Chapter 4

Status and trends of land degradation and restoration and associated changes in biodiversity and ecosystem functions

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This chapter should be cited as:

Prince, S., Von Maltitz, G., Zhang, F., Byrne, K., Driscoll, C., Eshel, G., Kust, G., Martínez-Garza, C., Metzger, J. P., Midgley, G., Moreno-Mateos, D., Sghaier, M., and Thwin, S. Chapter 4: Status and trends of land degradation and restoration and associated changes in biodiversity and ecosystem fundtions. In IPBES (2018): The IPBES assessment report on land degradation and restoration. Montanarella, L., Scholes, R., and Brainich, A. (eds.). Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem services, Bonn, Germany, pp. xx-xx.

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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.

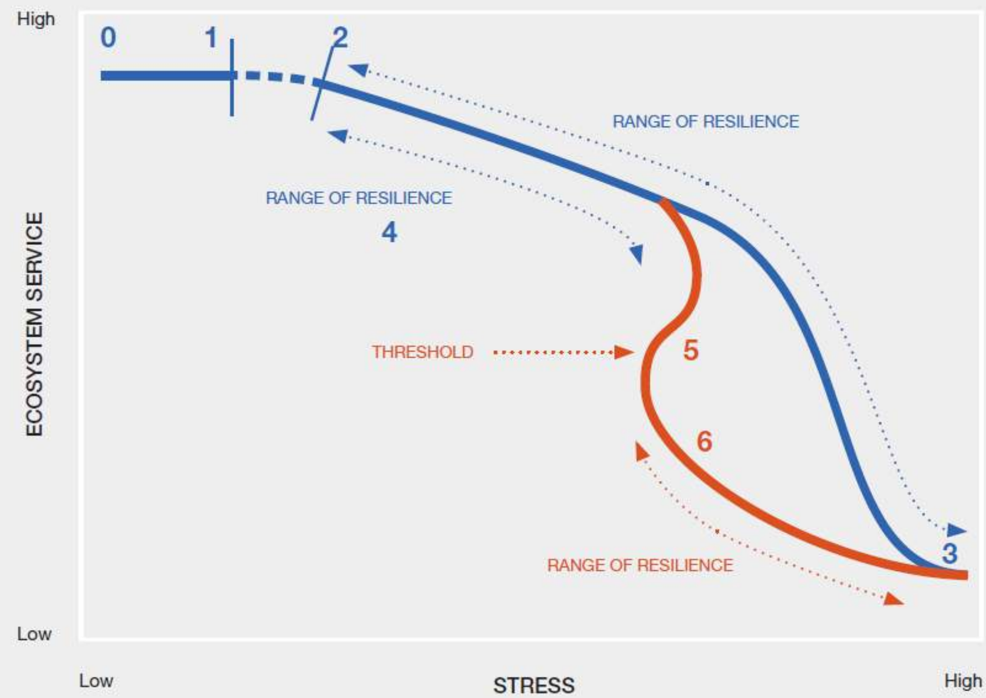
4.1.2. The degradation process

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).

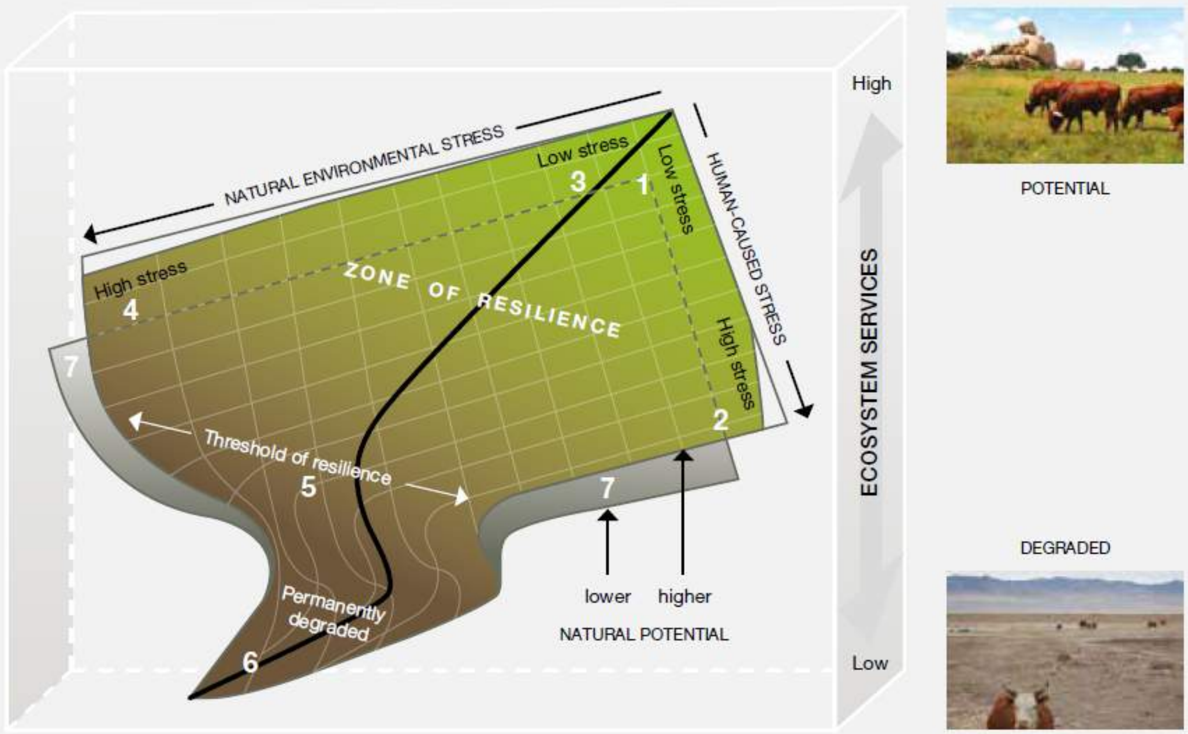


The analogy of response curves is helpful only when one anthropogenic stress is involved, but normally there are many that affect ecosystem services, such as soil type, pollution, soil compaction, loss of palatable species for livestock, and reduced productivity – all in one location. These stresses 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.

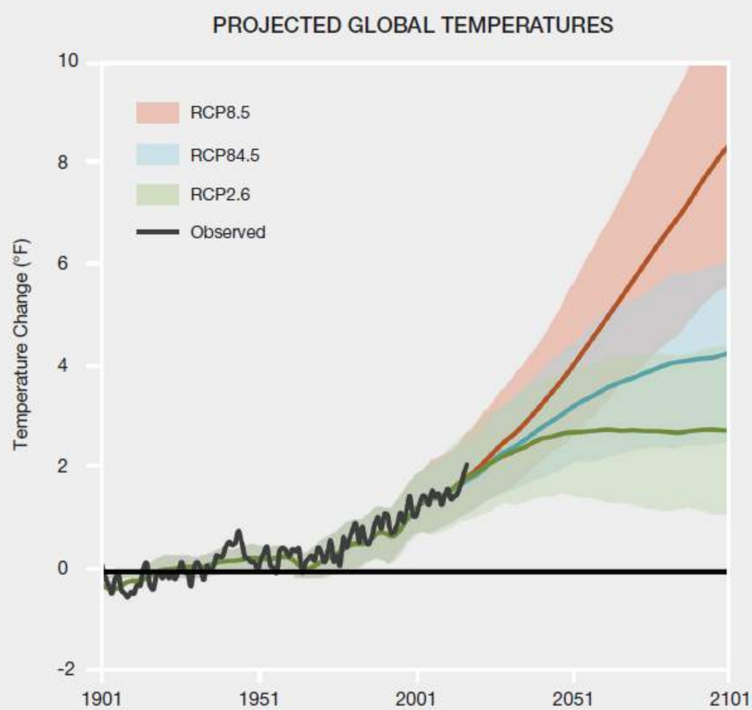
Scenarios of human activities and their effects on climate (Bjørnæs, 2015) use Integrated Assessment Models (IAMs) that estimate the combined effects of human activities (e.g., land use and fossil fuel emissions) on the carbon-climate system. IAMs such as the IMAGE model (Integrated Model to Assess the 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 3[rd] 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		

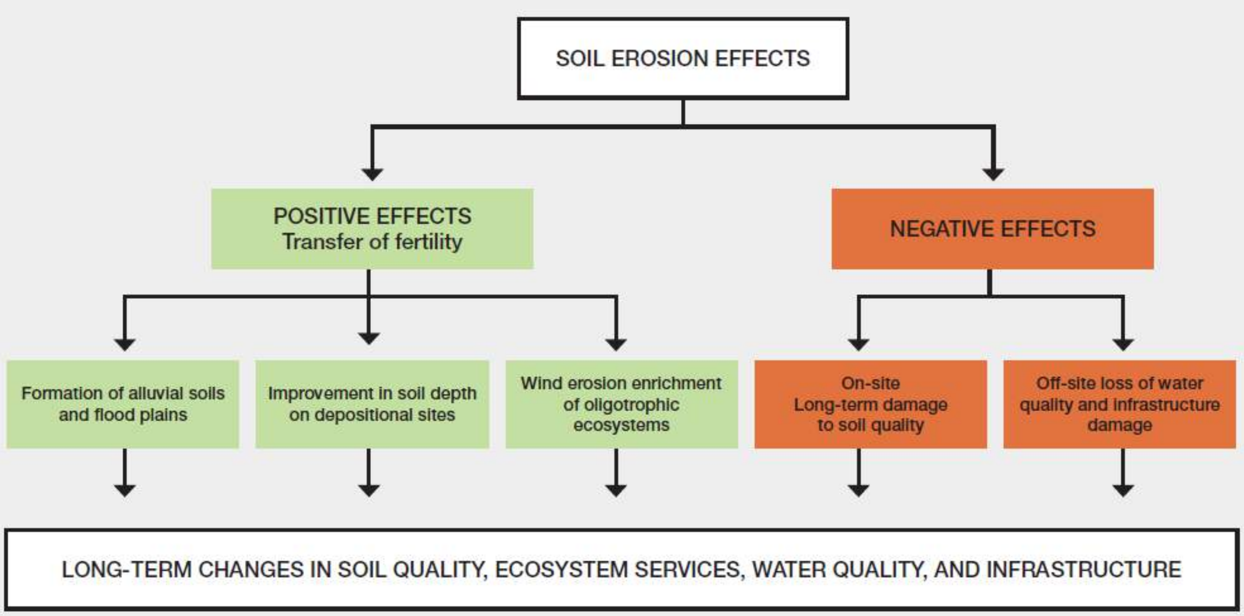
Europe	158	403	94	65	104	60	d		
North America	140	796	429 *a*	469	96	79	d		
South America	139	851	306 *b*	398	156	69	56 *b*		
World (Total)	1,216	6,140	3,592	2,740 *c*	991	470	76 *d*	57,560	15

4.2 Individual degradation processes

4.2.1 Soil erosion

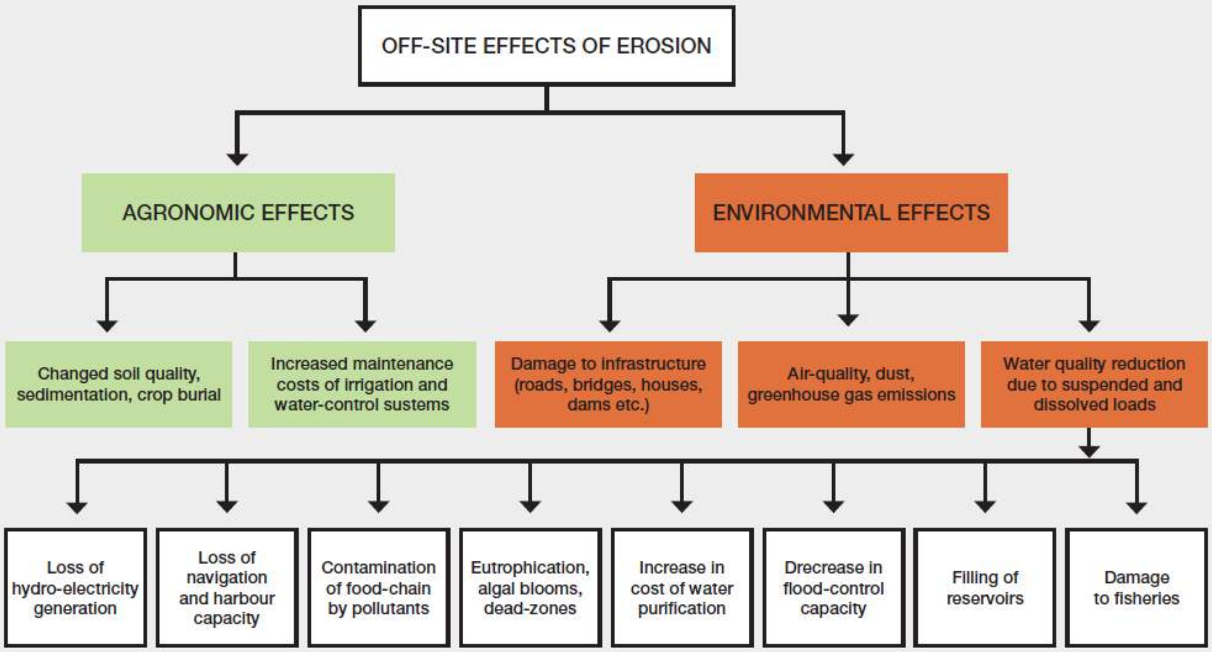
Soil is the basis for provision of many essential ecosystem (Costanza & Daly, 1987; Hassan *et al.*, 2005b) yet the soil resources of the world are finite and non-renewable in the human-time scale (Lal, 1998) and so extensive loss through erosion is a serious problem (Montgomery, 2007b). Nevertheless, 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

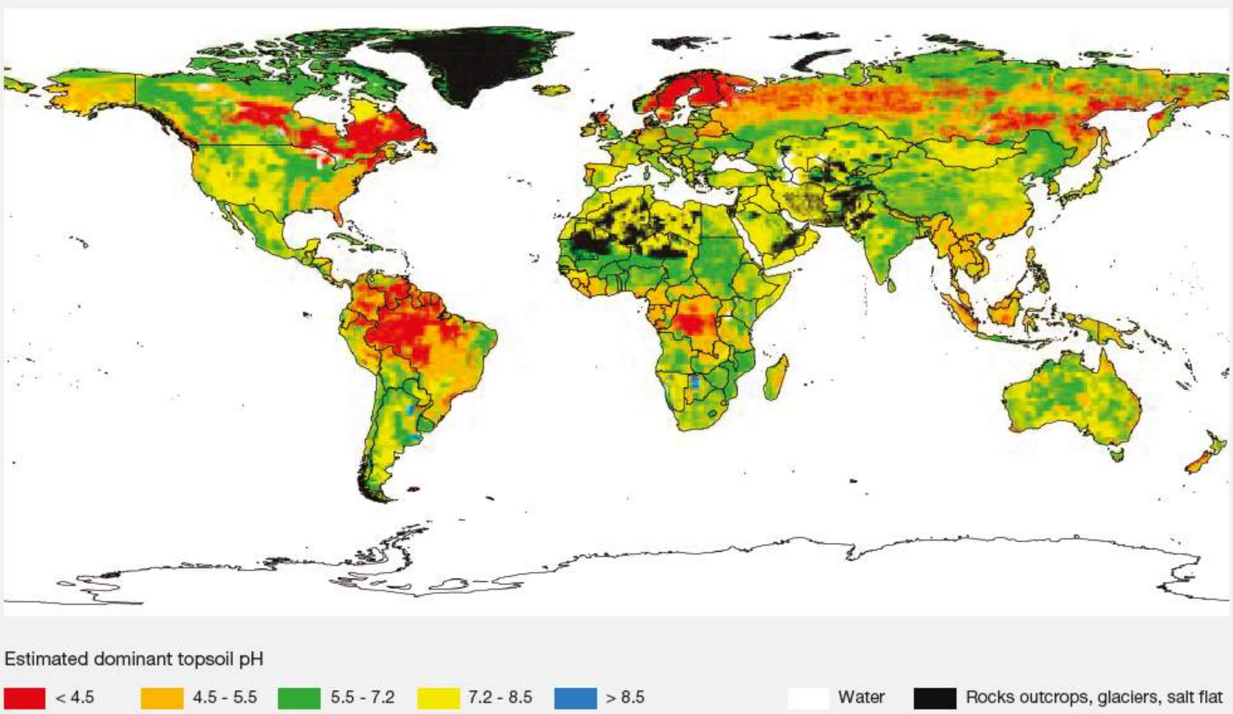
Human activities can exacerbate acidification that occurs with natural soil development. The common causes are: wet and dry deposition of acidic atmospheric pollutants ("acid rain") emitted from fossil fuel combustion; excessive application of ammonium-based fertilizers and intensive agricultural cropping; 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.

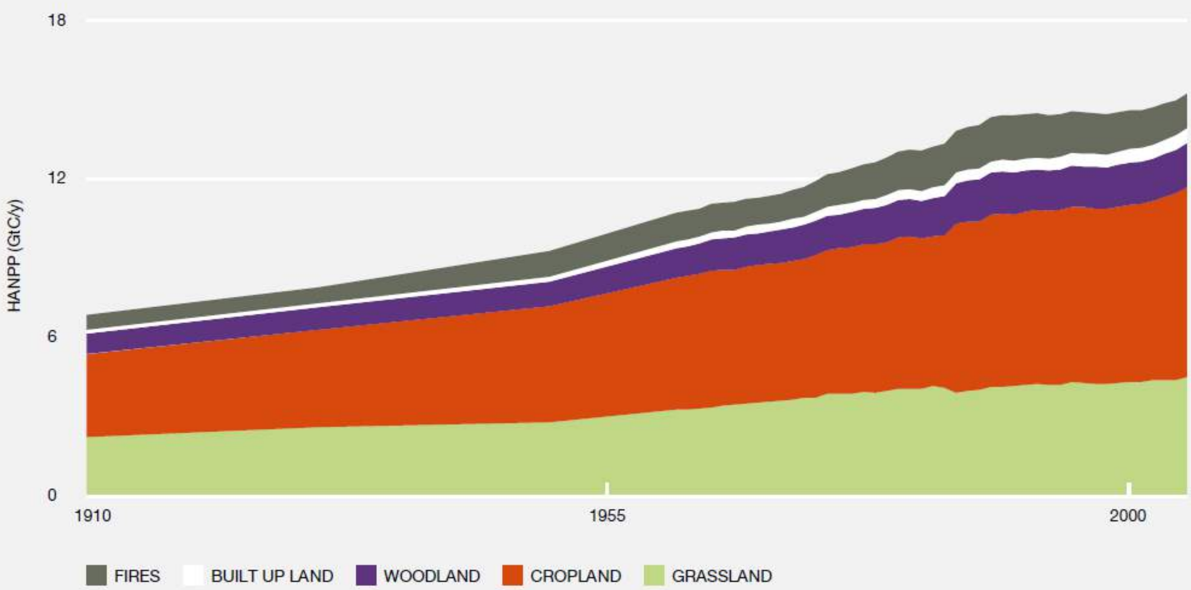
Overall, total net primary production of the terrestrial biosphere has increased by 0.02-0.04% yr^{-1} (an increase of 20 to 40 TgC yr^{-1} relative to a total global terrestrial NPP of around 100 PgC yr^{-1}) over the past several decades (many lines of evidence, including for example, Donohue *et al.*, 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).

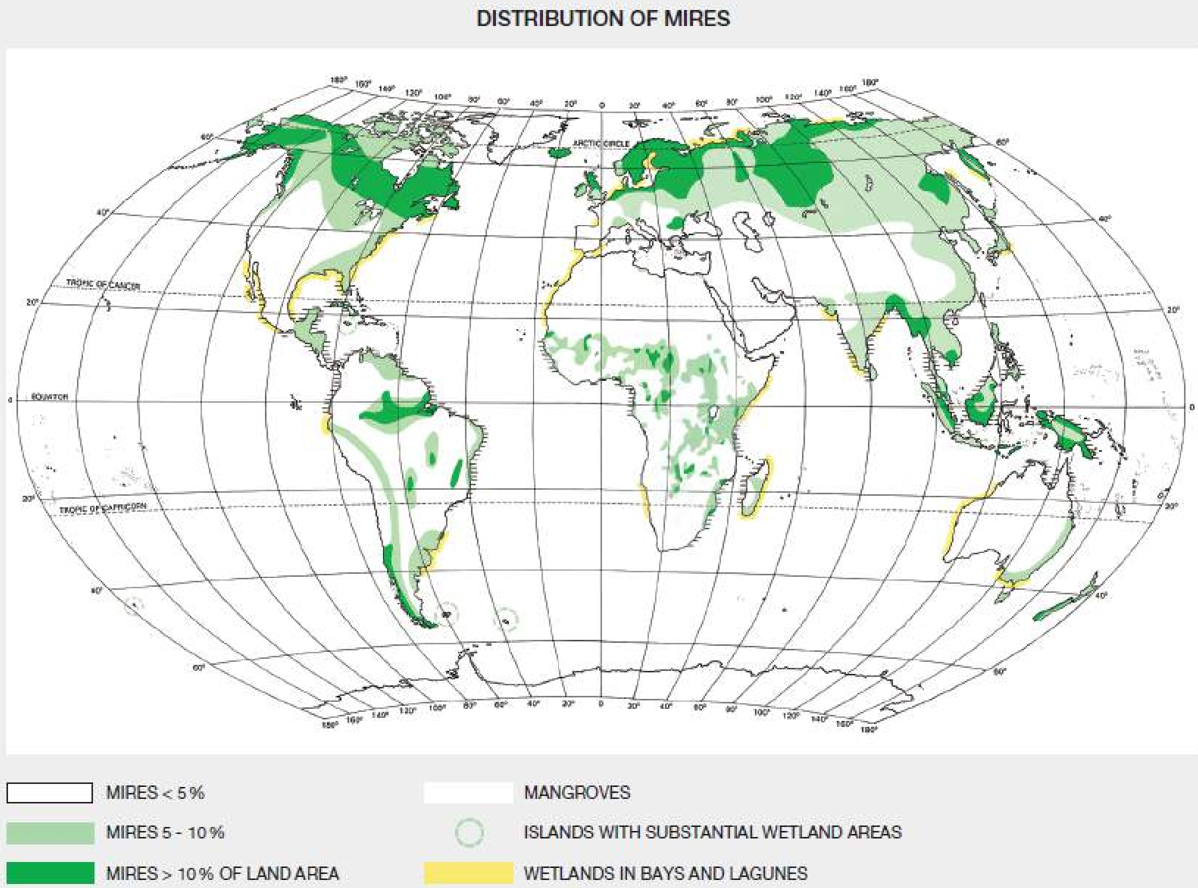
The broad control of biomass stocks is determined by changes in net primary production minus disturbances such as harvest and fire. The current growth of the land biomass stock in untransformed areas (e.g., Running *et al.*, 2004) can only result from increased net primary production, decreased in respiration 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.

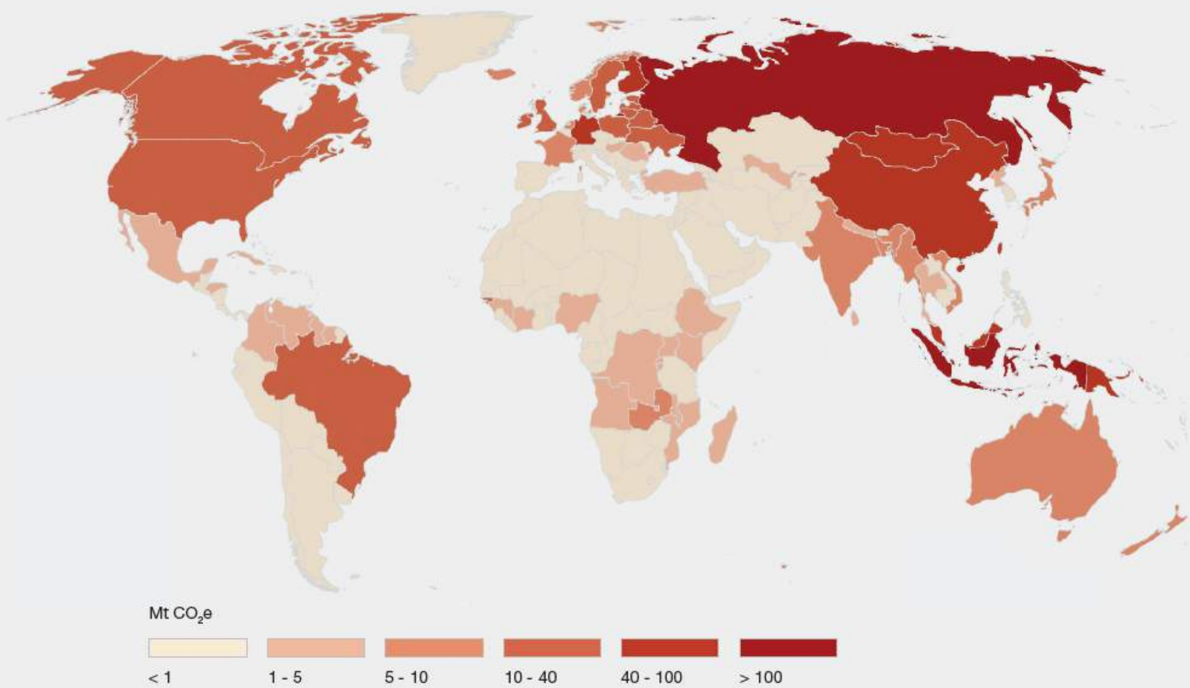


Peatlands are the most extensive form of terrestrial and coastal wetland (Section 4.2.5.2). Davidson, (2014) shows that wetland losses of 87% are typical of some regions, although Joosten, (2009) indicates that only 11.6% of the world's peatlands are currently considered to be degraded, this estimate 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

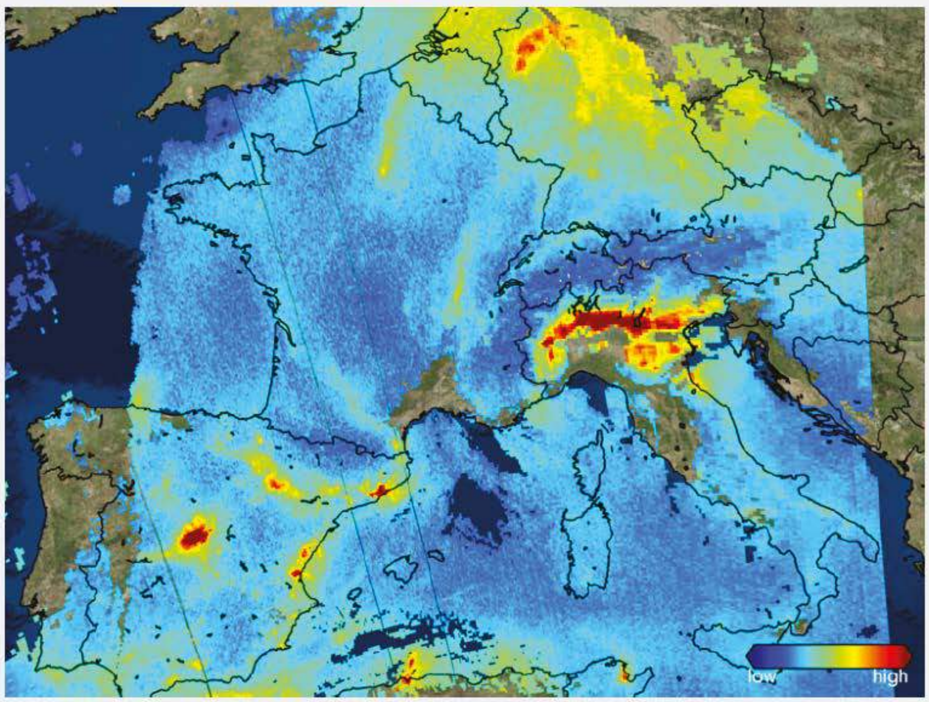
4.2.4.1 Atmospheric pollution

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).

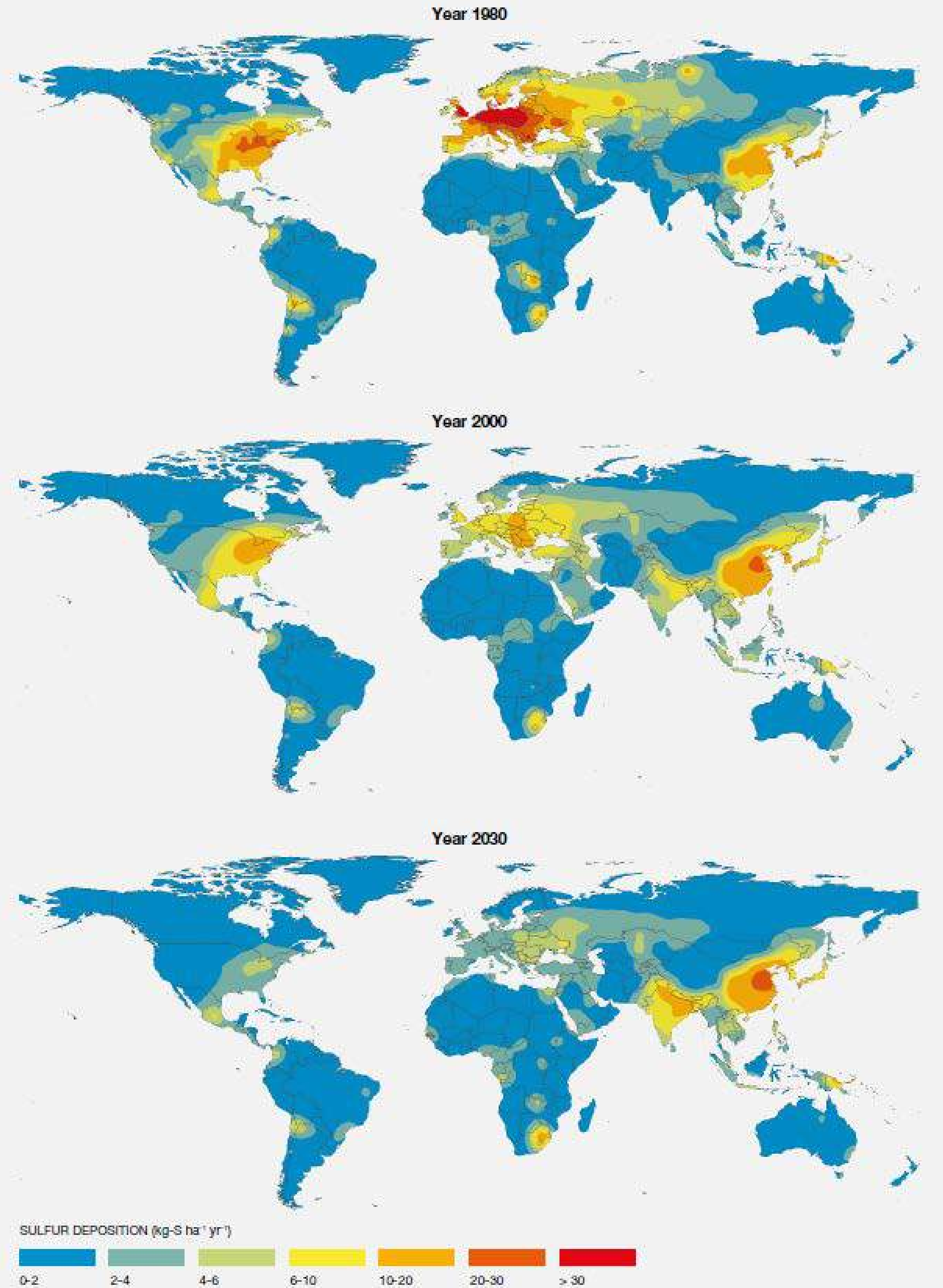


Figure 4 13 **Total nitrate deposition in kg N ha^{-1} yr^{-1} for 1980, 2000 and 2030.**

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).

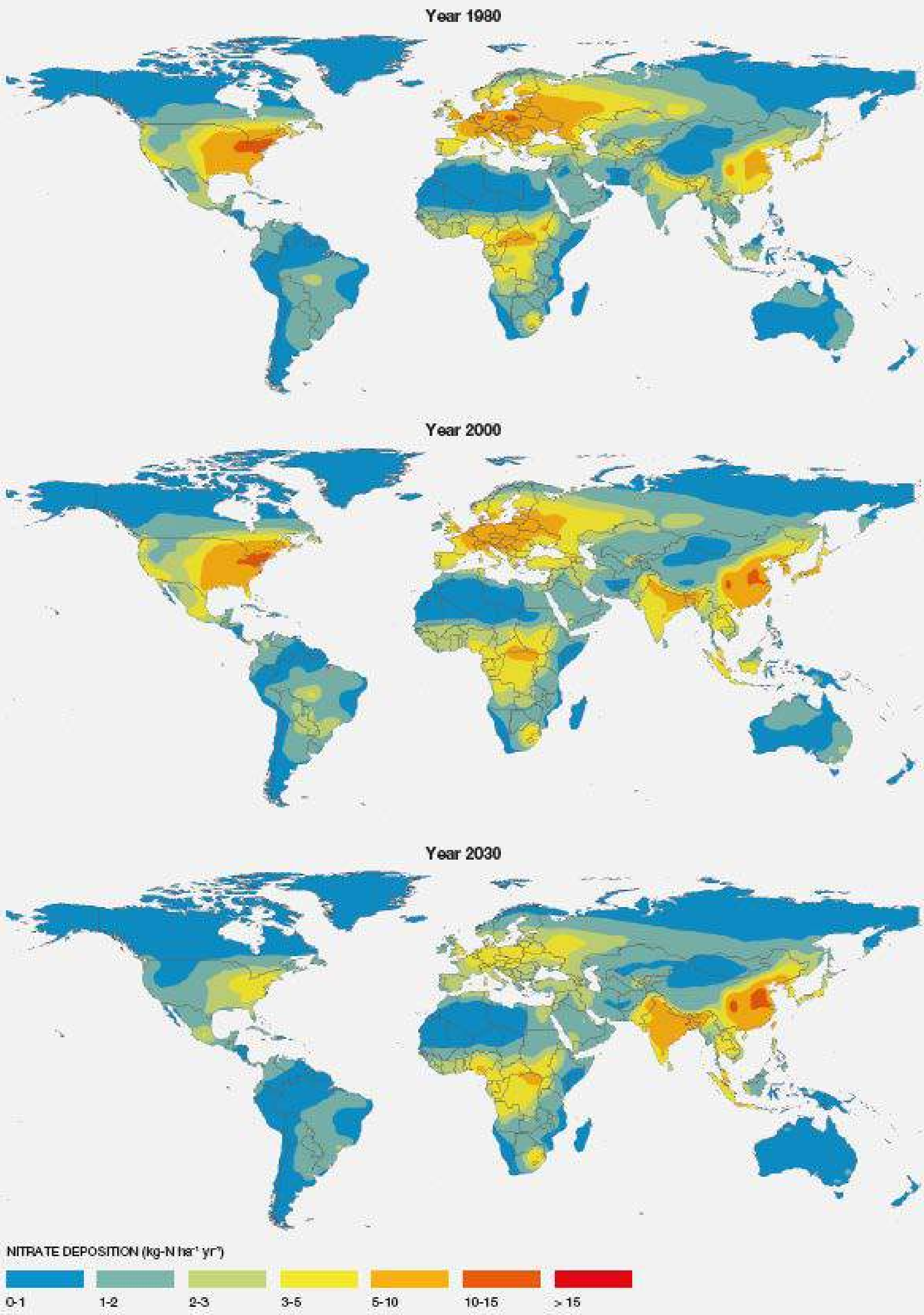
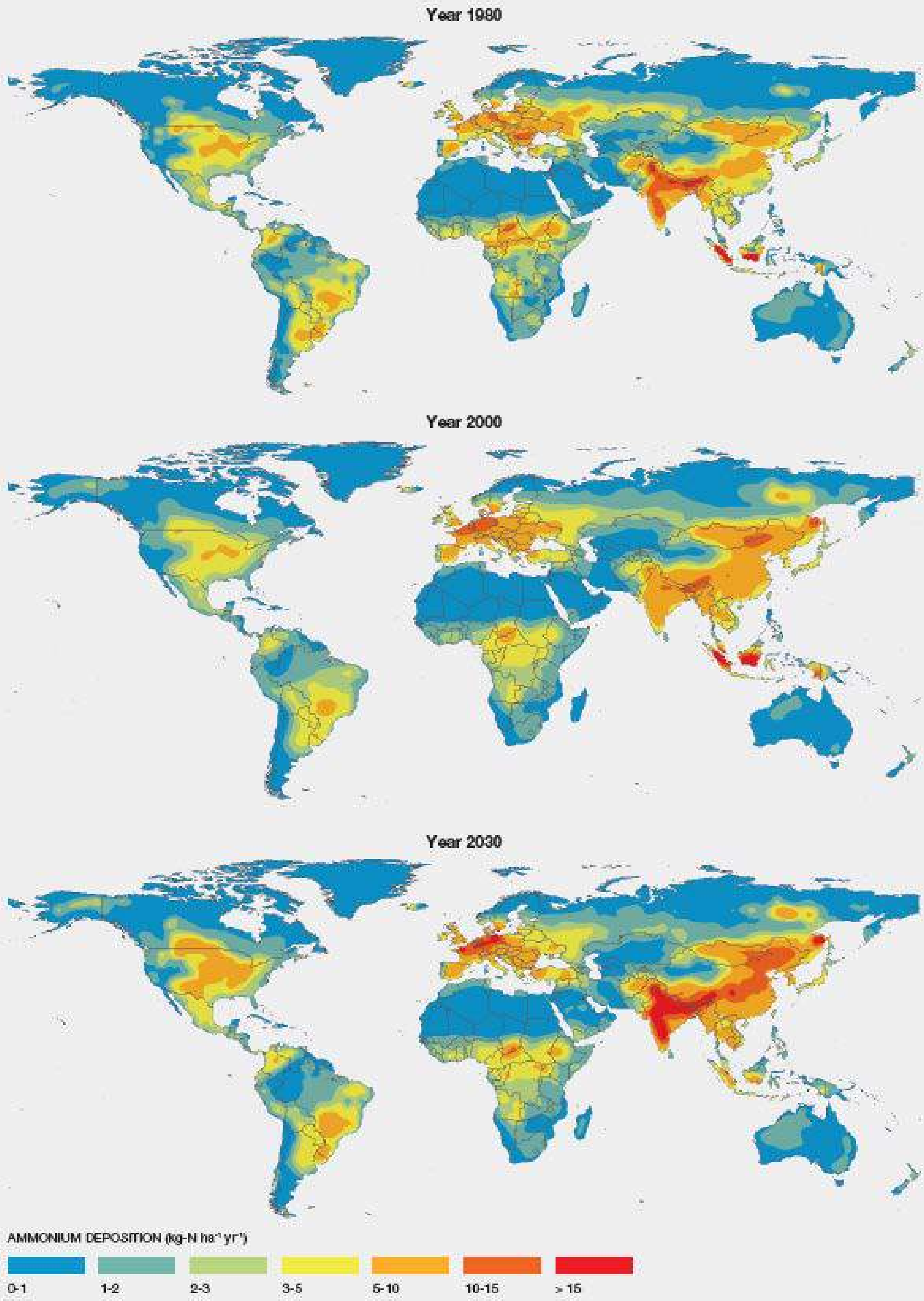


Figure 4 14 **Total ammonium deposition in kg N ha-1 yr-1 for 1980, 2000 and 2030.**

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).

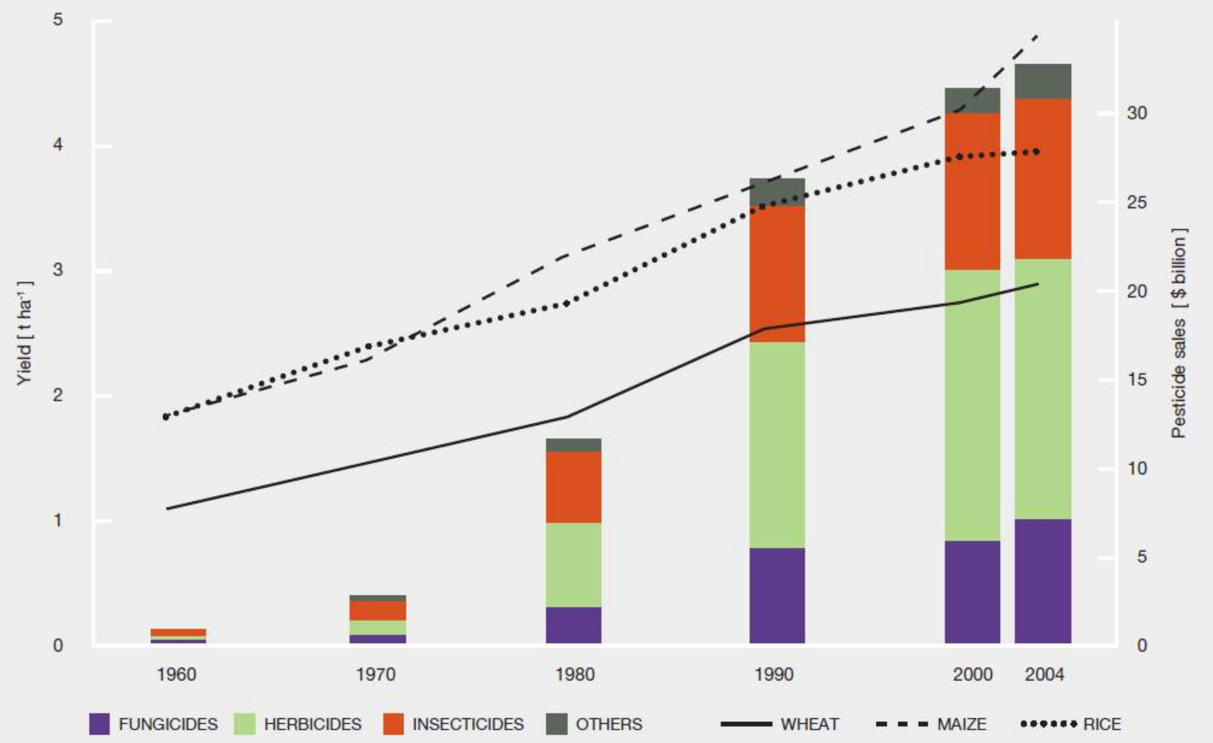
Persistent organic pollutants

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

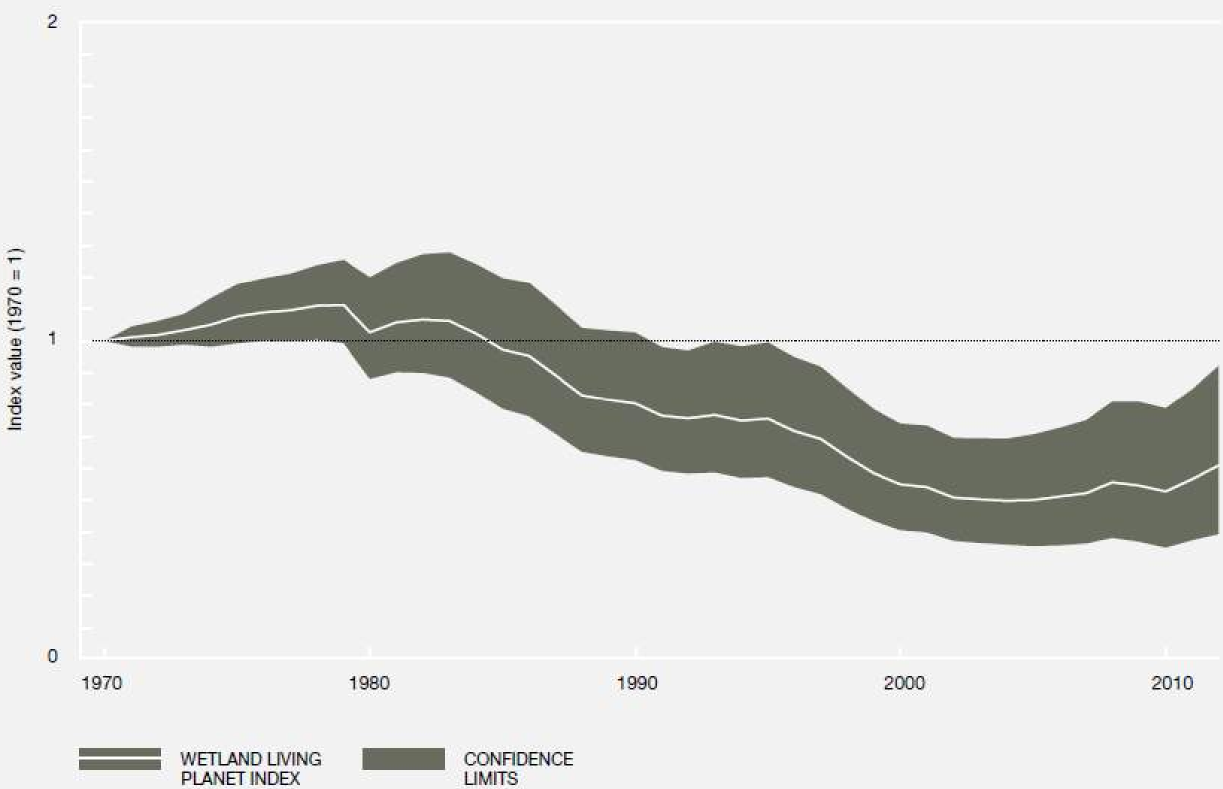
Status and trends in degradation

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.

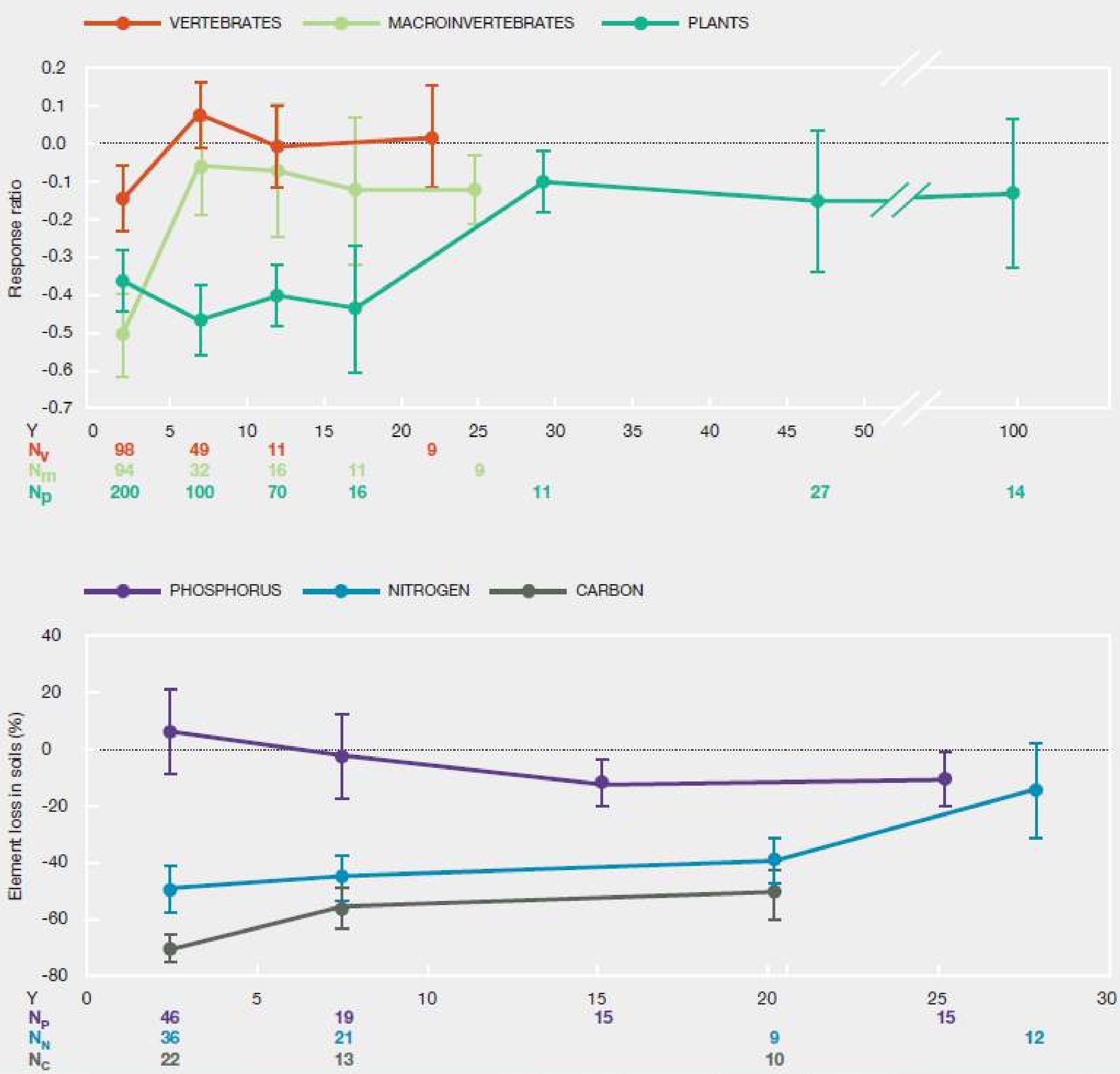
Impact on biodiversity, ecosystem process and function

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

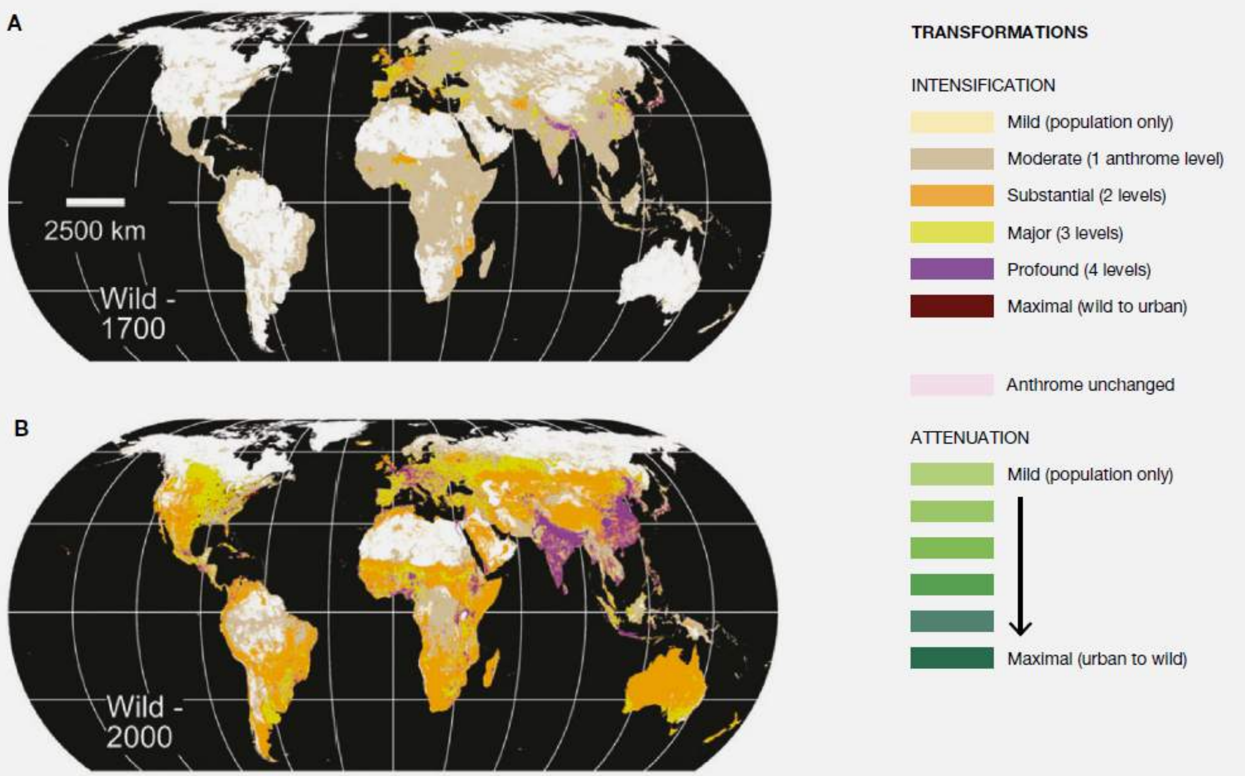
Land-cover changes are pervasive and, when aggregated globally, they may significantly affect basic processes of the global system's functioning (Lambin & Geist, 2006). They encompass the many types of deforestation, conversion of forests, grasslands and drained wetlands to cultivation as well as changes 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).



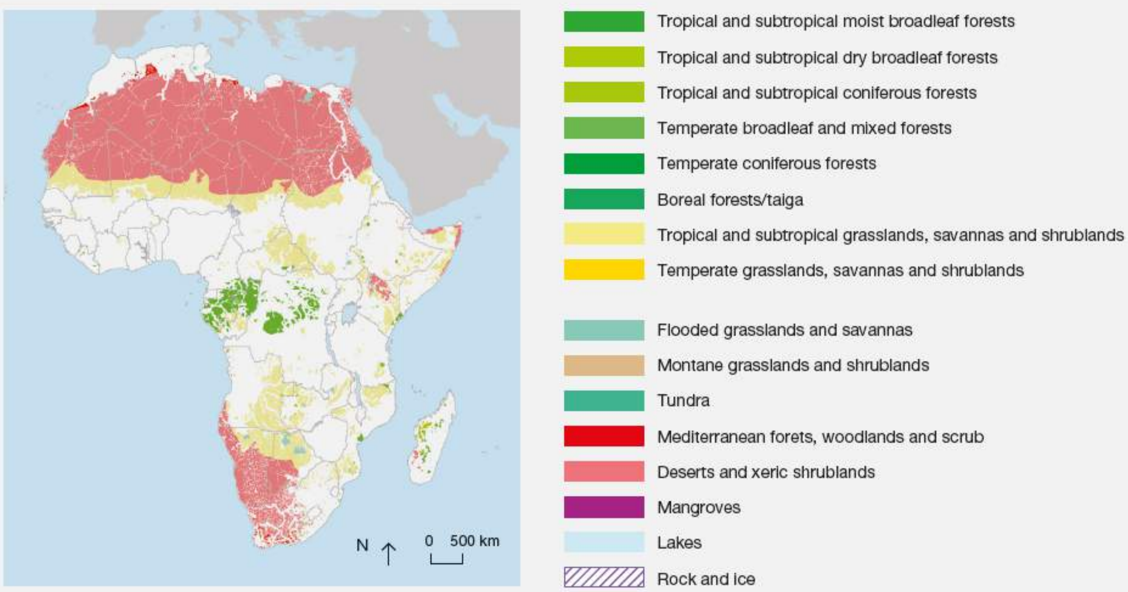
Over the past 300 years, more than 50% of the land surface has been substantively altered by land-use activities, over 25% of forests have been permanently cleared, over 30% of the land surface is occupied by agriculture, and 10–44 10^6 km^2 of land is globally recovering from previous human land-use 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).

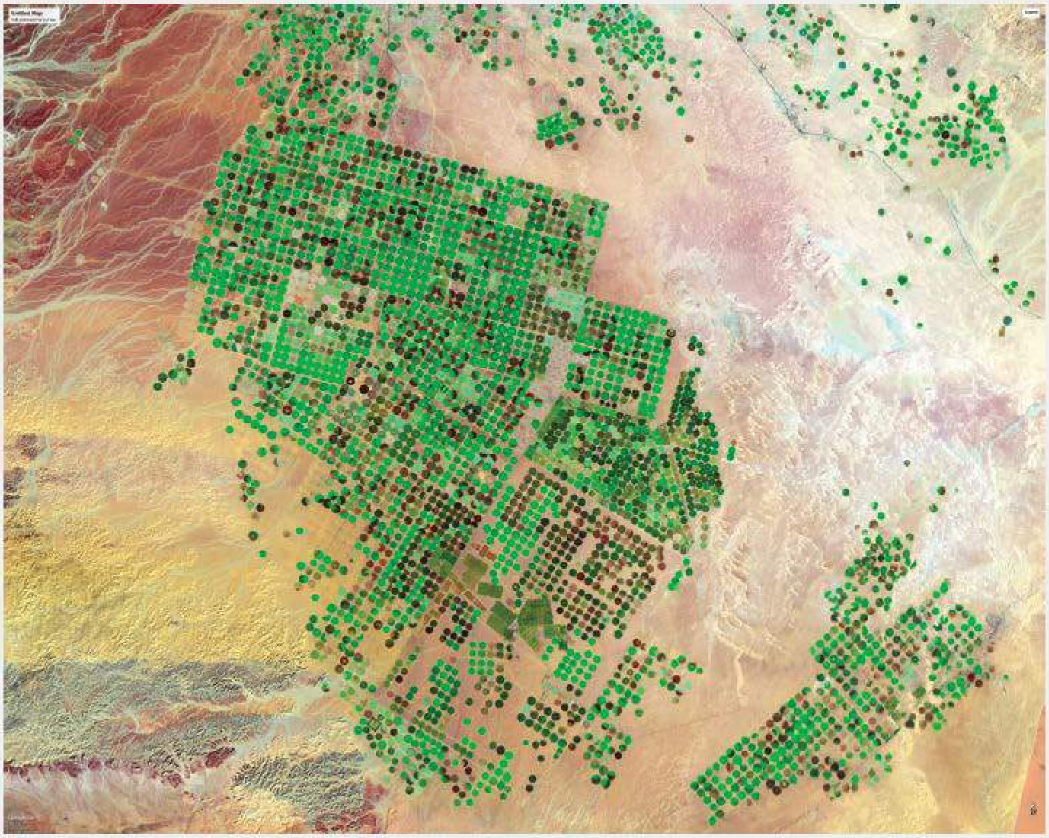
Climate

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).

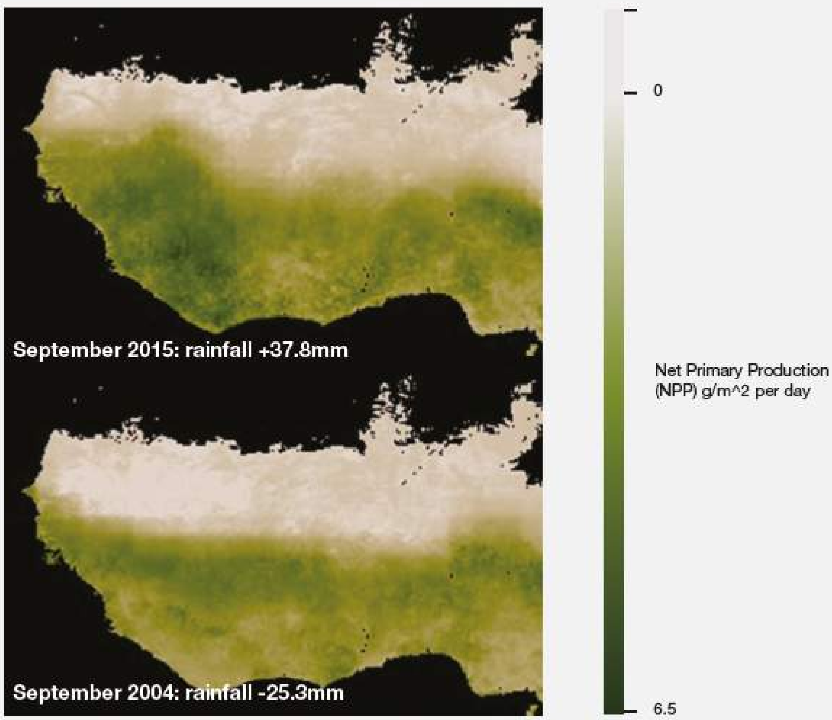


Figure 4 22 Temporal profiles of A field observations of herbaceous mass and B GIMMS-3g normalized difference vegetation index (NDVI) over the Gourma region of Mali.

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).

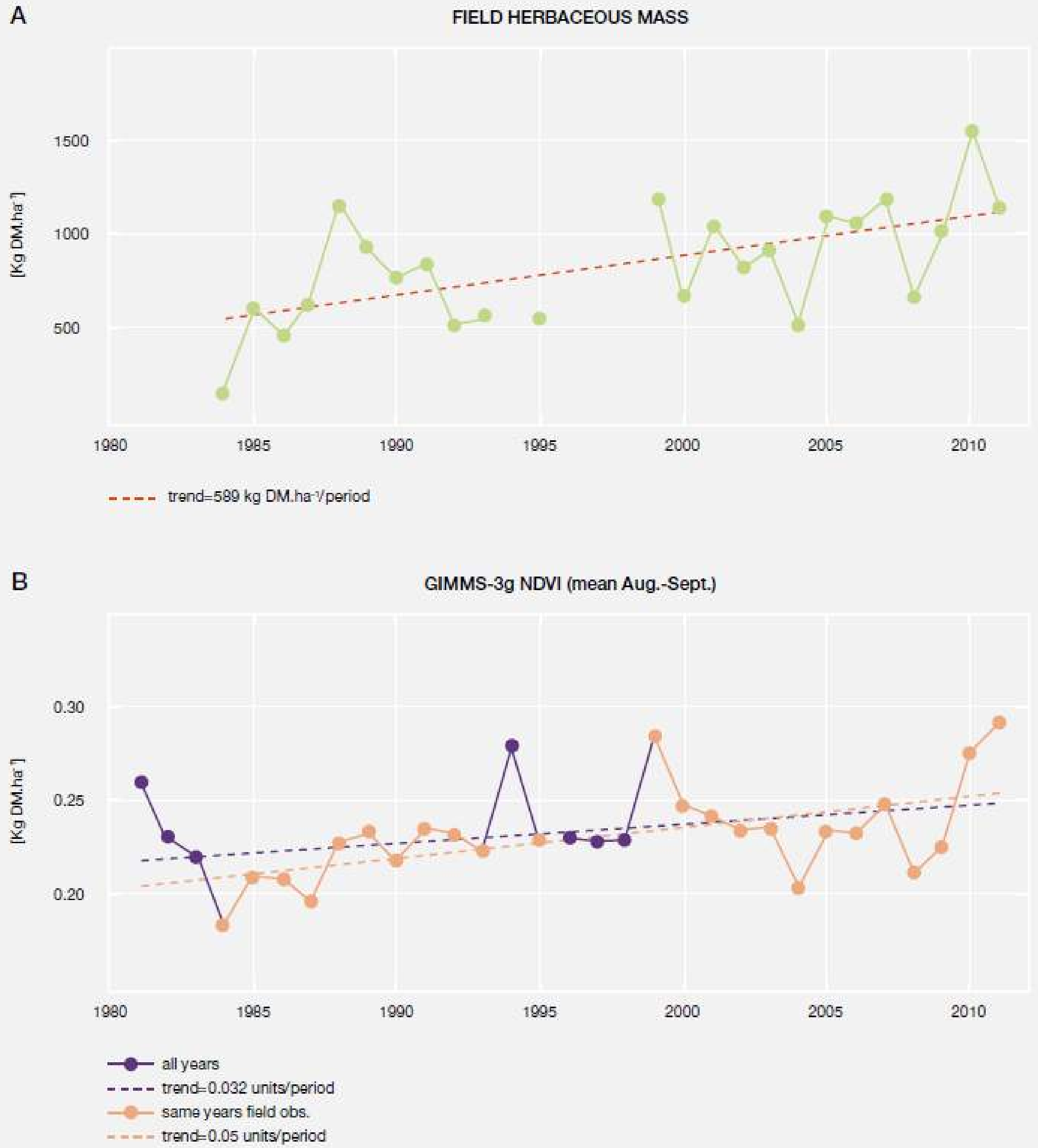


Figure 4 23 Sahel precipitation June–October from 1900 to 2011 shown as anomalies (deviations) from the mean of all dates.

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).

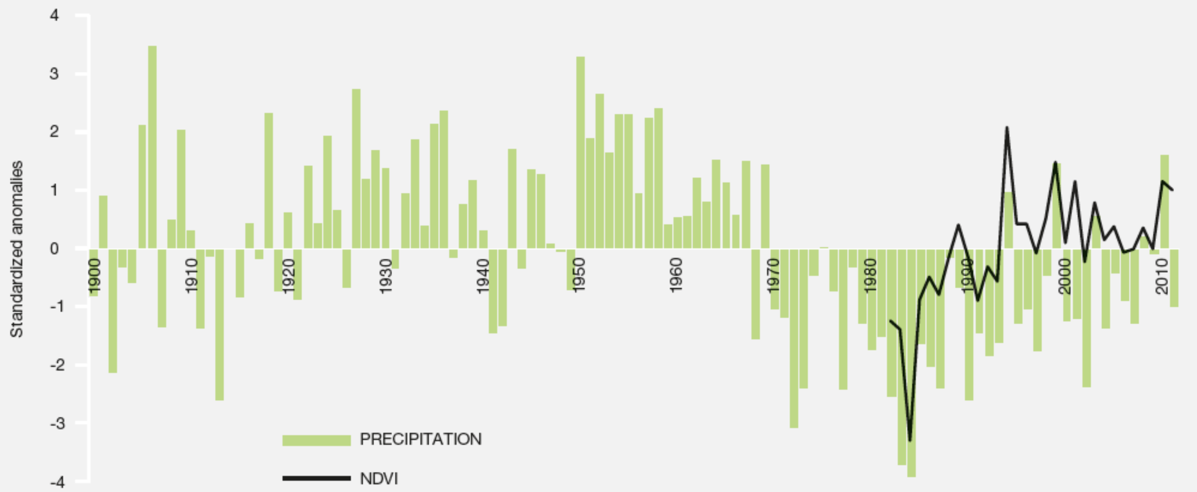
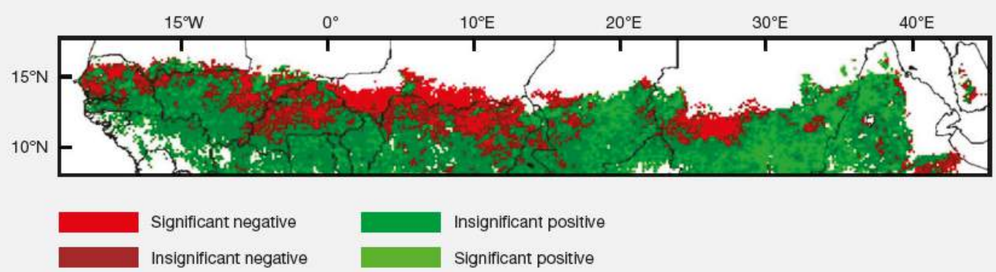


Figure 4 24 Potential areas where anthropogenic degradation may be in progress in the Sahel (shown in red).

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).

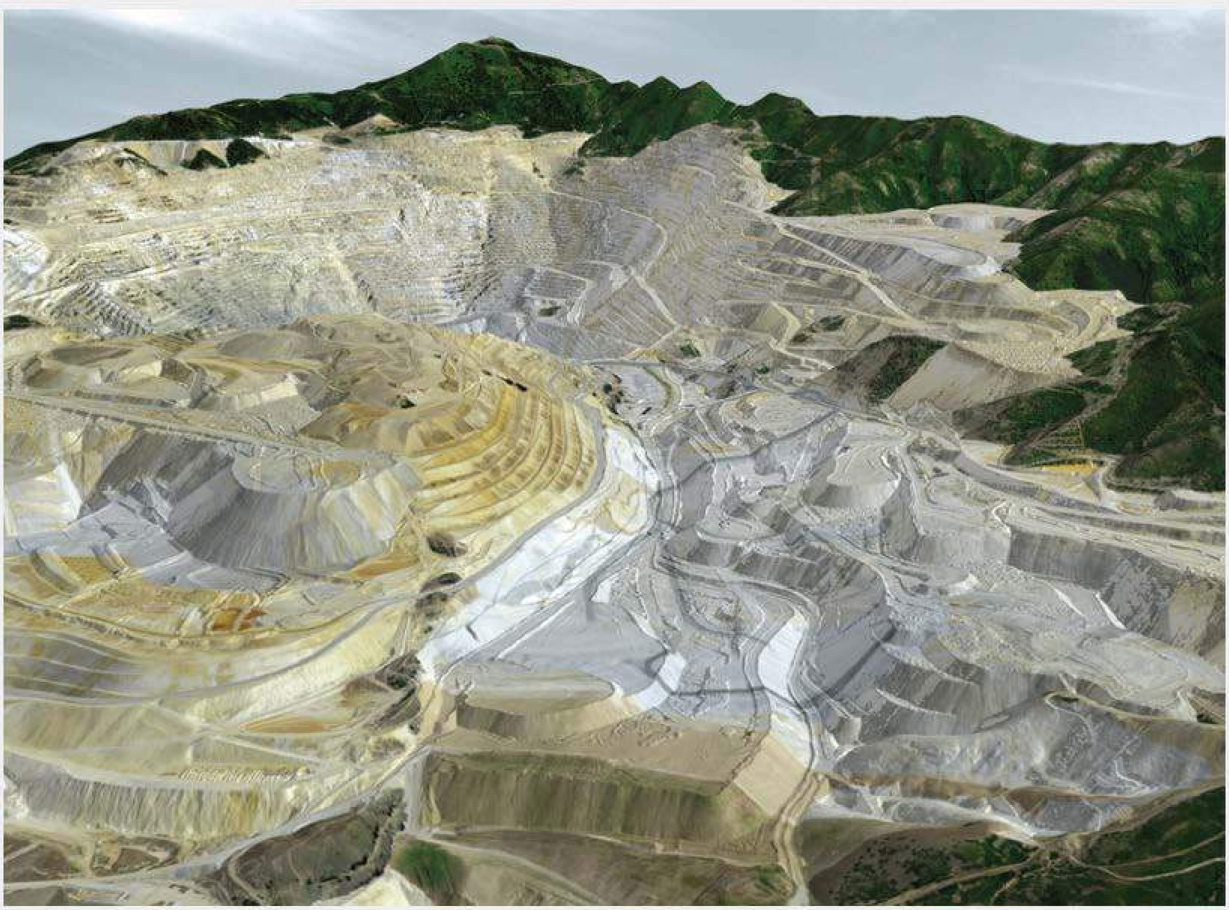
As discussed in Section 4.2.8, climate change is anticipated to have major, but uncertain, impacts on fire regimes.

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.

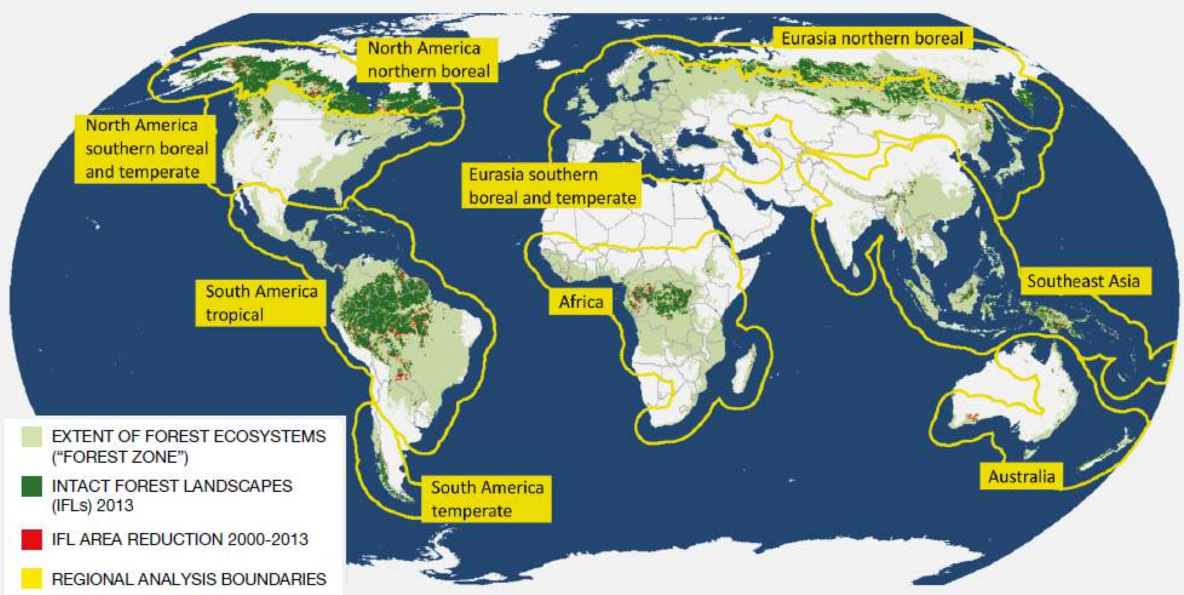
Composition

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

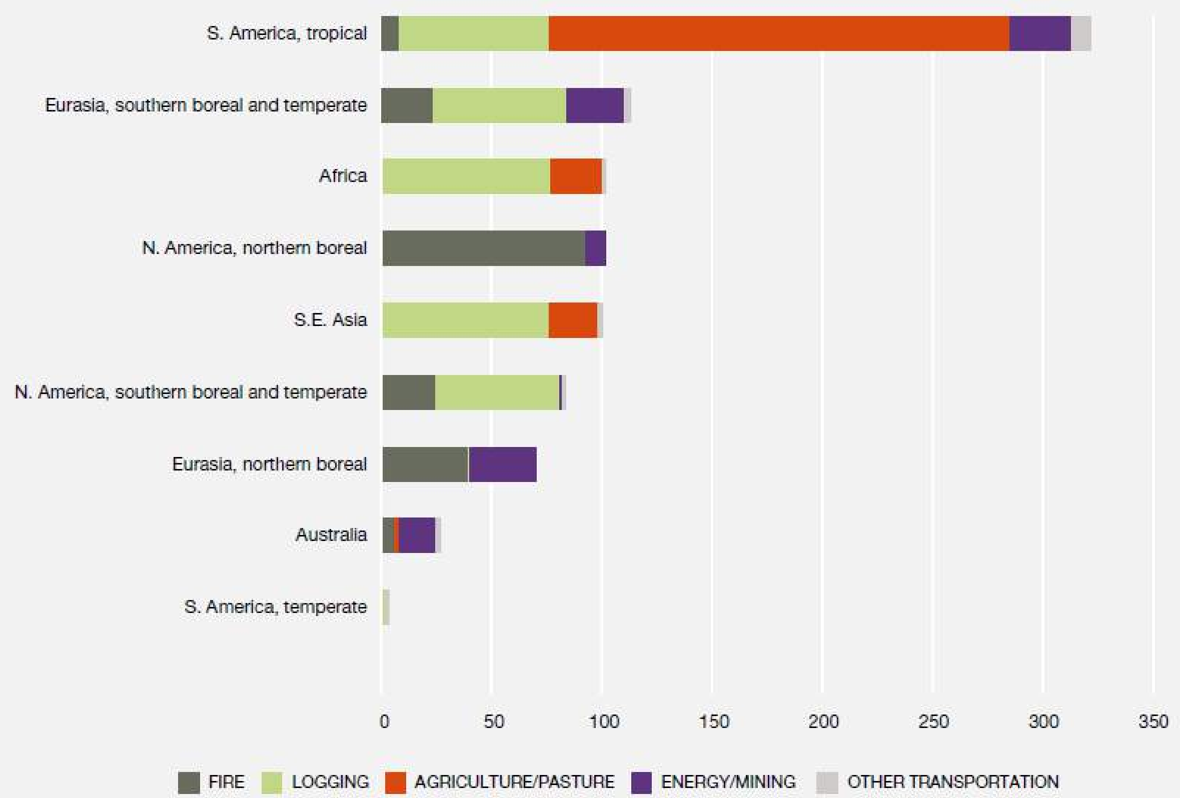
An individual land-cover type in a landscape can form one patch or many fragmented patches and the patches themselves can have complex shapes and, therefore, boundaries between them and the other land-cover types. The degree of fragmentation has important effects on biodiversity and some aspects of the 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 ($km^2 \times 10^3$) and causes of change. Source: Potapov *et al.* (2017). http://www.intactforests.org



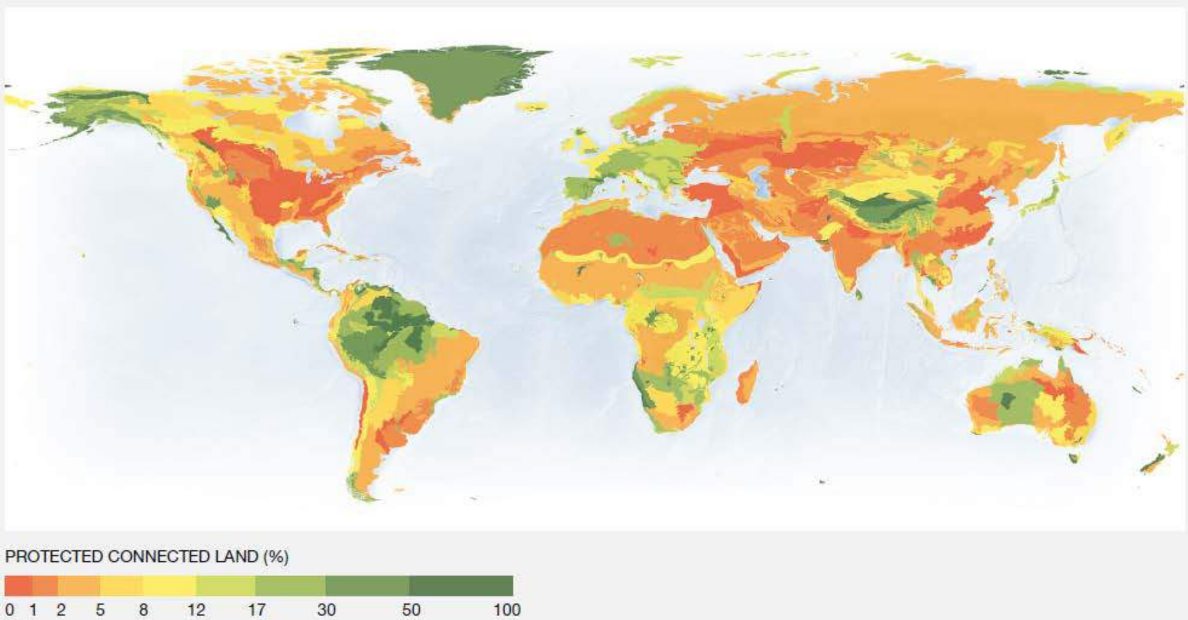
Based on a review of seven large-scale fragmentation experiments, running in five continents through the last 35 years, Haddad *et al.* (2015) estimated that habitat fragmentation reduces biodiversity by 13% to 75%. The ecosystem services that depend on native species, such as pollination, pest control 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.

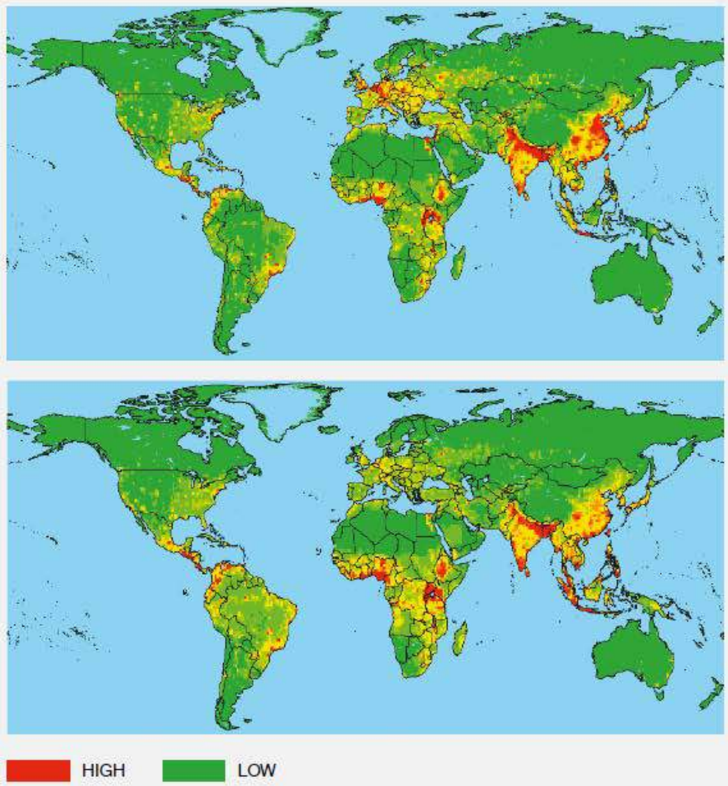
Most emerging diseases are driven by human activities that modify the environment or spread pathogens into new ecological niches (Table 4.10). The magnitude and direction of altered disease incidence due to anthropogenic disturbance differ globally and between ecosystems (Figure 4.30). Biophysical drivers 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.

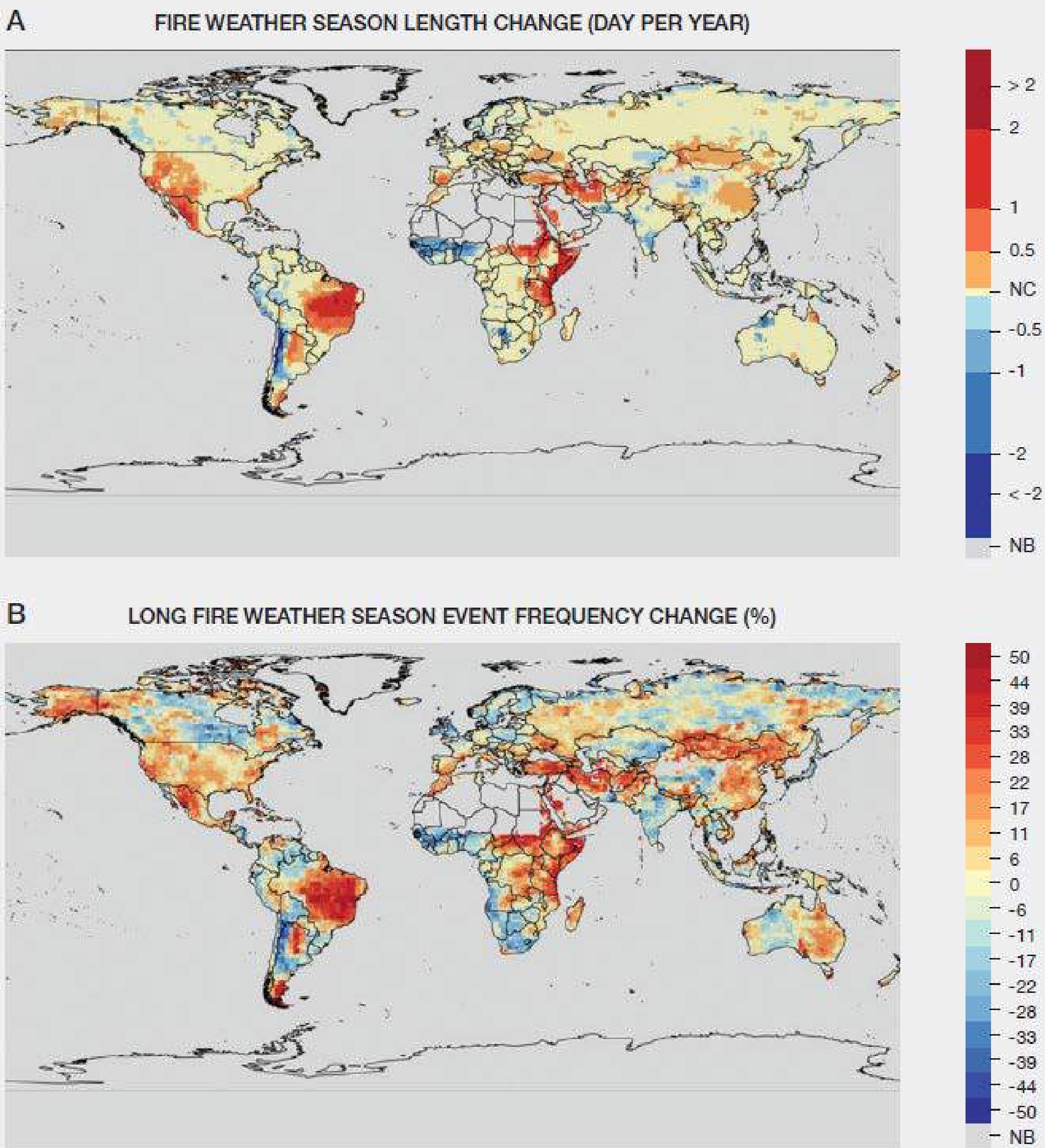
More frequent extreme weather events can reduce the resistance and defences of many organisms and make them vulnerable to diseases and predation that normally cause little harm. Damaged plants can facilitate transmission of viruses and bacteria (Mina & Sinha, 2008). Spider mites, grasshoppers and aphids 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.

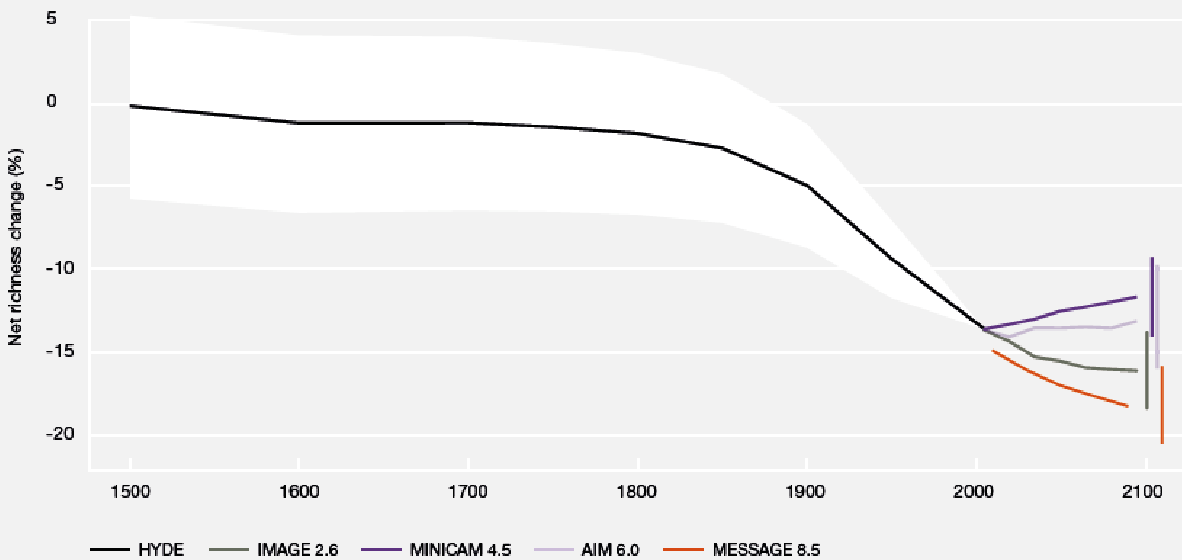
4.2.9 Biodiversity Loss

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).



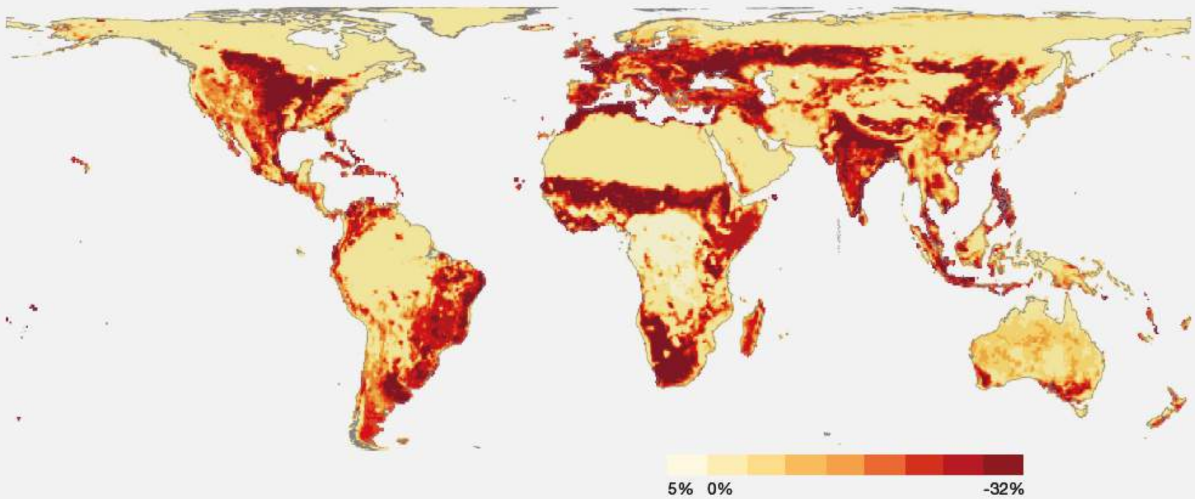
The distribution of declines in biodiversity is not geographically uniform. Croplands, pastures and urban areas have been found to have suffered the highest decrease in species richness and abundance compared to primary ecosystems and secondary growth, in a review of data from 284 publications including 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).

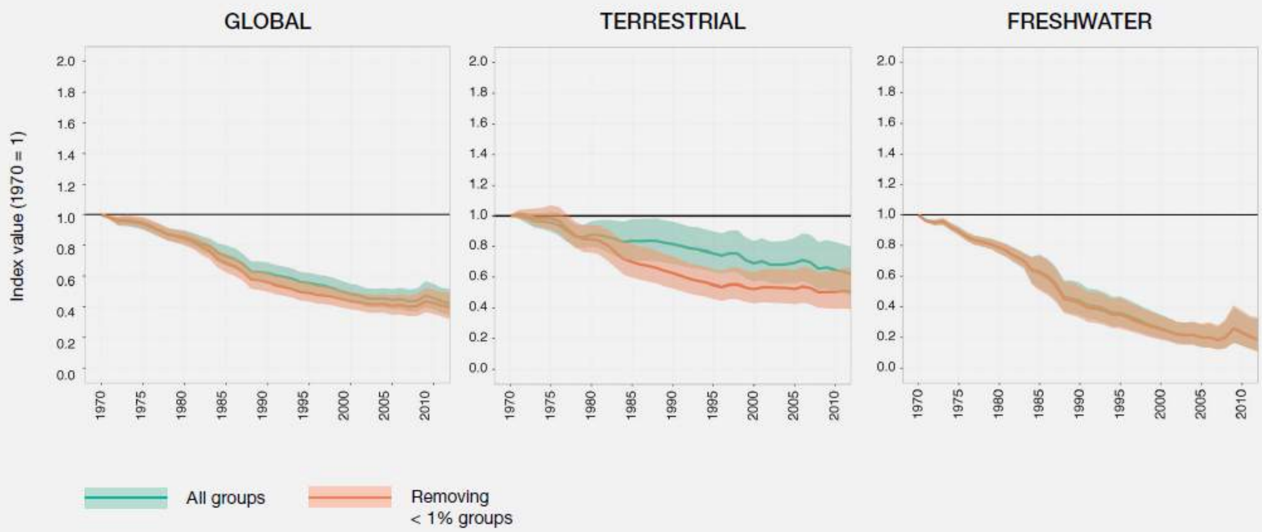


Table 4.12 Distribution of species facing imminent extinction (i.e., trigger species) and historically extinct species among taxa and island, mountain and low mainland areas (from Ricketts *et al.*, 2005). Trigger species meet the criteria necessary to trigger sites for this analysis. Historically extinct 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.

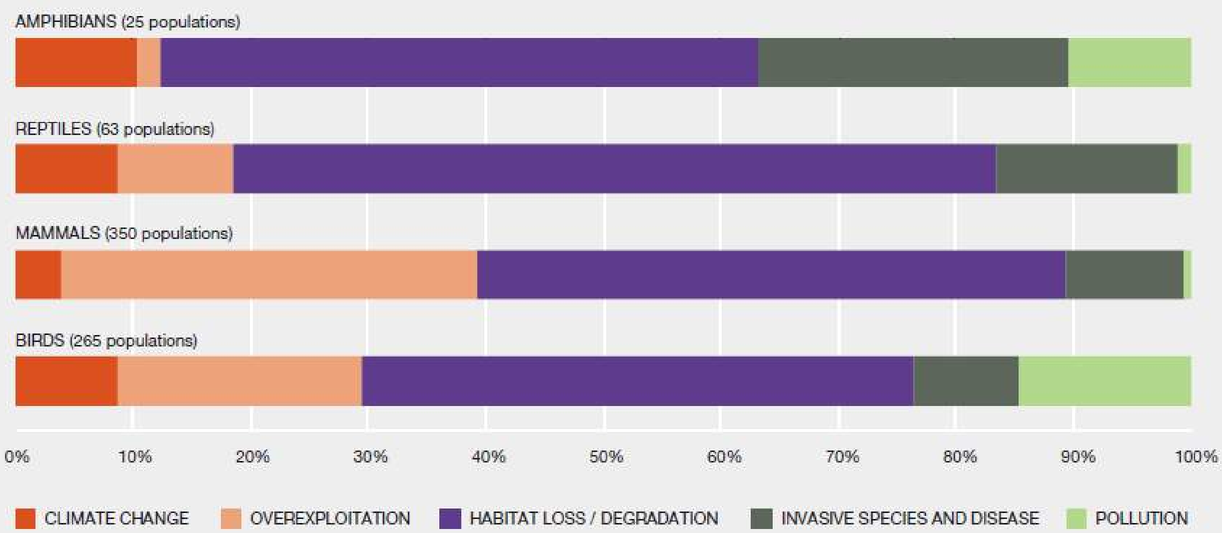
[3] Reptiles include only taxa that have been globally assessed by the 2004 IUCN Red List: order *Testudines,* order *Crocodylia*, and family *Iguanidae*.

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

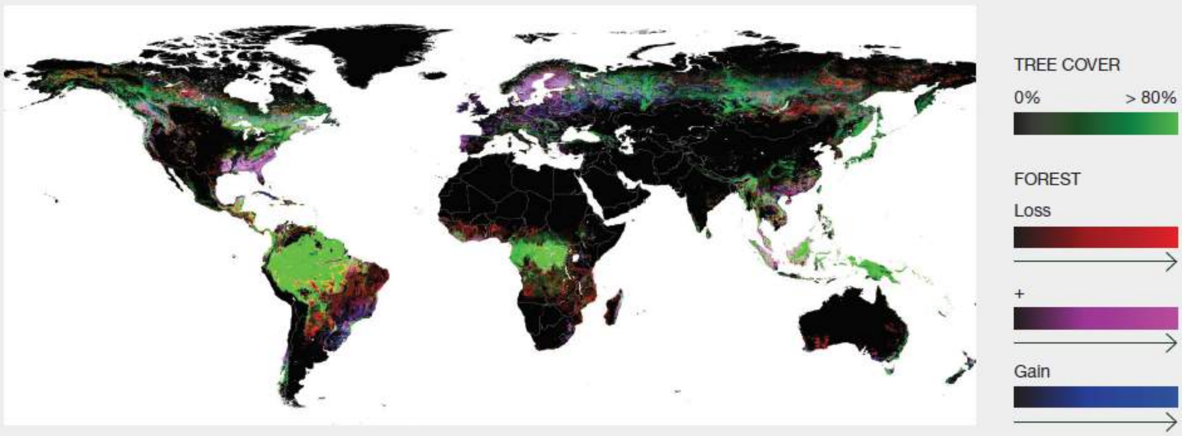
Forests worldwide are in a state of flux, with accelerating losses in some regions and gains in others (see Chapter 3, Section 3.3.3 for additional information on drivers). From 2000 to 2012 there was a net loss in global forest area of 2.3 million km^2 and a gain of 0.8 million km^2 (Hansen *et al.*, 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.



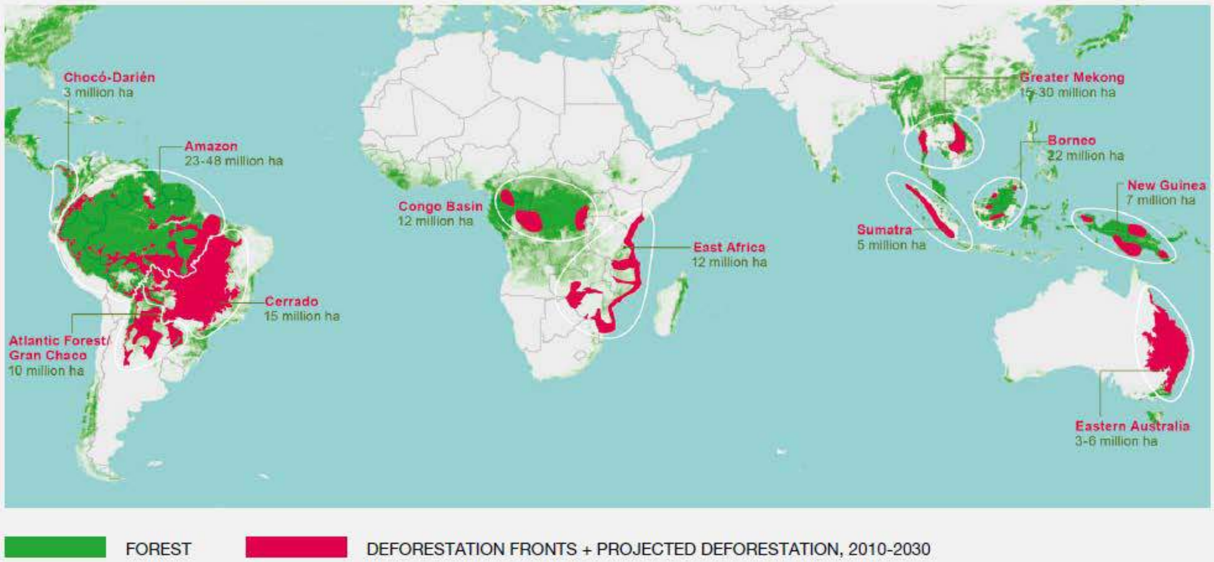
Planted forests (see Box 4.2) account for 25-100% of gains and increasingly substitute for natural forests, particularly in Africa. The global rate of planted-forest expansion since 1990 is close to a target of 2.4% per annum necessary to replace wood supplied from natural forests in the medium term, 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.**

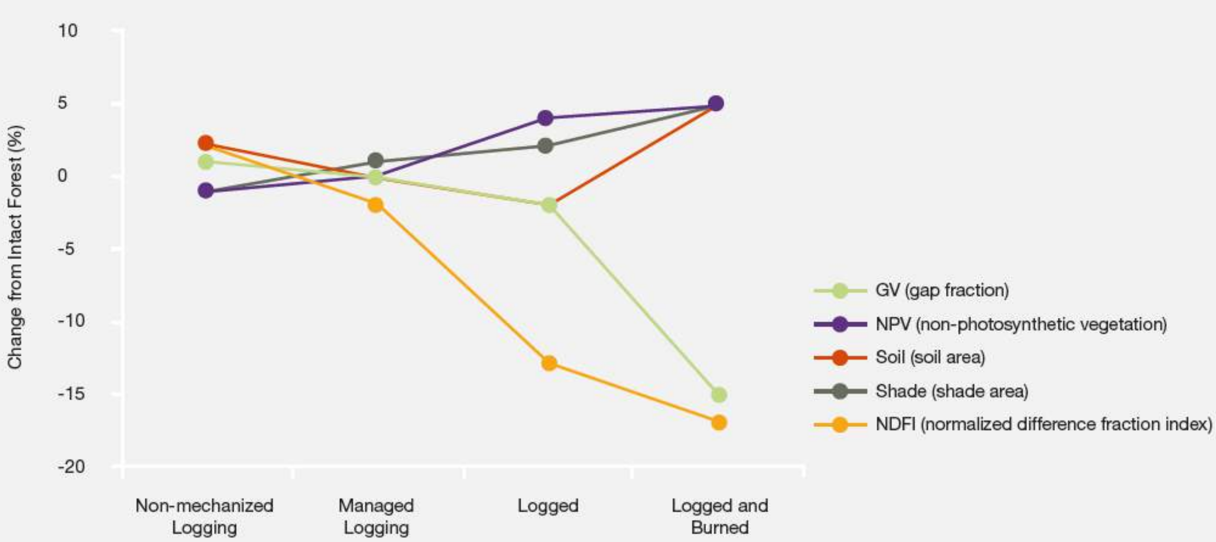
The 11 regions where the losses are expected to be greatest are circled. Source: WWF (2015).



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.

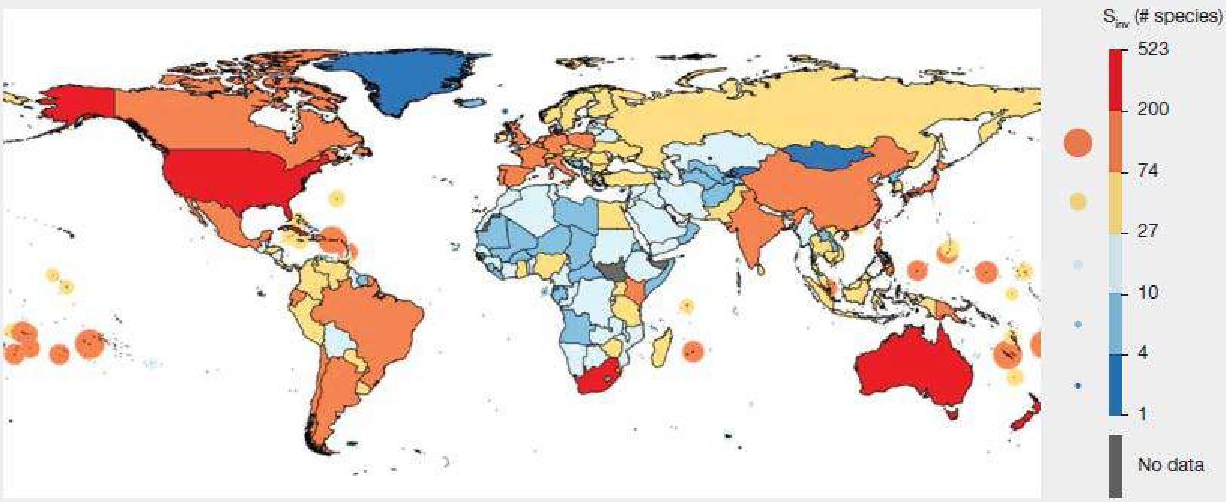
A

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).

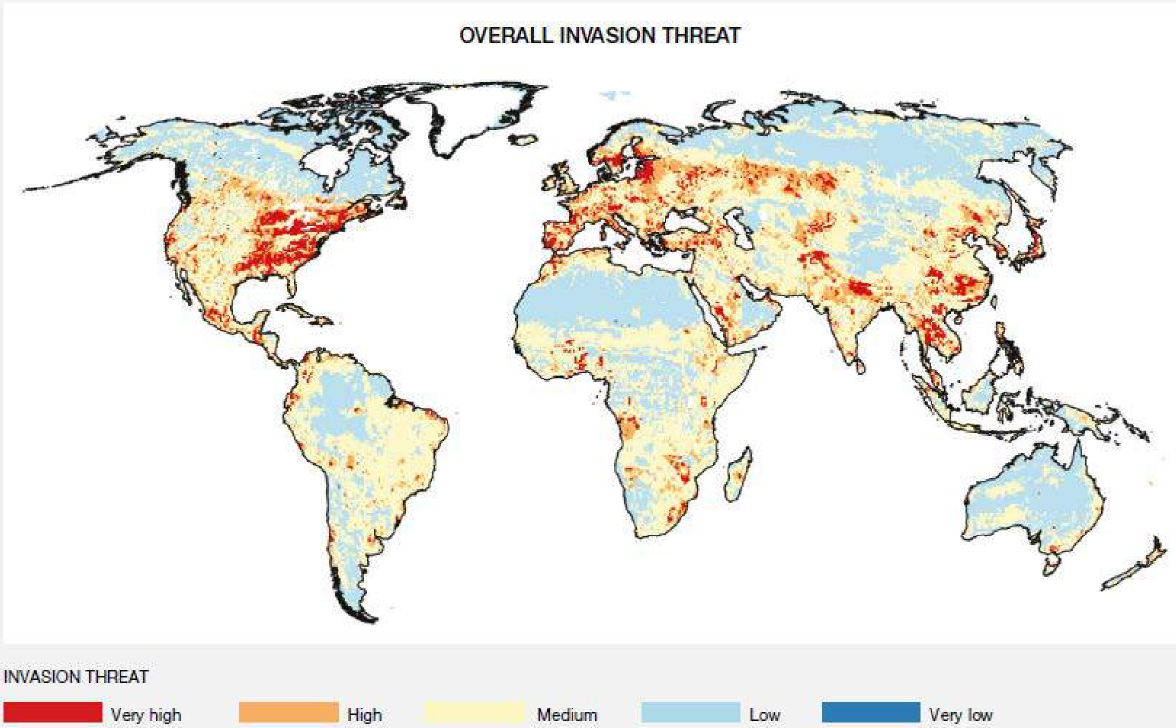
Figure 4 40 Number of invasive alien species per country, excluding overseas territories.

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).

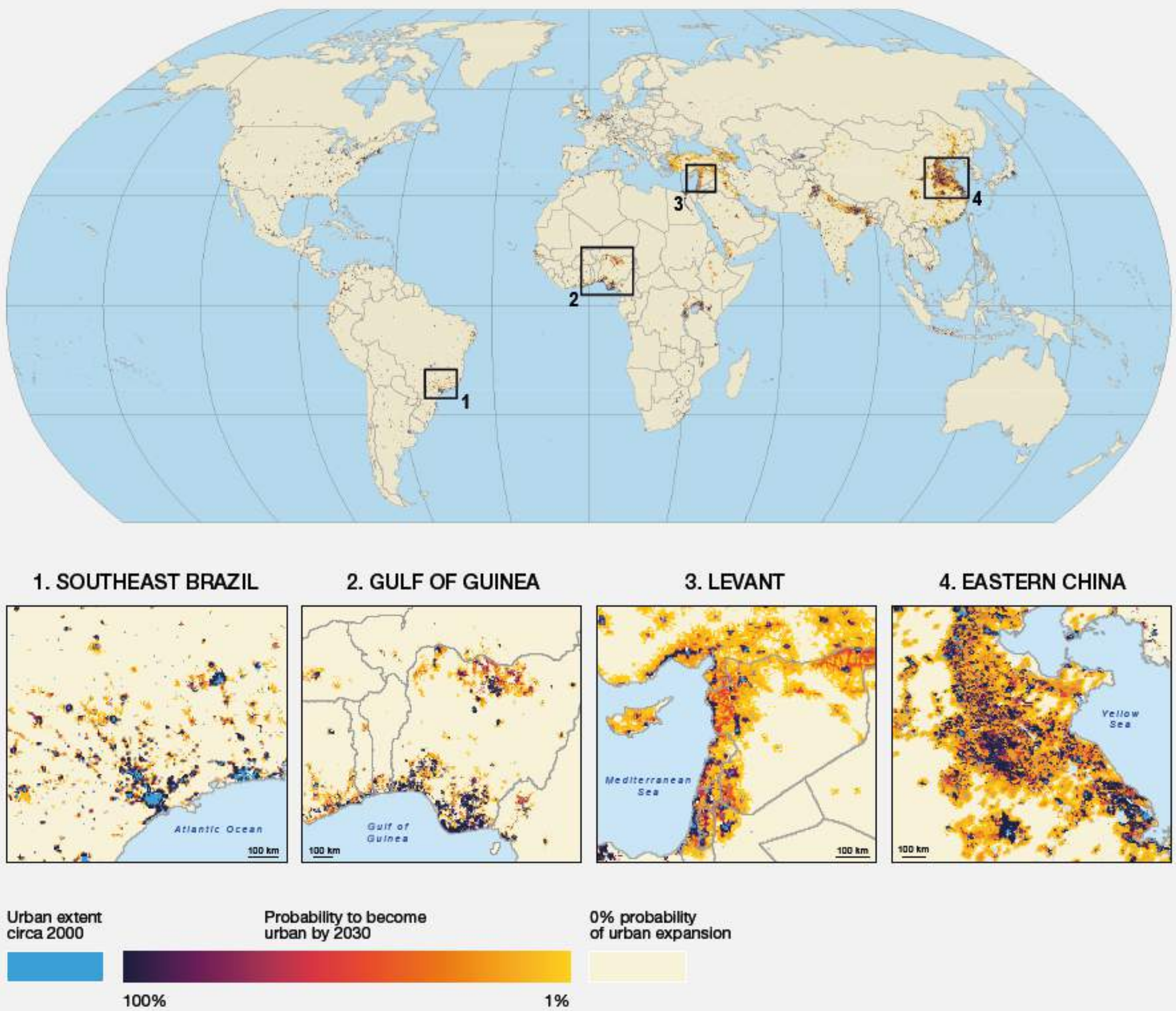
Particularly in richer countries, urban and suburban development has led to high nutrient loads in many streams and rivers due to run off from over-fertilization of lawns and golf-courses, faulty septic systems and cracked sewer pipes.

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).

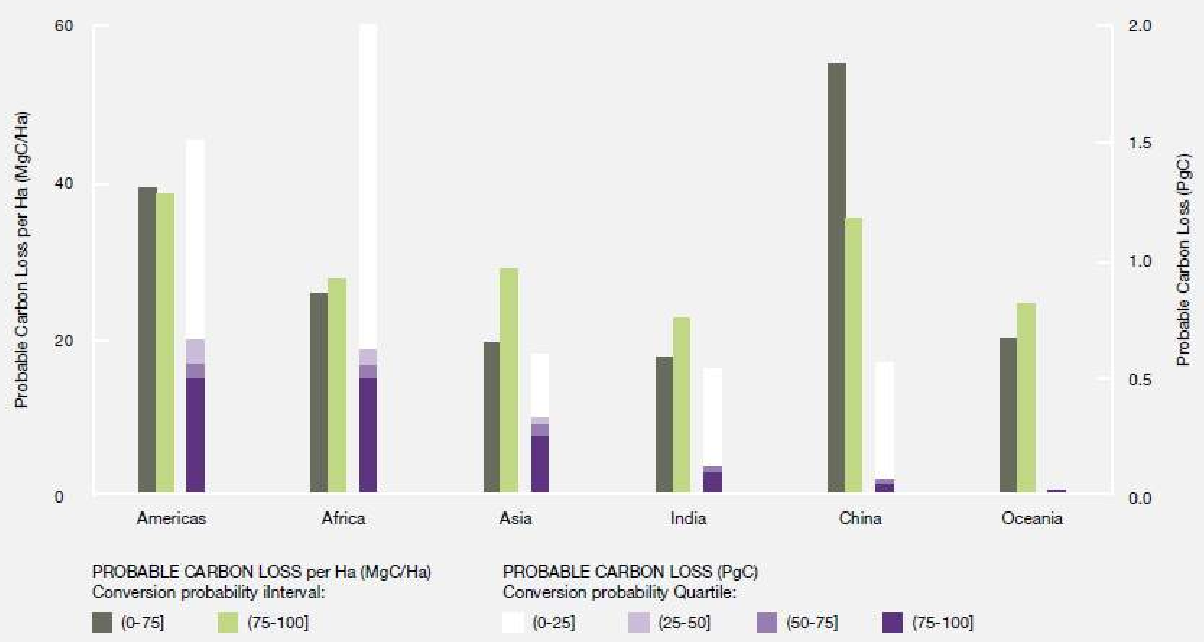


Figure 4 44 Average primary production (MgC ha^{-1}) and total carbon (PgC) loss in 2030 by region within the pan-tropics based on the probability of conversion. Source: Seto *et al.* (2012).

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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UNITED NATIONS

Food and Agriculture Organization of the United Nations

BES

IPBES/6/INF/1/Rev.1

Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services

Distr.: General
22 June 2018

English only

Plenary of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services
Sixth session
Medellin, Colombia, 18–24 March 2018
Agenda item 7

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.