farming area. Agrochemicals include: (i) fertilizers; (ii) liming and acidifying agents; (iii) soil conditioners; (iv) pesticides; and (v) chemicals used in animal husbandry, such as antibiotics and hormones.
Agroecology	The science and practice of applying ecological concepts, principles and knowledge (i.e., the interactions of, and explanations for, the diversity, abundance and activities of organisms) to the study, design and management of sustainable agroecosystems. It includes the roles of human beings as a central organism by way of social and economic processes in farming systems. agroecology examines the roles and interactions among all relevant biophysical, technical and socioeconomic components of farming systems and their surrounding landscapes.
Agroecosystem	An ecosystem, dominated by agriculture, containing assets and functions such as biodiversity, ecological succession and food webs. An agroecosystem is not restricted to the immediate site of agricultural activity (e.g. the farm), but rather includes the region that is impacted by this activity, usually by changes to the complexity of species assemblages and energy flows, as well as to the net nutrient balance.
Agroforestry	A collective name for land-use systems and technologies where woody perennials (trees, shrubs, palms, bamboos and so on) are deliberately used on the same land-management units as agricultural crops and/or animals, in some form of spatial arrangement or temporal sequence. Agroforestry can enhance the food supply, income and health of smallholder farmers and other rural people.
Aichi Biodiversity Targets	The 20 targets set by the Conference of the Parties to the Convention for Biological Diversity (CBD) at its tenth meeting, under the Strategic Plan for Biodiversity 2011-2020.
Alien species	See "invasive alien species".

Alluvial soil	Soils deposited by water.
Amorphous	Without a clearly defined shape or form.
Anaerobic	Descriptive of a condition in which molecular oxygen is not available.
Anthrome	Neologism for Anthropogenic biome, i.e. an ecosystem produced by humans.
Anoxic	Depleted of dissolved oxygen.
Anthropocentric value	See "Values".
Anthropocentrism (or anthropocentric)	In an anthropocentric view of nature, nature is valued for its benefits to human beings. See "Ecocentric".
Anthropogenic	Originating from human activity.
Anthropogenic assets	Built-up infrastructure, health facilities, or knowledge - including indigenous and local knowledge systems and technical or scientific knowledge - as well as formal and non-formal education, technology (both physical objects and procedures), and financial assets. Anthropogenic assets have been highlighted to emphasize that a good quality of life is achieved by a co-production of benefits between nature and people.
Approval	Approval of IPBES outputs signifies that the material has been subject to detailed, line-by-line discussion and agreement by consensus at a session of the Plenary.
Aqueous slurries	A semi-liquid mixture, typically of fine particles of manure, cement, or coal suspended in water.
Aquifer	A body of permeable rock which can contain or transmit groundwater.
Arid ecosystem	Those in which water availability severely constrains ecological activity.
Aridification	A chronic reduction in soil moisture caused by an increase of mean annual temperature or a decrease in yearly precipitation.
Assessment reports	Published outputs of scientific, technical and socioeconomic issues that take into account different approaches, visions and knowledge systems, including global assessments of biodiversity and ecosystem services with a defined geographical scope, and thematic or methodological assessments based on the standard or the fast-track approach. They are composed of two or more sections including a summary for policymakers, an optional technical summary and individual chapters and their executive summaries. Assessments are the major output of IPBES, and they contain syntheses of findings on topics that have been selected by the IPBES Plenary.
Assisted colonization	Also known as assisted migration or managed relocation, is the act of deliberately moving plants or animals to a different habitat. The destination habitat may have either historically held the species or it may not have hosted the species, but the habitat provides the bioclimatic requirements to support it. Assisted colonization may also supplement an existing population in a site where their numbers are dwindling (McLachlan et al, 2007). This is especially the case where the assisted species are unable to disperse at a rate which keeps pace with the shifting bio-climatic, bio-physical envelope.
Available water capacity	Soil water content useable by plants, based on the effective root penetration depth.
Badlands	Areas where most soil has been eroded away.

Bare soil	A land cover class that includes any geographic area dominated by natural abiotic surfaces (bare soil, sand, rocks and so on) where the natural vegetation is absent or almost absent (covers less than 2%).
Baseline	A minimum or starting point with which to compare other information (e.g., for comparisons between past and present or before and after an intervention).
Behavioural economics	The study of the influence of emotions and opinions on the decisions people and organizations make in spending and saving. Behavioural economics suggests that human decisions are strongly influenced by context, including the way in which choices are presented to us. Behaviour varies across time and space, and it is subject to cognitive biases, emotions, and social influences. Decisions are the result of less deliberative, linear and controlled processes.
Beneficiary pays principle	The beneficiary pay principle aims to compensate providers for costs involved in production of beneficial environmental goods and services.
Benefit sharing	Distribution of benefits between stakeholders.
Benefits	Advantage that contributes to wellbeing from the fulfilment of needs and wants Advantage that contributes to wellbeing from the fulfilment of needs and wants. In the context of nature's contributions to people (see "Nature's contributions to people"), a benefit is a positive contribution. (There may also be negative contributions, dis-benefits, or costs, from Nature, such as diseases).
Bioaccumulation	The accumulation of environmental pollutants such as isotopes of elements, inorganic and organic compounds in organisms or the environment.
Biocapacity	The capacity of a country, a region, or the world, to produce useful biological materials for its human population and to absorb waste materials.
Biocentrism	See "Ecocentrism"
Biochar	Charcoal made from biomass via pyrolysis and used for soil enhancement.
Biocultural diversity	The diversity exhibited collectively by natural and cultural systems. It incorporates three concepts: firstly, that the diversity of life includes human cultures and languages; secondly, that links exist between biodiversity and human cultural diversity; and finally, that these links have developed over time through mutual adaptation and possibly co-evolution between humans, plants and animals.
Biodegradation	Physical and chemical breakdown of a substance by living organisms, mainly bacteria and/or fungi.
Biodiesel	A fuel that is similar to diesel fuel and is derived from usually vegetable sources (such as soybean oil).
Biodiversity	The variability among living organisms from all sources including, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems.
Biodiversity dilution effect	A high number of species present in defined areas protects humans from infection from pathogens with an animal reservoir.
Biodiversity hotspot	A generic term for an area high in such biodiversity attributes as species richness or endemism. It may also be used in assessments as a precise term applied to geographic areas defined according to two criteria: (i) containing

	at least 1,500 species of the world's 300,000 vascular plant species as endemics; and (ii) having lost 70% of its primary vegetation.
Biodiversity loss	The reduction of any aspect of biological diversity (i.e., diversity at the genetic, species and ecosystem levels) is lost in a particular area through death (including extinction), destruction or manual removal; it can refer to many scales, from global extinctions to population extinctions, resulting in decreased total diversity at the same scale.
Biodiversity offset	A tool proposed by developers and planners for compensating for the loss of biodiversity in one place by biodiversity gains in another.
Biodynamic agriculture (or biodynamics)	A holistic, ecological, and ethical approach to farming, gardening, food, and nutrition. Biodynamic agriculture has been practiced for nearly a century, on every continent on Earth. Biodynamic principles and practices are based on the spiritual insights and practical suggestions of Dr. Rudolf Steiner, and have been developed through the collaboration of many farmers and researchers since the early 1920s. See also "Conservation Agriculture",
Bioenergy	Energy for industrial or commercial use that is derived from biological sources (such as plant matter or animal waste).
Bioenergy with Carbon Capture and Storage (BECCS)	A future greenhouse gas mitigation technology which produces negative carbon dioxide emissions by combining bioenergy (energy from biomass) use with geologic carbon capture and storage.
Biofuel	Fuel made from biomass.
Biological control (or biocontrol)	A method of controlling pests such as insects, mites, weeds and plant diseases using other organisms. It relies on predation, parasitism, herbivory, or other natural mechanisms, but typically also involves an active human management role. It can be an important component of integrated pest management (IPM) programs.
Biomass	The mass of non-fossilized and biodegradable organic material originating from plants, animals and micro-organisms in a given area or volume.
Biome	Global-scale zones, generally defined by the type of plant life that they support in response to average rainfall and temperature patterns. For example, tundra, coral reefs or savannahs.
Bioprospecting	The process of searching for and subsequently developing new drugs based on biological resources.
Bioremediation	The use of microorganisms to clean up polluted soil and water.
Biosecurity	Strategy, efforts and planning to protect human, animal and environmental health against biological threats
Biosphere	The sum of all the ecosystems of the world. It is both the collection of organisms living on the Earth and the space that they occupy on part of the Earth’s crust (the lithosphere), in the oceans (the hydrosphere) and in the atmosphere. The biosphere is all the planet’s ecosystems.
Bio-technical stabilization	A method for mitigating land degradation using mechanical (structures) and biological elements to prevent severe erosion.
Biotechnology	A method for mitigating land degradation using mechanical (structures) and biological elements.

Bioterrorism	The deliberate, private use of biological agents to harm and frighten the people of a state or society, is related to the military use of biological, chemical, and nuclear weapons.
Biota	All living organisms of an area; the flora and fauna considered as a unit.
Bog	An entirely rainfed wetland area that typically accumulate peat.
Brackish water	Inland water with a high salt concentration.
Built environment	Comprises urban design, land use and the transportation system, and encompasses patterns of human activity within the physical environment.
Bureau	The IPBES Bureau is a subsidiary body established by the Plenary which carries out the governance functions of IPBES. It is made up of representatives nominated from each of the United Nations regions, and is chaired by the Chair of IPBES.
Bush encroachment	An increase in density of shrubby or bushy tree vegetation in savannah or grassland systems.
Bushmeat	Meat for human consumption derived from wild animals.
Bushmeat (or wild meat) hunting	A form of subsistence hunting that entails the harvesting of wild animals for food and for non-food purposes, including for medicinal use.
Cap-and-trade	An economic policy instrument in which the State sets an overall environmental target (the cap) and assigns environmental impact allowances (or quotas) to actors that they can trade among each other.
Capacity-building (or capacity development)	Defined by the United Nations Development Programme as “the process through which individuals, organisations and societies obtain, strengthen and maintain their capabilities to set and achieve their own development objectives over time”. IPBES promotes and facilitates capacity-building, to improve the capacity of countries to make informed policy decisions on biodiversity and ecosystem-services.
Carbon cycle	The process by which carbon is exchanged among the ecosystems of the Earth.
Carbon sequestration	The long-term storage of carbon in plants, soils, geologic formations, and the ocean. Carbon sequestration occurs both naturally and as a result of anthropogenic activities and typically refers to the storage of carbon that has the immediate potential to become carbon dioxide gas.
Carbon storage	The technological process of capturing waste carbon dioxide from industry or power generation, and storing it so that it will not enter the atmosphere.
Carrying capacity	In ecology, the carrying capacity of a species in an environment is the maximum population size of the species that the environment can sustain indefinitely. The term is also used more generally to refer to the upper limit of habitats, ecosystems, landscapes, waterscapes or seascapes to provide tangible and intangible goods and services (including aesthetic and spiritual services) in a sustainable way.
Catalogue of policy support tools and methodologies	The IPBES catalogue of policy support tools and methodologies is an evolving online resource with two main goals. The first goal is to enable decision-makers to gain easy access to information on policy support tools and methodologies to better inform and assist the different phases of policy-making and implementation. The second goal is to allow a range of users to provide input to the catalogue and assess the usability of tools and

	methodologies in their specific contexts, including resources required and types of outputs that can be obtained, thus helping to identify and bridge gaps with respect to available tools and methodologies.
Causal chains	When the cause produces its effects in a remote and indirect manner, an explanation has to rely on causal chains, i.e., a continuous chain of causal mechanisms, where each step links a cause or combination of causes with its direct outcome, the latter being a direct cause of the subsequent outcome.
Causal effect	A causal effect can be defined in many ways, but essentially it amounts to the change in an outcome Y brought about by the change in a factor X. If X is a cause of Y then knowing something about X should help to predict something about Y that cannot be provided by another variable.
Certainty	In the context of IPBES, the summary terms to describe the state of knowledge are the following: • Well established (certainty term): comprehensive meta-analysis or other synthesis or multiple independent studies that agree. • Established but incomplete (certainty term): general agreement although only a limited number of studies exist but no comprehensive synthesis and, or the studies that exist imprecisely address the question. • Unresolved (certainty term): multiple independent studies exist but conclusions do not agree. • Inconclusive (certainty term): limited evidence, recognising major knowledge gaps
Civil society	A grouping that is broader than the institutionally recognized organizations, unions, associations and other pressure groups. The Internet and other new information and communication technologies facilitate the rise of self-organized, leaderless movements, allowing a rapid and efficient mobilization of citizens to understand and change the world around them.
Clean Development Mechanism (CDM)	Defined in Article 12 of the Protocol, allows a country with an emission-reduction or emission-limitation commitment under the Kyoto Protocol (Annex B Party) to implement an emission-reduction project in developing countries. Such projects can earn saleable certified emission reduction (CER) credits, each equivalent to one tone of CO2, which can be counted towards meeting Kyoto targets.
Climate change	Refers to a change of climate that is attributed directly or indirectly to human activity that alters the composition of the global atmosphere and that is in addition to natural climate variability observed over comparable time periods.
Climate change adaptation	Adjustment in natural or human systems in response to actual or expected climatic stimuli or their effects, which moderates harm or exploits beneficial opportunities.
Climate change mitigation	Climate change mitigation consists of actions to limit the magnitude or rate of long-term climate change. Climate change mitigation generally involves reductions in human (anthropogenic) emissions of greenhouse gases (GHGs). Mitigation may also be achieved by increasing the capacity of carbon sinks, e.g., through reforestation. Mitigation policies can substantially reduce the risks associated with human-induced global warming.

Climate envelope	A subset of the more general family of species distribution models that correlate species occurrence or abundance with climate variables to make spatially-explicit predictions of potential distribution.
Climate regulation	The influence of land cover and biological mediated processes that regulate atmospheric processes and weather patterns which in turn create the microclimate in which different plants and animals (including humans) live and function.
Climate smart agriculture	Aims to tackle three main objectives: sustainably increasing agricultural productivity and incomes; adapting and building resilience to climate change; and reducing and/or removing greenhouse gas emissions, where possible.
Co-management	Process of management in which government shares power with resource users, with each given specific rights and responsibilities relating to information and decision-making.
Comminution	The action of reducing a material, especially a mineral ore, to minute particles or fragments.
Commons	A concept whereby some forms of wealth belong to all, and that these community resources must be actively protected and managed for the good of all. It consists of land and services of common property (forests, rivers, fields and arable land) used and managed by a given community (mainly traditional, local or indigenous). The commons also consist of gifts of nature such as air, oceans and wildlife ("global commons") as well as shared social creations such as libraries, public spaces, scientific research and creative works. See also "Common Pool Resources" and "Tragedy of the commons".
Common Pool Resource (CPR)	Resources for which the exclusion of users is difficult (referred to as excludability), and the use of such a resource by one user decreases resource benefits for other users (referred to as subtractability). Common CPR examples include fisheries, forests, irrigation systems, and pastures. Global CPR examples include the earth's oceans and atmosphere. Difficulty in excluding users, combined with a CPR's subtractability, create management vulnerabilities that can result in resource degradation, often referred to as the "tragedy of the commons". See also "Tragedy of commons" and "Commons".
Community-based natural resource management (CBNRM)	An approach to natural resource management that involves the full participation of indigenous peoples' and local communities and resource users in decision-making activities, and the incorporation of local institutions, customary practices, and knowledge systems in management, regulatory, and enforcement processes. Under this approach, community-based monitoring and information systems are initiatives by indigenous peoples and local community organisations to monitor their community's well-being and the state of their territories and natural resources, applying a mix of traditional knowledge and innovative tools and approaches.
Concepts	The second stage of cognitive process. Perceptions are selected, organized, classified and hierarchized into concepts. This process is influenced by collective filters which are human systems of values, norms, and beliefs. Concepts do not come alone, but as integrated networks. See also 'Reality'; "Perceptions"; "Worldviews".
Conceptual Framework	The Platform's conceptual framework is a tool for building shared understanding across disciplines, knowledge systems and stakeholders of

	the interplay between biodiversity and ecosystem drivers, and of the role they play in building a good quality of life.
Confidence	See “certainty”.
Conservation agriculture	Approach to managing agro-ecosystems for improved and sustained productivity, increased profits and food security while preserving and enhancing the resource base and the environment. It is characterized by three linked principles, namely: (i) continuous minimum mechanical soil disturbance; (ii) permanent organic soil cover; and (iii) diversification of crop species grown in sequences and/or associations. This covers a wide range of approaches from minimum till to permaculture/“mimicking nature”.
Conservation tender (or conservation auction)	A financial mechanism to deliver funding to community groups and individuals for conservation works and, sometimes, permanently protect biodiversity (Australian Government, Department of the Environment and Energy).
Contaminant	Substance or agent present in the soil as a result of human.
Corridor	A geographically-defined area which allows species to move between landscapes, ecosystems and habitats, natural or modified, and ensures the maintenance of biodiversity and ecological and evolutionary processes.
Cost-benefit analysis	A technique designed to determine the feasibility of a project or plan by quantifying its costs and benefits.
Cropland	A land cover/use category that includes areas used for the production of crops for harvest.
Cross-scale analysis	Cross-scale effects are the result of spatial and/or temporal processes interacting with other processes at another scale. These interactions create emergent effects that can be difficult to predict.
Cross-sectoral	Relating to interactions between sectors (that is, the distinct parts of society, or of a nation's economy), such as how one sector affects another sector, or how a factor affects two or more sectors.
Cultural (ecosystem) services	The Millennium Ecosystem Assessment (Sarukhán & Whyte, 2005) defined cultural ecosystem services as “the nonmaterial benefits people obtain from ecosystems through spiritual enrichment, cognitive development, reflection, recreation, and aesthetic experiences”. Cultural ecosystem services have been included in many other typologies of ecosystem services and referred to variously as cultural services (Constanza, 1997), life-fulfilling functions (Daily, 1999), information functions (de Groot et al., 2002), amenities and fulfilment (Boyd & Banzhaf, 2007), cultural and amenity services (de Groot *et al.,* 2010, Kumar 2010), or socio-cultural fulfilment (Wallace, 2007).
Cumulative impacts	An impact produced over a period of time.
Customary law	Law based on tradition in communities where the authority of traditional leadership is recognised. It exists where there is a commonly repeated practice which is accepted as law by the members of a community.
Customary practices	See "Customary law".
Decision support tools	Approaches and techniques based on science and other knowledge systems, including indigenous and local knowledge, that can inform, assist and enhance relevant decisions, policy-making and implementation at the local, national, regional and international levels.

Decomposition	Breakdown of complex organic substances into simpler molecules or ions by physical, chemical and/or biological processes.
Deflation (wind)	Wind erosion.
Deforestation	Human-induced conversion of forested land to non-forested land. Deforestation can be permanent, when this change is definitive, or temporary when this change is part of a cycle that includes natural or assisted regeneration.
Degraded land	Land in a state that results from persistent decline or loss of biodiversity and ecosystem functions and services that cannot fully recover unaided.
Degrowth (or downscaling)	A theoretical frame invoking the necessity of downscaling and re-localizing production.
Denitrification	A heterotrophic process of anaerobic microbial respiration conducted by bacteria. Denitrification is the microbial oxidation of organic matter in which nitrate or nitrite is the terminal electron acceptor, and the end product is N2.
Densification	The increase in woody plants in a savanna, grassland or woodland.
Depositional sites	The places where eroded soils are deposited.
Desertification	Land degradation in arid, semi-arid and dry sub-humid areas resulting from various factors, including climatic variations and human activities.
Direct driver	See “driver”.
Disability-Adjusted Life Year (DALY)	One DALY can be thought of as one lost year of "healthy" life. The sum of these DALYs across the population, or the burden of disease, can be thought of as a measurement of the gap between current health status and an ideal health situation where the entire population lives to an advanced age, free of disease and disability.
Disaster Risk Reduction	The concept and practice of reducing disaster risks through systematic efforts to analyze and manage the causal factors of disasters, including through reduced exposure to hazards, lessened vulnerability of people and property, wise management of land and the environment, and improved preparedness for adverse events.
Downscaling	The transformation of information from coarser to finer spatial scales through statistical modelling or spatially nested linkage of structural models.
Drivers	In the context of IPBES, drivers of change are all the factors that, directly or indirectly, cause changes in nature, anthropogenic assets, nature’s contributions to people and a good quality of life. Direct drivers of change can be both natural and anthropogenic. **Direct drivers** have direct physical (mechanical, chemical, noise, light etc.) and psychological (disturbance etc.) impacts on nature and its functioning, and on people and their interaction. Direct drivers unequivocally influence biodiversity and ecosystem processes. They are also referred to as ‘pressures’. Direct drivers include, inter alia, climate change, pollution, land use change, invasive alien species and zoonoses, including their effects across regions. **Indirect drivers** are drivers that operate diffusely by altering and influencing direct drivers as well as other indirect drivers (also referred to as ‘underlying causes’). Interactions between indirect and direct drivers create different chains of relationship, attribution, and impacts, which may vary according to type, intensity, duration, and distance. These relationships can also lead to

	different types of spill-over effects. Global indirect drivers include economic, demographic, governance, technological and cultural ones, among others. Special attention is given, among indirect drivers, to the role of institutions (both formal and informal) and impacts of the patterns of production, supply and consumption on nature, nature's contributions to people and good quality of life.
Dry forest	Tropical and sub-tropical dry forests occur in climates that are warm year-round, and may receive several hundred centimetres or rain per year, they deal with long dry seasons which last several months and vary with geographic location.
Drylands	Tropical and temperate areas with an aridity index (annual rainfall/annual potential evaporation) of less than 0.65.
Ecocentrism (or biocentrism)	A concept that nature and natural things have a value in and of themselves, independent of any benefits they may have for human beings. See also "Anthropocentrism" and "Reality".
Ecological (or socio-ecological) breakpoint or threshold	The point at which a relatively small change in external conditions causes a rapid change in an ecosystem. When an ecological threshold has been passed, the ecosystem may no longer be able to return to its state by means of its inherent resilience.
Ecological footprint	A measure of the amount of biologically productive land and water required to support the demands of a population or productive activity. Ecological footprints can be calculated at any scale: for an activity, a person, a community, a city, a region, a nation or humanity as a whole.
Ecological infrastructure	The natural or semi-natural structural elements of ecosystems and landscapes that are important in delivering ecosystem services. It is similar to "green infrastructure", a term sometimes applied in a more urban context. The ecological infrastructure needed to support pollinators and improve pollination services includes patches of semi-natural habitats, including hedgerows, grassland and forest, distributed throughout productive agricultural landscapes, providing nesting and floral resources. Larger areas of natural habitat are also ecological infrastructure, although these do not directly support agricultural pollination in areas more than a few kilometres away from pollinator-dependent crops.
Ecological integrity	The ability of an ecosystem to support and maintain ecological processes and a diverse community of organisms.
Ecological marginalization	The take-over of local natural resources by private and/or state interests, and the gradual or immediate disorganization of the ecosystem via withdrawals and additions.
Ecological solidarity	As explained by Thompson *et al.* (2011): "From ecology based on interactions to solidarity based on links between individuals united around a common goal and conscious of their common interests and their moral obligation and responsibility to help others, we define ecological solidarity as the reciprocal interdependence of living organisms amongst each other and with spatial and temporal variation in their physical environment".
Economic and financial instruments	Economic and financial instruments can be used to change people's behaviour towards desired policy objectives. Instruments typically encompass a wide range of designs and implementation approaches. They include traditional fiscal instruments, including for example subsidies, taxes, charges and fiscal transfers. Additionally, instruments such as tradable

	pollution permits or tradable land development rights rely on the creation of new markets. Further instruments represent conditional and voluntary incentive schemes such as payments for ecosystem services. All these can in principle be used to correct for policy or/and market failures and reinstate full-cost pricing. They aim at reflecting social costs or benefits of the conservation and use of biodiversity and ecosystem services of a public good nature ("getting the price right"). Financial instruments, in contrast, are often extra-budgetary and can be financed from domestic sources or foreign aid, external borrowing, debt for nature swaps and so on. It should be noted that economic instruments do not necessarily imply that commodification of environmental functions is promoted. Generally, they are meant to change behaviour of individuals (e.g., consumers and producers) and public actors (e.g., local and regional governments).
Economic valuation	See "values".
Ecoregion	A large area of land or water that contains a geographically distinct assemblage of natural communities that: (a) Share a large majority of their species and ecological dynamics; (b) Share similar environmental conditions, and; (c) Interact ecologically in ways that are critical for their long-term persistence. In contrast to biomes, an ecoregion is generally geographically specific, at a much finer scale. For example, the "East African Montane Forest" ecoregion of Kenya (WWF ecoregion classification) is a geographically specific and coherent example of the globally occurring "tropical and subtropical forest" biome.
Ecosystem	A dynamic complex of plant, animal and micro-organism communities and their non-living environment interacting as a functional unit.
Ecosystem degradation	A persistent (long-time) reduction in the capacity to provide ecosystem services.
Ecosystem function(s)	The flow of energy and materials through the biotic and abiotic components of an ecosystem. It includes many processes such as biomass production, trophic transfer through plants and animals, nutrient cycling, water dynamics and heat transfer.
Ecosystem health	A state or condition of an ecosystem that expresses attributes of biodiversity within "normal" ranges, relative to its ecological stage of development. Ecosystem health depends inter alia on ecosystem resilience and resistance. Note that there is no universally accepted benchmark for a healthy ecosystem. Rather, the apparent health status of an ecosystem can vary, depending upon which metrics are employed in judging it, and which societal aspirations are driving the assessment.
Ecosystem management	An approach to maintaining or restoring the composition, structure, function and delivery of services of natural and modified ecosystems for the goal of achieving sustainability. It is based on an adaptive, collaboratively developed vision of desired future conditions that integrates ecological, socioeconomic, and institutional perspectives, applied within a geographic framework, and defined primarily by natural ecological boundaries.
Ecosystem services	The benefits people obtain from ecosystems. In the Millennium Ecosystem Assessment, ecosystem services can be divided into supporting, regulating, provisioning and cultural.

Ecotone	A transition area between two biomes or vegetation types.
Ecotourism	Sustainable travel undertaken to access sites or regions of unique natural or ecological quality, promoting their conservation, low visitor impact, and socio-economic involvement of local populations.
Enabling conditions	The institutional, policy and governance responses to create enabling conditions to implement direct responses or actions on the ground to halt land degradation or to restore degraded lands.
Endemism	The ecological state of a species being unique to a defined geographic location, such as an island, nation, country or other defined zone, or habitat type; organisms that are indigenous to a place are not endemic to it if they are also found elsewhere.
Energy security	Access to clean, reliable and affordable energy services for cooking and heating, lighting, communications and productive uses.
Environmental hazards	The potential occurrence of a natural or human-induced physical event or trend or physical impact that may cause loss of life, injury, or other health impacts, as well as damage and loss to property, infrastructure, livelihoods, service provision, ecosystems and environmental resources. In this report, the term hazard usually refers to climate-related physical events or trends or their physical impacts.
Environmental Impact Assessment (EIA)	An assessment that assesses the impacts of planned activity on the environment in advance, thereby allowing avoidance measures to be taken: prevention is better than cure.
Environmental incomes	An extraction from non-cultivated sources: natural forests, other non-forest wildlands such as grass-, bush- and wetlands, fallows, but also wild plants and animals harvested from croplands.
Environmental Kuznets Curves (EKC)	A hypothesized relationship between environmental quality and economic development: various indicators of environmental degradation tend to get worse as modern economic growth occurs until average income reaches a certain point over the course of development.
Epizootics	A disease outbreak affecting a species' population at the same time.
Erodibility	The ease with which a soil erodes, defined by its resistance to two energy sources: the impact of raindrops on the soil surface, and the shearing action of runoff between clods in grooves or rills.
Erosion hotspots	Places identified with as having a high erosion potential
Eutrophic (or eutrophication)	A condition of an aquatic system in which increased nutrient loading leads to progressively increasing amounts of algal growth and biomass accumulation. When the algae die off and decompose, the amount of dissolved oxygen in the water becomes reduced. In lakes, eutrophication leads to seasonal algal blooms, reduced water clarity, and, often, periodic fish mortality as a consequence of oxygen depletion.
Ex-ante assessment	The use of policy-screening scenarios to forecast the effects of alternative policy or management options (interventions) on environmental outcomes.
Ex-post assessment	The use of policy-evaluation scenarios to assess the extent to which outcomes actually achieved by an implemented policy match those expected based on modelled projections, thereby informing policy review.

Extensive grazing (lands)	A form of grazing in which livestock are raised on food that comes mainly from natural grasslands, shrublands, woodlands, wetlands and deserts. It differs from intensive grazing, where the animal feed comes mainly from artificial, seeded pastures.
Externality	A positive or negative consequence (benefits or costs) of an action that affects someone other than the agent undertaking that action and for which the agent is neither compensated nor penalized through the markets.
Extinction debt	The future extinction of species due to events in the past, owing to a time lag between an effect such as habitat destruction or climate change, and the subsequent disappearance of species.
Fire regime	A term used to describe the characteristics of fires that occur in a particular ecosystem over a period of time. Fire regimes are characterized based on a combination of factors including the frequency, intensity, size, pattern, season and severity of fires.
Food security	When all people, at all times, have physical, social and economic access to sufficient, safe and nutritious food that meets their dietary needs and food preferences for an active and healthy life.
Food self-sufficiency	The ability of a region or country to produce enough food (especially staple crops) without needing to buy or import additional food.
Food sovereignty (paradigm)	The right to define own policies and strategies for the sustainable production, distribution and consumption of food that guarantee the right to food for the entire population, on the basis of small and medium-sized production, respecting their own cultures and the diversity of peasant, fishing and indigenous forms of agricultural production, marketing and management of rural areas, in which women play a fundamental role.
Forest	A minimum area of land of 0.05 - 1.0 hectares with tree crown cover (or equivalent stocking level) of more than 10–30 per cent with trees with the potential to reach a minimum height of 2–5 m at maturity in situ. A forest may consist either of closed forest formations where trees of various stories and undergrowth cover a high proportion of the ground or open forest.
Forest transition	A shift, usually assessed at the national scale, from net forest loss to net forest gain, whether through natural recovery or planted forests.
Fossil fuel	Fuels such as petroleum derived for fossil oil sources.
Functional diversity	Value, range and relative abundance of functional traits in a given ecosystem.
General Circulation Models (GCMs)	A numerical representation of the physical processes in the atmosphere, ocean, cryosphere and land surface based on the physical, chemical and biological properties of their components, their interactions and feedback processes, and accounting for all or some of its known properties.
Geographic Information Systems (GIS)	A computer-based tool that analyses, stores, manipulates and visualizes geographic information on a map.
Geographic range	The geographic range of a species is the geographic boundary within which it occurs.
Gini index	In economics, the Gini coefficient (sometimes expressed as a Gini ratio or a normalized Gini index) is a measure of statistical dispersion intended to

	represent the income or wealth distribution of a nation's residents, and is the most commonly used measure of inequality.
Good quality of life	Within the context of the IPBES conceptual framework – the achievement of a fulfilled human life, a notion which may vary strongly across different societies and groups within societies. It is a context-dependent state of individuals and human groups, comprising aspects such as access to food, water, energy and livelihood security, and also health, good social relationships and equity, security, cultural identity, and freedom of choice and action. "Living in harmony with nature", "living-well in balance and harmony with Mother Earth" and "human well-being" are examples of different perspectives on a "good quality of life".
Governance	The way the rules, norms and actions in a given organization are structured, sustained and regulated.
Grassland	A land cover class that includes any geographic area dominated by natural herbaceous plants (grasslands, prairies, steppes and savannahs) with a cover of 10% or more, irrespective of different human and/or animal activities (e.g., grazing).
Grazing land management	The strategies used by people to promote both high quality and quantity of forage for domesticated livestock.
Green Revolution	A set of research and the development of technology transfer initiatives occurring between the 1930s and the late 1960s (with prequels in the work of the agrarian geneticist Nazareno Strampelli in the 1920s and 1930s), that increased agricultural production worldwide, particularly in the developing world, beginning most markedly in the late 1960s. The initiatives resulted in the adoption of new technologies, including: new, high-yielding varieties (HYVs) of cereals, especially dwarf wheats and rices, in association with chemical fertilizers and agro-chemicals, and with controlled water-supply (usually involving irrigation) and new methods of cultivation, including mechanization. All of these together were seen as a "package of practices" to supersede "traditional" technology and to be adopted as a whole.
Green water	Water transpired through plants to the atmosphere.
Greenhouse Gas	Those gaseous constituents of the atmosphere, both natural and anthropogenic, that absorb and emit radiation at specific wavelengths within the spectrum of infrared radiation emitted by the Earth's surface, the atmosphere, and clouds. This property causes the greenhouse effect.
Grey water	Any wastewater that is not contaminated with faecal matter.
Gross primary production (GPP)	Total terrestrial Gross Primary Production (GPP) is the total mass of carbon taken out of the atmosphere by plant photosynthesis.
Habitat	The place or type of site where an organism or population naturally occurs. Also used to mean the environmental attributes required by a particular species or its ecological niche.
Habitat connectivity	The degree to which the landscape facilitates the movement of organisms (animals, plant reproductive structures, pollen, pollinators, spores and so on) and other environmentally important resources (e.g., nutrients and moisture) between similar habitats. Connectivity is hampered by fragmentation.
Habitat degradation	A general term describing the set of processes by which habitat quality is reduced. Habitat degradation may occur through natural processes (e.g.

	drought, heat, cold) and through human activities (forestry, agriculture, urbanization).
Habitat ecosystem functions	The ability of soil or soil materials to serve as a habitat for micro-organisms, plants, soil-living animals and their interactions.
Habitat fragmentation	A general term describing the set of processes by which habitat loss results in the division of continuous habitats into a greater number of smaller patches of lesser total and isolated from each other by a matrix of dissimilar habitats. Habitat fragmentation may occur through natural processes (e.g., forest and grassland fires, flooding) and through human activities (forestry, agriculture, urbanization).
Habitat Service	The importance of ecosystems to provide living space for resident and migratory species (thus maintaining the gene pool and nursery service).
Homogenization	When used in the ecological sense “homogenization” means a decrease in the extent to which communities differ in species composition.
Human appropriation of net primary production (HANPP)	The aggregate impact of land use on biomass available each year in ecosystems.
Human capital	All the knowledge, talents, skills, abilities, experience, intelligence, training, judgment and wisdom possessed individually and collectively by individuals in a population.
Human rights	Rights inherent to all human beings, regardless of race, colour, sex, language, religion or other opinion, national or social origin, property, birth or any other status. These rights are interrelated, interdependent and indivisible.
Human rights-based approach	A conceptual framework for the process of human development that is normatively based on international human rights standards and operationally directed to promoting and protecting human rights.
Human Rights Instruments	Instruments for the protection and promotion of human rights, including general instruments, instruments concerning specific issues, and instruments relating to the protection of particular groups.
Humification	Decomposition of organic material followed by a synthesis of humic substances.
Humanistic economics	Humanistic economics intend to show that humankind is perfectly capable of living without the profit motive, and has done so for most of its history. It goes again the tendency to consider the profit motive as self-evident, an idea that underlies many political decisions. See also "Behavioural economics".
Hydraulic fracturing (or fracking)	An oil and gas well development process that typically involves injecting water, sand, and chemicals under high pressure into a bedrock formation via the well. This process is intended to create new fractures in the rock as well as increase the size, extent, and connectivity of existing fractures. Hydraulic fracturing is a well-stimulation technique used commonly in low-permeability rocks like tight sandstone, shale, and some coal beds to increase oil and/or gas flow to a well from petroleum-bearing rock formations.
Immaterial patrimony	Non-tangible aspects of cultural value that are passed from one human generation to the next.

Impact assessment	A formal, evidence-based procedure that assesses the economic, social and environmental effects of public policy or of any human activity
Indicators	A quantitative or qualitative factor or variable that provides a simple, measurable and quantifiable characteristic or attribute responding in a known and communicable way to a changing environmental condition, to a changing ecological process or function, or to a changing element of biodiversity.
Indigenous and local knowledge (ILK) systems	Social and ecological knowledge practices and beliefs pertaining to the relationship of living beings, including people, with one another and with their environments. Such knowledge can provide information, methods, theory and practice for sustainable ecosystem management.
Indigenous peoples and local communities (IPLC)	Typically, ethnic groups who are descended from and identify with the original inhabitants of a given region, in contrast to groups that have settled, occupied or colonized the area more recently. IPBES does not intend to create or develop new definitions of what constitutes “indigenous peoples and local communities”.
Indirect driver	See “driver”.
Institution	Encompasses all formal and informal interactions among stakeholders and social structures that determine how decisions are taken and implemented, how power is exercised, and how responsibilities are distributed.
Institutional competencies	The set of abilities which a given institution can use to achieve policy goals. Examples include the ability to collaborate with local communities, design scientifically sound restoration interventions, or foresee secondary effects of policies.
Integrated assessment model (IAM)	Interdisciplinary models that aim to describe the complex relationships between environmental, social, and economic drivers that determine current and future state of the ecosystem and the effects of global change, in order to derive policy-relevant insights. One of the essential characteristics of integrated assessments is the simultaneous consideration of the multiple dimensions of environmental problems.
Integrated landscape management	Refers to long-term collaboration among different groups of land managers and stakeholders to achieve the multiple objectives required from the landscape.
Integrated pest management (IPM) (or integrated pest control)	A broadly-based approach that integrates various practices for economic control of pests. IPM aims to suppress pest populations below the economic injury level (i.e., to below the level that the costs of further control outweigh the benefits derived). It involves careful consideration of all available pest control techniques and then integration of appropriate measures to discourage development of pest populations while keeping pesticides and other interventions to economically justifiable levels with minimal risks to human health and the environment. IPM emphasizes the growth of a healthy crop with the least possible disruption to agro-ecosystems and encourages natural pest control mechanisms.
Invasive alien species	Species whose introduction and/or spread by human action outside their natural distribution threatens biological diversity, food security, and human health and well-being. “Alien” refers to the species’ having been introduced outside its natural distribution (“exotic”, “non-native” and “non-indigenous” are synonyms for “alien”). “Invasive” means “tending to expand into and modify ecosystems to which it has been introduced”. Thus, a species may be

	alien without being invasive, or, in the case of a species native to a region, it may increase and become invasive, without actually being an alien species.
Intensive grazing lands	Grazing lands that are managed primarily for livestock production with few other uses of the land other than dispersed crops.
IPBES conceptual framework	The IPBES conceptual framework has been designed to build shared understanding across disciplines, knowledge systems and stakeholders of the interplay between biodiversity and ecosystem drivers, and of the role they play in building a good quality of life.
Knowledge systems	A body of propositions that are adhered to, whether formally or informally, and are routinely used to claim truth. They are organized structures and dynamic processes (a) generating and representing content, components, classes, or types of knowledge, that are (b) domain-specific or characterized by domain-relevant features as defined by the user or consumer, (c) reinforced by a set of logical relationships that connect the content of knowledge to its value (utility), (d) enhanced by a set of iterative processes that enable the evolution, revision, adaptation, and advances, and (e) subject to criteria of relevance, reliability and quality.
Land abandonment	Land abandonment occurs when a particular land use ceases, and there is no clearly-defined subsequent land use practice. It is often associated with poorly defined ownership and/or land use governance.
Land cover	The observed (bio)physical cover on the earth's surface.
Land degradation	Refers to the many processes that drive the decline or loss in biodiversity, ecosystem functions or services and includes the degradation of all terrestrial ecosystems.
land degradation neutrality	A state whereby the amount of healthy and productive land resources, necessary to support ecosystem services, remains stable or increases within specified temporal and spatial scales.
Land grabbing	The large-scale acquisition of land (especially in developing countries), driven primarily by concerns about food and energy security of high-income countries and often executed by the private sector.
Land sharing	A situation where low-yield farming enables biodiversity to be maintained within agricultural landscapes.
Land sparing	Land sparing, also called "land separation" involves restoring or creating non-farmland habitat in agricultural landscapes at the expense of field-level agricultural production - for example, woodland, natural grassland, wetland, and meadow on arable land. This approach does not necessarily imply high-yield farming of the non-restored, remaining agricultural land. (From Rey Benayas & Bullock, 2012). See also “Conservation agriculture” in this Glossary.
Land tenure	The relationship, whether legally or customarily defined, among people, as individuals or groups, with respect to land.
Land transformation	A process whereby the biotic community of an area is substantially altered or substituted by another, along with the underlying ecological and human processes responsible for its persistence, often as a result of a deliberate decision to change the purpose for which the land is used.
Land use	The human use of a specific area for a certain purpose (such as residential, agriculture, recreation, industrial, and so on). Influenced by, but not

	synonymous with, land cover. Land-use change refers to a change in the use or management of land by humans, which may lead to a change in land cover.
Landscape	A human-defined area ranging in size from c. 3 km2 to c. 3002 km. Landscape is spatially heterogeneous in at least one factor of interest and often consists of a mosaic of interacting ecosystems.
Landscape socio-ecological approach	The landscape scale approach incorporates the socio-ecological system, including natural and human-modified ecosystems, influenced by ecological, historical, economic, and socio-cultural processes. The landscape includes an array of stakeholders small enough to be manageable, but large enough to deliver multiple functions for stakeholders with differing interests.
Livelihood resilience	The capacity of all people across generations to sustain and improve their livelihood opportunities and well-being despite environmental, economic, social and political disturbances.
Livelihood security	Adequate and sustainable access to income and resources to meet basic needs (including adequate access to food, potable water, health facilities, educational opportunities, housing, time for community participation and social integration).
Mangrove	Group of trees and shrubs that live in the coastal intertidal zone. Mangrove forests only grow at tropical and subtropical latitudes near the equator because they cannot withstand freezing temperatures.
Marginal lands	Lands less suited for crop or livestock production.
Mass balance (analysis)	Comparison between input and output mass of materials to solve for losses such as oxidation.
Meta-analysis	A quantitative statistical analysis of several separate but similar experiments or studies in order to test the pooled data for statistical significance.
Millennium Ecosystem Assessment	The Millennium Ecosystem Assessment is a major assessment of the human impact on the environment published in 2005.
Mineral resource extraction	The removal of a mineral resource in or on the Earth's crust, which has appropriate form, quality and quantity to allow economic extraction.
Mineralization	Mineralization in soil science is the decomposition or oxidation of the chemical compounds in organic matter releasing the nutrients contained in those compounds into soluble inorganic forms that may be plant-accessible.
Mitigation	In the context of IPBES, an intervention to reduce negative or unsustainable uses of biodiversity and ecosystems.
Models	Qualitative or quantitative representations of key components of a system and of relationships between these components. Benchmarking (of models) is the process of systematically comparing sets of model predictions against measured data in order to evaluate model performance. Validation (of models) typically refers to checking model outputs for consistency with observations. However, since models cannot be validated in the formal sense of the term (i.e. proven to be true), some scientists prefer to use the words "benchmarking" or “evaluation". A dynamic model is a model that describes changes through time of a specific process.

A process-based model (also known as "mechanistic model") is a model in which relationships are described in terms of explicitly stated processes or mechanisms based on established scientific understanding, and model parameters therefore have clear ecological interpretation, defined beforehand.

Hybrid models are models that combine correlative and process-based modelling approaches.

A correlative model (also known as "statistical model") is a model in which available empirical data are used to estimate values for parameters that do not have predefined ecological meaning, and for which processes are implicit rather than explicit.

Integrated assessment models are interdisciplinary models that aim to describe the complex relationships between environmental, social, and economic drivers that determine current and future state of the ecosystem and the effects of global change, in order to derive policy-relevant insights. One of the essential characteristics of integrated assessments is the simultaneous consideration of the multiple dimensions of environmental problems.

Monitoring The repeated observation of a system in order to detect signs of change.

Monoculture The agricultural practice of producing or growing a single crop, plant, or livestock species, variety, or breed in a field or farming system at a time.

Moral economy A moral economy, initially based on peasants' sense of belonging and sharing, is an economy that is based on goodness, fairness, and justice. Such an economy is generally only stable in small, closely knit communities, where the principles of mutuality operate.

Mosaic restoration Landscape scale restoration efforts that do not rely on a single restoration mechanism for an entire landscape, or it is a single mechanism, deploying it in a spatially variable manner that creates patches of restored and non-restored landscape units.

Mother Earth An expression used in a number of countries and regions to refer to the planet Earth and the entity that sustains all living things found in nature with which humans have an indivisible, interdependent physical and spiritual relationship (see "nature").

Multifunctional agriculture The concept was adopted by FAO (1999) to foster an approach integrating landscape, biological connections, and less damageable practices. Multifunctional agriculture is meant to integrate the economic, social and ecological aspects of land management.

Native forests Forests that are made up of native tree species, and are either primary (have never been clear-cut) or secondary (regenerating naturally).

Native species Indigenous species of animals or plants that naturally occur in a given region or ecosystem.

Natural capital The world's stocks of natural assets which include geology, soil, air, water and all living things. It is from this natural capital that humans derive a wide range of services, often called ecosystem services, which make human life possible.

Natural Capital Accounts (NCA) Sets of linked accounts that contain information about the type and quantities and, where possible, the value of the stocks of natural assets and

	the flows of services generated by them (ONS, 2017,). The accounts contain two main components: physical accounts - types, quantities and condition of assets; and monetary accounts - application of monetary units of valuation to selected flows of services on an annual basis and associated values of stocks
Naturalized species/naturalization	A species that, once it is introduced outside its native distributional range, establishes self-sustaining populations.
Nature	In the context of the Platform, refers to the natural world with an emphasis on its living components. Within the context of Western science, it includes categories such as biodiversity, ecosystems (both structure and functioning), evolution, the biosphere, humankind’s shared evolutionary heritage, and biocultural diversity. Within the context of other knowledge systems, it includes categories such as Mother Earth and systems of life, and it is often viewed as inextricably linked to humans, not as a separate entity (see “Mother Earth”).
Nature’s non-material benefits	Benefits from nature that do not take a physical form such as spiritual enrichment, intellectual development, recreation and aesthetic values.
Nature's contribution to people (NCP)	All the contributions, both positive and negative, of nature (i.e., biodiversity, ecosystems, and their associated ecological and evolutionary processes) to good quality of life of people. Beneficial contributions from nature include such things as food provision, water purification, flood control, and artistic inspiration, whereas detrimental contributions include disease transmission and predation that damages people or their assets. Many NCP may be perceived as benefits or detriments depending on the cultural, temporal or spatial context.
Near surface ozone	Ozone near the earth surface formed photochemically during the oxidation of hydrocarbons in the presence of nitrogen oxides.
Net Biome Production (NBP)	The amount of carbon accumulating or lost in ecosystems at the regional scale is the Net Biome Production (NBP), defined as the NEP corrected for lateral transfers of carbon to adjacent biomes, due to process such as trade in agricultural products, export of organic matter in rivers and losses due to disturbances, including land clearing and wildfire.
Net Ecosystem Production (NEP)	The amount of NPP left in the ecosystem after the additional respiration by microbes and animals is the Net Ecosystem Production (NEP).
Net Positive Impact (NPI)	A net gain to biodiversity features measured in quality hectares (for habitats), number or percentage of individuals (for species), or other metrics appropriate to the feature.
Net Primary Production (NPP)	The total mass of carbon taken out of the atmosphere by plant photosynthesis (Gross Primary Production) minus return to the atmosphere of carbon due to autotrophic respiration.
Night Light Development Index (NLDI)	A spatially explicit and globally available empirical measurement of human development derived solely from night-time satellite imagery and population density.
Non-anthropogenic	A non-anthropocentric value is a value centred on something other than human beings. These values can be non-instrumental (e.g. a value ascribed to the existence of specific species for their own sake) or instrumental to non-human ends (e.g. the instrumental value a habitat has for the existence of a specific species).

Non-Indigenous Species or Non-native species or Alien species	See “Invasive Alien Species”.
Non-timber resource	A multitude of natural products (excluding timber) selectively harvested from the terrestrial environment for subsistence and commercial purposes.
Opportunity costs	"The added cost of using resources (as for production or speculative investment) that is the difference between the actual value resulting from such use and that of an alternative (such as another use of the same resources or an investment of equal risk but greater return)".
Organic agriculture	Any system that emphasizes the use of techniques such as crop rotation, compost or manure application, and biological pest control in preference to synthetic inputs. Most certified organic farming schemes prohibit all genetically modified organisms and almost all synthetic inputs. Its origins are in a holistic management system that avoids off-farm inputs, but some organic agriculture now uses relatively high levels of off-farm inputs.
Overstocking	Placing a number of animals on a given area that will result in overuse if continued to the end of the planned grazing period.
Paleological data	Information on environment event and trends (e.g., paleoclimate).
Participatory governance	A variant or subset of governance which puts emphasis on democratic engagement, in particular through deliberative practices.
Passive restoration	See “restoration”.
Participatory scenario development (and planning)	Approaches characterized by more interactive, and inclusive, involvement of stakeholders in the formulation and evaluation of scenarios. Aimed at improving the transparency and relevance of decision-making, by incorporating demands and information of each stakeholder, and negotiating outcomes between stakeholders.
Payments for Ecosystem Services (PES)	A payment mechanism that involves a series of payments to land or other natural resource owners in return for a guaranteed flow of ecosystem services or certain actions likely to enhance their provision over-and-above what would otherwise be provided in the absence of payment.
Peatland(s)	Wetlands which accumulate organic plant matter in situ because waterlogging prevents aerobic decomposition and the much slower rate of the resulting anaerobic decay is exceeded by the rate of accumulation.
Pedosphere	A part of the Earth's surface that contains the soil layer.
Perceptions	The first stage of the human cognitive process. Perceptions are not neutral as they pass through rational and emotional filters which assess and interpret the relevancy of what people see. These filters are conditioned by individual experience, education, and by collective worldviews. See also “Reality”; “Concepts”; “Worldviews”.
Permaculture	See "Conservation agriculture".
Permafrost	Perennially frozen ground that occurs wherever the temperature remains below 0°C for several years. Ground (soil or rock and included ice and organic material) that remains at or below 0°C for at least two consecutive years.
Permeability	The porosity of soils to allow water to pass through it.

Persistent organic pollutants (POPs)	Chemicals of global concern due to their potential for long-range transport, persistence in the environment, ability to bio-magnify and bio-accumulate in ecosystems, as well as their significant negative effects on human health and the environment.
Phenology	The timing of seasonal activities of animals and plants such as bud burst, flowering, fruiting, migration. Also used to refer to the study of such natural phenomena that recur periodically (e.g., development stages, migration) and their relation to climate and seasonal changes.
Phenotypic plasticity	An ability to alter growth form to suit current conditions without genetic change.
Plenary	Within the context of IPBES – the decision-making body comprising all of the members of IPBES.
Planetary boundaries	The safe operating space for humanity with respect to the Earth system and are associated with the planet's biophysical subsystems or processes.
Planning and zoning	Zoning is a planning control tool for regulating the built environment and creating functional real estate markets.
Plantation forests	Forests where trees have been deliberately planted (i.e., have not regenerated naturally) and are typically grown for the production of wood or fibre, in some cases they may replace grasslands or other non-forest vegetation types. They are often of exotic tree species.
Policy coherence	The systematic promotion of mutually reinforcing policy actions across government departments and agencies creating synergies towards achieving the agreed objectives.
Policy instrument	Set of means or mechanisms to achieve a policy goal.
Policy support tools	Approaches and techniques based on science and other knowledge systems that can inform, assist and enhance relevant decisions, policymaking and implementation at local, national, regional and global levels to protect nature, thereby promoting nature's benefits to people and a good quality of life.
Polluter-pays principle	The commonly accepted practice that those who produce pollution should bear the costs of managing it to prevent damage to human health or the environment. A polluter pays principle aims at preventing anybody from reaping the benefits at the expense of (or even considerable harm to) other members of the society.
Poverty	A state of deprivation that is multidimensional in nature. Poverty is more than the lack of income and resources to ensure a sustainable livelihood. Its manifestations include hunger and malnutrition, limited access to education and other basic services, social discrimination and exclusion as well as the lack of participation in decision-making
Precautionary principle	Pertains to risk management and states that if an action or policy has a suspected risk of causing harm to the public or to the environment, in the absence of scientific consensus that the action or policy is not harmful, the burden of proof that it is not harmful falls on those taking an action. The principle is used to justify discretionary decisions when the possibility of harm from making a certain decision (e.g., taking a particular course of action) is not, or has not been, established through extensive scientific knowledge. The principle implies that there is a social responsibility to

	protect the public from exposure to harm, when scientific investigation has found a plausible risk or if a potential plausible risk has been identified.
Preventive response	Conservation measures that maintain land and its environmental and productive functions.
Prior informed consent (PIC)	Consent given before access to knowledge or genetic resources takes place, based on truthful information about the use that will be made of the resources, which is adequate for the stakeholders or rights holders giving consent to understand the implications.
Protected area	A clearly defined geographical space, recognized, dedicated and managed, through legal or other effective means, to achieve the long-term conservation of nature with associated ecosystem services and cultural values.
Public -private partnerships (PPP)	A long-term contract between a private party and a government entity, for providing a public asset or service, in which the private party bears significant risk and management responsibility and remuneration is linked to performance.
Radiative forcing (RF)	The measurement of the capacity of a gas or other forcing agents to affect that energy balance, thereby contributing to climate change. Put more simply, RF expresses the change in energy in the atmosphere due to GHG emissions.
Ramsar site(s)	A wetland site designated of international importance especially as Waterfowl Habitat under the Ramsar Convention, an intergovernmental environment treaty established in 1975 by UNESCO, coming into force in 1975. Ramsar site refers to wetland of international significance in terms of ecology, botany, zoology, limnology or hydrology. Such a site meets at least one of the criteria of identifying wetlands of international importance set by Ramsar Convention and is designated by appropriate national authority to be added to Ramsar list.
Rangeland	Natural grasslands used for livestock grazing.
Reality	Current state of biodiversity and ecosystem functions independent of human knowledge and perceptions and ecosystem services (Nature in IPBES conceptual framework). See also “Perceptions”; “Concepts”; “Worldviews”.
Reclamation	The stabilization of the terrain, assurance of public safety, aesthetic improvement, and usually a return of the land to what, within the regional context, is considered to be a useful purpose.
REDD+	Reducing emissions from deforestation and forest degradation (REDD+) is a mechanism developed by Parties to the United Nations Framework Convention on Climate Change (UNFCCC). It creates a financial value for the carbon stored in forests by offering incentives for developing countries to reduce emissions from forested lands and invest in low-carbon paths to sustainable development. Developing countries would receive results-based payments for results-based actions. REDD+ goes beyond simply deforestation and forest degradation, and includes the role of conservation, sustainable management of forests and enhancement of forest carbon stocks.
Reforestation	Intentional replanting of trees and re-establishing a forest in areas that have been deforested.

Regime shift	Substantial reorganization in system structure, functions and feedbacks that often occurs abruptly and persists over time.
Rehabilitation	Restoration activities that may fall short of fully restoring a biotic community to its pre-degradation state, including natural regeneration and emergent ecosystems.
Remediation	Any action taken to rehabilitate ecosystems.
Renewable energy	Energy derived from natural processes (e.g., sunlight and wind) that are replenished at a faster rate than they are consumed. Solar, wind, geothermal, hydro, and some forms of biomass are common sources of renewable energy.
Replexity	Rapid and complex change.
Representative Concentration Pathways (RCPs)	Scenarios that include time series of emissions and concentrations of the full suite of greenhouse gases (GHGs) and aerosols and chemically active gases, as well as land use/land cover. The word representative signifies that each RCP provides only one of many possible scenarios that would lead to the specific radiative forcing characteristics. The term pathway emphasizes that not only the long-term concentration levels are of interest, but also the trajectory taken over time to reach that outcome. RCPs usually refer to the portion of the concentration pathway extending up to 2100, for which Integrated Assessment Models produced corresponding emission scenarios.
Resilience	The level of disturbance that an ecosystem or society can undergo without crossing a threshold to a situation with different structure or outputs. Resilience depends on factors such as ecological dynamics as well as the organizational and institutional capacity to understand, manage and respond to these dynamics.
Restoration	Any intentional activity that initiates or accelerates the recovery of an ecosystem from a degraded state. **Active restoration** includes a range of human interventions aimed at influencing and accelerating natural successional processes to recover biodiversity ecosystem service provision. **Passive restoration** includes reliance primarily on natural process of ecological succession to restore degraded ecosystems, but may include measures to protect a site from processes that currently prevent natural recovery (e.g., protection of degraded forests from overgrazing by livestock or unintentional human-induced fire).
Rewilding	Rewilding ensures natural processes and wild species play a much more prominent role in the land-and seascapes, meaning that after initial support, nature is allowed to take more care of itself. Rewilding helps landscapes become wilder, whilst also providing opportunities for modern society to reconnect with such wilder places for the benefits of all life.
Rotational grazing	A grazing scheme where animals are moved from one grazing unit (paddock) in the same group of grazing units to another without regard to specific graze: rest periods or levels of plant defoliation. cf. grazing system
Rubin Causal Model (RCM)	Also known as the Neyman–Rubin causal model, is an approach to the statistical analysis of cause and effect based on the framework of potential outcomes, named after Donald Rubin.
Salinization	The process of increasing the salt content in soil is known as salinization. Salinization can be caused by natural processes such as mineral weathering

	or by the gradual withdrawal of an ocean. It can also come about through artificial processes such as irrigation.
Savannah	Ecosystem characterized by a continuous layer of herbaceous plants, mostly grasses, and a discontinuous upper layer of trees that may vary in density.
Scale	The spatial, temporal, quantitative and analytical dimensions used to measure and study any phenomenon. The temporal scale is comprised of two properties: (i) temporal extent – the total length of the time period of interest for a particular study (e.g., 10 years, 50 years, or 100 years); and 2) temporal grain (or resolution) – the temporal frequency with which data are observed or projected within this total period (e.g. at 1-year, 5-year or 10-year intervals). The spatial scale is comprised of two properties: 1) spatial extent – the size of the total area of interest for a particular study (e.g., a watershed, a country, the entire planet); and (ii) spatial grain (or resolution) – the size of the spatial units within this total area for which data are observed or predicted (e.g., fine-grained or coarse-grained grid cells).
Scale paradox	Process in which land use outcomes vary (often counterintuitively) according to the geographic location and spatial scale under consideration.
Scenarios	Representations of possible futures for one or more components of a system, particularly for drivers of change in nature and nature's benefits, including alternative policy or management options. Exploratory scenarios (also known as "explorative scenarios" or "descriptive scenarios") are scenarios that examine a range of plausible futures, based on potential trajectories of drivers – either indirect (e.g., socio-political, economic and technological factors) or direct (e.g., habitat conversion, climate change). Target-seeking scenarios (also known as "goal-seeking scenarios" or "normative scenarios") are scenarios that start with the definition of a clear objective, or a set of objectives, specified either in terms of achievable targets, or as an objective function to be optimized, and then identify different pathways to achieving this outcome (e.g., through backcasting). Intervention scenarios are scenarios that evaluate alternative policy or management options – either through target seeking (also known as "goal seeking" or "normative scenario analysis") or through policy screening (also known as "ex-ante assessment"). Policy-evaluation scenarios are scenarios, including counterfactual scenarios, used in ex-post assessments of the gap between policy objectives and actual policy results, as part of the policy-review phase of the policy cycle. Policy-screening scenarios are scenarios used in ex-ante assessments, to forecast the effects of alternative policy or management options (interventions) on environmental outcomes.
Sector	A distinct part of society, or of a nation's economy.
Sedentarization	The process by which a nomadic group transitions to a lifestyle of living in one place.
Shared socioeconomic pathways (SSPs)	Narratives outlining broad characteristics of the global future and country-level population, Gross Domestic Product (GDP), urbanisation projections based on five alternative socio-economic developments (i.e., sustainable development), regional rivalry, inequality, fossil-fuelled development, and middle-of-the-road development. The SSPs are supported by key

	quantitative indicators and metrics, describing trends in demographics, human development, economy and lifestyle, policies and institutions, technology, environment and natural resources.
Silviculture	The applied science of forest ecology and management. The foundation is based on silvics, which is concerned with the development and growth of trees and forests. The practice of silviculture is rooted in a broad understanding of forested ecosystems, which includes biometeorology, hydrology, geology and soils and ecology.
Social capital	Networks together with shared norms, values and understandings that facilitate co-operation within or among groups. Social capital represents the capacity of a community (local or international like the UN) to gather and achieve common goals.
Social inequality	A state whereby resources in a given society are distributed unevenly, typically through norms of allocation, that engender specific patterns along lines of socially defined categories of persons.
Social marginalization	The process in which individuals or people are systematically blocked from (or denied full access to) various rights, opportunities and resources that are normally available to members of a different group, and which are fundamental to social integration and observance of human rights within that particular group (e.g., housing, employment, healthcare, civic engagement, democratic participation and due process).
Social-ecological resilience	The capacity of a social-ecological system to absorb or withstand perturbations and other stressors such that the system remains within the same regime, essentially maintaining its structure and functions. It describes the degree to which the system is capable of self-organization, learning and adaptation (Holling, 1973, Gunderson & Holling, 2002, Walker et al. 2004).
Socioecological system	An ecosystem, the management of this ecosystem by actors and organizations, and the rules, social norms, and conventions underlying this management.
Soil	The upper layer of the Earth's crust transformed by weathering and physical/chemical and biological processes. It is composed of mineral particles, organic matter, water, air and living organisms organized in genetic soil horizons.
Soil acidification	Soil acidification is caused by a number of factors including acidic precipitation and the deposition from the atmosphere of acidifying gases or particles, such as sulphur dioxide, ammonia and nitric acid. The most important causes of soil acidification on agricultural land, however, are the application of ammonium-based fertilizers and urea, elemental S fertilizer and the growth of legumes.
Soil biodiversity loss	Decline in the diversity of (micro- and macro-) organisms present in a soil. In turn, this prejudices the ability of soil to provide critical ecosystem services.
Soil compaction	An increase in density and a decline of porosity in a soil that impedes root penetration and movements of water and gases.
Soil contamination	An increase of toxic compounds (heavy metals, pesticides and so on) in a soil that constitute, directly or indirectly (via the food chain), a hazard for human health and/or for the provision of ecosystem services assured by the soil.
Soil degradation	The diminishing capacity of the soil to provide ecosystem goods and services.

Soil ecosystem functions	A description of the significance of soils to humans and the environment. Examples are: (i) control of substance and energy cycles within ecosystems; (ii) basis for the life of plants, animals and man; (iii) basis for the stability of buildings and roads; (iv) basis for agriculture and forestry; (v) carrier of genetic reservoir; (vi) document of natural history; and (vii) archaeological and paleo-ecological document.
Soil fertility	The quality of a soil that enables it to provide compounds in adequate amounts and proper balance to promote growth of plants when other factors (such as light, moisture, temperature and soil structure) are favourable.
Soil formation rates	The process of rock weathering though which soil is formed.
Soil health	The continued capacity of the soil to function as a vital living system, within ecosystem and land-use boundaries, to sustain biological productivity, promote the quality of air and water environments, and maintain plant, animal and human health (Doran, Stamatiadis and Haberern, 2002).
Soil organic carbon (SOC)	A summarizing parameter including all of the carbon forms for dissolved (DOC: Dissolved Organic Carbon) and total organic compounds (TOC: Total Organic Carbon) in soils.
Soil organic matter (SOM)	Matter consisting of plant and/or animal organic materials, and the conversion products of those materials in soils.
Soil pollution	Process of soil contamination by chemicals (fertilizers, petroleum products, pesticides, herbicides, mining) which has affected agricultural productivity and other ecosystem services negatively.
Soil processes	Physical or reactive geochemical and biological processes which may attenuate, concentrate, immobilize, liberate, degrade or otherwise transform substances in soil (ISO, 2013).
Soil quality	All current positive or negative properties with regard to soil utilization and soil functions.
Soil salinization	Increase in water-soluble salts in soil which is responsible for increasing the osmotic pressure of the soil. In turn, this negatively affects plant growth because less water is made available to plants.
Soil structure	The arrangement of soil particles in a variety of recognized shapes and sizes.
Soil sealing	The covering of the soil surface with materials like concrete and stone, as a result of new buildings, roads, parking places, but also other public and private space.
Sovereignty principle	Sovereignty in the sense of contemporary public international law denotes the basic international legal status of a state that is not subject, within its territorial jurisdiction, to the governmental, executive, legislative, or judicial jurisdiction of a foreign state or to foreign law other than public international law. A sovereign entity can decide and administer its own laws, can determine the use of its land and can do pretty much as it pleases, free of external influence (within the limitations of international law).
Soil stability	The integrity of soil aggregates, degree of soil structural development, and erosion resistance.
Species	An interbreeding group of organisms that is reproductively isolated from all other organisms, although there are many partial exceptions to this rule in particular taxa. Operationally, the term species is a generally agreed

	fundamental taxonomic unit, based on morphological or genetic similarity, that once described and accepted is associated with a unique scientific name.
Species composition	The array of species in a specific region, area, or assembly.
Species richness	The number of species within a given sample, community, or area.
Species/ecological community	An assemblage or association of populations of two or more different species occupying the same geographical area and in a particular time.
Stakeholder(s)	Any individuals, groups or organizations who affect, or could be affected (whether positively or negatively) by a particular issue and its associated policies, decisions and action.
Strategic environmental assessment (SEA)	A mechanism that attempts to assess systematically the environmental impacts of decisions made at, what is conventionally called, levels of strategic decisions.
Summary for policymakers	Is a component of any report, providing a policy-relevant but not policy prescriptive summary of that report.
Surface mining	Includes strip mining, open-pit mining and mountaintop removal mining, is a broad category of mining in which soil and rock overlying the mineral deposit (the overburden) are removed.
Sustainable Development Goals (SDGs)	Also, the “Global Goals,” are a universal call to action to end poverty, protect the planet and ensure that all people enjoy peace and prosperity. These 17 Goals build on the successes of the Millennium Development Goals, while including new areas such as climate change, economic inequality, innovation, sustainable consumption, peace and justice, among other priorities. The goals are interconnected; often the key to success on one will involve tackling issues more commonly associated with another.
Sustainable forest management (SFM)	Can mean many things to many people – yet a common thread is the production of forest goods and services for the present and future generations. The concept provides guidance on how to manage forests to provide for today’s needs (as best as possible) and not compromise (i.e., reduce) the options of future generations.
Sustainable intensification	A process or system where agricultural yields are increased without adverse environmental impact and without the conversion of additional non-agricultural land.
Sustainable intensive agriculture	Process or system where agricultural yields are increased without adverse environmental impact and without the conversion of additional non-agricultural land.
Sustainable land management	The use of land resources, including soils, water, animals and plants for the production of goods to meet changing human needs while ensuring the long-term productive potential of these resources and the maintenance of their environmental functions
Sustainable land use	The land use that serves the needs (for food, energy, housing, recreation etc.) of all human beings living on Earth today and in the future, respecting the boundaries and the resilience of ecological systems.
Sustainable soil management	Sets of activities that maintain or enhance the supporting, provisioning, regulating and cultural services provided by soils without significantly impairing either the soil functions that enable those services or biodiversity.

Sustainable use (of biodiversity and its components)	The use of components of biological diversity in a way and at a rate that does not lead to the long-term decline of biological diversity, thereby maintaining its potential to meet the needs and aspirations of present and future generations.
Sustainability	A characteristic or state whereby the needs of the present and local population can be met without compromising the ability of future generations or populations in other locations to meet their needs.
Swidden/slash and burn agriculture	Swidden farming, also known as shifting cultivation or milpa in Latin America, is conventionally defined as an agricultural system in which temporary clearings are cropped for fewer years than they are allowed to remain fallow.
Target	A choice by people of a desired contemporary or future outcome.
Target condition	A condition that maximizes the desired mix of ecosystem services.
Telecoupling	Socioeconomic and environmental interactions over distances. It involves distant exchanges of information, energy and matter (e.g., people, goods, products, capital) at multiple spatial, temporal and organizational scales.
Tenure security	An agreement between an individual or group to land and residential property, which is governed and regulated by a legal and administrative framework includes both customary and statutory systems.
Terrestrial productivity	Net Primary Production (NPP) from the terrestrial environment.
Thermodynamics	The science of the relationship between heat, work, temperature, and energy. In broad terms, thermodynamics deals with the transfer of energy from one place to another and from one form to another. The key concept is that heat is a form of energy corresponding to a definite amount of mechanical work. The behaviour of a complex thermodynamic system, such as Earth's atmosphere, can be understood by first applying the principles of states and properties to its component parts—in this case, water, water vapour, and the various gases making up the atmosphere. By isolating samples of material whose states and properties can be controlled and manipulated, properties and their interrelations can be studied as the system changes from state to state.
Threatened species	In the IUCN Red List terminology, a threatened species is any species listed in the Red List categories critically endangered, endangered, or vulnerable.
Tillage	In agriculture, the preparation of soil for planting and the cultivation of soil after planting.
Tipping point	A set of conditions of an ecological or social system where further perturbation will cause rapid change and prevent the system from returning to its former state.
Topsoil	The upper part of a natural soil that is generally dark coloured and has a higher content of organic matter and nutrients when compared to the (mineral) horizons below. It excludes the litter layer.
Trade-off	A situation where an improvement in the status of one aspect of the environment or of human well-being is necessarily associated with a decline in or loss of a different aspect. Trade-offs characterize most complex systems, and are important to consider when making decisions that aim to improve environmental and/or socio-economic outcomes. Trade-offs are distinct from synergies (the latter are also referred to as "win-win"

	scenarios): synergies arise when the enhancement of one desirable outcome leads to enhancement of another.
Traditional knowledge	See “Indigenous and local knowledge”.
Tragedy of the Commons	Title of an influential 1968 essay by biologist Garrett Hardin, which argued that overuse of common resources is a leading cause of environmental degradation. This was interpreted by some, especially economists and free-market libertarians, to mean that private ownership is preferable to the commons for the stewardship of land, water, minerals, etc. Yet in recent years many have challenged this view on both empirical and philosophical grounds. Professor Elinor Ostrom of Indiana University has been a leading figure in demonstrating the practical utility and sustainability of commons governance regimes, particularly in developing countries. This suggests that the vision of human behaviour implicit in the tragedy of the commons metaphor is not as immutable as many economists assert, and that collective management is an eminently practical governance strategy in many circumstances. The tragedy of the “anti-commons” is now frequently invoked to describe the problems associated with excessive privatization and fragmentation of property rights, such that collective action for the common good is thwarted. See also "Commons" and "Common pool resources".
Transboundary pollution	Pollution that originates in one country but, by crossing the border through pathways of water or air, can cause damage to the environment in another country.
Transhumance	Form of pastoralism or nomadism organized around the migration of livestock between mountain pastures in warm seasons and lower altitudes the rest of the year.
Tree-covered area	A land cover class that includes any geographic area dominated by natural tree plants with a cover of 10 percent or more. Areas planted with trees for afforestation purposes and forest plantations are included in this class.
Trends	A general development or change in a situation or in the way that people are behaving.
Trophic level	The level in the food chain in which one group of organisms serves as a source of nutrition for another group of organisms.
Uncertainty	Any situation in which the current state of knowledge is such that: (i) the order or nature of things is unknown; (ii) the consequences, extent, or magnitude of circumstances, conditions, or events is unpredictable; and (iii) credible probabilities to possible outcomes cannot be assigned. Uncertainty can result from lack of information or from disagreement about what is known or even knowable. Uncertainty can be represented by quantitative measures (e.g., a range of values calculated by various models) or by qualitative statements (e.g., reflecting the judgment of a team of experts).
Upscaling	The process of scaling information from local, fine-grained resolution to global, coarse-grained resolution.
Urban heat island effect	The term "heat island" describes built up areas that are hotter than nearby rural areas.
Values	• Values systems: Set of values according to which people, societies and organizations regulate their behaviour. Value systems can be identified in

	both individuals and social groups. • Value (as principles): A value can be a principle or core belief underpinning rules and moral judgments. Values as principles vary from one culture to another and also between individuals and groups. • Value (as preference): A value can be the preference someone has for something or for a particular state of the world. Preference involves the act of making comparisons, either explicitly or implicitly. Preference refers to the importance attributed to one entity relative to another one. • Value (as importance): A value can be the importance of something for itself or for others, now or in the future, close by or at a distance. This importance can be considered in three broad classes. 1. The importance that something has subjectively, and may be based on experience. 2. The importance that something has in meeting objective needs. 3. The intrinsic value of something. • Value (as measure): A value can be a measure. In the biophysical sciences, any quantified measure can be seen as a value. • Non-anthropocentric value: A non-anthropocentric value is a value centred on something other than human beings. These values can be non-instrumental or instrumental to non-human ends. • Intrinsic value: The value inherent to nature, independent of human experience and evaluation, and therefore beyond the scope of anthropocentric valuation approaches. • Anthropocentric value: Human-centred, the value that something has for human beings and human purposes. • Instrumental value: The direct and indirect contribution of nature's benefits to the achievement of a good quality of life. Within the specific framework of the total economic value, instrumental values can be classified into use (direct and indirect use values) on the one hand, and non-use values (option, bequest and existence values) on the other. Sometimes option values are considered as use values as well. • Non-instrumental value: The value attributed to something as an end in itself, regardless of its utility for other ends. • Relational value: The values that contribute to desirable relationships, such as those among people and between people and nature, as in "living in harmony with nature". • Integrated valuation: The process of collecting, synthesizing, and communicating knowledge about the ways in which people ascribe importance and meaning of nature's contribution, to facilitate deliberation and agreement for decision making and planning.
Vector-borne pathogens	Disease causing agents that are spread from host to host by living or non-living agent. For example, malaria is transmitted to humans by mosquitos.
Virtual water	The volume of freshwater used to produce the commodity, good or service, measured at the place where the product was actually produced.
Virtual water balance	In global trade, the difference between water used to produce export products and the water used to produce import products.
Volatilization	The process of converting a chemical substance from a liquid or solid state to a gaseous or vapour state.
Vulnerability reduction	The propensity or predisposition to be adversely affected. Vulnerability encompasses a variety of concepts and elements including sensitivity or susceptibility to harm and lack of capacity to cope and adapt.

Water footprint	The measure of humanity's use of fresh water as represented in volumes of water consumed and/or polluted.
Water logging	An excess of water on top and/or within the soil, leading to reduced air availability in the soil for long periods.
Water purification	Vegetation, and specially aquatic plants, can assist in removing sediments and nutrients and other impurities from water.
Water security	The capacity of a population to safeguard sustainable access to adequate quantities of and acceptable quality water for sustaining livelihoods, human well-being, and socio-economic development, for ensuring protection against water-borne pollution, water-related disasters, and for preserving ecosystems.
Water Security Index (WSI)	The ratio of total water withdrawal to the water availability including environmental flow requirements. Higher WSI values lead to decreasing water security.
Water table	The upper surface of the zone of ground water.
Well-being	A perspective on a good life that comprises access to basic resources, freedom and choice, health and physical well-being, good social relationships, security, peace of mind and spiritual experience. Human well-being is a state of being with others and the environment. Well-being is achieved when individuals and communities can act meaningfully to pursue their goals and everyone can enjoy a good quality of life. The concept of human well-being is used in many Western societies and its variants, together with living in harmony with nature, and living well in balance and harmony with Mother Earth.
Western cultures/western science	(Also called modern science, Western scientific knowledge or international science) is used in the context of the IPBES conceptual framework as a broad term to refer to knowledge typically generated in universities, research institutions and private firms following paradigms and methods typically associated with the “scientific method” consolidated in Post-Renaissance Europe on the basis of wider and more ancient roots. It is typically transmitted through scientific journals and scholarly books. Some of its central tenets are observer independence, replicable findings, systematic scepticism, and transparent research methodologies with standard units and categories.
Wetlands	Areas that are subject to inundation or soil saturation at a frequency and duration, such that the plant communities present are dominated by species adapted to growing in saturated soil conditions, and/or that the soils of the area are chemically and physically modified due to saturation and indicate a lack of oxygen; such areas are frequently termed peatlands, marshes, swamps, sloughs, fens, bogs, wet meadows and so on.
Woody encroachment	Increasing dominance of shrubs in grasslands and trees in shrublands.
Worldviews	Defined by the connections between networks of concepts and systems of knowledge, values, norms and beliefs. Individual person’s worldviews are moulded by the community to which the person belongs. Practices are embedded in worldviews and are intrinsically part of them (e.g. through rituals, institutional regimes, social organization, but also in environmental policies, in development choices, etc.). See also “Perceptions”; “Concepts”; “Reality”.

Acronyms

ADB	Asian Development Bank
ASSOD	Assessment of Soil Degradation in South and Southeast Asia
BAU	Business-as-usual
BBOP	Business and Biodiversity Offsets Programme
BECCS	Bioenergy with Carbon Capture and Storage
BRC	UK Biological Records Centre
C	Carbon
CBA	Cost–Benefit Analysis
CBD	Convention on Biological Diversity
CGIAR	Consultative Group for International Agricultural Research
CO2	Carbon Dioxide
CO2e	Carbon Dioxide Equivalent
EC	European Commission
EEA	European Environment Agency
ELD	Economics of Land Degradation Initiative
EPA	Environmental Protection Agency (USA)
ESVD	Ecosystem Service Valuation Database
EU	European Union
EVRI	Environmental Valuation Reference Inventory
FAO	Food and Agriculture Organization
FAOSTAT	Food and Agriculture Organization Corporate Statistical Database
FSC	Forest Stewardship Council
GBO4	4th Global Biodiversity Outlook
GDP	Gross Domestic Product
Gg	Gigagram (1g x 109)
GHG	Greenhouse Gas
GIS	Geographical Information System
GIZ	Deutsche Gesellschaft für Internationale Zusammenarbeit
GLADA	Global Assessment of Land Degradation and Improvement
GLADIS	Global Land Degradation Information System
GLASOD	Global Assessment of Human-induced Soil Degradation
GLOBIO	Global Biodiversity model
GMO	Genetically Modified Organism
Ha	Hectare (10,000 square meters, 0.01 km-1)
IAASTD	International Assessment of Agricultural Knowledge, Science and Technology for Development
IAM	Integrated Assessment Model
ICMM	International Council on Mining and Metals
IEA	International Energy Agency
IIASA	International Institute for Applied Systems Analysis
IIFB	International Indigenous Forum on Biodiversity
IIPFCC	International Indigenous Peoples Forum on Climate Change
ILK	Indigenous and Local Knowledge

ILKP	Indigenous and Local Knowledge and Practices
IMAGE	Integrated Model to Assess the Environment
InVEST	Integrated Valuation of Ecosystem Services and Tradeoffs
IPBES	Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services
IPCC	Intergovernmental Panel on Climate Change
IPES-Food	International Panel of Experts on Sustainable Food Systems
IPLC	Indigenous Peoples and Local Communities
ITPS	Intergovernmental Technical Panel on Soils
ITTO	International Tropical Timber Organization
IUCN	International Union for Conservation of Nature
LADA	Land Degradation Assessment in Drylands
LCA	Life Cycle Analysis
MCDA	Multi-Criteria Decision Analysis
MEA or MA	Millennium Ecosystem Assessment
MIMES	Multi-scale Integrated Models of Ecosystem Services
MODIS	Moderate-resolution Imaging Spectroradiometer
MSA	Mean Species Abundance
NCP	Nature's Contributions to People
NDVI	Normalized Difference Vegetation Index
NGOs	Non-Governmental Organizations
NPP	Net Primary Productivity
OECD	Organization for Economic Co-operation and Development
ONS	Office for National Statistics (UK)
PA	Protected Area
PBL	Planbureau voor de Leefomgeving/Netherlands Environmental Assessment Agency
PEFC	Programme for the Endorsement of Forest Certification
PES	Payments for Ecosystem/Environmental Services
Pg	Petagram (1g x 1015 or 1 Gigatonne)
PPP	Public-Private Partnerships
RCP	Representative Concentration Pathway
RECARE	Preventing and Remediating degradation of soils in Europe through Land Care
REDD+	Reducing Emissions from Deforestation and forest Degradation
RUSLE	Revised Universal Soil Loss Equation
SDGs	Sustainable Development Goals
SOC	Soil Organic Carbon
SolVES	Social Values for Ecosystem Services
SPLASH	Simple process-led algorithms for simulating habitats
SSP	Shared Socioeconomic Pathway
Tg	Teragram (g x 1012)
TNC	The Nature Conservancy
UKNEA	UK National Ecosystem Assessment
UN	United Nations
UNCCD	United Nations Convention to Combat Desertification
UNCTAD	United Nations Conference on Trade and Development
UNDP	United Nations Development Programme
UNECE	United Nations Economic Commission for Europe
UNEP	United Nations Environment Programme
UNEP-FI	United Nations Environment Programme - Finance Initiative

UNFCCC	United Nations Framework Convention on Climate Change
UNFCCC	United Nations Framework Convention on Climate Change
UNICEF	United Nations Children's Fund
UNISDR	United Nations International Strategy for Disaster Risk Reduction
UNPFII	United Nations Permanent Forum on Indigenous Issues
UN-WATER	United Nations Water
USA	United States of America
USLE	Universal Soil Loss Equation (also RUSLE: Revised, and MUSCLE: Modified)
WHO	World Health Organization
WOCAT	World Overview of Conservation Approaches and Technologies
WRI	World Resources Institute
WTO	World Trade Organization
Yr	Year