1.1 Introduction

1.1.1 Scope of the chapter

This chapter examines the scientific understanding of how climate change impacts land degradation, and vice versa, with a focus on non-drylands. Land degradation of drylands is covered in Chapter 3. After providing definitions and the context (Section 4.1) we proceed with a theoretical explanation of the different processes of land degradation and how they are related to climate and to climate change, where possible (Section 4.2). Two sections are devoted to a systematic assessment of the scientific literature on status and trend of land degradation (Section 4.3) and projections of land degradation (Section 4.4). Then follows a section where we assess the impacts of climate change mitigation options, bioenergy and land-based technologies for carbon dioxide removal (CDR), on land degradation (Section 4.5). The ways in which land degradation can impact on climate and climate change are assessed in Section 4.6. The impacts of climate-related land degradation on human and natural systems are assessed in Section 4.7. The remainder of the chapter assesses land degradation mitigation options based on the concept of sustainable land management: avoid, reduce and reverse land degradation (Section 4.8), followed by a presentation of eight illustrative case studies of land degradation and remedies (Section 4.9). The chapter ends with a discussion of the most critical knowledge gaps and areas for further research (Section 4.10).

1.1.2 Perspectives of land degradation

Land degradation has accompanied humanity at least since the widespread adoption of agriculture during Neolithic time, some 10,000 to 7,500 years ago (Dotterweich 2013; Butzer 2005; Dotterweich 2008) and the associated population increase (Bocquet-Appel 2011). There are indications that the levels of greenhouse gases (GHGs) – particularly carbon dioxide (CO_2) and methane (CH_4) – in the atmosphere already started to increase more than 3,000 years ago as a result of expanding agriculture, clearing of forests, and domestication of wild animals (Fuller et al. 2011; Kaplan et al. 2011; Vavrus et al. 2018; Ellis et al. 2013). While the development of agriculture (cropping and animal husbandry) underpinned the development of civilisations, political institutions and prosperity, farming practices led to conversion of forests and grasslands to farmland, and the heavy reliance on domesticated annual grasses for our food production meant that soils started to deteriorate through seasonal mechanical disturbances (Turner et al. 1990; Steffen et al. 2005; Ojima et al. 1994; Ellis et al. 2013). More recently, urbanisation has significantly altered ecosystems (Cross-Chapter Box 4 in Chapter 2). Since around 1850, about 35% of human-caused CO_2 emissions to the atmosphere has come from land as a combined effect of land degradation and land-use change (Foley et al. 2005) and about 38% of the Earth's land area has been converted to agriculture (Foley et al. 2011). See Chapter 2 for more details.

Not all human impacts on land result in degradation according to the definition of land degradation used in this report (Section 4.1.3). There are many examples of long-term sustainably managed land around the world (such as terraced agricultural systems and sustainably managed forests) although degradation and its management are the focus of this chapter. We also acknowledge that human use of land and ecosystems provides essential goods and services for society (Foley et al. 2005; Millennium Ecosystem Assessment 2005).

Land degradation was long subject to a polarised scientific debate between disciplines and perspectives in which social scientists often proposed that natural scientists exaggerated land degradation as a global problem (Blaikie and Brookfield 1987; Forsyth 1996; Lukas 2014; Zimmerer 1993). The elusiveness of the concept in combination with the difficulties of measuring and monitoring land degradation at global and regional scales by extrapolation and aggregation of empirical studies at local scales, such as the Global Assessment of Soil Degradation database (GLASOD) (Sonneveld and Dent 2009) contributed to conflicting views. The conflicting views were not confined to science only, but also caused tension between the scientific understanding of land degradation and policy (Andersson et al. 2011; Behnke and Mortimore 2016; Grainger 2009; Toulmin and Brock 2016). Another weakness of many land degradation studies is the exclusion of the views and experiences of the land users, whether farmers or forest-dependent communities (Blaikie and Brookfield 1987; Fairhead and Scoones 2005; Warren 2002; Andersson et al. 2011). More recently, the polarised views described above have been reconciled under the umbrella of Land Change Science, which has emerged as an interdisciplinary field aimed at examining the dynamics of land cover and land-use as a coupled human-environment system (Turner et al. 2007). A comprehensive discussion about concepts and different perspectives of land degradation was presented in Chapter 2 of the recent report from the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) on land degradation (Montanarella et al. 2018).

In summary, agriculture and clearing of land for food and wood products have been the main drivers of land degradation for millennia (*high confidence*). This does not mean, however, that agriculture and forestry always cause land degradation (*high confidence*); sustainable management is possible but not always practised (*high confidence*). Reasons for this are primarily economic, political and social.

1.1.3 Definition of land degradation

To clarify the scope of this chapter, it is important to start by defining land itself. The Special Report on Climate Change and Land (SRCCL) defines land as 'the terrestrial portion of the biosphere that comprises the natural resources (soil, near surface air, vegetation and other biota, and water), the ecological processes, topography, and human settlements and infrastructure that operate within that system' (Henry et al. 2018, adapted from FAO 2007; UNCCD 1994).

Land degradation is defined in many different ways within the literature, with differing emphases on biodiversity, ecosystem functions and ecosystem services (e.g., Montanarella et al. 2018). In this report, land degradation is defined as a *negative trend in land condition, caused by direct or indirect human-induced processes including anthropogenic climate change, expressed as long-term* *reduction or loss of at least one of the following: biological productivity, ecological integrity or value to humans*. This definition applies to forest and non-forest land: forest degradation is land degradation that occurs in forest land. Soil degradation refers to a subset of land degradation processes that directly affect soil.

The SRCCL definition is derived from the IPCC AR5 definition of desertification, which is in turn taken from the United Nations Convention to Combat Desertification (UNCCD): 'Land degradation in arid, semi-arid, and dry sub-humid areas resulting from various factors, including climatic variations and human activities. Land degradation in arid, semi-arid, and dry sub-humid areas is a reduction or loss of the biological or economic productivity and integrity of rainfed cropland, irrigated cropland, or range, pasture, forest, and woodlands resulting from land uses or from a process or combination of processes, including processes arising from human activities and habitation patterns, such as (i) soil erosion caused by wind and/or water; (ii) deterioration of the physical, chemical, biological, or economic properties of soil; and (iii) long-term loss of natural vegetation' (UNCCD 1994, Article 1).

For this report, the SRCCL definition is intended to complement the more detailed UNCCD definition above, expanding the scope to all regions, not just drylands, providing an operational definition that emphasises the relationship between land degradation and climate. Through its attention to the three aspects – biological productivity, ecological integrity and value to humans – the SRCCL definition is consistent with the Land Degradation Neutrality (LDN) concept, which aims to maintain or enhance the land-based natural capital, and the ecosystem services that flow from it (Cowie et al. 2018).

In the SRCCL definition of land degradation, changes in land condition resulting solely from natural processes (such as volcanic eruptions and tsunamis) are not considered land degradation, as these are not direct or indirect human-induced processes. Climate variability exacerbated by human-induced climate change can contribute to land degradation. Value to humans can be expressed in terms of ecosystem services or Nature's Contributions to People.

The definition recognises the reality presented in the literature that land-use and land management decisions often result in trade-offs between time, space, ecosystem services, and stakeholder groups (e.g., Dallimer and Stringer 2018). The interpretation of a negative trend in land condition is somewhat subjective, especially where there is a trade-off between ecological integrity and value to humans. The definition also does not consider the magnitude of the negative trend or the possibility that a negative trend in one criterion may be an acceptable trade-off for a positive trend in another criterion. For example, reducing timber yields to safeguard biodiversity by leaving on site more wood that can provide habitat, or vice versa, is a trade-off that needs to be evaluated based on context (i.e. the broader landscape) and society's priorities. Reduction of biological productivity *or* ecological integrity *or* value to humans *can* constitute degradation, but any one of these changes need not necessarily be considered degradation. Thus, a land-use change that reduces ecological integrity and enhances sustainable food production at a specific location is not necessarily degradation. Different stakeholder groups with different world views value ecosystem services differently. As Warren (2002) explained: land degradation is contextual. Further, a decline in biomass carbon stock does not always signify degradation, such as when caused by periodic forest harvest. Even a decline in productivity may not equate to land degradation, such as when a high-intensity agricultural system is converted to a lower-input, more sustainable production system.

In the SRCCL definition, degradation is indicated by a negative trend in land condition during the period of interest, thus the baseline is the land condition at the start of this period. The concept of baseline is theoretically important but often practically difficult to implement for conceptual and methodological reasons (Herrick et al. 2019; Prince et al. 2018; also Sections 4.3.1 and 4.4.1). Especially in biomes characterised by seasonal and interannual variability, the baseline values of the indicators to be assessed should be determined by averaging data over a number of years prior to the commencement of the assessment period (Orr et al. 2017) (Section 4.2.4).

Forest degradation is land degradation in forest remaining forest. In contrast, deforestation refers to the conversion of forest to non-forest that involves a loss of tree cover and a change in land use. Internationally accepted definitions of forest (FAO 2015; UNFCCC 2006) include lands where tree cover has been lost temporarily, due to disturbance or harvest, with an expectation of forest regrowth. Such temporary loss of forest cover, therefore, is not deforestation.

1.1.4 Land degradation in previous IPCC reports

Several previous IPCC assessment reports include brief discussions of land degradation. In AR5 WGIII land degradation is one factor contributing to uncertainties of the mitigation potential of land-based ecosystems, particularly in terms of fluxes of soil carbon (Smith et al. 2014, p. 817). In AR5 WGI, soil carbon was discussed comprehensively but not in the context of land degradation, except forest degradation (Ciais et al. 2013) and permafrost degradation (Vaughan et al. 2013). Climate change impacts were discussed comprehensively in AR5 WGII, but land degradation was not prominent. Land-use and land-cover changes were treated comprehensively in terms of effects on the terrestrial carbon stocks and flows (Settele et al. 2015) but links to land degradation were, to a large extent, missing. Land degradation was discussed in relation to human security as one factor which, in combination with extreme weather events, has been proposed to contribute to human migration (Adger et al. 2014), an issue discussed more comprehensively in this chapter (Section 4.7.3). Drivers and processes of degradation by which land-based carbon is released to the atmosphere and/or the long-term reduction in the capacity of the land to remove atmospheric carbon and to store this in biomass and soil carbon, have been discussed in the methodological reports of IPCC (IPCC 2006, 2014a) but less so in the assessment reports.

The Special Report on Land Use, Land-Use Change and Forestry (SR-LULUCF) (Watson et al. 2000) focused on the role of the biosphere in the global cycles of GHG. Land degradation was not addressed in a comprehensive way. Soil erosion was discussed as a process by which soil carbon is lost and the productivity of the land is reduced. Deposition of eroded soil carbon in marine sediments was also mentioned as a possible mechanism for permanent sequestration of terrestrial carbon (Watson et al. 2000, p. 194). The possible impacts of climate change on land productivity and degradation were not discussed comprehensively. Much of the report was about how to account for sources and sinks of terrestrial carbon under the Kyoto Protocol.

The IPCC Special Report on Managing the Risks of Extreme Events and Disasters to Advance Climate Change Adaptation (SREX) (IPCC 2012) did not provide a definition of land degradation. Nevertheless, it addressed different aspects related to some types of land degradation in the context of weather and climate extreme events. From this perspective, it provided key information on both observed and projected changes in weather and climate (extremes) events that are relevant to extreme impacts on socio-economic systems and on the physical components of the environment, notably on permafrost in mountainous areas and coastal zones for different geographic regions, but few explicit links to land degradation. The report also presented the concept of sustainable land management as an effective risk-reduction tool.

Land degradation has been treated in several previous IPCC reports, but mainly as an aggregated concept associated with GHG emissions, or as an issue that can be addressed through adaptation and mitigation.

1.1.5 Sustainable land management (SLM) and sustainable forest management (SFM)

Sustainable land management (SLM) is defined as 'the stewardship and use of land resources, including soils, water, animals and plants, to meet changing human needs, while simultaneously ensuring the long-term productive potential of these resources and the maintenance of their environmental functions' – adapted from World Overview of Conservation Approaches and Technologies (WOCAT n.d.). Achieving the objective of ensuring that productive potential is maintained in the long term will require implementation of adaptive management and 'triple loop learning', that seeks to monitor outcomes, learn from experience and emerging new knowledge, modifying management accordingly (Rist et al. 2013).

Sustainable Forest Management (SFM) is defined as 'the stewardship and use of forests and forest lands in a way, and at a rate, that maintains their biodiversity, productivity, regeneration capacity, vitality and their potential to fulfill, now and in the future, relevant ecological, economic and social functions, at local, national, and global levels, and that does not cause damage to other ecosystems' (Forest Europe 1993; Mackey et al. 2015). This SFM definition was developed by the Ministerial Conference on the Protection of Forests in Europe and has since been adopted by the Food and Agriculture Organization. Forest management that fails to meet these sustainability criteria can contribute to land degradation.

Land degradation can be reversed through restoration and rehabilitation. These terms are defined in the Glossary, along with other terms that are used but not explicitly defined in this section of the report. While the definitions of SLM and SFM are very similar and could be merged, both are included to maintain the subtle differences in the existing definitions. SFM can be considered a subset of SLM – that is, SLM applied to forest land.

Climate change impacts interact with land management to determine sustainable or degraded outcome (Figure 4.1). Climate change can exacerbate many degradation processes (Table 4.1) and introduce novel ones (e.g., permafrost thawing or biome shifts). To avoid, reduce or reverse degradation, land management activities can be selected to mitigate the impact of, and adapt to, climate change. In some cases, climate change impacts may result in increased productivity and carbon stocks, at least in the short term. For example, longer growing seasons due to climate warming can lead to higher forest productivity (Henttonen et al. 2017; Kauppi et al. 2014; Dragoni et al. 2011), but warming alone may not increase productivity where other factors such a water supply are limiting (Hember et al. 2017).

The types and intensity of human land-use and climate change impacts on lands affect their carbon stocks and their ability to operate as carbon sinks. In managed agricultural lands, degradation can result in reductions of soil organic carbon stocks, which also adversely affects land productivity and carbon sinks (Figure 4.1).

The transition from natural to managed forest landscapes usually results in an initial reduction of landscape-level carbon stocks. The magnitude of this reduction is a function of the differential in frequency of stand-replacing natural disturbances (e.g., wildfires) and harvest disturbances, as well as the age-dependence of these disturbances (Harmon et al. 1990; Kurz et al. 1998; Trofymow et al. 2008).

SFM applied at the landscape scale to existing unmanaged forests can first reduce average forest carbon stocks and subsequently increase the rate at which CO_2 is removed from the atmosphere, because net ecosystem production of forest stands is highest in intermediate stand ages (Kurz et al. 2013; Volkova et al. 2018; Tang et al. 2014). The net impact on the atmosphere depends on the magnitude of the reduction in carbon stocks, the fate of the harvested biomass (i.e. use in short – or long-lived products and for bioenergy, and therefore displacement of emissions associated with GHG-intensive building materials and fossil fuels), and the rate of regrowth. Thus, the impacts of SFM on one indicator (e.g., past reduction in carbon stocks in the forested landscape) can be negative, while those on another indicator (e.g., current forest productivity and rate of CO_2 removal from the atmosphere, avoided fossil fuel emissions) can be positive. Sustainably managed forest landscapes can have a lower biomass carbon density than unmanaged forest, but the younger forests can have a higher growth rate, and therefore contribute stronger carbon sinks than older forests (Trofymow et al. 2008; Volkova et al. 2018; Poorter et al. 2016).

Selective logging and thinning can maintain and enhance forest productivity and achieve co-benefits when conducted with due care for the residual stand and at intensity and frequency that does not exceed the rate of regrowth (Romero and Putz 2018). In contrast, unsustainable logging practices can lead to stand-level degradation. For example, degradation occurs when selective logging

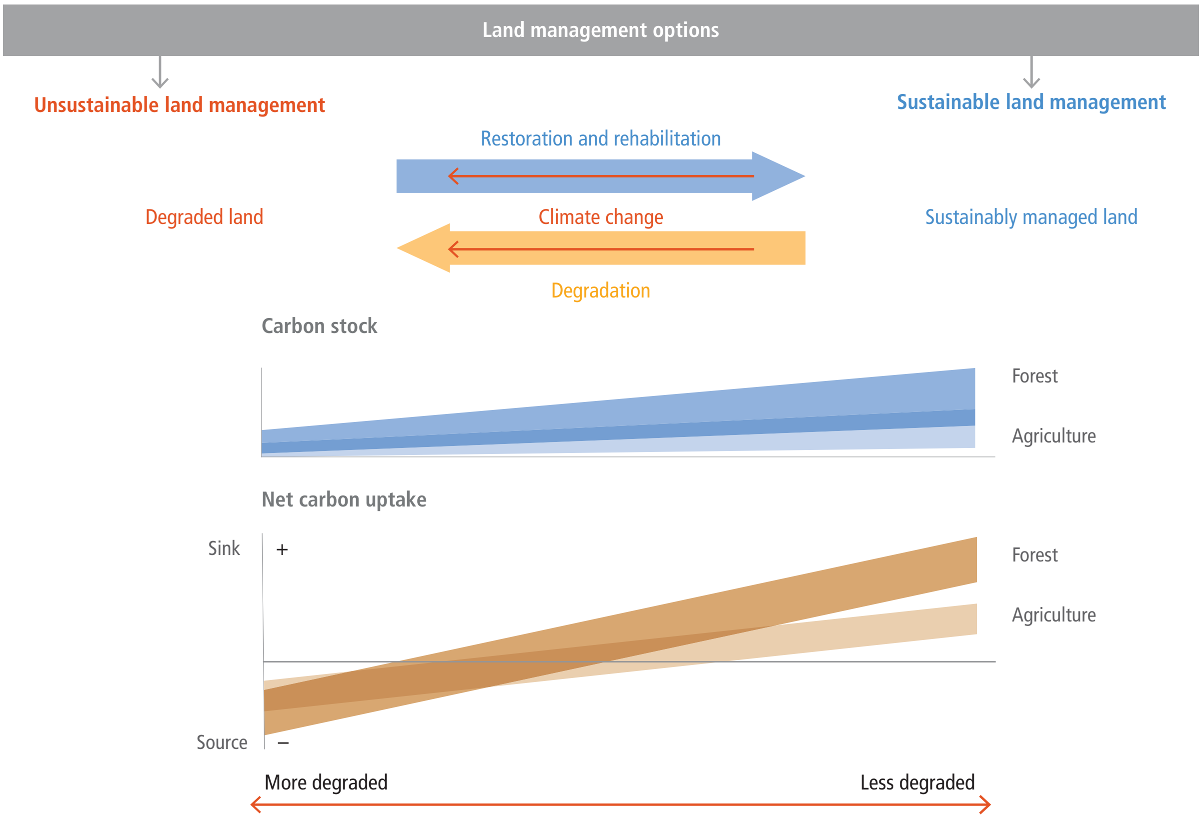

Figure 4.1 | Conceptual figure illustrating that climate change impacts interact with land management to determine sustainable or degraded outcome. Climate change can exacerbate many degradation processes (Table 4.1) and introduce novel ones (e.g., permafrost thawing or biome shifts), hence management needs to respond to climate impacts in order to avoid, reduce or reverse degradation. The types and intensity of human land-use and climate change impacts on lands affect their carbon stocks and their ability to operate as carbon sinks. In managed agricultural lands, degradation typically results in reductions of soil organic carbon stocks, which also adversely affects land productivity and carbon sinks. In forest land, reduction in biomass carbon stocks alone is not necessarily an indication of a reduction in carbon sinks. Sustainably managed forest landscapes can have a lower biomass carbon density but the younger forests can have a higher growth rate, and therefore contribute stronger carbon sinks, than older forests. Ranges of carbon sinks in forest and agricultural lands are overlapping. In some cases, climate change impacts may result in increased productivity and carbon stocks, at least in the short term.

(high-grading) removes valuable large-diameter trees, leaving behind damaged, diseased, non-commercial or otherwise less productive trees, reducing carbon stocks and also adversely affecting subsequent forest recovery (Belair and Ducey 2018; Nyland 1992).

SFM is defined using several criteria (see above) and its implementation will typically involve trade-offs among these criteria. The conversion of primary forests to sustainably managed forest ecosystems increases relevant economic, social and other functions but often with adverse impacts on biodiversity (Barlow et al. 2007). In regions with infrequent or no stand-replacing natural disturbances, the timber yield per hectare harvested in managed secondary forests is typically lower than the yield per hectare from the first harvest in the primary forest (Romero and Putz 2018).

The sustainability of timber yield has been achieved in temperate and boreal forests where intensification of management has resulted in increased growing stocks and increased harvest rates in countries where forests had previously been overexploited (Henttonen et al. 2017; Kauppi et al. 2018). However, intensification of management to increase forest productivity can be associated with reductions in biodiversity. For example, when increased productivity is achieved by periodic thinning and removal of trees that would otherwise die due to competition, thinning reduces the amount of dead organic matter of snags and coarse woody debris that can provide habitat, and this loss reduces biodiversity (Spence 2001; Ehnström 2001) and forest carbon stocks (Russell et al. 2015; Kurz et al. 2013). Recognition of adverse biodiversity impacts of high-yield forestry is leading to modified management aimed at increasing habitat availability through, for example, variable retention logging and continuous cover management (Roberts et al. 2016) and through the re-introduction of fire disturbances in landscapes where fires have been suppressed (Allen et al. 2002). Biodiversity losses are also observed during the transition from primary to managed forests in tropical regions (Barlow et al. 2007) where tree species diversity can be very high – for example, in the Amazon region, about 16,000 tree species are estimated to exist (ter Steege et al. 2013).

Forest certification schemes have been used to document SFM outcomes (Rametsteiner and Simula 2003) by assessing a set of criteria and indicators (e.g., Lindenmayer et al. 2000). While many of the certified forests are found in temperate and boreal countries (Rametsteiner and Simula 2003; MacDicken et al. 2015), examples from the tropics also show that SFM can improve outcomes. For example, selective logging emits 6% of the tropical GHG annually and improved logging practices can reduce emissions by 44% while maintaining timber production (Ellis et al. 2019). In the Congo Basin, implementing reduced impact logging (RIL-C) practices can cut emissions in half without reducing the timber yield (Umunay et al. 2019). SFM adoption depends on the socio-economic and political context, and its improvement depends mainly on better reporting and verification (Siry et al. 2005).

The successful implementation of SFM requires well-established and functional governance, monitoring, and enforcement mechanisms to eliminate deforestation, illegal logging, arson, and other activities that are inconsistent with SFM principles (Nasi et al. 2011). Moreover, following human and natural disturbances, forest regrowth must be ensured through reforestation, site rehabilitation activities or natural regeneration. Failure of forests to regrow following disturbances will lead to unsustainable outcomes and long-term reductions in forest area, forest cover, carbon density, forest productivity and land-based carbon sinks (Nasi et al. 2011).

Achieving all of the criteria of the definitions of SLM and SFM is an aspirational goal that will be made more challenging where climate change impacts, such as biome shifts and increased disturbances, are predicted to adversely affect future biodiversity and contribute to forest degradation (Warren et al. 2018). Land management to enhance land sinks will involve trade-offs that need to be assessed within their spatial, temporal and societal context.

1.1.6 The human dimension of land degradation and forest degradation

Studies of land and forest degradation are often biased towards biophysical aspects, both in terms of its processes, such as erosion or nutrient depletion, and its observed physical manifestations, such as gullying or low primary productivity. Land users' own perceptions and knowledge about land conditions and degradation have often been neglected or ignored by both policymakers and scientists (Reed et al. 2007; Forsyth 1996; Andersson et al. 2011). A growing body of work is nevertheless beginning to focus on land degradation through the lens of local land users (Kessler and Stroosnijder 2006; Fairhead and Scoones 2005; Zimmerer 1993; Stocking et al. 2001) and the importance of local and indigenous knowledge within land management is starting to be appreciated (Montanarella et al. 2018). Climate change impacts directly and indirectly on the social reality, the land users, and the ecosystem, and vice versa. Land degradation can also have an impact on climate change (Section 4.6).

The use and management of land is highly gendered and is expected to remain so for the foreseeable future (Kristjanson et al. 2017). Women often have less formal access to land than men and less influence over decisions about land, even if they carry out many of the land management tasks (Jerneck 2018a; Elmhirst 2011; Toulmin 2009; Peters 2004; Agarwal 1997; Jerneck 2018b). Many oft-cited general statements about women's subordination in agriculture are difficult to substantiate, yet it is clear that gender inequality persists (Doss et al. 2015). Even if women's access to land is changing formally (Kumar and Quisumbing 2015), the practical outcome is often limited due to several other factors related to both formal and informal institutional arrangements and values (Lavers 2017; Kristjanson et al. 2017; Djurfeldt et al. 2018). Women are also affected differently than men when it comes to climate change, having lower adaptive capacities due to factors such as prevailing land tenure frameworks, less access to other capital assets and dominant cultural practices (Vincent et al. 2014; Antwi-Agyei et al. 2015; Gabrielsson et al. 2013). This affects the options available to women to respond to both land degradation and climate change. Indeed, access to land and other assets (e.g., education and training) is key in shaping land-use and land management strategies (Liu et al. 2018b; Lambin et al. 2001). Young people are also often disadvantaged in terms of access to resources and decision-making power, even though they carry out much of the day-to-day work (Wilson et al. 2017; Kosec et al. 2018; Naamwintome and Bagson 2013).

Land rights differ between places and are dependent on the political-economic and legal context (Montanarella et al. 2018). This means that there is no universally applicable best arrangement. Agriculture in highly erosion-prone regions requires site-specific and long-lasting soil and water conservation measures, such as terraces (Section 4.8.1), which may benefit from secure private land rights (Tarfasa et al. 2018; Soule et al. 2000). Pastoral modes of production and community-based forest management systems are often dominated by, and benefit from, communal land tenure arrangements, which may conflict with agricultural/forestry modernisation policies implying private property rights (Antwi-Agyei et al. 2015; Benjaminsen and Lund 2003; Itkonen 2016; Owour et al. 2011; Gebara 2018).

Cultural ecosystem services, defined as the non-material benefits people obtain from ecosystems through spiritual enrichment, cognitive development, reflection, recreation and aesthetic experiences (Millennium Ecosystem Assessment 2005) are closely linked to land and ecosystems, although often under-represented in the literature on ecosystem services (Tengberg et al. 2012; Hernández-Morcillo et al. 2013). Climate change interacting with land conditions can impact on cultural aspects, such as sense of place and sense of belonging (Olsson et al. 2014).

1.2 Land degradation in the context of climate change

Land degradation results from a complex chain of causes making the clear distinction between direct and indirect drivers difficult. In the context of climate change, an additional complex aspect is brought by the reciprocal effects that both processes have on each other (i.e. climate change influencing land degradation and vice versa). In this chapter, we use the terms 'processes' and 'drivers' with the following meanings:

Processes of land degradation are those direct mechanisms by which land is degraded and are similar to the notion of 'direct drivers' in the Millennium Ecosystem Assessment framework (Millennium Ecosystem Assessment, 2005). A comprehensive list of land degradation processes is presented in Table 4.1.

Drivers of land degradation are those indirect conditions which may drive processes of land degradation and are similar to the notion of 'indirect drivers' in the Millennium Ecosystem Assessment framework. Examples of indirect drivers of land degradation are changes in land tenure or cash crop prices, which can trigger land-use or management shifts that affect land degradation.

An exact demarcation between processes and drivers is not possible. Drought and fires are described as drivers of land degradation in the next section but they can also be a process: for example, if repeated fires deplete seed sources, they can affect regeneration and succession of forest ecosystems. The responses to land degradation follow the logic of the LDN concept: avoiding, reducing and reversing land degradation (Orr et al. 2017; Cowie et al. 2018).

In research on land degradation, climate and climate variability are often intrinsic factors. The role of climate change, however, is less articulated. Depending on what conceptual framework is used, climate change is understood either as a process or a driver of land degradation, and sometimes both.

1.2.1 Processes of land degradation

A large array of interactive physical, chemical, biological and human processes lead to what we define in this report as land degradation (Johnson and Lewis 2007). The biological productivity, ecological integrity (which encompasses both functional and structural attributes of ecosystems) or the human value (which includes any benefit that people get from the land) of a given territory can deteriorate as the result of processes triggered at scales that range from a single furrow (e.g., water erosion under cultivation) to the landscape level (e.g., salinisation through raising groundwater levels under irrigation). While pressures leading to land degradation are often exerted on specific components of the land systems (i.e., soils, water, biota), once degradation processes start, other components become affected through cascading and interactive effects. For example, different pressures and degradation processes can have convergent effects, as can be the case of overgrazing leading to wind erosion, landscape drainage resulting in wetland drying, and warming causing more frequent burning; all of which can independently lead to reductions of the soil organic matter (SOM) pools as a second-order process. Still, the reduction of organic matter pools is also a first-order process triggered directly by the effects of rising temperatures (Crowther et al. 2016) as well as other climate changes such as precipitation shifts (Viscarra Rossel et al. 2014). Beyond this complexity, a practical assessment of the major land degradation processes helps to reveal and categorise the multiple pathways in which climate change exerts a degradation pressure (Table 4.1).

Conversion of freshwater wetlands to agricultural land has historically been a common way of increasing the area of arable land. Despite the small areal extent – about 1% of the earth's surface (Hu et al. 2017; Dixon et al. 2016) – freshwater wetlands provide a very large number of ecosystem services, such as groundwater replenishment, flood protection and nutrient retention, and are biodiversity hotspots (Reis et al. 2017; Darrah et al. 2019; Montanarella et al. 2018). The loss of wetlands since 1900 has been estimated at about 55% globally (Davidson 2014) (*low confidence*) and 35% since 1970 (Darrah et al. 2019) (*medium confidence*) which in many situations pose a problem for adaptation to climate change. Drainage causes loss of wetlands, which can be exacerbated by climate change, further reducing the capacity to adapt to climate change (Barnett et al. 2015; Colloff et al. 2016; Finlayson et al. 2017) (*high confidence*).

1.2.1.1 Types of land degradation processes

Land degradation processes can affect the soil, water or biotic components of the land as well as the reactions between them (Table 4.1). Across land degradation processes, those affecting the soil have received more attention. The most widespread and studied land degradation processes affecting soils are water and wind erosion, which have accompanied agriculture since its onset and are still dominant (Table 4.1). Degradation through erosion processes is not restricted to soil loss in detachment areas but includes impacts on transport and deposition areas as well (less commonly, deposition areas can have their soils improved by these inputs). Larger-scale degradation processes related to the whole continuum of soil erosion, transport and deposition include dune field expansion/displacement, development of gully networks and the accumulation of sediments in natural and artificial water-bodies (siltation) (Poesen and Hooke 1997; Ravi et al. 2010). Long-distance sediment transport during erosion events can have remote effects on land systems, as documented for the fertilisation effect of African dust on the Amazon (Yu et al. 2015).

Coastal erosion represents a special case among erosional processes, with reports linking it to climate change. While human interventions in coastal areas (e.g., expansion of shrimp farms) and rivers (e.g., upstream dams cutting coastal sediment supply), and economic activities causing land subsidence (Keogh and Törnqvist 2019; Allison et al. 2016) are dominant human drivers, storms and sea-level rise have already left a significant global imprint on coastal erosion (Mentaschi et al. 2018). Recent projections that take into account geomorphological and socioecological feedbacks suggest that coastal wetlands may not be reduced by sea level rise if their inland growth is accommodated with proper management actions (Schuerch et al. 2018).

Other physical degradation processes in which no material detachment and transport are involved include soil compaction, hardening, sealing and any other mechanism leading to the loss of porous space crucial for holding and exchanging air and water (Hamza and Anderson 2005). A very extreme case of degradation through pore volume loss, manifested at landscape or larger scales, is ground subsidence. Typically caused by the lowering of groundwater or oil levels, subsidence involves a sustained collapse of the ground surface, which can lead to other degradation processes such as salinisation and permanent flooding. Chemical soil degradation processes include relatively simple changes, like nutrient depletion resulting from the imbalance of nutrient extraction on harvested products and fertilisation, and more complex ones, such as acidification and increasing metal toxicity. Acidification in croplands is increasingly driven by excessive nitrogen fertilisation and, to a lower extent, by the depletion of cation like calcium, potassium or magnesium through exports in harvested biomass (Guo et al. 2010). One of the most relevant chemical degradation processes of soils in the context of climate change is the depletion of its organic matter pool. Reduced in agricultural soils through the increase of respiration rates by tillage and the decline of below-ground plant biomass inputs, SOM pools have been diminished also by the direct effects of warming, not only in cultivated land, but also under natural vegetation (Bond-Lamberty et al. 2018). Debate persists, however, on whether in more humid and carbon-rich ecosystems the simultaneous stimulation of decomposition and productivity may result in the lack of effects on soil carbon (Crowther et al. 2016; van Gestel et al. 2018). In the case of forests, harvesting – particularly if it is exhaustive, as in the case of the use of residues for energy generation – can also lead to organic matter declines (Achat et al. 2015). Many other degradation processes (e.g., wildfire increase, salinisation) have negative effects on other pathways of soil degradation (e.g., reduced nutrient availability, metal toxicity). SOM can be considered a 'hub' of degradation processes and a critical link with the climate system (Minasny et al. 2017).

Land degradation processes can also start from alterations in the hydrological system that are particularly important in the context of climate change. Salinisation, although perceived and reported in soils, is typically triggered by water table-level rises, driving salts to the surface under dry to sub-humid climates (Schofield and Kirkby 2003). While salty soils occur naturally under these climates (primary salinity), human interventions have expanded their distribution, secondary salinity with irrigation without proper drainage being the predominant cause of salinisation (Rengasamy 2006). Yet, it has also taken place under non-irrigated conditions where vegetation changes (particularly dry forest clearing and cultivation) have reduced the magnitude and depth of soil water uptake, triggering water table rises towards the surface. Changes in evapotranspiration and rainfall regimes can exacerbate this process (Schofield and Kirkby 2003). Salinisation can also result from the intrusion of sea water into coastal areas, both as a result of sea level rise and ground subsidence (Colombani et al. 2016).

Recurring flood and waterlogging episodes (Bradshaw et al. 2007; Poff 2002), and the more chronic expansion of wetlands over dryland ecosystems, are mediated by the hydrological system, on occasions aided by geomorphological shifts as well (Kirwan et al. 2011). This is also the case for the drying of continental water bodies and wetlands, including the salinisation and drying of lakes and inland seas (Anderson et al. 2003; Micklin 2010; Herbert et al. 2015). In the context of climate change, the degradation of peatland ecosystems is particularly relevant given their very high carbon storage and their sensitivity to changes in soils, hydrology and/or vegetation (Leifeld and Menichetti 2018). Drainage for land-use conversion together with peat mining are major drivers of peatland degradation, yet other factors such as the extractive use of their natural vegetation and the interactive effects of water table levels and fires (both sensitive to climate change) are important (Hergoualc'h et al. 2017a; Lilleskov et al. 2019).

The biotic components of the land can also be the focus of degradation processes. Vegetation clearing processes associated with land-use changes are not limited to deforestation but include other natural and seminatural ecosystems such as grasslands (the most cultivated biome on Earth), as well as dry steppes and shrublands, which give place to croplands, pastures, urbanisation or just barren land. This clearing process is associated with net carbon losses from the vegetation and soil pool. Not all biotic degradation processes involve biomass losses. Woody encroachment of open savannahs involves the expansion of woody plant cover and/or density over herbaceous areas and often limits the secondary productivity of rangelands (Asner et al. 2004; Anadon et al. 2014). These processes have accelerated since the mid-1800s over most continents (Van Auken 2009). Change in plant composition of natural or semi-natural ecosystems without any significant vegetation structural changes is another pathway of degradation affecting rangelands and forests. In rangelands, selective grazing and its interaction with climate variability and/or fire can push ecosystems to new compositions with lower forage value and a higher proportion of invasive species (Illius and O´Connor 1999; Sasaki et al. 2007), in some cases with higher carbon sequestration potential, yet with very complex interactions between vegetation and soil carbon shifts (Piñeiro et al. 2010). In forests, extractive logging can be a pervasive cause of degradation, leading to long-term impoverishment and, in extreme cases, a full loss of the forest cover through its interaction with other agents such as fires (Foley et al. 2007) or progressive intensification of land use. Invasive alien species are another source of biological degradation. Their arrival into cultivated systems is constantly reshaping crop production strategies, making agriculture unviable on occasions. In natural and seminatural systems such as rangelands, invasive plant species not only threaten livestock production through diminished forage quality, poisoning and other deleterious effects, but have cascading effects on other processes such as altered fire regimes and water cycling (Brooks et al. 2004). In forests, invasions affect primary productivity and nutrient availability, change fire regimes, and alter species composition, resulting in long-term impacts on carbon pools and fluxes (Peltzer et al. 2010).

Other biotic components of ecosystems have been shown as a focus of degradation processes. Invertebrate invasions in continental waters can exacerbate other degradation processes such as eutrophication, which is the over-enrichment of nutrients, leading to excessive algal growth (Walsh et al. 2016a). Shifts in soil microbial and mesofaunal composition – which can be caused by pollution with pesticides or nitrogen deposition and by vegetation or disturbance regime shifts – alter many soil functions, including respiration rates and carbon release to the atmosphere (Hussain et al. 2009; Crowther et al. 2015). The role of the soil biota in modulating the effects of climate change on soil carbon has been recently demonstrated (Ratcliffe et al. 2017), highlighting the importance of this lesser-known component of the biota as a focal point of land degradation. Of special relevance as both indicators and agents of land degradation recovery are mycorrhiza, which are root-associated fungal organisms (Asmelash et al. 2016; Vasconcellos et al. 2016). In natural dry ecosystems, biological soil crusts composed of a broad range of organisms, including mosses, are a particularly sensitive focus for degradation (Field et al. 2010) with evidenced sensitivity to climate change (Reed et al. 2012).

1.2.1.2 Land degradation processes and climate change

While the subdivision of individual processes is challenged by their strong interconnectedness, it provides a useful setting to identify the most important 'focal points' of climate change pressures on land degradation. Among land degradation processes, those responding more directly to climate change pressures include all types of erosion and SOM declines (soil focus), salinisation, sodification and permafrost thawing (soil/water focus), waterlogging of dry ecosystems and drying of wet ecosystems (water focus), and a broad group of biologically-mediated processes like woody encroachment, biological invasions, pest outbreaks (biotic focus), together with biological soil crust destruction and increased burning (soil/biota focus) (Table 4.1). Processes like ground subsidence can be affected by climate change indirectly through sea level rise (Keogh and Törnqvist 2019).

Even when climate change exerts a direct pressure on degradation processes, it can be a secondary driver subordinated to other overwhelming human pressures. Important exceptions are three processes in which climate change is a dominant global or regional pressure and the main driver of their current acceleration. These are: coastal erosion as affected by sea level rise and increased storm frequency/intensity (*high agreement*, *medium evidence*) (Johnson et al. 2015; Alongi 2015; Harley et al. 2017; Nicholls et al. 2016); permafrost thawing responding to warming (*high agreement*, *robust evidence*) (Liljedahl et al. 2016; Peng et al. 2016; Batir et al. 2017); and increased burning responding to warming and altered precipitation regimes (*high agreement*, *robust evidence*) (Jolly et al. 2015; Abatzoglou and Williams 2016; Taufik et al. 2017; Knorr et al. 2016). The previous assessment highlights the fact that climate change not only exacerbates many of the well-acknowledged ongoing land degradation processes of managed ecosystems (i.e., croplands and pastures), but becomes a dominant pressure that introduces novel degradation pathways in natural and seminatural ecosystems. Climate change has influenced species invasions and the degradation that they cause by enhancing the transport, colonisation, establishment, and ecological impact of the invasive species, and also by impairing their control practices (*medium agreement*, *medium evidence*) (Hellmann et al. 2008).

Table 4.1 | Major land degradation processes and their connections with climate change. For each process a 'focal point' (soil, water, biota) on which degradation occurs in the first place is indicated, acknowledging that most processes propagate to other land components and cascade into or interact with some of the other processes listed below. The impact of climate change on each process is categorised based on the proximity (very direct = high, very indirect = low) and dominance (dominant = high, subordinate to other pressures = low) of effects. The major effects of climate change on each process are highlighted together with the predominant pressures from other drivers. Feedbacks of land degradation processes on climate change are categorised according to the intensity (very intense = high, subtle = low) of the chemical (GHG emissions or capture) or physical (energy and momentum exchange, aerosol emissions) effects. Warming effects are indicated in red and cooling effects in blue. Specific feedbacks on climate change are highlighted.

Processes	Focal point	Impacts of climate change				Feedbacks on climate change			
		Proximity	Dominance	Climate change pressures	Other pressures	Intensity of chemical effects	Intensity of physical effects	Global extent	Specific impacts
Wind erosion	Soil	high	medium	Altered wind/drought patterns (*high confidence* on effect, *medium-low confidence* on trend) (1). Indirect effect through vegetation type and biomass production shifts	Tillage, leaving low cover, overgrazing, deforestation/vegetation clearing, large plot sizes, vegetation and fire regime shifts	low	medium	high	Radiative cooling by dust release (*medium confidence*). Ocean and land fertilisation and carbon burial (*medium confidence*). Albedo increase. Dust effect as condensation nuclei (19)
Water erosion	Soil	high	medium	Increasing rainfall intensity (*high confidence* on effect and trend) (2). Indirect effects on fire frequency/intensity, permafrost thawing, biomass production	Tillage, cultivation leaving low cover, overgrazing, deforestation/vegetation clearing, vegetation burning, poorly designed roads and paths	medium	medium	high	Net carbon release. Net release is probably less than site-specific loss due to deposition and burial (*high confidence*). Albedo increase (20)
Coastal erosion	Soil/water	high	high	Sea level rise, increasing intensity/frequency of storm surges (*high confidence* on effects and trends) (3)	Retention of sediments by upstream dams, coastal aquiculture, elimination of mangrove forests, subsidence	high	low	low	Release of old buried carbon pools (*medium confidence*) (21)

Processes	Focal point	Impacts of climate change				Feedbacks on climate change			
		Proximity	Dominance	Climate change pressures	Other pressures	Intensity of chemical effects	Intensity of physical effects	Global extent	Specific impacts
Subsidence	Soil/water	low	low	Indirect through increasing drought leading to higher ground water use. Indirect through enhanced decomposition (e.g., through drainage) in organic soils	Groundwater depletion/ overpumping, peatland drainage	low/high	low	low	Unimportant in the case of groundwater depletion. Very high net carbon release in the case of drained peatlands
Compaction/ hardening	Soil	low	low	Indirect through reduced organic matter content	Land-use conversion, machinery overuse, intensive grazing, poor tillage/grazing management (e.g., under wet or waterlogged conditions)	low	low	medium	Contradictory effects of reduced aeration on N_2O emissions
Nutrient depletion	Soil	low	low	Indirect (e.g., shifts in cropland distribution, BECCS)	Insufficient replenishment of harvested nutrients	low	low	medium	Net carbon release due shrinking SOC pools. Larger reliance on soil liming with associated CO_2 releases
Acidification/ overfertilisation	Soil	low	low	Indirect (e.g., shifts in cropland distribution, BECCS). Sulfidic wetland drying due to increased drought as special direct effect	High nitrogen fertilisation, high cation depletion, acid rain/deposition	medium	low	medium	N_2O release from overfertilised soils, increased by acidification. Inorganic carbon release from acidifying soils (*medium to high confidence*) (22)
Pollution	Soil/biota	low	low	Indirect (e.g., increased pest and weed incidence)	Intensifying chemical control of weed and pests	low	low	medium	Unknown, probably unimportant
Organic matter decline	Soil	high	medium	Warming accelerates soil respiration rates (*medium confidence* on effects and trends) (4). Indirect effects through changing quality of plant litter or fire/waterlogging regimes	Tillage. reduced plant input to soil. Drainage of waterlogged soils. Influenced by most of the other soil degradation processes.	high	low	high	Net carbon release (*high confidence*) (23)
Metal toxicity	Soil	low	low	Indirect	High cation depletion, fertilisation, mining activities	low	low	low	Unknown, probably unimportant
Salinisation	Soil/water	high	low	Sea level rise (*high confidence* on effects and trends) (5). Water balance shifts (*medium confidence* on effects and trends) (6). Indirect effects through irrigation expansion	Irrigation without good drainage infrastructure. Deforestation and water table-level rises under dryland agriculture	low	medium	medium	Reduced methane emissions with high sulfate load. Albedo increase
Sodification (increased sodium and associated physical degradation in soils)	Soil/water	high	low	Water balance shifts (*medium confidence* on effects and trends) (7). Indirect effects through irrigation expansion	Poor water management	low	medium	low	Net carbon release due to soil structure and organic matter dispersion. Albedo increase
Permafrost thawing	Soil/water	high	high	Warming (*very high confidence* on effects and trends) (8), seasonality shifts and accelerated snow melt leading to higher erosivity.		high	low	high	Net carbon release. CH_4 release (*high confidence*) (24)

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Processes	Focal point	Impacts of climate change				Feedbacks on climate change			
		Proximity	Dominance	Climate change pressures	Other pressures	Intensity of chemical effects	Intensity of physical effects	Global extent	Specific impacts
Waterlogging of dry systems	Water	high	medium	Water balance shifts (*medium confidence* on effects and trends) (9). Indirect effects through vegetation shifts	Deforestation. Irrigation without good drainage infrastructure	medium	medium	low	CH_4 release. Albedo decrease
Drying of continental waters/wetland/lowlands	Water	high	medium	Increasing extent and duration of drought (*high confidence* on effects, *medium confidence* on trends) (10). Indirect effects through vegetation shifts	Upstream surface and groundwater consumption. Intentional drainage. Trampling/overgrazing	medium	medium	medium	Net carbon release. N_2O release. Albedo increase
Flooding	Water	high	medium	Sea level rise, increasing intensity/frequency of storm surges, increasing rainfall intensity causing flash floods (*high confidence* on effects and trends) (11)	Land clearing. Increasing impervious surface. Transport infrastructure	medium	medium	low	CH_4 and N_2O release. Albedo decrease
Eutrophication of continental waters	Water/biota	low	low	Indirect through warming effects on nitrogen losses from the land or climate change effects on erosion rates. Interactive effects of warming and nutrient loads on algal blooms	Excess fertilisation. Erosion. Poor management of livestock/human sewage	medium	low	low	CH_4 and N_2O release
Woody encroachment	Biota	high	medium	Rainfall shifts (*medium confidence* on effects and trends), CO_2 rise (*medium confidence* on effects, *very high confidence* on trends) (12)	Overgrazing. Altered fire regimes, fire suppression. Invasive alien species	high	high	high	Net carbon storage. Albedo decrease
Species loss, compositional shifts	Biota	high	medium	Habitat loss as a result of climate shifts (*medium confidence* on effects and trends) (13)	Selective grazing and logging causing plant species loss, Pesticides causing soil microbial and soil faunal losses, large animal extinctions, interruption of disturbance regimes	low	low	medium	Unknown
Soil microbial and mesofaunal shifts	Biota	high	low	Habitat loss as a result of climate shifts (*medium confidence* on effects and trends) (14)	Altered fire regimes, nitrogen deposition, pesticide pollution, vegetation shifts, disturbance regime shifts	low	low	medium	Unknown
Biological soil crust destruction	Biota/soil	high	medium	Warming. Changing rainfall regimes. (*medium confidence* on effects, high confidence and trends). Indirect through fire regime shifts and/or invasions (15)	Overgrazing and trampling. Land-use conversion	low	high	high	Radiative cooling through albedo rise and dust release (*high confidence*) (25)
Invasions	Biota	high	medium	Habitat gain as a result of climate shifts (*medium confidence* on effects and trends) (16)	Intentional and unintentional species introductions	low	low	medium	Unknown
Pest outbreaks	Biota	high	medium	Habitat gain and accelerated reproduction as a result of climate shifts (*medium confidence* on effects and trends) (17)	Large-scale monocultures. Poor pest management practices	medium	low	medium	Net carbon release

Processes	Focal point	Impacts of climate change				Feedbacks on climate change			
		Proximity	Dominance	Climate change pressures	Other pressures	Intensity of chemical effects	Intensity of physical effects	Global extent	Specific impacts
Increased burning	Soil/biota	high	high	Warming, drought, shifting precipitation regimes, also wet spells rising fuel load (*high confidence* on effects and trends) (18)	Fire suppression policies increasing wildfire intensity. Increasing use of fire for rangeland management. Agriculture introducing fires in humid climates without previous fire history. Invasions	high	medium	medium	Net carbon release. CO, CH_4, N_2O release. Albedo increase (*high confidence*). Long-term decline of NPP in non-adapted ecosystems (26)

References in Table 4.1:

(1) Bärring et al. 2003; Munson et al. 2011; Sheffield et al. 2012, (2) Nearing et al. 2004; Shakesby 2011; Panthou et al. 2014, (3) Johnson et al. 2015; Alongi 2015; Harley et al. 2017, (4) Bond-Lamberty et al. 2018; Crowther et al. 2016; van Gestel et al. 2018, (5) Colombani et al. 2016, (6) Schofield and Kirkby 2003; Aragüés et al. 2015; Benini et al. 2016, (7) Jobbágy et al. 2017, (8) Liljedahl et al. 2016; Peng et al. 2016; Batir et al. 2017, (9) Piovano et al. 2004; Osland et al. 2016, (10) Burkett and Kusler 2000; Nielsen and Brock 2009; Johnson et al. 2015; Green et al. 2017, (11) Panthou et al. 2014; Arnell and Gosling 2016; Vitousek et al. 2017, (12) Van Auken 2009; Wigley et al. 2010, (13) Vincent et al. 2014; Gonzalez et al. 2010; Scheffers et al. 2016, (14) Pritchard 2011; Ratcliffe et al. 2017, (15) Reed et al. 2012; Maestre et al. 2013, (16) Hellmann et al. 2008; Hulme 2017, (17) Pureswaran et al. 2015; Cilas et al. 2016; Macfadyen et al. 2018, (18) Jolly et al. 2015; Abatzoglou and Williams 2016; Taufik et al. 2017; Knorr et al. 2016, (19) Davin et al. 2010; Pinty et al. 2011, (20) Wang et al. 2017b; Chappell et al. 2016, (21) Pendleton et al. 2012, (22) Oertel et al. 2016, (23) Houghton et al. 2012; Eglin et al. 2010, (24) Schuur et al. 2015; Christensen et al. 2004; Walter Anthony et al. 2016; Abbott et al. 2016, (25) Belnap, Walker, Munson & Gill, 2014; Rutherford et al. 2017, (26) Page et al. 2002; Pellegrini et al. 2018.

1.2.2 Drivers of land degradation

Drivers of land degradation and land improvement are many and they interact in multiple ways. Figure 4.2 illustrates how some of the most important drivers interact with the land users. It is important to keep in mind that natural and human factors can drive both degradation and improvement (Kiage 2013; Bisaro et al. 2014).

Land degradation is driven by the entire spectrum of factors, from very short and intensive events, such as individual rain storms of 10 minutes removing topsoil or initiating a gully or a landslide (Coppus and Imeson 2002; Morgan 2005b) to century-scale slow depletion of nutrients or loss of soil particles (Johnson and Lewis 2007, pp. 5–6). But, instead of focusing on absolute temporal variations, the drivers of land degradation can be assessed in relation to the rates of possible recovery. Unfortunately, this is impractical to do in a spatially explicit way because rates of soil formation are difficult to measure due to the slow rate, usually <5mm/century (Delgado and Gómez 2016). Studies suggest that erosion rates of conventionally tilled agricultural fields exceed the rate at which soil is generated by one to two orders of magnitude (Montgomery 2007a).

The landscape effects of gully erosion from one short intensive rainstorm can persist for decades and centuries (Showers 2005). Intensive agriculture under the Roman Empire in occupied territories in France is still leaving its marks and can be considered an example of irreversible land degradation (Dupouey et al. 2002).

The climate-change-related drivers of land degradation are gradual changes of temperature, precipitation and wind, as well as changes of the distribution and intensity of extreme events (Lin et al. 2017). Importantly, these drivers can act in two directions: land improvement and land degradation. Increasing CO_2 level in the atmosphere is a driver of land improvement, even if the net effect is modulated by other factors, such as the availability of nitrogen (Terrer et al. 2016) and water (Gerten et al. 2014; Settele et al. 2015; Girardin et al. 2016).

The gradual and planetary changes that can cause land degradation/improvement have been studied by global integrated models and Earth observation technologies. Studies of global land suitability for agriculture suggest that climate change will increase the area suitable for agriculture by 2100 in the Northern high latitudes by 16% (Ramankutty et al. 2002) or 5.6 million km^2 (Zabel et al. 2014), while tropical regions will experience a loss (Ramankutty et al. 2002; Zabel et al. 2014).

Temporal and spatial patterns of tree mortality can be used as an indicator of climate change impacts on terrestrial ecosystems. Episodic mortality of trees occurs naturally even without climate change, but more widespread spatio-temporal anomalies can be a sign of climate-induced degradation (Allen et al. 2010). In the absence of systematic data on tree mortality, a comprehensive meta-analysis of 150 published articles suggests that increasing tree mortality around the world can be attributed to increasing drought and heat stress in forests worldwide (Allen et al. 2010).

Other and more indirect drivers can be a wide range of factors such as demographic changes, technological change, changes of consumption patterns and dietary preferences, political and economic changes, and social changes (Mirzabaev et al. 2016). It is important to stress that there are no simple or direct relationships between underlying drivers and land degradation, such as poverty or high population density, that are necessarily causing land degradation (Lambin et al. 2001). However, drivers of land degradation need to be studied in the context

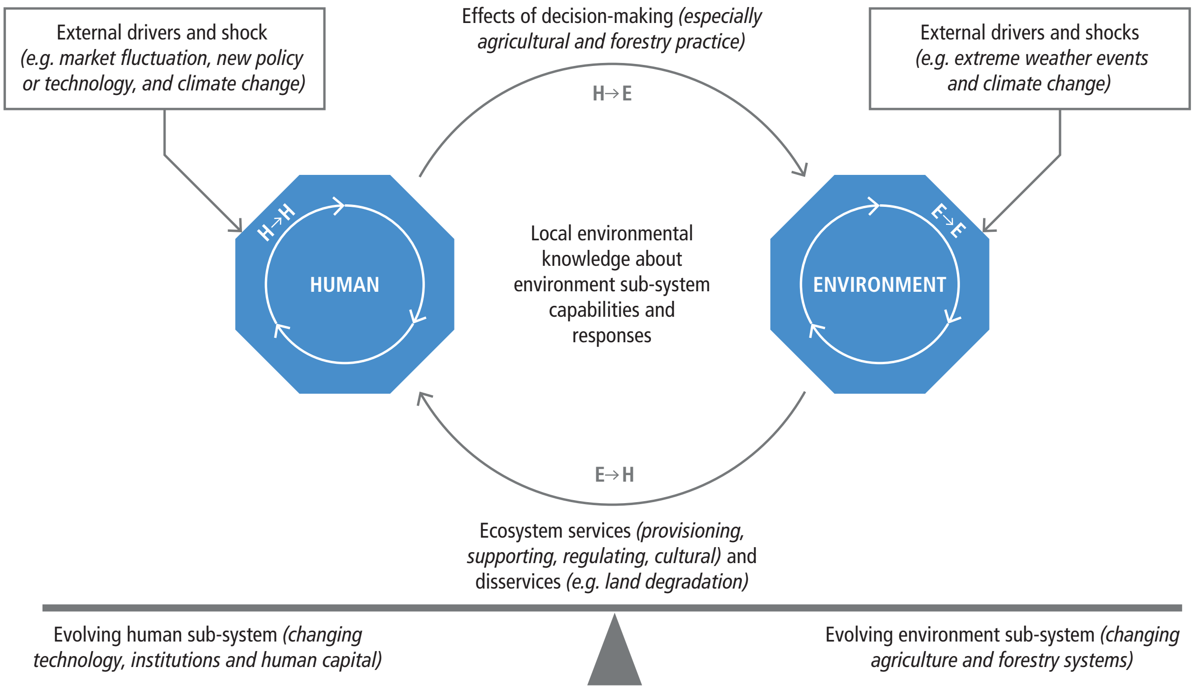

Figure 4.2 | Schematic representation of the interactions between the human (H) and environmental (E) components of the land system showing decision-making and ecosystem services as the key linkages between the components (moderated by an effective system of local and scientific knowledge), and indicating how the rates of change and the way these linkages operate must be kept broadly in balance for functional coevolution of the components. Modified with permission from Stafford Smith et al. (2007).

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of spatial, temporal, economic, environmental and cultural aspects (Warren 2002). Some analyses suggest an overall negative correlation between population density and land degradation (Bai et al. 2008) but we find many local examples of both positive and negative relationships (Brandt et al. 2018a, 2017). Even if there are correlations in one or the other direction, causality is not always the same.

Land degradation is inextricably linked to several climate variables, such as temperature, precipitation, wind, and seasonality. This means that there are many ways in which climate change and land degradation are linked. The linkages are better described as a web of causality rather than a set of cause–effect relationships.

1.2.3 Attribution in the case of land degradation

The question here is whether or not climate change can be attributed to land degradation and vice versa. Land degradation is a complex phenomenon often affected by multiple factors such as climatic (rainfall, temperature, and wind), abiotic ecological factors (e.g., soil characteristics and topography), type of land use (e.g., farming of various kinds, forestry, or protected area), and land management practices (e.g., tilling, crop rotation, and logging/thinning). Therefore, attribution of land degradation to climate change is extremely challenging. Because land degradation is highly dependent on land management, it is even possible that climate impacts would trigger land management changes reducing or reversing land degradation, sometimes called transformational adaptation (Kates et al. 2012). There is not much research on attributing land degradation explicitly to climate change, but there is more on climate change as a threat multiplier for land degradation. However, in some cases, it is possible to infer climate change impacts on land degradation, both theoretically and empirically. Section 4.2.3.1 outlines the potential direct linkages of climate change on land degradation based on current theoretical understanding of land degradation processes and drivers. Section 4.2.3.2 investigates possible indirect impacts on land degradation.

1.2.3.1 Direct linkages with climate change

The most important direct impacts of climate change on land degradation are the results of increasing temperatures, changing rainfall patterns, and intensification of rainfall. These changes will, in various combinations, cause changes in erosion rates and the processes driving both increases and decreases of soil erosion. From an attribution point of view, it is important to note that projections of precipitation are, in general, more uncertain than projections of temperature changes (Murphy et al. 2004; Fischer and Knutti 2015; IPCC 2013a). Precipitation involves local processes of larger complexity than temperature, and projections are usually less robust than those for temperature (Giorgi and Lionello 2008; Pendergrass 2018).

Theoretically the intensification of the hydrological cycle as a result of human-induced climate change is well established (Guerreiro et al. 2018; Trenberth 1999; Pendergrass et al. 2017; Pendergrass and Knutti 2018) and also empirically observed (Blenkinsop et al. 2018; Burt et al. 2016a; Liu et al. 2009; Bindoff et al. 2013). AR5 WGI concluded that heavy precipitation events have increased in frequency, intensity, and/or amount since 1950 (*likely*) and that further changes in this direction are *likely* to *very likely* during the 21st century (IPCC 2013). The IPCC Special Report on 1.5°C concluded that human-induced global warming has already caused an increase in the frequency, intensity and/or amount of heavy precipitation events at the global scale (Hoegh-Guldberg et al. 2018). As an example, in central India, there has been a threefold increase in widespread extreme rain events during 1950–2015 which has influenced several land degradation processes, not least soil erosion (Burt et al. 2016b). In Europe and North America, where observation networks are dense and extend over a long time, it is *likely* that the frequency or intensity of heavy rainfall have increased (IPCC 2013b). It is also expected that seasonal shifts and cycles such as monsoons and El Niño–Southern Oscillation (ENSO) will further increase the intensity of rainfall events (IPCC 2013).

When rainfall regimes change, it is expected to drive changes in vegetation cover and composition, which may be a cause of land degradation in and of itself, as well as impacting on other aspects of land degradation. Vegetation cover, for example, is a key factor in determining soil loss through water (Nearing et al. 2005) and wind erosion (Shao 2008). Changing rainfall regimes also affect below-ground biological processes, such as fungi and bacteria (Meisner et al. 2018; Shuab et al. 2017; Asmelash et al. 2016).

Changing snow accumulation and snow melt alter volume and timing of hydrological flows in and from mountain areas (Brahney et al. 2017; Lutz et al. 2014), with potentially large impacts on downstream areas. Soil processes are also affected by changing snow conditions with partitioning between evaporation and streamflow and between subsurface flow and surface runoff (Barnhart et al. 2016). Rainfall intensity is a key climatic driver of soil erosion. Early modelling studies and theory suggest that light rainfall events will decrease while heavy rainfall events increase at about 7% per degree of warming (Liu et al. 2009; Trenberth 2011). Such changes result in increased intensity of rainfall, which increases the erosive power of rainfall (erosivity) and hence enhances the likelihood of water erosion. Increases in rainfall intensity can even exceed the rate of increase of atmospheric moisture content (Liu et al. 2009; Trenberth 2011). Erosivity is highly correlated to the product of total rainstorm energy and the maximum 30-minute rainfall intensity of the storm (Nearing et al. 2004) and increased erosivity will exacerbate water erosion substantially (Nearing et al. 2004). However, the effects will not be uniform, but highly variable across regions (Almagro et al. 2017; Mondal et al. 2016). Several empirical studies around the world have shown the increasing intensity of rainfall (IPCC 2013b; Ma et al. 2015, 2017) and also suggest that this will be accentuated with future increased global warming (Cheng and AghaKouchak 2015; Burt et al. 2016b; O'Gorman 2015).

The very comprehensive database of direct measurements of water erosion presented by García-Ruiz et al. (2015) contains 4377 entries (North America: 2776, Europe: 847, Asia: 259, Latin America: 237, Africa: 189, Australia and Pacific: 67), even though not all entries are complete (Figure 4.3).

An important finding from that database is that almost any erosion rate is possible under almost any climatic condition (García-Ruiz et al. 2015). Even if the results show few clear relationships between erosion and land conditions, the authors highlighted four observations (i) the highest erosion rates were found in relation to agricultural activities – even though moderate erosion rates were also found in agricultural settings, (ii) high erosion rates after forest fires were not observed (although the cases were few), (iii) land covered by shrubs showed generally low erosion rates, (iv) pasture land showed generally medium rates of erosion. Some important

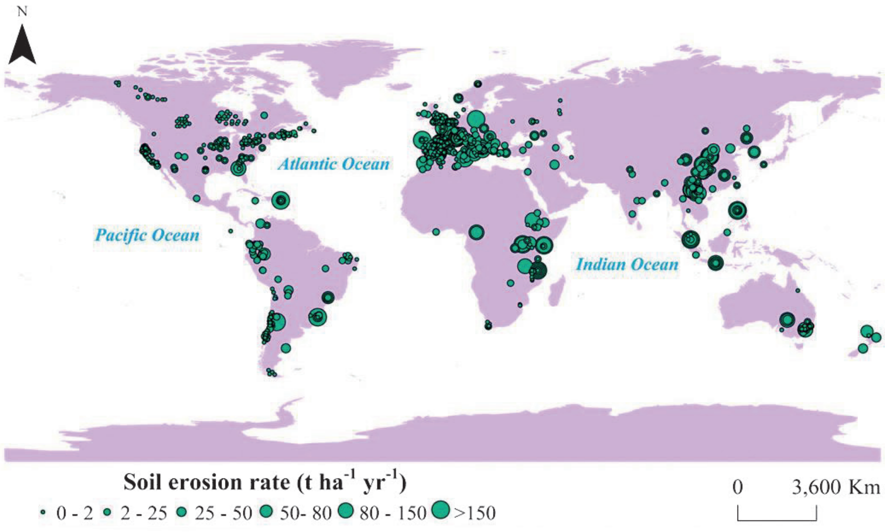

Figure 4.3.| Map of observed soil erosion rates in database of 4,377 entries by García-Ruiz et al. (2015). The map was published by Li and Fang (2016).

findings for the link between soil erosion and climate change can be noted from erosion measurements: erosion rates tend to increase with increasing mean annual rainfall, with a peak in the interval of 1000 to 1400 mm annual rainfall (García-Ruiz et al. 2015) (*low confidence*). However, such relationships are overshadowed by the fact that most rainfall events do not cause any erosion, instead erosion is caused by a few high-intensity rainfall events (Fischer et al. 2016; Zhu et al. 2019). Hence, mean annual rainfall is not a good predictor of erosion (Gonzalez-Hidalgo et al. 2012, 2009). In the context of climate change, it means that the tendency for rainfall patterns to change towards more intensive precipitation events is serious. Such patterns have already been observed widely, even in cases where the total rainfall is decreasing (Trenberth 2011). The findings generally confirm the strong consensus about the importance of vegetation cover as a protection against soil erosion, emphasising how extremely important land management is for controlling erosion.

In the Mediterranean region, the observed and expected decrease in annual rainfall due to climate change is accompanied by an increase of rainfall intensity, and hence erosivity (Capolongo et al. 2008). In tropical and sub-tropical regions, the on-site impacts of soil erosion dominate, and are manifested in very high rates of soil loss, in some cases exceeding 100 t ha^{-1} yr^{-1} (Tadesse 2001; García-Ruiz et al. 2015). In temperate regions, the off-site costs of soil erosion are often a greater concern, for example, siltation of dams and ponds, downslope damage to property, roads and other infrastructure (Boardman 2010). In cases where water erosion occurs, the downstream effects, such as siltation of dams, are often significant and severe in terms of environmental and economic damages (Kidane and Alemu 2015; Reinwarth et al. 2019; Quiñonero-Rubio et al. 2016; Adeogun et al. 2018; Ben Slimane et al. 2016).

The distribution of wet and dry spells also affects land degradation, although uncertainties remain depending on resolution of climate models used for prediction (Kendon et al. 2014). Changes in timing of rainfall events may have significant impacts on processes of soil erosion through changes in wetting and drying of soils (Lado et al. 2004).

Soil moisture content is affected by changes in evapotranspiration and evaporation, which may influence the partitioning of water into surface and subsurface runoff (Li and Fang 2016; Nearing et al. 2004). This portioning of rainfall can have a decisive effect on erosion (Stocking et al. 2001).

Wind erosion is a serious problem in agricultural regions, not only in drylands (Wagner 2013). Near-surface wind speeds over land areas have decreased in recent decades (McVicar and Roderick 2010), partly as a result of changing surface roughness (Vautard et al. 2010). Theoretically (Bakun 1990; Bakun et al. 2015) and empirically (Sydeman et al. 2014; England et al. 2014) average winds along coastal regions worldwide have increased with climate change (*medium evidence*, *high agreement*). Other studies of wind and wind erosion have not detected any long-term trend, suggesting that climate change has altered wind patterns outside drylands in a way that can significantly affect the risk of wind erosion (Pryor and Barthelmie 2010; Bärring et al. 2003). Therefore, the findings regarding wind erosion and climate change are inconclusive, partly due to inadequate measurements.

Global mean temperatures are rising worldwide, but particularly in the Arctic region (*high confidence*) (IPCC 2018a). Heat stress from extreme temperatures and heatwaves (multiple days of hot weather in a row) have increased markedly in some locations in the last three decades (*high confidence*), and are *virtually certain* to continue during the 21st century (Olsson et al. 2014a). The IPCC Special Report on Global Warming of 1.5°C concluded that human-induced global warming has already caused more frequent heatwaves in most of land regions, and that climate models project robust differences between present-day and global warming up to 1.5°C and between 1.5°C and 2°C (Hoegh-Guldberg et al. 2018). Direct temperature effects on soils are of two kinds. Firstly, permafrost thawing leads to soil degradation in boreal and high-altitude regions (Yang et al. 2010; Jorgenson and Osterkamp 2005). Secondly, warming alters the cycling of nitrogen and carbon in soils, partly due to impacts on soil microbiota (Solly et al. 2017). There are many studies with particularly strong experimental evidence, but a full understanding of cause and effect is contextual and elusive (Conant et al. 2011a,b; Wu et al. 2011). This is discussed comprehensively in Chapter 2.

Climate change, including increasing atmospheric CO_2 levels, affects vegetation structure and function and hence conditions for land degradation. Exactly how vegetation responds to changes remains a research task. In a comparison of seven global vegetation models under four representative concentration pathways, Friend et al. (2014) found that all models predicted increasing vegetation carbon storage, however, with substantial variation between models. An important insight compared with previous understanding is that structural dynamics of vegetation seems to play a more important role for carbon storage than vegetation production (Friend et al. 2014). The magnitude of CO_2 fertilisation of vegetation growth, and hence conditions for land degradation, is still uncertain (Holtum and Winter 2010), particularly in tropical rainforests (Yang et al. 2016). For more discussion on this topic, see Chapter 2 in this report.

In summary, rainfall changes attributed to human-induced climate change have already intensified drivers of land degradation (*robust evidence*, *high agreement*) but attributing land degradation to climate change is challenging because of the importance of land management (*medium evidence*, *high agreement*). Changes in climate variability modes, such as in monsoons and El Niño–Southern Oscillation (ENSO) events, can also affect land degradation (*low evidence*, *low agreement*).

1.2.3.2 Indirect and complex linkages with climate change

Many important indirect linkages between land degradation and climate change occur via agriculture, particularly through changing outbreaks of pests (Rosenzweig et al. 2001; Porter et al. 1991; Thomson et al. 2010; Dhanush et al. 2015; Lamichhane et al. 2015), which is covered comprehensively in Chapter 5. More negative impacts have been observed than positive ones (IPCC 2014b). After 2050, the risk of yield loss increases as a result of climate change in combination with other drivers (*medium confidence*) and such risks will increase dramatically if global mean temperatures increase by about 4°C (*high confidence*) (Porter et al. 2014). The reduction (or plateauing) in yields in major production areas (Brisson et al. 2010; Lin and Huybers 2012; Grassini et al. 2013) may trigger cropland expansion elsewhere, either into natural ecosystems, marginal arable lands or intensification on already cultivated lands, with possible consequences for increasing land degradation.

Precipitation and temperature changes will trigger changes in land and crop management, such as changes in planting and harvest dates, type of crops, and type of cultivars, which may alter the conditions for soil erosion (Li and Fang 2016).

Much research has tried to understand how plants are affected by a particular stressor, for example, drought, heat, or waterlogging, including effects on below-ground processes. But less research has tried to understand how plants are affected by several simultaneous stressors – which of course is more realistic in the context of climate change (Mittler 2006; Kerns et al. 2016) and from a hazards point of view (Section 7.2.1). From an attribution point of view, such a complex web of causality is problematic if attribution is only done through statistically-significant correlation. It requires a combination of statistical links and theoretically informed causation, preferably integrated into a model. Some modelling studies have combined several stressors with geomorphologically explicit mechanisms – using the Water Erosion Prediction Project (WEPP) model – and realistic land-use scenarios, and found severe risks of increasing erosion from climate change (Mullan et al. 2012; Mullan 2013). Other studies have included various management options, such as changing planting and harvest dates (Zhang and Nearing 2005; Parajuli et al. 2016; Routschek et al. 2014; Nunes and Nearing 2011), type of cultivars (Garbrecht and Zhang 2015), and price of crops (Garbrecht et al. 2007; O'Neal et al. 2005) to investigate the complexity of how new climate regimes may alter soil erosion rates.

In summary, climate change increases the risk of land degradation, both in terms of likelihood and consequence, but the exact attribution to climate change is challenging due to several confounding factors. But since climate change exacerbates most degradation processes, it is clear that, unless land management is improved, climate change will result in increasing land degradation (*very high confidence*).

1.2.4 Approaches to assessing land degradation

In a review of different approaches and attempts to map global land degradation, Gibbs and Salmon (2015) identified four main approaches to map the global extent of degraded lands: expert opinions (Oldeman and van Lynden 1998; Dregne 1998; Reed 2005; Bot et al. 2000); satellite observation of vegetation greenness – for example, remote sensing of Normalized Difference Vegetation Index (NDVI), Enhanced Vegetation Index (EVI), Plant Phenology Index (PPI) – (Yengoh et al. 2015; Bai et al. 2008c; Shi et al. 2017; Abdi et al. 2019; JRC 2018); biophysical models (biogeographical/topological) (Cai et al. 2011b; Hickler et al. 2005; Steinkamp and Hickler 2015; Stoorvogel et al. 2017); and inventories of land use/condition. Together they provide a relatively complete evaluation, but none on its own assesses the complexity of the process (Vogt et al. 2011; Gibbs and Salmon 2015). There is, however, a robust consensus that remote sensing and field-based methods are critical to assess and monitor land degradation, particularly over large areas (such as global, continental and sub-continental) although there are still knowledge gaps to be filled (Wessels et al. 2007, 2004; Prince 2016; Ghazoul and Chazdon 2017) as well as the problem of baseline values (Section 4.1.3).

Remote sensing can provide meaningful proxies of land degradation in terms of severity, temporal development, and areal extent. These proxies of land degradation include several indexes that have been used to assess land conditions, and monitoring changes of land conditions – for example, extent of gullies, severe forms of rill and sheet erosion, and deflation. The presence of open-access, quality controlled and continuously updated global databases of remote sensing data is invaluable, and is the only method for consistent monitoring of large areas over several decades (Sedano et al. 2016; Brandt et al. 2018b; Turner 2014). The NDVI, as a proxy for Net Primary Production (NPP) (see Glossary), is one of the most commonly used methods to assess land degradation, since it indicates land cover, an important factor for soil protection. Although NDVI is not a direct measure of vegetation biomass, there is a close coupling between NDVI integrated over a season and in situ NPP (*high agreement*, *robust evidence*) (see Higginbottom et al. 2014; Andela et al. 2013; Wessels et al. 2012).

Distinction between land degradation/improvement and the effects of climate variation is an important and contentious issue (Murthy and Bagchi 2018; Ferner et al. 2018). There is no simple and straightforward way to disentangle these two effects. The interaction of different determinants of primary production is not well understood. A key barrier to this is a lack of understanding of the inherent interannual variability of vegetation (Huxman et al. 2004; Knapp and Smith 2001; Ruppert et al. 2012; Bai et al. 2008a; Jobbágy and Sala 2000). One possibility is to compare potential land productivity modelled by vegetation models and actual productivity measured by remote sensing (Seaquist et al. 2009; Hickler et al. 2005; van der Esch et al. 2017), but the difference in spatial resolution, typically 0.5 degrees for vegetation models compared to 0.25–0.5 km for remote sensing data, is hampering the approach. The Moderate Resolution Imaging Spectroradiometer (MODIS) provides higher spatial resolution (up to 0.25 km), delivers data for the EVI, which is calculated in the same way as NDVI, and has showed a robust approach to estimate spatial patterns of global annual primary productivity (Shi et al. 2017; Testa et al. 2018).

Another approach to disentangle the effects of climate and land use/management is to use the Rain Use Efficiency (RUE), defined as the biomass production per unit of rainfall, as an indicator (Le Houerou 1984; Prince et al. 1998; Fensholt et al. 2015). A variant of the RUE approach is the residual trend (RESTREND) of a NDVI time series, defined as the fraction of the difference between the observed NDVI and the NDVI predicted from climate data (Yengoh et al. 2015; John et al. 2016). These two metrics aim to estimate the NPP, rainfall and the time dimensions. They are simple transformations of the same three variables: RUE shows the NPP relationship with rainfall for individual years, while RESTREND is the interannual change of RUE; also, both consider that rainfall is the only variable that affects biomass production. They are legitimate metrics when used appropriately, but in many cases they involve oversimplifications and yield misleading results (Fensholt et al. 2015; Prince et al. 1998).

Furthermore, increases in NPP do not always indicate improvement in land condition/reversal of land degradation, since this does not account for changes in vegetation composition. It could, for example, result from conversion of native forest to plantation, or due to bush encroachment, which many consider to be a form of land degradation (Ward 2005). Also, NPP may be increased by irrigation, which can enhance productivity in the short to medium term while increasing risk of soil salinisation in the long term (Niedertscheider et al. 2016).

Recent progress and expanding time series of canopy characterisations based on passive microwave satellite sensors have offered rapid progress in regional and global descriptions of forest degradation and recovery trends (Tian et al. 2017). The most common proxy is vertical optical depth (VOD) and has already been used to describe global forest/savannah carbon stock shifts over two decades, highlighting strong continental contrasts (Liu et al. 2015a) and demonstrating the value of this approach to monitor forest degradation at large scales. Contrasting with NDVI, which is only sensitive to vegetation 'greenness', from which primary production can be modelled, VOD is also sensitive to water in woody parts of the vegetation and hence provides a view of vegetation dynamics that can be complementary to NDVI. As well as the NDVI, VOD also needs to be corrected to take into account the rainfall variation (Andela et al. 2013).

Even though remote sensing offers much potential, its application to land degradation and recovery remains challenging as structural changes often occur at scales below the detection capabilities of most remote-sensing technologies. Additionally, if the remote sensing is based on vegetation index data, other forms of land degradation, such as nutrient depletion, changes of soil physical or biological properties, loss of values for humans, among others, cannot be inferred directly by remote sensing. The combination of remotely sensed images and field-based approach can give improved estimates of carbon stocks and tree biodiversity (Imai et al. 2012; Fujiki et al. 2016).

Additionally, the majority of trend techniques employed would be capable of detecting only the most severe of degradation processes, and would therefore not be useful as a degradation early-warning system (Higginbottom et al. 2014; Wessels et al. 2012). However, additional analyses using higher-resolution imagery, such as the Landsat and SPOT satellites, would be well suited to providing further localised information on trends observed (Higginbottom et al. 2014). New approaches to assess land degradation using high spatial resolution are developing, but the need for time series makes progress slow. The use of synthetic aperture radar (SAR) data has been shown to be advantageous for the estimation of soil surface characteristics, in particular, surface roughness and soil moisture (Gao et al. 2017; Bousbih et al. 2017), and detecting and quantifying selective logging (Lei et al. 2018). Continued research effort is required to enable full assessment of land degradation using remote sensing.

Computer simulation models can be used alone or combined with the remote sensing observations to assess land degradation. The Revised Universal Soil Loss Equation (RUSLE) can be used, to some extent, to predict the long-term average annual soil loss by water erosion. RUSLE has been constantly revisited to estimate soil loss based on the product of rainfall–runoff erosivity, soil erodibility, slope length and steepness factor, conservation factor, and support practice parameter (Nampak et al. 2018). Inherent limitations of RUSLE include data-sparse regions, inability to account for soil loss from gully erosion or mass wasting events, and that it does not predict sediment pathways from hillslopes to water bodies (Benavidez et al. 2018). Since RUSLE models only provide gross erosion, the integration of a further module in the RUSLE scheme to estimate the sediment yield from the modelled hillslopes is needed. The spatially distributed sediment delivery model, WaTEM/SEDEM, has been widely tested in Europe (Borrelli et al. 2018). Wind erosion is another factor that needs to be taken into account in the modelling of soil erosion (Webb et al. 2017a, 2016). Additional models need to be developed to include the limitations of the RUSLE models.

Regarding the field-based approach to assess land degradation, there are multiple indicators that reflect functional ecosystem processes linked to ecosystem services and thus to the value for humans. These indicators are a composite set of measurable attributes from different factors, such as climate, soil, vegetation, biomass, management, among others, that can be used together or separately to develop indexes to better assess land degradation (Allen et al. 2011; Kosmas et al. 2014).

Declines in vegetation cover, changes in vegetation structure, decline in mean species abundances, decline in habitat diversity, changes in abundance of specific indicator species, reduced vegetation health and productivity, and vegetation management intensity and use, are the most common indicators in the vegetation condition of forest and woodlands (Stocking et al. 2001; Wiesmair et al. 2017; Ghazoul and Chazdon 2017; Alkemade et al. 2009).

Several indicators of the soil quality (SOM, depth, structure, compaction, texture, pH, C:N ratio, aggregate size distribution and stability, microbial respiration, soil organic carbon, salinisation, among others) have been proposed (Schoenholtz et al. 2000) (Section 2.2). Among these, SOM directly and indirectly drives the majority of soil functions. Decreases in SOM can lead to a decrease in fertility and biodiversity, as well as a loss of soil structure, causing reductions in water-holding capacity, increased risk of erosion (both wind and water) and increased bulk density and hence soil compaction (Allen et al. 2011; Certini 2005; Conant et al. 2011a). Thus, indicators related with the quantity and quality of the SOM are necessary to identify land degradation (Pulido et al. 2017; Dumanski and Pieri 2000). The composition of the microbial community is *very likely* to be positive impacted by both climate change and land degradation processes (Evans and Wallenstein 2014; Wu et al. 2015; Classen et al. 2015), thus changes in microbial community composition can be very useful to rapidly reflect land degradation (e.g., forest degradation increased the bacterial alpha-diversity indexes) (Flores-Rentería et al. 2016; Zhou et al. 2018). These indicators might be used as a set of site-dependent indicators, and in a plant-soil system (Ehrenfeld et al. 2005).

Useful indicators of degradation and improvement include changes in ecological processes and disturbance regimes that regulate the flow of energy and materials and that control ecosystem dynamics under a climate change scenario. Proxies of dynamics include spatial and temporal turnover of species and habitats within ecosystems (Ghazoul et al. 2015; Bahamondez and Thompson 2016). Indicators in agricultural lands include crop yield decreases and difficulty in maintaining yields (Stocking et al. 2001). Indicators of landscape degradation/improvement in fragmented forest landscapes include the extent, size and distribution of remaining forest fragments, an increase in edge habitat, and loss of connectivity and ecological memory (Zahawi et al. 2015; Pardini et al. 2010).

In summary, as land degradation is such a complex and global process, there is no single method by which land degradation can be estimated objectively and consistently over large areas (*very high confidence*). However, many approaches exist that can be used to assess different aspects of land degradation or provide proxies of land degradation. Remote sensing, complemented by other kinds of data (i.e., field observations, inventories, expert opinions), is the only method that can generate geographically explicit and globally consistent data over time scales relevant for land degradation (several decades).

1.3 Status and current trends of land degradation

The scientific literature on land degradation often excludes forest degradation, yet here we attempt to assess both issues. Because of the different bodies of scientific literature, we assess land degradation and forest degradation under different sub-headings and, where possible, draw integrated conclusions.

1.3.1 Land degradation

There are no reliable global maps of the extent and severity of land degradation (Gibbs and Salmon 2015; Prince et al. 2018; van der Esch et al. 2017), despite the fact that land degradation is a severe problem (Turner et al. 2016). The reasons are both conceptual – that is, how land degradation is defined, using what baseline (Herrick et al. 2019) or over what time period – and methodological – that is, how it can be measured (Prince et al. 2018). Although there is a strong consensus that land degradation is a reduction in productivity of the land or soil, there are diverging views regarding the spatial and temporal scales at which land degradation occurs (Warren 2002), and how this can be quantified and mapped. Proceeding from the definition in this report, there are also diverging views concerning ecological integrity and the value to humans. A comprehensive treatment of the conceptual discussion about land degradation is provided by the recent report on land degradation from the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) (Montanarella et al. 2018).

A review of different attempts to map global land degradation, based on expert opinion, satellite observations, biophysical models and a database of abandoned agricultural lands, suggested that between <10 Mkm^2 to 60 Mkm^2 (corresponding to 8–45% of the ice-free land area) have been degraded globally (Gibbs and Salmon, 2015) (*very low confidence*).

One often-used global assessment of land degradation uses trends in NDVI as a proxy for land degradation and improvement during the period 1983 to 2006 (Bai et al. 2008b,c) with an update to 2011 (Bai et al. 2015). These studies, based on very coarse resolution satellite data (NOAA AVHRR data with a resolution of 8 km), indicated that, between 22% and 24% of the global ice-free land area was subject to a downward trend, while about 16% showed an increasing trend. The study also suggested, contrary to earlier assessments (Middleton and Thomas 1997), that drylands were not among the most affected regions. Another study using a similar approach for the period 1981–2006 suggested that about 29% of the global land area is subject to 'land degradation hotspots', that is, areas with acute land degradation in need of particular attention. These hotspot areas were distributed over all agro-ecological regions and land cover types. Two different studies have tried to link land degradation, identified by NDVI as a proxy, and number of people affected: Le et al. (2016) estimated that at least 3.2 billion people were affected, while Barbier and Hochard (2016, 2018) estimated that 1.33 billion people were affected, of which 95% were living in developing countries.

Yet another study, using a similar approach and type of remote-sensing data, compared NDVI trends with biomass trends calculated by a global vegetation model over the period 1982–2010 and found that 17–36% of the land areas showed a negative NDVI trend, while a positive or neutral trend was predicted in modelled vegetation (Schut et al. 2015). The World Atlas of Desertification (3rd edition) includes a global map of land productivity change over the period 1999 to 2013, which is one useful proxy for land degradation (Cherlet et al. 2018). Over that period, about 20% of the global ice-free land area shows signs of declining or unstable productivity, whereas about 20% shows increasing productivity. The same report also summarised the productivity trends by land categories and found that most forest land showed increasing trends in productivity, while rangelands had more declining trends than increasing trends (Figure 4.4). These productivity assessments, however, do not distinguish between trends due to climate change and trends due to other factors. A recent analysis of 'greening' of the world using MODIS time series of NDVI 2000–2017, shows a striking increase in the greening over China and India. In China the greening is seen over forested areas, 42%, and cropland areas, in which 32% is increasing (Section 4.9.3). In India, the greening is almost entirely associated with cropland (82%) (Chen et al. 2019).

All these studies of vegetation trends show that there are regionally differentiated trends of either decreasing or increasing vegetation. When comparing vegetation trends with trends in climatic variables, Schut et al. (2015) found very few areas (1–2%) where an increase in vegetation trend was independent of the climate drivers, and that study suggested that positive vegetation trends are primarily caused by climatic factors.

In an attempt to go beyond the mapping of global vegetation trends for assessing land degradation, Borelli et al. (2017) used a soil erosion model (RUSLE) and suggested that soil erosion is mainly caused in areas of cropland expansion, particularly in Sub-Saharan Africa, South America and Southeast Asia. The method is controversial for conceptual reasons (i.e., the ability of the model to capture the most important erosion processes) and data limitations (i.e., the availability of relevant data at regional to global scales), and its validity for assessing erosion over large areas has been questioned by several studies (Baveye 2017; Evans and Boardman 2016a,b; Labrière et al. 2015).

An alternative to using remote sensing for assessing the state of land degradation is to compile field-based data from around the globe (Turner et al. 2016). In addition to the problems of definitions and baselines, this approach is also hampered by the lack of standardised methods used in the field. An assessment of the global severity of soil erosion in agriculture, based on 1673 measurements around the world (compiled from 201 peer-reviewed articles), indicated that the global net median rate of soil formation (i.e., formation minus erosion) is about 0.004 mm yr^{-1} (about 0.05 t ha^{-1} yr^{-1}) compared with the median net rate of soil loss in agricultural fields, 1.52 mm yr^{-1} (about 18 t ha^{-1} yr^{-1}) in tilled fields and 0.065 mm yr^{-1} (about 0.8 t ha^{-1} yr^{-1}) in no-till fields (Montgomery 2007a). This means that the rate of soil erosion from agricultural fields is between 380 (conventional tilling) and 16 times (no-till) the natural rate of soil formation (*medium agreement, limited evidence*). These approximate figures are supported by another large meta-study including over 4000 sites around the world (see Figure 4.4) where the average soil loss from agricultural plots was about 21 t ha^{-1} yr^{-1} (García-Ruiz et al. 2015). Climate change, mainly through the intensification of rainfall, will further increase these rates unless land management is improved (*high agreement*, *medium evidence*).

Soils contain about 1500 Gt of organic carbon (median across 28 different estimates presented by Scharlemann et al. (2014)), which is about 1.8 times more carbon than in the atmosphere (Ciais et al. 2013) and 2.3–3.3 times more than what is held in the terrestrial vegetation of the world (Ciais et al. 2013). Hence, land degradation, including land conversion leading to soil carbon losses, has the potential to impact on the atmospheric concentration of CO_2 substantially. When natural ecosystems are cultivated they lose soil carbon that accumulated over long time periods. The loss rate depends on the type of natural vegetation and how the soil is managed. Estimates of the magnitude of loss vary but figures between 20% and 59% have been reported in several meta studies (Poeplau and Don 2015; Wei et al. 2015; Li et al. 2012; Murty et al. 2002; Guo and Gifford 2002). The amount of soil carbon lost explicitly due to land degradation after conversion is hard to assess due to large variation in local conditions and management, see also Chapter 2.

From a climate change perspective, land degradation plays an important role in the dynamics of nitrous oxide (N_2O) and methane (CH_4). N_2O is produced by microbial activity in the soil and the dynamics are related to both management practices and weather

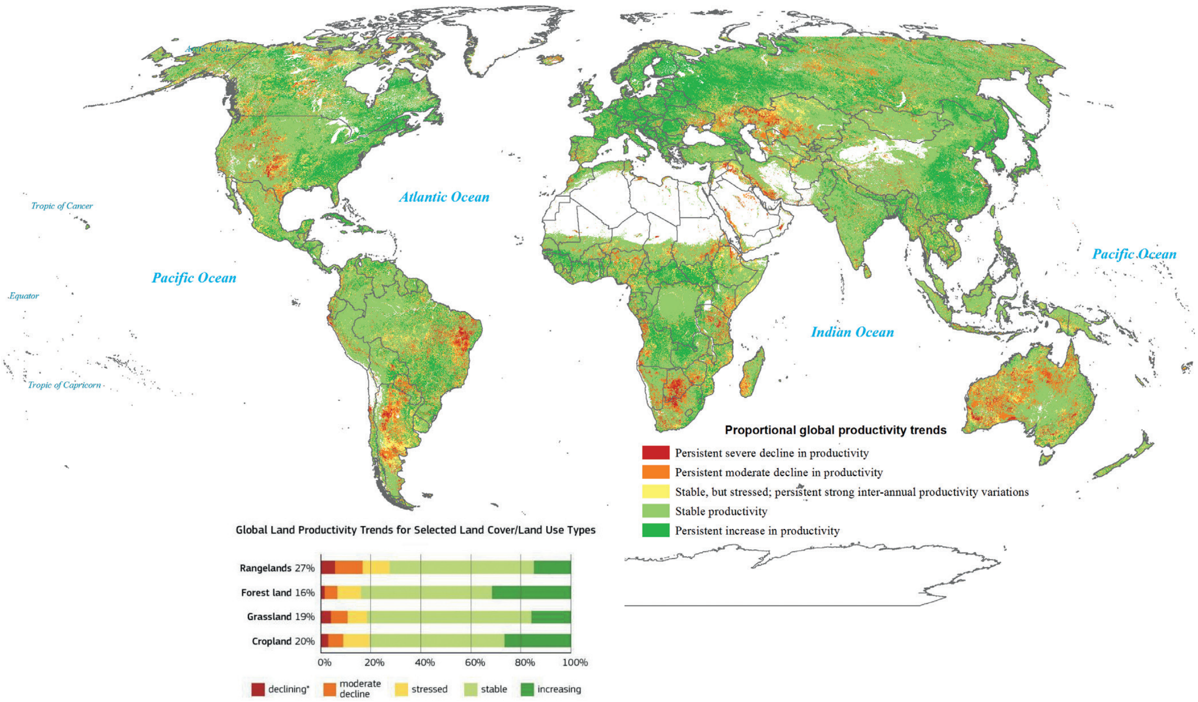

Figure 4.4 | Proportional global land productivity trends by land-cover/land-use class. (Cropland includes arable land, permanent crops and mixed classes with over 50% crops; grassland includes natural grassland and managed pasture land; rangelands include shrubland, herbaceous and sparsely vegetated areas; forest land includes all forest categories and mixed classes with tree cover greater than 40%.) Data source: Copernicus Global Land SPOT VGT, 1999–2013, adapted from (Cherlet et al. 2018).

conditions, while CH_4 dynamics are primarily determined by the amount of soil carbon and to what extent the soil is subject to waterlogging (Palm et al. 2014), see also Chapter 2.

Several attempts have been made to map the human footprint on the planet (Čuček et al. 2012; Venter et al. 2016) but, in some cases, they confuse human impact on the planet with degradation. From our definition it is clear that human impact (or pressure) is not synonymous with degradation, but information on the human footprint provides a useful mapping of potential non-climatic drivers of degradation.

In summary, there are no uncontested maps of the location, extent and severity of land degradation. Proxy estimates based on remote sensing of vegetation dynamics provide one important information source, but attribution of the observed changes in productivity to climate change, human activities, or other drivers is hard. Nevertheless, the different attempts to map the extent of global land degradation using remotely sensed proxies show some convergence and suggest that about a quarter of the ice-free land area is subject to some form of land degradation (*limited evidence*, *medium agreement*) affecting about 3.2 billion people (*low confidence*). Attempts to estimate the severity of land degradation through soil erosion estimates suggest that soil erosion is a serious form of land degradation in croplands closely associated with unsustainable land management in combination with climatic parameters, some of which are subject to climate change (*limited evidence*, *high agreement*). Climate change is one among several causal factors in the status and current trends of land degradation (*limited evidence*, *high agreement*).

1.3.2 Forest degradation

Quantifying degradation in forests has also proven difficult. Remote sensing based inventory methods can measure reductions in canopy cover or carbon stocks more easiliy than reductions in biological productivity, losses of ecological integrity or value to humans. However, the causes of reductions in canopy cover or carbon stocks can be many (Curtis et al. 2018), including natural disturbances (e.g., fires, insects and other forest pests), direct human activities (e.g., harvest, forest management) and indirect human impacts (such as climate change) and these may not reduce long-term biological productivity. In many boreal, some temperate and other forest types natural disturbances are common, and consequently these disturbance-adapted forest types are comprised of a mosaic of stands of different ages and stages of stand recovery following natural disturbances. In those managed forests where natural disturbances are uncommon or suppressed, harvesting is the primary determinant of forest age-class distributions.

Quantifying forest degradation as a reduction in productivity, carbon stocks or canopy cover also requires that an initial condition (or baseline) is established, against which this reduction is assessed (Section 4.1.4). In forest types with rare stand-replacing disturbances, the concept of 'intact' or 'primary' forest has been used to define the initial condition (Potapov et al. 2008) but applying a single metric can be problematic (Bernier et al. 2017). Moreover, forest types with frequent stand-replacing disturbances, such as wildfires, or with natural disturbances that reduce carbon stocks, such as some insect outbreaks, experience over time a natural variability of carbon stocks or canopy density, making it more difficult to define the appropriate baseline carbon density or canopy cover against which to assess degradation. In these systems, forest degradation cannot be assessed at the stand level, but requires a landscape-level assessment that takes into consideration the stand age-class distribution of the landscape, which reflects natural and human disturbance regimes over past decades to centuries and also considers post-disturbance regrowth (van Wagner 1978; Volkova et al. 2018; Lorimer and White 2003).

The lack of a consistent definition of forest degradation also affects the ability to establish estimates of the rates or impacts of forest degradation because the drivers of degradation are not clearly defined (Sasaki and Putz 2009). Moreover, the literature at times confounds estimates of forest degradation and deforestation (i.e., the conversion of forest to non-forest land uses). Deforestation is a change in land use, while forest degradation is not, although severe forest degradation can ultimately lead to deforestation.

Based on empirical data provided by 46 countries, the drivers for deforestation (due to commercial agriculture) and forest degradation (due to timber extraction and logging) are similar in Africa, Asia and Latin America (Hosonuma et al. 2012). More recently, global forest disturbance over the period 2001–2015 was attributed to commodity-driven deforestation (27 ± 5%), forestry (26 ± 4%), shifting agriculture (24 ± 3%) and wildfire (23 ± 4%). The remaining 0.6 ± 0.3% was attributed to the expansion of urban centres (Curtis et al. 2018).

The trends of productivity shown by several remote-sensing studies (see previous section) are largely consistent with mapping of forest cover and change using a 34-year time series of coarse resolution satellite data (NOAA AVHRR) (Song et al. 2018). This study, based on a thematic classification of satellite data, suggests that (i) global tree canopy cover increased by 2.24 million km^2 between 1982 and 2016 (corresponding to +7.1%) but with regional differences that contribute a net loss in the tropics and a net gain at higher latitudes, and (ii) the fraction of bare ground decreased by 1.16 million km^2 (corresponding to –3.1%), mainly in agricultural regions of Asia (Song et al. 2018), see Figure 4.5. Other tree or land cover datasets show opposite global net trends (Li et al. 2018b), but high agreement in terms of net losses in the tropics and large net gains in the temperate and boreal zones (Li et al. 2018b; Song et al. 2018; Hansen et al. 2013). Differences across global estimates are further discussed in Chapter 1 (Section 1.1.2.3) and Chapter 2.

The changes detected from 1982 to 2016 were primarily linked to direct human action, such as land-use changes (about 60% of the observed changes), but also to indirect effects, such as human-induced climate change (about 40% of the observed changes) (Song et al. 2018), a finding also supported by a more recent study (Chen et al. 2019). The climate-induced effects were clearly discernible in some regions, such as forest decline in the US Northwest due to increasing pest infestation and increasing fire frequency (Lesk et al. 2017; Abatzoglou and Williams 2016; Seidl et al. 2017), warming-induced

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vegetation increase in the Arctic region, general greening in the Sahel probably as a result of increasing rainfall and atmospheric CO_2, and advancing treelines in mountain regions (Song et al. 2018).

Keenan et al. (2015) and Sloan and Sayer (2015) studied the 2015 Forest Resources Assessment (FRA) of the Food and Agriculture Organization of the United Nations (FAO) (FAO 2016) and found that the total forest area from 1990 to 2015 declined by 3%, an estimate that is supported by a global remote-sensing assessment of forest area change that found a 2.8% decline between 1990–2010 (D'Annunzio et al. 2017; Lindquist and D'Annunzio 2016). The trend in deforestation is, however, contradicted between these two global assessments, with FAO (2016) suggesting that deforestation is slowing down, while the remote sensing assessments finds it to be accelerating (D'Annunzio et al. 2017). Recent estimates (Song et al. 2018) owing to semantic and methodological differences (see Chapter 1, Section 1.1.2.3) suggest that global tree cover has increased over the period 1982–2016, which contradicts the forest area dynamics assessed by FAO (2016) and Lindquist and D'Annunzio (2016). The loss rate in tropical forest areas from 2010 to 2015 is 55,000 $km^2\ yr^{-1}$. According to the FRA, the global natural forest area also declined from 39.61 Mkm^2 to 37.21 Mkm^2 during the period 1990 to 2015 (Keenan et al. 2015).

Since 1850, deforestation globally contributed 77% of the emissions from land-use and land-cover change while degradation contributed 10% (with the remainder originating from non-forest land uses) (Houghton and Nassikas 2018). That study also showed large temporal and regional differences with northern mid-latitude forests currently contributing to carbon sinks due to increasing forest area and forest management. However, the contribution to carbon emissions of degradation as percentage of total forest emissions (degradation and deforestation) are uncertain, with estimates varying from 25% (Pearson et al. 2017) to nearly 70% of carbon losses (Baccini et al. 2017). The 25% estimate refers to an analysis of 74 developing countries within tropical and subtropical regions covering 22 million km^2 for the period 2005–2010, while the 70% estimate refers to an analysis of the tropics for the period 2003–2014, but, by and large, the scope of these studies is the same. Pearson et al. (2017) estimated annual gross emissions of 2.1 $GtCO_2$, of which 53% were derived from timber harvest, 30% from woodfuel harvest and 17% from forest fire. Estimating gross emissions only, creates a distorted representation of human impacts on the land sector carbon cycle. While forest harvest for timber and fuelwood and land-use change (deforestation) contribute to gross emissions, to quantify impacts on the atmosphere, it is necessary to estimate net emissions, that is, the balance of gross emissions and gross removals of carbon from the atmosphere through forest regrowth (Chazdon et al. 2016a; Poorter et al. 2016; Sanquetta et al. 2018).

Current efforts to reduce atmospheric CO_2 concentrations can be supported by reductions in forest-related carbon emissions and increases in sinks, which requires that the net impact of forest management on the atmosphere be evaluated (Griscom et al. 2017). Forest management and the use of wood products in GHG mitigation strategies result in changes in forest ecosystem carbon stocks, changes in harvested wood product carbon stocks, and potential changes in emissions resulting from the use of wood products and forest biomass that substitute for other emissions-intensive materials such as concrete, steel and fossil fuels (Kurz et al. 2016; Lemprière et al. 2013; Nabuurs et al. 2007). The net impact of these changes on GHG emissions and removals, relative to a scenario without forest mitigation actions, needs to be quantified, (e.g., Werner et al. 2010; Smyth et al. 2014; Xu et al. 2018). Therefore, reductions in forest ecosystem carbon stocks alone are an incomplete estimator of the impacts of forest management on the atmosphere (Nabuurs et al. 2007; Lemprière et al. 2013; Kurz et al. 2016; Chen et al. 2018b). The impacts of forest management and the carbon storage in long-lived products and landfills vary greatly by region, however, because of the typically much shorter lifespan of wood products produced from tropical regions compared to temperate and boreal regions (Earles et al. 2012; Lewis et al. 2019; Iordan et al. 2018) (Section 4.8.4).

Assessments of forest degradation based on remote sensing of changes in canopy density or land cover, (e.g., Hansen et al. 2013; Pearson et al. 2017) quantify changes in above-ground biomass carbon stocks and require additional assumptions or model-based analyses to also quantify the impacts on other ecosystem carbon pools including below-ground biomass, litter, woody debris and soil carbon. Depending on the type of disturbance, changes in above-ground biomass may lead to decreases or increases in other carbon pools, for example, windthrow and insect-induced tree mortality may result in losses in above-ground biomass that are (initially) offset by corresponding increases in dead organic matter carbon pools (Yamanoi et al. 2015; Kurz et al. 2008), while deforestation will reduce the total ecosystem carbon pool (Houghton et al. 2012).

A global study of current vegetation carbon stocks (450 Gt C), relative to a hypothetical condition without land use (916 Gt C), attributed 42–47% of carbon stock reductions to land management effects without land-use change, while the remaining 53–58% of carbon stock reductions were attributed to deforestation and other land-use changes (Erb et al. 2018). While carbon stocks in European forests are lower than hypothetical values in the complete absence of human land use, forest area and carbon stocks have been increasing over recent decades (McGrath et al. 2015; Kauppi et al. 2018). Studies by Gingrich et al. (2015) on the long-term trends in land use over nine European countries (Albania, Austria, Denmark, Germany, Italy, the Netherlands, Romania, Sweden and the United Kingdom) also show an increase in forest land and reduction in cropland and grazing land from the 19th century to the early 20th century. However, the extent to which human activities have affected the productive capacity of forest lands is poorly understood. Biomass Production Efficiency (BPE), i.e. the fraction of photosynthetic production used for biomass production, was significantly higher in managed forests (0.53) compared to natural forests (0.41) (and it was also higher in managed (0.63) compared to natural (0.44) grasslands) (Campioli et al. 2015). Managing lands for production may involve trade-offs. For example, a larger proportion of NPP in managed forests is allocated to biomass carbon storage, but lower allocation to fine roots is hypothesised to reduce soil carbon stocks in the long term (Noormets et al. 2015). Annual volume increment in Finnish forests has more than doubled over the last century, due to increased growing stock, improved forest management and environmental changes (Henttonen et al. 2017).

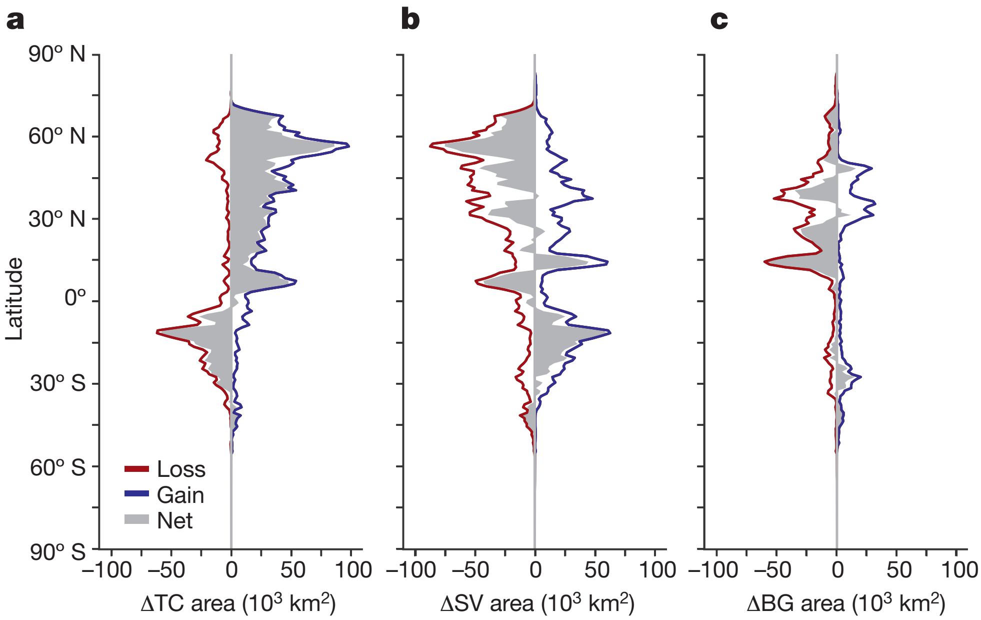

Figure 4.5 | Diagrams showing latitudinal profiles of land cover change over the period 1982 to 2016 based on analysis of time-series of NOAA AVHRR imagery: a) tree canopy cover change (ΔTC); **b)** short vegetation cover change (ΔSV); **c)** bare ground cover change (ΔBG). Area statistics were calculated for every 1° of latitude (Song et al. 2018). Source of data: NOAA AVHRR.

As economies evolve, the patterns of land-use and carbon stock changes associated with human expansion into forested areas often include a period of rapid decline of forest area and carbon stocks, recognition of the need for forest conservation and rehabilitation, and a transition to more sustainable land management that is often associated with increasing carbon stocks, (e.g., Birdsey et al. 2006). Developed and developing countries around the world are in various stages of forest transition (Kauppi et al. 2018; Meyfroidt and Lambin 2011). Thus, opportunities exist for SFM to contribute to atmospheric carbon targets through reduction of deforestation and degradation, forest conservation, forest restoration, intensification of management, and enhancements of carbon stocks in forests and harvested wood products (Griscom et al. 2017) (*medium evidence*, *medium agreement*).

1.4 Projections of land degradation in a changing climate

Land degradation will be affected by climate change in both direct and indirect ways, and land degradation will, to some extent, also feed back into the climate system. The direct impacts are those in which climate and land interact directly in time and space. Examples of direct impacts are when increasing rainfall intensity exacerbates soil erosion, or when prolonged droughts reduce the vegetation cover of the soil, making it more prone to erosion and nutrient depletion. The indirect impacts are those where climate change impacts and land degradation are separated in time and/or space. Examples of such impacts are when declining agricultural productivity due to climate change drives an intensification of agriculture elsewhere, which may cause land degradation. Land degradation, if sufficiently widespread, may also feed back into the climate system by reinforcing ongoing climate change.

Although climate change is exacerbating many land degradation processes (*high to very high confidence*), prediction of future land degradation is challenging because land management practices determine, to a very large extent, the state of the land. Scenarios of climate change in combination with land degradation models can provide useful knowledge on what kind and extent of land management will be necessary to avoid, reduce and reverse land degradation.

1.4.1 Direct impacts on land degradation

There are two main levels of uncertainty in assessing the risks of future climate-change-induced land degradation. The first level, where uncertainties are comparatively low, involves changes of the degrading agent, such as erosive power of precipitation, heat stress from increasing temperature extremes (Hüve et al. 2011), water stress from droughts, and high surface wind speed. The second level of uncertainties, and where the uncertainties are much larger, relates to the above – and below-ground ecological changes as a result of changes in climate, such as rainfall, temperature, and increasing level of CO_2. Vegetation cover is crucial to protect against erosion (Mullan et al. 2012; García-Ruiz et al. 2015).

Changes in rainfall patterns, such as distribution in time and space, and intensification of rainfall events will increase the risk of land degradation, both in terms of likelihood and consequences (*high agreement*, *medium evidence*). Climate-induced vegetation changes will increase the risk of land degradation in some areas (where vegetation cover will decline) (*medium confidence*). Landslides are a form of land degradation, induced by extreme rainfall events. There is a strong theoretical reason for increasing landslide activity due to intensification of rainfall, but so far, the empirical evidence that climate change has contributed to landslides is lacking (Crozier 2010; Huggel et al. 2012; Gariano and Guzzetti 2016). Human disturbance may be a more important future trigger than climate change (Froude and Petley 2018).

Erosion of coastal areas as a result of sea level rise will increase worldwide (*very high confidence*). In cyclone-prone areas (such as the Caribbean, Southeast Asia, and the Bay of Bengal) the combination of sea level rise and more intense cyclones (Walsh et al. 2016b) and, in some areas, land subsidence (Yang et al. 2019; Shirzaei and Bürgmann 2018; Wang et al. 2018; Fuangswasdi et al. 2019; Keogh and Törnqvist 2019), will pose a serious risk to people and livelihoods (*very high confidence*), in some cases even exceeding limits to adaption (Sections 4.8.4.1, 4.9.6 and 4.9.8).

1.4.1.1 Changes in water erosion risk due to precipitation changes

The hydrological cycle is intensifying with increasing warming of the atmosphere. The intensification means that the number of heavy rainfall events is increasing, while the total number of rainfall events tends to decrease (Trenberth 2011; Li and Fang 2016; Kendon et al. 2014; Guerreiro et al. 2018; Burt et al. 2016a; Westra et al. 2014; Pendergrass and Knutti 2018) (*robust evidence*, *high agreement*). Modelling of the changes in land degradation that are a result of climate change alone is hard because of the importance of local contextual factors. As shown above, actual erosion rate is extremely dependent on local conditions, primarily vegetation cover and topography (García-Ruiz et al. 2015). Nevertheless, modelling of soil erosion risks has advanced substantially in recent decades, and such studies are indicative of future changes in the risk of soil erosion, while actual erosion rates will still primarily be determined by land management. In a review article, Li and Fang (2016) summarised 205 representative modelling studies around the world where erosion models were used in combination with downscaled climate models to assess future (between 2030 to 2100) erosion rates. The meta-study by Li and Fang, where possible, considered climate change in terms of temperature increase and changing rainfall regimes and their impacts on vegetation and soils. Almost all of the sites had current soil loss rates above 1 t ha^{-1} (assumed to be the upper limit for acceptable soil erosion in Europe) and 136 out of 205 studies predicted increased soil erosion rates. The percentage increase in erosion rates varied between 1.2% to as much as over 1600%, whereas 49 out of 205 studies projected more than 50% increase. Projected soil erosion rates varied substantially between studies because the important of local factors, hence climate change impacts on soil erosion, should preferably be assessed at the local to regional scale, rather than the global (Li and Fang 2016).

Mesoscale convective systems (MCS), typically thunder storms, have increased markedly in the last three to four decades in the USA and Australia and they are projected to increase substantially (Prein et al. 2017). Using a climate model with the ability to represent MCS, Prein and colleagues were able to predict future increases in frequency, intensity and size of such weather systems. Findings include the 30% decrease in number of MCS of <40 mm h^{-1}, but a sharp increase of 380% in the number of extreme precipitation events of >90 mm h^{-1} over the North American continent. The combined effect of increasing precipitation intensity and increasing size of the weather systems implies that the total amount of precipitation from these weather systems is expected to increase by up to 80% (Prein et al. 2017), which will substantially increase the risk of land degradation in terms of landslides, extreme erosion events, flashfloods, and so on.

The potential impacts of climate change on soil erosion can be assessed by modelling the projected changes in particular variables of climate change known to cause erosion, such as erosivity of rainfall. A study of the conterminous United States based on three climate models and three scenarios (A2, A1B, and B1) found that rainfall erosivity will increase in all scenarios, even if there are large spatial differences – a strong increase in the north-east and north-west, and either weak or inconsistent trends in the south-west and mid-west (Segura et al. 2014).

In a study of how climate change will impact on future soil erosion processes in the Himalayas, Gupta and Kumar (2017) estimated that soil erosion will increase by about 27% in the near term (2020s) and 22% in the medium term (2080s), with little difference between scenarios. A study from Northern Thailand estimated that erosivity will increase by 5% in the near term (2020s) and 14% in the medium term (2080s), which would result in a similar increase of soil erosion, all other factors being constant (Plangoen and Babel 2014). Observed rainfall erosivity has increased significantly in the lower Niger Basin (Nigeria) and is predicted to increase further based on statistical downscaling of four General Circulation Models (GCM) scenarios, with an estimated increase of 14%, 19% and 24% for the 2030s, 2050s, and 2070s respectively (Amanambu et al. 2019).

Many studies from around the world where statistical downscaling of GCM results have been used in combination with process-based erosion models show a consistent trend of increasing soil erosion.

Using a comparative approach, Serpa et al. (2015) studied two Mediterranean catchments (one dry and one humid) using a spatially explicit hydrological model – soil and water assessment tool (SWAT) – in combination with land-use and climate scenarios for 2071–2100. Climate change projections showed, on the one hand, decreased rainfall and streamflow for both catchments, whereas sediment export decreased only for the humid catchment; projected land-use change, from traditional to more profitable, on the other hand, resulted in increase in streamflow. The combined effect of climate and land-use change resulted in reduced sediment export for the humid catchment (–29% for A1B; –22% for B1) and increased sediment export for the dry catchment (+222% for A1B; +5% for B1). Similar methods have been used elsewhere, also showing the dominant effect of land-use/land cover for runoff and soil erosion (Neupane and Kumar 2015).

A study of future erosion rates in Northern Ireland, using a spatially explicit erosion model in combination with downscaled climate projections (with and without sub-daily rainfall intensity changes), showed that erosion rates without land management changes would decrease by the 2020s, 2050s and 2100s, irrespective of changes in intensity, mainly as a result of a general decline in rainfall (Mullan et al. 2012). When land management scenarios were added to the modelling, the erosion rates started to vary dramatically for all three time periods, ranging from a decrease of 100% for no-till land use, to an increase of 3621% for row crops under annual tillage and sub-days intensity changes (Mullan et al. 2012). Again, it shows how crucial land management is for addressing soil erosion, and the important role of rainfall intensity changes.

There is a large body of literature based on modelling future land degradation due to soil erosion concluding that, in spite of the increasing trend of erosive power of rainfall, (*medium evidence*, *high agreement*) land degradation is primarily determined by land management (*very high confidence*).

1.4.1.2 Climate-induced vegetation changes, implications for land degradation

The spatial mosaic of vegetation is determined by three factors: the ability of species to reach a particular location, how species tolerate the environmental conditions at that location (e.g., temperature, precipitation, wind, the topographic and soil conditions), and the interaction between species (including above/below ground species (Settele et al. 2015). Climate change is projected to alter the conditions and hence impact on the spatial mosaic of vegetation, which can be considered a form of land degradation. Warren et al. (2018) estimated that only about 33% of globally important biodiversity conservation areas will remain intact if global mean temperature increases to 4.5°C, while twice that area (67%) will remain intact if warming is restricted to 2°C. According to AR5, the clearest link between climate change and ecosystem change is when temperature is the primary driver, with changes of Arctic tundra as a response to significant warming as the best example (Settele et al. 2015). Even though distinguishing climate-induced changes from land-use changes is challenging, Boit et al. (2016) suggest that 5–6% of biomes in South America will undergo biome shifts until 2100, regardless of scenario, attributed to climate change. The projected biome shifts are primarily forests shifting to shrubland and dry forests becoming fragmented and isolated from more humid forests (Boit et al. 2016). Boreal forests are subject to unprecedented warming in terms of speed and amplitude (IPCC 2013b), with significant impacts on their regional distribution (Juday et al. 2015). Globally, tree lines are generally expanding northward and to higher elevations, or remaining stable, while a reduction in tree lines was rarely observed, and only where disturbances occurred (Harsch et al. 2009). There is *limited evidence* of a slow northward migration of the boreal forest in eastern North America (Gamache and Payette 2005). The thawing of permafrost may increase drought-induced tree mortality throughout the circumboreal zone (Gauthier et al. 2015).

Forests are a prime regulator of hydrological cycling, both fluxes of atmospheric moisture and precipitation, hence climate and forests are inextricably linked (Ellison et al. 2017; Keys et al. 2017). Forest management influences the storage and flow of water in forested watersheds. In particular, harvesting, forest thinning and the construction of roads increase the likelihood of floods as an outcome of extreme climate events (Eisenbies et al. 2007). Water balance of at least partly forested landscapes is, to a large extent, controlled by forest ecosystems (Sheil and Murdiyarso 2009; Pokam et al. 2014). This includes surface runoff, as determined by evaporation and transpiration and soil conditions, and water flow routing (Eisenbies et al. 2007). Water-use efficiency (i.e., the ratio of water loss to biomass gain) is increasing with increased CO_2 levels (Keenan et al. 2013), hence transpiration is predicted to decrease which, in turn, will increase surface runoff (Schlesinger and Jasechko 2014). However, the interaction of several processes makes predictions challenging (Frank et al. 2015; Trahan and Schubert 2016). Surface runoff is an important agent in soil erosion.

Generally, removal of trees through harvesting or forest death (Anderegg et al. 2012) will reduce transpiration and hence increase the runoff during the growing season. Management-induced soil disturbance (such as skid trails and roads) will affect water flow routing to rivers and streams (Zhang et al. 2017; Luo et al. 2018; Eisenbies et al. 2007).

Climate change affects forests in both positive and negative ways (Trumbore et al. 2015; Price et al. 2013) and there will be regional and temporal differences in vegetation responses (Hember et al. 2017; Midgley and Bond 2015). Several climate-change-related drivers interact in complex ways, such as warming, changes in precipitation and water balance, CO_2 fertilisation, and nutrient cycling, which makes projections of future net impacts challenging (Kurz et al. 2013; Price et al. 2013) (Section 2.3.1.2). In high latitudes, a warmer climate will extend the growing seasons. However, this could be constrained by summer drought (Holmberg et al. 2019), while increasing levels of atmospheric CO_2 will increase water-use efficiency but not necessarily tree growth (Giguère-Croteau et al. 2019). Improving one growth-limiting factor will only enhance tree growth if other factors are not limiting (Norby et al. 2010; Trahan and Schubert 2016; Xie et al. 2016; Frank et al. 2015). Increasing forest productivity has been observed in most of Fennoscandia (Kauppi et al. 2014; Henttonen et al. 2017), Siberia and the northern reaches of North America as a response to a warming trend (Gauthier et al. 2015) but increased warming may also decrease forest productivity and increase risk of tree mortality and natural disturbances (Price et al. 2013; Girardin et al. 2016; Beck et al. 2011; Hember et al. 2016; Allen et al. 2011). The climatic conditions in high latitudes are changing at a magnitude faster than the ability of forests to adapt with detrimental, yet unpredictable, consequences (Gauthier et al. 2015).

Negative impacts dominate, however, and have already been documented (Lewis et al. 2004; Bonan et al. 2008; Beck et al. 2011) and are predicted to increase (Miles et al. 2004; Allen et al. 2010; Gauthier et al. 2015; Girardin et al. 2016; Trumbore et al. 2015). Several authors have emphasised a concern that tree mortality (forest dieback) will increase due to climate-induced physiological stress as well as interactions between physiological stress and other stressors, such as insect pests, diseases, and wildfires (Anderegg et al. 2012; Sturrock et al. 2011; Bentz et al. 2010; McDowell et al. 2011). Extreme events such as extreme heat and drought, storms, and floods also pose increased threats to forests in both high – and low-latitude forests (Lindner et al. 2010; Mokria et al. 2015). However, comparing observed forest dieback with modelled climate-induced damages did not show a general link between climate change and forest dieback (Steinkamp and Hickler 2015). Forests are subject to increasing frequency and intensity of wildfires which is projected to increase substantially with continued climate change (Price et al. 2013) (Cross-Chapter Box 3 in Chapter 2, and Chapter 2). In the tropics, interaction between climate change, CO_2 and fire could lead to abrupt shifts between woodland – and grassland-dominated states in the future (Shanahan et al. 2016).

Within the tropics, much research has been devoted to understanding how climate change may alter regional suitability of various crops. For example, coffee is expected to be highly sensitive to both temperature and precipitation changes, both in terms of growth and yield, and in terms of increasing problems of pests (Ovalle-Rivera et al. 2015). Some studies conclude that the global area of coffee production will decrease by 50% (Bunn et al. 2015). Due to increased heat stress, the suitability of Arabica coffee is expected to deteriorate in Mesoamerica, while it can improve in high-altitude areas in South America. The general pattern is that the climatic suitability for Arabica coffee will deteriorate at low altitudes of the tropics as well as at the higher latitudes (Ovalle-Rivera et al. 2015). This means that climate change in and of itself can render unsustainable previously sustainable land-use and land management practices, and vice versa (Laderach et al. 2011).

Rangelands are projected to change in complex ways due to climate change. Increasing levels of atmospheric CO_2 directly stimulate plant growth and can potentially compensate for negative effects from drying by increasing rain-use efficiency. But the positive effect of increasing CO_2 will be mediated by other environmental conditions, primarily water availability, but also nutrient cycling, fire regimes and invasive species. Studies over the North American rangelands suggest, for example, that warmer and dryer climatic conditions will reduce NPP in the southern Great Plains, the Southwest, and northern Mexico, but warmer and wetter conditions will increase NPP in the northern Plains and southern Canada (Polley et al. 2013).

1.4.1.3 Coastal erosion

Coastal erosion is expected to increase dramatically by sea level rise and, in some areas, in combination with increasing intensity of cyclones (highlighted in Section 4.9.6) and cyclone-induced coastal erosion. Coastal regions are also characterised by high population density, particularly in Asia (Bangladesh, China, India, Indonesia, Vietnam), whereas the highest population increase in coastal regions is projected in Africa (East Africa, Egypt, and West Africa) (Neumann et al. 2015). For coastal regions worldwide, and particularly in developing countries with high population density in low-lying coastal areas, limiting the warming to 1.5°C to 2.0°C will have major socio-economic benefits compared with higher temperature scenarios (IPCC 2018a; Nicholls et al. 2018). For more in-depth discussions on coastal process, please refer to Chapter 4 of the IPCC Special Report on the Ocean and Cryosphere in a Changing Climate (IPCC SROCC).

Despite the uncertainty related to the responses of the large ice sheets of Greenland and west Antarctica, climate-change-induced sea level rise is largely accepted and represents one of the biggest threats faced by coastal communities and ecosystems (Nicholls et al. 2011; Cazenave and Cozannet 2014; DeConto and Pollard 2016; Mengel et al. 2016). With significant socio-economic effects, the physical impacts of projected sea level rise, notably coastal erosion, have received considerable scientific attention (Nicholls et al. 2011; Rahmstorf 2010; Hauer et al. 2016).

Rates of coastal erosion or recession will increase due to rising sea levels and, in some regions, also in combination with increasing oceans waves (Day and Hodges 2018; Thomson and Rogers 2014; McInnes et al. 2011; Mori et al. 2010), lack or absence of sea-ice (Savard et al. 2009; Thomson and Rogers 2014) thawing of permafrost (Hoegh-Guldberg et al. 2018), and changing cyclone paths (Tamarin-Brodsky and Kaspi 2017; Lin and Emanuel 2016a). The respective role of the different climate factors in the coastal erosion process will vary spatially. Some studies have shown that the role of sea level rise on the coastal erosion process can be less important than other climate factors, like wave heights, changes in the frequency of the storms, and the cryogenic processes (Ruggiero 2013; Savard et al. 2009). Therefore, in order to have a complete picture of the potential effects of sea level rise on rates of coastal erosion, it is crucial to consider the combined effects of the aforementioned climate controls and the geomorphology of the coast under study.

Coastal wetlands around the world are sensitive to sea level rise. Projections of the impacts on global coastlines are inconclusive, with some projections suggesting that 20% to 90% (depending on sea level rise scenario) of present day wetlands will disappear during the 21st century (Spencer et al. 2016). Another study, which included natural feedback processes and management responses, suggested that coastal wetlands may actually increase (Schuerch et al. 2018).

Low-lying coastal areas in the tropics are particularly subject to the combined effect of sea level rise and increasing intensity of tropical cyclones, conditions that, in many cases, pose limits to adaptation (Section 4.8.5.1).

Many large coastal deltas are subject to the additional stress of shrinking deltas as a consequence of the combined effect of reduced sediment loads from rivers due to damming and water use, and land subsidence resulting from extraction of ground water or natural gas, and aquaculture (Higgins et al. 2013; Tessler et al. 2016; Minderhoud et al. 2017; Tessler et al. 2015; Brown and Nicholls 2015; Szabo et al. 2016; Yang et al. 2019; Shirzaei and Bürgmann 2018; Wang et al. 2018; Fuangswasdi et al. 2019). In some cases the rate of subsidence can outpace the rate of sea level rise by one order of magnitude (Minderhoud et al. 2017) or even two (Higgins et al. 2013). Recent findings from the Mississippi Delta raise the risk of a systematic underestimation of the rate of land subsidence in coastal deltas (Keogh and Törnqvist 2019).

In sum, from a land degradation point of view, low-lying coastal areas are particularly exposed to the nexus of climate change and increasing concentration of people (Elliott et al. 2014) (*robust* *evidence*, *high agreement*) and the situation will become particularly acute in delta areas shrinking from both reduced sediment loads and land subsidence (*robust evidence*, *high agreement*).

1.4.2 Indirect impacts on land degradation

Indirect impacts of climate change on land degradation are difficult to quantify because of the many conflating factors. The causes of land-use change are complex, combining physical, biological and socio-economic drivers (Lambin et al. 2001; Lambin and Meyfroidt 2011). One such driver of land-use change is the degradation of agricultural land, which can result in a negative cycle of natural land being converted to agricultural land to sustain production levels. The intensive management of agricultural land can lead to a loss of soil function, negatively impacting on the many ecosystem services provided by soils, including maintenance of water quality and soil carbon sequestration (Smith et al. 2016a). The degradation of soil quality due to cropping is of particular concern in tropical regions, where it results in a loss of productive potential of the land, affecting regional food security and driving conversion of non-agricultural land, such as forestry, to agriculture (Lambin et al. 2003; Drescher et al. 2016; Van der Laan et al. 2017). Climate change will exacerbate these negative cycles unless sustainable land management practices are implemented.

Climate change impacts on agricultural productivity (see Chapter 5) will have implications for the intensity of land use and hence exacerbate the risk of increasing land degradation. There will be both localised effects (i.e., climate change impacts on productivity affecting land use in the same region) and teleconnections (i.e., climate change impacts and land-use changes that are spatially and temporally separate) (Wicke et al. 2012; Pielke et al. 2007). If global temperature increases beyond 3°C it will have negative yield impacts on all crops (Porter et al. 2014) which, in combination with a doubling of demands by 2050 (Tilman et al. 2011), and increasing competition for land from the expansion of negative emissions technologies (IPCC 2018a; Schleussner et al. 2016), will exert strong pressure on agricultural lands and food security.

In sum, reduced productivity of most agricultural crops will drive land-use changes worldwide (*robust evidence*, *medium agreement*), but predicting how this will impact on land degradation is challenging because of several conflating factors. Social change, such as widespread changes in dietary preferences, will have a huge impact on agriculture and hence land degradation (*medium evidence*, *high agreement*).

1.5 Impacts of bioenergy and technologies for CO_2 removal (CDR) on land degradation

1.5.1 Potential scale of bioenergy and land-based CDR

In addition to the traditional land-use drivers (e.g., population growth, agricultural expansion, forest management), a new driver will interact to increase competition for land throughout this century: the potential large-scale implementation of land-based technologies for CO_2 removal (CDR). Land-based CDR includes afforestation and reforestation, bioenergy with carbon capture and storage (BECCS), soil carbon management, biochar and enhanced weathering (Smith et al. 2015; Smith 2016).

Most scenarios, including two of the four pathways in the IPCC Special Report on 1.5°C (IPCC 2018a), compatible with stabilisation at 2°C involve substantial areas devoted to land-based CDR, specifically afforestation/reforestation and BECCS (Schleussner et al. 2016; Smith et al. 2016b; Mander et al. 2017). Even larger land areas are required in most scenarios aimed at keeping average global temperature increases to below 1.5°C, and scenarios that avoid BECCS also require large areas of energy crops in many cases (IPCC 2018b), although some options with strict demand-side management avoid this need (Grubler et al. 2018). Consequently, the addition of carbon capture and storage (CCS) systems to bioenergy facilities enhances mitigation benefits because it increases the carbon retention time and reduces emissions relative to bioenergy facilities without CCS. The IPCC SR15 states that, 'When considering pathways limiting warming to 1.5°C with no or limited overshoot, the full set of scenarios shows a conversion of 0.5–11 Mkm^2 of pasture into 0–6 Mkm^2 for energy crops, a 2 Mkm^2 reduction to 9.5 Mkm^2 increase [in] forest, and a 4 Mkm^2 decrease to a 2.5 Mkm^2 increase in non-pasture agricultural land for food and feed crops by 2050 relative to 2010.' (Rogelj et al. 2018, p. 145). For comparison, the global cropland area in 2010 was 15.9 Mkm^2 (Table 1.1), and Woods et al. (2015) estimate that the area of abandoned and degraded land potentially available for energy crops (or afforestation/reforestation) exceeds 5 Mkm^2. However, the area of available land has long been debated, as much marginal land is subject to customary land tenure and used informally, often by impoverished communities (Baka 2013, 2014; Haberl et al. 2013; Young 1999). Thus, as noted in SR15, 'The implementation of land-based mitigation options would require overcoming socio-economic, institutional, technological, financing and environmental barriers that differ across regions.' (IPCC, 2018a, p. 18).

The wide range of estimates reflects the large differences among the pathways, availability of land in various productivity classes, types of negative emission technology implemented, uncertainties in computer models, and social and economic barriers to implementation (Fuss et al. 2018; Nemet et al. 2018; Minx et al. 2018).

1.5.2 Risks of land degradation from expansion of bioenergy and land-based CDR

The large-scale implementation of high-intensity dedicated energy crops, and harvest of crop and forest residues for bioenergy, could contribute to increases in the area of degraded lands: intensive land management can result in nutrient depletion, over-fertilisation and soil acidification, salinisation (from irrigation without adequate drainage), wet ecosystems drying (from increased evapotranspiration), as well as novel erosion and compaction processes (from high-impact biomass harvesting disturbances) and other land degradation processes described in Section 4.2.1.

Global integrated assessment models used in the analysis of mitigation pathways vary in their approaches to modelling CDR (Bauer et al. 2018) and the outputs have large uncertainties due to their limited capability to consider site-specific details (Krause et al. 2018). Spatial resolutions vary from 11 world regions to 0.25 degrees gridcells (Bauer et al. 2018). While model projections identify potential areas for CDR implementation (Heck et al. 2018), the interaction with climate-change-induced biome shifts, available land and its vulnerability to degradation are unknown. The crop/forest types and management practices that will be implemented are also unknown, and will be influenced by local incentives and regulations. While it is therefore currently not possible to project the area at risk of degradation from the implementation of land-based CDR, there is a clear risk that expansion of energy crops at the scale anticipated could put significant strain on land systems, biosphere integrity, freshwater supply and biogeochemical flows (Heck et al. 2018). Similarly, extraction of biomass for energy from existing forests, particularly where stumps are utilised, can impact on soil health (de Jong et al. 2017). Reforestation and afforestation present a lower risk of land degradation and may in fact reverse degradation (Section 4.5.3) although potential adverse hydrological and biodiversity impacts will need to be managed (Caldwell et al. 2018; Brinkman et al. 2017). Soil carbon management can deliver negative emissions while reducing or reversing land degradation. Chapter 6 discusses the significance of context and management in determining environmental impacts of implementation of land-based options.

1.5.3 Potential contributions of land-based CDR to reducing and reversing land degradation

Although large-scale implementation of land-based CDR has significant potential risks, the need for negative emissions and the anticipated investments to implement such technologies can also create significant opportunities. Investments into land-based CDR can contribute to halting and reversing land degradation, to the restoration or rehabilitation of degraded and marginal lands (Chazdon and Uriarte 2016; Fritsche et al. 2017) and can contribute to the goals of LDN (Orr et al. 2017).

Estimates of the global area of degraded land range from less than 10 to 60 Mkm^2 (Gibbs and Salmon 2015) (Section 4.3.1). Additionally, large areas are classified as marginal lands and may be suitable for the implementation of bioenergy and land-based CDR (Woods et al. 2015). The yield per hectare of marginal and degraded lands is lower than on fertile lands, and if CDR will be implemented on marginal and degraded lands, this will increase the area demand and costs per unit area of achieving negative emissions (Fritsche et al. 2017). The selection of lands suitable for CDR must be considered carefully to reduce conflicts with existing users, to assess the possible trade-offs in biodiversity contributions of the original and the CDR land uses, to quantify the impacts on water budgets, and to ensure sustainability of the CDR land use.

Land use and land condition prior to the implementation of CDR affect climate change benefits (Harper et al. 2018). Afforestation/reforestation on degraded lands can increase carbon stocks in vegetation and soil, increase carbon sinks (Amichev et al. 2012), and deliver co-benefits for biodiversity and ecosystem services, particularly if a diversity of local species are used. Afforestation and reforestation on native grasslands can reduce soil carbon stocks, although the loss is typically more than compensated by increases in biomass and dead organic matter carbon stocks (Bárcena et al. 2014; Li et al. 2012; Ovalle-Rivera et al. 2015; Shi et al. 2013), and may impact on biodiversity (Li et al. 2012).

Strategic incorporation of energy crops into agricultural production systems, applying an integrated landscape management approach, can provide co-benefits for management of land degradation and other environmental objectives. For example, buffers of Miscanthus and other grasses can enhance soil carbon and reduce water pollution (Cacho et al. 2018; Odgaard et al. 2019), and strip-planting of short-rotation tree crops can reduce the water table where crops are affected by dryland salinity (Robinson et al. 2006). Shifting to perennial grain crops has the potential to combine food production with carbon sequestration at a higher rate than annual grain crops and avoid the trade-off between food production and climate change mitigation (Crews et al. 2018; de Olivera et al. 2018; Ryan et al. 2018) (Section 4.9.2).

Changes in land cover can affect surface reflectance, water balances and emissions of volatile organic compounds and thus the non-GHG impacts on the climate system from afforestation/reforestation or planting energy crops (Anderson et al. 2011; Bala et al. 2007; Betts 2000; Betts et al. 2007) (see Section 4.6 for further details). Some of these impacts reinforce the GHG mitigation benefits, while others offset the benefits, with strong local (slope, aspect) and regional (boreal vs. tropical biomes) differences in the outcomes (Li et al. 2015). Adverse effects on albedo from afforestation with evergreen conifers in boreal zones can be reduced through planting of broadleaf deciduous species (Astrup et al. 2018; Cai et al. 2011a; Anderson et al. 2011).

Combining CDR technologies may prove synergistic. Two soil management techniques with an explicit focus on increasing the soil carbon content rather than promoting soil conservation more broadly have been suggested: addition of biochar to agricultural soils (Section 4.9.5) and addition of ground silicate minerals to soils in order to take up atmospheric CO_2 through chemical weathering (Taylor et al. 2017; Haque et al. 2019; Beerling 2017; Strefler et al. 2018). The addition of biochar is comparatively well understood and also field tested at large scale, see Section 4.9.5 for a comprehensive discussion. The addition of silicate minerals to soils is still highly uncertain in terms of its potential (from 95 $GtCO_2\ yr^{-1}$ (Strefler et al. 2018) to only 2–4 $GtCO_2\ yr^{-1}$ (Fuss et al. 2018)) and costs (Schlesinger and Amundson 2018).

Effectively addressing land degradation through implementation of bioenergy and land-based CDR will require site-specific local knowledge, matching of species with the local land, water balance, nutrient and climatic conditions, ongoing monitoring and, where necessary, adaptation of land management to ensure sustainability under global change (Fritsche et al. 2017). Effective land governance mechanisms including integrated land-use planning, along with strong sustainability standards could support deployment of energy crops and afforestation/reforestation at appropriate scales and geographical contexts (Fritsche et al. 2017). Capacity-building and technology transfer through the international cooperation mechanisms of the Paris Agreement could support such efforts. Modelling to inform policy development is most useful when undertaken with close interaction between model developers and other stakeholders including policymakers to ensure that models account for real world constraints (Dooley and Kartha 2018).

International initiatives to restore lands, such as the Bonn Challenge (Verdone and Seidl 2017) and the New York Declaration on Forests (Chazdon et al. 2017), and interventions undertaken for LDN and implementation of NDCs (see Glossary) can contribute to NET objectives. Such synergies may increase the financial resources available to meet multiple objectives (Section 4.8.4).

1.5.4 Traditional biomass provision and land degradation

Traditional biomass (fuelwood, charcoal, agricultural residues, animal dung) used for cooking and heating by some 2.8 billion people (38% of global population) in non-OECD countries accounts for more than half of all bioenergy used worldwide (IEA 2017; REN21 2018) (Cross-Chapter Box 7 in Chapter 6). Cooking with traditional biomass has multiple negative impacts on human health, particularly for women, children and youth (Machisa et al. 2013; Sinha and Ray 2015; Price 2017; Mendum and Njenga 2018; Adefuye et al. 2007) and on household productivity, including high workloads for women and youth (Mendum and Njenga 2018; Brunner et al. 2018; Hou et al. 2018; Njenga et al. 2019). Traditional biomass is land-intensive due to reliance on open fires, inefficient stoves and overharvesting of woodfuel, contributing to land degradation, losses in biodiversity and reduced ecosystem services (IEA 2017; Bailis et al. 2015; Masera et al. 2015; Specht et al. 2015; Fritsche et al. 2017; Fuso Nerini et al. 2017). Traditional woodfuels account for 1.9–2.3% of global GHG emissions, particularly in 'hotspots' of land degradation and fuelwood depletion in eastern Africa and South Asia, such that one-third of traditional woodfuels globally are harvested unsustainably (Bailis et al. 2015). Scenarios to significantly reduce reliance on traditional biomass in developing countries present multiple co-benefits (*high evidence*, *high agreement*), including reduced emissions of black carbon, a short-lived climate forcer that also causes respiratory disease (Shindell et al. 2012).

A shift from traditional to modern bioenergy, especially in the African context, contributes to improved livelihoods and can reduce land degradation and impacts on ecosystem services (Smeets et al. 2012; Gasparatos et al. 2018; Mudombi et al. 2018). In Sub-Saharan Africa, most countries mention woodfuel in their Nationally Determined Contribution (NDC) but fail to identify transformational processes to make fuelwood a sustainable energy source compatible with improved forest management (Amugune et al. 2017). In some regions, especially in South and Southeast Asia, a scarcity of woody biomass may lead to excessive removal and use of agricultural wastes and residues, which contributes to poor soil quality and land degradation (Blanco-Canqui and Lal 2009; Mateos et al. 2017).

In Sub-Saharan Africa, forest degradation is widely associated with charcoal production, although in some tropical areas rapid re-growth can offset forest losses (Hoffmann et al. 2017; McNicol et al. 2018). Overharvesting of wood for charcoal contributes to the high rate of deforestation in Sub-Saharan Africa, which is five times the world average, due in part to corruption and weak governance systems (Sulaiman et al. 2017). Charcoal may also be a by-product of forest clearing for agriculture, with charcoal sale providing immediate income when the land is cleared for food crops (Kiruki et al. 2017; Ndegwa et al. 2016). Besides loss of forest carbon stock, a further concern for climate change is methane and black carbon emissions from fuelwood burning and traditional charcoal-making processes (Bond et al. 2013; Patange et al. 2015; Sparrevik et al. 2015).

A fundamental difficulty in reducing environmental impacts associated with charcoal lies in the small-scale nature of much charcoal production in Sub-Saharan Africa, leading to challenges in regulating its production and trade, which is often informal, and in some cases illegal, but nevertheless widespread since charcoal is the most important urban cooking fuel (Zulu 2010; Zulu and Richardson 2013; Smith et al. 2015; World Bank 2009). Urbanisation combined with population growth has led to continuously increasing charcoal production. Low efficiency of traditional charcoal production results in a four-fold increase in raw woody biomass required and thus much greater biomass harvest (Hojas-Gascon et al. 2016; Smeets et al. 2012). With continuing urbanisation anticipated, increased charcoal production and use will probably contribute to increasing land pressures and increased land degradation, especially in Sub-Saharan Africa (*medium evidence*, *high agreement*).

Although it could be possible to source this biomass more sustainably, the ecosystem and health impacts of this increased demand for cooking fuel would be reduced through use of other renewable fuels or, in some cases, non-renewable fuels (LPG), as well as through improved efficiency in end-use and through better resource and supply chain management (Santos et al. 2017; Smeets et al. 2012; Hoffmann et al. 2017). Integrated response options such as agro-forestry (Chapter 6) and good governance mechanisms for forest and agricultural management (Chapter 7) can support the transition to sustainable energy for households and reduce the environmental impacts of traditional biomass.

1.6 Impacts of land degradation on climate

While Chapter 2 has its focus on land cover changes and their impacts on the climate system, this chapter focuses on the influences of individual land degradation processes on climate (see Table 4.1) which may or may not take place in association with land cover changes. The effects of land degradation on CO_2 and other GHGs as well as those on surface albedo and other physical controls of the global radiative balance are discussed.

1.6.1 Impact on greenhouse gases (GHGs)

Land degradation processes with direct impact on soil and terrestrial biota have great relevance in terms of CO_2 exchange with the atmosphere, given the magnitude and activity of these reservoirs in the global carbon cycle. As the most widespread form of soil degradation, erosion detaches the surface soil material, which typically hosts the highest organic carbon stocks, favouring the mineralisation and release as CO_2. Yet complementary processes such as carbon burial may compensate for this effect, making soil erosion a long-term carbon sink (*low agreement, limited evidence*), (Wang et al. (2017b), but see also Chappell et al. (2016)). Precise estimation of the CO_2 released from eroded lands is challenged by the fact that only a fraction of the detached carbon is eventually lost to the atmosphere. It is important to acknowledge that a substantial fraction of the eroded material may preserve its organic carbon load in field conditions. Moreover, carbon sequestration may be favoured through the burial of both the deposited material and the surface of its hosting soil at the deposition location (Quinton et al. 2010). The cascading effects of erosion on other environmental processes at the affected sites can often cause net CO_2 emissions through their indirect influence on soil fertility, and the balance of organic carbon inputs and outputs, interacting with other non-erosive soil degradation processes (such as nutrient depletion, compaction and salinisation), which can lead to the same net carbon effects (see Table 4.1) (van de Koppel et al. 1997).

As natural and human-induced erosion can result in net carbon storage in very stable buried pools at the deposition locations, degradation in those locations has a high C-release potential. Coastal ecosystems such as mangrove forests, marshes and seagrasses are at typical deposition locations, and their degradation or replacement with other vegetation is resulting in a substantial carbon release (0.15 to 1.02 GtC yr^{-1}) (Pendleton et al. 2012), which highlights the need for a spatially integrated assessment of land degradation impacts on climate that considers in-situ but also ex-situ emissions.

Cultivation and agricultural management of cultivated land are relevant in terms of global CO_2 land–atmosphere exchange (Section 4.8.1). Besides the initial pulse of CO_2 emissions associated with the onset of cultivation and associated vegetation clearing (Chapter 2), agricultural management practices can increase or reduce carbon losses to the atmosphere. Although global croplands are considered to be at a relatively neutral stage in the current decade (Houghton et al. 2012), this results from a highly uncertain balance between coexisting net losses and gains. Degradation losses of soil and biomass carbon appear to be compensated by gains from soil protection and restoration practices such as cover crops, conservation tillage and nutrient replenishment favouring organic matter build-up. Cover crops, increasingly used to improve soils, have the potential to sequester 0.12 GtC yr^{-1} on global croplands with a saturation time of more than 150 years (Poeplau and Don 2015). No-till practices (i.e., tillage elimination favouring crop residue retention in the soil surface) which were implemented to protect soils from erosion and reduce land preparation times, were also seen with optimism as a carbon sequestration option, which today is considered more modest globally and, in some systems, even less certain (VandenBygaart 2016; Cheesman et al. 2016; Powlson et al. 2014). Among soil fertility restoration practices, lime application for acidity correction, increasingly important in tropical regions, can generate a significant net CO_2 source in some soils (Bernoux et al. 2003; Desalegn et al. 2017).

Land degradation processes in seminatural ecosystems driven by unsustainable uses of their vegetation through logging or grazing lead to reduced plant cover and biomass stocks, causing net carbon releases from soils and plant stocks. Degradation by logging activities is particularly prevalent in developing tropical and subtropical regions, involving carbon releases that exceed by far the biomass of harvested products, including additional vegetation and soil sources that are estimated to reach 0.6 GtC yr^{-1} (Pearson et al. 2014, 2017). Excessive grazing pressures pose a more complex picture with variable magnitudes and even signs of carbon exchanges. A general trend of higher carbon losses in humid overgrazed rangelands suggests a high potential for carbon sequestration following the rehabilitation of those systems (Conant and Paustian 2002) with a global potential sequestration of 0.045 GtC yr^{-1}. A special case of degradation in rangelands is the process leading to the woody encroachment of grass-dominated systems, which can be responsible for declining animal production but high carbon sequestration rates (Asner et al. 2003; Maestre et al. 2009).

Fire regime shifts in wild and seminatural ecosystems can become a degradation process in itself, with high impact on net carbon emission and with underlying interactive human and natural drivers such as burning policies (Van Wilgen et al. 2004), biological invasions (Brooks et al. 2009), and plant pest/disease spread (Kulakowski et al. 2003). Some of these interactive processes affecting unmanaged forests have resulted in massive carbon release, highlighting how degradation feedbacks on climate are not restricted to intensively used land but can affect wild ecosystems as well (Kurz et al. 2008).

Agricultural land and wetlands represent the dominant source of non-CO_2 greenhouse gases (GHGs) (Chen et al. 2018d). In agricultural land, the expansion of rice cultivation (increasing CH_4 sources), ruminant stocks and manure disposal (increasing CH_4, N_2O and NH_3 fluxes) and nitrogen over-fertilisation combined with soil acidification (increasing N_2O fluxes) are introducing the major impacts (*medium agreement*, *medium evidence*) and their associated emissions appear to be exacerbated by global warming (*medium agreement*, *medium evidence*) (Oertel et al. 2016).

As the major sources of global N_2O emissions, over-fertilisation and manure disposal are not only increasing in-situ sources but also stimulating those along the pathway of dissolved inorganic nitrogen transport all the way from draining waters to the ocean (*high agreement*, *medium evidence*). Current budgets of anthropogenically fixed nitrogen on the Earth System (Tian et al. 2015; Schaefer et al. 2016; Wang et al. 2017a) suggest that N_2O release from terrestrial soils and wetlands accounts for 10–15% of the emissions, yet many further release fluxes along the hydrological pathway remain uncertain, with emissions from oceanic 'dead-zones' being a major aspect of concern (Schlesinger 2009; Rabalais et al. 2014).

Environmental degradation processes focused on the hydrological system, which are typically manifested at the landscape scale, include both drying (as in drained wetlands or lowlands) and wetting trends (as in waterlogged and flooded plains). Drying of wetlands reduces CH_4 emissions (Turetsky et al. 2014) but favours pulses of organic matter mineralisation linked to high N_2O release (Morse and Bernhardt 2013; Norton et al. 2011). The net warming balance of these two effects is not resolved and may be strongly variable across different types of wetlands. In the case of flooding of non-wetland soils, a suppression of CO_2 release is typically overcompensated in terms of net greenhouse impact by enhanced CH_4 fluxes that stem from the lack of aeration but are aided by the direct effect of extreme wetting on the solubilisation and transport of organic substrates (McNicol and Silver 2014). Both wetlands rewetting/restoration and artificial wetland creation can increase CH_4 release (Altor and Mitsch 2006; Fenner et al. 2011). Permafrost thawing is another major source of CH_4 release, with substantial long-term contributions to the atmosphere that are starting to be globally quantified (Christensen et al. 2004; Schuur et al. 2015; Walter Anthony et al. 2016).

1.6.2 Physical impacts

Among the physical effects of land degradation, surface albedo changes are those with the most evident impact on the net global radiative balance and net climate warming/cooling. Degradation processes affecting wild and semi-natural ecosystems, such as fire regime changes, woody encroachment, logging and overgrazing, can trigger strong albedo changes before significant biogeochemical shifts take place. In most cases these two types of effects have opposite signs in terms of net radiative forcing, making their joint assessment critical for understanding climate feedbacks (Bright et al. 2015).

In the case of forest degradation or deforestation, the albedo impacts are highly dependent on the latitudinal/climatic belt to which they belong. In boreal forests, the removal or degradation of the tree cover increases albedo (net cooling effect) (*medium evidence*, *high agreement*) as the reflective snow cover becomes exposed, which can exceed the net radiative effect of the associated carbon release to the atmosphere (Davin et al. 2010; Pinty et al. 2011). On the other hand, progressive greening of boreal and temperate forests has contributed to net albedo declines (*medium agreement*, *medium evidence*) (Planque et al. 2017; Li et al. 2018a). In the northern treeless vegetation belt (tundra), shrub encroachment leads to the opposite effect as the emergence of plant structures above the snow cover level reduce winter-time albedo (Sturm 2005).

The extent to which albedo shifts can compensate for carbon storage shifts at the global level has not been estimated. A significant but partial compensation takes place in temperate and subtropical dry ecosystems in which radiation levels are higher and carbon stocks smaller compared to their more humid counterparts (*medium agreement*, *medium evidence*). In cleared dry woodlands, half of the net global warming effect of net carbon release has been compensated by albedo increase (Houspanossian et al. 2013), whereas in afforested dry rangelands, albedo declines cancelled one-fifth of the net carbon sequestration (Rotenberg and Yakir 2010). Other important cases in which albedo effects impose a partial compensation of carbon exchanges are the vegetation shifts associated with wildfires, as shown for the savannahs, shrublands and grasslands of Sub-Saharan Africa (Dintwe et al. 2017). Besides the net global effects discussed above, albedo shifts can play a significant role in local climate (*high agreement*, *medium evidence*), as exemplified by the effect of no-till agriculture reducing local heat extremes in European landscapes (Davin et al. 2014) and the effects of woody encroachment causing precipitation rises in the North American Great Plains (Ge and Zou 2013). Modelling efforts that integrate ground data from deforested areas worldwide accounting for both physical and biogeochemical effects, indicate that massive global deforestation would have a net warming impact (Lawrence and Vandecar 2015) at both local and global levels with highlight non-linear effects of forest loss on climate variables.

Beyond the albedo effects presented above, other physical impacts of land degradation on the atmosphere can contribute to global and regional climate change. Of particular relevance, globally and continentally, are the net cooling effects of dust emissions (*low agreement*, *medium evidence*) (Lau and Kim (2007), but see also Huang et al. (2014)). Anthropogenic emission of mineral particles from degrading land appear to have a similar radiative impact than all other anthropogenic aerosols (Sokolik and Toon 1996). Dust emissions may explain regional climate anomalies through reinforcing feedbacks, as suggested for the amplification of the intensity, extent and duration of the low precipitation anomaly of the North American Dust Bowl in the 1930s (Cook et al. 2009). Another source of physical effects on climate are surface roughness changes which, by affecting atmospheric drag, can alter cloud formation and precipitation (*low agreement*, *low evidence*), as suggested by modelling studies showing how the massive deployment of solar panels in the Sahara could increase rainfall in the Sahel (Li et al. 2018c), or how woody encroachment in the Arctic tundra could reduce cloudiness and raise temperature (Cho et al. 2018). The complex physical effects of deforestation, as explored through modelling, converge into general net regional precipitation declines, tropical temperature increases and boreal temperature declines, while net global effects are less certain (Perugini et al. 2017). Integrating all the physical effects of land degradation and its recovery or reversal is still a challenge, yet modelling attempts suggest that, over the last three decades, the slow but persistent net global greening caused by the average increase of leaf area in the land has caused a net cooling of the Earth, mainly through the rise in evapotranspiration (Zeng et al. 2017) (*low confidence*).

1.7 Impacts of climate-related land degradation on poverty and livelihoods

Unravelling the impacts of climate-related land degradation on poverty and livelihoods is highly challenging. This complexity is due to the interplay of multiple social, political, cultural and economic factors, such as markets, technology, inequality, population growth, (Barbier and Hochard 2018) each of which interact and shape the ways in which social-ecological systems respond (Morton 2007). We find *limited evidence* attributing the impacts of climate-related land degradation to poverty and livelihoods, with climate often not distinguished from any other driver of land degradation. Climate is nevertheless frequently noted as a risk multiplier for both land degradation and poverty (*high agreement*, *robust evidence*) and is one of many stressors people live with, respond to and adapt to in their daily lives (Reid and Vogel 2006). Climate change is considered to exacerbate land degradation and potentially accelerate it due to heat stress, drought, changes to evapotranspiration rates and biodiversity, as well as a result of changes to environmental conditions that allow new pests and diseases to thrive (Reed and Stringer 2016). In general terms, the climate (and climate change) can increase human and ecological communities' sensitivity to land degradation. Land degradation then leaves livelihoods more sensitive to the impacts of climate change and extreme climatic events (*high agreement*, *robust evidence*). If human and ecological communities exposed to climate change and land degradation are sensitive and cannot adapt, they can be considered vulnerable to it; if they are sensitive and can adapt, they can be considered resilient (Reed and Stringer 2016). The impacts of land degradation will vary under a changing climate, both spatially and temporally, leading some communities and ecosystems to be more vulnerable or more resilient than others under different scenarios. Even within communities, groups such as women and youth are often more vulnerable than others.

1.7.1 Relationships between land degradation, climate change and poverty

This section sets out the relationships between land degradation and poverty, and climate change and poverty, leading to inferences about the three-way links between them. Poverty is multidimensional and includes a lack of access to the whole range of capital assets that can be used to pursue a livelihood. Livelihoods constitute the capabilities, assets and activities that are necessary to make a living (Chambers and Conway 1992; Olsson et al. 2014b).

The literature shows *high agreement* in terms of speculation that there are *potential* links between land degradation and poverty. However, studies have not provided robust quantitative assessments of the extent and incidence of poverty within populations affected by land degradation (Barbier and Hochard 2016). Some researchers, for example, Nachtergaele et al. (2011) estimate that 1.5 billion people were dependent upon degraded land to support their livelihoods in 2007, while >42% of the world's poor population inhabit degraded areas. However, there is overall *low confidence* in the evidence base, a lack of studies that look beyond the past and present, and the literature calls for more in-depth research to be undertaken on these issues (Gerber et al. 2014). Recent work by Barbier and Hochard (2018) points to biophysical constraints such as poor soils and limited rainfall, which interact to limit land productivity, suggesting that those farming in climatically less-favourable agricultural areas are challenged by poverty. Studies such as those by Coomes et al. (2011), focusing on an area in the Amazon, highlight the importance of the initial conditions of land holding in the dominant (shifting) cultivation system in terms of long-term effects on household poverty and future forest cover, showing that initial land tenure and socio-economic aspects can make some areas less favourable too.

Much of the qualitative literature is focused on understanding the livelihood and poverty impacts of degradation through a focus on subsistence agriculture, where farms are small, under traditional or informal tenure and where exposure to environmental (including climate) risks is high (Morton 2007). In these situations, poorer people lack access to assets (financial, social, human, natural and physical) and in the absence of appropriate institutional supports and social protection, this leaves them sensitive and unable to adapt, so a vicious cycle of poverty and degradation can ensue. To further illustrate the complexity, livelihood assessments often focus on a single snapshot in time. Livelihoods are dynamic and people alter their livelihood activities and strategies depending on the internal and external stressors to which they are responding (O'Brien et al. 2004). When certain livelihood activities and strategies are no longer tenable as a result of land degradation (and may push people into poverty), land degradation can have further effects on issues such as migration (Lee 2009), as people adapt by moving (Section 4.7.3); and may result in conflict (Section 4.7.3), as different groups within society compete for scarce resources, sometimes through non-peaceful actions. Both migration and conflict can lead to land-use changes elsewhere that further fuel climate change through increased emissions.

Similar challenges as for understanding land degradation–poverty linkages are experienced in unravelling the relationship between climate change and poverty. A particular issue in examining climate change–poverty links relates to the common use of aggregate economic statistics like GDP, as the assets and income of the poor constitute a minor proportion of national wealth (Hallegatte et al. 2018). Aggregate quantitative measures also fail to capture the distributions of costs and benefits from climate change. Furthermore, people fall into and out of poverty, with climate change being one of many factors affecting these dynamics, through its impacts on livelihoods. Much of the literature on climate change and poverty tends to look backward rather than forward (Skoufias et al. 2011), providing a snapshot of current or past relationships (for example, Dell et al. (2009) who examine the relationship between temperature and income (GDP) using cross-sectional data from countries in the Americas). Yet, simulations of future climate change impacts on income or poverty are largely lacking.

Noting the *limited evidence* that exists that explicitly focuses on the relationship between land degradation, climate change and poverty, Barbier and Hochard (2018b) suggest that those people living in less-favoured agricultural areas face a poverty–environment trap that can result in increased land degradation under climate change conditions. The emergent relationships between land degradation, climate change and poverty are shown in Figure 4.6 (see also Figure 6.1).

The poor have access to few productive assets – so land, and the natural resource base more widely, plays a key role in supporting the livelihoods of the poor. It is, however, hard to make generalisations about how important income derived from the natural resource base is for rural livelihoods in the developing world (Angelsen et al. 2014). Studies focusing on forest resources have shown that approximately one quarter of the total rural household income in developing countries stems from forests, with forest-based income shares being tentatively higher for low-income households (Vedeld et al. 2007; Angelsen et al. 2014). Different groups use land in different ways within their overall livelihood portfolios and are, therefore, at different levels of exposure and sensitivity to climate shocks and stresses. The literature nevertheless displays *high evidence* and *high agreement* that those populations whose livelihoods are more sensitive to climate change and land degradation are often more dependent on environmental assets, and these people are often the poorest members of society. There is further *high evidence* and *high agreement* that both climate change and land degradation can affect livelihoods and poverty through their threat multiplier effect. Research in Bellona, in the Solomon Islands in the South Pacific (Reenberg et al. 2008) examined event-driven impacts on livelihoods, taking into account weather events as one of many drivers of land degradation and links to broader land use and land cover changes that have taken place. Geographical locations experiencing land degradation are often the same locations that are directly affected by poverty, and also by extreme events linked to climate change and variability.

Much of the assessment presented above has considered place-based analyses examining the relationships between poverty, land degradation and climate change in the locations in which these outcomes have occurred. Altieri and Nicholls (2017) note that, due to the globalised nature of markets and consumption systems, the impacts of changes in crop yields linked to climate-related land degradation (manifest as lower yields) will be far reaching, beyond the sites and livelihoods experiencing degradation. Despite these teleconnections, farmers living in poverty in developing countries will be especially vulnerable due to their exposure, dependence on the environment for income and limited options to engage in other ways to make a living (Rosenzweig and Hillel 1998). In identifying ways in which these interlinkages can be addressed, Scherr (2000) observes that key actions that can jointly address poverty and environmental improvement often seek to increase access to natural resources, enhance the productivity of the natural resource assets of the poor, and engage stakeholders in addressing public natural resource management issues. In this regard, it is increasingly recognised that those suffering from, and being vulnerable to, land degradation and poverty need to have a voice and play a role in the development of solutions, especially where the natural resources and livelihood activities they depend on are further threatened by climate change.

1.7.2 Impacts of climate-related land degradation on food security

How and where we grow food, compared to where and when we need to consume it, is at the crux of issues surrounding land degradation, climate change and food security, especially because more than 75% of the global land surface (excluding Antarctica) faces rain-fed crop production constraints (Fischer et al. 2009), see also Chapter 5. Taken separately, knowledge on land degradation processes and human-induced climate change has attained a great level of maturity. However, their combined effects on food security, notably food supply, remain underappreciated (Webb et al. 2017b), and quantitative information is lacking. Just a few studies have shown how the interactive effects of the aforementioned challenging, interrelated phenomena can impact on crop productivity and hence food security and quality (Karami et al. 2009; Allen et al. 2001; Högy and Fangmeier 2008) (*low evidence*). Along with socio-economic drivers, climate change accelerates land degradation due to its influence on land-use systems (Millennium Ecosystem Assessment 2005; UNCCD 2017), potentially leading to a decline in agri-food system productivity, particularly on the supply side. Increases in temperature and changes in precipitation patterns are expected to have impacts on soil quality, including nutrient availability and assimilation (St.Clair and Lynch 2010). Those climate-related changes are expected to have net negative impacts on agricultural productivity, particularly in tropical regions, though the magnitude of impacts depends on the models used. Anticipated supply-side issues linked to land and climate relate to biocapacity factors (including e.g., whether there is enough water to support agriculture); production factors (e.g., chemical pollution of soil and water resources or lack of soil nutrients) and distribution issues (e.g., decreased availability of and/or accessibility to the necessary diversity of quality food where and when it is needed) (Stringer et al. 2011). Climate-sensitive transport infrastructure is

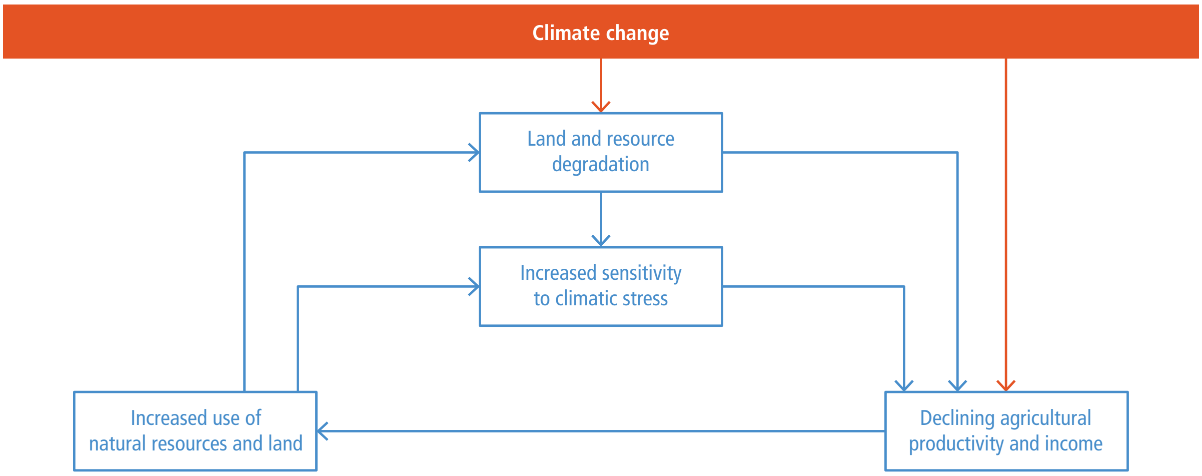

Figure 4.6 | Schematic representation of links between climate change, land management and socio-economic conditions.

also problematic for food security (Islam et al. 2017), and can lead to increased food waste, while poor siting of roads and transport links can lead to soil erosion and forest loss (Xiao et al. 2017), further feeding back into climate change.

Over the past decades, crop models have been useful tools for assessing and understanding climate change impacts on crop productivity and food security (White et al. 2011; Rosenzweig et al. 2014). Yet, the interactive effects of soil parameters and climate change on crop yields and food security remain limited, with *low evidence* of how they play out in different economic and climate settings (e.g., Sundström et al. 2014). Similarly, there have been few meta-analyses focusing on the adaptive capacity of land-use practices such as conservation agriculture in light of climate stress (see e.g., Steward et al. 2018), as well as *low evidence* quantifying the role of wild foods and forests (and, by extension, forest degradation) in both the global food basket and in supporting household-scale food security (Bharucha and Pretty 2010; Hickey et al. 2016).

To be sustainable, any initiative aimed at addressing food security – encompassing supply, diversity and quality – must take into consideration the interactive effects between climate and land degradation in a context of other socio-economic stressors. Such socio-economic factors are especially important if we look at demand-side issues too, which include lack of purchasing power, large-scale speculation on global food markets, leading to exponential price rises (Tadesse et al. 2014), competition in appropriation of supplies and changes to per capita food consumption (Stringer et al. 2011) (Chapter 5). Lack of food security, combined with lack of livelihood options, is often an important manifestation of vulnerability, and can act as a key trigger for people to migrate. In this way, migration becomes an adaptation strategy.

1.7.3 Impacts of climate-related land degradation on migration and conflict

Land degradation may trigger competition for scarce natural resources, potentially leading to migration and/or conflict, though, even with *medium evidence*, there is *low agreement* in the literature. Linkages between land degradation and migration occur within a larger context of multi-scale interaction of environmental and non-environmental drivers and processes, including resettlement projects, searches for education and/or income, land shortages, political turmoil, and family-related reasons (McLeman 2017; Hermans and Ide 2019). The complex contribution of climate to migration and conflict hampers retrieving any level of confidence on climate-migration and climate-conflict linkages, therefore constituting a major knowledge gap (Cramer et al. 2014; Hoegh-Guldberg et al. 2018).

There is *low evidence* on the causal linkages between climate change, land degradation processes (other than desertification) and migration. Existing studies on land degradation and migration – particularly in drylands – largely focus on the effect of rainfall variability and drought, and show how migration serves as adaptation strategy (Piguet et al. 2018; McLeman 2017; Chapter 3). For example, in the Ethiopian highlands, severe topsoil erosion and forest degradation is a major environmental stressor which is amplified by recurring droughts, with migration being an important household adaptation strategy (Morrissey 2013). In the humid tropics, land degradation, mainly as a consequence of deforestation, has been a reported reason for people leaving their homes during the Amazonian colonisation (Hecht 1983) but was also observed more recently, for example in Guatemala, where soil degradation was one of the most frequently cited migration pushes (López-Carr 2012) and Kenya, where households respond to low soil quality by sending temporary migrants for additional income generation (Gray 2011). In contrast, in the Andean highlands and the Pacific coastal plain, migration increased with land quality, probably because revenues from additional agricultural production was invested in costly forms of migration (Gray and Bilsborrow 2013). These mixed results illustrate the complex, non-linear relationship of land degradation–migration linkages and suggest that explaining land degradationand migration linkages requires considering a broad range of socio-ecological conditions (McLeman 2017).

In addition to people moving away from an area due to 'lost' livelihood activities, climate-related land degradation can also reduce the availability of livelihood safety nets – environmental assets that people use during times of shocks or stress. For example, Barbier (2000) notes that wetlands in north-east Nigeria around Hadejia–Jama'are floodplain provide dry season pastures for seminomadic herders, agricultural surpluses for Kano and Borno states, groundwater recharge of the Chad formation aquifer and 'insurance' resources in times of drought. The floodplain also supports many migratory bird species. As climate change and land degradation combine, delivery of these multiple services can be undermined, particularly as droughts become more widespread, reducing the utility of this wetland environment as a safety net for people and wildlife alike.

Early studies conducted in Africa hint at a significant causal link between land degradation and violent conflict (Homer-Dixon et al. 1993). For example, Percival and Homer-Dixon (1995) identified land degradation as one of the drivers of the crisis in Rwanda in the early 1990s, which allowed radical forces to stoke ethnic rivalries. With respect to the Darfur conflict, some scholars and United Nations Environment Programme (UNEP) concluded that land degradation, together with other environmental stressors, constitute a major security threat for the Sudanese people (Byers and Dragojlovic 2004; Sachs 2007; UNEP 2007). Recent studies show *low agreement*, suggesting that climate change can increase the likelihood of civil violence if certain economic, political and social factors, including low development and weak governance mechanisms, are present (Scheffran et al. 2012; Benjaminsen et al. 2012). In contrast, Raleigh and Urdal (2007) found in a global study that land degradation is a weak predictor for armed conflict. As such, studies addressing possible linkages between climate change – a key driver of land degradation – and the risks of conflict have yielded contradictory results, and it remains largely unclear whether land degradation resulting from climate change leads to conflict or cooperation (Salehyan 2008; Solomon et al. 2018).

Land degradation–conflict linkages can be bi-directional. Research suggests that households experiencing natural resource degradation often engage in migration for securing livelihoods (Kreamer 2012), which potentially triggers land degradation at the destination, leading to conflict there (Kassa et al. 2017). While this indeed holds true for some cases, it may not for others, given the complexity of processes, contexts and drivers. Where conflict and violence do ensue, it is often as a result of a lack of appreciation for the cultural practices of others.

1.8 Addressing land degradation in the context of climate change

Land degradation in the form of soil carbon loss is estimated to have been ongoing for at least 12,000 years, but increased exponentially in the last 200 years (Sanderman et al. 2017). Before the advent of modern sources of nutrients, it was imperative for farmers to maintain and improve soil fertility through the prevention of runoff and erosion, and management of nutrients through vegetation residues and manure. Many ancient farming systems were sustainable for hundreds and even thousands of years, such as raised-field agriculture in Mexico (Crews and Gliessman 1991), tropical forest gardens in Southeast Asia and Central America (Ross 2011; Torquebiau 1992; Turner and Sabloff 2012), terraced agriculture in East Africa, Central America, Southeast Asia and the Mediterranean basin (Turner and Sabloff 2012; Preti and Romano 2014; Widgren and Sutton 2004; Håkansson and Widgren 2007; Davies and Moore 2016; Davies 2015), and integrated rice–fish cultivation in East Asia (Frei and Becker 2005).

Such long-term sustainable farming systems evolved in very different times and geographical contexts, but they share many common features, such as: the combination of species and structural diversity in time and space (horizontally and vertically) in order to optimise the use of available land; recycling of nutrients through biodiversity of plants, animals and microbes; harnessing the full range of site-specific micro-environments (e.g., wet and dry soils); biological interdependencies which help suppression of pests; reliance on mainly local resources; reliance on local varieties of crops, and sometimes incorporation of wild plants and animals; the systems are often labour and knowledge intensive (Rudel et al. 2016; Beets 1990; Netting 1993; Altieri and Koohafkan 2008). Such farming systems have stood the test of time and can provide important knowledge for adapting farming systems to climate change (Koohafkann and Altieri 2011).

In modern agriculture, the importance of maintaining the biological productivity and ecological integrity of farmland has not been a necessity in the same way as in pre-modern agriculture because nutrients and water have been supplied externally. The extreme land degradation in the US Midwest during the Dust Bowl period in the 1930s became an important wake-up call for agriculture and agricultural research and development, from which we can still learn much in order to adapt to ongoing and future climate change (McLeman et al. 2014; Baveye et al. 2011; McLeman and Smit 2006).

SLM is a unifying framework for addressing land degradation and can be defined as the stewardship and use of land resources, including soils, water, animals and plants, to meet changing human needs, while simultaneously ensuring the long-term productive potential of these resources and the maintenance of their environmental functions. It is a comprehensive approach comprising technologies combined with social, economic and political enabling conditions (Nkonya et al. 2011). It is important to stress that farming systems are informed by both scientific and local/traditional knowledge. The power of SLM in small-scale diverse farming was demonstrated effectively in Nicaragua after the severe cyclone Mitch in 1998 (Holt-Giménez 2002). Pairwise analysis of 880 fields with and without implementation of SLM practices showed that the SLM fields systematically fared better than the fields without SLM in terms of more topsoil remaining, higher field moisture, more vegetation, less erosion and lower economic losses after the cyclone. Furthermore, the difference between fields with and without SLM increased with increasing levels of storm intensity, slope gradient, and age of SLM (Holt-Giménez 2002).

When addressing land degradation through SLM and other approaches, it is important to consider feedbacks that impact on climate change. Table 4.2 shows some of the most important land degradation issues, their potential solutions, and their impacts on climate change. This table provides a link between the comprehensive lists of land degradation processes (Table 4.1) and land management solutions.

1.8.1 Actions on the ground to address land degradation

Concrete actions on the ground to address land degradation are primarily focused on soil and water conservation. In the context of adaptation to climate change, actions relevant for addressing land degradation are sometimes framed as ecosystem-based adaptation (Scarano 2017) or Nature-Based Solutions (Nesshöver et al. 2017), and in an agricultural context, agroecology (see Glossary) provides an important frame. The site-specific biophysical and social conditions, including local and indigenous knowledge, are important for successful implementation of concrete actions.

Responses to land degradation generally take the form of agronomic measures (methods related to managing the vegetation cover), soil management (methods related to tillage, nutrient supply), and mechanical methods (methods resulting in durable changes to the landscape) (Morgan 2005a). Measures may be combined to reinforce benefits to land quality, as well as improving carbon sequestration that supports climate change mitigation. Some measures offer adaptation options and other co-benefits, such as agroforestry, involving planting fruit trees that can support food security in the face of climate change impacts (Reed and Stringer 2016), or application of compost or biochar that enhances soil water-holding capacity, so increases resilience to drought.

There are important differences in terms of labour and capital requirements for different technologies, and also implications for land tenure arrangements. Agronomic measures and soil management require generally little extra capital input and comprise activities repeated annually, so have no particular implication for land tenure

Table 4.2 | Interaction of human and climate drivers can exacerbate desertification and land degradation. Climate change exacerbates the rate and magnitude of several ongoing land degradation and desertification processes. Human drivers of land degradation and desertification include expanding agriculture, agricultural practices and forest management. In turn, land degradation and desertification are also drivers of climate change through GHG emissions, reduced rates of carbon uptake, and reduced capacity of ecosystems to act as carbon sinks into the future. Impacts on climate change are either warming (in red) or cooling (in blue).

Issue/ syndrome	Impact on climate change	Human driver	Climate driver	Land management options	References
Erosion of agricultural soils	Emission: CO_2, N_2O			Increase soil organic matter, no-till, perennial crops, erosion control, agroforestry, dietary change	3.1.4, 3.4.1, 3.5.2, 3.7.1, 4.8.1, 4.8.5, 4.9.2, 4.9.5
Deforestation	Emission of CO_2			Forest protection, sustainable forest management and dietary change	4.1.5, 4.5, 4.8.3, 4.8.4, 4.9.3
Forest degradation	Emission of CO_2 Reduced carbon sink			Forest protection, sustainable forest management	4.1.5, 4.5, 4.8.3, 4.8.4, 4.9.3
Overgrazing	Emission: CO_2, CH_4 Increasing albedo			Controlled grazing, rangeland management	3.1.4.2, 3.4.1, 3.6.1, 3.7.1, 4.8.1.4
Firewood and charcoal production	Emission: CO_2, CH_4 Increasing albedo			Clean cooking (health co-benefits, particularly for women and children)	3.6.3, 4.5.4, 4.8.3, 4.8.4
Increasing fire frequency and intensity	Emission: CO_2, CH_4, N_2O Emission: aerosols, increasing albedo			Fuel management, fire management	3.1.4, 3.6.1, 4.1.5, 4.8.3, Cross-Chapter Box 3 in Chp 2
Degradation of tropical peat soils	Emission: CO_2, CH_4			Peatland restoration, erosion control, regulating the use of peat soils	4.9.4
Thawing of permafrost	Emission: CO_2, CH_4			Relocation of settlement and infrastructure	4.8.5.1
Coastal erosion	Emission: CO_2, CH_4			Wetland and coastal restoration, mangrove conservation, long-term land-use planning	4 .9.6, 4.9. 7, 4.9.8
Sand and dust storms, wind erosion	Emission: aerosols			Vegetation management, afforestation, windbreaks	3.3.1, 3.4.1, 3.6.1, 3.7.1, 3.7.2
Bush encroachment	Capturing: CO_2, Decreasing albedo			Grazing land management, fire management	3.6.1.3, 3.7.3.2

Human driver	Climate driver
Grazing pressure	Warming trend
Agriculture practice	Extreme temperature
Expansion of agriculture	Drying trend
Forest clearing	Extreme rainfall
Wood fuel	Shifting rains
	Intensifying cyclones
	Sea level rise

arrangements. Mechanical methods require substantial upfront investments in terms of capital and labour, resulting in long-lasting structural change, requiring more secure land tenure arrangements (Mekuriaw et al. 2018). Agroforestry is a particularly important strategy for SLM in the context of climate change because of the large potential to sequester carbon in plants and soil and enhance resilience of agricultural systems (Zomer et al. 2016).

Implementation of SLM practices has been shown to increase the productivity of land (Branca et al. 2013) and to provide good economic returns on investment in many different settings around the world (Mirzabaev et al. 2015). Giger et al. (2018) showed, in a meta study of 363 SLM projects over the period 1990 to 2012, that 73% of the projects were perceived to have a positive or at least neutral cost-benefit ratio in the short term, and 97% were perceived to have a positive or very positive cost-benefit ratio in the long term (*robust evidence*, *high agreement*). Despite the positive effects, uptake is far from universal. Local factors, both biophysical conditions (e.g., soils, drainage, and topography) and socio-economic conditions (e.g., land tenure, economic status, and land fragmentation) play decisive roles in the interest in, capacity to undertake, and successful implementation of SLM practices (Teshome et al. 2016; Vogl et al. 2017; Tesfaye et al. 2016; Cerdà et al. 2018; Adimassu et al. 2016). From a landscape perspective, SLM can generate benefits, including adaptation to and mitigation of climate change, for entire watersheds, but challenges remain regarding coordinated and consistent implementation (*medium evidence*, *medium agreement*) (Kerr et al. 2016; Wang et al. 2016a).

1.8.1.1 Agronomic and soil management measures

Rebuilding soil carbon is an important goal of SLM, particularly in the context of climate change (Rumpel et al. 2018). The two most important reasons why agricultural soils have lost 20–60% of the soil carbon they contained under natural ecosystem conditions are the frequent disturbance through tillage and harvesting, and the change from deep-rooted perennial plants to shallow-rooted annual plants (Crews and

Rumsey 2017). Practices that build soil carbon are those that increase organic matter input to soil, or reduce decomposition of SOM.

Agronomic practices can alter the carbon balance significantly, by increasing organic inputs from litter and roots into the soil. Practices include retention of residues, use of locally adapted varieties, inter-cropping, crop rotations, and green manure crops that replace the bare field fallow during winter and are eventually ploughed before sowing the next main crop (Henry et al. 2018). Cover crops (green manure crops and catch crops that are grown between the main cropping seasons) can increase soil carbon stock by between 0.22 and 0.4 t C $ha^{-1}yr^{-1}$ (Poeplau and Don 2015; Kaye and Quemada 2017).

Reduced tillage (or no-tillage) is an important strategy for reducing soil erosion and nutrient loss by wind and water (Van Pelt et al. 2017; Panagos et al. 2015; Borrelli et al. 2016). But the evidence that no-till agriculture also sequesters carbon is not compelling (VandenBygaart 2016). Soil sampling of only the upper 30 cm can give biased results, suggesting that soils under no-till practices have higher carbon content than soils under conventional tillage (Baker et al. 2007; Ogle et al. 2012; Fargione et al. 2018; VandenBygaart 2016).

Changing from annual to perennial crops can increase soil carbon content (Culman et al. 2013; Sainju et al. 2017). A perennial grain crop (intermediate wheatgrass) was, on average, over four years a net carbon sink of about 13.5 tCO_2 ha^{-1} yr^{-1} (de Oliveira et al. 2018). Sprunger et al. (2018) compared an annual winter wheat crop with a perennial grain crop (intermediate wheatgrass) and found that the perennial grain root biomass was 15 times larger than winter wheat, however, there was no significant difference in soil carbon pools after the four-year experiment. Exactly how much, and over what time period, carbon can be sequestered through changing from annual to perennial crops depends on the degree of soil carbon depletion and other local biophysical factors (Section 4.9.2).

Integrated soil fertility management is a sustainable approach to nutrient management that uses a combination of chemical and organic amendments (manure, compost, biosolids, biochar), rhizobial nitrogen fixation, and liming materials to address soil chemical constraints (Henry et al. 2018). In pasture systems, management of grazing pressure, fertilisation, diverse species including legumes and perennial grasses can reduce erosion and enhance soil carbon (Conant et al. 2017).

1.8.1.2 Mechanical soil and water conservation

In hilly and mountainous terrain, terracing is an ancient but still practised soil conservation method worldwide (Preti and Romano 2014) in climatic zones from arid to humid tropics (Balbo 2017). By reducing the slope gradient of hillsides, terraces provide flat surfaces. Deep, loose soils that increase infiltration, reduce erosion and thus sediment transport. They also decrease the hydrological connectivity and thus reduce hillside runoff (Preti et al. 2018; Wei et al. 2016; Arnáez et al. 2015; Chen et al. 2017). In terms of climate change, terraces are a form of adaptation that helps in cases where rainfall is increasing or intensifying (by reducing slope gradient and the hydrological connectivity), and where rainfall is decreasing (by increasing infiltration and reducing runoff) (*robust evidence*, *high agreement*). There are several challenges, however, to continued maintenance and construction of new terraces, such as the high costs in terms of labour and/or capital (Arnáez et al. 2015) and disappearing local knowledge for maintaining and constructing new terraces (Chen et al. 2017). The propensity of farmers to invest in mechanical soil conservation methods varies with land tenure; farmers with secure tenure arrangements are more willing to invest in durable practices such as terraces (Lovo 2016; Sklenicka et al. 2015; Haregeweyn et al. 2015). Where the slope is less severe, erosion can be controlled by contour banks, and the keyline approach (Duncan 2016; Stevens et al. 2015) to soil and water conservation.

1.8.1.3 Agroforestry

Agroforestry is defined as a collective name for land-use systems in which woody perennials (trees, shrubs, etc.) are grown in association with herbaceous plants (crops, pastures) and/or livestock in a spatial arrangement, a rotation, or both, and in which there are both ecological and economic interactions between the tree and non-tree components of the system (Young, 1995, p. 11). At least since the 1980s, agroforestry has been widely touted as an ideal land management practice in areas vulnerable to climate variations and subject to soil erosion. Agroforestry holds the promise of improving soil and climatic conditions, while generating income from wood energy, timber and non-timber products – sometimes presented as a synergy of adaptation and mitigation of climate change (Mbow et al. 2014).

There is strong scientific consensus that a combination of forestry with agricultural crops and/or livestock, agroforestry systems can provide additional ecosystem services when compared with monoculture crop systems (Waldron et al. 2017; Sonwa et al. 2011, 2014, 2017; Charles et al. 2013). Agroforestry can enable sustainable intensification by allowing continuous production on the same unit of land with higher productivity without the need to use shifting agriculture systems to maintain crop yields (Nath et al. 2016). This is especially relevant where there is a regional requirement to find a balance between the demand for increased agricultural production and the protection of adjacent natural ecosystems such as primary and secondary forests (Mbow et al. 2014). For example, the use of agroforestry for perennial crops such as coffee and cocoa is increasingly promoted as offering a route to sustainable farming, with important climate change adaptation and mitigation co-benefits (Sonwa et al. 2001; Kroeger et al. 2017). Reported co-benefits of agroforestry in cocoa production include increased carbon sequestration in soils and biomass, improved water and nutrient use efficiency and the creation of a favourable micro-climate for crop production (Sonwa et al. 2017; Chia et al. 2016). Importantly, the maintenance of soil fertility using agroforestry has the potential to reduce the practice of shifting agriculture (of cocoa) which results in deforestation (Gockowski and Sonwa 2011). However, positive interactions within these systems can be ecosystem and/or species specific, but co-benefits such as increased resilience to extreme climate events, or improved soil fertility are not always observed (Blaser et al. 2017; Abdulai et al. 2018). These contrasting outcomes indicate the importance of field-scale research programmes to inform agroforestry system design, species selection and management practices (Sonwa et al. 2014).

Despite the many proven benefits, adoption of agroforestry has been low and slow (Toth et al. 2017; Pattanayak et al. 2003; Jerneck and Olsson 2014). There are several reasons for the slow uptake, but the perception of risks and the time lag between adoption and realisation of benefits are often important (Pattanayak et al. 2003; Mercer 2004; Jerneck and Olsson 2013).

An important question for agroforestry is whether it supports poverty alleviation, or if it favours comparatively affluent households. Experiences from India suggest that the overall adoption is low, with a differential between rich and poor households. Brockington el al. (2016), studied agroforestry adoption over many years in South India and found that, overall, only 18% of the households adopted agroforestry. However, among the relatively rich households who adopted agroforestry, 97% were still practising it after six to eight years, and some had expanded their operations. Similar results were obtained in Western Kenya, where food-secure households were much more willing to adopt agroforestry than food-insecure households (Jerneck and Olsson 2013, 2014). Other experiences from Sub-Saharan Africa illustrate the difficulties (such as local institutional support) of having a continued engagement of communities in agroforestry (Noordin et al. 2001; Matata et al. 2013; Meijer et al. 2015).

1.8.1.4 Crop–livestock interaction as an approach to managing land degradation

The integration of crop and livestock production into 'mixed farming' for smallholders in developing countries became an influential model, particularly for Africa, in the early 1990s (Pritchard et al. 1992; McIntire et al. 1992). Crop–livestock integration under this model was seen as founded on three pillars: improved use of manure for crop fertility management; expanded use of animal traction (draught animals); and promotion of cultivated fodder crops. For Asia, emphasis was placed on draught power for land preparation, manure for soil fertility enhancement, and fodder production as an entry point for cultivation of legumes (Devendra and Thomas 2002). Mixed farming was seen as an evolutionary process to expand food production in the face of population increase, promote improvements in income and welfare, and protect the environment. The process could be further facilitated and steered by research, agricultural advisory services and policy (Pritchard et al. 1992; McIntire et al. 1992; Devendra 2002).

Scoones and Wolmer (2002) place this model in historical context, including concern about population pressure on resources and the view that mobile pastoralism was environmentally damaging. The latter view had already been critiqued by developing understandings of pastoralism, mobility and communal tenure of grazing lands (e.g., Behnke 1994; Ellis 1994). They set out a much more differentiated picture of crop–livestock interactions, which can take place either within a single-farm household, or between crop and livestock producers, in which case they will be mediated by formal and informal institutions governing the allocation of land, labour and capital, with the interactions evolving through multiple place-specific pathways (Ramisch et al. 2002; Scoones and Wolmer 2002). Promoting a diversity of approaches to crop–livestock interactions does not imply that the integrated model necessarily leads to land degradation, but increases the space for institutional support to local innovation (Scoones and Wolmer 2002).

However, specific managerial and technological practices that link crop and livestock production will remain an important part of the repertoire of on-farm adaptation and mitigation. Howden and coauthors (Howden et al. 2007) note the importance of innovation within existing integrated systems, including use of adapted forage crops. Rivera-Ferre et al. (2016) list as adaptation strategies with high potential for grazing systems, mixed crop–livestock systems or both: crop–livestock integration in general; soil management, including composting; enclosure and corralling of animals; improved storage of feed. Most of these are seen as having significant co-benefits for mitigation, and improved management of manure is seen as a mitigation measure with adaptation co-benefits.

1.8.2 Local and indigenous knowledge for addressing land degradation

In practice, responses are anchored in scientific research, as well as local, indigenous and traditional knowledge and know-how. For example, studies in the Philippines by Camacho et al. (2016) examine how traditional integrated watershed management by indigenous people sustain regulating services vital to agricultural productivity, while delivering co-benefits in the form of biodiversity and ecosystem resilience at a landscape scale. Although responses can be site specific and sustainable at a local scale, the multi-scale interplay of drivers and pressures can nevertheless cause practices that have been sustainable for centuries to become less so. Siahaya et al. (2016) explore the traditional knowledge that has informed rice cultivation in the uplands of East Borneo, grounded in sophisticated shifting cultivation methods (*gilir balik*) which have been passed on for generations (more than 200 years) in order to maintain local food production. Gilir balik involves temporary cultivation of plots, after which, abandonment takes place as the land user moves to another plot, leaving the natural (forest) vegetation to return. This approach is considered sustainable if it has the support of other subsistence strategies, adapts to and integrates with the local context, and if the carrying capacity of the system is not surpassed (Siahaya et al. 2016). Often gilir balik cultivation involves intercropping of rice with bananas, cassava and other food crops. Once the abandoned plot has been left to recover such that soil fertility is restored, clearance takes place again and the plot is reused for cultivation. Rice cultivation in this way plays an important role in forest management, with several different types of succession forest being found in the study by Siahaya et al. (2016). Nevertheless, interplay of these practices with other pressures (large-scale land acquisitions for oil palm plantation, logging and mining), risk their future sustainability. Use of fire is critical in processes of land clearance, so there are also trade-offs for climate change mitigation, which have been sparsely assessed.

Interest appears to be growing in understanding how indigenous and local knowledge inform land users' responses to degradation, as scientists engage farmers as experts in processes of knowledge co-production and co-innovation (Oliver et al. 2012; Bitzer and Bijman 2015). This can help to introduce, implement, adapt and promote the use of locally appropriate responses (Schwilch et al. 2011). Indeed, studies strongly agree on the importance of engaging local populations in both sustainable land and forest management. Meta-analyses in tropical regions that examined both forests in protected areas and community-managed forests suggest that deforestation rates are lower, with less variation in deforestation rates presenting in community-managed forests compared to protected forests (Porter-Bolland et al. 2012). This suggests that consideration of the social and economic needs of local human populations is vital in preventing forest degradation (Ward et al. 2018). However, while disciplines such as ethnopedology seek to record and understand how local people perceive, classify and use soil, and draw on that information to inform its management (Barrera-Bassols and Zinck 2003), links with climate change and its impacts (perceived and actual) are not generally considered.

1.8.3 Reducing deforestation and forest degradation and increasing afforestation

Improved stewardship of forests through reduction or avoidance of deforestation and forest degradation, and enhancement of forest carbon stocks can all contribute to land-based natural climate solutions (Angelsen et al. 2018; Sonwa et al. 2011; Griscom et al. 2017). While estimates of annual emissions from tropical deforestation and forest degradation range widely from 0.5 to 3.5 GtC yr^{-1} (Baccini et al. 2017; Houghton et al. 2012; Mitchard 2018; see also Chapter 2), they all indicate the large potential to reduce annual emissions from deforestation and forest degradation. Recent estimates of forest extent for Africa in 1900 may result in downward adjustments of historic deforestation and degradation emission estimates (Aleman et al. 2018). Emissions from forest degradation in non-Annex I countries have declined marginally from 1.1 $GtCO_2$ yr^{-1} in 2001–2010 to 1 $GtCO_2$ yr^{-1} in 2011–2015, but the relative emissions from degradation compared to deforestation have increased from a quarter to a third (Federici et al. 2015). Forest sector activities in developing countries were estimated to represent a technical mitigation potential in 2030 of 9 $GtCO_2$ (Miles et al. 2015). This was partitioned into reduction of deforestation (3.5 $GtCO_2$), reduction in degradation and forest management (1.7 $GtCO_2$) and afforestation and reforestation (3.8 $GtCO_2$). The economic mitigation potential will be lower than the technical potential (Miles et al. 2015).

Natural regeneration of second-growth forests enhances carbon sinks in the global carbon budget (Chazdon and Uriarte 2016). In Latin America, Chazdon et al. (2016) estimated that, in 2008, second-growth forests (up to 60 years old) covered 2.4 Mkm^2 of land (28.1% of the total study area). Over 40 years, these lands can potentially accumulate 8.5 GtC in above-ground biomass via low-cost natural regeneration or assisted regeneration, corresponding to a total CO_2 sequestration of 31.1 $GtCO_2$ (Chazdon et al. 2016b). While above-ground biomass carbon stocks are estimated to be declining in the tropics, they are increasing globally due to increasing stocks in temperate and boreal forests (Liu et al. 2015b), consistent with the observations of a global land sector carbon sink (Le Quéré et al. 2013; Keenan et al. 2017; Pan et al. 2011).

Moving from technical mitigation potentials (Miles et al. 2015) to real reduction of emissions from deforestation and forest degradation required transformational changes (Korhonen-Kurki et al. 2018). This transformation can be facilitated by two enabling conditions: the presence of already initiated policy change; or the scarcity of forest resources combined with an absence of any effective forestry framework and policies. These authors and others (Angelsen et al. 2018) found that the presence of powerful transformational coalitions of domestic pro-REDD+ (the United Nations Collaborative Programme on Reducing Emissions from Deforestation and Forest Degradation in Developing Countries) political actors combined with strong ownership and leadership, regulations and law enforcement, and performance-based funding, can provide a strong incentive for achieving REDD+ goals.

Implementing schemes such as REDD+ and various projects related to the voluntary carbon market is often regarded as a no-regrets investment (Seymour and Angelsen 2012) but the social and ecological implications (including those identified in the Cancun Safeguards) must be carefully considered for REDD+ projects to be socially and ecologically sustainable (Jagger et al. 2015). In 2018, 34 countries have submitted a REDD+ forest reference level and/or forest reference emission level to the United Nations Framework Convention on Climate Change (UNFCCC). Of these REDD+ reference levels, 95% included the activity 'reducing deforestation' while 34% included the activity 'reducing forest degradation' (FAO 2018). Five countries submitted REDD+ results in the technical annex to their Biennial Update Report totalling an emission reduction of 6.3 $GtCO_2$ between 2006 and 2015 (FAO 2018).

Afforestation is another mitigation activity that increases carbon sequestration (Cross-Chapter Box 2 in Chapter 1). Yet, it requires careful consideration about where to plant trees to achieve potential climatic benefits, given an altering of local albedo and turbulent energy fluxes and increasing night-time land surface temperatures (Peng et al. 2014). A recent hydro-climatic modelling effort has shown that forest cover can account for about 40% of the observed decrease in annual runoff (Buendia et al. 2016). A meta-analysis of afforestation in Northern Europe (Bárcena et al. 2014) concluded that significant soil organic carbon sequestration in Northern Europe occurs after afforestation of croplands but not grasslands. Additional sequestration occurs in forest floors and biomass carbon stocks. Successful programmes of large-scale afforestation activities in South Korea and China are discussed in-depth in a special case study (Section 4.9.3).

The potential outcome of efforts to reduce emissions from deforestation and degradation in Indonesia through a 2011 moratorium on concessions to convert primary forests to either timber or palm oil uses was evaluated against rates of emissions over the period 2000 to 2010. The study concluded that less than 7% of emissions would have been avoided had the moratorium been implemented in 2000 because it only curtailed emissions due to a subset of drivers of deforestation and degradation (Busch et al. 2015).

In terms of ecological integrity of tropical forests, the policy focus on carbon storage and tree cover can be problematic if it leaves out other aspects of forests ecosystems, such as biodiversity – and particularly fauna (Panfil and Harvey 2016; Peres et al. 2016; Hinsley et al. 2015). Other concerns of forest-based projects under the voluntary carbon market are potential negative socio-economic side effects (Edstedt and Carton 2018; Carton and Andersson 2017; Osborne 2011; Scheidel and Work 2018; Richards and Lyons 2016; Borras and Franco 2018; Paladino and Fiske 2017) and leakage (particularly at the subnational scale), that is, when interventions to reduce deforestation or degradation at one site displace pressures and increase emissions elsewhere (Atmadja and Verchot 2012; Phelps et al. 2010; Lund et al. 2017; Balooni and Lund 2014).

Maintaining and increasing forest area, in particular native forests rather than monoculture and short-rotation plantations, contributes to the maintenance of global forest carbon stocks (Lewis et al. 2019) (*robust evidence*, *high agreement*).

1.8.4 Sustainable forest management (SFM) and CO_2 removal (CDR) technologies

While reducing deforestation and forest degradation may directly help to meet mitigation goals, SFM aimed at providing timber, fibre, biomass and non-timber resources can provide long-term livelihood for communities, reduce the risk of forest conversion to non-forest uses (settlement, crops, etc.), and maintain land productivity, thus reducing the risks of land degradation (Putz et al. 2012; Gideon Neba et al. 2014; Sufo Kankeu et al. 2016; Nitcheu Tchiadje et al. 2016; Rossi et al. 2017).

Developing SFM strategies aimed at contributing towards negative emissions throughout this century requires an understanding of forest management impacts on ecosystem carbon stocks (including soils), carbon sinks, carbon fluxes in harvested wood, carbon storage in harvested wood products, including landfills and the emission reductions achieved through the use of wood products and bioenergy (Nabuurs et al. 2007; Lemprière et al. 2013; Kurz et al. 2016; Law et al. 2018; Nabuurs et al. 2017). Transitions from natural to managed forest landscapes can involve a reduction in forest carbon stocks, the magnitude of which depends on the initial landscape conditions, the harvest rotation length relative to the frequency and intensity of natural disturbances, and on the age-dependence of managed and natural disturbances (Harmon et al. 1990; Kurz et al. 1998). Initial landscape conditions, in particular the age-class distribution and therefore carbon stocks of the landscape, strongly affect the mitigation potential of forest management options (Ter-Mikaelian et al. 2013; Kilpeläinen et al. 2017). Landscapes with predominantly mature forests may experience larger reductions in carbon stocks during the transition to managed landscapes (Harmon et al. 1990; Kurz et al. 1998; Lewis et al. 2019). In landscapes with predominantly young or recently disturbed forests, SFM can enhance carbon stocks (Henttonen et al. 2017).

Forest growth rates, net primary productivity, and net ecosystem productivity are age-dependent, with maximum rates of CO_2 removal (CDR) from the atmosphere occurring in young to medium-aged forests and declining thereafter (Tang et al. 2014). In boreal forest ecosystem, estimation of carbon stocks and carbon fluxes indicate that old growth stands are typically small carbon sinks or carbon sources (Gao et al. 2018; Taylor et al. 2014; Hadden and Grelle 2016). In tropical forests, carbon uptake rates in the first 20 years of forest recovery were 11 times higher than uptake rates in old-growth forests (Poorter et al. 2016). Age-dependent increases in forest carbon stocks and declines in forest carbon sinks mean that landscapes with older forests have accumulated more carbon but their sink strength is diminishing, while landscapes with younger forests contain less carbon but they are removing CO_2 from the atmosphere at a much higher rate (Volkova et al. 2017; Poorter et al. 2016). The rates of CDR are not just age-related but also controlled by many biophysical factors and human activities (Bernal et al. 2018). In ecosystems with uneven-aged, multispecies forests, the relationships between carbon stocks and sinks are more difficult and expensive to quantify.

Whether or not forest harvest and use of biomass is contributing to net reductions of atmospheric carbon depends on carbon losses during and following harvest, rates of forest regrowth, and the use of harvested wood and carbon retention in long-lived or short-lived products, as well as the emission reductions achieved through the substitution of emissions-intensive products with wood products (Lempriѐre et al. 2013; Lundmark et al. 2014; Xu et al. 2018b; Olguin et al. 2018; Dugan et al. 2018; Chen et al. 2018b; Pingoud et al. 2018; Seidl et al. 2007). Studies that ignore changes in forest carbon stocks (such as some lifecycle analyses that assume no impacts of harvest on forest carbon stocks), ignore changes in wood product pools (Mackey et al. 2013) or assume long-term steady state (Pingoud et al. 2018), or ignore changes in emissions from substitution benefits (Mackey et al. 2013; Lewis et al. 2019) will arrive at diverging conclusions about the benefits of SFM. Moreover, assessments of climate benefits of any mitigation action must also consider the time dynamics of atmospheric impacts, as some actions will have immediate benefits (e.g., avoided deforestation), while others may not achieve net atmospheric benefits for decades or centuries. For example, the climate benefits of woody biomass use for bioenergy depend on several factors, such as the source and alternate fate of the biomass, the energy type it substitutes, and the rates of regrowth of the harvested forest (Laganière et al. 2017; Ter-Mikaelian et al. 2014; Smyth et al. 2017). Conversion of primary forests in regions of very low stand-replacing disturbances to short-rotation plantations where the harvested wood is used for short-lived products with low displacement factors will increase emissions. In general, greater mitigation benefits are achieved if harvested wood products are used for products with long carbon retention time and high displacement factors.

With increasing forest age, carbon sinks in forests will diminish until harvest or natural disturbances, such as wildfire, remove biomass carbon or release it to the atmosphere (Seidl et al. 2017). While individual trees can accumulate carbon for centuries (Köhl et al. 2017), stand-level carbon accumulation rates depend on both tree growth and tree mortality rates (Hember et al. 2016; Lewis et al. 2004). SFM, including harvest and forest regeneration, can help maintain active carbon sinks by maintaining a forest age-class distribution that includes a share of young, actively growing stands (Volkova et al.

2018; Nabuurs et al. 2017). The use of the harvested carbon in either long-lived wood products (e.g., for construction), short-lived wood products (e.g., pulp and paper), or biofuels affects the net carbon balance of the forest sector (Lemprière et al. 2013; Matthews et al. 2018). The use of these wood products can further contribute to GHG emission-reduction goals by avoiding the emissions from the products with higher embodied emissions that have been displaced (Nabuurs et al. 2007; Lemprière et al. 2013). In 2007 the IPCC concluded that '[i]n the long term, a sustainable forest management strategy aimed at maintaining or increasing forest carbon stocks, while producing an annual sustained yield of timber, fibre or energy from the forest, will generate the largest sustained mitigation benefit' (Nabuurs et al. 2007). The apparent trade-offs between maximising forest carbon stocks and maximising ecosystem carbon sinks are at the origin of ongoing debates about optimum management strategies to achieve negative emissions (Keith et al. 2014; Kurz et al. 2016; Lundmark et al. 2014). SFM, including the intensification of carbon-focused management strategies, can make long-term contributions towards negative emissions if the sustainability of management is assured through appropriate governance, monitoring and enforcement. As specified in the definition of SFM, other criteria such as biodiversity must also be considered when assessing mitigation outcomes (Lecina-Diaz et al. 2018). Moreover, the impacts of changes in management on albedo and other non-GHG factors also need to be considered (Luyssaert et al. 2018) (Chapter 2). The contribution of SFM for negative emissions is strongly affected by the use of the wood products derived from forest harvest and the time horizon over which the carbon balance is assessed. SFM needs to anticipate the impacts of climate change on future tree growth, mortality and disturbances when designing climate change mitigation and adaptation strategies (Valade et al. 2017; Seidl et al. 2017).

1.8.5 Policy responses to land degradation

The 1992 United Nations Conference on Environment and Development (UNCED), also known as the Rio de Janeiro Earth Summit, recognised land degradation as a major challenge to sustainable development, and led to the establishment of the UNCCD, which specifically addressed land degradation in the drylands. The UNCCD emphasises sustainable land use to link poverty reduction on one hand and environmental protection on the other. The two other 'Rio Conventions' emerging from the UNCED – the UNFCCC and the Convention on Biological Diversity (CBD) – focus on climate change and biodiversity, respectively. The land has been recognised as an aspect of common interest to the three conventions, and SLM is proposed as a unifying theme for current global efforts on combating land degradation, climate change and loss of biodiversity, as well as facilitating land-based adaptation to climate change and sustainable development.

The Global Environmental Facility (GEF) funds developing countries to undertake activities that meet the goals of the conventions and deliver global environmental benefits. Since 2002, the GEF has invested in projects that support SLM through its Land Degradation Focal Area Strategy, to address land degradation within and beyond the drylands.

Under the UNFCCC, parties have devised National Adaptation Plans (NAPs) that identify medium- and long-term adaptation needs. Parties have also developed their climate change mitigation plans, presented as NDCs. These programmes have the potential of assisting the promotion of SLM. It is understood that the root causes of land degradation and successful adaptation will not be realised until holistic solutions to land management are explored. SLM can help address root causes of low productivity, land degradation, loss of income-generating capacity, as well as contribute to the amelioration of the adverse effects of climate change.

The '4 per 1000' (4p1000) initiative (Soussana et al. 2019) launched by France during the UNFCCC COP21 in 2015 aims at capturing CO_2 from the atmosphere through changes to agricultural and forestry practices at a rate that would increase the carbon content of soils by 0.4% per year (Rumpel et al. 2018). If global soil carbon content increases at this rate in the top 30–40 cm, the annual increase in atmospheric CO_2 would be stopped (Dignac et al. 2017). This is an illustration of how extremely important soils are for addressing climate change. The initiative is based on eight steps: stop carbon loss (priority #1 is peat soils); promote carbon uptake; monitor, report and verify impacts; deploy technology for tracking soil carbon; test strategies for implementation and upscaling; involve communities; coordinate policies; and provide support (Rumpel et al. 2018). Questions remain, however, about the extent that the 4p1000 is achievable as a universal goal (van Groenigen et al. 2017; Poulton et al. 2018; Schlesinger and Amundson 2018).

LDN was introduced by the UNCCD at Rio +20, and adopted at UNCCD COP12 (UNCCD 2016a). LDN is defined as 'a state whereby the amount and quality of land resources necessary to support ecosystem functions and services and enhance food security remain stable or increase within specified temporal and spatial scales and ecosystems'(Cowie et al. 2018). Pursuit of LDN requires effort to avoid further net loss of the land-based natural capital relative to a reference state, or baseline. LDN encourages a dual-pronged effort involving SLM to reduce the risk of land degradation, combined with efforts in land restoration and rehabilitation, to maintain or enhance land-based natural capital, and its associated ecosystem services (Orr et al. 2017; Cowie et al. 2018). Planning for LDN involves projecting the expected cumulative impacts of land-use and land management decisions, then counterbalancing anticipated losses with measures to achieve equivalent gains, within individual land types (where land type is defined by land potential). Under the LDN framework developed by UNCCD, three primary indicators are used to assess whether LDN is achieved by 2030: land cover change; net primary productivity; and soil organic carbon (Cowie et al. 2018; Sims et al. 2019). Achieving LDN therefore requires integrated landscape management that seeks to optimise land use to meet multiple objectives (ecosystem health, food security, human well-being) (Cohen-Shacham et al. 2016). The response hierarchy of Avoid > Reduce > Reverse land degradation articulates the priorities in planning LDN interventions. LDN provides the impetus for widespread adoption of SLM and efforts to restore or rehabilitate land. Through its focus, LDN ultimately provides tremendous potential for mitigation of, and adaptation to, climate change by halting and reversing land degradation and transforming land from a carbon source to a sink. There are strong synergies between the concept of LDN and the NDCs of many countries, with linkages to national climate plans. LDN is also closely related to many Sustainable Development Goals (SDG) in the areas of poverty, food security, environmental protection and sustainable use of natural resources (UNCCD 2016b). The GEF is supporting countries to set LDN targets and implement their LDN plans through its land degradation focal area, which encourages application of integrated landscape approaches to managing land degradation (GEF 2018).

The 2030 Agenda for Sustainable Development, adopted by the United Nations in 2015, comprises 17 SDGs. Goal 15 is of direct relevance to land degradation, with the objective to protect, restore and promote sustainable use of terrestrial ecosystems, sustainably manage forests, combat desertification and halt and reverse land degradation and halt biodiversity loss. Target 15.3 specifically addresses LDN. Other goals that are relevant for land degradation include Goal 2 (Zero hunger), Goal 3 (Good health and well-being), Goal 7 (Affordable and clean energy), Goal 11 (Sustainable cities and communities), and Goal 12 (Responsible production and consumption). Sustainable management of land resources underpins the SDGs related to hunger, climate change and environment. Further goals of a cross-cutting nature include 1 (No poverty), 6 (Clean water and sanitation) and 13 (Climate action). It remains to be seen how these interconnections are dealt with in practice.

With a focus on biodiversity, IPBES published a comprehensive assessment of land degradation in 2018 (Montanarella et al. 2018). The IPBES report, together with this report focusing on climate change, may contribute to creating a synergy between the two main global challenges for addressing land degradation in order to help achieve the targets of SDG 15 (protect, restore and promote sustainable use of terrestrial ecosystems, sustainably manage forests, combat desertification, and halt and reverse land degradation and halt biodiversity loss).

Market-based mechanisms like the Clean Development Mechanism (CDM) under the UNFCCC and the voluntary carbon market provide incentives to enhance carbon sinks on the land through afforestation and reforestation. Implications for local land use and food security have been raised as a concern and need to be assessed (Edstedt and Carton 2018; Olsson et al. 2014b). Many projects aimed at reducing emissions from deforestation and forest degradations (not to be confused with the national REDD+ programmes in accordance with the UNFCCC Warsaw Framework) are being planned and implemented to primarily target countries with high forest cover and high deforestation rates. Some parameters of incentivising emissions reduction, quality of forest governance, conservation priorities, local rights and tenure frameworks, and sub-national project potential are being looked into, with often very mixed results (Newton et al. 2016; Gebara and Agrawal 2017).

Besides international public initiatives, some actors in the private sector are increasingly aware of the negative environmental impacts of some global value chains producing food, fibre, and energy products (Lambin et al. 2018; van der Ven and Cashore 2018; van der Ven et al. 2018; Lyons-White and Knight 2018). While improvements are underway in many supply chains, measures implemented so far are often insufficient to be effective in reducing or stopping deforestation and forest degradation (Lambin et al. 2018). The GEF is investing in actions to reduce deforestation in commodity supply chains through its Food Systems, Land Use, and Restoration Impact Program (GEF 2018).

1.8.5.1 Limits to adaptation

SLM can be deployed as a powerful adaptation strategy in most instances of climate change impacts on natural and social systems, yet there are limits to adaptation (Klein et al. 2014; Dow, Berhout and Preston 2013). Such limits are dynamic and interact with social and institutional conditions (Barnett et al. 2015; Filho and Nalau 2018). Exceeding adaptation limits will trigger escalating losses or require undesirable transformational change, such as forced migration. The rate of change in relation to the rate of possible adaptation is crucial (Dow et al. 2013). How limits to adaptation are defined, and how they can be measured, is contextual and contested. Limits must be assessed in relation to the ultimate goals of adaptation, which is subject to diverse and differential values (Dow et al. 2013; Adger et al. 2009). A particularly sensitive issue is whether migration is accepted as adaptation or not (Black et al. 2011; Tacoli 2009; Bardsley and Hugo 2010). If migration were understood and accepted as a form of successful adaptation, it would change the limits to adaptation by reducing, or even avoiding, future humanitarian crises caused by climate extremes (Adger et al. 2009; Upadhyay et al. 2017; Nalau et al. 2018).

In the context of land degradation, potential limits to adaptation exist if land degradation becomes so severe and irreversible that livelihoods cannot be maintained, and if migration is either not acceptable or not possible. Examples are coastal erosion where land disappears (Gharbaoui and Blocher 2016; Luetz 2018), collapsing livelihoods due to thawing of permafrost (Landauer and Juhola 2019), and extreme forms of soil erosion, (e.g., landslides (Van der Geest and Schindler 2016) and gully erosion leading to badlands (Poesen et al. 2003)).

1.8.6 Resilience and thresholds

Resilience refers to the capacity of interconnected social, economic and ecological systems, such as farming systems, to absorb disturbance (e.g., drought, conflict, market collapse), and respond or reorganise, to maintain their essential function, identity and structure. Resilience can be described as 'coping capacity'. The disturbance may be a shock – sudden events such as a flood or disease epidemic – or it may be a trend that develops slowly, like a drought or market shift. The shocks and trends anticipated to occur due to climate change are expected to exacerbate risk of land degradation. Therefore, assessing and enhancing resilience to climate change is a critical component of designing SLM strategies.

Resilience as an analytical lens is particularly strong in ecology and related research on natural resource management (Folke et al. 2010; Quinlan et al. 2016) while, in the social sciences, the relevance of resilience for studying social and ecological interactions is contested

(Cote and Nightingale 2012; Olsson et al. 2015; Cretney 2014; Béné et al. 2012; Joseph 2013). In the case of adaptation to climate change (and particularly regarding limits to adaptation), a crucial ambiguity of resilience is the question of whether resilience is a normative concept (i.e., resilience is good or bad) or a descriptive characteristic of a system (i.e., neither good nor bad). Previous IPCC reports have defined resilience as a normative (positive) attribute (see AR5 Glossary), while the wider scientific literature is divided on this (Weichselgartner and Kelman 2015; Strunz 2012; Brown 2014; Grimm and Calabrese 2011; Thorén and Olsson 2018). For example, is outmigration from a disaster-prone area considered a successful adaptation (high resilience) or a collapse of the livelihood system (lack of resilience) (Thorén and Olsson 2018)? In this report, resilience is considered a positive attribute when it maintains capacity for adaptation, learning and/or transformation.

Furthermore, 'resilience' and the related terms 'adaptation' and 'transformation' are defined and used differently by different communities (Quinlan et al. 2016). The relationship and hierarchy of resilience with respect to vulnerability and adaptive capacity are also debated, with different perspectives between disaster management and global change communities, (e.g., Cutter et al. 2008). Nevertheless, these differences in usage need not inhibit the application of 'resilience thinking' in managing land degradation; researchers using these terms, despite variation in definitions, apply the same fundamental concepts to inform management of human-environment systems, to maintain or improve the resource base, and sustain livelihoods.

Applying resilience concepts involves viewing the land as a component of an interlinked social-ecological system; identifying key relationships that determine system function and vulnerabilities of the system; identifying thresholds or tipping points beyond which the system transitions to an undesirable state; and devising management strategies to steer away from thresholds of potential concern, thus facilitating healthy systems and sustainable production (Walker et al. 2009).

A threshold is a non-linearity between a controlling variable and system function, such that a small change in the variable causes the system to shift to an alternative state. Bestelmeyer et al. (2015) and Prince et al. (2018) illustrate this concept in the context of land degradation. Studies have identified various biophysical and socio-economic thresholds in different land-use systems. For example, 50% ground cover (living and dead plant material and biological crusts) is a recognised threshold for dryland grazing systems (e.g., Tighe et al. 2012); below this threshold, the infiltration rate declines, risk of erosion causing loss of topsoil increases, a switch from perennial to annual grass species occurs and there is a consequential sharp decline in productivity. This shift to a lower-productivity state cannot be reversed without significant human intervention. Similarly, the combined pressure of water limitations and frequent fire can lead to transition from closed forest to savannah or grassland: if fire is too frequent, trees do not reach reproductive maturity and post-fire regeneration will fail; likewise, reduced rainfall/increased drought prevents successful forest regeneration (Reyer et al. 2015; Thompson et al. 2009) (Cross-Chapter Box 3 in Chapter 2).

In managing land degradation, it is important to assess the resilience of the existing system, and the proposed management interventions. If the existing system is in an undesirable state or considered unviable under expected climate trends, it may be desirable to promote adaptation or even transformation to a different system that is more resilient to future changes. For example, in an irrigation district where water shortages are predicted, measures could be implemented to improve water use efficiency, for example, by establishing drip irrigation systems for water delivery, although transformation to pastoralism or mixed dryland cropping/livestock production may be more sustainable in the longer term, at least for part of the area. Application of SLM practices, especially those focused on ecological functions (e.g., agroecology, ecosystem-based approaches, regenerative agriculture, organic farming), can be effective in building resilience of agro-ecosystems (Henry et al. 2018). Similarly, the resilience of managed forests can be enhanced by SFM that protects or enhances biodiversity, including assisted migration of tree species within their current range limit (Winder et al. 2011; Pedlar et al. 2012) or increasing species diversity in plantation forests (Felton et al. 2010; Liu et al. 2018a). The essential features of a resilience approach to management of land degradation under climate change are described by O'Connell et al. (2016) and Simonsen et al. (2014).

Consideration of resilience can enhance effectiveness of interventions to reduce or reverse land degradation (*medium agreement, limited evidence*). This approach will increase the likelihood that SLM/SFM and land restoration/rehabilitation interventions achieve long-term environmental and social benefits. Thus, consideration of resilience concepts can enhance the capacity of land systems to cope with climate change and resist land degradation, and assist land-use systems to adapt to climate change.

1.8.7 Barriers to implementation of sustainable land management (SLM)

There is a growing recognition that addressing barriers and designing solutions to complex environmental problems, such as land degradation, requires awareness of the larger system into which the problems and solutions are embedded (Laniak et al. 2013). An ecosystem approach to sustainable land management (SLM) based on an understanding of land degradation processes has been recommended to separate multiple drivers, pressures and impacts (Kassam et al. 2013), but large uncertainty in model projections of future climate, and associated ecosystem processes (IPCC 2013a) pose additional challenges to the implementation of SLM. As discussed earlier in this chapter, many SLM practices, including technologies and approaches, are available that can increase yields and contribute to closing the yield gap between actual and potential crop or pasture yield, while also enhancing resilience to climate change (Yengoh and Ardö 2014; WOCAT n.d.). However, there are often systemic barriers to adoption and scaling up of SLM practices, especially in developing countries.

Uitto (2016) identified areas that the GEF, the financial mechanism of the UNCCD, UNFCCC and other multilateral environmental agreements, can address to solve global environmental problems. These include:

removal of barriers related to knowledge and information; strategies for implementation of technologies and approaches; and institutional capacity. Strengthening these areas would drive transformational change, leading to behavioural change and broader adoption of sustainable environmental practices. Detailed analysis of barriers as well as strategies, methods and approaches to scale up SLM have been undertaken for GEF programmes in Africa, China and globally (Tengberg and Valencia 2018; Liniger et al. 2011; Tengberg et al. 2016). A number of interconnected barriers and bottlenecks to the scaling up of SLM have been identified in this context and are related to:

- limited access to knowledge and information, including new SLM technologies and problem-solving capacities
- weak enabling environment, including the policy, institutional and legal framework for SLM, and land tenure and property rights
- inadequate learning and adaptive knowledge management in the project cycle, including monitoring and evaluation of impacts
- limited access to finance for scaling up, including public and private funding, innovative business models for SLM technologies and financial mechanisms and incentives, such as payment for ecosystem services (PES), insurance and micro-credit schemes (see also Shames et al. 2014).

Adoption of innovations and new technologies are increasingly analysed using the transition theory framework (Geels 2002), the starting point being the recognition that many global environmental problems cannot be solved by technological change alone, but require more far-reaching change of social-ecological systems. Using transition theory makes it possible to analyse how adoption and implementation follow the four stages of sociotechnical transitions, from predevelopment of technologies and approaches at the niche level, take-off and acceleration, to regime shift and stabilisation at the landscape level. According to a recent review of sustainability transitions in developing countries (Wieczorek 2018), three internal niche processes are important, including the formation of networks that support and nurture innovation, the learning process, and the articulation of expectations to guide the learning process. While technologies are important, institutional and political aspects form the major barriers to transition and upscaling. In developing and transition economies, informal institutions play a pivotal role, and transnational linkages are also important, such as global value chains. In these countries, it is therefore more difficult to establish fully coherent regimes or groups of individuals who share expectations, beliefs or behaviour, as there is a high level of uncertainty about rules and social networks or dominance of informal institutions, which creates barriers to change. This uncertainty is further exacerbated by climate change. Landscape forces comprise a set of slow-changing factors, such as broad cultural and normative values, long-term economic effects such as urbanisation, and shocks such as war and crises that can lead to change.

A study on SLM in the Kenyan highlands using transition theory concluded that barriers to adoption of SLM included high poverty levels, a low-input/low-output farming system with limited potential to generate income, diminishing land sizes, and low involvement of the youth in farming activities. Coupled with a poor coordination of government policies for agriculture and forestry, these barriers created negative feedbacks in the SLM transition process. Other factors to consider include gender issues and lack of secure land tenure. Scaling up of SLM technologies would require collaboration of

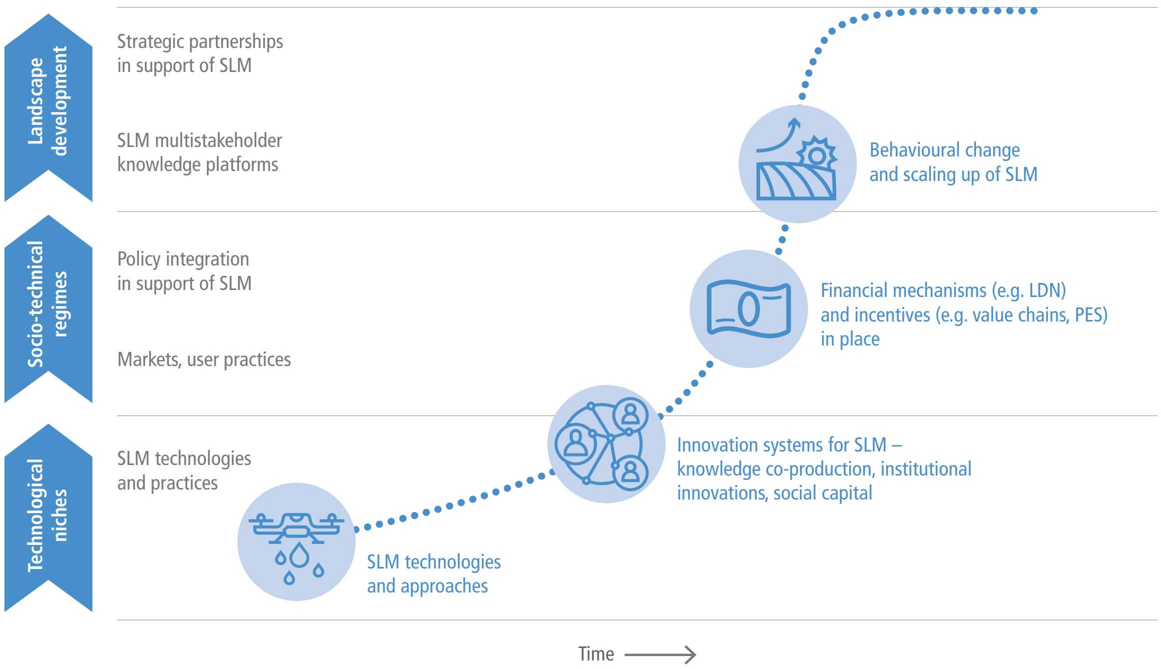

Figure 4.7 | The transition from SLM niche adoption to regime shift and landscape development. Figure draws inspiration from Geels (2002), adapted from Tengberg and Valencia (2018).

diverse stakeholders across multiple scales, a more supportive policy environment and substantial resource mobilisation (Mutoko et al. 2014). Tengberg and Valencia (2018) analysed the findings from a review of the GEF's integrated natural resources management portfolio of projects using the transition theory framework (Figure 4.7). They concluded that to remove barriers to SLM, an agricultural innovations systems approach that supports co-production of knowledge with multiple stakeholders, institutional innovations, a focus on value chains and strengthening of social capital to facilitate shared learning and collaboration could accelerate the scaling up of sustainable technologies and practices from the niche to the landscape level. Policy integration and establishment of financial mechanisms and incentives could contribute to overcoming barriers to a regime shift. The new SLM regime could, in turn, be stabilised and sustained at the landscape level by multi-stakeholder knowledge platforms and strategic partnerships. However, transitions to more sustainable regimes and practices are often challenged by lock-in mechanisms in the current system (Lawhon and Murphy 2012) such as economies of scale, investments already made in equipment, infrastructure and competencies, lobbying, shared beliefs, and practices, that could hamper wider adoption of SLM.

Adaptive, multi-level and participatory governance of social-ecological systems is considered important for regime shifts and transitions to take place (Wieczorek 2018) and essential to secure the capacity of environmental assets to support societal development over longer time periods (Folke et al. 2005). There is also recognition that effective environmental policies and programmes need to be informed by a comprehensive understanding of the biophysical, social, and economic components and processes of a system, their complex interactions, and how they respond to different changes (Kelly (Letcher) et al. 2013). But blueprint policies will not work, due to the wide diversity of rules and informal institutions used across sectors and regions of the world, especially in traditional societies (Ostrom 2009).

The most effective way of removing barriers to funding of SLM has been mainstreaming of SLM objectives and priorities into relevant policy and development frameworks, and combining SLM best practices with economic incentives for land users. As the short-term costs for establishing and maintaining SLM measures are generally high and constitute a barrier to adoption, land users may need to be compensated for generation of longer-term public goods, such as ecosystem services. Cost-benefit analyses can be conducted on SLM interventions to facilitate such compensations (Liniger et al. 2011; Nkonya et al. 2016; Tengberg et al. 2016). The landscape approach is a means to reconcile competing demands on the land and remove barriers to implementation of SLM (e.g., Sayer et al. 2013; Bürgi et al. 2017). It involves an increased focus on participatory governance, development of new SLM business models, and innovative funding schemes, including insurance (Shames et al. 2014). The LDN Fund takes a landscape approach and raises private finance for SLM and promotes market-based instruments, such as PES, certification and carbon trading, that can support scaling up of SLM to improve local livelihoods, sequester carbon and enhance the resilience to climate change (Baumber et al. 2019).

1.9 Case studies

Climate change impacts on land degradation can be avoided, reduced or even reversed, but need to be addressed in a context-sensitive manner. Many of the responses described in this section can also provide synergies of adaptation and mitigation. In this section we provide more in-depth analysis of a number of salient aspects of how land degradation and climate change interact. Table 4.3 is a synthesis of how of these case studies relate to climate change and other broader issues in terms of co-benefits.

1.9.1 Urban green infrastructure

Over half of the world's population now lives in towns and cities, a proportion that is predicted to increase to about 70% by the middle of the century (United Nations 2015). Rapid urbanisation is a severe threat to land and the provision of ecosystem services (Seto et al. 2012). However, as cities expand, the avoidance of land degradation, or the maintenance/enhancement of ecosystem services is rarely considered in planning processes. Instead, economic development and the need for space for construction is prioritised, which can result in substantial pollution of air and water sources, the degradation of existing agricultural areas and indigenous, natural or semi-natural ecosystems both within and outside of urban areas. For instance, urban areas are characterised by extensive impervious surfaces. Degraded, sealed soils beneath these surfaces do not provide the same quality of water retention as intact soils. Urban landscapes comprising 50–90% impervious surfaces can therefore result in 40–83% of rainfall becoming surface water runoff (Pataki et al. 2011). With rainfall intensity predicted to increase in many parts of the world under climate change (Royal Society 2016), increased water runoff is going to get worse. Urbanisation, land degradation and climate change are therefore strongly interlinked, suggesting the need for common solutions (Reed and Stringer 2016).

There is now a large body of research and application demonstrating the importance of retaining urban green infrastructure (UGI) for the delivery of multiple ecosystem services (DG Environment News Alert Service, 2012; Wentworth, 2017) as an important tool to mitigate and adapt to climate change. UGI can be defined as all green elements within a city, including, but not limited to, retained indigenous ecosystems, parks, public greenspaces, green corridors, street trees, urban forests, urban agriculture, green roofs/walls and private domestic gardens (Tzoulas et al. 2007). The definition is usually extended to include 'blue' infrastructure, such as rivers, lakes, bioswales and other water drainage features. The related concept of Nature-Based Solutions (defined as: *living solutions inspired by, continuously supported by and using nature, which are designed to address various societal challenges in a resource-efficient and adaptable manner and to provide simultaneously economic, social, and environmental benefits*) has gained considerable traction within the European Commission as one approach to mainstreaming the importance of UGI (Maes and Jacobs 2017; European Union 2015).

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Table 4.3 | Synthesis of how the case studies interact with climate change and a broader set of co-benefits.

Case studies (4.9)	Mitigation benefits and potential	Adaptation benefits	Co-benefits
Urban green infrastructure (4.9.1) An increasing majority of the world population live in cities and land degradation is an urgent matter for urban areas	↓	≋↓	human health, recreation
Perennial grains (4.9.2) After 40 years of breeding, perennial grains now seem to have the potential of reducing climate impacts of agriculture while increasing its overall sustainability	↓ ↑	≋↓	reduced use of herbicides, reduced soil erosion and nutrient leakage
Reforestation (4.9.3) Two cases of successful reforestation serve as illustrations of the potential of sustained efforts into reforestation	↓	≋↓	economic return from sustainable forestry, reduced flood risk downstream
Management of peat soils (4.9.4) Degradation of peat soils in tropical and Arctic regions is a major source of greenhouse gases, hence an urgent mitigation option	↑		improved air quality in tropical regions
Biochar (4.9.5) Biochar is a land-management technique of high potential, but controversial	↓ ↑		improved soil fertility
Protection against hurricane damages (4.9.6) More severe tropical cyclones increase the risk of land degradation in some areas, hence the need for increased adaptation		≋↓	reduced losses (human lives, livelihoods, and assets)
Responses to saltwater intrusion (4.9.7) The combined effect of climate-induced sea level rise and land-use change in coastal regions increases the risk of saltwater intrusion in many coastal regions		≋	improved food and water security
Avoiding coastal maladaptation (4.9.8) Low-lying coastal areas are in urgent need of adaptation, but examples have resulted in maladaptation		≋	reduced losses (human lives, livelihoods, and assets)

Legend	
↓	carbon sink
↑	reduced emission
≋↓	reduced flood risk
	reduced heat stress
	drought resistance
	storm protection
≋	protection against sea level rise

Through retaining existing vegetation and ecosystems, revegetating previous developed land or integrating vegetation into buildings in the form of green walls and roofs, UGI can play a direct role in mitigating climate change through carbon sequestration. However, compared to overall carbon emissions from cities, effects will be small. Given that UGI necessarily involves the retention and management of non-sealed surfaces, co-benefits for land degradation (e.g., soil compaction avoidance, reduced water runoff, carbon storage and vegetation productivity (Davies et al. 2011; Edmondson et al. 2011, 2014; Yao et al. 2015) will also be apparent. Although not currently a priority, its role in mitigating land degradation could be substantial. For instance, appropriately managed innovative urban agricultural production systems, such as vertical farms, could have the potential to meet some of the food needs of cities and reduce the production (and therefore degradation) pressure on agricultural land in rural areas, although thus far this is unproven (for a recent review, see Wilhelm and Smith 2018).

The importance of UGI as part of a climate change adaptation approach has received greater attention and application (Gill et al. 2007; Fryd et al. 2011; Demuzere et al. 2014; Sussams et al. 2015). The EU's Adapting to Climate Change white paper emphasises the 'crucial role in adaptation in providing essential resources for social and economic purposes under extreme climate conditions' (CEC, 2009, p. 9). Increasing vegetation cover, planting street trees and maintaining/expanding public parks reduces temperatures (Cavan et al. 2014; Di Leo et al. 2016; Feyisa et al. 2014; Tonosaki and Kawai 2014; Zölch et al. 2016). Further, the appropriate design and spatial distribution of greenspaces within cities can help to alter urban climates to improve human health and comfort (e.g., Brown and Nicholls 2015; Klemm et al. 2015). The use of green walls and roofs can also reduce energy use in buildings (e.g., Coma et al. 2017). Similarly, natural flood management and ecosystem-based approaches of providing space for water, renaturalising rivers and reducing surface runoff through the presence of permeable surfaces and vegetated features (including walls and roofs) can manage flood risks, impacts and vulnerability (e.g., Gill et al. 2007; Munang et al. 2013). Access to UGI in times of environmental stresses and shock can provide safety nets for people, and so can be an important adaptation mechanism, both to climate change (Potschin et al. 2016) and land degradation.

Most examples of UGI implementation as a climate change adaptation strategy have centred on its role in water management for flood risk reduction. The importance for land degradation is either not stated, or not prioritised. In Beira, Mozambique, the government is using UGI to mitigate increased flood risks predicted to occur under climate change and urbanisation, which will be done by improving the natural water capacity of the Chiveve River. As part of the UGI approach, mangrove habitats have been restored, and future phases include developing new multi-functional urban green spaces along the river (World Bank 2016). The retention of green spaces within the city will have the added benefit of halting further degradation in those areas. Elsewhere, planning mechanisms promote the retention and expansion of green areas within cities to ensure ecosystem service delivery, which directly halts land degradation, but are largely viewed and justified in the context of climate change adaptation and mitigation. For instance, the Berlin Landscape Programme includes five plans, one of which covers adapting to climate change through the recognition of the role of UGI (Green Surge 2016). Major climate-related challenges facing Durban, South Africa, include sea level rise, urban heat island, water runoff and conservation (Roberts and O'Donoghue 2013). Now considered a global leader in climate adaptation planning (Roberts 2010), Durban's Climate Change Adaptation plan includes the retention and maintenance of natural ecosystems, in particular those that are important for mitigating flooding, coastal erosion, water pollution, wetland siltation and climate change (eThekwini Municipal Council 2014).

1.9.2 Perennial grains and soil organic carbon (SOC)

The severe ecological perturbation that is inherent in the conversion of native perennial vegetation to annual crops, and the subsequent high frequency of perturbation required to maintain annual crops, results in at least four forms of soil degradation that will be exacerbated by the effects of climate change (Crews et al. 2016). First, soil erosion is a very serious consequence of annual cropping, with median losses exceeding rates of formation by one to two orders of magnitude in conventionally plowed agroecosystems, and while erosion is reduced with conservation tillage, median losses still exceed formation by several fold (Montgomery 2007). More severe storm intensity associated with climate change is expected to cause even greater losses to wind and water erosion (Nearing et al. 2004). Second, the periods of time in which live roots are reduced or altogether absent from soils in annual cropping systems allow for substantial losses of nitrogen from fertilised croplands, averaging 50% globally (Ladha et al. 2005). This low retention of nitrogen is also expected to worsen with more intense weather events (Bowles et al. 2018). A third impact of annual cropping is the degradation of soil structure caused by tillage, which can reduce infiltration of precipitation, and increase surface runoff. It is predicted that the percentage of precipitation that infiltrates into agricultural soils will decrease further under climate-change scenarios (Basche and DeLonge 2017; Wuest et al. 2006). The fourth form of soil degradation that results from annual cropping is the reduction of soil organic matter (SOM), a topic of particular relevance to climate change mitigation and adaptation.

Undegraded cropland soils can theoretically hold far more SOM (which is about 58% carbon) than they currently do (Soussana et al. 2006). We know this deficiency because, with few exceptions, comparisons between cropland soils and those of proximate mature native ecosystems commonly show a 40–75% decline in soil carbon attributable to agricultural practices. What happens when native ecosystems are converted to agriculture that induces such significant losses of SOM? Wind and water erosion commonly results in preferential removal of light organic matter fractions that can accumulate on or near the soil surface (Lal 2003). In addition to the effects of erosion, the fundamental practices of growing annual food and fibre crops alters both inputs and outputs of organic matter from most agroecosystems, resulting in net reductions in soil carbon equilibria (Soussana et al. 2006; McLauchlan 2006; Crews et al. 2016). Native vegetation of almost all terrestrial ecosystems is dominated by perennial plants, and the below-ground carbon allocation of these perennials is a key variable in determining formation rates of stable soil organic carbon (SOC) (Jastrow et al. 2007; Schmidt et al. 2011). When perennial vegetation is replaced by annual crops, inputs of root-associated carbon (roots, exudates, mycorrhizae) decline substantially. For example, perennial grassland species allocate around 67% of productivity to roots, whereas annual crops allocate between 13–30% (Saugier 2001; Johnson et al. 2006).

At the same time, inputs of SOC are reduced in annual cropping systems, and losses are increased because of tillage, compared to native perennial vegetation. Tillage breaks apart soil aggregates which, among other functions, are thought to inhibit soil bacteria, fungi and other microbes from consuming and decomposing SOM (Grandy and Neff 2008). Aggregates reduce microbial access to organic matter by restricting physical access to mineral-stabilised organic compounds as well as reducing oxygen availability (Cotrufo et al. 2015; Lehmann and Kleber 2015). When soil aggregates are broken open with tillage in the conversion of native ecosystems to agriculture, microbial consumption of SOC and subsequent respiration of CO_2 increase dramatically, reducing soil carbon stocks (Grandy and Robertson 2006; Grandy and Neff 2008).

Many management approaches are being evaluated to reduce soil degradation in general, especially by increasing mineral-protected forms of SOC in the world's croplands (Paustian et al. 2016). The menu of approaches being investigated focuses either on increasing below-ground carbon inputs, usually through increases in total crop productivity, or by decreasing microbial activity, usually through reduced soil disturbance (Crews and Rumsey 2017). However, the basic biogeochemistry of terrestrial ecosystems managed for production of annual crops presents serious challenges to achieving the standing stocks of SOC accumulated by native ecosystems that preceded agriculture. A novel new approach that is just starting to receive significant attention is the development of perennial cereal, legume and oilseed crops (Glover et al. 2010; Baker 2017).

There are two basic strategies that plant breeders and geneticists are using to develop new perennial grain crop species. The first involves making wide hybrid crosses between existing elite lines of annual crops, such as wheat, sorghum and rice, with related wild perennial species in order to introgress perennialism into the genome of the annual (Cox et al. 2018; Huang et al. 2018; Hayes et al. 2018). The other approach is *de novo* domestication of wild perennial species that have crop-like traits of interest (DeHaan et al. 2016; DeHaan and Van Tassel 2014). New perennial crop species undergoing *de novo* domestication include intermediate wheatgrass, a relative of wheat that produces grain known as Kernza (DeHaan et al. 2018; Cattani and Asselin 2018) and *Silphium integrifolium*, an oilseed crop in the sunflower family (Van Tassel et al. 2017). Other grain crops receiving attention for perennialisation include pigeon pea, barley, buckwheat and maize (Batello et al. 2014; Chen et al. 2018c) and a number of legume species (Schlautman et al. 2018). In most cases, the seed yields of perennial grain crops under development are well below those of elite modern grain varieties. During the period that it will take for intensive breeding efforts to close the yield and other trait gaps between annual and perennial grains, perennial

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proto-crops may be used for purposes other than grain, including forage production (Ryan et al. 2018). Perennial rice stands out as a high-yielding exception, as its yields matched those of elite local varieties in the Yunnan Province for six growing seasons over three years (Huang et al. 2018).

In a perennial agroecosystem, the biogeochemical controls on SOC accumulation shift dramatically, and begin to resemble the controls that govern native ecosystems (Crews et al. 2016). When erosion is reduced or halted, and crop allocation to roots increases by 100–200%, and when soil aggregates are not disturbed thus reducing microbial respiration, SOC levels are expected to increase (Crews and Rumsey 2017). Deep roots growing year round are also effective at increasing nitrogen retention (Culman et al. 2013; Jungers et al. 2019). Substantial increases in SOC have been measured where croplands that had historically been planted to annual grains were converted to perennial grasses, such as in the US Conservation Reserve Program or in plantings of second-generation perennial biofuel crops. Two studies have assessed carbon accumulation in soils when croplands were converted to the perennial grain Kernza. In one, researchers found no differences in soil labile (permanganate-oxidisable) carbon after four years of cropping to perennial Kernza versus annual wheat in a sandy textured soil. Given that coarse textured soils do not offer the same physicochemical protection against microbial attack as many finer textured soils, these results are not surprising, but these results do underscore how variable the rates of carbon accumulation can be (Jastrow et al. 2007). In the second study, researchers assessed the carbon balance of a Kernza field in Kansas, USA over 4.5 years using eddy covariance observations (de Oliveira et al. 2018). They found that the net carbon accumulation rate of about 1500 gC m^{-2} yr^{-1} in the first year of the study corresponding to the biomass of Kernza, increasing to about 300 gC m^{-2} yr^{-1} in the final year, where CO_2 respiration losses from the decomposition of roots and SOM approached new carbon inputs from photosynthesis. Based on measurements of soil carbon accumulation in restored grasslands in this part of the USA, the net carbon accumulation in stable organic matter under a perennial grain crop might be expected to sequester 30–50 gC m^{-2} yr^{-1} (Post and Kwon 2000) until a new equilibrium is reached. Sugar cane, a highly productive perennial, has been shown to accumulate a mean of 187 gC m^{-2} yr^{-1} in Brazil (La Scala Júnior et al. 2012).

Reduced soil erosion, increased nitrogen retention, greater water uptake efficiency and enhanced carbon sequestration represent improved ecosystem functions, made possible in part by deep and extensive root systems of perennial crops (Figure 4.8).

When compared to annual grains like wheat, single species stands of deep-rooted perennial grains such as Kernza are expected to reduce soil erosion, increase nitrogen retention, achieve greater water uptake efficiency and enhance carbon sequestration (Crews et al. 2018) (Figure 4.8). An even higher degree of ecosystem services can, at least theoretically, be achieved by strategically combining different functional groups of crops such as a cereal and a nitrogen-fixing legume (Soussana and Lemaire 2014). Not only is there evidence from plant-diversity experiments that communities with higher species richness sustain higher concentrations of SOC (Hungate et al. 2017; Sprunger and Robertson 2018; Chen, S. 2018; Yang et al. 2019), but other valuable ecosystem services such as pest suppression, lower GHG emissions, and greater nutrient retention may be enhanced (Schnitzer et al. 2011; Culman et al. 2013).

Similar to perennial forage crops such as alfalfa, perennial grain crops are expected to have a definite productive lifespan, probably in the range of three to 10 years. A key area of research on perennial grains cropping systems is to minimise losses of SOC during conversion of one stand of perennial grains to another. Recent work demonstrates that no-till conversion of a mature perennial grassland to another perennial crop will experience several years of high net CO_2 emissions as decomposition of copious crop residues exceed ecosystem uptake of carbon by the new crop (Abraha et al. 2018). Most, if not all, of this lost carbon will be recaptured in the replacement crop. It is not known whether mineral-stabilised carbon that is protected in soil aggregates is vulnerable to loss in perennial crop succession.

Figure 4.8 | Comparison of root systems between the newly domesticated intermediate wheatgrass (left) and annual wheat (right). Photo: Copyright © Jim Richardson.

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Perennial grains hold promises of agricultural practices, which can significantly reduce soil erosion and nutrient leakage while sequestering carbon. When cultivated in mixes with N-fixing species (legumes) such polycultures also reduce the need for external inputs of nitrogen – a large source of GHG from conventional agriculture.

1.9.3 Reversing land degradation through reforestation

1.9.3.1 South Korea case study on reforestation success

In the first half of the 20th century, forests in the Republic of Korea (South Korea) were severely degraded and deforested during foreign occupations and the Korean War. Unsustainable harvest for timber and fuelwood resulted in severely degraded landscapes, heavy soil erosion and large areas denuded of vegetation cover. Recognising that South Korea's economic health would depend on a healthy environment, South Korea established a national forest service (1967) and embarked on the first phase of a 10-year reforestation programme in 1973 (Forest Development Program), which was followed by subsequent reforestation programmes that ended in 1987, after 2.4 Mha of forests were restored (Figure 4.9).

As a consequence of reforestation, forest volume increased from 11.3 m^3 ha^{-1} in 1973 to 125.6 m^3 ha^{-1} in 2010 and 150.2 m^3 ha^{-1} in 2016 (Korea Forest Service 2017). Increases in forest volume had significant co-benefits such as increasing water yield by 43% and reducing soil losses by 87% from 1971 to 2010 (Kim et al. 2017).

The forest carbon density in South Korea has increased from 5–7 MgC ha^{-1} in the period 1955–1973 to more than 30 MgC ha^{-1} in the late 1990s (Choi et al. 2002). Estimates of carbon uptake rates in the late 1990s were 12 TgC yr^{-1} (Choi et al. 2002). For the period 1954 to 2012, carbon uptake was 8.3 TgC yr^{-1} (Lee et al. 2014), lower than other estimates because reforestation programmes did not start until 1973. Net ecosystem production in South Korea was 10.55 ± 1.09 TgC yr^{-1} in the 1980s, 10.47 ± 7.28 Tg C yr^{-1} in the 1990s, and 6.32 ± 5.02 Tg C yr^{-1} in the 2000s, showing a gradual decline as average forest age increased (Cui et al. 2014). The estimated past and projected future increase in the carbon content of South Korea's forest area during 1992–2034 was 11.8 TgC yr^{-1} (Kim et al. 2016).

During the period of forest restoration, South Korea also promoted inter-agency cooperation and coordination, especially between the energy and forest sectors, to replace firewood with fossil fuels, and to reduce demand for firewood to help forest recovery (Bae et al. 2012). As experience with forest restoration programmes has increased, emphasis has shifted from fuelwood plantations, often

Figure 4.9 | Example of severely degraded hills in South Korea and stages of forest restoration. The top two photos are taken in the early 1970s, before and after restoration, the third photo about five years after restoration, and the bottom photo was taken about 20 years after restoration. Many examples of such restoration success exist throughout South Korea. (Photos: Copyright © Korea Forest Service)

with exotic species and hybrid varieties to planting more native species and encouraging natural regeneration (Kim and Zsuffa 1994; Lee et al. 2015). Avoiding monocultures in reforestation programmes can reduce susceptibility to pests (Kim and Zsuffa 1994). Other important factors in the success of the reforestation programme were that private landowners were heavily involved in initial efforts (both corporate entities and smallholders) and that the reforestation programme was made part of the national economic development programme (Lamb 2014).

The net present value and the cost-benefit ratio of the reforestation programme were 54.3 billion and 5.84 billion USD in 2010, respectively. The breakeven point of the reforestation investment appeared within two decades. Substantial benefits of the reforestation programme included disaster risk reduction and carbon sequestration (Lee et al. 2018a).

In summary, the reforestation programme was a comprehensive technical and social initiative that restored forest ecosystems, enhanced the economic performance of rural regions, contributed to disaster risk reduction, and enhanced carbon sequestration (Kim et al. 2017; Lee et al. 2018a; UNDP 2017).

The success of the reforestation programme in South Korea and the associated significant carbon sink indicate a high mitigation potential that might be contributed by a potential future reforestation programme in the Democratic People's Republic of Korea (North Korea) (Lee et al. 2018b).

1.9.3.2 China case study on reforestation success

The dramatic decline in the quantity and quality of natural forests in China resulted in land degradation, such as soil erosion, floods, droughts, carbon emission, and damage to wildlife habitat (Liu and Diamond 2008). In response to failures of previous forestry and land policies, the severe droughts in 1997, and the massive floods in 1998, the central government decided to implement a series of land degradation control policies, including the National Forest Protection Program (NFPP), Grain for Green or the Conversion of Cropland to Forests and Grassland Program (GFGP) (Liu et al. 2008; Yin 2009; Tengberg et al. 2016; Zhang et al. 2000). The NFPP aimed to completely ban logging of natural forests in the upper reaches of the Yangtze and Yellow rivers as well as in Hainan Province by 2000 and to substantially reduce logging in other places (Xu et al. 2006). In 2011, NFPP was renewed for the 10-year second phase, which also added another 11 counties around Danjiangkou Reservoir in Hubei and Henan Provinces, the water source for the middle route of the South-to-North Water Diversion Project (Liu et al. 2013). Furthermore, the NFPP afforested 31 Mha by 2010 through aerial seeding, artificial planting, and mountain closure (i.e., prohibition of human activities such as fuelwood collection and lifestock grazing) (Xu et al. 2006). China banned commercial logging in all natural forests by the end of 2016, which imposed logging bans and harvesting reductions in 68.2 Mha of forest land – including 56.4 Mha of natural forest (approximately 53% of China's total natural forests).

GFGP became the most ambitious of China's ecological restoration efforts, with more than 45 billion USD devoted to its implementation since 1990 (Kolinjivadi and Sunderland 2012) The programme involves the conversion of farmland on slopes of 15–25° or greater to forest or grassland (Bennett 2008). The pilot programme started in three provinces – Sichuan, Shaanxi and Gansu – in 1999 (Liu and Diamond 2008). After its initial success, it was extended to 17 provinces by 2000 and finally to all provinces by 2002, including the headwaters of the Yangtze and Yellow rivers (Liu et al. 2008).

NFPP and GFGP have dramatically improved China's land conditions and ecosystem services, and thus have mitigated the unprecedented land degradation in China (Liu et al. 2013; Liu et al. 2002; Long et al. 2006; Xu et al. 2006). NFPP protected 107 Mha forest area and increased forest area by 10 Mha between 2000 and 2010. For the second phase (2011–2020), the NFPP plans to increase forest cover by a further 5.2 Mha, capture 416 million tons of carbon, provide 648,500 forestry jobs, further reduce land degradation, and enhance biodiversity (Liu et al. 2013). During 2000–2007, sediment concentration in the Yellow River had declined by 38%. In the Yellow River basin, it was estimated that surface runoff would be reduced by 450 million m^3 from 2000 to 2020, which is equivalent to 0.76% of the total surface water resources (Jia et al. 2006). GFGP had cumulatively increased vegetative cover by 25 Mha, with 8.8 Mha of cropland being converted to forest and grassland, 14.3 Mha barren land being afforested, and 2.0 Mha of forest regeneration from mountain closure. Forest cover within the GFGP region has increased 2% during the first eight years (Liu et al. 2008). In Guizhou Province, GFGP plots had 35–53% less loss of phosphorus than non-GFGP plots (Liu et al. 2002). In Wuqi County of Shaanxi Province, the Chaigou Watershed had 48% and 55% higher soil moisture and moisture-holding capacity in GFGP plots than in non-GFGP plots, respectively (Liu et al. 2002). According to reports on China's first national ecosystem assessment (2000–2010), for carbon sequestration and soil retention, coefficients for the GFGP targeting forest restoration and NFPP are positive and statistically significant. For sand fixation, GFGP targeting grassland restoration is positive and statistically significant. Remote sensing observations confirm that vegetation cover increased and bare soil declined in China over the period 2001 to 2015 (Qiu et al. 2017). But, where afforestation is sustained by drip irrigation from groundwater, questions about plantation sustainability arise (Chen et al. 2018a). Moreover, greater gains in biodiversity could be achieved by promoting mixed forests over monocultures (Hua et al. 2016).

NFPP-related activities received a total commitment of 93.7 billion yuan (about 14 billion USD at 2018 exchange rate) between 1998 and 2009. Most of the money was used to offset economic losses of forest enterprises caused by the transformation from logging to tree plantations and forest management (Liu et al. 2008). By 2009, the cumulative total investment through the NFPP and GFGP exceeded 50 billion USD2009 and directly involved more than 120 million farmers in 32 million households in the GFGP alone (Liu et al. 2013). All programmes reduce or reverse land degradation and improve human well-being. Thus, a coupled human and natural systems perspective (Liu et al. 2008) would be helpful to understand the complexity of policies and their impacts, and to establish long-term management mechanisms to improve the livelihood of participants in these programmes and other land management policies in China and many other parts of the world.

1.9.4 Degradation and management of peat soils

Globally, peatlands cover 3–4% of the Earth's land area (about 430 Mha) (Xu et al. 2018a) and store 26–44% of estimated global SOC (Moore 2002). They are most abundant in high northern latitudes, covering large areas in North America, Russia and Europe. At lower latitudes, the largest areas of tropical peatlands are located in Indonesia, the Congo Basin and the Amazon Basin in the form of peat swamp forests (Gumbricht et al. 2017; Xu et al. 2018a). It is estimated that, while 80–85% of the global peatland areas is still largely in a natural state, they are such carbon-dense ecosystems that degraded peatlands (0.3% of the terrestrial land) are responsible for a disproportional 5% of global anthropogenic CO_2 emissions – that is, an annual addition of 0.9–3 $GtCO_2$ to the atmosphere (Dommain et al. 2012; IPCC 2014c).

Peatland degradation is not well quantified globally, but regionally peatland degradation can involve a large percentage of the areas. Land-use change and degradation in tropical peatlands have primarily been quantified in Southeast Asia, where drainage and conversion to plantation crops is the dominant transition (Miettinen et al. 2016). Degradation of peat swamps in Peru is also a growing concern and one pilot survey showed that more than 70% of the peat swamps were degraded in one region surveyed (Hergoualc'h et al. 2017a). Around 65,000 km^2 or 10% of the European peatland area has been lost and 44% of the remaining European peatlands are degraded (Joosten, H., Tanneberger 2017). Large areas of fens have been entirely 'lost' or greatly reduced in thickness due to peat wastage (Lamers et al. 2015).

The main drivers of the acceleration of peatland degradation in the 20th century were associated with drainage for agriculture, peat extraction and afforestation related activities (burning, over-grazing, fertilisation) with a variable scale and severity of impact depending on existing resources in the various countries (O'Driscoll et al. 2018; Cobb, A.R. et al. Dommain et al. 2018; Lamers et al. 2015). New drivers include urban development, wind farm construction (Smith et al. 2012), hydroelectric development, tar sands mining and recreational uses (Joosten and Tanneberger 2017). Anthropogenic pressures are now affecting peatlands in previously geographically isolated areas with consequences for global environmental concerns and impacts on local livelihoods (Dargie et al. 2017; Lawson et al. 2015; Butler et al. 2009).

Drained and managed peatlands are GHG-emission hotspots (Swails et al. 2018; Hergoualc'h et al. 2017a, 2017b; Roman-Cuesta et al. 2016). In most cases, lowering of the water table leads to direct and indirect CO_2 and N_2O emissions to the atmosphere, with rates dependent on a range of factors, including the groundwater level and the water content of surface peat layers, nutrient content, temperature, and vegetation communities. The exception is nutrient-limited boreal peatlands (Minkkinen et al. 2018; Ojanen et al. 2014). Drainage also increases erosion and dissolved organic carbon loss, removing stored carbon into streams as dissolved and particulate organic carbon, which ultimately returns to the atmosphere (Moore et al. 2013; Evans et al. 2016).

In tropical peatlands, oil palm is the most widespread plantation crop and, on average, it emits around 40 $tCO_2\ ha^{-1}\ yr^{-1}$; Acacia plantations for pulpwood are the second most widespread plantation crop and emit around 73 $tCO_2\ ha^{-1}\ yr^{-1}$ (Drösler et al. 2013). Other land uses typically emit less than 37 $tCO_2\ ha^{-1}\ yr^{-1}$. Total emissions from peatland drainage in the region are estimated to be between 0.07 and 1.1 $GtCO_2\ yr^{-1}$ (Houghton and Nassikas 2017; Frolking et al. 2011). Land-use change also affects the fluxes of N_2O and CH_4. Undisturbed tropical peatlands emit about 0.8 $MtCH_4\ yr^{-1}$ and 0.002 $MtN_2O\ yr^{-1}$, while disturbed peatlands emit 0.1 $MtCH_4\ yr^{-1}$and 0.2 $MtN_2O–N\ yr^{-1}$ (Frolking et al. 2011). These N_2O emissions are probably low, as new findings show that emissions from fertilised oil palm can exceed 20 $kgN_2O–N\ ha^{-1}\ yr^{-1}$ (Oktarita et al. 2017).

In the temperate and boreal zones, peatland drainage often leads to emissions in the order of 0.9 to 9.5 $tCO_2\ ha^{-1}\ y^{-1}$ in forestry plantations and 21 to 29 $tCO_2\ ha^{-1}\ y^{-1}$ in grasslands and croplands. Nutrient-poor sites often continue to be CO_2 sinks for long periods (e.g., 50 years) following drainage and, in some cases, sinks for atmospheric CH_4, even when drainage ditch emissions are considered (Minkkinen et al. 2018; Ojanen et al. 2014). Undisturbed boreal and temperate peatlands emit about 30 $MtCH_4\ yr^{-1}$ and 0.02 $MtN_2O–N\ yr^{-1}$, while disturbed peatlands emit 0.1 $MtCH_4\ yr^{-1}$and 0.2 $MtN_2O–N\ yr^{-1}$ (Frolking et al. 2011).

Fire emissions from tropical peatlands are only a serious issue in Southeast Asia, where they are responsible for 634 (66–4070) $MtCO_2\ yr^{-1}$ (van der Werf et al. 2017). Much of the variability is linked with the El Niño–Southern Oscillation (ENSO), which produces drought conditions in this region. Anomalously active fire seasons have also been observed in non-drought years and this has been attributed to the increasing effect of high temperatures that dry vegetation out during short dry spells in otherwise normal rainfall years (Fernandes et al. 2017; Gaveau et al. 2014). Fires have significant societal impacts; for example, the 2015 fires caused more than 100,000 additional deaths across Indonesia, Malaysia and Singapore, and this event was more than twice as deadly as the 2006 El Niño event (Koplitz et al. 2016).

Peatland degradation in other parts of the world differs from Asia. In Africa, for large peat deposits like those found in the Cuvette Centrale in the Congo Basin or in the Okavango inland delta, the principle threat is changing rainfall regimes due to climate variability and change (Weinzierl et al. 2016; Dargie et al. 2017). Expansion of agriculture is not yet a major factor in these regions. In the Western Amazon, extraction of non-timber forest products like the fruits of *Mauritia flexuosa* (moriche palm) and Suri worms are major sources of degradation that lead to losses of carbon stocks (Hergoualc'h et al. 2017a).

The effects of peatland degradation on livelihoods have not been systematically characterised. In places where plantation crops are driving the conversion of peat swamps, the financial benefits can be considerable. One study in Indonesia found that the net present value of an oil palm plantation is between 3,835 and 9,630 USD per ha to land owners (Butler et al. 2009). High financial returns are creating incentives for the expansion of smallholder production in peatlands. Smallholder plantations extend over 22% of the peatlands in insular Southeast Asia compared to 27% for industrial plantations (Miettinen et al. 2016). In places where income is generated from extraction of marketable products, ecosystem degradation probably has a negative effect on livelihoods. For example, the sale of fruits of *M. flexuosa* in some parts of the western Amazon constitutes as much as 80% of the winter income of many rural households, but information on trade values and value chains of *M. flexuosa* is still sparse (Sousa et al. 2018; Virapongse et al. 2017).

There is little experience with peatland restoration in the tropics. Experience from northern latitudes suggests that extensive damage and changes in hydrological conditions mean that restoration in many cases is unachievable (Andersen et al. 2017). In the case of Southeast Asia, where peatlands form as raised bogs, drainage leads to collapse of the dome, and this collapse cannot be reversed by rewetting. Nevertheless, efforts are underway to develop solutions, or at least partial solutions in Southeast Asia, for example, by the Indonesian Peatland Restoration Agency. The first step is to restore the hydrological regime in drained peatlands, but so far experiences with canal blocking and reflooding of the peat have been only partially successful (Ritzema et al. 2014). Market incentives with certification through the Roundtable on Sustainable Palm Oil have also not been particularly successful as many concessions seek certification only after significant environmental degradation has occurred (Carlson et al. 2017). Certification had no discernible effect on forest loss or fire detection in peatlands in Indonesia. To date there is no documentation of restoration methods or successes in many other parts of the tropics. However, in situations where degradation does not involve drainage, ecological restoration may be possible. In South America, for example, there is growing interest in restoration of palm swamps, and as experiences are gained it will be important to document success factors to inform successive efforts (Virapongse et al. 2017).

In higher latitudes where degraded peatlands have been drained, the most effective option to reduce losses from these large organic carbon stocks is to change hydrological conditions and increase soil moisture and surface wetness (Regina et al. 2015). Long-term GHG monitoring in boreal sites has demonstrated that rewetting and restoration noticeably reduce emissions compared to degraded drained sites and can restore the carbon sink function when vegetation is re-established (Wilson et al. 2016; IPCC 2014a; Nugent et al. 2018) although, restored ecosystems may not yet be as resilient as their undisturbed counterparts (Wilson et al. 2016). Several studies have demonstrated the co-benefits of rewetting specific degraded peatlands for biodiversity, carbon sequestration, (Parry et al. 2014; Ramchunder et al. 2012; Renou-Wilson et al. 2018) and other ecosystem services, such as improvement of water storage and quality (Martin-Ortega et al. 2014) with beneficial consequences for human well-being (Bonn et al. 2016; Parry et al. 2014).

1.9.5 Biochar

Biochar is organic matter that is carbonised by heating in an oxygen-limited environment, and used as a soil amendment. The properties of biochar vary widely, dependent on the feedstock and the conditions of production. Biochar could make a significant contribution to mitigating both land degradation and climate change, simultaneously.

1.9.5.1 Role of biochar in climate change mitigation

Biochar is relatively resistant to decomposition compared with fresh organic matter or compost, so represents a long-term carbon store (*very high confidence*). Biochars produced at higher temperature (>450°C) and from woody material have greater stability than those produced at lower temperature (300–450°C), and from manures (*very high confidence*) (Singh et al. 2012; Wang et al. 2016b). Biochar stability is influenced by soil properties: biochar carbon can be further stabilised by interaction with clay minerals and native SOM (*medium evidence*) (Fang et al. 2015). Biochar stability is estimated to range from decades to thousands of years, for different biochars in different applications (Singh et al. 2015; Wang et al. 2016). Biochar stability decreases as ambient temperature increases (*limited evidence*) (Fang et al. 2017).

Biochar can enhance soil carbon stocks through 'negative priming', in which rhizodeposits are stabilised through sorption of labile carbon on biochar, and formation of biochar-organo-mineral complexes (Weng et al. 2015, 2017, 2018; Wang et al. 2016b). Conversely, some studies show increased turnover of native soil carbon ('positive priming') due to enhanced soil microbial activity induced by biochar. In clayey soils, positive priming is minor and short-lived compared to negative priming effects, which dominate in the medium to long term (Singh and Cowie 2014; Wang et al. 2016b). Negative priming has been observed particularly in loamy grassland soil (Ventura et al. 2015) and clay-dominated soils, whereas positive priming is reported in sandy soils (Wang et al. 2016b) and those with low carbon content (Ding et al. 2018).

Biochar can provide additional climate-change mitigation by decreasing nitrous oxide (N_2O) emissions from soil, due in part to decreased substrate availability for denitrifying organisms, related to the molar H/C ratio of the biochar (Cayuela et al. 2015). However, this impact varies widely: meta-analyses found an average decrease in N_2O emissions from soil of 30–54%, (Cayuela et al. 2015; Borchard et al. 2019; Moore 2002), although another study found no significant reduction in field conditions when weighted by the inverse of the number of observations per site (Verhoeven et al. 2017). Biochar has been observed to reduce methane emissions from flooded soils, such as rice paddies, though, as for N_2O, results vary between studies and increases have also been observed (He et al. 2017; Kammann et al. 2017). Biochar has also been found to reduce methane uptake by dryland soils, though the effect is small in absolute terms (Jeffery et al. 2016).

Additional climate benefits of biochar can arise through: reduced nitrogen fertiliser requirements, due to reduced losses of nitrogen through leaching and/or volatilisation (Singh et al. 2010) and enhanced biological nitrogen fixation (Van Zwieten et al. 2015); increased yields of crop, forage, vegetable and tree species (Biederman and Harpole 2013), particularly in sandy soils and acidic tropical soils (Simon et al. 2017); avoided GHG emissions from manure that would otherwise be stockpiled, crop residues that would be burned or processing residues that would be landfilled; and reduced GHG emissions from compost when biochar is added (Agyarko-Mintah et al. 2017; Wu et al. 2017a).

Climate benefits of biochar could be substantially reduced through reduction in albedo if biochar is surface-applied at high rates to light-coloured soils (Genesio et al. 2012; Bozzi et al. 2015; Woolf et al. 2010), or if black carbon dust is released (Genesio et al. 2016). Pelletising or granulating biochar, and applying below the soil surface or incorporating into the soil, minimises the release of black carbon dust and reduces the effect on albedo (Woolf et al. 2010).

Biochar is a potential 'negative emissions' technology: the thermochemical conversion of biomass to biochar slows mineralisation of the biomass, delivering long-term carbon storage; gases released during pyrolysis can be combusted for heat or power, displacing fossil energy sources, and could be captured and sequestered if linked with infrastructure for CCS (Smith 2016). Studies of the lifecycle climate change impacts of biochar systems generally show emissions reduction in the range 0.4 –1.2 $tCO_2e\ t^{-1}$ (dry) feedstock (Cowie et al. 2015). Use of biomass for biochar can deliver greater benefits than use for bioenergy, if applied in a context where it delivers agronomic benefits and/or reduces non-CO_2 GHG emissions (Ji et al. 2018; Woolf et al. 2010, 2018; Xu et al. 2019). A global analysis of technical potential, in which biomass supply constraints were applied to protect against food insecurity, loss of habitat and land degradation, estimated technical potential abatement of 3.7–6.6 $GtCO_2e\ yr^{-1}$ (including 2.6–4.6 $GtCO_2e\ yr^{-1}$ carbon stabilisation), with theoretical potential to reduce total emissions over the course of a century by 240–475 $GtCO_2e$ (Woolf et al. 2010). Fuss et al. (2018) propose a range of 0.5–2 $GtCO_2e$ per year as the sustainable potential for negative emissions through biochar. Mitigation potential of biochar is reviewed in Chapter 2.

1.9.5.2 Role of biochar in management of land degradation

Biochars generally have high porosity, high surface area and surface-active properties that lead to high absorptive and adsorptive capacity, especially after interaction in soil (Joseph et al. 2010). As a result of these properties, biochar could contribute to avoiding, reducing and reversing land degradation through the following documented benefits:

- Improved nutrient use efficiency due to reduced leaching of nitrate and ammonium (e.g., Haider et al. 2017) and increased availability of phosphorus in soils with high phosphorus fixation capacity (Liu et al. 2018c), potentially reducing nitrogen and phosphorus fertiliser requirements.
- Management of heavy metals and organic pollutants: through reduced bioavailability of toxic elements (O'Connor et al. 2018; Peng et al. 2018), by reducing availability, through immobilisation due to increased pH and redox effects (Rizwan et al. 2016) and adsorption on biochar surfaces (Zhang et al. 2013) thus providing a means of remediating contaminated soils, and enabling their utilisation for food production.
- Stimulation of beneficial soil organisms, including earthworms and mycorrhizal fungi (Thies et al. 2015).
- Improved porosity and water-holding capacity (Quin et al. 2014), particularly in sandy soils (Omondi et al. 2016), enhancing microbial function during drought (Paetsch et al. 2018).
- Amelioration of soil acidification, through application of biochars with high pH and acid-neutralising capacity (Chan et al. 2008; Van Zwieten et al. 2010).

Biochar systems can deliver a range of other co-benefits, including destruction of pathogens and weed propagules, avoidance of landfill, improved handling and transport of wastes such as sewage sludge, management of biomass residues such as environmental weeds and urban greenwaste, reduction of odours and management of nutrients from intensive livestock facilities, reduction in environmental nitrogen pollution and protection of waterways. As a compost additive, biochar has been found to reduce leaching and volatilisation of nutrients, increasing nutrient retention through absorption and adsorption processes (Joseph et al. 2018).

While many studies report positive responses, some studies have found negative or zero impacts on soil properties or plant response (e.g., Kuppusamy et al. 2016). The risk that biochar may enhance polycyclic aromatic hydrocarbon (PAH) in soil or sediments has been raised (Quilliam et al. 2013; Ojeda et al. 2016), but bioavailability of PAH in biochar has been shown to be very low (Hilber et al. 2017) Pyrolysis of biomass leads to losses of volatile nutrients, especially nitrogen. While availability of nitrogen and phosphorus in biochar is lower than in fresh biomass, (Xu et al. 2016) the impact of biochar on plant uptake is determined by the interactions between biochar, soil minerals and activity of microorganisms (e.g., Vanek and Lehmann 2015; Nguyen et al. 2017). To avoid negative responses, it is important to select biochar formulations to address known soil constraints, and to apply biochar prior to planting (Nguyen et al. 2017). Nutrient enrichment improves the performance of biochar from low nutrient feedstocks (Joseph et al. 2013). While there are many reports of biochar reducing disease or pest incidence, there are also reports of nil or negative effects (Bonanomi et al. 2015). Biochar may induce systemic disease resistance (e.g., Elad et al. 2011), though Viger et al. (2015) reported down-regulation of plant defence genes, suggesting increased susceptibility to insect and pathogen attack. Disease suppression where biochar is applied is associated with increased microbial diversity and metabolic potential of the rhizosphere microbiome (Kolton et al. 2017). Differences in properties related to feedstock (Bonanomi et al. 2018) and differential response to biochar dose, with lower rates more effective (Frenkel et al. 2017), contribute to variable disease responses.

The constraints on biochar adoption include: the high cost and limited availability due to limited large-scale production; limited amount of unutilised biomass; and competition for land for growing biomass. While early biochar research tended to use high rates of application (10 t ha^{-1} or more) subsequent studies have shown that biochar can be effective at lower rates, especially when combined with chemical or organic fertilisers (Joseph et al. 2013). Biochar can be produced at many scales and levels of engineering sophistication, from simple cone kilns and cookstoves to large industrial-scale units processing several tonnes of biomass per hour (Lehmann and Joseph 2015). Substantial technological development has occurred recently, though large-scale deployment is limited to date.

Governance of biochar is required to manage climate, human health and contamination risks associated with biochar production in poorly designed or operated facilities that release methane or particulates (Downie et al. 2012; Buss et al. 2015), to ensure quality control of biochar products, and to ensure that biomass is sourced sustainably and is uncontaminated. Measures could include labelling standards, sustainability certification schemes and regulation of biochar production and use. Governance mechanisms should be tailored to context, commensurate with risks of adverse outcomes.

In summary, application of biochar to soil can improve soil chemical, physical and biological attributes, enhancing productivity and resilience to climate change, while also delivering climate-change mitigation through carbon sequestration and reduction in GHG emissions (*medium agreement*, *robust evidence*). However, responses to biochar depend on the biochar's properties, which are in turn dependent on feedstock and biochar production conditions, and the soil and crop to which it is applied. Negative or nil results have been recorded. Agronomic and methane-reduction benefits appear greatest in tropical regions, where acidic soils predominate and suboptimal rates of lime and fertiliser are common, while carbon stabilisation is greater in temperate regions. Biochar is most effective when applied in low volumes to the most responsive soils and when properties are matched to the specific soil constraints and plant needs. Biochar is thus a practice that has potential to address land degradation and climate change simultaneously, while also supporting sustainable development. The potential of biochar is limited by the availability of biomass for its production. Biochar production and use requires regulation and standardisation to manage risks (*strong agreement*).

1.9.6 Management of land degradation induced by tropical cyclones

Tropical cyclones are normal disturbances that natural ecosystems have been affected by and recovered from for millennia. Climate models mostly predict decreasing frequency of tropical cyclones, but dramatically increasing intensity of the strongest storms, as well as increasing rainfall rates (Bacmeister et al. 2018; Walsh et al. 2016b). Large amplitude fluctuations in the frequency and intensity complicate both the detection and attribution of tropical cyclones to climate change (Lin and Emanuel 2016b). Yet, the force of high-intensity cyclones has increased and is expected to escalate further due to global climate change (*medium agreement*, *robust evidence*) (Knutson et al. 2010; Bender et al. 2010; Vecchi et al. 2008; Bhatia et al. 2018; Tu et al. 2018; Sobel et al. 2016). Tropical cyclone paths are also shifting towards the poles, increasing the area subject to tropical cyclones (Sharmila and Walsh 2018; Lin and Emanuel 2016b). Climate change alone will affect the hydrology of individual wetland ecosystems, mostly through changes in precipitation and temperature regimes with great global variability (Erwin 2009). Over the last seven decades, the speed at which tropical cyclones move has decreased significantly, as expected from theory, exacerbating the damage on local communities from increasing rainfall amounts and high wind speed (Kossin 2018). Tropical cyclones will accelerate changes in coastal forest structure and composition. The heterogeneity of land degradation at coasts that are affected by tropical cyclones can be further enhanced by the interaction of its components (for example, rainfall, wind speed, and direction) with topographic and biological factors (for example, species susceptibility) (Luke et al. 2016).

Small Island Developing States (SIDS) are particularly affected by land degradation induced by tropical cyclones; recent examples are Matthew (2016) in the Caribbean, and Pam (2015) and Winston (2016) in the Pacific (Klöck and Nunn 2019; Handmer and Nalau 2019). Even if the Pacific Ocean has experienced cyclones of unprecedented intensity in recent years, their geomorphological effects may not be unprecedented (Terry and Lau 2018).

Cyclone impacts on coastal areas is not restricted to SIDS, but a problem for all low-lying coastal areas (Petzold and Magnan 2019). The Sundarbans, one of the world's largest coastal wetlands, covers about one million hectares between Bangladesh and India. Large areas of the Sundarbans mangroves have been converted into paddy fields over the past two centuries and, more recently, into shrimp farms (Ghosh et al. 2015). In 2009, cyclone Aila caused incremental stresses on the socio-economic conditions of the Sundarbans coastal communities through rendering huge areas of land unproductive for a long time (Abdullah et al. 2016). The impact of Aila was widespread throughout the Sundarbans mangroves, showing changes between the pre- and post-cyclonic period of 20–50% in the enhanced vegetation index (Dutta et al. 2015), although the magnitude of the effects of the Sundarbans mangroves derived from climate change is not yet defined (Payo et al. 2016; Loucks et al. 2010; Gopal and Chauhan 2006; Ghosh et al. 2015; Chaudhuri et al. 2015). There is *high agreement* that the joint effect of climate change and land degradation will be very negative for the area, strongly affecting the environmental services provided by these forests, including the extinction of large mammal species (Loucks et al. 2010). The changes in vegetation are mainly due to inundation and erosion (Payo et al. 2016).

Tropical cyclone Nargis unexpectedly hit the Ayeyarwady River delta (Myanmar) in 2008 with unprecedented and catastrophic damages to livelihoods, destruction of forests and erosion of fields (Fritz et al. 2009) as well as eroding the shoreline 148 m compared with the long-term average (1974–2015) of 0.62 m yr^{-1}. This is an example of the disastrous effects that changing cyclone paths can have on areas previously not affected by cyclones (Fritz et al. 2010).

1.9.6.1 Management of coastal wetlands

Tropical cyclones mainly, but not exclusively, affect coastal regions, threatening maintenance of the associated ecosystems, mangroves, wetlands, seagrasses, and so on. These areas not only provide food, water and shelter for fish, birds and other wildlife, but also provide important ecosystem services such as water-quality improvement, flood abatement and carbon sequestration (Meng et al. 2017).

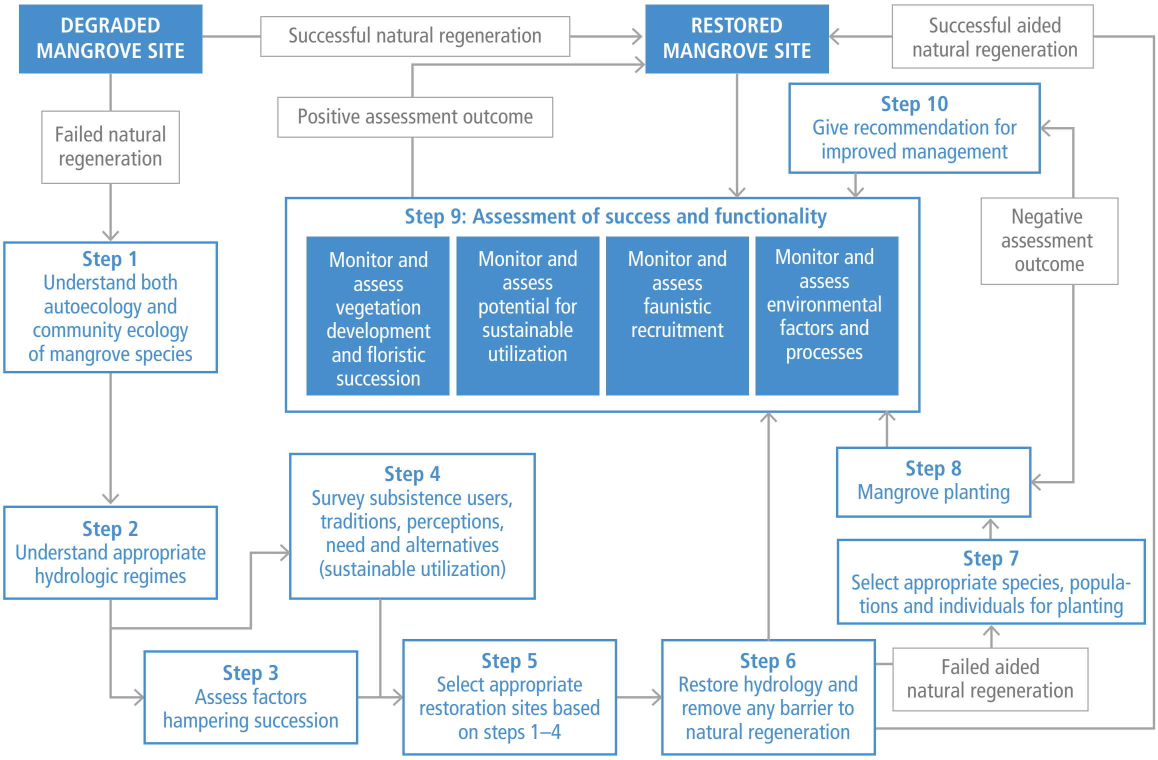

Figure 4.10 | Decision tree showing recommended steps and tasks to restore a mangrove wetland based on original site conditions. (Modified from Bosire et al. 2008.)

Despite their importance, coastal wetlands are listed amongst the most heavily damaged of natural ecosystems worldwide. Starting in the 1990s, wetland restoration and re-creation became a 'hotspot' in the ecological research fields (Zedler 2000). Coastal wetland restoration and preservation is an extremely cost-effective strategy for society, for example, the preservation of coastal wetlands in the USA provides storm protection services, with a cost of 23.2 billion USD yr^{-1} (Costanza et al. 2008).

There is a *high agreement* with *medium evidence* that the success of wetland restoration depends mainly on the flow of the water through the system, the degree to which re-flooding occurs, disturbance regimes, and the control of invasive species (Burlakova et al. 2009; López-Rosas et al. 2013). The implementation of the Ecological Mangrove Rehabilitation protocol (López-Portillo et al. 2017) that includes monitoring and reporting tasks, has been proven to deliver successful rehabilitation of wetland ecosystem services.

1.9.7 Saltwater intrusion

Current environmental changes, including climate change, have caused sea levels to rise worldwide, particularly in tropical and subtropical regions (Fasullo and Nerem 2018). Combined with scarcity of water in river channels, such rises have been instrumental in the intrusion of highly saline seawater inland, posing a threat to coastal areas and an emerging challenge to land managers and policymakers. Assessing the extent of salinisation due to sea water intrusion at a global scale nevertheless remains challenging. Wicke et al. (2011) suggest that across the world, approximately 1.1 Gha of land is affected by salt, with 14% of this categorised as forest, wetland or some other form of protected area. Seawater intrusion is generally caused by (i) increased tidal activity, storm surges, cyclones and sea storms due to changing climate, (ii) heavy groundwater extraction or land-use changes as a result of changes in precipitation, and droughts/floods, (iii) coastal erosion as a result of destruction of mangrove forests and wetlands, (iv) construction of vast irrigation canals and drainage networks leading to low river discharge in the deltaic region; and (v) sea level rise contaminating nearby freshwater aquifers as a result of subsurface intrusion (Uddameri et al. 2014).

The Indus Delta, located in the south-eastern coast of Pakistan near Karachi in the North Arabian Sea, is one of the six largest estuaries in the world, spanning an area of 600,000 ha. The Indus delta is a clear example of seawater intrusion and land degradation due to local as well as up-country climatic and environmental conditions (Rasul et al. 2012). Salinisation and waterlogging in the up-country areas including provinces of Punjab and Sindh is, however, caused by the irrigation network and over-irrigation (Qureshi 2011).

Such degradation takes the form of high soil salinity, inundation and waterlogging, erosion and freshwater contamination. The interannual

4

variability of precipitation with flooding conditions in some years and drought conditions in others has caused variable river flows and sediment runoff below Kotri Barrage (about 200 km upstream of the Indus delta). This has affected hydrological processes in the lower reaches of the river and the delta, contributing to the degradation (Rasul et al. 2012).

Over 480,000 ha of fertile land is now affected by sea water intrusion, wherein eight coastal subdivisions of the districts of Badin and Thatta are mostly affected (Chandio et al. 2011). A very high intrusion rate of 0.179 ± 0.0315 km yr^{-1}, based on the analysis of satellite data, was observed in the Indus delta during the 10 years between 2004 and 2015 (Kalhoro et al. 2016). The area of agricultural crops under cultivation has been declining, with economic losses of millions of USD (IUCN 2003). Crop yields have reduced due to soil salinity, in some places failing entirely. Soil salinity varies seasonally, depending largely on the river discharge: during the wet season (August 2014), salinity (0.18 mg L^{-1}) reached 24 km upstream, while during the dry season (May 2013), it reached 84 km upstream (Kalhoro et al. 2016). The freshwater aquifers have also been contaminated with sea water, rendering them unfit for drinking or irrigation purposes. Lack of clean drinking water and sanitation causes widespread diseases, of which diarrhoea is most common (IUCN 2003).

Lake Urmia in northwest Iran, the second-largest saltwater lake in the world and the habitat for endemic Iranian brine shrimp, *Artemia urmiana*, has also been affected by salty water intrusion. During a 17-year period between 1998 and 2014, human disruption, including agriculture and years of dam building affected the natural flow of freshwater as well as salty sea water in the surrounding area of Lake Urmia. Water quality has also been adversely affected, with salinity fluctuating over time, but in recent years reaching a maximum of 340 g L^{-1} (similar to levels in the Dead Sea). This has rendered the underground water unfit for drinking and agricultural purposes and risky to human health and livelihoods. Adverse impacts of global climate change as well as direct human impacts have caused changes in land use, overuse of underground water resources and construction of dams over rivers, which resulted in the drying-up of the lake in large part. This condition created sand, dust and salt storms in the region which affected many sectors including agriculture, water resources, rangelands, forests and health, and generally presented desertification conditions around the lake (Karbassi et al. 2010; Marjani and Jamali 2014; Shadkam et al. 2016).

Rapid irrigation expansion in the basin has, however, indirectly contributed to inflow reduction. Annual inflow to Lake Urmia has dropped by 48% in recent years. About three-fifths of this change was caused by climate change and two-fifths by water resource development and agriculture (Karbassi et al. 2010; Marjani and Jamali 2014; Shadkam et al. 2016).

In the drylands of Mexico, intensive production of irrigated wheat and cotton using groundwater (Halvorson et al. 2003) resulted in sea water intrusion into the aquifers of La Costa de Hermosillo, a coastal agricultural valley at the centre of Sonora Desert in Northwestern Mexico. Production of these crops in 1954 was on 64,000 ha of cultivated area, increasing to 132,516 ha in 1970, but decreasing to 66,044 ha in 2009 as a result of saline intrusion from the Gulf of California (Romo-Leon et al. 2014). In 2003, only 15% of the cultivated area was under production, with around 80,000 ha abandoned due to soil salinisation whereas in 2009, around 40,000 ha was abandoned (Halvorson et al. 2003; Romo-Leon et al. 2014). Salinisation of agricultural soils could be exacerbated by climate change, as Northwestern Mexico is projected to be warmer and drier under climate change scenarios (IPCC 2013a).

In other countries, intrusion of seawater is exacerbated by destruction of mangrove forests. Mangroves are important coastal ecosystems that provide spawning bed for fish, timber for building, and livelihoods to dependent communities. They also act as barriers against coastal erosion, storm surges, tropical cyclones and tsunamis (Kalhoro et al. 2017) and are among the most carbon-rich stocks on Earth (Atwood et al. 2017). They nevertheless face a variety of threats: climatic (storm surges, tidal activities, high temperatures) and human (coastal developments, pollution, deforestation, conversion to aquaculture, rice culture, oil palm plantation), leading to declines in their areas. In Pakistan, using remote sensing, the mangrove forest cover in the Indus delta decreased from 260,000 ha in 1980s to 160,000 ha in 1990 (Chandio et al. 2011). Based on remotely sensed data, a sharp decline in the mangrove area was also found in the arid coastal region of Hormozgan province in southern Iran during 1972, 1987 and 1997 (Etemadi et al. 2016). Myanmar has the highest rate (about 1% yr^{-1}) of mangrove deforestation in the world (Atwood et al. 2017). Regarding global loss of carbon stored in the mangrove due to deforestation, four countries exhibited high levels of loss: Indonesia (3410 $GgCO_2$ yr^{-1}), Malaysia (1288 $GgCO_2$ yr^{-1}), US (206 $GgCO_2$ yr^{-1}) and Brazil (186 $GgCO_2$ yr^{-1}). Only in Bangladesh and Guinea Bissau was there no decline in the mangrove area from 2000 to 2012 (Atwood et al. 2017).

Frequency and intensity of average tropical cyclones will continue to increase (Knutson et al. 2015) and global sea level will continue to rise. The IPCC (2013) projected with *medium confidence* that the sea level in the Asia Pacific region will rise from 0.4 to 0.6 m, depending on the emission pathway, by the end of this century. Adaptation measures are urgently required to protect the world's coastal areas from further degradation due to saline intrusion. A viable policy framework is needed to ensure that the environmental flows to deltas in order to repulse the intruding seawater.

1.9.8 Avoiding coastal maladaptation

Coastal degradation – for example, beach erosion, coastal squeeze, and coastal biodiversity loss – as a result of rising sea levels is a major concern for low lying coasts and small islands (*high confidence*). The contribution of climate change to increased coastal degradation has been well documented in AR5 (Nurse et al. 2014; Wong et al. 2014) and is further discussed in Section 4.4.1.3 as well as in the IPCC Special Report on the Ocean and Cryosphere in a Changing Climate (SROCC). However, coastal degradation can also be indirectly induced by climate change as the result of adaptation measures that involve changes to the coastal environment, for example, coastal protection measures against increased flooding and erosion due to sea level rise, and storm surges transforming the natural coast to a 'stabilised' coastline (Cooper and Pile 2014; French 2001). Every kind of adaptation response option is context-dependent, and, in fact, sea walls play an important role for adaptation in many places. Nonetheless, there are observed cases where the construction of sea walls can be considered 'maladaptation' (Barnett and O'Neill 2010; Magnan et al. 2016) by leading to increased coastal degradation, such as in the case of small islands where, due to limitations of space, coastal retreat is less of an option than in continental coastal zones. There is emerging literature on the implementation of alternative coastal protection measures and mechanisms on small islands to avoid coastal degradation induced by sea walls (e.g., Mycoo and Chadwick 2012; Sovacool 2012).

In many cases, increased rates of coastal erosion due to the construction of sea walls are the result of the negligence of local coastal morphological dynamics and natural variability as well as the interplay of environmental and anthropogenic drivers of coastal change (*medium evidence*, *high agreement*). Sea walls in response to coastal erosion may be ill-suited for extreme wave heights under cyclone impacts and can lead to coastal degradation by keeping overflowing sea water from flowing back into the sea, and therefore affect the coastal vegetation through saltwater intrusion, as observed in Tuvalu (Government of Tuvalu 2006; Wairiu 2017). Similarly, in Kiribati, poor construction of sea walls has resulted in increased erosion and inundation of reclaimed land (Donner 2012; Donner and Webber 2014). In the Comoros and Tuvalu, sea walls have been constructed from climate change adaptation funds and 'often by international development organisations seeking to leave tangible evidence of their investments' (Marino and Lazrus 2015, p. 344). In these cases, they have even increased coastal erosion, due to poor planning and the negligence of other causes of coastal degradation, such as sand mining (Marino and Lazrus 2015; Betzold and Mohamed 2017; Ratter et al. 2016). On the Bahamas, the installation of sea walls as a response to coastal erosion in areas with high wave action has led to the contrary effect and has even increased sand loss in those areas (Sealey 2006). The reduction of natural buffer zones – such as beaches and dunes – due to vertical structures, such as sea walls, increased the impacts of tropical cyclones on Reunion Island (Duvat et al. 2016). Such a process of 'coastal squeeze' (Pontee 2013) also results in the reduction of intertidal habitat zones, such as wetlands and marshes (Zhu et al. 2010). Coastal degradation resulting from the construction of sea walls, however, is not only observed in SIDS, as described above, but also on islands in the Global North, for example, the North Atlantic (Muir et al. 2014; Young et al. 2014; Cooper and Pile 2014; Bush 2004).

The adverse effects of coastal protection measures may be avoided by the consideration of local social-ecological dynamics, including critical study of the diverse drivers of ongoing shoreline changes, and the appropriate implementation of locally adequate coastal protection options (French 2001; Duvat 2013). Critical elements for avoiding maladaptation include profound knowledge of local tidal regimes, availability of relative sea level rise scenarios and projections for extreme water levels. Moreover, the downdrift effects of sea walls need to be considered, since undefended coasts may be exposed to increased erosion (Zhu et al. 2010). In some cases, it may be possible to keep intact and restore natural buffer zones as an alternative to the construction of hard engineering solutions. Otherwise, changes in land use, building codes, or even coastal realignment can be an option in order to protect and avoid the loss of the buffer function of beaches (Duvat et al. 2016; Cooper and Pile 2014). Examples in Barbados show that combinations of hard and soft coastal protection approaches can be sustainable and reduce the risk of coastal ecosystem degradation while keeping the desired level of protection for coastal users (Mycoo and Chadwick 2012). Nature-based solutions and approaches such as 'building with nature' (Slobbe et al. 2013) may allow for more sustainable coastal protection mechanisms and avoid coastal degradation. Examples from the Maldives, several Pacific islands and the North Atlantic show the importance of the involvement of local communities in coastal adaptation projects, considering local skills, capacities, as well as demographic and socio-political dynamics, in order to ensure the proper monitoring and maintenance of coastal adaptation measures (Sovacool 2012; Muir et al. 2014; Young et al. 2014; Buggy and McNamara 2016; Petzold 2016).

1.10 Knowledge gaps and key uncertainties

The co-benefits of improved land management, such as mitigation of climate change, increased climate resilience of agriculture, and impacts on rural areas/societies are well known in theory, but there is a lack of a coherent and systematic global inventory of such integrated efforts. Both successes and failures are important to document systematically.

Efforts to reduce climate change through land-demanding mitigation actions aimed at removing atmospheric carbon, such as afforestation, reforestation, bioenergy crops, intensification of land management and plantation forestry can adversely affect land conditions and lead to degradation. However, they may also lead to avoidance, reduction and reversal of degradation. Regionally differentiated, socially and ecologically appropriate SLM strategies need to be identified, implemented, monitored and the results communicated widely to ensure climate effective outcomes.

Impacts of new technologies on land degradation and their social and economic ramifications need more research.

Improved quantification of the global extent, severity and rates of land degradation by combining remote sensing with a systematic use of ancillary data is a priority. The current attempts need better scientific underpinning and appropriate funding.

Land degradation is defined using multiple criteria but the definition does not provide thresholds or the magnitude of acceptable change. In practice, human interactions with land will result in a variety of changes; some may contribute positively to one criterion while adversely affecting another. Research is required on the magnitude of impacts and the resulting trade-offs. Given the urgent need to remove carbon from the atmosphere and to reduce climate change impacts, it is important to reach agreement on what level of reduction in one criterion (biological productivity, ecological integrity) may be acceptable for a given increase in another criterion (ecological integrity, biological productivity).

Attribution of land degradation to the underlying drivers is a challenge because it is a complex web of causality rather than a simple cause–effect relationship. Also, diverging views on land degradation in relation to other challenges is hampering such efforts.

A more systematic treatment of the views and experiences of land users would be useful in land degradation studies.

Much research has tried to understand how social and ecological systems are affected by a particular stressor, for example, drought, heat, or waterlogging. But less research has tried to understand how such systems are affected by several simultaneous stressors – which is more realistic in the context of climate change (Mittler 2006).

More realistic modelling of carbon dynamics, including better appreciation of below-ground biota, would help us to better quantify the role of soils and soil management for soil carbon sequestration.

Frequently Asked Questions

FAQ 4.1 | How do climate change and land degradation interact with land use?

Climate change, land degradation and land use are linked in a complex web of causality. One important impact of climate change on land degradation is that increasing global temperatures intensify the hydrological cycle, resulting in more intense rainfall, which is an important driver of soil erosion. This means that sustainable land management (SLM) becomes even more important with climate change. Land-use change in the form of clearing of forest for rangeland and cropland (e.g., for provision of bio-fuels), and cultivation of peat soils, is a major source of greenhouse gas (GHG) emission from both biomass and soils. Many SLM practices (e.g., agroforestry, perennial crops, organic amendments, etc.) increase carbon content of soil and vegetation cover and hence provide both local and immediate adaptation benefits, combined with global mitigation benefits in the long term, while providing many social and economic co-benefits. Avoiding, reducing and reversing land degradation has a large potential to mitigate climate change and help communities to adapt to climate change.

FAQ 4.2 | How does climate change affect land-related ecosystem services and biodiversity?

Climate change will affect land-related ecosystem services (e.g., pollination, resilience to extreme climate events, water yield, soil conservation, carbon storage, etc.) and biodiversity, both directly and indirectly. The direct impacts range from subtle reductions or enhancements of specific services, such as biological productivity, resulting from changes in temperature, temperature variability or rainfall, to complete disruption and elimination of services. Disruptions of ecosystem services can occur where climate change causes transitions from one biome to another, for example, forest to grassland as a result of changes in water balance or natural disturbance regimes. Climate change will result in range shifts and, in some cases, extinction of species. Climate change can also alter the mix of land-related ecosystem services, such as groundwater recharge, purification of water, and flood protection. While the net impacts are specific to time as well as ecosystem types and services, there is an asymmetry of risk such that overall impacts of climate change are expected to reduce ecosystem services. Indirect impacts of climate change on land-related ecosystem services include those that result from changes in human behaviour, including potential large-scale human migrations or the implementation of afforestation, reforestation or other changes in land management, which can have positive or negative outcomes on ecosystem services.