Chapter 4: Land Degradation

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4.1 Executive summary

What is the problem?

Climate change exacerbates many land degradation processes (4.4.1, 4.4.2, Table 4.1) (*high confidence*). In particular, the following climate change induced processes are projected to increase the risk of land degradation (there is much evidence that these changes have already started to take effect):

- Intensification of hydrological cycle leads to more intensive rainfall which increases the risk of soil erosion (*high confidence*) PRECIP
- Increasing fire frequency resulting from heatwaves and reduction in rainfall drives land degradation of forest ecosystems (*high confidence*) TEMP & PRECIP
- Shifting vegetation patterns leads to land degradation in rapidly warming regions, such as the Arctic and sub-Arctic regions (*medium confidence*) TEMP & PRECIP
- In coastal regions, land degradation increases due to the combination of sea level rise and increasing wave actions. Particularly an increase of the most severe hurricanes will cause much damage (*high confidence*), but highly site specific. SLR & WIND
- Land degradation as a result of permafrost melting (*high confidence*) TEMP

Observed land degradation outcomes, however, are mediated by land use and land management which makes projections of future land degradation trends challenging (4.4.3) (*high confidence*)

Unless land management is improved, trends and projections of climate induced land degradation will seriously threaten livelihoods and ecosystem services in some biomes/anthromes (4.5, 4.6) (*medium confidence*). But the complex web of causality between climate change and many other local, regional and global trends and changes makes attribution difficult.

- Particularly illustrate how land degradation threatens human values and livelihoods in particular geographic settings (the analysis is in progress)

Land degradation in most biomes/anthromes is highly site specific and varies with differences in socio-economic conditions and land management (4.9) *(high confidence)*

- The following land management practices have been particularly successful in avoiding and preventing climate induced land degradation (still in progress)
- Future approaches with a potential to address both climate change and land degradation globally include the deployment of biochar as a soil additive, shifting from annual to perennial grain crops (in progress)

What are the potential solutions?

Sustainable Land Management (SLM) can reduce the risk of land degradation while simultaneously contributing to mitigation of climate change through carbon sequestration (Section 4.9). Proven methods exist for avoiding, reducing, and reversing land degradation while providing economic and social benefits to land managers *(high confidence)*.

What needs to be done?

Deployment, adoption, and maintenance of methods for SLM have been slow and needs further attention and resources (*high confidence*). Particularly the social and economic conditions, including gender and other equity concerns, must be addressed in order to make full use of the potential of SLM for realising the synergies between improving land productivity and mitigating climate change.

A range of legal and regulatory, economic and financial (4.10.1, 4.10.2, 4.10.5), social and cultural measures (4.8.1) can be used to create an enabling environment that equips land users and other

stakeholders to better manage land degradation and climate change impacts. Several options offer the possibility to deliver co-benefits in terms of mitigation and adaptation (4.10.1), and wider co-benefits such as biodiversity, health and well-being (4.10.2, 4.10.3) *(high confidence).*

Particular measures have proven effective in different contexts. (the analysis is in progress)

What can we do next?

Avoiding, reducing, and reversing land degradation is urgent for ensuring food security and improving human wellbeing. Early actions will generate both site specific and immediate benefits to affected communities as well as global benefits in terms of climate change adaptation and mitigation in the medium and long term (*high confidence*).

Note: we plan to use a 7x6 matrix of biomes (i.e. biophysical categories) and anthromes (i.e. land use intensity categories) as an implicit representation of spatial distribution. We intend to use the Land Degradation Neutrality (LDN) concepts as an organising principle (where applicable): avoid, reduce, and reverse land degradation.

4.2 Introduction

Land degradation has accompanied humanity since time immemorial but has accelerated since the transition from hunters and gatherers to farmers some 10,000 years ago. This change of livelihoods, the Neolithic revolution, has even been proposed as the onset of Anthropocene (Lewis and Maslin 2015). There is evidence that the levels of greenhouse gases (carbon dioxide and methane) of the atmosphere started to increase already 8000 to 5000 years ago as a result of expanding agriculture and clearing of forests (Garcin et al. 2018; Ruddiman 2003). 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 (Crews 2017). In a long historical perspective, say millennia, most scientists would agree that our planet has been subject to extensive and severe land degradation, mainly as a result of agriculture and forestry, even if detailed evidence are scattered (Dupouey et al. 2002; Xinying et al. 2012; Kates et al. 1990). In a shorter time perspective, say decades, science has been able to more accurately detect and describe significant changes of the face of the Earth. In terms of climate change, since 1850, about 35% of the human caused emissions of CO_2 to the atmosphere comes from land use change (Foley et al. 2005) and nearly 40% of Earth's land area has been converted to agriculture (Foley et al. 2011).

Not all human impacts on land are considered degradation, according to the definition of land degradation used in this report; some impacts are positive, 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). Land use is a socio-economic process which moves land from a natural to a used state, but how the land is used determines whether the land use is sustainable or will lead to degradation over time.

Land degradation was long subject to a polarised scientific debate between disciplines and perspectives in which social scientists often perceived 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). There are three important reasons for including the land users' views and experiences in assessing land degradation: because of the complexity of land degradation processes, measurements become more realistic; the assessment becomes more integrated and hence relevant for the land users, this also includes the perception of potential links to climate change; assessments that include the users' views increases the chances of implementing any measures (Stocking and Murnaghan 2001).

Calls for integrating policies of land degradation with those of climate change and biodiversity were made in 2005 (Gisladottir and Stocking 2005), but realised more than 10 years later.

4.3 Land degradation in previous IPCC reports

Several previous IPCC assessment reports include brief discussion 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 is 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 are discussed comprehensively in AR5 WGII, but land degradation is not prominent. Land use and land cover changes are treated comprehensively in terms of effects on the terrestrial carbon stocks and flows (Settele et al. 2015) but links to land degradation are 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 be contributing to human migration (Adger et al. 2014), an issue discussed more comprehensively in this chapter (4.9.3,4.9.4). Neither drivers nor 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, has been discussed comprehensively in previous IPCC reports.

The Special report on land use, land use change and forestry (Watson et al. 2000) focused on the role of the biosphere in the global cycles of greenhouse gases (GHG). Land degradation is not addressed in a comprehensive way. Soil erosion is discussed as a possible mechanism for reducing the loss of terrestrial carbon. The possible impacts of climate change on land productivity and degradation is not discussed comprehensively. Much of the report is about how to account for sources and sinks of terrestrial carbon under the Kyoto Protocol.

The SREX report (IPCC 2012) did not provide a definition of land degradation. Nevertheless, it has 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 little explicit links to land degradation. The report also presented the concept of sustainable land management as an effective risk reduction tool.

4.3.1 Definitions of land degradation and land management

In this report, land degradation is defined as a *negative trend* (or persistent decline) *in land condition resulting in long term reduction or loss of the biological productivity of land, its ecological complexity,* *and/or its human values, caused by direct and/or indirect human-induced processes, including climate change*.

The SRCCL definition is derived from IPCC AR5 definition of desertification:

"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 complexity of rainfed cropland, irrigated cropland, or range, pasture, forest, and woodlands resulting from land uses or from a process or combination of processes, including processes arising from human activities and habitation patterns, such as (1) soil erosion caused by wind and/or water; (2) deterioration of the physical, chemical, biological, or economic properties of soil; and (3) long-term loss of natural vegetation" (IPCC WGII 2014; UNCCD 1994, Article 1).

The SRCCL definition is not intended to replace this more detailed definition, but rather to provide an operational definition, that emphasises the relationship between land degradation and climate, for use in this report.

In the SRCCL definition, changes in land condition resulting solely from natural processes (such as earthquakes and volcanic eruptions) are not considered land degradation. Climate variability exacerbated by human induced climate change can contribute to land degradation. The definition recognises the reality that land use decisions are likely to result in trade-offs between time, space, ecosystem services, and stakeholder groups. The interpretation of a negative trend in land condition is somewhat subjective, especially where trade-offs between ecological complexity and human values occur. The use of "and/or" specifies that *either* loss of biological productivity, *or* ecological complexity, *or* human values can constitute degradation, and any one of these changes need not be considered degradation. Thus, a land transformation that reduces ecological complexity and enhances sustainable food production need not be classed as degradation. Different stakeholder groups with different worldviews are likely to value ecosystem services differently.

Land degradation is defined differently in the IPBES Land Degradation and Restoration Assessment (LDRA) as "the many human-caused processes that drive the decline or loss in biodiversity, ecosystem functions or ecosystem services in any terrestrial and associated aquatic ecosystems". The IPBES Thematic Report on Land Degradation (IPBES LDRA) defines degraded land as: "the state of land which results from the persistent decline or loss of biodiversity and ecosystem functions and services that cannot fully recover unaided within decadal time scales" (IPBES 2018). The IPBES LDRA adds further that "Degraded land takes many forms: in some cases, all biodiversity, ecosystem functions and services are adversely affected; in others, only some aspects are negatively affected while others have been increased." Thus, compared to the SRCCL definition, the IPBES LDRA focuses on processes of land degradation, and emphasises biodiversity impacts, while the SRCCL definition focuses on productivity outcomes and implications for human wellbeing. The IPBES LDRA definition (SPM1 and SPM10) appears to convey that all land altered by human management, compared with its natural condition, is considered degraded. The SRCCL, in contrast, views land to be degraded only if its productive potential is diminished. Furthermore, the baseline for SRCCL is not the "pristine state" but rather the condition at the start of the assessment, as it refers only to the trend during the period of interest. The LDRA discusses alternative baselines, but generally favours the natural state.

To clarify the scope of this chapter it is important to define land itself. The SRCCL defines land as "the terrestrial system 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 (adapted from (FAO 2007; UNCCD 1994). Sustainable land management is defined as "the 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 WOCAT). Achieving the objective of ensuring long-term productive potential will require implementation of adaptive management and "triple loop learning", that seeks to learn from experience, modifying management and adjusting accordingly, as new knowledge emerges and impacts are monitored.

4.3.2 Sustainable land and forest management

The SRCCL definitions of land and land degradation are intended to apply equally to forests as well as non-forested land. Nevertheless, explicit definitions for forest degradation and sustainable forest management are provided, to highlight the specific issues relevant to forest management. A conceptual illustration of sustainable land and forest management is shown in Figure 4.1.

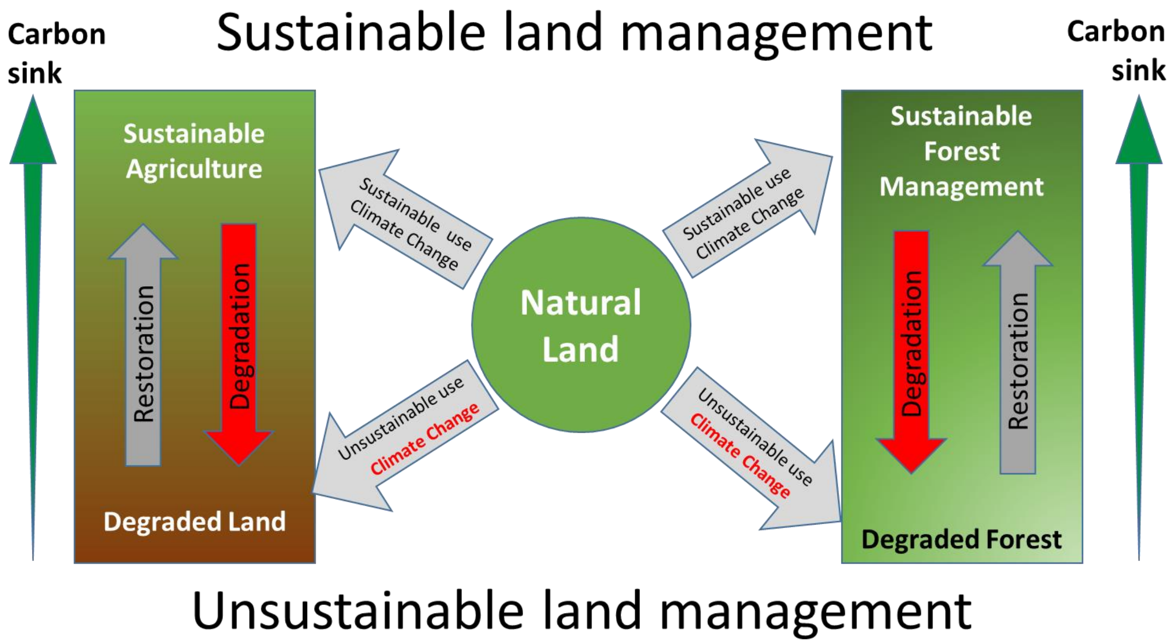

Figure 4.1 Land-use moves land from a natural to a used state – but how the land is managed determines sustainable or degraded outcomes. The types and intensity of human land use and climate change impacts on natural lands affect their carbon stocks and their ability to operate as carbon sinks. In agricultural 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 an indication of a reduction in carbon sinks. Sustainably managed forest landscapes can have a lower biomass carbon density but can contribute stronger carbon sinks then natural forests. To assess the net impact on the atmosphere, the contributions of products removed from agricultural and forest lands as carbon stores (e.g. long-lived wood products in buildings) and to substitute other emissions-intensive products (e.g. though bioenergy use) also need to be evaluated. Climate change impacts can enhance carbon sinks (e.g. through longer growing seasons or higher atmospheric CO_2 concentrations) or have strong adverse effects on productivity through prolonged droughts, extreme events, fires and other disturbances.

Initial attempts to define forest degradation have taken an approach analogue to those used to define land degradation, that is forest degradation is defined as a reduction in the productive capacity of forests, (e.g., IPCC 2013). However, the difficulties in measuring and operationally implementing this definition have been recognised (IPCC 2013) and have resulted in attempts to develop alternate definitions (e.g. (Penman et al. 2003)). More recent definitions focus on reductions in canopy cover or carbon stocks (IPBES 2018), both indicators that remote sensing or other forest inventory methods can measure more easily than reductions in productive capacity. However, the causes of reductions in canopy cover or carbon stocks can be many, including natural disturbances (fires, and insects), direct human activities (harvest, forest management) and indirect human impacts (such as climate change) and these may not reduce long-term forest productivity. In many boreal and some temperate forests 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.

Defining forest degradation as a reduction in productivity, carbon stocks or canopy cover also requires that a baseline is established against which this reduction is assessed. In forest types with rare stand-replacing disturbances, the concept of "intact" or "primary" forest has been used to define a baseline (Potapov et al. 2008; Bernier et al. 2017). 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 range in variability of carbon stocks or canopy density making it more difficult to define the appropriate baseline carbon density or canopy cover to assess degradation. In these systems, forest degradation cannot be defined at the stand level, but requires a landscape-level assessment that takes into consideration the stand age-class distribution of the landscape, which reflects disturbance regimes over past decades (Wagner 1978).

Stand-level degradation can occur in all forest types when selective logging (high-grading) removes valuable large-diameter trees, leaving behind damaged, diseased or otherwise less productive trees and conditions that reduce not only carbon stocks but also adversely affect subsequent forest recovery (Belair and Ducey 2018; Nyland 1992).

The term forest degradation is typically used to describe activities with undesirable outcomes, including losses in productive capacity, losses in biodiversity, losses in the ability to provide goods and services, and other losses (Barlow et al. 2007). However, sustainable forest management applied at the landscape scale can reduce average forest carbon stocks, while increasing the rate at which carbon dioxide is removed from the atmosphere, because Net Ecosystem Production of forest stands is highest in intermediate stand ages (Kurz et al. 2013). Thus, the impacts of sustainable forest management on one indicator (C stocks in the forest) can be negative, while those on another indicator (forest productivity and rate of C removal from the atmosphere) can simultaneously be positive. Moreover, increases in forest productivity can be associated with reductions in biodiversity, as increased productivity can be achieved by periodic thinning and removal of trees that would otherwise die due to competition, and the dead organic matter of snags and coarse woody debris can contribute to biodiversity (Spence 2001; Ehnström 2001).

Instead of seeking to quantify the rates of forest degradation based on vague definitions and weakly defined baselines, scientific and policy communities would be better supported by information on changes in specific forest characteristics which together can identify forest degradation, as this would allow for the assessment of the trade-offs among the various forest characteristics. For example, carbon stocks per hectare, net ecosystem productivity, net biome productivity, albedo and (to some extent) biodiversity are indicators that can be quantified and reported. Improved understanding of past trends and projections of these indicators will enhance the ability to design and implement land management strategies aimed at achieving desired outcomes, including sustainable forest management and activities aimed at reducing atmospheric GHG concentrations as outlined in the Paris Agreement. As long as any form of human impacts on forests is considered degradation and thus undesirable, the opportunities will remain limited to identify and implement sustainable land-use and land-management strategies that allow for the co-existence of forest ecosystems and humans with their requirements for food, fibre, timber and shelter.

The successful implementation of sustainable forest management (SFM) requires well established and functional governance, monitoring, and enforcement mechanisms to eliminate deforestation, illegal logging and other activities that are inconsistent with SFM principles. 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 unstainable outcomes and long-term reductions in forest area, carbon density and forest productivity.

A definition of SFM was developed by the Ministerial Conference on the Protection of Forests in Europe and has since been adopted by the Food and Agriculture Organization. It defines sustainable forest management 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 2016).

Other terms pertinent to this chapter are:

Land potential: The inherent, long-term potential of the land to sustainably generate ecosystem services, which reflects the capacity and resilience of the land-based natural capital, in the face of ongoing environmental change. (UNEP 2016)

Land Restoration: The process of assisting the recovery of an ecosystem that has been degraded. Restoration seeks to re-establish the pre-existing state, in terms of ecological integrity (adapted from (McDonald et al. 2016))

Land Rehabilitation: Actions undertaken with the aim of reinstating ecosystem functionality, where the focus is on provision of goods and services rather than restoration to the pre-existing state (adapted from (McDonald et al. 2016))

4.3.3 The human dimension of land 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 land degradation have often been neglected or ignored (Reed et al. 2007; Forsyth 1996; Andersson et al. 2011). The omission of such perspectives has led to policies which are characterised by scientism (Warren and Olsson 2003) and sometimes neo-Malthusian perspectives (Stringer 2009; Stringer and Reed 2007). A growing body of work is nevertheless beginning to focus on land degradation through the lens of local land users (Kessler and Stroosnijder 2006; Stocking and Murnaghan 2001; Fairhead and Scoones 2005; Zimmerer 1993) and the importance of local and indigenous knowledge within land management decision making is starting to be better appreciated (IPBES 2018). In this report we treat both land degradation and people's responses to it as a relational problem in which land users are interacting with the local ecosystem and climate, while embedded in a multi-scalar social reality. Climate change impacts directly and indirectly the social reality, the land users, and the ecosystem and vice versa. In some cases, land degradation can also have an impact on climate change.

Important aspects of these relationships will be highlighted throughout the chapter. For example, women have often less formal access to land than men and hence 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). The use and management of land is therefore highly gendered. 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, lower access to other capital assets and dominant cultural practices (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. 2018; Lambin et al. 2001a). Land rights are highly context specific and dependent upon the political-economic and legal context (IPBES 2018). This means there is no universally applicable best arrangement. Agriculture in highly erosion prone regions require site specific investments which may benefit from secure private land rights (Tarfasa et al. 2018) while pastoral modes of production are often dominated by communal land tenure arrangements which may conflict with agricultural modernisation policies implying private property rights (Antwi-Agyei et al. 2015; Benjaminsen and Lund 2003; Itkonen 2016; Owour et al. 2011).

4.4 Land degradation in the context of climate change

Several conceptual frameworks have been used in previous scientific assessments. This chapter borrows from frameworks used in other assessments, see (Tomich et al. 2010). The distinction between drivers and processes is clear but the boundary between drivers and pressures is somewhat blurry in the literature. In the DPSIR framework, Drivers are both natural and anthropogenic driving forces (e.g. climate change, population growth), Pressures are human activities affecting the environment, resulting from drivers (deforestation, burning fossil fuels). Processes are the natural phenomena that link pressures to State for example, overgrazing (pressure) leads to erosion (process) which reduces soil fertility (state).

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 (MA) framework and “pressure” in the DPSIR framework (Tomich et al. 2010).

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 MA framework and “drivers” in the DPSIR framework, (Tomich et al. 2010).

An exact demarcation between processes and drivers is impossible to make, for example 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.

4.4.1 Processes of land degradation

A large array of interactive physical, chemical, biological and human processes can lead to what we define in this report as land degradation (Johnson and Lewis 2007). The biological productivity, ecological complexity or the human value of a given territory can be deteriorated as a result of processes triggered at scales that range from a single furrow (e.g. water erosion under cultivation) to the landscape level region (e.g. salinisation through raising groundwater levels under irrigation). While the "entry" point of these land degradation processes can be the soil, water, or plant and animal populations, most land degradation phenomena propagate to the rest of the components, turning into more complex phenomena. Hence, the influence of climate variability and change on land degradation can originate from its direct effects on any of these spatial scales and entry points. or from its indirect effects in the way humans use and treat the land.

4.4.1.1 Types of land degradation processes

Soil degradation has received more attention than other forms of land degradation. The most widespread and studied soil degradation processes are water and wind erosion, which have accompanied cultivation since its onset and are still dominant (Table 4.1). Degradation through erosion processes is not restricted to soil loss in the eroded areas but can also include impacts on transport and deposition areas as well. 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 siltation of natural and artificial water bodies (Poesen and Hooke 1997; Ravi et al. 2010). Other physical degradation process in which no material detachment and transport are involved include soil compaction, hardening, sealing and any other mechanism leading to the loss of pore space. Chemical soil degradation process ranges from nutrient depletion, resulting from the imbalance of extraction and fertilisation, to more complex processes of acidification and increasing metal toxicity. One of the most relevant chemical degradation processes of soils in the context of climate change is the depletion of its organic matter pool. Favoured by increasing respiration rates (e.g. through cultivation) and reduced organic inputs (e.g. diminished plant inputs under agriculture), declining soil organic matter pools have cascading effects on the degradation of soil health and the preservation of ecosystem carbon stocks.

Not all land degradation processes start in the soil and those starting from alterations in the hydrological system 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). Recurring flood and waterlogging episodes (Bradshaw et al. 2007; Poff 2002), and the more chronic expansion of wetlands over dryland ecosystems (e.g. paludification) 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, for example terrestrialisation, wetland drainage and salinisation, drying of lakes and inland seas (Anderson et al. 2003; Micklin 2010; Herbert et al. 2015).

Land degradation can also be initiated by purely biotic processes. Vegetation alterations in natural or semi-natural ecosystems are a widespread mechanism of land degradation. Woody encroachment and the "thicketization" of open savannahs involve the expansion of woody plant cover and/or density over herbaceous areas and often limits the secondary productivity of rangelands (Asner et al. 2004). These processes have been accelerated since the mid-1800s over most continents (Van Auken 2009). Change in plant composition of natural or semi-natural ecosystems without 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 stable compositions with lower forage value (Illius et al.; Sasaki et al. 2007) but with higher carbon sequestration potential. In forests, selective logging is a pervasive cause of degradation which can lead 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.). Invasive exotic species are another source of biological degradation. Their arrival into cultivated systems is constantly reshaping crop production strategies making agriculture unviable on occasions. In rangelands invasive species not only threaten livestock production through diminished forage quality, poisoning and other deleterious effects, but have cascading effects on other processes such as fire regimes and water cycling.

4.4.1.2 Land degradation processes and climate change

Many land degradation processes are affected and/or affect climate change. Here we identify the most accepted and well documented links across the broad groups of soil, water and plant mediated degradation processes. Importantly, soil erosion is not only a geomorphological agent, but it can also cause mobilisation of soil contaminants such as heavy metals, persistent organic pollutants, herbicides and pesticides, as well as excess synthetic fertilisers (Li and Fang 2016).

The most important land degradation processes are listed in Table 4.1(this table is in progress).

Table 4.1 describes land degradation processes, how they are impacted by climate change, and how they may feedback on the climate system

Process	Focus	Proximate Drivers	Influence OF Climate Change	Influence ON Climate Change	References for CC->LD	References for LD->CC	General Reviews of the process
Compaction / Hardening	Soil	Machinery overuse, intensive grazing , poor tillage/grazing management (e.g. under wet or waterlogged conditions)	Indirect, reduced SOC due to higher temperatures - More frequent wet/waterlogged periods	Complex effects on GHG emissions. Poor aereation has ambiguous effects on N_2O emissions (Ball 2013).		(Ball 2013)	(Hamza and Anderson 2005)
Wind erosion	Soil	Cultivation with poor cover, overgrazing, deforestation/vegetation clearing, Larger plot sizes, vegetation shifts - Documented reversal with vegetation restoration projects (Guo, et al 2017)	Altered wind/drough patterns. Yet, no strong trends for combined humidity/wind speed/available energy assessments (Sheffield et al. 2012). Land use/cover more important than climate in US plains (Nordstrom & Hotta 2004). No clear long term trend on wind climate driving erosion in Sweden (Barring et al. 2003 - Catena), Climate change induced vegetation change enhance wind erosion (Munson et al. 2011)	Radiative cooling by aerosols (Tegen et al. 1996). Enhanced weathering + Ocean and Land fertilisation + SOC burial (Quinton & Govers 2010)	Sheffield et al. 2012 - Nature, Barring et al. 2003 - Catena), Munson et al. 2011 - PNAS	Tegen et al. 1996 - Nature, Quinton & Govers 2010 - Nature Geosciences	Nordstrom & Hotta 2004 - Geoderma, Sterk 2003 - LDD

Water erosion	Soil/Water	Cultivation with poor cover, overgrazing, deforestation/vegetation clearing, poor tillage practices - More indirect: fires, vegetation shifts	Increasing rainfall intensity, drought and vegetation shifts (i.e. forest to woodland with climate change (Allen & Breshears 1998). Rainfall amount increases are amplified by erosion and run-off (Nearing et al. 2004). Increasing fire activity raises erosion (Shakesby 2011), Permaforst melting causing erosion (Jorgenson & Osterkamp 2005), Climate change effects mediated by biomass production appear more important than those through rainfall in modeling study (Pruski and Nearing 2002)	C emissions may be significant globally (approx 1 PgC yr^{-1}) Lal 2003	Allen & Breshears 1998 - PNAS, Nearing et al. 2004 - J Soil and Water Conservation, Shakesby 2011 - Earth Science Reviews, Jorgenson & Osterkamp 2005 - Can J Forest Research, Pruski and Nearing 2002 - WRR	Lal 2003 - Environment	Pesen et al. 2003 - Catena (Gully erosion),
Nutrient depletion	Soil	Insufficient replenishment of harvested nutrients		Reduced SOC, C release from soils - NO SYNTHESIS WORK FOUND			
Acidification	Soil	High cation depletion, Fertilisation, Acid rain	--	Inorganic soil C release?			
Toxicity	Soil	High cation depletion, Fertilisation	--	--			
Organic matter decline	Soil	Cultivation, reduced plant input, higher decomposition	Raising temperature accelerating SOC turnover (Knorr et al 2002), warming explaining widespread	Cultivation release of C through global cultivation expansion	Knorr et al 2002 - Nature, Bellamy et al. 2005 - Nature, Bond-	Houghton 2003 - Tellus, Kurganova et al. 2014 -	Conant et al 2011 - Global Change Biology (response of

			SOC decline in UK (Bellamy et al. 2005), Accelerated global soil respiration due to warming (Bond-Lamberty & Thompson 2010)	(Houghton 2003). Reversal due to land abandonement in Russia (Kurganova et al. 2014)	Lamberthy & Thompson 2010 - Nature	Global Change Biology	SOC decomposition to raising temperature)
Sodification	Soil / Water	Poor water management	Water balance shifts (precipitation/potential evapotranspiration shifts)	Effects of high alkalinity on GHG release			
Salinisation	Soil / Water	Irrigation with poor leaching+drainage, Deforestation	Sea level raise, Water balance shifts	Reduced methane emissions with high sulfate			
Waterlogging of dry systems	Water	Deforestation, Irrigation with poor drainage	Water balance shifts, rainfall increase, vegetation changes (forest to cropland-grassland)	Rewetting of dry peatland releases of CH_4 (Fenner et al. 2011), Artificial riparian wetlands release CH_4 (Altor and Mitsch 2006)		Fenner et al. 2011 - Hydrobiologia, Altor and Mitsch 2006 - Ecological Engineering, Hahn-Schofl et al. 2011 - Biogeosciences	
Drying of continental waters/wetland/lowland systems	Water	Upstream or Groundwater water consumption, intentional drainage, trampling/overgrazing, droughts	Extended drought causing vegetation dieback and soil degradation (McKee et al. 2004)	C stock reduction / C release in drying montane meadows (Norton et al. 2011), N_2O release from dried wetlands (Morse & Bernhardt 2013)	Mc Kee et al. 2004 - Global Ecology and Biogeography	Norton et al. 2011 - Ecosystems, Morse and Bernhardt 2013 - Soil Biol & Bioch	

Flooding	Water	Deforestation, Increasing impervious surface	Sea level raise, Water balance shifts, Increasing flood frequency (Milly et al 2002), Increased rainfall extremes in UK (Pall et al. 2011), Rapid climate change causes more floods now and in paleorecords (Knox 2000)	CH_4 and CO_2 release from flooded soils. Boreal soils under reservoirs (Oelbermann & Schiff 2010). Abundant literature on paddy rice and methane.	Milly et al. 2002 - Nature, Pall et al. 2011 (attribution of extremes to CC), Knox 2000 - Quaternary Science Reviews	Oelbermann & Schiff 2010 - Ecoscience
Eutrophication of continental waters	Water	Excess fertilisation, Poor management of livestock/human sewage	Only indirectly (e.g. warming favouring N losses in the land) or interactively (warming x eutrophication compensating effects - Audet et al. 2017), Warming and algal blooms (Paerl & Huisman 2008)	Complex interaction between warming and eutrophication modulating GHG release in shallow lakes (Audet et al. 2017). High potential release of GHG with EUTROPH in humic lakes (Fenner & Freeman 2013). Also in reservoirs (Hattunen et al. 2003) - Vast literature on constructed wetlands for nutrient treatment and GHG emissions (e.g. Gui et al. 2007)	Audet et al. 2017, Paerl & Huisman 2008 - Science	Audet et al. 2017 - Freshwater Biology, Fenner and Freeman 2013 - Global Change Biology. Gui et al. 2007 - Water Science and Technology. Hattunen et al. 2003 - Chemosphere

Woody encroachment	Plant	Overgrazing, Altered fire regimes	CO_2 raise, Climate change (moderation?) (Van Auken 2009, Wigley et al. 2010)	C sequestration interactive with climate gradients (Knapp et al 2008, Sala & Maestre 2014)	Van Auken 2009 - J Env Management, Wigley et al. 2010 - Global Change Biology	Knapp et al. 2008 - Global Change Biology, Sala & Maestre 2014 - J Ecology	
Valued species loss	Plant /Animal	Selective grazing and logging	Multiple	Multiple			
Invasions	Plant /Animal	Intentional and unintentional species introductions	Multiple	Multiple			
Insect outbreaks	Plant /Animal	Poor pest management practices	Multiple	Multiple			
Fire regime shift	Plant /Soil	Biomass accumulation, intentional burning, fire suppression policies	Warming, Intensification of dry/wet alternative spells	massive C release in tropical peat fires (Page et al. 2002)		Page et al. 2002 - Nature	Certini 2005 - Oecologia (effects on soils)

4.4.2 Drivers of land degradation

Drivers of land degradation and land improvement are many and they interact in multiple ways. In Figure 4.2 we illustrate how some of the most important drivers are interacting with the land users. It is important to keep in mind that both natural and human drivers can drive both degradation and improvement (Kiage 2013).

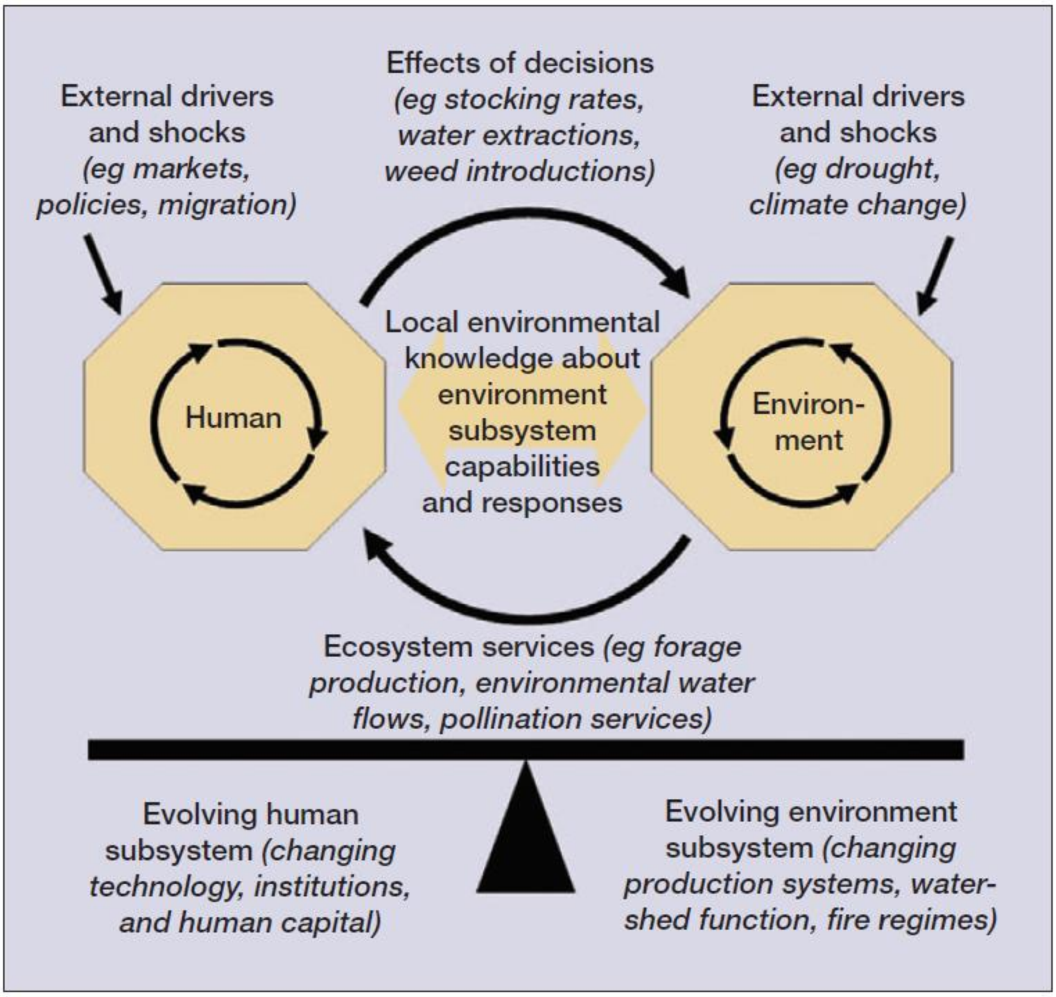

Figure 4.2 Drivers of land degradation (Verstraete et al. 2009) [*we will develop our own version of this figure, but this is a good starting point*]

Land degradation is sometimes considered to be a creeping phenomenon, controlled by slow variables (Walker et al. 2012; Reynolds et al. 2011). Examples of such slow variables are depletion of nutrients or a gradual reduction of ecosystem services such as water holding capacity. But it is important to realise that land degradation is driven by the entire spectrum of factors, from very short and intensive events such as an individual rain storm of 10 minutes (Coppus and Imeson 2002; Morgan 1995) to century scale slow depletion of nutrients or loss of soil particles (Johnson and Lewis 2007, p. 5-6). But instead of focusing on absolute temporal variations, the drivers of land degradation should more appropriately be assessed in relation to the rates of possible recovery. Studies suggest for example, that erosion rates of conventionally tilled agricultural fields exceed the rate at which soil is generated by one to two magnitudes of order (Crews et al. 2018). 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 both gradual changes of temperature and precipitation, and changes of the distribution and intensity of extreme events. Importantly, these drivers can act in two directions: land improvement and land degradation.

The gradual and planetary changes that can cause land degradation/improvement have been studied by global integrated models. Studies of global land suitability for agriculture suggest that climate change will increase the area suitable for agriculture 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).

In the study of recent trends in vegetation dynamics over South America, Barbosa et al. (2015) found the vegetation degradation is coupled to decline in amount of rainfall in some areas. Douglas (2006) studied local drivers of land degradation in South East Asia and identified long drought and deficit rainfall are the major causes.

It is also worth noting that a rise in air temperature and subsequent increase in potential and actual evapotranspiration will have an impact on land degradation through impeding vegetation growth (Li et al. 2013; Madhu et al. 2015). Barbosa and Lakshmi Kumar, (Barbosa et al. 2015) used the Sea Surface Temperatures of Nino 3.4 region and Atlantic Dipole regions to study the persistent droughts to understand the long term land degradation over Brazil and found a strong linkage between the El Nino and droughts between 1979 and 2000.

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 paint a very bleak picture in which 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 previously sustainable land use and land management practices unsustainable and vice versa (Laderach et al. 2011).

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, are necessarily causing land degradation (Lambin et al. 2001b). However, drivers of land degradation need to be studied in the context 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 can also be affected indirectly by climate change through changing patterns of wildlife habitats and wildlife densities (Ims and Fuglei 2009; Aryal et al. 2014; Beschta et al. 2013).

4.4.3 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. There is not much explicit research that addresses this question (there is more on climate change as a threat multiplier for land degradation) but we may be able to infer climate change impacts on land degradation. Section 4.4.3.1 will outline the potential direct linkages of climate change on land degradation based on current theoretical understanding of land degradation processes and drivers. Section 4.3.3.2 will investigate possible indirect impacts on land degradation.

4.4.3.1 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. The conceptual illustration below (Figure 4.3) shows how the climatic factors temperature and rainfall, may influence soil erosion rates (Li and Fang 2016). 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).

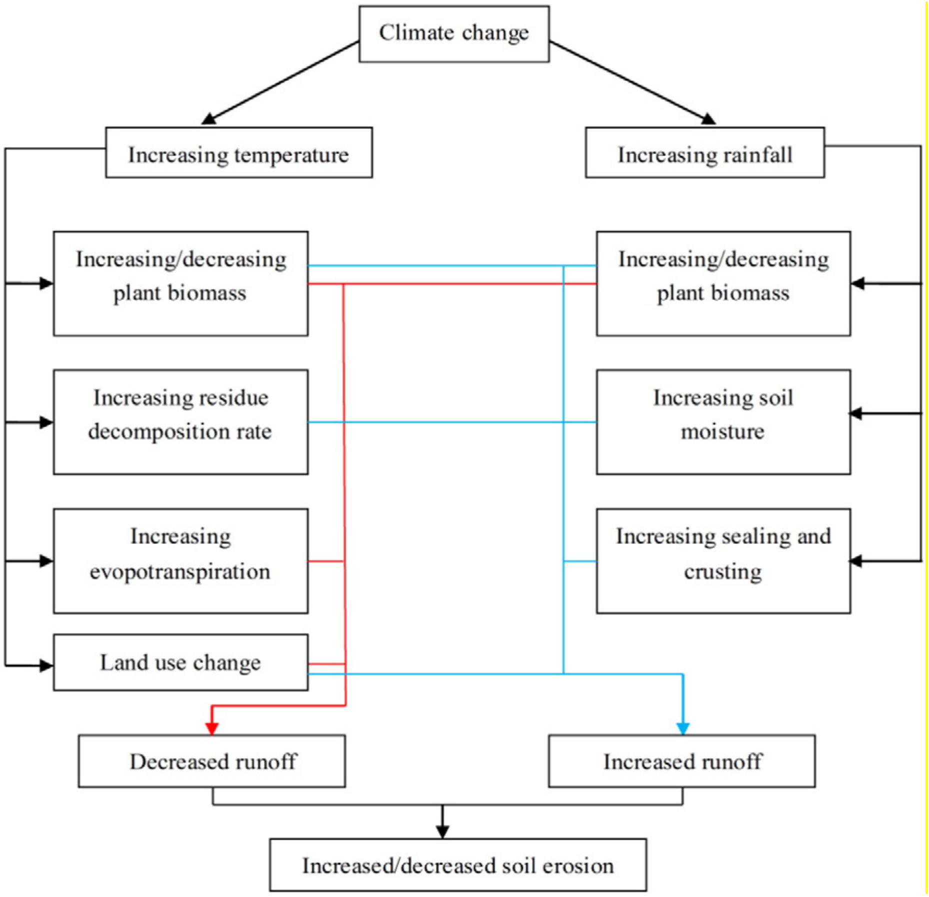

Figure 4.3 Conceptual illustration of how climate change interacts with different land degradation processes (Li and Fang 2016).

Theoretically the intensification of the hydrological cycle as a result of human induced climate change is well established (Trenberth 1999) and also empirically observed (IPCC 2014; Burt et al. 2016a; Liu et al. 2009). AR5 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, 2014, p. 7). In central India, there has been a threefold increase in widespread extreme rain events during 1950–2015. This has influenced several land degradation processes, not least soil erosion (Burt et al. 2016b).

Climate change may alter regional rainfall regimes. The idea that already wet regions get wetter and already dry regions get drier (Held and Soden 2006; Trenberth 2011) is contested (Knapp et al. 2015; Huang et al. 2015; Byrne et al. 2015; Greve et al. 2014). But if 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 other aspects of land degradation. Vegetation cover, for example is a key factor in determining soil loss through both water (Nearing et al. 2005) and wind erosion (Shao 2008).

Rainfall intensity is a key determinant of soil erosion. There are reasons to believe that increases in rainfall intensity can even exceed the rate of increase of atmospheric moisture content (Liu et al. 2009; Trenberth 2011). 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). This will result in increases in the intensity of rainfall which will increase the erosive power of rainfall (erosivity) and hence increase the risk of water erosion. Erosivity is highly correlated to the product of total rainstorm energy and the maximum 30 minute rainfall intensity of the storm (Nearing et al. 2004a) and increases of erosivity will exacerbate water erosion substantially (Nearing et al. 2004b). However, the effects will not be uniform but highly variable across regions (Almagro et al. 2017; Mondal et al. 2016a).

The most comprehensive database of direct measurements of water erosion to our knowledge contains 4377 entries, even though not all entries are complete. An important finding from that database is that almost any erosion rate is possible under almost any climatic condition (García-Ruiz et al. 2015a), a finding which emphasises how extremely important land management is for controlling erosion. Some important 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 (*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 annual events. 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 the tendency of 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).

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 is manifested in very high rates of soil loss, it can reach over 100 tons ha^{-1} yr^{-1} (García-Ruiz et al. 2015b; Tadesse 2001). In temperate regions, the off-site effects 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).

The distribution over time of wet and dry spells is also expected to be affected although uncertainties still remain depending on for example 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 evaporation (evapotranspiration and evaporation) and may influence the partitioning of water into surface and subsurface runoff (Li and Fang 2016; Nearing et al. 2004c). This portioning of can have a decisive effect on erosion (Stocking and Murnaghan 2001, p. 5-6).

Wind erosion is a serious problem in agricultural regions, but studies in Europe suggest that climate change will not alter wind patterns in a way that can significantly affect the risk of wind erosion (Pryor and Barthelmie 2010).

Direct temperature effects on soils are of two kinds. Firstly, permafrost melting 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 (N) and carbon (C) in soils. 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,c; Wu et al. 2011).

4.4.3.2 Indirect and complex linkages with climate change

Many important indirect linkages between land degradation and climate change occur via agriculture. More negative impacts have been observed than positive ones. After 2050 the risk of severe yield losses increase as a result of climate change in combination with other drivers (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 intensification of land use 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 water logging. 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). From an attribution point of view, such a complex web of causality is problematic if attribution is only done through statistical 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 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 the new climate regime may alter soil erosion rates.

4.4.4 Approaches to assessing land degradation

Processes that lead to land degradation and their biophysical, socio-economic, and cultural drivers across multiple temporal (historical dimension important) and spatial scales (both bottom-up and top-down), including concepts of resilience and tipping points.

In a review of different approaches and attempts to map global land degradation, (Gibbs and Salmon 2015) identified three main approaches to map the global extent of degraded lands (for the purpose of estimating the extent and locations for possible expansion of bioenergy): expert opinions (Oldeman and van Lynden 1998; Dregne 1998; Reed 2005; Bot et al. 2000), satellite derived trends of vegetation greenness (NDVI) (Yengoh et al. 2015; Bai et al. 2008c), and biophysical models (Cai et al. 2011). There were large differences between the approaches except that they generally agreed about the extent and location of non-degraded areas. There is, however, a strong consensus that satellite based remote sensing is the only affordable and practical way to assess and monitor land degradation even if there are still knowledge gaps to be filled (Wessels et al. 2007, 2004; Prince 2016).

4.4.4.1 Assessment by modelling (LO)

There are now a myriad erosion models available for various scales varying from individual patches, to catchments and landscape levels and even to the globe. See further (Gliński et al. 2011; Morgan and Nearing 2011) and the homepage: http://soilerosion.net/dd_models.html. Erosion models can be divided into empirical and process-based models.

At the top-down approach there are several indexes that have been used to assess land conditions and monitoring the changes of land condition. The RUSLE (Revised Universal Soil Loss Equation) can be used to predict the long-term average annual soil loss by water erosion.

4.4.4.2 Assessment by remote sensing

Some forms of land degradation can be measured directly at local scales by remote sensing methods, for example extent of gullies, severe forms of rill and sheet erosion, deflation etc. Changes in frequency, intensity, duration and spatial extent of fires as drivers of land degradation can also be monitored directly. Other forms of land degradation, such as nutrient depletion or other forms of changes to soil physical or biological properties cannot be measured directly by remote sensing. In such cases and for monitoring of land degradation over large areas remote sensing offers proxies for land degradation and is the only method for consistent monitoring of large areas over several decades. The presence of open-access, quality controlled and continuously updated global databases of remote sensing data is invaluable. Remote sensing can provide meaningful proxies in terms of severity, temporal development, and aral extent, but to understand the processes and drivers at hand require other types of data, or at least remote sensing data of much higher spatial and/or spectral resolution, and ground observations or measurements (Sedano et al. 2016; Brandt et al. 2018b; Turner 2014)

Inter-annual vegetation dynamics, such as above-ground net primary productivity and vegetation phenology, can effectively and accurately be measured by satellite born sensors. Several vegetation indices have been described and evaluated which can be used as scales from the global level to sub-national level (Yengoh et al. 2015).

The NDVI (Normalised difference vegetation index) is one of the most commonly used proxies to assess land degradation. Moreover, there are major factors confounding the relationship between NDVI (NPP) trend and human-induced land degradation. First, the effect of inter-annual rainfall variation that can be corrected by different methods, considering this factor. On the other hand, the effect of atmospheric fertilisation caused by elevated levels of CO_2 and NOx in the atmosphere (Dentener 2006; Reay et al. 2008) also complicates the global assessment of land degradation using the NDVI-based approach. The rising level of atmospheric fertilisation of CO_2 stimulates photosynthesis in plants' leaves, thus increasing NPP, but the soil fertility may not necessarily be proportional to the above ground biomass improvement (Lee et al. 2016). This is also necessary to correct the NDVI data for the current and predicted atmospheric fertilisation of CO_2. Additionally NDVI values can be affected by several site- and land cover-specific factors (Smith et al. 2014; Mbow et al. 2013), different locations with the same NDVI value are not necessarily have the same biomass productivity. There is ample evidence of regionally-differentiated responses to environmental changes – growth enhancement in forests due to CO_2 fertilisation is strongly controlled by water availability.

It is important to emphasise that an increase in NPP does not always indicate improvement in land condition/reversal of land degradation. It could for example result from 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-medium term but may reduce resilience, by increasing risk of soil salinisation (Niedertscheider et al. 2016), hence the importance of corroborating remote sensing data with other sources of information.

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 2016 - Global Change Biology). The most common proxy is VOD (vertical optical depth) and has already been used to describe global forest/savannah carbon stock shifts over two decades highlighting strong continental contrasts (Liu et al. 2015 - Nature Climate Change) and demonstrating the value of this approach to monitor forest degradation at large scales.

Distinction between land degradation/improvement and the effects of climate variation is an important and contentious issue. There is no simple and straightforward way to disentangle these two effects. The interaction of different determinants of primary production is not well understood and a critical limitation to such disentangling is a lack of understanding of the inherent inter-annual variability of vegetation (Huxman et al. 2004; Knapp and Smith 2001; Ruppert et al. 2012; Bai et al. 2008a; Jobbágy and Sala 2000).

In some studies, based on remote sensing, the authors may have misunderstood the relationship between rainfall and above-ground net primary production (ANPP). The most commonly quoted source of the concept Rain Use Efficiency is Le Houerou who worked extensively on methods and theories for estimating range productivity in drylands (Le Houérou 1996; Le Houerou 1984; Houerou and Hoste 1977). He noted a strong correlation between mean annual rainfall and plant productivity, on average 4 kg dry matter per hectare and mm rainfall (Houerou and Hoste 1977; Le Houerou 1984). Similar empirical studies had been conducted earlier with very similar results (Cook and Sims 1975). It is important to note that both Le Houerou and Cook & Sims highlighted that the high correlation was between mean annual rainfall (or in some cases mean seasonal rainfall) and plant productivity, and they stressed that the relationship between annual rainfall and annual plant productivity was much weaker (Cook and Sims 1975; Houerou and Hoste 1977), which has been confirmed in later studies (Lauenroth and Sala 1992; Gamoun 2016). In a study of LTER sites across the US and Latin America, Huxman et al. (2004) found that ANPP was more strongly correlated with the maximum temperature and the ANPP the previous year than with annual rainfall for the most productive sites, while ANPP was most strongly correlated with annual rainfall for the least productive sites.

Rainfall influences vegetation dynamics in different ways across biomes and across species. As shown by several studies above, ANPP might for some biomes be indirectly driven by rainfall over several years (Ponce Campos et al. 2013; Michaletz et al. 2014; Jones et al. 2016) while the direct impacts on ANPP might be negligible (Michaletz et al. 2014). This will cause problems for assessing how rainfall and ANPP interact at the landscape level.

The rainfall – ANPP relationship varies across biomes, soil and vegetation types. For example woody / herbaceous and annual / perennial, will respond differently to rainfall (Ruppert et al. 2012). Most of the studies above are valid for woody vegetation, but also herbaceous vegetation composition seems to be responsive to rainfall regimes in a similar way (Jones et al. 2016). In Tunisia, vegetation on loamy soils responded much more to drought conditions than other soils.

The rainfall – ANPP relationship varies across eco-climatic zones. Rainfall is a stronger driver of ANPP in drylands (Dardel et al. 2014) than in more humid areas (Huxman et al. 2004). But according to Huxman, all biomes seem to converge to a common (Gamoun et al. 2011) RUE, corresponding to 3.6 kg DM ha^{-1} yr^{-1} mm^{-1} annual rainfall, during the driest year for each biomes (Huxman et al. 2004), a value which is very similar to what Le Houerou came up with (4 kg ha^{-1} yr^{-1} mm^{-1} rainfall).

Rain-use efficiency is often assumed to be a conservative measure, that is, it is constant over time for a given biome in the absence of any non-climatic stressor, but there are often substantial time lags between rainfall anomalies and vegetation response (Lauenroth and Sala 1992).

4.4.4.3 Assessment by field-based methods

Direct measurements of soil erosion has been undertaken in many parts of the world since the early 20th century, but such cases are spatially limited and very unevenly distributed across regions, with most studies from the USA, followed by Southwestern Europe (The Mediterranean region), and only scatted studies in other parts of the world (García-Ruiz et al. 2015a). Such data are nevertheless crucial as reference cases for studies using proxies, either from ground-based methods or from remotely sensed methods.

At the ground (bottom-up) scale there are multiple indicators that reflect functional ecosystem processes linked to ecosystem services. These indicators are a composite set of measurable attributes from different factors, such as climate, soil, vegetation, management, among others, that can be used together or to develop indexes to better assess land degradation (Allen et al. 2011; Kosmas et al. 2014).

From these indicators, we can go deep in some with high relevance to assess climate change impacts: Changes in the rainfall seasonality and evapotranspiration as an effect of the climate change will alter the plant and microbial activity and productivity, producing affectations at the landscape scale such as water stress, water erosion, forest fires and overgrazing (Table 4.2)(Kosmas et al. 2014).

Among the indicators that has been proposed in the soil properties, the soil organic matter (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 on water holding capacity, increased risk of erosion and increased bulk density and hence soil compaction (Allen et al. 2011; Conant et al. 2011b; Certini 2005). Although there is not a consensus in the response of SOM to elevated temperature (Ågren and Wetterstedt 2007), in general, increases in temperature have been reported to enhance decomposition of SOM, but the combined effect of rising temperature and precipitation, CO_2 fertilisation and atmospheric N deposition may support high plant productivity and organic matter input to soil and consequently increase SOM (Allen et al. 2011).

Soil microbes are the main drivers for the nutrient cycling and C dynamics (Singh et al. 2010a; McGuire and Treseder 2010). The composition of the microbial community is very likely to be impacted by both the climate change and the land degradation processes (Holden and Treseder 2013; Pérez-Valera et al. 2015; Lau and Lennon 2012; Evans and Wallenstein 2014; Wu et al. 2015; Classen et al. 2015). Abiotic disturbances resulting from climate change effects (e.g. alteration of precipitation regimes and temperature increases) may significantly decrease soil microbial abundance, with corresponding consequences for microbial activity (Holden and Treseder 2013). The effects of these reductions on soil microbial abundance will depend on its functional redundancy, but are likely to have a negative impact on the ecosystem functioning of most parts of ecosystems (Holden and Treseder 2013; Pérez-Valera et al. 2015; Lladó et al. 2017; Singh et al. 2010a). On the other hand, the altered soil moisture and temperature regimes in present climatic scenarios directly or indirectly affect soil microbes which can have positive or negative feedback to climate changes. Elevated CO_2 significantly affects photosynthesis, which modifies the rhizodeposition, soil C pools, and nutrient dynamics, and subsequently alters microbial activities (Singh et al. 2010a).

Table 4.2 Land degradation: key indicators to process and functions under projected climate change scenarios

Type	**Indicator**	**Process affected**	**Landscape scale (direct determination or estimated from functions)**
Climate	Rainfall seasonality	Plant productivity, microbial activity	Water erosion, water stress, overgrazing, forest fires
	Evapotranspiration	Plant productivity, soil water availability	Water erosion, water stress, forest fires
	Water runoff	Relative field capacity, nutrients leaching	Soil salinisation, water stress
Soil	Porosity	Air capacity, plant available water capacity, relative field capacity	Soil crusting, reduced seed germination, aeration, water entry, compaction

	pH	Biological and chemical activity thresholds	Soil acidification, soil salinisation, soil structure
	Soil organic matter	Plant residue decomposition, metabolic activity of soil microorganisms, mineralization and immobilization of nutrients	Loss of organic matter, soil aggregate formation, nutrient supply, tillage erosion, compaction
	Soil respiration	Microbial and root activity	Microbial and root activity
	Microbial biomass C and N	Microbial activity	Soil structure, nutrient supply, pesticide degradation
	Plant cover	Plant productivity, mineralization and immobilization of nutrients	Water erosion, soil structure, microbial and root activity, nutrient supply, overgrazing
	Plant productivity	Mineralization and immobilization of nutrients	Biomass production, nutrient supply, overgrazing, water stress, water erosion
Vegetation	Forest fires (risk and frequency)	Plant productivity, plant residue, metabolic activity of soil microorganisms	Overgrazing, water stress, water erosion
	Land abandonment	Plant productivity, nutrient cycling	Overgrazing, water erosion, water stress
	Land use intensity	Plant productivity, nutrient cycling and availability	Water erosion, tillage erosion, forest fires
Management	Population density	Nutrient cycling	Soil salinization, water erosion, water stress
	Policy implementation	Nutrient cycling	Water erosion, tillage erosion, water stress

Adapted from (Kosmas et al. 2014; Allen et al. 2011; Bünemann et al. 2018)

4.5 Status and trends of land conditions

4.5.1 Land

There are no reliable global maps of the extent and severity of land degradation. The reasons are both conceptual, that is, how is land degradation defined, over what time period, etc. and methodological. Even if 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, if land degradation is an ongoing process or is an end result, and of course how this can be studied. One widely used global assessment of land degradation used 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 indicated that between 22% and 24% of the global land area was subject to a downward trend, while about 16% showed an increasing trend. There was no simple relationship between downward trends and population density. The study also suggested, contrary to earlier assessments (Middleton and Thomas 1997), that drylands were not among the most affected regions.

An assessment of the global severity of soil erosion in agriculture, based on a large number of published scientific studies around the world, indicated that the global net median rate of soil formation (i.e. formation minus erosion) is about 0.004 mm yr^{-1} compared with the median net rate of soil loss in agricultural fields, 1.52 mm yr^{-1} in tilled fields and 0.065 mm yr^{-1} in no-till fields (Montgomery 2007a). This means that the rate of soil erosion from agricultural fields is between 360 and 16 times the natural rate of soil formation. Climate change, mainly through the intensification of rainfall, may further increase this rate unless land management is improved.

It is known that the land degradation occurs due to inadequate land management practices, often in combination with unfavorable climatic resources. The stressful interaction between land surface and atmosphere changes the surface energy budget. It is also reported that destructive utilisation of wild life, plant production and destructive vegetation which affects the water cycle have profound impacts on human life by contribution to land degradation (Wala et al. 2012; MA (Millennium Ecosystem Assessment) 2005). Recurring droughts coinciding with high temperatures, heat waves, is conducive to bush-fires which have tremendous impact on land degradation Watkins (2005) and the recent example from California). The trends of land degradation are alarming and it is essential to understand the contribution of climate effects and human induced (Evans et al, 2004). Barrio et al. (2016) studied the land degradation trends in the northwestern Maghred Drylands during 1998 to 2008 and found the semiarid zones are highly vulnerable to land degradation. Schulz et al, 2011 reported that the controlling factors of vegetation dynamics will be useful in predicting the land degradation trends. It is also reported that the general trends for the native forest loss is mainly emerge from the predominant biophysical conditions such as shifting the areas of cultivation, human density variations (Geist and Lambin 2002). It is also interested to note that the abandoned crop lands in Chile turned to be regenerative for native forests (Díaz et al. 2011). Dimobe et al. (2015) studied the driving factors of land degradation in the wildlife reserve of Bontioli (West Africa) and reported the protected areas are threatened by land degradation and deforestation. They also reported that the agricultural expansion and wood cutting activities are the direct causes of land degradation and deforestation. Safirel and Adeel (2005) and Stafford Smith et al. (2009) studied the dry land dynamics and found the increasing drought frequency, food insecurity, poverty, migration and social disintegration are the driving indicators for the land degradation. The impact on livestock production, irrigation and land cover reduction due to demographic, technical and climate conditions led to the different syndromes of desertification (Petschel-Held, et al. 1999; Reynolds et al. 2007). (Huber-Sannwald et al. 2012) reported the over usage of land for several reasons for so many years changed the landscape of Amapola, Mexico. Drylands are frequently experiencing droughts where in the biomass production and crop yields mainly affect. Anthropogenic activities viz-a-viz over cultivation, overgrazing may also show significant degradation of vegetation, thus land (Keller and Goldstein 1998). Hence, the trends in the land degradation can be detected by the trends in vegetation dynamics (Barbosa et al. 2015), prolonged deficit rainfall conditions (Stroosnijder 2007) and climate change and variability (Barbosa and Kumar 2012). (Madhu et al. 2015) used the Standardised Difference Vegetation Index (SDVI) to study the drought conditions in North East Brazil and reported that the rainfall is an important parameter in understanding the dynamics of vegetation. Increased fire weather-risk enhance the hot fires due to accumulated dry highly flammable biomass that become susceptible to land degradation (Dube 2007). Barbosa and Lakshmi Kumar (2012) stressed the importance of improving remote sensing capacities to understand the long-term land degradation in the context of climate change and variability.

A survey on land degradation by (Graham 1992) over New South Wales, Australia reports the rill erosion in cropping lands, dry land and irrigation salinity in irrigation and induced soil acidity in pasture lands, resulted the increased land degradation. As the agricultural production in the present years has been affected by the soil degradation, the Australian National Soil Research Development and Extension Strategy, noticed that the soil security will be of a major concern for future agricultural productivity in Australia. It is reported that the effect of soil acidification is about 50% of the cropping land (Australia State of Environment, 2011), the effect of soil carbon loss is about 40% to 60% over the Australian agricultural soil (Sanderman and Baldock 2010). (Koch et al. 2015) reported that future projections of climate such as high intensity rainfall events, increasing temperatures, prolonged drought conditions will accelerate the land degradation in Australia. It is also reported that soil acidification, compaction also affected the soil degradation in Australia (Lal 2001).

In North America, the soil degradation was mainly due to the catastrophic wind erosion accompanied by the dust bowl during 1930s and it has become the soil-chemical degradation due to industrial processing during the latter half of the 20th century. Extracting minerals, coal, oil and gases through mining causes the soil degradation and the mining has been drastically increased from 4.4 million ha (Lal 2004) to 27 million ha (Nickerson et al, 2011) by the year 2007. Baumhardt et al. (2015) found that the human induced causes to the soil degradation in North America are mainly viz a viz industrial dislocation through mining and urban sprawl. The study pointed out that the continued expansion of infrastructure such as hi-way development, construction of housing etc. Decomposition of soil organic carbon is another threat to land degradation. The decomposition lead to loss of C from CO_2 and other nutrients which will be insufficient to plant growth. (Baumhardt et al. 2015)found that in North America, the forests were being converted to farm lands and as a result, the soil organic carbon content is only 505 at present. Ausubel et al. (2013) reported that the land capable of producing crops was declined by 65% during the period 1961 to 2009 whereas the global population has been doubled during the same period. Romero-sanchez et al. (2016) used the regional trend indicators with satellite data to assess the land degradation in Mexico, They found that the net primary productivity and canopy covers as the two indicators of vegetation status. (Barbosa and Kumar 2012) also used Normalised Difference Vegetation Index (NDVI) to study the temporal dynamics of some portions of Brazil and found a strong linkage with the climate variability.

Santibañez and Santibáñez 2007) found that 45% of the crop lands in South America and 74% in Meso America were degraded. For the agricultural practices, some temperate forests in Chile, Argentina and Brazil have reduced to small patches. This stress on the ecosystems are due to the lack of good agricultural practices and climatic fluctuations such as increase in climate variability, frequent droughts and floods. The major threat is on the biomes of Catnaga, Brazil and highlands of Puna and Amazon rain forest.

It is reported by FAO (2010) that the total deforestation in Brazil is about 55.3 million ha, in Venezula is about 5.8 million ha and in Mexico, it is about 5.5 million ha during the period 1990 to 2010. This deforestation is mainly to convert the natural forests and shrubs in to the pastures. This over usage of forest land might be due to the extensive livestock grazing, illegal crop planting etc. (Williarts et al, 2014). The intensification of soil degradation due to climate change is one of the major concerns (IPCC, 2014B).

Jiang et al. (2014) studied the impact of climate change on land degradation in Mount Elgon region, Uganda, and reported that the trends in land degradation are linked to the precipitation changes. They have studied the trends of land degradation from 2000 to 2012 and found that the annual mean soil loss showed the fluctuations during the study period.

4.5.2 Forests

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. The contribution of degradation to carbon emissions is thus uncertain, with estimates varying from 10%

(Houghton and Nassikas 2018) to nearly 70% of carbon losses (Baccini et al. 2017), although these two estimates are not strictly comparable. The "10%" estimate refers to emissions from land-use change, while the "70%" estimate refers to all changes in biomass (from both losses from land-use change and gains from environmental factors). Asner et al. (Asner et al. 2004) estimated emissions from selective logging in a portion of Amazonia. Baccini et al. (2017) found that degradation within forests accounted for 69% of the losses of forest biomass for the entire tropics. Pearson et al. (Pearson et al. 2017) defined degradation as "a direct, human-induced decrease in carbon stocks in forests resulting from a loss of canopy cover that is insufficient to be classed as deforestation" and estimated rates of gross emissions for 74 developing countries from changes in canopy density. They estimated annual gross emissions of 2.1 billion tons of carbon dioxide, of which 53% were derived from timber harvest, 30% from wood fuel harvest and 17% from forest fire (Pearson et al. 2017). Estimating gross emissions only, creates a distorted representation of human impacts on forest carbon cycles. While there is no doubt that in most developing countries the impacts of forest harvest for timber and fuel wood and land–use change (deforestation) contribute gross emissions, it is also necessary to quantify net emissions, that is, the balance of gross emissions and gross removals of carbon from the atmosphere through forest regrowth.

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. Forest management and the use of wood products in GHG mitigation strategies result in the changes in forest ecosystem C stocks, changes in harvested wood product (HWP) C stocks, and 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 (Nabuurs et al. 2007; Lemprière et al. 2013; Kurz et al. 2016). 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)). Definitions of forest degradation, which focus only on reductions in forest ecosystem C stocks can lead to conclusions about forest management impacts on the atmosphere that are incomplete because they do not quantify increases in C stocks in harvested wood products or reductions of emissions in other sectors, that result from the use of wood products and bioenergy (Nabuurs et al. 2007; Lemprière et al. 2013; Kurz et al. 2016).

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 aboveground biomass C stocks and require additional assumptions or model-bases analyses to also quantify the impacts on the other carbon stocks defined by the IPCC, including belowground biomass, litter, woody debris and soil carbon. Depending on the type of disturbance, changes in aboveground biomass may lead to decreases or increases in other carbon pools, for example windthrow may result in losses in aboveground biomass that are (initially) off-set by corresponding increases in dead organic matter carbon pools, while deforestation will reduce all ecosystem carbon pools.

Impacts of deforestation and forest degradation, including forest management have resulted in carbon stock reductions in global forests (Erb et al. 2017) relative to the hypothetical natural forest conditions. However, the extent to which human activities have reduced the productive capacity of forest lands is poorly understood. Moreover, 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). Thus opportunities exist for sustainable forest management to contribute to atmospheric carbon targets through avoidance of deforestation and degradation, forest conservation, and enhancements of carbon stocks.

4.6 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. The direct impacts are those in which climate and land interact directly in time and space, for example increasing rainfall intensity may exacerbate soil erosion. The indirect impacts are those where climate change impacts and land degradation are separated in time and/or space, for example if declining agricultural productivity due to climate change impacts drive an intensification of agriculture elsewhere, which may cause land degradation. Land degradation if sufficiently widespread may also feed back into the climate system by either reinforce or balance ongoing climate change.

4.6.1 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, is the changes of the erosive power of precipitation. The second level of uncertainties, and where the uncertainties are much larger, relates to the vegetation changes as a result of rainfall changes. The protective function of vegetation is crucial for erosion (Mullan et al. 2012; García-Ruiz et al. 2015b).

4.6.1.1 Changes in erosion risk due to precipitation changes

The hydrological cycle is intensifying with increasing warming of the atmosphere (Trenberth 2011). The intensification means that the number of heavy rainfall events is increasing while the total number of rainfall events tends to decrease.

Modelling of changes in land degradation as 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 (García-Ruiz et al. 2015b). 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 & Fang (Li and Fang 2016) summarised 205 representative modelling studies around the world where erosion models had been used in combination with down-scaled climate models to assess future (between 2030 to 2100) erosion rates. Almost all of the sites had current soil loss rates above 1t ha^{-1} (often assumed to be the upper limit for acceptable soil erosion) 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.

Mesoscale convective systems (MCS), typically thunder storms, have increased markedly in recent 3-4 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 etc.

Using a comparative approach Serpa and colleagues (Serpa et al. 2015) studied two Mediterranean catchments (one dry and one humid) using a spatially explicit hydrological model (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 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 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.

4.6.1.2 Climate induced vegetation changes, implications for land degradation

Forests influence the storage and flow of water in watersheds (Eisenbies et al. 2007) and are therefore important for regulating how climate change will impact landscapes. Hence the ways in which forests are impacted by climate change and other dynamics, such as forest management, are keys to understanding future impacts on land degradation. 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 (Eisenbies et al. 2007).

Climate change affects forests in both positive and negative ways. In high latitudes, a warmer climate will increase water use efficiency and extend the growing seasons, while increasing levels of atmospheric CO_2 will potentially increase vigor and growth (Allen et al. 2010). Negative impacts dominate, however, and have already been documented (Bonan et al. 2008) and are predicted to increase (Allen et al. 2010). Several authors have emphasised a concern that tree mortality will increase due to climate induced physiological stress as well as interactions between physiological stress and other stressors, such as insect outbreaks 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 risks to forests (Lindner et al. 2010).

Water balance of at least partly forested landscapes is to a large extent controlled by forest ecosystems. 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). Surface runoff is an important agent in soil erosion

Rangelands are projected to change in complex ways due to climate change. Increasing levels of atmospheric CO_2 stimulate directly plant growth and can potentially compensate 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).

4.6.1.3 Changes of hydrological regimes

In mountainous regions more precipitation tend to fall as rain instead of snow, and snow melts earlier in a warmer climate (Trenberth 2011). This is projected to alter hydrological regimes.

Changes in river runoff between different climate scenarios can be studied through hydrological models in combination with climate scenarios and models. Comparisons of different hydrological models and different climate models suggest that the variability in the runoff results is considerably larger between climate models than the variability among hydrological models (Gosling et al. 2011; Teng et al. 2012). Such studies in Australia predict a drier future for the Southeastern part of the continent and that the hydrological response of reduced rainfall is amplified (Teng et al. 2012).

4.6.1.4 Coastal erosion

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). 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). In different regions of the world, it has been found a relation between the rates of relative sea-level rise and coastal erosion or recession (Meeder and Parkinson 2018; Shearman et al. 2013; Savard et al. 2009; Yates et al. 2013).

Rates of coastal erosion or recession will increase due to rising sea levels and in some regions also in combination with increasing oceans waves (McInnes et al. 2011; Mori et al. 2010), lack or absence of sea-ice (Savard et al. 2009) and changing hurricane paths (Tamarin-Brodsky and Kaspi 2017). 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 that 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 also the lithostratigraphy of the coast under study.

4.6.2 Indirect impacts on land degradation

Indirect impacts of climate change on land degradation are difficult to quantify because of the many conflating factors. 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 bioenergy (Schleussner et al. 2016) will exert strong pressure on agricultural lands.

4.6.2.1 Changing agricultural practices

Placeholder. To be completed

4.7 Impacts of bioenergy provision on land degradation

4.7.1 Provision of bioenergy and negative emission technologies (NETs)

The most profound driver of change in land-use and land-cover in the coming decades is the expected provision of bioenergy. Most scenarios compatible with the goal of the Paris Agreement, to stay below 2°C, rely on large-scale deployment of negative emission technologies involving bioenergy (Schleussner et al. 2016; Smith et al. 2016a; Mander et al. 2017). However, many of the assumptions built into such scenarios and modelling results are insensitive to a wide range of real-world constraints related to land use, such as agricultural efficiency gains, land area requirement, crop yields, availability of crop residues, and infrastructure for transporting large quantities of biomass (Mander et al. 2017), the economic viability of the technologies (Faran and Olsson 2018), and the social premises for their deployment (Dooley and Kartha 2018).

Published scenarios for meeting the targets of the Paris Agreement include a wide range of requirements (accumulated during the 21st century) for NETs, between 450 and 1000 $GtCO_2$ for 1.5°C and between 0 and 900 $GtCO_2$ for 2°C (Rogelj et al. 2015; Dooley and Kartha 2018). Estimates of the biophysical possibilities of such amounts indicate a range of 370-480 $GtCO_2$. Even this range must be considered optimistic from a social and ethical point of view (Gough and Vaughan 2015).

[Discussion to be added: risks of increased land degradation through expansion of biomass production; - if possible, based on information of how much area, and where – and. opportunities for biomass production through rehabilitation of degraded land.]

4.7.2 Reducing deforestation and forest degradation

Placeholder. To be completed

4.8 Impacts of land degradation on climate systems

(Coordinate with Ch2)

Land degradation processes have multiple links with the atmosphere that can translate into an array of local to global effects on climate. These effects include those mediated by the fluxes of greenhouse gases from the land to the atmosphere (i.e. biogeochemical effects), which are global, but also those affecting radiation fluxes and water cycling (i.e. biophysical effects) through albedo shifts, aerosol loading, evapotranspiration changes, or surface roughness modifications, which have both global and local effects.

While Chapter 2 has its focus on land cover changes and their impacts on the climate system, this section focuses on the influence of land degradation process (see Table 4.3.1) which may take place in association or not to land cover changes. The effects of land degradation on CO_2 and other greenhouse gases as well as those on surface albedo and other physical controls of the global radiative balance are treated.

4.8.1 Impacts on net CO_2 emissions

Land degradation process with direct impact on soil and terrestrial vegetation have great relevance in terms of CO_2 exchange with the atmosphere given the magnitude and activity of these reservoirs in the global C cycle. As the most widespread form of soil degradation, erosion detaches the surface soil material which typically hosts the highest organic C stocks, favoring the mineralisation and release as CO_2. Laboratory experiments suggest that more than 20% of the organic C of transported material can be released (Lal 2003). However, a substantial fraction of the eroded material may preserve its organic C load in field conditions and, in occasions, even favor C sequestration through the burial of both the transported material and the surface soils at the deposition location and or the (Quinton et al. 2010). In this regard the "side-effects" of erosion at the affected sites are more likely to cause net CO2 emissions through their indirect influence on soil fertility and the balance of organic C inputs and outputs, converging with other non-erosive soil degradation processes such as nutrient depletion, compaction and salinisation, which can lead to the same net C effects (see table 4.3.1)(van de Koppel et al. 1997).

As natural and human-induced erosion can result in net C storage on 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 a typical deposition location and their degradation or replacement with other vegetation is resulting in a substantial C release (Pendleton et al. 2012), what 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 particularly relevant in terms of global CO_2 emissions. (To be completed after coordination with Chapter 2. This is also mentioned in section 4.10.4 on Perennial grains). Besides the initial pulse of CO_2 emissions associated with the onset of cultivation, soil protection and restoration practices such as conservation tillage and nutrient replenishment can have net C benefits as long as they reverse soil degradation and favor its organic matter build-up. The direct C-sequestration benefits of no-till practices (i.e. tillage elimination favoring crop residue retention in the soil surface) appear uncertain after recent assessments (VandenBygaart 2016). Among soil fertility restoration practices, lime application for acidity correction can generate a significant net CO_2 source (Bernoux et al. 2003). Land degradation processes in seminatural ecosystems driven by unsustainable uses of their vegetation such as overgrazing often lead to partial denudation and reduced biomass stocks, causing net C releases from soils and plant stocks, yet the reverse arises in the case of degradation processes leading to the woody encroachment or thicketization of grass-dominated systems **Error! Reference source not found.** (Maestre et al. 2009). Fire regime shifts in wild and seminatural ecosystems are a degradation process in itself, with high impact on net C emission and with underlaying 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 this interactive processes affecting unmanaged forests have resulted in massive C 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).

In this section we will attempt to calculate the reduction in C storage as a result of degradation, and also the reduction in NPP or NEP as a result of degradation if the data exist (or if the global models would agree – but we could at least make the point that these numbers are measured in Pg C (i.e. they are meaningful in the context of the global climate system).

4.8.2 Impacts on non-CO_2 greenhouse gases

In agricultural land overfertilisation and poor nitrogen retention is a major source of N_2O to the atmosphere (Oertel et al. 2016), not only in-situ but also along the pathway of dissolved inorganic N transport by draining waters all the way to the dead-zones in the ocean. Current budgets of anthropogenically fixed nitrogen on the Earth System suggest that N_2O release from terrestrial soils and wetlands accounts for 10-15% of the inputs yet many gaps on further release along hydrological the hydrological pathway remain, with ocean emissions being a major aspect of concern (Schlesinger 2009).

Hydrological degradations processes, 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 wetland and meadow creates intense pulses of mineralisation linked with high N_2O release (Morse and Bernhardt 2013; Norton et al. 2011). In the case of flooding of non-wetland soils, a suppression of CO_2 release is typically out-compensated in terms of net greenhouse impact by enhanced CH_4 fluxes, that stem from anaerobiosis 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 creation can produce intense pulses of CH_4 release (Altor and Mitsch 2006; Fenner et al. 2011).

4.8.3 Albedo-related 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. In the case of forest degradation or elimination, 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 raises albedo (net cooling effect) as the reflective snow cover becomes exposed, what can overwhelm the net radiative effect of the associated C release to the atmosphere (Davin et al. 2010). This high compensation appear also significant in temperate and subtropical dry forests in which radiation levels are higher and C stocks smaller compared to their more humid counterparts (Houspanossian et al. 2013). Reciprocally, woody encroachment over highly reflective grasslands can introduce net warming effects though albedo cuts and fires on the other hand appear to have a long-lasting effect on albedo raises (Dintwe et al. 2017). Degradation processes affecting wild and semi-natural ecosystems such as fire regime changes, woody encroachment, selective logging and overgrazing can trigger strong albedo changes before significant biogeochemical shifts take place, in some cases these two types of effects have opposite signs in terms of net radiative forcing, making their join assessment critical for understanding climate feedbacks (Bright et al. 2015). Besides global the net global effects discussed above, albedo shifts can play a significant role on local climate, as exemplified by the effect of no-till agriculture reducing local heat extremes in European landscapes (Davin et al. 2014).

4.8.4 Other biophysical impacts

Other Biophysical effects of degradation (heat exchange, water exchange, cloudiness shifts, local convective activity)

4.8.5 Integrating multiple global impacts

Need for full accounting, difficulty of global/local trade-offs

4.9 Impacts of climate-related land degradation on people and nature

Unravelling the impacts of climate-related land degradation on people and nature is highly complex. This complexity is due to the interplay of multiple social, political, cultural, and economic factors which shape the ways in which social-ecological systems respond (Morton 2007). Consequently, attribution of the impacts of *climate-related land degradation* on people and nature is rarely undertaken within the literature, with climate often not distinguished from any other driver of land degradation. Climate is nevertheless frequently noted as a risk multiplier for land degradation 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 resulting from 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' exposure and sensitivity to land degradation, with land degradation then leaving livelihoods more sensitive to the impacts of climate change and extreme climatic events. If human and ecological communities are exposed and sensitive and cannot adapt, they can be considered vulnerable, while if they are exposed, sensitive and can adapt, they may be considered resilient.

4.9.1 Poverty, livelihood and vulnerability impacts of climate related land degradation

Poverty is multidimensional and includes a lack of access to the whole range of capital assets that can be used to pursue a livelihood. Poverty also contributes to vulnerability, while vulnerability to climate change can deliver outcomes that exacerbate poverty (Eriksen, Siri and O'Brien 2007). However, poverty and vulnerability do not always mutually reinforce in the same way. Those who are vulnerable are not necessarily in poverty; and those living in poverty can exhibit heterogeneous patterns of vulnerability, with failure to secure wellbeing and livelihood outcomes taking a variety of different forms.

Livelihoods constitute the capabilities, assets, and activities that are necessary to make a living (Chambers and Conway 1992; Olsson et al. 2014). The sustainable livelihoods framework (Chambers and Conway 1992) is commonly employed to understand how specific actions may support livelihood improvement in an overall attempt to reduce poverty, and to understand how elements of the vulnerability context (the wide range of trends, trajectories, shocks and stresses operating over multiple time frames) can exacerbate poverty. Vulnerability can be assessed in many different ways, at different scales, using different indicators, taking into account specific vulnerability (in relation to identified drivers) or overall vulnerability (Vincent and Cull 2014). Generally, vulnerability is considered to be a function of exposure, sensitivity and adaptive capacity. Spatially and temporally, the impacts of land degradation will vary under a changing climate, leading some communities and ecosystems to be more vulnerable or more resilient than others under different scenarios. Even within communities, some groups such as women and the youth are often more vulnerable than others.

Climate and land degradation can both affect poverty and vulnerability through their threat multiplier effect. Often the geographical locations experiencing degradation are the same locations that are directly affected by poverty and vulnerability and are also affected by extreme events linked to climate change and variability. Altieri et al (Altieri and Nicholls 2017) however 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).

Although livelihood assessments often focus on a single snapshot in time, livelihoods are dynamic and people alter their livelihood activities and strategies depending the on internal and external stressors to which they are responding (O’Brien et al. 2004). When certain livelihood activities and strategies become no longer tenable as a result of land degradation, it can have further effects on issues such as migration (Lee 2009), as people seek to reduce their vulnerability and adapt by moving (see 4.8.5); and conflict (see 4.8.6), 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 as a result of for example, land use change.

Some authors (Okpara et al. 2016a,b) note that a vulnerability lens can be a useful way to understand the impacts of environmental and social changes taking place, allowing a focus on livelihood systems and the interlinkages between poverty, vulnerability, livelihoods and environmental changes. A large body of 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, the poor 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.

Methodologically, timeline approaches can be useful in unpacking the links between climate, land degradation and other livelihood impacts over specific temporal scales. For example, research in Bellona, in the Solomon Islands in the south Pacific (Reenberg et al. 2008) has 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 (Figure 4.4).

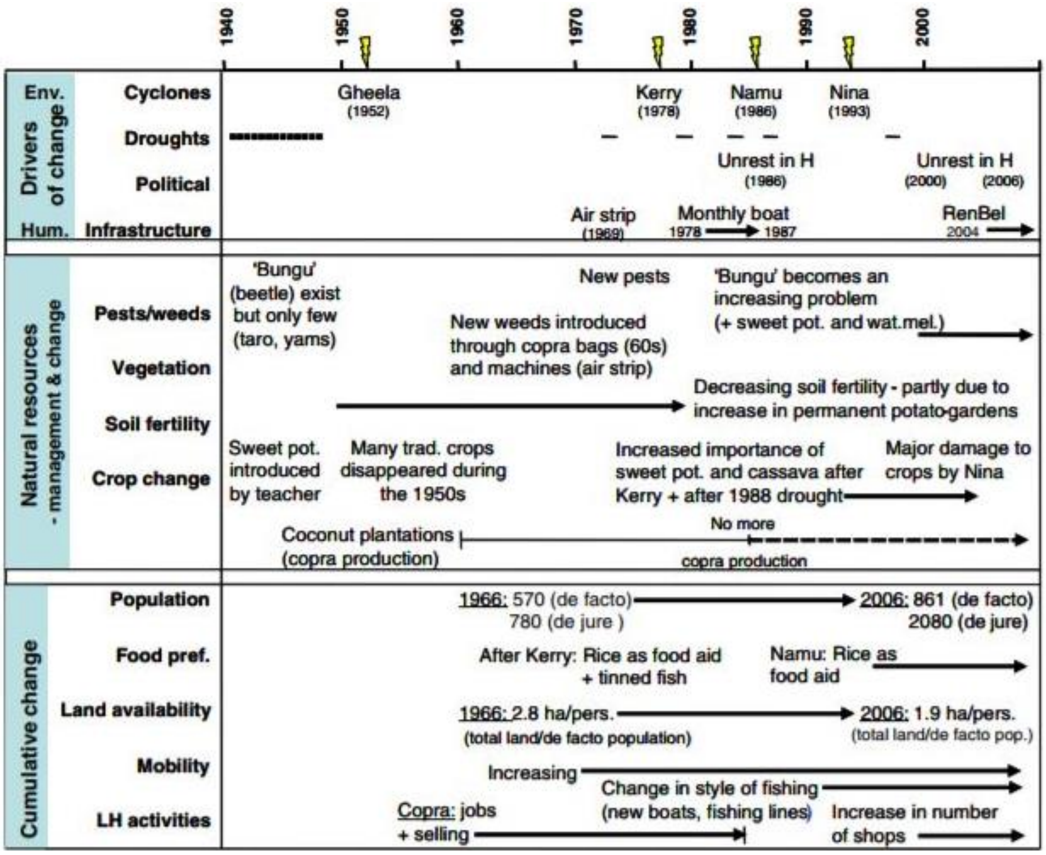

Figure 4.4 Coupled human-environment timelines of Bellona, 1940s to 2006. Source: Group interviews with selected households from the general HH-questionnaire (Reenberg et al. 2008)**.**

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 to engage stakeholders in addressing public natural resource management issues. In this regard, it is increasingly recognised that those suffering from and 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.

4.9.2 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). 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. 2017). Just a few studies have shown how the interactive effects of the aforementioned challenging, interrelated phenomena can impact crop productivity and hence food security and quality (Karami et al. 2009; Allen et al. 2001; Högy and Fangmeier 2008). Along with socio-economic drivers climate change accelerates land degradation due to its influence on land use systems (Millennium Assessment 2005; UNCCD 2017), potentially leading to a decline in agri-food systems, 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. 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 also problematic for food security, and can lead to increased food waste, while poor siting of roads and transport links can lead to soil erosion and forest loss, further feeding back into climate change.

Over the past decades, crop models have been useful tools for assessing and understanding of climate change impacts on crop productivity and food security (Rosenzweig et al. 2014; White et al. 2011). Yet, the interactive effects of soil parameters and climate change on crop yields and food security remain limited, with few studies assessing 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). To be sustainable, any initiative aiming 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, competition in appropriation of supplies and changes to per capita food consumption (Stringer et al. 2011). 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.

4.9.3 Impacts of climate-related land degradation on migration

Land degradation may trigger competition for scarce natural resources potentially leading to migration and/or conflict. However, 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, search for education and/or income, land shortage, political turmoil, and family-related reasons (McLeman 2017). Consequently, impacts of land degradation on migration are largely context-specific and vary across the globe.

One of the global ecosystems that is particularly prone to climate-related land degradation is drylands, which is mainly because of its fragile ecological conditions (UNCCD 2017). A significant share of existing studies on land degradation and migration – particularly in drylands – focuses on the effect of rainfall variability and drought and shows how migration serves as adaptation strategy (Piguet et al. 2018; McLeman 2017). For example, in the dry Ethiopian highlands severe soil erosion and forest degradation is a major environmental stressor which is amplified by re-occurring droughts, with migration being an important 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 and B. 1983) but was also observed more recently, for example in Guatemala (López-Carr 2012). In contrast, in Equator migration increased with land quality, likely because increased land 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 explaining land degradation-migration linkages requires considering a broad socio-ecological embedding.

In many corners of the world it is commonplace for household members to migrate seasonally, temporarily or permanently to diversify income sources and raising funds that can be invested in their home land to secure or improve livelihood outcomes. Remittances therefore may provide a vital lifeline for many households that are suffering from land degradation, with the financial support provided helping to increase wealth (Lambin and Meyfroidt 2011). Studies have shown that remittances help to slow down degradation and contribute to recovering degraded land (Hecht and Saatchi 2007), yet, may also lead to an expansion of land use (Davis and Lopez-Carr 2014), potentially contributing to (additional) land degradation. Again, these mixed results demonstrate the complexity of land degradation-migration linkages and suggest that recovery of degraded land in the context of migration may require governmental interventions.

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 (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 environment as a safety net for people and wildlife alike.

4.9.4 Impacts of climate-related land degradation on conflict

The relationships between land degradation, drought and conflict are highly controversially debated among scientists with the recent conflict in Syria further fuelling these debates (Selby et al. 2017a,b; Seager et al. 2017; Hendrix 2018). Much of the existing studies on natural resource degradation and conflict focuses on the effect of rainfall variability, drought and water scarcity and analyses their contribution to conflicts.

Early studies conducted in Africa hint at a significant causal link between land degradation and violent conflict (Homer-Dixon et al. 1993). For example, Val Percival & Homer-Dixon (Percival and Homer-Dixon 1995) identified land degradation as one of the drivers of the crisis in Rwanda in the early 1990ies which allowed radical forces to stoke ethnic rivalries. With respect to the Darfur conflict some scholars and 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 mixed results, yet, in many instances suggesting that climate change can increase the likelihood of civil violence if certain context factors are present (Scheffran et al. 2012; Benjaminsen et al. 2012). In contrast, Raleigh (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, although not always, 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 ensure, it is often as a result of a lack of appreciation for the cultural practices of others.

4.9.5 Impacts of climate-related land degradation on cultural practices

The nature of land-based practices, such as farming and forestry, are often closely linked to cultural practices. Forests in particular often have cultural and spiritual values (Bhagwat et al. 2017; Bhagwat and Rutte 2006). Although there are many drivers of deforestation and forest degradation, climate change stresses on agricultural land is one contributor (Dissanayake et al. 2017). The rate of deforestation and conversion of land for farming is significantly lower in sacred forests, for example, in countries as diverse as Ethiopia and China (Daye and Healey 2015; Brandt et al. 2015).

Studies on resolving the long-term and structural problems created by climate- related land degradation have revealed that the loss of cultural identity and heritage is a significant impact in a variety of contexts. Bush encroachment in Kalahari rangelands in southwest Botswana has adversely affected the forage availability and production of cattle, which play a central role in Batswana culture (Reed and Stringer 2016; Reed 2005). In Swaziland, droughts followed by heavy rainfall events cause soil erosion and loss of soil fertility and this land degradation is considered a threat to Swazi culture, in which soil has cultural and spiritual value (Stringer and Reed 2007). The indicators for cultural impacts of climate-related land degradation vary regionally due to differences in climate, culture, soil and ecology in different parts of the world. In Australia the increasing frequency of drought and its effect on productivity of land has increased the levels of farmer suicide, particularly among men (Alston 2011) but ultimately affecting whole communities. Some positive trends have nevertheless emerged from the situation: recognition of the importance of managing land degradation and the need to adapt to climate change has galvanised community action by local land care groups, undertaking for example,revegetation projects, counteracting to some extent the negative social impacts of climate related land degradation.

The nature of people's relationship with land is also determined by gender norms. Where land-based livelihoods are concerned, typically men have privileged control over the benefits from production (for example in terms of commercialisation), whilst women are responsible for household reproduction and ensuring food security, although the socially constructed norms and behaviors expected by men and women differ from place to place (Carr 2008; Doss 2002; Gladwin et al. 2001; Bryceson 1995; Dixon 1982). This is both reflected in, and reinforced by, gender differences in land tenure that are pervasive in the development world (e.g. (Agarwal 2003). However, it is not just gender but also other social identifiers that play a role – such that in Malawi age was found to be as important as gender in determining access to land (Carr and Thompson 2014). An intersectional approach to analysis is thus now preferred to a binary dichotomy between men and women, recognising that it is the intersection of many social identifiers that creates the norms that govern practices and, thus, in turn the effects of land degradation (Djoudi et al. 2016; Thompson-Hall et al. 2016).

Gender norms around access to, and use of, land also determine the nature of effects of climate-driven degradation and the nature of adaptation to it (Djoudi and Brockhaus 2011). Implicit assumptions also tend to prioritise technological solutions to adapt to climate change, which are better aligned to men than women (Alston 2013). Solutions to climate-related land degradation also often mean moving away from unproductive land and seeking alternative income sources, which also typically preference men, leaving women burdened by additional household tasks if they are left behind by migration (Alston and Whittenbury 2011). There is increasing recognition that land management plans and actions should be designed in a participatory process, involving local people and incorporating local knowledge and culture, to enhance effectiveness and ensure gender-responsiveness of efforts to address land degradation under climate change (Armesto et al. 2007; Altieri and Nicholls 2017).

4.9.6 Impacts of climate-related land degradation on nature

One example of climate interacting with human activities and other stressors to deliver land degradation impacts which then feed back into nature (in this case causing climate impacts) comes from the Aral Sea in Central Asia. The Aral Sea is found between southern Uzbekistan and northern Kazakhstan and was previously the fourth largest saline lake on Earth (Izhitskiy et al. 2016). Large-scale irrigation investments under the governance of the USSR led to substantial drying of the Sea, leaving land exposed to water and wind erosion, following initial increases in land productivity due to irrigation. As the lake levels diminished, dust entrainment increased and was deposited in other areas. Much of this dust was contaminated with pesticides used during the successful cropping years. Human health impacts from this have been stark, with the area experiencing high rates of tuberculosis, anaemia, kidney and liver disease, respiratory infections, allergies and cancer. Whish-Wilson (2002) estimated that 3.5 million people have been affected by the drying of the Aral Sea and the wider land degradation impacts it caused. The climate has also been affected. Land next to large water bodies is usually warmer in winter and cooler in summer than land further away (Reed and Stringer 2016). As the Aral Sea shrunk, the climate became more extreme, with resulting drier, shorter summers and longer, colder winters. In turn this has reduced growing seasons and rangeland productivity and had negative impacts on livelihoods. This example demonstrates the close relationship not just between people and environment but between people, land degradation and nature (in this case climate).

Other feedbacks between climate related land degradation on nature relate to amplified impacts of extreme events on the environment. For example, mangroves, which are found in 123 countries around the world, are being degraded at a phenomenal rate, with 20-35% of global mangrove area lost since 1980 (Polidoro et al. 2010). This forest degradation leaves coastlines largely unprotected as mangroves act as 'first line' defences. As the frequency of hurricanes, storms and typhoons increases with climate change, greater areas of land are threatened by storm surges. In turn, this can exacerbate degradation issues such as saline intrusion and erosion.

There are similar multidirectional relationships between climate, land degradation and fire, with changes to the patterns of land degradation affecting the susceptibility of an area to fire, whilst occurrence of fire can shape the quality of the land. These interlinkages may be further amplified under future land use change and climate scenarios (Bajocco et al. 2010), as soil, vegetation, climate, fire relationships are disrupted. This impacts upon aspects such as biodiversity, biomass, soil erosion, landscape productivity and therefore the land quality status of the affected area (Shakesby et al. 2007).

The literature contains lots of information about the role of biodiversity in helping to mitigate and reduce the impacts of climate change and land degradation, including climate related land degradation. In general, both climate change and land degradation have similar effects on biodiversity, leading to simplified ecosystems and an increased ratio of generalist species to specialists (Clavel et al. 2011) as well as losses in genetic diversity. In turn, simplification of ecosystems can reduce environmental safety nets available to people during times of environmental stress. Research suggests that the more biodiverse agroecosystems are, the lower the impacts of extreme events on those systems (Altieri and Nicholls 2017) and therefore the less vulnerable systems are to climate driven degradation. This is partly attributed to additions of organic matter which improve below ground biodiversity, creating good conditions for plant roots, improving soil water retention capacity and enhancing drought tolerance, while also improving infiltration/reducing erosive runoff (Liniger et al. 2007), while also supporting redundancy and wider tolerance amongst the broader genotypes present in more biodiverse farms. This reduces exposure of the land to degradation processes that are caused primarily due to climatic factors. More diverse cropping systems can also improve resilience and act as a buffer to climate variability and extreme events. Crop diversification further supports suppression of pest outbreaks and reduces pathogen transmission, both of which may worsen under climate scenarios of the future (Lin 2011).

Sustaining biodiversity within agricultural systems can boost the delivery of ecosystem services such as pollination, essential for the successful production of many food crops. The literature highlights the combined challenges of intensifying land use (and subsequent degradation, especially of forest areas), climate change and the spread of alien diseases and species as key threats to pollinators (Vanbergen and Initiative 2013). Land degradation in the form of habitat loss, combined with climate change, is also highlighted as problematic for natural enemies of agricultural pests (Thomson et al. 2010).

4.10 Addressing/targeting land degradation in the context of climate change

4.10.1 Actions on the ground to address land degradation

Placeholder. Section in progress.

4.10.2 Higher-level responses to land degradation

Placeholder. Section in progress.

4.10.3 Contributions of land restoration and rehabilitation to mitigation of climate change

Placeholder. Section in progress.

4.10.4 Resilience and tipping points

Resilience refers to the capacity of a system, such as a farming system, to absorb disturbance (e.g. drought, conflict, market collapse), and recover, to retain the same identity. It 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. Resilience as an analytical lens is particularly strong in ecology and related research on natural resource management while in the social sciences the use of resilience for studying social and ecological interactions is contested (Cote and Nightingale 2012; Olsson et al. 2015; Cretney 2014; Béné et al. 2012). Further description of the debate will be added.

The shocks and trends anticipated to occur due to climate change are likely to exacerbate risk of land degradation; assessing and enhancing resilience to climate change is a critical component of designing sustainable land management strategies.

Figure 4.5 illustrates how resilience and the related terms adaptation and transformation are used differently by different communities. The relationship and hierarchy of resilience with respect to vulnerability and adaptive capacity are also debated, with different perspectives between the disaster management, and global change communities, (e.g. (Cutter et al. 2008). These differences in usage need not inhibit the application of "resilience thinking" in managing land degradation; researchers using these terms, despite variation in definitions, are applying 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 using understanding of the key variables, relationships and vulnerabilities of the system; knowledge of the thresholds or tipping points beyond which the system may transition to an undesirable state; and devising management strategies to steer away from thresholds of potential concern, thus facilitating long-lasting environmental and production benefits.

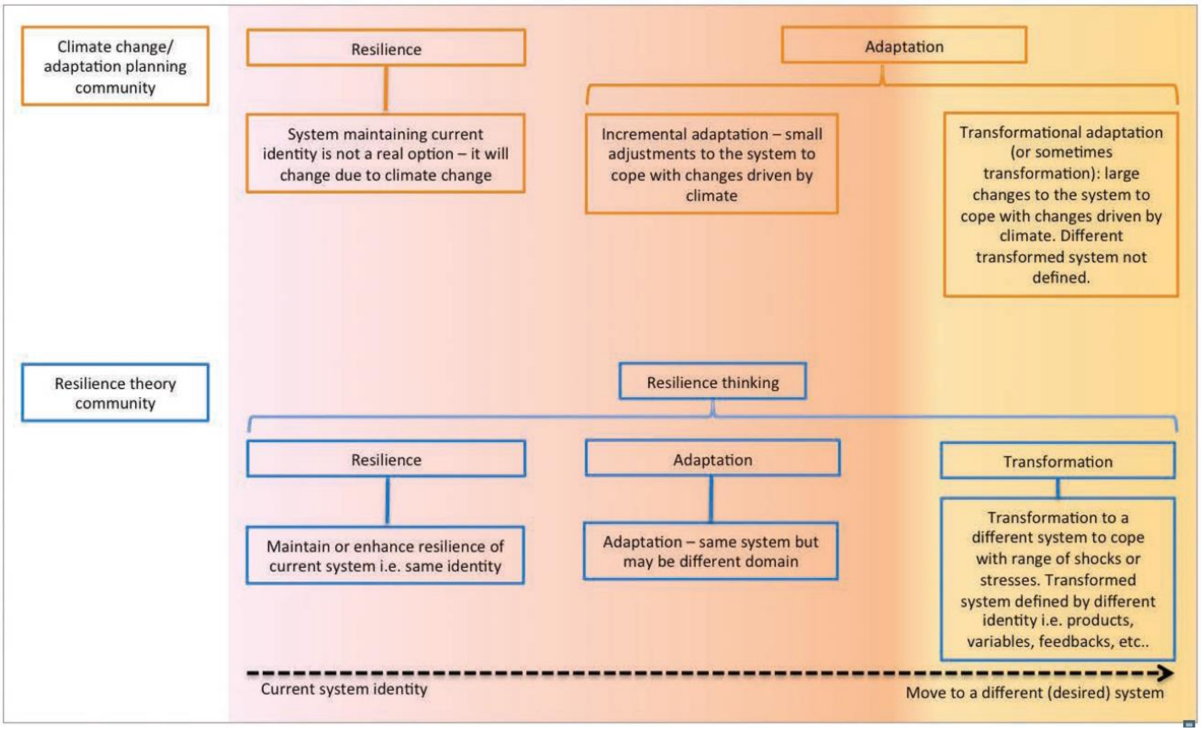

Figure 4.5 Usage of the terms resilience, adaptation and transformation by the climate change adaptation and ecological resilience communities Source: (O'Connell et al. 2016)**.**

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 (e.g. ground cover threshold below which pasture shifts to low productivity state). Studies have identified various biophysical and socio-economic thresholds in different land use systems – see Box 4.1

In managing land degradation, it is important to assess the resilience of the existing system, and the proposed management interventions. It may be beneficial to enhance resilience of the existing system, or if the existing system is in an undesirable state or considered unviable under expected climate trends, it may be more useful to promote adaptation or even transformation to a different system that is more resilient to future changes. For example, in irrigation areas where water shortages are increasingly likely, measures could be implemented to improve water use efficiency, for example by establishing drip irrigation systems for water delivery, but transformation to pastoralism or mixed dryland cropping/livestock production may be more sustainable in the longer term, at least for part of the area.

The essential features of a resilience approach to management of land degradation under climate change are (adapted from (O'Connell et al. 2016; Simonsen et al. 2014):

- Primacy of "systems thinking" Systems thinking is based on understanding of relationships between the elements of the targeted social-ecological system, the valued products and services delivered by the system, key controlling variables and feedback loops. It identifies the fundamental "slow variables" such as soil organic matter content, that influence the state of the system and respond gradually to change. It identifies non-linear responses - thresholds – and cross-scale interactions. Systems thinking aids the identification of the key drivers of land degradation. It allows trade-offs and synergies to be recognised and managed.
- Focus on avoiding crossing thresholds that lead to an undesirable state, by assessing current condition and trajectory, and proximity to thresholds; identifying the key determinants or controlling variables of resilience in the system, and how management can influence them; managing to enhance 'specified resilience" (to anticipated climate change including slowly changing variables and extreme climate events) and general resilience (promote improved

coping capacity through education, public health, gender equity, equality in power, secure land tenure; diversity in land use at landscape scale; agrobiodiversity at farm scale).

- Articulation of the theory of change, that specifies the intended impact pathways for interventions planned to manage land degradation.
- Management of land according to its identified land potential, which reflects inherent land and soil features aligned with key ecosystem functions, and determines the inherent, long-term capacity of the land to sustainably generate ecosystem services (UNEP 2016).
- Monitoring suitable indicators of resilience. While there are some broadly applicable indicators of general resilience, there is no single globally-relevant indicator of resilience of land management systems (O'Connell et al. 2015). For example, connectivity of transhumance routes is likely to be a key indicator for communities relying on seasonal movement of grazing livestock, while maintenance of soil carbon could be a key indicator for cropping systems. Indicators of resilience should be identified for each context, based on knowledge of the key variables, risks and thresholds.
- Effective stakeholder engagement, applying participatory approaches to land use decision-making; design of land management strategies; collection and interpretation of monitoring data.
- Application of adaptive management based on results of monitoring and new research.
- Effective knowledge management with emphasis on learning, integrating indigenous, local, and traditional knowledge, applying "triple loop learning", where the first learning loop can lead to incremental changes in routine actions, the second leads to revisiting underlying assumptions, and the third may influence underlying values and core beliefs (Stafford Smith et al. 2009).

Box 4.1: Examples of documented thresholds

(*medium confidence*) 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 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.

limited evidence: above 20% deforestation, reduction of rainfall and forest resilience (measured as the probability of high tree-cover in function of the rainfall regime) in the southwestern Amazon forest (Zemp et al. 2017).

There is high agreement that in the humid tropics, soil loss is considerably higher in bare soils such as cropland in fallow (erosion rate is one-150th in forest, also substantially reduced in grassland (Labrière et al. 2015). Of most relevance to land management, soil erosion was reduced by 99% where conservation practices were employed (e.g. no-till farming, mulch and hedgerows) compared to recently planted crops with no conservation measures. Thus, when establishing crops or plantations, ground cover should be restored and/or maintained using mulches or cover crops (Labrière et al. 2015).

(*limited evidence – one paper*) Soil erosion in coffee grown under shade trees: threshold for erosion is 60-65% litter cover (Blanco Sepúlveda and Aguilar Carrillo 2015).

Closed forest to savannah or grassland: transition resulting from the combined pressure of water limitations and fire disturbance frequency: if fire is too frequent trees do not reach reproductive maturity and post-fire regeneration will fail; similarly reduced rainfall / increased droughts prevents successful forest regeneration (Reyer et al. 2015; Thompson et al. 2009).

Other examples: die-back caused by drought; bark beetle attack caused by warmer winters

4.10.5 Barriers to implementation and "limits to adaptation"

Placeholder. Section in progress.

4.11 Hotspots and case-studies

4.11.1 The role of urban green infrastructure in land degradation, climate change adaptation and mitigation

Over half the world's population lives in towns and cities, a proportion that is predicted to increase to about 70% by 2050 (United Nations 2015). This 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 as economic development and the need for space for construction is prioritised. This can result in the degradation of existing agricultural areas and 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 (The Royal Society 2014) this issue is only likely 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) and as an important tool to mitigate and adapt to climate change. UGI can be defined as all green elements such as parks, public greenspaces, green corridors, street trees, urban forests, green roofs/walls and private domestic gardens (Tzoulas et al. 2007); a definition often extended to include 'blue' infrastructure, such as rivers, lakes, swales and other water drainage features.

Through retaining vegetation, revegetating previous developed land or using 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, even though, compared to overall carbon emissions from cities, this effect is comparatively small. Given that UGI necessarily involves the retention and management of non-sealed surfaces, co-benefits for reducing land degradation (e.g. improved water retention, carbon storage and vegetation productivity) are also apparent.

The importance of UGI as part of a climate change adaptation approach has received greater attention and application (Gill et al. 2007; Demuzere et al. 2014; Sussams et al. 2015; Fryd et al. 2011). 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. 5. Increasing vegetation cover, planting street trees and maintaining/expanding public parks reduces temperatures locally (Cavan et al. 2014; Di Leo et al. 2016; Feyisa et al. 2014; Zölch et al. 2016). Similarly, natural flood management and ecosystem based approaches such as providing space for water, renaturalising rivers and reducing surface run-off through the presence of impermeable surfaces and vegetated features (including walls and roofs) can manage flood risks, impacts and vulnerability (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 can, therefore, be an important adaptation mechanism, both to climate change (Potschin et al. 2016) and land degradation.

Most examples of UGI implementation have centred on its role in water management for flood risk reduction as a climate change adaptation strategy. The importance for land degradation is commonly either not stated, or not prioritised within the literature. In Beira, Mozambique, the government is using UGI to mitigate against increased flood risks predicted to occur under climate change and urbanisation, by improving the natural water capacity of the Chiveve River. As part of this process, 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 land degradation in those areas. Elsewhere, planning mechanisms promote the retention and expansion of green areas within cities to protect ecosystem service delivery, which directly reduces land degradation, but UGI efforts more generally are largely viewed and justified in the context of climate change adaptation and mitigation. For instance, the Landscape Programme in Berlin includes five plans, one of which covers adapting to climate change through the recognition of the role of UGI (Hansen et al. 2016). Major climate related challenges facing Durban, South Africa, now considered a global leader in climate adaptation planning (Roberts 2010), include sea level rise, urban heat island, water runoff and conservation (Roberts and O'Donoghue 2013). Durban's Climate Change Adaptation plan includes the retention and maintenance of natural ecosystems, in particular those which are important for mitigating flooding, coastal erosion, water pollution, wetland siltation and climate change (eThekwini Municipality 2014).

4.11.2 Approaches to agricultural intensification (Sustainable/ecological intensification, agro-ecology) and the implications for LD, food security, livelihoods, socio-cultural aspects, etc. (mitigation and adaptation)

The requirement to produce more food and fibre for a growing human population requires intensification of current land use, while also ensuring the sustainability of agricultural systems and meeting mitigation goals (Tilman et al. 2011). One of the challenges of intensified land use is that this can lead to ecological and environmental damage as well as degradation of soil resulting in a loss of function which underpins many ecosystem services (Smith et al. 2016b). There is risk that the short-term becomes the enemy of the long-term. Agroecosystems which maintain or improve the environmental, ecological and human capital services they provide may be defined as sustainable systems, those which deplete these assets, leaving fewer for the future are unsustainable (Pretty and Bharucha 2014). Producing more food and fibre without the conversion of additional non-agricultural land and while also reducing environmental impacts requires what has been termed sustainable intensification (Godfray et al. 2010); Figure 4.6). Sustainable intensification may be achieved through a wide variety of means; from improved nutrient and water use efficiency via plant and animal breeding programs, to the implementation of integrated soil and pest management practices. To achieve these goals a combination of land sparing and land sharing options could be utilised. Under a land sparing strategy intensification of land use in some areas, generating higher productivity per unit area of land, can allow other land to be set aside to meet wider SDG's such as increased carbon sequestration and the conservation of natural ecosystems and biodiversity. Conversely under a land sharing strategy less land is set aside, but lower levels of intensification are applied to agricultural land, providing a combination of food and fibre production and other functions such as biodiversity conservation from the same land (Green et al. 2005). The two approaches are not mutually exclusive and the suitability of their application can be system and/or location specific (Fischer et al. 2014).

4.11.2.1 Closing yield gaps through crop and livestock breeding / efficient irrigation

Section to address the drive for more nutrient efficient crops and livestock through breeding programs.

While it is important to breed new more nutrient and water efficient cultivars that close existing yields gaps, it also vital that food security is ensured under future climate conditions, requiring an interdisciplinary approach across plant science and animal sciences (McKersie 2015).

4.11.2.2 Precision agriculture

Precision farming usually refers to optimising production in fields through site-specific choices of crop varieties, agrochemical application, and water management (Hedley 2015). This type of farming takes advantage of the natural variability of soil and terrain in a field rather than ignoring it. In modern agriculture it is a technologically advanced approach to combine continuously monitoring instruments with plant management devices, often replacing manual labour with technology. Precise monitoring of crop performance over the course of the growing season will enable farmers to economise on their inputs in terms of water, nutrients and pest management. Precision agriculture has the potential to achieve higher yields in a more efficient and sustainable manner compared with traditional methods. Therefore, it can contribute to both the food security and land sparing goals associated with sustainable intensification. However, barriers to the uptake of these precision agriculture technologies remain. In what is described at the 'implementation problem', despite the potential to collect vast quantities of data on crop or livestock performance, converting this data to management practice is a challenge (Lindblom et al. 2017). The principle of precision agriculture can be applied equally to low capital-input farming such as small-scale farming in the global South. The principle is the same but instead of employing monitoring instruments they use the skilled eye of an experienced farmer, and instead of automatic plant management devices they use manual labour to supply water, nutrients, and pest management in a timely and site-specific manner (Mondal and Basu 2009).

4.11.2.3 Agroforestry as a Mode of Sustainable Intensification

As well as improving regional food security in a sustainable manner, agroforestry can provide additional ecosystem services when compared with monoculture crop systems. This is especially relevant where there is a requirement to find a balance between the demand for increased agricultural production and the protection of natural ecosystems such as primary and secondary forests (Mbow et al. 2014). Co-benefits include increased carbon sequestration in soils and biomass, improved water and nutrient use efficiency, the creation of a favourable micro-climate as well as contributing to many broader SDG's (Waldron et al. 2017). In crops such as coffee and cocoa, agroforestry systems are increasingly being promoted as offering a route to sustainable farming with important adaptation and mitigation co-benefits (Vaast et al. 2016). However, positive interactions within these systems can be ecosystem and/or species specific, and co-benefits such increased resilience to extreme climate events, or improved soil fertility are not always observed (Abdulai et al. 2018; Blaser et al. 2017). Another mode by which agroforestry can enable sustainable intensification is by allowing continuous production on the same piece of land with higher productivity without the need to use shifting agriculture systems to maintain crop yields (Nath et al. 2016). This provides an example of sustainable land use coupled with increased agricultural yields and co-benefits for other ecosystem services. Nevertheless, the adoption of agroforestry has been low (Coe et al. 2014), and many barriers exist, especially for small holder farmers who often have limited access to finance for inputs and mechanisation (Jerneck and Olsson 2013, 2014).

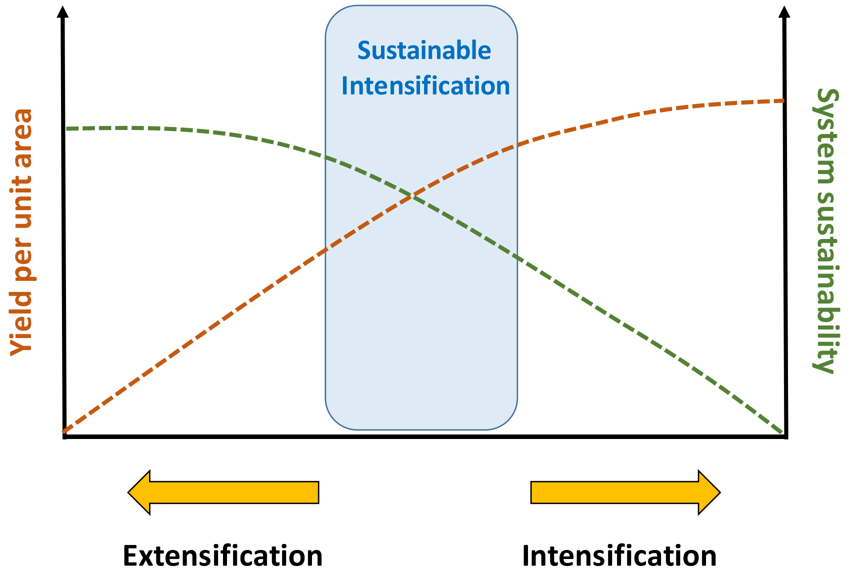

Figure 4.6 Sustainable intensification – there is a need to balance the increasing demands for food and fibre with long term sustainability of land use. Sustainable intensification can offer a window of opportunity between intensification of land use without causing degradation. This allows the sparing of land to provide other ecosystem services, including carbon sequestration and the protection of biodiversity.

4.11.2.4 Intensively Managed Grasslands

In temperate regions, highly productive agricultural grasslands used to produce meat and dairy products are characterised by monoculture cropping of agronomic grasses with high agrochemical inputs. This intensive management has been related to environmental and ecological degradation and any further intensification of these systems is therefore limited. Multi-species grasslands may provide a route to sustainable intensification, as even a modest increase in species richness in intensively managed grasslands can result in higher forage yields without increased inputs (Finn et al. 2013; Sanderson et al. 2013; Tilman et al. 2011). Increases in yield are primarily driven through replacement of mineral nitrogen requirements, which are fossil fuel dependant, with symbiotically fixed nitrogen from legume species. Other co-benefits from multi-species grasslands include the production of forage with a higher protein content, contributing to on-farm protein self-sufficiency, and increased resistance to weed invasion compared to monocultures reducing the intensity of chemical weed control required (Connolly et al. 2018; Lüscher et al. 2014a). Evidence is also emerging that by combining grassland species with complementary functional traits, it is possible to provide an adaptation option to extreme weather events, such as drought (Hofer et al. 2016). However, there are barriers to the uptake of multi-species grassland systems by farmers, these include low temporal persistence of legume species in intensively managed swards and the requirement for farmers to implement new management practices (Lüscher et al. 2014b).

4.11.2.5 Conclusion

- A combination of land sparing and sharing options can be utilised – their application is case specific (flexibility in policy required).
- Improved crop and livestock genetics can close yield gaps – co-benefits for both land sharing and sparing options.
- Intensification needs to be achieved sustainability – or the short term becomes the enemy of the long term.

4.11.3 Case study showing the patchiness of LD

PLACEHOLDER. To be completed

4.11.4 Perennial Grains and Soil Organic Carbon

Cropland soils can theoretically hold far more carbon than they currently do. We know this because with few exceptions, comparisons between cropland soils and those of proximate mature native ecosystems show a 20-70% decline in soil carbon attributable to agricultural practices. What happens when native ecosystems are converted to agriculture that induces such significant loses of soil organic carbon (SOC)? On most landscapes, the removal of vegetation and direct exposure of soil to wind and rain greatly accelerates loses to erosion (Montgomery 2007a,b). In processes of both wind and water erosion, light organic matter fractions that can accumulate on or near the soil surface are commonly lost preferentially resulting in an overall decline in SOC (Lal 2003).

Aside from the effects of erosion, the fundamental practices of growing annual food and fiber crops alters both inputs and outputs of carbon from most agroecosystems resulting in net reductions in soil carbon equilibria (McLauchlan 2006; Crews et al. 2016a). Native vegetation of almost all terrestrial ecosystems is dominated by perennial plants, and the belowground carbon allocation of these perennials is a key variable in determining formation rates of stable SOC (Jastrow et al. 2007a; 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).

Reliance on annual crop species necessitates the control of competing vegetation (weeds) in agroecosystems. This control has traditionally been achieved using various types of tillage that promote an intensity and frequency of soil disturbance unmatched in native ecosystems (Crews et al. 2016b). Plowing, disking or other forms of cultivation alter the physical structure of topsoils, breaking open aggregates that had been formed over time through a combination of biological and physical pedogenic processes. Among other functions, well-developed soil aggregates are thought to deter soil bacteria, fungi and other microbes from consuming soil organic matter (Grandy and Neff 2008). When native ecosystems are converted to agriculture, and tillage breaks open soil aggregates, microbial consumption of SOC and subsequent respiration of CO_2 increase dramatically, reducing soil carbon stocks (Grandy and Robertson 2006a; Grandy and Neff 2008).

Many management approaches are being evaluated to increase stable forms of SOC in the world's croplands (Paustian et al. 2016). The menu of approaches being investigated focus either on increasing belowground 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 approaches 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 marketed 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 perennial grain crops receiving attention include pigeon pea, barley, buckwheat and maize (Batello et al. 2014; Chen et al. 2018a) and a number of legume species (Schlautman et al. 2018).

In a perennial agroecosystem, the biogeochemical controls on soil carbon accumulation shift dramatically, and begin to resemble the controls that govern native ecosystems (Crews et al. 2016b). 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 increase (Crews and Rumsey 2017). 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 CRP program of the U.S., 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-oxidizable) C after 4 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 rates of carbon accumulation can be (Jastrow et al. 2007b). 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 the net C accumulation rate of about 1500 g C m^{-2} yr^{-1} in the first year of the study corresponding to the biomass of Kernza increasing, to about 300 g C m^{-2} yr^{-1}in the final year where CO_2 respiration loses from the decomposition of roots and soil organic matter approached new carbon inputs from photosynthesis. Based on measurements of soil carbon accumulation in restored grasslands in this part of USA, the net carbon accumulation in stable organic matter under a perennial grain crop might be expected to sequester on the order of 30-50 g C m^{-2} yr^{-1}(Post and Kwon 2000).

The reason for the high rate of carbon sequestration is primarily attributed to the very deep and extensive root system (Figure 4.7).

Figure 4.7 Comparison of root systems between the newly domesticated intermediate wheatgrass (left) and annual wheat (right). Photo and copyright: Jim Richardson.

Single species stands of perennial grains would be expected to accumulate more soil C than comparable annual grains (e.g., Kernza and wheat), but an even higher degree of ecosystem services should at least theoretically be achieved by strategically combining different functional groups of crops such as a grain and a legume. Not only is there evidence from plant diversity experiments that communities with higher species richness appear to sustain higher concentrations of soil organic carbon (Hungate et al. 2017; Chen et al. 2018b), but other valuable ecosystem services such as pest suppression, lower greenhouse gas 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 life span, most likely in the range of 3-10 years. A key area of research on perennial grains cropping systems is to minimise losses of soil organic carbon during conversion of one stand of perennial grains to another. Work in the upper Midwest USA has demonstrated large net increases in CO_2 and N_2O emissions for several years following initial cultivation of a perennial grassland and conversion to annual crops (Grandy and Robertson 2006b). Large emissions are to be expected in such a conversion given the disruption of soil aggregates caused by tillage and the replacement of perennial vegetation with annuals. More recent work, however suggests that even no-till conversion of a mature perennial grassland stand to another perennial crop will experience several years of high net CO_2 emissions before net carbon uptake from the atmosphere occurs (Abraha et al. 2018).

4.11.5 Reversing Land Degradation, examples of large scale restoration of degraded lands (adaptation and mitigation) (China, South Korea)

4.11.5.1 Korea Case Study on Reforestation Success

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

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. 2017a).

The forest carbon density in Korea has increased from 5–7 Mg C ha^{-1} in the period 1955–1973 to more than 30 Mg C ha^{-1} in the late 1990s (Choi et al. 2002). Estimates of C uptake rates in the late 1990s were 12 Tg C yr^{-1} (Choi et al. 2002). For the period 1954 to 2012 C uptake was 8.3 Tg C yr^{-1} (Lee et al. 2014), lower than other estimates because reforestation programs did not start until 1973. NEP in South Korea was 10.55 ± 1.09 Tg C 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 Korea's forest area during 1992-2034 was 11.8 Tg C yr^{-1} (Kim et al. 2017b).

Figure 4.8 Example of severely degraded hills in South Korea and stages of forest restoration. Many examples of such restoration success exist throughout South Korea (Source: Korea Forest Service).

During the period of forest restoration, Korea also promoted inter-agency cooperation and coordination, especially between the energy and forest sectors, to replace firewood with fossil fuels, and in doing so reduced the demand for firewood and helped forest recovery (Bae et al. 2012). As experience with forest restoration programs has increased, emphasis has shifted from fuelwood plantations, often with exotic species and hybrid varieties to planning more native species and encouraging natural regeneration (Kim and Zsuffa 1994; Lee et al. 2015). Additionally, from this experience, should be highlighted the importance of avoid monocultures in the reforestation programs, since this can become the focal points for the spread of pests and less nutrients mineralisation (Kim and Zsuffa 1994). Another important factor in the success of the reforestation program is that private landowner were heavily involved in initial efforts (both corporate entities and smallholders) and the fact that the reforestation program was made part of the national economic development program (Lamb 2014). In summary, the forestation program was a comprehensive technical and social framework that recovered the forest ecosystems and enhanced the economic performance of rural regions (Kim et al. 2017a).

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

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

4.11.6 Degradation of tropical peat soils

Globally, peatlands cover about 3% of the Earth's land area (about 400 million hectares) and store one-third of global soil carbon. They are most abundant in high northern latitudes, where they extend over 4Mkm^2 and store between 550 and 700 Pg C. Tropical peatlands are thought to cover about 400,000 km^2 and store an additional 50 to 100 PgC, primarily in Southeast Asia (Yu 2012; Page et al. 2011;Yu et al. 2010). However, given the lack of data and the difficulty tracing the origins of some estimates in many countries, tropical estimates in particular are highly uncertain (Page et al. 2011).

Recognising these constraints, Gumbricht et al. (2017) developed a novel method for mapping tropical wetlands and peatlands using a hybrid expert system approach with hydrological modelling, time series analysis of soil moisture phenology from optical satellite data, and hydro-geomorphology from topographic data. This approach yielded surprisingly high areas and volumes of tropical peatlands (1.7 Mkm^2 and 7,268 km^3), which were more than threefold higher than previous estimates. The new map suggests that South America rather than Asia harbors the largest tropical peatland area and volume (around 44% for both), partly related to some previously unaccounted deep deposits, but mainly to the prediction of the existence of extensive shallow peat in the Amazon Basin. The model needs further validation, but recent data from Central Africa (Dargie et al. 2017), the Western Amazon (Lähteenoja et al. 2012), and the Central Amazon (Lähteenoja et al. 2013) show that tropical peat deposits are much more widespread than previously thought.

Tropical peatland dynamics are most well understood in Southeast Asia. Over the last 8000 years, long-term carbon accumulation rates estimated using ^{14}C to date specific layers of the peat profile in the coastal peatlands have been between 0.59 and 0.77 M C ha^{-1} yr^{-1}, while inland peatlands have accumulated carbon at about 0.3 Mg C ha^{-1} yr^{-1} (Kurnianto et al. 2015; Dommain et al. 2011). Analyses of the long-term dynamics and accumulation rates are needed for other regions with extensive peatlands. Sampling is much less extensive in western Amazonia, but data suggest that peat accumulation in that region began more recently than in Southeast Asia and that peat accumulation in that region has been punctuated with periods of no accumulation. Carbon accumulation estimates are not available for peatlands in the region (Kelly et al. 2017).

The climate impacts of 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. The CO_2 sink is lost when peatlands are drained and converted into a net source to the atmosphere. Oil palm is the most widespread plantation crop and on average it emits 11 MgC/ha yr^{-1};

Acacia plantations for pulpwood are the second most widespread plantation crop and emit 20 MgC ha^{-1} yr^{-1} (Drösler et al. 2013). Other land uses typically emit less than 10 MgC ha^{-1} yr^{-1}. Total emissions from peatland drainage in the region are estimated to be between 0.03 and 0.2 $PgCyr^{-1}$ (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 Tg CH_4 yr^{-1} and 0.002 TgN_2O yr^{-1}, while disturbed peatlands emit 0.1 Tg CH_4 $yr^{-1}$and 0.2 Tg $yr^{-1}N_2O$ (Frolking et al. 2011). These N_2O emissions are likely low as new findings show that emissions from fertilised oil palm can exceed 20 kg N_2O–N ha^{-1} yr^{-1} (Oktarita et al. 2017).

Fire emissions from tropical peatlands are only a serious issue in Southeast Asia, where they are responsible for 173 (18–1110) TgC yr^{-1} (van der Werf et al. 2017). Much of the variability is linked with the El Niño Southern Oscillation, 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). These fires have significant societal impacts. Koplitz et al. (2016), for example, used smoke exposure to estimate that the 2015 fires caused over 100,000 additional deaths across Indonesia, Malaysia and Singapore and that this event was more than twice as deadly as the 2006 El Niño event.

Peatland degradation in other parts of the world differ from Asia. In Africa 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* and Suri worms are major sources of degradation that lead to losses of carbon stocks (Hergoualc'h et al. 2017).

The effects of peatland conversion and degradation on livelihoods has 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 USD 3,835 and 9,630 to land owners (Butler et al. 2009). High financial returns are creating the 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 is likely to have 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. The first step is to restore the hydrological regime in drained peatlands and 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 been accomplished (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, but 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).

4.11.7 Increasing frequencies and intensities of woodlands forest fires

Potentially a cross-chapter box: Ch4, Ch2, Ch6, Ch7

In many forest biomes, wildfires are an essential component of forest ecology and the life cycle of trees. In boreal biomes, wildfires release nutrients that have accumulated in slowly decomposing organic matter over decades. Wildfires also reduce the organic layer depth, thus altering the thermal regime, allowing mineral soil to warm, nutrient availability to increase and where permafrost exists, the depth of the active layer increases in the post-fire years. Increased soil temperatures, greater nutrient availability, and deeper active layer all contribute to increased site productivity allowing for the regeneration of forests. Many boreal species have serotinous cones, in which seeds are protected from fire inside cones covered by resin. The heat of the fire causes these cones to open after the fire allowing for seeds to be released to the ground where thinning or removal of organic layers during the fire has created micro-sites suitable for regeneration. Thus, under past climatic conditions boreal ecosystems are resilient to fires, which can destroy the existing stand but rejuvenate the site allowing for forest regeneration and renewal.

Temperate forests especially eucalypt forests are adapted to regular fire of low to moderate intensity – the overstorey is largely unaffected, and fire stimulates germination of understorey. Severe stand-replacing fire is infrequent (typically decades or in some cases centuries between events) but important for regeneration of some species (e.g. wet eucalypt forests in Southern Australia). Increasing frequency of high-intensity fire is causing range retreat of fire-sensitive species (Holz et al. 2015) and also fire-dependent species (Bowman et al. 2014) changing species composition of forests (Bowman et al. 2014).

Climate change is predicted to increase both the fire frequency and intensity in boreal and temperate forests (Flannigan, M.D., Logan, K.A., Amiro, B.D., Skinner, W.R., and Stocks 2005; Flannigan et al. 2005, 2009; Balshi et al. 2009; Enright et al. 2015). Land degradation and forest loss can occur when the fire return frequency is shorter than the time required for trees to reach reproductive maturity and produce sufficient seeds to support post-fire regeneration. Repeated fires will destroy the existing forest but the absence of seeds can force succession to alternate, non-forest pathways (Lavoie and Sirois 1998; Girard et al. 2008, 2009; Côté et al. 2013). In dry boreal regions, forests can switch to grassland ecosystems (Thompson et al. 2009) while in moister regions lichen shurblands may be established that can prevent forest regeneration for decades and beyond (Girard et al. 2009, 2008). Ground fires are dominant in Russia. But fires in the forests, which kill trees also occur, especially in Siberia. Normally, only part of the forest area is burned and seeds from neighboring areas contribute to the regeneration of the burnt areas.

Ground fires can lead to weakening of forests and the rapid spread of insects. Trees affected by pests in managed forests are cut down, while in unmanaged forests trees affected by insects can die.

In boreal conditions in Russia, after the burning out of coniferous forests, their restoration takes place. In this case, a sphagnum swamp is formed at the first stage, then it is replaced by a small-leaved forest, and after this the coniferous forest begins to regenerate. The whole process takes about 30-40 years. But this is possible if the climatic conditions remain favorable for coniferous forests. If the climatic conditions become unfavorable, then the forest itself is not restored (good example – forest fires in southern France in Mediterranean zone).

Regeneration failure following forest fires in areas where the climate has become unfavorable for forest vegetation is a climate-change induced mechanism for changing the type of vegetation.

Wild fires are considered to be one of the main reasons land degradation (Ferreira et al. 2008; Mataix-Solera et al. 2011). Wild fires also have impact on soil erosion by reducing the rainfall interception and storage and infiltration capacity (Swanson 1981). Post wild fire intense rainfall events also increase the soil erosion which will in turn lead to large–scale, long-term land degradation. It is reported that the sub-humid climatic conditions of European Mediterranean Countries are vulnerable to wild fires (Rulli et al. 2006). Esteves and co-authors (2012) have used the PESERA model to study land degradation caused by wild fires in Central Portugal and found that the models have the capability to suggest the mitigation measures. It is also reported that the frequent forest fires in northern Ethiopian region cause soil erosion and decline in soil fertility, which leads to land degradation (Descheemaeker et al. 2009).

Due to lack of consistent definitions and data on fire events across jurisdictions in Australia, there is little direct evidence of changing climate on bushfires from fire statistics, but there are studies on changes in fire weather and fire behavior (e.g., (Sullivan 2010; Matthews et al. 2012), from which changes in fire frequency and severity can be inferred/predicted. These studies showed that in forests, the state of the forest fuel condition (i.e. indirect climate effect) has a far greater impact than direct effects through weather and moisture (though (Price and Bradstock 2012) concluded the opposite – weather conditions were the major cause of Australia's worst-ever bushfires).

Experts do not expect forest to savannah conversions in Australia—there is not the consistency of ignition sources and fuel availability in temperate Australia as is found in northern Australia. The changes (positive and negative) to the shrubby component of temperate forests will play a very large role in species shift but currently it is not clear which mechanisms will dominate here. Shrubs need extended fire free-periods to establish, which may not occur under climate change, although given the weather variability it is possible that a five to ten-year period of wetter than 'new' normal may arise, allowing shrub establishment to happen.

Beringer (2015) show how savannah fire influences on the rest of the earth system via biophysical and biogeochemical cycles with feedbacks to regional and global climate.

Combined impacts of *fire* and drought: "Field studies in the southwestern Yukon Territory showed that recent warming and drying have led to poor regeneration of spruce-dominated forests following fire and are leading to a shift toward aspen on south-facing slopes and at low elevations (Johnstone et al. 2011). At some sites, former spruce-dominated forests have been replaced by scattered clones of aspen interspersed by grassland, thus resembling the vegetation of the parkland zone of the Prairie provinces much further south (Hogg and Wein 2005)." (Price et al. 2013) – but he does not specifically address repeated fires and grassland conversion.

Table 1 of (Reyer et al. 2015) includes examples of degradation impacts around the world which we will draw on for the fire section.

"Drought and fire have induced transition from forest and savannahs to shrublands in south Portugal (Acácio et al. 2007, 2009; Acácio and Holmgren 2014)".

4.11.8 Hurricane-induced land degradation

Despite tropical cyclones being normal disturbances that natural ecosystems have been affected by and recovered from for millennia, their characteristics have changed or will change in a warming climate (Knutson et al. 2010). Large amplitude fluctuations in the frequency and intensity complicate both the detection and attribution of tropical cyclones to climate change. However, it is likely that the frequency of high-intensity hurricanes is expected to increase due to global climate change (Knutson et al. 2010; Bender et al. 2010; Vecchi et al. 2008). Tropical cyclones could therefore 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 (Boose et al. 2004; Luke et al. 2016).

Coastal wetland regions are under serious threat and have been suffering from severe degradation. Coastal wetlands function as valuable, self-maintaining "horizontal levees" for storm protection, and also provide a host of other ecosystem services that vertical levees do not. Their restoration and preservation is an extremely cost-effective strategy for society. The US were estimated to provide 23.2 billion USD yr^{-1} in storm protection services (Costanza et al. 2009). Thus, the maintenance of these wetlands is critical to prevent coastal degradation. Floodplains, mangroves, seagrasses, saltmarshes, arctic wetlands, peatlands, freshwater marshes and forests are very diverse habitats, with different stressors and hence different management and restoration techniques are needed.

Starting in the 1990s, wetland restoration and re-creation became a "hotspot" in the ecological research fields (Zedler 2000). The US government enforced the regulatory policy of 'no net loss' of wetlands, combined with a focus on wetlands banking to ensure minimum impacts on wetlands.

Over the last decade, the natural coastal wetlands in China have deeply declined. The reasons for China's wetland degradation include climate change, sea level rise, biological invasions, marine disasters, and other natural factors such as hurricanes (Jiang et al. 2015).

Data regarding China's successful efforts in wetland restoration (PLACEHOLDER)

Data regarding Caribbean successful efforts in wetland restoration (PLACEHOLDER)

4.11.9 Saltwater intrusion

Current environmental changes including climate change have led sea levels to rise worldwide, particularly in the tropical and subtropical regions. Combined with scarcity of water in river channels, such rise has been instrumental in intrusion of highly saline seawater inland posing a threat to coastal areas and an emerging challenge to managers and policymakers. Assessing the extent of salinisation due to sea water intrusion at a global scale nevertheless remains challenging. Wicke et al. (2011) suggested 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.

The seawater intrusion is generally mediated by the following factors: i) increased tidal activity including storm surges, hurricanes, sea storms, etc. 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 mangroves forests and wetlands; and iv) construction of vast irrigation canals and drainage networks leading to low river discharge in the deltaic region. The Indus delta, located in the southeastern coast of Pakistan near Karachi in the North Arabian sea and one of the six largest estuaries in the world spanning over an area of 600,000 ha (Figure 4.9) is a clear example of sea water intrusion and land degradation due to local as well as upcountry climatic and environmental conditions (Rasul et al. 2012).

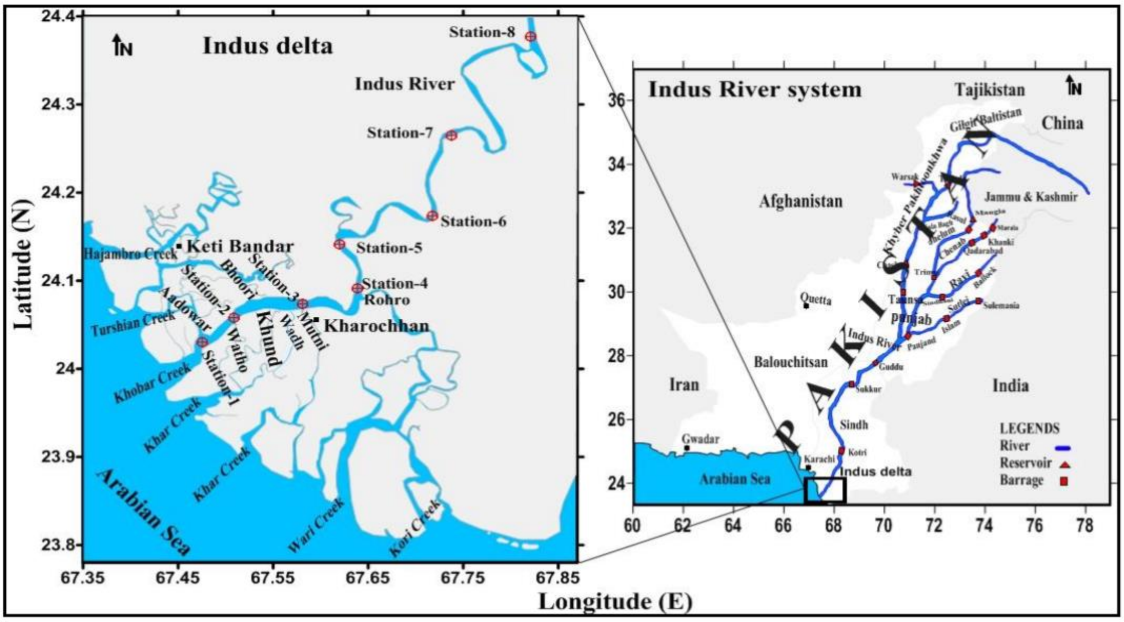

Figure 4.9 Indus delta with a network of major creeks and tributaries (Source Kalhoro, N.A.; He, Z; D. Xu, I.; Muhammad, A.F.; Sohoo 2017).

Such degradation takes the form of high soil salinity, inundation and waterlogging, erosion and freshwater contamination. The inter-annual variability of precipitation with flooding conditions in some years and drought conditions in others has caused variable river flows and sediment runoffs below Kotri barrage (about 200 km upstream of the Indus delta) (Figure 4.10). This has affected hydrological processes in the lower reaches of river and the delta, contributing to the degradation (Rasul et al. 2012).

Figure 4.10 A view of the dry Indus river at Kotri barrage, about 200 km upstream of Indus delta taken from (Kalhoro, N.A;, He, Z.; Xu, D.; Faiz, M.;Yafei, L.V. Sohoo 2016).

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; Anwar. & Chandio 2011) A very high erosion rate of 0.179±0.0315 km yr^{-1}, based on the analysis of satellite data, was observed in the Indus delta during the past 10 years (2004-15).

The area of agricultural crops under cultivation has been declining with economic losses of millions of dollars (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 (Aug 2014), the salinity (0.18 mg L^{-1}) reached 24 km upstream while during the dry season (May 2013), the salinity reached 84 km upstream. 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 is causing widespread diseases, of which diarrhoea is most common (IUCN 2003).

The seawater intrusion has been experienced by many other countries. The Lake Urmia in northwest Iran, the second largest saltwater lake in the world and the habitat for endemic Iranian brine shrimp, *Artemia urmiana*, has been affected by sea water ingression. During the 17-year period from 1998 to 2014, human disruption and years of dam building has attracted the natural flow of sweet water as well as salty sea water from the surrounding area into Urmia Lake (Figure 4.11). The quality of water has also been adversely affected with its salinity fluctuating over time, but in recent years reaching a maximum of 340 g L^{-1}. This has rendered the underground water unfit for drinking and agricultural purposes and risky for human health and livelihood.

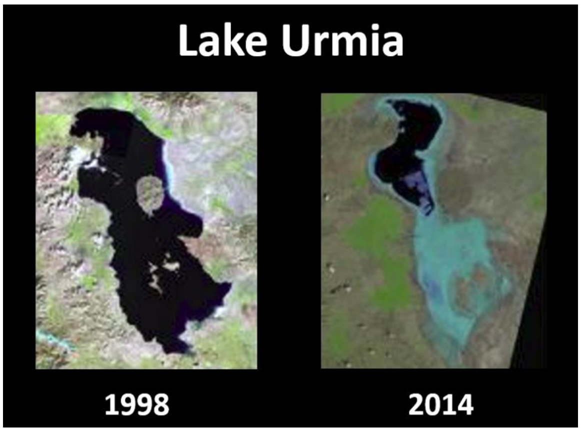

Figure 4.11 Urmia Lake condition in years 1998 and 2014

The rapid irrigation expansion in the basin has, however, indirectly contributed to inflow reduction. Annual inflow to Urmia Lake 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 (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 center of Sonora Desert in the northwest of Mexico. The 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 (Halvorson et al. 2003).

Intrusion of seawater is exacerbated by destruction of mangrove forests. Mangroves are important coastal ecosystems that provide spawning bed for fish, timber for building, livelihood to dependent communities, 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. Unfortunately, they 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, etc.) leading to declines in their areas. In Pakistan, using remote sensing (RS) techniques, the mangrove forest cover in the Indus delta has been found to decrease from 260,000 ha in 1980s to 160,000 ha in 1990 (Chandio et al 2011). Based on non-linearity analysis and RS data, a sharp decline in the mangrove area was found in the arid coastal region of Hormozgan province in southern Iran during 1972, 1987 and 1987 periods (Etemadi et al. 2016). Myanmar has the highest rate (about 1%) of mangrove deforestation in the world (Atwood et al. 2017), and more than one-third of mangroves have vanished worldwide (Hamilton and Casey 2016). Regarding global loss of carbon stored in the mangrove soil due to deforestation, four countries were on top; Indonesia (3,410 Gg CO_2 yr^{-1}), Malaysia (1,288 Gg CO_2 yr^{-1}), US (206 Gg CO_2 yr^{-1}) and Brazil (186 Gg CO_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 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. Also, a viable policy framework is needed to ensure the environmental flows to Indus delta, to repulse the intruding sea water in consultation with different stakeholders.

4.11.10 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 addressing both land degradation and climate change, simultaneously.

4.11.10.1 Role of biochar in climate change mitigation

Biochar is relatively resistant to decomposition compared with fresh organic matter, so represents a long-term C 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, Cowie, & Smernik, 2012; Wang, Xiong, & Kuzyakov, 2016). Biochar stability is influenced by soil properties: biochar carbon can be further stabilised by interaction with clay minerals and native soil organic matter (*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 C on biochar, and formation of biochar-organo-mineral complexes (Weng et al., 2015;Wang et al., 2016; Han Weng et al., 2017; Weng et al., 2018). 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 & Cowie, 2014; Wang et al., 2016). Negative priming has been observed particularly in clay-dominated soils, whereas positive priming is observed in sandy soils (Wang et al. 2016). Thus, biochar carbon stability is greatest in clay soils in temperate environments; biochar stimulates negative priming and therefore builds soil carbon in clay soils especially in temperate environments but stimulates loss of native soil carbon in sandy soils and at higher ambient temperatures.

Biochar can provide additional climate change mitigation by lowering nitrous oxide (N_2O) emissions from soil: meta-analyses found an average decrease in emissions from soil of 30%-54%, though the impact varies widely (Cayuela, Jeffery, & van Zwieten, 2015; He et al., 2017). The effect is due in part to decreased substrate availability for denitrifying organisms, driven by the molar H/C ratio of the biochar (Cayuela et al. 2015).

Biochar reduces methane emissions from flooded soils, such as rice paddies, but also reduces methane uptake by dryland soils (*medium evidence*) though the latter is a small effect and will have limited impact in absolute terms (Jeffery et al. 2016).

Additional benefits of biochar may arise through reduced N fertiliser requirements, due to reduced losses of nutrients through leaching and/or volatilization (Singh, Hatton, Balwant, & Cowie, 2010); increased plant yield, particularly in light-textured soils, and acidic tropical soils (Simon et al. 2017); avoided emissions from decomposition of organic wastes that are instead used for biochar, such as manure that would otherwise be stockpiled, crop residues that would be burned or processing residues that would be landfilled; reduced emissions from compost (Agyarko-Mintah et al., 2017; Wu et al., 2017).

Biochar is a potential "negative emissions" technology: the thermochemical conversion of biomass to biochar slows mineralisation of the biomass, delivering long term C 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 carbon capture and storage (Smith 2016). Studies of the life cycle climate change impacts of biochar systems generally show emissions reduction in the range 0.4 -1.2Mg CO_2e Mg-1 (dry) feedstock (Cowie, A.; Woolf, A.D.; Gaunt, J.; Brandão, M.; de la Rosa 2015). A global analysis in which sustainability constraints were applied to protect against food insecurity, loss of habitat and land degradation, found that annual net GHG emissions could be reduced by 3.7 - 6.6Pg CO_2e yr^{-1} (7% to 12% of 2012 anthropogenic GHG emissions), with total net emissions over the course of a century reduced by 240 – 475 Pg CO_2e (Woolf et al. 2010).

4.11.10.2 Role of biochar in management of land degradation

Biochar can contribute to management of land degradation through the following documented benefits:

- Improved nutrient use efficiency: biochars can enhance retention of N and availability of phosphorus (P) in soils with high P fixation capacity, potentially reducing fertiliser requirements. Furthermore, biochar produced from nutrient dense feedstocks, such as poultry litter, can substitute chemical fertiliser.
- Management of heavy metals: application of biochar can substantially reduce plant uptake of toxic elements (O'Connor et al., 2018; Peng ; Deng, ; Peng, & Yue, 2018), by reducing availability, through immobilisation due to increased pH and redox effects (Rizwan et al. 2016) thus providing an affordable means of remediating contaminated soils, and enabling the continued utilisation of such soils for food production.
- Improved water holding capacity, particularly in sandy soils (Omondi et al. 2016).
- Biochar systems can deliver a range of other co-benefits, such as waste management, destruction of pathogens and weed propagules, avoidance of landfill, improved ease of handling, management of odors, reduction in environmental N pollution, protection of waterways and soil remediation.

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).

In summary, biochar is a technology that can simultaneously enhance soil productivity and contribute to climate change mitigation and sustainable development. Studies have found that biochars can improve plant yields, enhance soil water holding capacity and reduce fertiliser requirements, though results vary widely between different biochars, soil types, climates and target crops. Agronomic and methane reduction benefits appear greatest in tropical regions, while carbon stabilisation is greater in temperate regions. Biochar will be most beneficial if made from biomass residues, formulated to address identified soil constraints, and applied in low volumes to the most responsive soils. Scaling-up of biochar will be limited by biomass availability and cost of biochar production.

4.11.11 Avoiding coastal degradation induced by maladaptation to sea-level rise on small islands

Coastal degradation—for example, beach erosion, coastal squeeze, 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 4.5.1 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 (*medium confidence*) (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 the specific case of adaptation to sea-level rise, it can be considered maladaptation when the construction of adaptation measures leads to coastal degradation, as can be observed on many small islands (*high agreement, medium evidence*). In many cases, increased rates of coastal erosion by 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. 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 even increased sand loss in those areas (Sealey 2006). The reduction of natural buffer zones—for example, beaches and dunes—due to vertical structures, such as sea walls, increased the impacts of tropical cyclones on Reunion Island (Duvat et al. 2016). Coastal degradation issues from the construction of sea walls, however, are not only observed in Small Island Developing States (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).

The adverse effects of coastal protection measures may be avoided by the consideration of local social-ecological dynamics, including the critical studying of diverse drivers of ongoing shoreline changes, and the according implementation of locally adequate coastal protection options (*medium confidence*) (French 2001; Duvat 2013). In some cases, it may be possible to keep intact and restore natural buffer zones as 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 of Barbados show that combined approaches 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).

4.12 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.

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

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

Attribution is a challenge because a complex web of causality rather than simple cause-effect relationships. 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.

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