

1 **Chapter 4: Land Degradation**

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1 **4.1 Executive summary**

2 **Land degradation resulting in a decline in productive capacity, ecological integrity or value to**
3 **humans, is linked to climate change in many complex ways** (*very high confidence*). In this report,
4 land degradation is defined as a *negative trend in land condition resulting in long term reduction or*
5 *loss of the biological productivity of land, its ecological integrity or its value to humans, caused by*
6 *direct or indirect human-induced processes, including climate change*. Therefore, in this context, if it
7 is not accompanied by a decline in productive capacity, a decrease in carbon stock does not necessarily
8 indicate land degradation. Forest degradation is land degradation which occurs in forested land {4.3.1,
9 4.3.2}.

10 **Climate change exacerbates many land degradation processes in terms of rates and magnitudes**
11 **of degradation, therefore sustainable land management is even more urgently required to avoid,**
12 **reduce and reverse degradation** (*very high confidence*). Climate change exacerbates processes
13 through increasing heat stress and drought, more intense rainfall, sea-level rise, and increased wind and
14 wave action, yet land management determines whether the land becomes degraded or not (*high*
15 *agreement, robust evidence*). Agriculture and clearing of land for food and timber production are key
16 drivers of land degradation. However, this does not necessarily mean that agriculture and forestry
17 always cause land degradation; sustainable management is possible but not always practiced. Reasons
18 for this are economic, political, and social, including lack of knowledge {4.4.1, 4.4.2, 4.4.3, 4.2.2}.

19 **Land degradation occurs as a result of conversion of tropical forests to agriculture** (*very high*
20 *confidence*) **and in agriculture worldwide wherever land management is unsustainable** (*very high*
21 *confidence*). There are no reliable global estimates of the extent and severity of land degradation due to
22 both conceptual and methodological reasons. Proxies based on remote sensing indicate that 20-25% of
23 the global land area is subject to some form of degradation while about 15-20% shows signs of
24 increasing vegetation cover (*low confidence*). Estimates of tropical forest cover from remote sensing
25 and collated national statistics diverge making implementation and enforcement of policies difficult
26 {4.5.1, 4.5.2}.

27 **Changes in rainfall distribution in time and space, and intensification of rainfall events increase**
28 **the risk of land degradation, both in terms of likelihood and consequences** (*very high confidence*).
29 Changes of hydrological regimes as a combined result of human land/water use and climate change will
30 impact floodplains and delta areas with detrimental effects on livelihoods, human habitats, and
31 infrastructure (*very high confidence*). Climate induced vegetation changes will pose increasing risk of
32 land degradation in some areas (*medium confidence*). Erosion of coastal areas because of sea level rise
33 will increase worldwide (*very high confidence*). In hurricane prone areas the combination of sea level
34 rise and more intense hurricanes will pose a serious risk to people and livelihoods (*very high confidence*)
35 {4.6.1}.

36 **Most forms of land degradation can be prevented and reversed with adequate actions** (*very high*
37 *confidence*). Whether the land becomes degraded or not depends on the interplay between biophysical
38 factors and human factors. Biophysical factors include climate, landscape, vegetation, and soil
39 conditions; human factors include land use and management, land tenure, the socio-economic and
40 livelihood context. Land degradation also has political and cultural (including gendered) aspects in
41 terms of causes, consequences and remedies. Proven methods and approaches exist for avoiding,
42 reducing and reversing land degradation under the umbrella term Sustainable Land Management (SLM)
43 {4.10.1, 4.10.2}.

44 **While climate change exacerbates many of the ongoing land degradation processes of managed**
45 **ecosystems, it also introduces novel degradation pathways in wild and semi-natural ecosystems**
46 (*high confidence*). Novel degradation pathways include changes in the water balance (drought,

1 rainstorms); intensified and more frequent fires; permafrost thawing and coastal erosion damage in wild
2 and semi-natural systems; stimulation of novel land uses aimed at mitigating climate change with poor
3 understanding of their local impacts. Climate change can also contribute to improved land productivity,
4 notably in high latitudes. The net balance of positive impacts in some regions and negative impacts in
5 others is not well known yet, the majority of affected population is located in the global South (*high*
6 *agreement, medium evidence*) {4.4.1, 4.4.2}.

7 **Intensification of land management associated with large-scale deployment of land-based NETs,**
8 **including fertilizer additions, irrigation, and the use of fast growing (monoculture) species**
9 **increases the risk of land degradation** (*medium confidence*). Few studies have specifically addressed
10 the impacts of land-based NETs on land degradation. Poorly implemented intensification of land
11 management can contribute to land degradation (e.g. salinization from irrigation) and disrupted
12 livelihoods. In areas where afforestation and reforestation occur on previously degraded lands,
13 opportunities exist to restore lands with potential significant co-benefits (*high agreement, robust*
14 *evidence*) {4.7.1, 4.7.2, 4.7.3, 4.7.6}.

15 **Land degradation is a driver of climate change through emission of terrestrial greenhouse gases**
16 **and deteriorating sinks of atmospheric greenhouse gases** (*very high confidence*). Over the past three
17 decades forest area has changed with net decreases in the tropics and increase in the extra-tropics, while
18 globally the forest area has increased (*medium confidence*). However, carbon density in areas lost was
19 higher than in regrowing forests resulting in net emissions to the atmosphere from land-use change.
20 These regional differences are due both biophysical differences, such as fire or drought occurrence or
21 human management such as; traditional wood-fuel collection, livestock grazing in forests, commercial
22 and smallholder agriculture, and logging. Agricultural soils have lost 60-75% of the original soil carbon
23 content prior to cultivation, and soils under conventional agriculture continues to be a source of
24 greenhouse gases (*medium confidence*) {4.5.1, 4.5.2, 4.8.1, 4.8.2, 4.8.3}.

25 **The current global extent and severity of land degradation is not well known.** Because land
26 degradation is such a complex and normative process there is no method by which land degradation can
27 be measured objectively and consistently over large areas (*very high confidence*). However, there exist
28 many approaches which can be used to assess different aspects of land degradation or provide proxies
29 of land degradation. Remote sensing, corroborated by other kinds of data, is the only method that can
30 generate geographically explicit and globally consistent data over time scales relevant for land
31 degradation (several decades). Simulation models are essential for better understanding how climate
32 change, ecosystem and landscape dynamics, and land use, interact and may or may not result in land
33 degradation. Yet, models have limited predictive capability due to the complex interactions and strong
34 effect of land management on the land degradation outcome {4.4.4, 4.5.1}.

35 **Land degradation processes affecting natural and semi-natural ecosystems such as deforestation,**
36 **increasing wild fires and permafrost thawing have their highest warming potential through the**
37 **release of greenhouse gases and the reduction in land carbon sinks where forests are lost** (*high*
38 *confidence*). Agricultural land is a dominant source of non-CO₂ greenhouse gases and these emissions
39 are exacerbated by climate change (*high evidence, medium to high confidence*). Degradation processes,
40 however, have additional physical effects on the global climate like those arising from albedo shifts
41 which can be significant and often opposed to those related to their carbon balance (*medium*
42 *confidence*). These interactions call for more integrative climate impact assessments {4.8.1, 4.8.2,
43 4.8.3}.

44 **Climate and land degradation can act as threat multipliers for poverty and vulnerability, both**
45 **individually and in combination** (*very high confidence*). Climate change, including increasing climate
46 variability, is increasing human and ecological communities' exposure and sensitivity to land
47 degradation, leaving livelihoods more sensitive to the impacts of climate change and extreme climatic

1 events. To adequately respond to food insecurity challenges, the interaction between climate and land
2 degradation need to be considered in the context of other socio-economic stressors (*high agreement,*
3 *limited evidence*). The impacts of climate change (drought, heat stress, more intensive rainfall) are
4 expected to negatively affect agricultural productivity and hence availability of and access to food,
5 particularly in tropical regions (*medium agreement, medium evidence*), which could be further amplified
6 by land use changes for NETs, with few studies assessing these impacts in different socio-economic
7 and climate settings {4.9.1, 4.9.2}.

8 **Land degradation exacerbated by climate change can have profound implications for human**
9 **population distributions, societal stability and cultural practices** (*low confidence*). There are diverse
10 links and outcomes between land degradation and human migration, conflict, and cooperation, as human
11 populations that are exposed and sensitive to degradation and climate change seek to reduce their
12 vulnerability and poverty and improve their development prospects. Land management practices
13 commonly stem from the intersection of cultural practices, gender and cultural norms and other social
14 identifiers, these aspects need central consideration in both understanding the impacts of climate related
15 land degradation and in the participatory design of responses {4.9.3, 4.9.4}.

16 **Land degradation can be addressed successfully in most cases by implementation of Sustainable**
17 **Land Management (SLM)** (*very high confidence*). SLM is a comprehensive approach of technologies
18 and enabling socio-economic conditions, which have proven to reduce and reverse land degradation at
19 scales from local farms (*high agreement, robust evidence*) to entire river basins (*medium agreement,*
20 *limited evidence*). Barriers to adoption of sustainable land/forestry management practices and
21 implementation of practices to reverse land degradation include lack of finance, skills and awareness
22 amongst land managers, and lack of incentives, such as where land users do not have secure tenure
23 (*very high confidence*). Culture and gender can also present barriers. Measures that facilitate
24 implementation of practices that reduce, or reverse land degradation include tenure reform, tax
25 incentives, payments for environmental services, integrated participatory land use planning, farmer
26 networks and extension officers {4.10.1, 4.10.2, 4.10.3, 4.10.4}.

28 4.2 Introduction

29 4.2.1 Scope of the chapter

30 This chapter examines the scientific understanding of how climate change impacts land degradation,
31 and vice versa, with a focus on non-drylands. Land degradation of drylands is covered in Chapter 3.
32 After providing definitions and the context (Section 4.3) we proceed with a theoretical explanation of
33 the different processes of land degradation and how they are related to climate and possibly to climate
34 change (Section 4.4). Two sections are devoted to a systematic review of the scientific literature on
35 status and trend of land degradation (Section 4.5) and projections of land degradation (Section 4.6).
36 Then follows a section where we examine the impacts of climate change mitigation options, bioenergy
37 and land-based negative emission technologies (NETs), on land degradation (Section 4.7). The ways in
38 which land degradation can have impacts on climate and climate change are examined in Section 4.8.
39 The impacts of climate related land degradation on human and natural systems is examined in Section
40 4.9. The remainder of the chapter discusses how land degradation can be addressed, based on the
41 concept of sustainable land management: avoid, reduce and reverse land degradation (Section 4.10),
42 followed by a presentation of nine illustrative case studies of land degradation and remedies (Section
43 4.11). The chapter ends with a discussion of the most important knowledge gaps and areas for further
44 research (Section 4.12).

1 **4.2.2 Perspectives of land degradation**

2 Land degradation has accompanied humanity since time immemorial but has accelerated after the
3 transition from hunters and gatherers to farmers some 10,000 years ago. This change of livelihoods, the
4 Neolithic revolution, has even been proposed as the onset of Anthropocene (Lewis and Maslin 2015).
5 There are indications that the levels of greenhouse gases (carbon dioxide and methane) of the
6 atmosphere started to increase already 8000 to 5000 years ago as a result of expanding agriculture,
7 clearing of forests, and domestication of livestock (Fuller et al. 2011; Kaplan et al. 2011; Vavrus et al.
8 2018). While the development of agriculture (cropping and animal husbandry) underpinned the
9 development of civilisations, political institutions, and prosperity, farming practices led to conversion
10 of forests and grasslands to farmland, and the heavy reliance on domesticated annual grasses for our
11 food production meant that soils started to deteriorate through seasonal mechanical disturbances (Turner
12 et al. 1990; Steffen et al. 2005; Ojima et al. 1994). In a long historical perspective, say millennia, a
13 significant proportion of ecosystems no longer function as they did before human induced land changes,
14 mainly as a result of agriculture and forestry, even if detailed evidence are scattered (Dupouey et al.
15 2002; Xinying et al. 2012; Kates et al. 1990). In a shorter time perspective, say decades, science has
16 been able to more accurately detect and describe significant changes of the face of the Earth. In terms
17 of climate change, since 1850, about 35% of the human caused emissions of CO₂ to the atmosphere
18 comes from land use change (Foley et al. 2005) and about 38% of Earth's land area has been converted
19 to agriculture (Foley et al. 2011), see further Chapter 2 for more details.

20 Not all human impacts on land result in degradation according to the definition of land degradation used
21 in this report (see Glossary), some impacts are positive, although degradation and its management are
22 the focus of this chapter. We also acknowledge that human use of land and ecosystems provides
23 essential goods and services for society (Foley et al. 2005; MA (Millennium Ecosystem Assessment)
24 2005). Land use is a socio-economic process which moves land from a natural to a used state, but how
25 the land is used determines whether the land use is sustainable or will lead to degradation over time.

26 Land degradation was long subject to a polarised scientific debate between disciplines and perspectives
27 in which social scientists often perceived that natural scientists exaggerated land degradation as a global
28 problem (Blaikie and Brookfield 1987; Forsyth 1996; Lukas 2014; Zimmerer 1993). The elusiveness
29 of the concept in combination with the difficulties of measuring and monitoring land degradation at
30 global and regional scales by extrapolation and aggregation of empirical studies at local scales, such as
31 the Global Assessment of Soil Degradation database (GLASOD) (Sonneveld and Dent 2009)
32 contributed to conflicting views. The conflicting views were not confined to science only but also
33 caused tension between the scientific understanding of land degradation and policy (Andersson et al.
34 2011; Behnke and Mortimore 2016; Grainger 2009; Toulmin and Brock 2016). Another weakness of
35 many land degradation studies is the exclusion of the views and experiences of the land users, whether
36 farmers or forest dependent communities (Blaikie and Brookfield 1987; Fairhead and Scoones 2005;
37 Warren 2002; Andersson et al. 2011). There are at least four important reasons for including the land
38 users' views and experiences in assessing land degradation: 1/ because of the complexity of land
39 degradation processes, measurements become more realistic; 2/ the assessment becomes more
40 integrated and hence relevant for the land users; 3/ the assessment also includes the perception of
41 potential links to climate change; 4/ assessments that include the users' views increases the chances of
42 implementing any measures (Stocking et al. 2001). More recently, the polarised views described above
43 have been reconciled under the umbrella of Land Change Science, which has emerged as an
44 interdisciplinary field aimed at examining the dynamics of land cover and land use as a coupled human-
45 environment system (Turner et al. 2007). A comprehensive discussion about concepts and different
46 perspectives of land degradation was presented in Chapter 2 of the recent report from the
47 Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES) on land degradation
48 (IPBES 2018).

1 Agriculture and clearing of land for food and timber production have been the main drivers of land
2 degradation since time immemorial (*high confidence*). This does not mean that agriculture and forestry
3 always cause land degradation, sustainable management is possible but not always practiced. Reasons
4 for this are primarily economic, political and social.
5

6 **4.3 Land degradation in previous IPCC reports**

7 Several previous IPCC assessment reports include brief discussions of land degradation. In AR5 WGIII
8 land degradation is one factor contributing to uncertainties of the mitigation potential of land-based
9 ecosystems, particularly in terms of fluxes of soil carbon (Smith et al., 2014, p. 817). In AR5 WGI, soil
10 carbon is discussed comprehensively but not in the context of land degradation, except forest
11 degradation (Ciais et al. 2013) and permafrost degradation (Vaughan et al. 2013). Climate change
12 impacts are discussed comprehensively in AR5 WGII, but land degradation is not prominent. Land use
13 and land cover changes are treated comprehensively in terms of effects on the terrestrial carbon stocks
14 and flows (Settele et al. 2015) but links to land degradation are to a large extent missing. Land
15 degradation was discussed in relation to human security as one factor which in combination with
16 extreme weather events has been proposed to be contributing to human migration (Adger et al. 2014),
17 an issue discussed more comprehensively in this chapter (4.9.3). Neither drivers nor processes of
18 degradation by which land-based carbon is released to the atmosphere and/or the long-term reduction
19 in the capacity of the land to remove atmospheric carbon and to store this in biomass and soil carbon,
20 has been discussed comprehensively in previous IPCC reports.

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

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

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

41 **4.3.1 Definitions of land degradation and land management**

42 In this report, land degradation is defined as a *negative trend in land condition resulting in long term*
43 *reduction or loss of the biological productivity of land, its ecological integrity or its value to humans,*
44 *caused by direct or indirect human-induced processes, including climate change.*

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

1 “Land degradation in arid, semi-arid, and dry sub-humid areas resulting from various factors, including
2 climatic variations and human activities. Land degradation in arid, semi-arid, and dry sub-humid areas
3 is a reduction or loss of the biological or economic productivity and integrity of rainfed cropland,
4 irrigated cropland, or range, pasture, forest, and woodlands resulting from land uses or from a process
5 or combination of processes, including processes arising from human activities and habitation patterns,
6 such as (1) soil erosion caused by wind and/or water; (2) deterioration of the physical, chemical,
7 biological, or economic properties of soil; and (3) long-term loss of natural vegetation” (IPCC WGII
8 2014; UNCCD 1994, Article 1).

9 The SRCCL definition is not intended to replace this more detailed definition, but rather to provide an
10 operational definition that is applicable to all regions, not just the drylands, and that emphasises the
11 relationship between land degradation and climate, for use in this report. Through its attention to the
12 three aspects biological productivity, ecological integrity and value to humans, the SRCCL definition
13 is consistent with the Land Degradation Neutrality (LDN) concept, which aims to maintain or enhance
14 the land-based natural capital, and the ecosystem services that flow from it (Cowie et al. 2018).

15 In the SRCCL definition, changes in land condition resulting solely from natural processes (such as
16 earthquakes and volcanic eruptions) are not considered land degradation. Climate variability
17 exacerbated by human-induced climate change can contribute to land degradation. The definition
18 recognises the reality that land-use decisions are likely to result in trade-offs between time, space,
19 ecosystem services, and stakeholder groups. The interpretation of a negative trend in land condition is
20 somewhat subjective, especially where there is a trade-off between ecological integrity and value to
21 humans. The use of “or” rather than “and” specifies that reduction of *either* biological productivity, *or*
22 ecological integrity *or* value to humans can constitute degradation, and any one of these changes need
23 not be considered degradation. Thus, a land transformation that reduces ecological integrity and
24 enhances sustainable food production need not be classed as degradation. Different stakeholder groups
25 with different worldviews are likely to value ecosystem services differently. Further, a decline in carbon
26 stock in biomass does not always signify degradation, such as when associated with periodic forest
27 harvest. Even a decline in productive potential may not equate to land degradation, such as when a high
28 intensity agricultural system is converted to a lower input more sustainable production system.

29 Land degradation is defined differently in the IPBES Land Degradation and Restoration Assessment
30 (LDRA) as “the many human-caused processes that drive the decline or loss in biodiversity, ecosystem
31 functions or ecosystem services in any terrestrial and associated aquatic ecosystems”. The IPBES
32 LDRA defines degraded land as: “the state of land which results from the persistent decline or loss of
33 biodiversity and ecosystem functions and services that cannot fully recover unaided within decadal time
34 scales” (IPBES 2018). The IPBES LDRA adds further that “Degraded land takes many forms: in some
35 cases, all biodiversity, ecosystem functions and services are adversely affected; in others, only some
36 aspects are negatively affected while others have been increased.” Thus, compared to the SRCCL
37 definition, the IPBES LDRA definition focuses on processes of land degradation, and emphasises
38 impacts on biodiversity and ecosystem functions and services, while the SRCCL definition focuses on
39 productivity outcomes and implications for human wellbeing. Under the IPBES LDRA definitions of
40 land degradation and degraded land, decline in biodiversity alone can be considered degradation, and
41 land altered by human management, compared with its natural condition, is generally considered
42 degraded. The SRCCL, in contrast, does not necessarily class such an impact as degradation.
43 Furthermore, the baseline for SRCCL is the condition at the start of the assessment, as it refers only to
44 the trend during the period of interest. In contrast, the IPBES LDRA discusses several alternative
45 baselines, and tends to favour the natural state, consistent with the mission of IPBES, that focusses on
46 conservation and sustainable use of biodiversity.

47 To clarify the scope of this chapter it is important to define land itself. The SRCCL defines land as “the
48 terrestrial system that comprises the natural resources (soil, near surface air, vegetation and other biota,

1 and water), the ecological processes, topography, and human settlements and infrastructure that operate
2 within that system” (Henry et al. 2018), adapted from (FAO 2007; UNCCD 1994). Sustainable land
3 management is defined as “the use of land resources, including soils, water, animals and plants, to meet
4 changing human needs, while simultaneously ensuring the long-term productive potential of these
5 resources and the maintenance of their environmental functions” (Adapted from World Overview of
6 Conservation Approaches and Technologies, WOCAT). Achieving the objective of ensuring productive
7 potential is maintained in the long term will require implementation of adaptive management and “triple
8 loop learning”, that seeks to monitor outcomes, learn from experience, and as new knowledge emerges
9 modifying management accordingly (Rist et al. 2013).

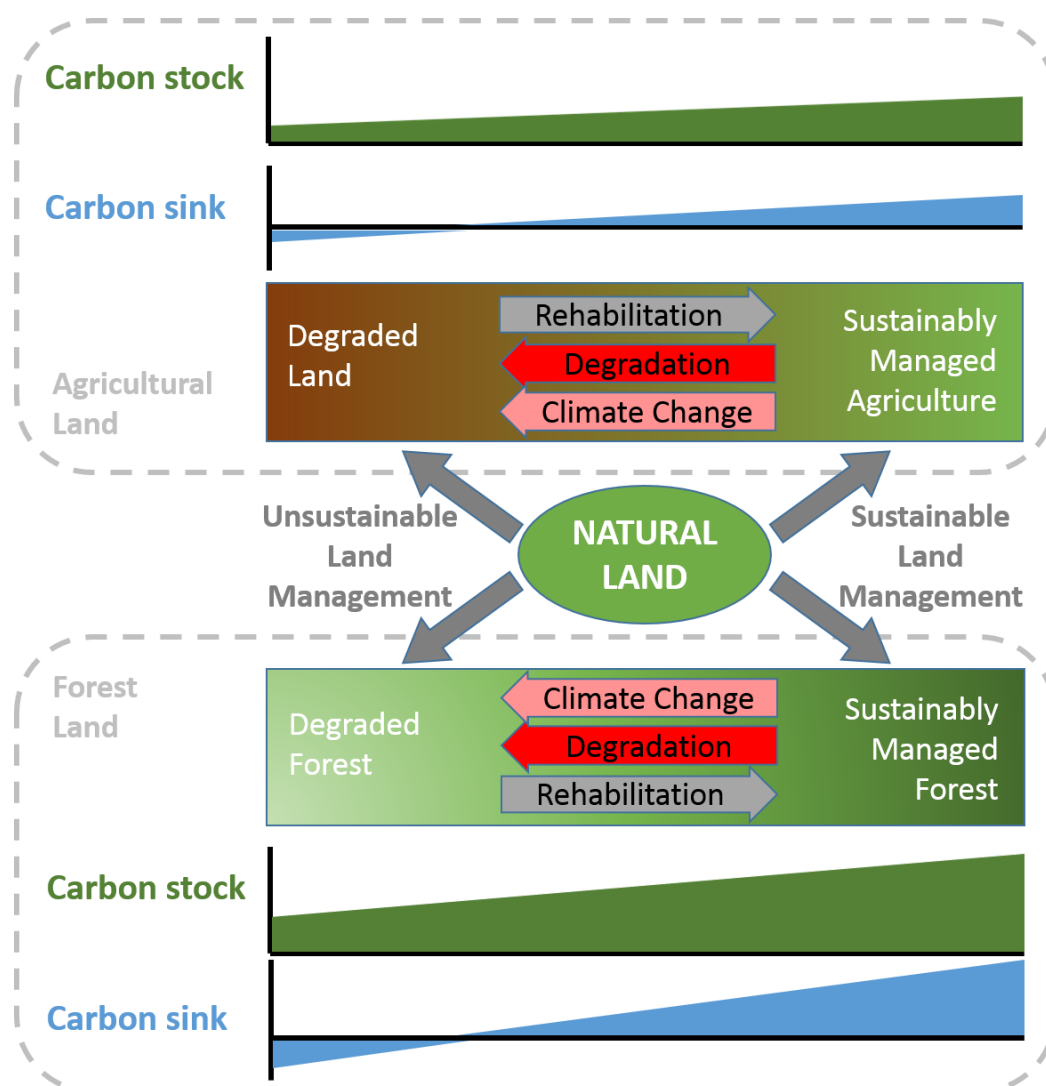
10 **4.3.2 Sustainable land and forest management**

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

15 Initial attempts to define forest degradation have taken an approach analogue to those used to define
16 land degradation, i.e. forest degradation is defined as a reduction in the productive capacity of forests,
17 (e.g., (Penman et al. 2003)). However, the difficulties in measuring and operationally implementing this
18 definition have been recognized and have resulted in attempts to develop alternate definitions (Penman
19 et al. 2003). More recent definitions focus on reductions in canopy cover or carbon stocks (IPBES
20 2018), both indicators that remote sensing or tory methods can measure more easily than reductions in
21 productive capacity. However, the causes of reductions in canopy cover or carbon stocks can be many,
22 including natural disturbances (fires and insects), direct human activities (harvest, forest management)
23 and indirect human impacts (such as climate change) and these may not reduce long-term forest
24 productivity. In many boreal, and some temperate, and other forest types natural disturbances are
25 common, and consequently these disturbance-adapted forest types are comprised of a mosaic of stands
26 of different ages and stages of stand recovery following natural disturbances.

27 Defining forest degradation as a reduction in productivity, carbon stocks or canopy cover also requires
28 that a baseline is established against which this reduction is assessed. In forest types with rare stand-
29 replacing disturbances, the concept of “intact” or “primary” forest has been used to define a baseline
30 (Potapov et al. 2008; Bernier et al. 2017). Forest types with frequent stand-replacing disturbances such
31 as wildfires or with natural disturbances that reduce carbon stocks such as some insect outbreaks,
32 experience over time a natural range in variability of carbon stocks or canopy density making it more
33 difficult to define the appropriate natural carbon density or canopy cover against which to assess
34 degradation. In these systems, forest degradation cannot be defined at the stand level, but requires a
35 landscape-level assessment that takes into consideration the stand age-class distribution of the
36 landscape, which reflects disturbance regimes over past decades and also considers post-disturbance
37 regrowth (Wagner 1978; Volkova et al. 2018).

38



1
2 **Figure 4.1** Conceptual figure illustrating how land use moves land from a natural to a used state. How the
3 land is managed in response to climate change determines sustainable or degraded outcome. Climate
4 change can exacerbate many degradation processes (Table 4.1) and introduce novel ones (e.g., permafrost
5 thawing or biome shifts), hence management needs to respond to climate impacts in order to avoid,
6 reduce or reverse degradation. The types and intensity of human land use and climate change impacts on
7 natural lands affect their carbon stocks and their ability to operate as carbon sinks. In managed
8 agricultural lands, degradation typically results in reductions of soil organic carbon stocks, which also
9 adversely affects land productivity and carbon sinks. In forest land, reduction in biomass carbon stocks
10 alone is not necessarily an indication of a reduction in carbon sinks. Sustainably managed forest
11 landscapes can have a lower biomass carbon density but the younger forests have a higher growth rate,
12 and therefore contribute stronger carbon sinks, than natural forests

13 Unsustainable logging practices and stand-level degradation can occur in all forest types, for example
14 when selective logging (high-grading) removes valuable large-diameter trees, leaving behind damaged,
15 diseased or otherwise less productive trees and conditions that reduce not only carbon stocks but also
16 adversely affect subsequent forest recovery (Belair and Ducey 2018; Nyland 1992). However,
17 sustainable selective logging such as thinning can maintain and enhance forest productivity.

18 The term forest degradation is typically used to describe activities with undesirable outcomes, including
19 losses in productive capacity, losses in biodiversity, losses in the ability to provide goods and services,
20 and other losses (Barlow et al. 2007). However, sustainable forest management applied at the landscape
21 scale can reduce average forest carbon stocks, while increasing the rate at which carbon dioxide is

1 removed from the atmosphere, because Net Ecosystem Production of forest stands is highest in
2 intermediate stand ages (Kurz et al. 2013; Volkova et al. 2018; Tang et al. 2014). Thus, the impacts of
3 sustainable forest management on one indicator (C stocks in the forested landscape) can be negative,
4 while those on another indicator (forest productivity and rate of C removal from the atmosphere) can
5 simultaneously be positive. Moreover, increases in forest productivity can be associated with reductions
6 in biodiversity, as increased productivity can be achieved by periodic thinning and removal of trees that
7 would otherwise die due to competition, and the dead organic matter of snags and coarse woody debris
8 can provide habitat that contributes to biodiversity (Spence 2001; Ehnström 2001). However, while the
9 sustainability for forest management has been demonstrated in temperate and boreal forests, questions
10 remain about the ability to implement sustainable forest harvesting in complex tropical forest
11 ecosystems.

12 Instead of seeking to quantify the rates of forest degradation based on vague definitions and weakly
13 defined baselines, scientific and policy communities would be better supported by information on
14 changes in specific forest characteristics, which together can identify forest degradation, as this would
15 allow for the assessment of the trade-offs among the various forest characteristics. For example, carbon
16 stocks per hectare, net ecosystem productivity, net biome productivity, and albedo are indicators that
17 can be quantified and reported. In contrast, assessing biodiversity changes is much more complicated
18 and requires a clear definition of the biodiversity targets. Improved understanding of past trends and
19 projections of these indicators will enhance the ability to design and implement land management
20 strategies aimed at achieving desired outcomes, including sustainable forest management and activities
21 aimed at reducing atmospheric GHG concentrations as outlined in the Paris Agreement. As long as any
22 form of human impacts on forests is considered degradation (IPBES 2018) and thus undesirable, the
23 opportunities will remain limited to identify and implement sustainable land-use and land-management
24 strategies that allow for the co-existence of forest ecosystems and humans with their requirements for
25 food, fibre, timber and shelter.

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

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

36 The stewardship and use of forests and forest lands in a way, and at a rate, that maintains their
37 biodiversity, productivity, regeneration capacity, vitality and their potential to fulfill, now and in the
38 future, relevant ecological, economic and social functions, at local, national, and global levels, and that
39 does not cause damage to other ecosystems (Forest Europe 2016; Mackey et al. 2015).

40 Other terms pertinent to this chapter are:

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

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

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

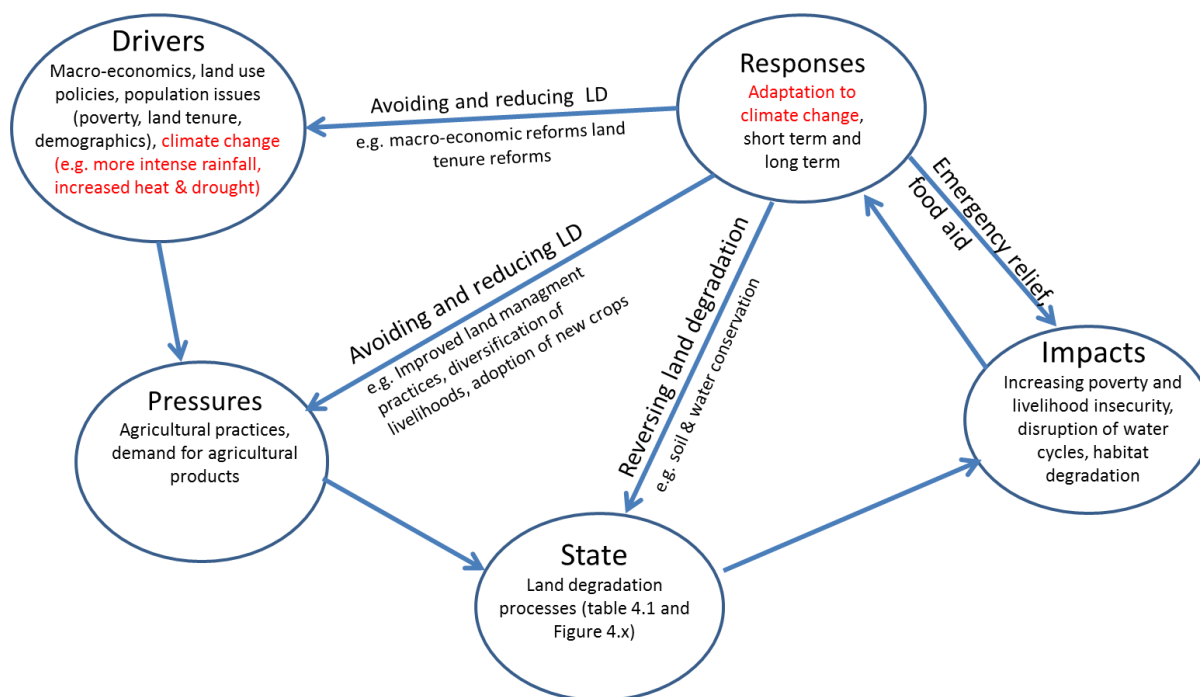
4 **4.3.3 The human dimension of land and forest degradation**

5 Studies of land and forest degradation are often biased towards biophysical aspects both in terms of its
6 processes, such as erosion or nutrient depletion, and its observed physical manifestations, such as
7 gullyng or low primary productivity. Land users' own perceptions and knowledge about land
8 conditions and land degradation have often been neglected or ignored (Reed et al. 2007; Forsyth 1996;
9 Andersson et al. 2011). The omission of such perspectives has led to policies which are characterised
10 by scientism, i.e. a narrow focus on natural science (Warren and Olsson 2003) and sometimes neo-
11 Malthusian perspectives, i.e. a narrow focus on population growth as the problem (Stringer 2009;
12 Stringer and Reed 2007). A growing body of work is nevertheless beginning to focus on land
13 degradation through the lens of local land users (Kessler and Stroosnijder 2006; Fairhead and Scoones
14 2005; Zimmerer 1993; Stocking et al. 2001) and the importance of local and indigenous knowledge
15 within land management decision making is starting to be better appreciated (IPBES 2018). In this
16 report we treat both land degradation and people's responses to it as a relational problem in which land
17 users are interacting with the local ecosystem and climate, while embedded in a multi-scalar social
18 reality. Climate change impacts directly and indirectly the social reality, the land users, and the
19 ecosystem and vice versa. In some cases, land degradation can also have an impact on climate change
20 (see Section 4.8).

21 Important aspects of these relationships will be highlighted throughout the chapter. For example,
22 women have often less formal access to land than men and hence less influence over decisions about
23 land, even if they carry out many of the land management tasks (Jerneck 2018a; Elmhirst 2011; Toulmin
24 2009; Peters 2004; Agarwal 1997; Jerneck 2018b). Many oft-cited sweeping statements about women's
25 subordination in agriculture are difficult to substantiate, yet it is clear that the gender gap is very large
26 (Doss et al. 2015). Even if women's access to land is changing formally (Kumar and Quisumbing 2015),
27 the practical outcome is often limited due to several other factors (Lavers 2017; Kristjanson et al. 2017;
28 Djurfeldt et al. 2018). The use and management of land is therefore still highly gendered and is expected
29 to remain so for the foreseeable future (Kristjanson et al. 2017). Women are also affected differently
30 than men when it comes to climate change, having lower adaptive capacities due to factors such as
31 prevailing land tenure frameworks, lower access to other capital assets and dominant cultural practices
32 (Antwi-Agyei et al. 2015; Gabrielsson et al. 2013). This affects the options available to women to
33 respond to both land degradation and climate change. Indeed, access to land and other assets (e.g.,
34 education and training) is key in shaping land use and land management strategies (Liu et al. 2018a;
35 Lambin et al. 2001). Land rights are highly context specific and dependent upon the political-economic
36 and legal context (IPBES 2018). This means there is no universally applicable best arrangement.
37 Agriculture in highly erosion prone regions require site specific investments which may benefit from
38 secure private land rights (Tarfasa et al. 2018). Pastoral modes of production and indigenous forest
39 management systems are often dominated by communal land tenure arrangements which often conflict
40 with agricultural/forestry modernisation policies implying private property rights (Antwi-Agyei et al.
41 2015; Benjaminsen and Lund 2003; Itkonen 2016; Owour et al. 2011; Gebara 2018).

43 **4.4 Land degradation in the context of climate change**

44 Several conceptual frameworks have been used in previous scientific assessments to describe land
45 degradation processes and their causality. This chapter borrows from the driver-pressure- state impact-
46 -response (DPSIR) framework (Tomich et al. 2010; Gisladottir and Stocking 2005), see Figure 4.2.



1
2 **Figure 4.2 Schematic illustration of land degradation using the Drivers, Pressures, State, Impacts,**
3 **Responses (DPSIR) framework. Climate change aspects are highlighted in red**

4 Establishing clear boundaries between drivers and pressures can often be difficult. In the DPSIR
5 framework, drivers can be both natural (e.g. climate change and variation) and human forces (e.g.
6 population growth, economic shocks, technological change). Pressures are human activities, such as
7 dominant land management practices, affecting the environment, resulting from drivers. Processes are
8 the natural phenomena that link pressures to state, as in the case if overgrazing leads to erosion which
9 reduces soil fertility.

10 In this chapter we use the terms processes and drivers with the following meanings:

11 **Processes of land degradation** are those direct mechanisms by which land is degraded and are similar
12 to the notion of “direct drivers” in the Millennium Ecosystem Assessment (MA) framework and “state”
13 in the DPSIR framework (Tomich et al. 2010). In this report, a comprehensive list of land degradation
14 processes is presented in Table 4.1.

15 **Drivers of land degradation** are those indirect conditions which may drive processes of land
16 degradation and are similar to the notion of “indirect drivers” in the MA framework and both “drivers”
17 and “pressures” in the DPSIR framework in Figure 4.2 (Tomich et al. 2010).

18 An exact demarcation between processes and drivers is impossible to make. Drought and fires are
19 described as drivers of land degradation in the next section but they can also be a process: for example,
20 if repeated fires deplete seed sources they can affect regeneration and succession of forest ecosystems.

21 The relationships between drivers and processes are illustrated by Figure 4.2. The responses to land
22 degradation follow the logic of the Land Degradation Neutrality concept: avoiding, reducing and
23 reversing land degradation.

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

1 **4.4.1 Processes of land degradation**

2 A large array of interactive physical, chemical, biological and human processes lead to what we define
3 in this report as land degradation (Johnson and Lewis 2007). The biological productivity, ecological
4 integrity or the human value of a given territory can be deteriorated as the result of processes triggered
5 at scales that range from a single furrow (e.g., water erosion under cultivation) to the landscape level
6 region (e.g., salinisation through raising groundwater levels under irrigation). While pressures leading
7 to land degradation are often exerted on specific components of the land systems (i.e., soils, water,
8 biota), once degradation processes start, other components become affected through cascading and
9 interactive effects. For example, different pressures and degradation processes can have convergent
10 effects, as it can be the case of overgrazing leading to wind erosion, landscape drainage resulting in
11 wetland drying, and warming causing more frequent burning; all of which can independently lead to
12 reductions of the soil organic matter pools as second order process. Still, the reduction of organic matter
13 pools is also a first order process triggered directly by the effects of rising temperatures (Bellamy et al.
14 2005). Beyond this complexity, a practical assessment of the major land degradation processes helps to
15 reveal and categorize the multiple “entry points” in which climate change exerts a degradation pressure
16 (Table 4.1).

17 **4.4.1.1 Types of land degradation processes**

18 Land degradation processes can have their origin or major “entry points” in the soil, water or biotic
19 components of the land or in their respective interfaces (Table 4.1). Across land degradation processes,
20 those having their “entry point” in the soil have received more attention. The most widespread and
21 studied soil degradation processes are water and wind erosion, which have accompanied cultivation
22 since its onset and are still dominant (Table 4.1). Degradation through erosion processes is not restricted
23 to soil loss in the eroded areas but can also include impacts on transport and deposition areas as well
24 (less commonly, deposition areas can have their soils improved by these inputs). Larger scale
25 degradation processes related to the whole continuum of soil erosion, transport and deposition include
26 dune field expansion/displacement, development of gully networks and siltation of natural and artificial
27 water bodies (Poesen and Hooke 1997; Ravi et al. 2010). Coastal erosion represents a special case
28 among erosional processes with particularly strong links to climate change. While human interventions
29 in the land-ocean interphase (e.g., expansion of shrimp farms) and rivers (e.g., upstream dams cutting
30 coastal sediment supply) are dominant global drivers, storms and land subsidence (a useful proxy of sea
31 level rise) have already left a significant global imprint on coastal erosion (Mentaschi et al. 2018).
32 Recent projections that take into account geomorphological and socioecological feedbacks suggest that
33 coastal wetlands may not get reduced by sea level rise if their inland growth is accommodated with
34 proper management actions (Schuerch et al. 2018).

35 Other physical degradation process in which no material detachment and transport are involved include
36 soil compaction, hardening, sealing and any other mechanism leading to the loss of pore volume. A
37 very extreme case of degradation through pore volume loss, manifested at landscape or larger scales, is
38 ground subsidence. Typically caused by the depletion of groundwater or oil reserves, subsidence
39 involves a sustained collapse of the ground surface, which can lead to other degradation processes such
40 as salinization and permanent flooding (see Section 4.2.2). Chemical soil degradation processes range
41 from nutrient depletion, resulting from the imbalance of nutrient extraction on harvested products and
42 fertilization, to more complex processes of acidification and increasing metal toxicity. Acidification in
43 croplands is increasingly driven by excessive N fertilisation and to a lower extent by cation depletion
44 through harvesting exports (Guo et al. 2010). One of the most relevant chemical degradation processes
45 of soils in the context of climate change is the depletion of its organic matter pool. Favoured in
46 agricultural soils through the increase of respiration rates by tillage and the reduction of belowground
47 plant biomass inputs, soil organic matter pools have been reduced also by the direct effects of warming
48 (Bellamy et al. 2005; Bond-Lamberty et al. 2018). Affected by many other degradation process and

1 having in turn cascading negative effects on other pathways of soil degradation, soil organic matter can
2 be considered a “hub” of degradation processes and also a critical link with the climate system on which
3 mitigation actions can be focused (Minasny et al. 2017).

4 Not all land degradation processes start in the soil and those starting from alterations in the hydrological
5 system are particularly important in the context of climate change. Salinisation, although perceived and
6 reported in soils, is typically triggered by water table level rises driving salts to the surface under dry to
7 sub-humid climates (Schofield and Kirkby 2003). While salty soils and ecosystems occur naturally
8 under these climates (primary salinity), human interventions have expanded their distribution
9 (secondary salinity). Irrigation without proper drainage has been the predominant cause of salinisation.
10 Yet, it has also taken place under non-irrigated conditions where vegetation changes (particularly dry
11 forest clearing and cultivation) had reduced the magnitude and depth of soil water uptake, triggering
12 water table rises towards the surface. Changes in evapotranspiration and rainfall regimes can exacerbate
13 this process (Schofield and Kirkby 2003). Recurring flood and waterlogging episodes (Bradshaw et al.
14 2007; Poff 2002), and the more chronic expansion of wetlands over dryland ecosystems (e.g.,
15 paludification) are mediated by the hydrological system, on occasions aided by geomorphological shifts
16 as well (Kirwan et al. 2011). This is also the case for the drying of continental water bodies and
17 wetlands, including the salinisation and drying of lakes and inland seas (Anderson et al. 2003; Micklin
18 2010; Herbert et al. 2015).

19 The biotic components of the land can also be the “entry point” of degradation processes. Vegetation
20 clearing processes associated with land use changes are not limited to deforestation but include other
21 natural and seminatural ecosystems such as grasslands (the most cultivated biome on Earth), as well as
22 dry steppes and shrublands, which give place to croplands, pastures, urbanisation or just barren land.
23 This clearing processes is associated with net C losses from the vegetation and soil pool. Not all biotic
24 degradation processes involve biomass losses. Woody encroachment and the "thicketization" of open
25 savannahs involve the expansion of woody plant cover and/or density over herbaceous areas and often
26 limits the secondary productivity of rangelands (Asner et al. 2004, Anadon et al. 2014). These processes
27 have been accelerated since the mid-1800s over most continents (Van Auken 2009). Change in plant
28 composition of natural or semi-natural ecosystems without significant vegetation structural changes is
29 another pathway of degradation affecting rangelands and forests. In rangelands, selective grazing and
30 its interaction with climate variability and/or fire can push ecosystems to new stable compositions with
31 lower forage value (Illius and O’Connor 1999, Sasaki et al. 2007) but with higher carbon sequestration
32 potential. In forests, selective logging is a pervasive cause of degradation which can lead to long-term
33 impoverishment and in extreme cases, a full loss of the forest cover through its interaction with other
34 agents such as fires (Foley et al. 2007) or progressive intensification of land use. Invasive alien species
35 are another source of biological degradation. Their arrival into cultivated systems is constantly
36 reshaping crop production strategies making agriculture unviable on occasions. In natural and
37 seminatural systems such as rangelands, invasive plant species not only threaten livestock production
38 through diminished forage quality, poisoning and other deleterious effects, but have cascading effects
39 on other processes such as altered fire regimes and water cycling (Brooks et al. 2004).

40 Other biotic components of ecosystems have been shown to act as “entry points” of degradation
41 processes. Invertebrate invasions in continental waters can exacerbate other degradations processes
42 such as eutrophication (Walsh et al. 2016). Shifts in soil microbial and mesofaunal composition, which
43 can be caused by pollution with pesticides or nitrogen deposition, alter many soil functions including
44 respiration rates and C release to the atmosphere (Hussain et al. 2009; Crowther et al. 2015). In natural
45 dry ecosystems, biological soil crusts composed by a broad range of organisms including mosses are a
46 particularly sensitive “entry point” for degradation (Field et al. 2010) with evidenced sensitivity to
47 climate change (Reed et al. 2012).

1 **4.4.1.2. Land degradation processes and climate change**

2 While the subdivision of individual processes is challenged by their strong interconnectedness, it
3 provides a useful setting to identify the most important “entry points” of climate change pressures on
4 land degradation. Among land degradation processes those responding more directly to climate change
5 pressures include all types of erosion and soil organic matter declines (soil entry point), salinization,
6 sodification and permafrost thawing (soil/water entry point), waterlogging of dry ecosystems and drying
7 of wet ecosystems (water entry point), and a broad group of biological mediated processes like woody
8 encroachment, biological invasions, pest outbreaks (biotic entry point), together with biological soil
9 crust destruction and increased burning (soil/biota entry point) (Table 4.1). Processes like ground
10 subsidence or deforestation can be affected by climate change only indirectly.

11 Even when climate change exerts a direct pressure on degradation processes, it can be a secondary
12 driver subordinated to other overwhelming human pressures. The most important exceptions are three
13 processes in which climate change is a dominant pressure and the main driver of their current
14 acceleration. These are coastal erosion as affected by sea level rise and increased storm
15 frequency/intensity (*high confidence*), permafrost thawing responding to warming (*very high*
16 *confidence*) and increased burning responding to warming and altered precipitation regimes (*high*
17 *confidence*). The previous assessment highlights the fact that climate change not only exacerbates many
18 of the well acknowledged ongoing land degradation process of managed ecosystem (i.e., croplands and
19 pastures), but becomes a dominant pressure that introduces novel degradation pathways in natural and
20 seminatural ecosystems.

1 **Table 4.1 Major land degradation processes and their connections with climate change. For each process an “entry point” (soil, water, biota) on which degradation**
 2 **pressures actuate on first place is indicated, acknowledging that most processes propagate to other land components and cascade into or interact with some of the**
 3 **other processes listed below. The impact of climate change on each process is categorized based on the proximity (very direct = high, very indirect=low) and**
 4 **dominance (dominant=high, subordinate to other pressures = low) of effects. The major intervening pressures of climate change on each process are highlighted**
 5 **together with the predominant pressures from other drivers. Feedbacks of land degradation processes on climate change are categorized according to the intensity**
 6 **(very intense=high, subtle=low) of the chemical (greenhouse gases emissions or capture) or physical (energy and momentum exchange, aerosol emissions) effects.**
 7 **Specific consequences on climate change are highlighted**

Processes	Entry point	Impacts of Climate Change				Feedbacks on Climate Change			
		proximity	dominance	Climate Change pressures	Other pressures	intensity of chemical effects	intensity of physical effects	global extent	Climate Change consequence
Wind erosion	Soil	high	medium	Altered wind/drought patterns (<i>high confidence</i> on effect, <i>medium-low confidence</i> on trend). Indirect effect through vegetation type and biomass production shifts	Tillage, Cultivation leaving low cover, overgrazing, deforestation/vegetation clearing, large plot sizes, vegetation and fire regime shifts	low	medium (cooling)	high	Radiative cooling by dust release (<i>high confidence</i>). Enhanced weathering leading to ocean and land fertilization and C burial (medium confidence). Albedo increase.
Water erosion	Soil	high	medium	Increasing rainfall intensity (<i>high confidence</i> on effect and trend). Indirect effects of climate change on fire frequency/intensity, permafrost melting, biomass production are likely to be more important.	Tillage, cultivation leaving low cover, overgrazing, deforestation/vegetation clearing, vegetation burning, poorly designed roads and paths.	medium	medium (cooling)	high	Net C release. Likely net release is less than site-specific loss due to redeposition and burial (<i>high confidence</i>). Albedo increase.
Coastal erosion	Soil/Water	high	high	Sea level raise, increasing intensity/frequency of storm surges (<i>high confidence</i> on effects and trends)	Retention of sediments by upstream dams, Coastal aquaculture, Elimination of mangrove forests, Subsidence	high	low	low	Release of old buried C pools (<i>medium confidence</i>)

Subsidence	Soil/Water	low	low	Indirect through increasing drought leading to higher ground water use. Indirect through enhanced decomposition (e.g. through drainage) in organic soils.	Groundwater depletion / overpumping. Peatland drainage.	low/high	low	low	Unimportant in the case of groundwater depletion. Very high net C release in the case of drained peatlands
Compaction/Hardening	Soil	low	low	Indirect through reduced organic matter content.	Land use conversion, machinery overuse, intensive grazing, poor tillage/grazing management (e.g. under wet or waterlogged conditions)	low	low	medium	Ambiguous effects of reduced aeration on N ₂ O emissions (<i>low confidence</i>)
Nutrient depletion	Soil	low	low	Indirect (e.g. shifts in cropland distribution, BECCS)	Insufficient replenishment of harvested nutrients	low	low	medium	Net C release due shrinking SOC pools (medium confidence)
Acidification/Overfertilisation	Soil	low	low	Indirect (e.g. shifts in cropland distribution, BECCS)	High N fertilisation. High cation depletion. Acid rain/deposition	medium	low	medium	N ₂ O release from overfertilised soils, increased by acidification. Inorganic C release from acidifying soils (<i>medium to high confidence</i>)
Pollution	Soil/Biota	low	low	Indirect (e.g. increased pest and weed incidence)	Intensifying chemical control of weed and pests	low	low	medium	Unknown, likely unimportant.
Organic matter decline	Soil	high	medium	Warming accelerates soil respiration rates (<i>high confidence</i> on effects and trends). Indirect effects through changing quality of plant litter or fire/waterlogging regimes.	Tillage. reduced plant input to soil. Drainage of waterlogged soils. Influenced by most of the other soil degradation processes.	high	low	high	Net C release (<i>high confidence</i>).
Metal toxicity	Soil	low	low	Indirect	High cation depletion, fertilisation, mining activities	low	low	low	unknown, likely unimportant.
Salinisation	Soil / Water	high	low	Sea level rise (<i>high confidence</i> on effects and trends). Water balance shifts (<i>medium confidence</i> on effects and trends). Indirect effects through irrigation expansion.	Irrigation without good drainage infrastructure. Deforestation and water table level raises under dryland agriculture	low	medium (cooling)	medium	Reduced methane emissions with high sulfate load. Albedo increase.

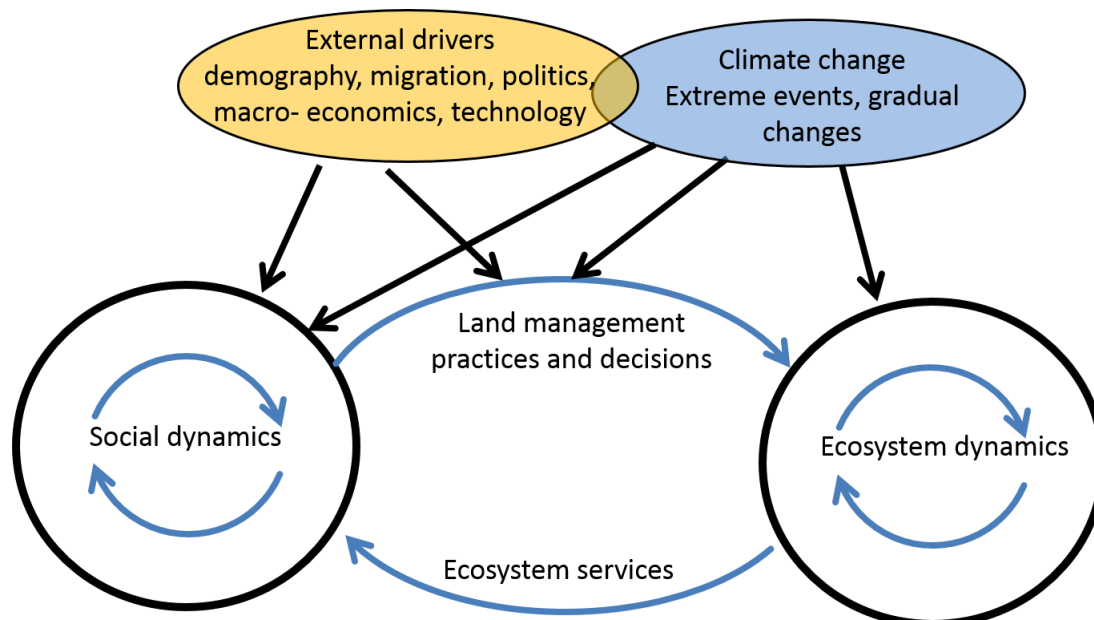
Sodification	Soil / Water	high	low	Water balance shifts (<i>medium confidence</i> on effects and trends). Indirect effects through irrigation expansion.	Poor water management	low	medium (cooling)	low	Net C release due to soil structure and organic matter dispersion. Albedo increase.
Permafrost melting	Soil / Water	high	high	Warming (<i>very high confidence</i> on effects and trends)		high	low	high	Net C release. CH ₄ release
Waterlogging of dry systems	Water	high	medium	Water balance shifts (<i>medium confidence</i> on effects and trends). Indirect effects through vegetation shifts.	Deforestation. Irrigation without good drainage infrastructure	medium	medium	low	CH ₄ release. Albedo decrease
Drying of continental waters/wetland/lowl and systems	Water	high	medium	Increasing extent and duration of drought (<i>high confidence</i> on effects, <i>medium confidence</i> on trends). Indirect effects through vegetation shifts.	Upstream surface and groundwater water consumption. Intentional drainage. Trampling/overgrazing.	medium	medium (warming)	medium	Net C release. N ₂ O release. Albedo increase
Flooding	Water	high	medium	Sea level raise (<i>high confidence</i> on effects and trends), increasing intensity/frequency of storm surges (<i>high confidence</i> on effects and trends), increasing rainfall intensity causing flash floods (<i>high confidence</i> on effects and trends)	Land clearing. Increasing impervious surface. Transport infrastructure.	medium	medium (cooling)	low	CH ₄ and N ₂ O release. Albedo decrease
Eutrophication of continental waters	Water/Biota	low	low	Indirect through warming effects on N losses from the land or climate change effects on erosion rates. Interactive effects of warming and nutrient loads on algal blooms.	Excess fertilization. Erosion. Poor management of livestock/human sewage.	medium	low	low	CH ₄ and N ₂ O release.
Woody encroachment	Biota	high	medium	Rainfall shifts (<i>medium confidence</i> on effects and trends), CO ₂ rise (<i>medium confidence</i> on effects, <i>very high confidence</i> on trends)	Overgrazing. Altered fire regimes, fire suppression. Invasive alien species.	high (cooling)	high (warming)	high	Net C storage. Albedo decrease

Deforestation / Land Clearing	Biota	low	low	Indirect through BECCS and their geographical leakage effects. Indirect through migration of agriculture into new areas.	Expansion of crops and pastures. Overexploitation of forests.	high	medium (cooling)	high	Net C release. Albedo increase
Species loss, compositional shifts	Biota	high	medium	Habitat loss as a result of climate shifts (<i>medium confidence</i> on effects and trends)	Selective grazing and logging causing plant species loss, Pesticides causing soil microbial and soil faunal losses, Large animal extinctions, Interruption of disturbance regimes	low	low	medium	Ambiguous.
Soil microbial and mesofaunal losses	Biota	high	low	Habitat loss as a result of climate shifts (<i>medium confidence</i> on effects and trends)	Altered fire regimes, nitrogen deposition, pesticide pollution	low	low	medium	Ambiguous.
Biological soil crust destruction	Biota/Soil	high	medium	Warming. Changing rainfall regimes. Indirect through fire regime shifts and/or invasions.	Overgrazing and trampling. Land use conversion.	low	high (cooling)	high	Radiative cooling through albedo rise and dust release (<i>high confidence</i>).
Invasions	Biota	high	medium	Habitat gain as a result of climate shifts (<i>medium confidence</i> on effects and trends)	Intentional and unintentional species introductions.	low	low	medium	Ambiguous.
Pest outbreaks	Biota	high	medium	Habitat gain and accelerated reproduction as a result of climate shifts (<i>medium confidence</i> on effects and trends)	Large scale monocultures. Poor pest management practices.	medium	low	medium	Net C release.
Increased burning	Soil/Biota	high	high	Warming, drought, shifting precipitation regimes (<i>high confidence</i> on effects and trends)	Fire suppression policies increasing wildfire intensity. Increasing use of fire for rangeland management. Agriculture introducing fires in humid climates without previous fire history. Invasions.	high	medium (cooling)	medium	Net C release. CO, CH ₄ , N ₂ O release. Albedo increase. Long term decline of NPP in grasslands, savannas and broadleaf forests

1

1 4.4.2 Drivers of land degradation

2 Drivers of land degradation and land improvement are many and they interact in multiple ways. Figure
 3 4.3 illustrates how some of the most important drivers are interacting with the land users. It is important
 4 to keep in mind that both natural and human drivers can drive both degradation and improvement (Kiage
 5 2013).



6
 7 **Figure 4.3 Drivers of land degradation and interactions between social and ecosystem dynamics (this is**
 8 **still a first draft)**

9 Land degradation is sometimes considered to be a creeping phenomenon, controlled by slow variables
 10 (Walker et al. 2012; Reynolds et al. 2011). Examples of such slow variables are depletion of nutrients
 11 or a gradual reduction of ecosystem services such as water holding capacity. But it is important to realise
 12 that land degradation is driven by the entire spectrum of factors, from very short and intensive events
 13 such as an individual rain storm of 10 minutes (Coppus and Imeson 2002; Morgan 2005) to century
 14 scale slow depletion of nutrients or loss of soil particles (Johnson and Lewis 2007, p. 5-6). But instead
 15 of focusing on absolute temporal variations, the drivers of land degradation should more appropriately
 16 be assessed in relation to the rates of possible recovery. Studies suggest for example, that erosion rates
 17 of conventionally tilled agricultural fields exceed the rate at which soil is generated by one to two orders
 18 of magnitude (Montgomery 2007a). The rate of soil formation varies across landscapes between about
 19 $0.3 \text{ t ha}^{-1} \text{ yr}^{-1}$ to $1.4 \text{ t ha}^{-1} \text{ yr}^{-1}$ (Verheijen et al. 2009). Guidelines in the USA for tolerable rate of erosion
 20 is typically $11 \text{ t ha}^{-1} \text{ yr}^{-1}$, which exceeds drastically soil formation rate. In Europe there is no universally
 21 accepted level, but it has been recommended that tolerable erosion rate should be about equal to soil
 22 formation rate, i.e. about $1 \text{ t ha}^{-1} \text{ yr}^{-1}$ (Verheijen et al. 2009).

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

27 The climate change related drivers of land degradation are both gradual changes of temperature and
 28 precipitation, and changes of the distribution and intensity of extreme events. Importantly, these drivers
 29 can act in two directions: land improvement and land degradation.

30 The gradual and planetary changes that can cause land degradation/improvement have been studied by
 31 global integrated models and Earth observation technologies. Studies of global land suitability for

1 agriculture suggest that climate change will increase the area suitable for agriculture in the Northern
2 high latitudes by 16% (Ramankutty et al. 2002) or 5.6 million km² (Zabel et al. 2014), while tropical
3 regions will experience a loss (Ramankutty et al. 2002; Zabel et al. 2014).

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

8 Temporal and spatial patterns of tree mortality can be used as an indicator of climate change impacts
9 on terrestrial ecosystems. Episodic mortality of trees occur naturally even without climate change, but
10 more widespread spatio-temporal anomalies can be sign of climate induced degradation (Allen et al.
11 2010). In the absence of systematic data on tree mortality, a comprehensive meta-analysis of 150
12 published articles suggests that increasing tree mortality around the world can be attributed to increasing
13 drought and heat stress in forests worldwide (Allen et al. 2010).

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

20 Within the tropics, much research has been devoted to understanding how climate change may alter
21 regional suitability of various crops. For example coffee is expected to be highly sensitive to both
22 temperature and precipitation changes, both in terms of growth and yield and in terms of increasing
23 problems of pests (Ovalle-Rivera et al. 2015). Some studies paint a very bleak picture in which the
24 global area of coffee production will decrease by 50% (Bunn et al. 2015). Due to increased heat stress,
25 the suitability of Arabica coffee is expected to deteriorate in Mesoamerica while it can improve in high
26 altitude areas in South America. The general pattern is that the climatic suitability for Arabica coffee
27 will deteriorate at low altitudes of the tropics as well as at the higher latitudes (Ovalle-Rivera et al.
28 2015). This means that climate change in and of itself can render previously sustainable land use and
29 land management practices unsustainable and vice versa (Laderach et al. 2011).

30 Other and more indirect drivers can be a wide range of factors such as demographic changes,
31 technological change, changes of consumption patterns and dietary preferences, political and economic
32 changes, and social changes (Mirzabaev et al. 2016). It is important to stress that there are no simple or
33 direct relationships between underlying drivers and land degradation, such as poverty or high population
34 density, that are necessarily causing land degradation (Lambin et al. 2001). However, drivers of land
35 degradation need to be studied in the context of spatial, temporal, economic, environmental and cultural
36 aspects (Warren 2002). Some analyses suggest an overall negative correlation between population
37 density and land degradation (Bai et al. 2008) but we find many local examples of both positive and
38 negative relationships (Brandt et al. 2018a, 2017). Even if there are correlations in one or the other
39 direction, causality is not always the same.

40 Land degradation can also be affected indirectly by climate change through changing patterns of
41 wildlife habitats and wildlife densities (Ims and Fuglei 2009; Aryal et al. 2014; Beschta et al. 2013).

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

46

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. Land degradation is a complex phenomenon affected by several independent factors: climatic (such as rainfall, temperature, and wind), abiotic ecological factors (such as soil characteristics and topography), type of land use (such as farming of various kinds, forestry, or protected area), and land management practices (such as tilling, crop rotation, and logging/thinning). Therefore, attribution of land degradation to climate change is extremely challenging. Because land degradation is highly dependent on land management, it is even possible that climate impacts would trigger land management changes reducing or reversing land degradation, sometimes called transformational adaptation (Kates et al. 2012). There is not much research on attributing land degradation explicitly to climate change, there is more on climate change as a threat multiplier for land degradation. Instead, we may be able to infer climate change impacts on land degradation both theoretically and empirically. 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 4.4.3.2 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. From an attribution point of view, it is important to note that projections of precipitation are in general more uncertain than projections of temperature changes (Murphy et al. 2004; Fischer and Knutti 2015; IPCC 2013a). Precipitation involves local processes of larger complexity than temperature and projections are usually less robust than those for temperature (Giorgi and Lionello 2008).

Theoretically the intensification of the hydrological cycle as a result of human induced climate change is well established (Guerreiro et al. 2018; Trenberth 1999) and also empirically observed (Blenkinsop et al. 2018; Burt et al. 2016; IPCC WGI AR5 2013; Liu et al. 2009). AR5 WGI concluded that heavy precipitation events have increased in frequency, intensity, and/or amount since 1950 (*Likely*) and that further changes in this direction are *likely to very likely* during the 21st century (IPCC 2013, p. 7). As an example, in central India, there has been a threefold increase in widespread extreme rain events during 1950–2015 which has influenced several land degradation processes, not least soil erosion (Burt et al. 2016). It is also expected that seasonal shifts and cycles such as monsoons and ENSO will further increase the intensity of rainfall events (IPCC 2013, p. 23).

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 climatic driver 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). Such changes result in increases in the intensity of rainfall which 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. 2004a). However, the effects will not be uniform but highly variable across regions (Almagro et al. 2017; Mondal et al. 2016).

1 The most comprehensive database of direct measurements of water erosion to our knowledge contains
2 4377 entries (the majority from North America and Europe), even though not all entries are complete.
3 An important finding from that database is that almost any erosion rate is possible under almost any
4 climatic condition (García-Ruiz et al. 2015). Even if the results show few clear relationships between
5 erosion and land conditions, the authors highlight four observations: 1/ the highest erosion rates were
6 found in relation to agricultural activities – even though moderate erosion rates were also found in
7 agricultural settings, 2/ high erosion rates after forest fires were not observed (although the cases were
8 few), 3/ land covered by shrubs showed generally low erosion rates, 4/ pasture land showed generally
9 medium rates of erosion. Some important findings for the link between soil erosion and climate change
10 can be noted from erosion measurements: erosion rates tend to increase with increasing mean annual
11 rainfall, with a peak in the interval of 1000 to 1400 mm annual rainfall (*low confidence*). However, such
12 relationships are overshadowed by the fact that most rainfall events do not cause any erosion, instead
13 erosion is caused by a few annual events. Hence mean annual rainfall is not a good predictor of erosion
14 (Gonzalez-Hidalgo et al. 2012, 2009). In the context of climate change, it means the tendency of rainfall
15 patterns to change towards more intensive precipitation events is serious. Such patterns have already
16 been observed widely, even in cases where the total rainfall is decreasing (Trenberth 2011). The findings
17 generally confirm the strong consensus about the importance of vegetation cover as a protection against
18 soil erosion, a finding emphasises how extremely important land management is for controlling erosion.

19 In the Mediterranean region, the observed and expected decrease in annual rainfall due to climate
20 change is accompanied by an increase of rainfall intensity and hence erosivity (Capolongo et al. 2008).
21 In tropical and sub-tropical regions, the on-site impacts of soil erosion dominate, and is manifested in
22 very high rates of soil loss, in some cases exceeding 100 tons ha⁻¹ yr⁻¹ (Tadesse 2001; García-Ruiz et
23 al. 2015). In temperate regions, the off-site effects of soil erosion are often a greater concern, for
24 example siltation of dams and ponds, downslope damage to property, roads and other infrastructure
25 (Boardman 2010).

26 The distribution over time of wet and dry spells is also expected to be affected although uncertainties
27 still remain depending on for example resolution of climate models used for prediction (Kendon et al.
28 2014). Changes in timing of rainfall events may have significant impacts on processes of soil erosion
29 through changes in wetting and drying of soils (Lado et al. 2004).

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

33 Wind erosion is a serious problem in agricultural regions, not only in drylands (Wagner 2013).
34 Theoretically (Bakun 1990) and empirically (Sydeman et al. 2014) winds along coastal regions
35 worldwide have increased with climate change. Other studies of wind and wind erosion have not
36 detected any long-term trend suggesting that climate change has altered wind patterns outside drylands
37 in a way that can significantly affect the risk of wind erosion (Pryor and Barthelmie 2010; Barring et
38 al. 2003).

39 Direct temperature effects on soils are of two kinds. Firstly, permafrost thawing leads to soil degradation
40 in boreal and high altitude regions (Yang et al. 2010; Jorgenson and Osterkamp 2005). Secondly,
41 warming alters the cycling of nitrogen (N) and carbon (C) in soils. There are many studies with
42 particularly strong experimental evidence, but a full understanding of cause and effect is contextual and
43 elusive (Conant et al. 2011a,b; Wu et al. 2011).

44 Climate change, including increasing atmospheric CO₂ levels, affects vegetation structure and function
45 and hence conditions for land degradation. Exactly how vegetation responds to changes remains a
46 research task. In a comparison of seven global vegetation models under four representative
47 concentration pathways Friend et al. (Friend et al. 2014) found that all models predicted increasing

1 vegetation carbon storage but with substantial variation between models. An important insight
2 compared with previous understanding is that structural dynamics of vegetation seems to play a more
3 important role than vegetation production (Friend et al. 2014). The magnitude of CO₂ fertilization of
4 vegetation growth, and hence conditions for land degradation is still uncertain (Holtum and Winter
5 2010), particularly in tropical rainforests (Yang et al. 2016). For more discussion on this topic, please
6 refer to Chapter 2 in this report.

7 In summary, rainfall changes attributed to human induced climate change have already intensified
8 drivers of land degradation (*high agreement, robust evidence*) but attributing land degradation to
9 climate change is challenging because the importance of land management (*high agreement, medium
10 evidence*). Other climate change impacts, such as changes in monsoons and ENSO events can affect
11 land degradation (*medium agreement, medium evidence*).

12 **4.4.3.2 Indirect and complex linkages with climate change**

13 Many important indirect linkages between land degradation and climate change occur via agriculture,
14 particularly through changing outbreaks of pests (Rosenzweig et al. 2001; Porter et al. 1991; Thomson
15 et al. 2010; Dhanush et al. 2015; Lamichhane et al. 2015), which is covered comprehensively in Chapter
16 5. More negative impacts have been observed than positive ones (IPCC 2014a). After 2050 the risk of
17 severe yield losses increase as a result of climate change in combination with other drivers (Porter et al.
18 2014). The reduction (or plateauing) in yields in major production areas (Brisson et al. 2010; Lin and
19 Huybers 2012; Grassini et al. 2013) may trigger intensification of land use elsewhere, either into natural
20 ecosystems, marginal arable lands or intensification on already cultivated lands, with possible
21 consequences for increasing land degradation.

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

25 Much research has tried to understand how plants are affected by a particular stressor, for example
26 drought, heat, or water logging. But less research has tried to understand how plants are affected by
27 several simultaneous stressors – which of course is more realistic in the context of climate change
28 (Mittler 2006). From an attribution point of view, such a complex web of causality is problematic if
29 attribution is only done through statistical significant correlation. It requires a combination of statistical
30 links and theoretically informed causation, preferably integrated into a model. Some modelling studies
31 have combined several stressors with geomorphologically explicit mechanisms (using the WEPP
32 model) and realistic land use scenarios, and found severe risks of increasing erosion from climate
33 change (Mullan et al. 2012; Mullan 2013). Other studies have included various management options,
34 such as changing planting and harvest dates (Zhang and Nearing 2005; Parajuli et al. 2016; Routschek
35 et al. 2014; Nunes and Nearing 2011), type of cultivars (Garbrecht and Zhang 2015), and price of crops
36 (Garbrecht et al. 2007; O’Neal et al. 2005) to investigate the complexity of how new climate regimes
37 may alter soil erosion rates.

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

43 **4.4.4 Approaches to assessing land degradation**

44 In a review of different approaches and attempts to map global land degradation, Gibbs and Salmon
45 (2015) identified four main approaches to map the global extent of degraded lands: expert opinions
46 (Oldeman and van Lynden 1998; Dregne 1998; Reed 2005; Bot et al. 2000), satellite observation of

1 vegetation greenness (e.g., remote sensing of NDVI, Normalised Difference Vegetation Index) (Yengoh
2 et al. 2015; Bai et al. 2008c), biophysical models (Cai et al. 2011b) and inventories of abandoned land.
3 There were large differences between the approaches except that they generally agreed about the extent
4 and location of non-degraded areas. Together they provide a relatively holistic evaluation but none on
5 its own assesses the complexity of the process (Vogt et al. 2011; Gibbs and Salmon 2015). There is,
6 however, a robust consensus that remote sensing and field-based methods are critical to assess and
7 monitor land degradation, particularly over large areas (such as global, continental and sub-continental)
8 even if there are still knowledge gaps to be filled (Wessels et al. 2007, 2004; Prince 2016; Ghazoul and
9 Chazdon 2017).

10 Remote sensing can provide meaningful proxies of land degradation in terms of severity, temporal
11 development, and areal extent. These proxies of land degradation include several indexes that have been
12 used to assess land conditions and monitoring the changes of land condition, for example extent of
13 gullies, severe forms of rill and sheet erosion, and deflation. The presence of open-access, quality
14 controlled and continuously updated global databases of remote sensing data is invaluable, and is the
15 only method for consistent monitoring of large areas over several decades (Sedano et al. 2016; Brandt
16 et al. 2018b; Turner 2014). The NDVI, as a proxy for Net Primary Production (NPP), is one of the most
17 commonly used method to assess land degradation. Even if NDVI is not a direct measure of vegetation
18 biomass, there is a close coupling between NDVI integrated over a season and in situ NPP (*high
19 agreement, robust evidence*) (see Higginbottom et al. 2014; Andela et al. 2013; Wessels et al. 2012).
20 The pre-processed (accounting corrections for solar zenith angle, volcanic aerosols, sensor
21 compatibility) long-term data sets available for this index is another important fact to fully assess the
22 status and trends of the land degradation.

23 Distinction between land degradation/improvement and the effects of climate variation is an important
24 and contentious issue (Murthy and Bagchi 2018; Ferner et al. 2018). There is no simple and
25 straightforward way to disentangle these two effects. The interaction of different determinants of
26 primary production is not well understood and a critical limitation to such disentangling is a lack of
27 understanding of the inherent inter-annual variability of vegetation (Huxman et al. 2004; Knapp and
28 Smith 2001; Ruppert et al. 2012; Bai et al. 2008a; Jobbágy and Sala 2000). One possibility is to compare
29 potential land productivity modelled by vegetation models and actual productivity measured by remote
30 sensing (Seaquist et al. 2009; Hickler et al. 2005), but the difference in spatial resolution, typically 0.5
31 degrees for vegetation models compared to 0.25–0.5 km for remote sensing data, is hampering the
32 approach.

33 Another approach to disentangle the effects of climate and land use/management is to use the Rain Use
34 Efficiency (RUE), defined as the biomass production per unit of rainfall, as an indicator (Le Houerou
35 1984; Prince et al. 1998; Fensholt et al. 2015). A variant of the RUE approach is the residual trend
36 (RESTREND) of a NDVI time-series, defined as the fraction of the difference between the observed
37 NDVI and the NDVI predicted from climate data (Yengoh et al. 2015; John et al. 2016). These two
38 metrics aim to measure the NPP, rainfall and the time dimensions. They are simple transforms of the
39 same three variables: RUE shows the NPP relationship with rainfall for individual years, while
40 RESTREND is the interannual change of RUE. They are legitimate metrics when used appropriately,
41 but in many cases they involve oversimplifications and yield misleading results (Fensholt et al. 2015;
42 Prince et al. 1998).

43 In theory, land degradation reduces the productive capacity of the land, i.e. the RUE declines. The ratio
44 of seasonally integrated NDVI over seasonal rainfall has often been used as a proxy of RUE, but the
45 scientific support is ambivalent. The original formulation of RUE (Le Houérou 1996; Le Houerou 1984;
46 Le Houerou and Hoste 1977) posited that ANPP was highly correlated with rainfall when the ANPP
47 was sampled from many sites along a rainfall gradient (a spatial approach). When RUE is used in
48 combination with remote sensing, RUE is typically derived as a ratio of time integrated NDVI over

1 annual/seasonal rainfall (Mbow et al. 2015) (a temporal approach), which is very different from the
2 original formulation of RUE (Knapp et al. 2017). The use of remote sensing based RUE is also
3 complicated by the fact that each pixel comprises an assemblage of vegetation (typically a mix of
4 herbaceous and woody vegetation) where different types of vegetation have different responses to
5 rainfall (Knapp et al. 2017). In summary, the use of RUE based assessment of land degradation is
6 problematic for vegetation types other than homogenous herbaceous vegetation.

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

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

22 Remote sensing offers much potential, but its application to land degradation and recovery remains
23 challenging as structural changes often occur at scales below the detection capabilities of most remote
24 sensing technologies. Additionally, if the remote sensing is based on vegetation indexes data, other
25 forms of land degradation, such as nutrient depletion, changes of soil physical or biological properties,
26 loss of values for humans, among others, cannot be measured directly by remote sensing.

27 Additionally, the majority of trend techniques employed would be capable of detecting only the most
28 severe of degradation processes, and would therefore not be useful as a degradation early-warning
29 system (Higginbottom et al. 2014; Wessels et al. 2012). However, additional analyses using higher
30 resolution imagery, such as the Landsat and SPOT satellites, would be well suited to provide further
31 localised information on trends observed (Higginbottom et al. 2014). In this sense, new approaches to
32 assess land degradation are in development the high (fine) spatial resolution synthetic aperture radar
33 (SAR) data, that has been shown to be advantageous for the estimation of soil surface characteristics,
34 in particular surface roughness and soil moisture (Gao et al. 2017; Bousbih et al. 2017). It is necessary
35 to maintain the efforts to fully assess land degradation using remote sensing.

36 Computer simulation models can be used alone or combined with the remote sensing observations to
37 assess land degradation. The RUSLE (Revised Universal Soil Loss Equation) can be used to predict the
38 long-term average annual soil loss by water erosion. RUSLE has been constantly revisited to estimate
39 soil loss based on the product of rainfall–runoff erosivity, soil erodibility, slope length and steepness
40 factor, conservation factor, and support practice parameter (Nampak et al. 2018). Inherent limitations
41 of RUSLE include data-sparse regions, inability to account for soil loss from gully erosion or mass
42 wasting events, and that it does not predict sediment pathways from hillslopes to water bodies
43 (Benavidez et al. 2018). In Ireland, RUSLE applied at the national scale showed poor correspondence
44 with field based methods to estimate soil loss (Rymaszewicz et al. 2015).

45 Regarding the field based approach to assess land degradation, there are multiple indicators that reflect
46 functional ecosystem processes linked to ecosystem services and, thus, to the value for humans. These
47 indicators are a composite set of measurable attributes from different factors, such as climate, soil,

1 vegetation, management, among others, that can be used together or to develop indexes to better assess
2 land degradation (Allen et al. 2011; Kosmas et al. 2014).

3 It is important to understand the history and the stage of the vegetation in natural ecosystems in order
4 to assess land degradation. Declines in vegetation cover, changes in vegetation structure and species
5 composition, decline in species and habitat diversity, changes in abundance of specific indicator species,
6 reduced vegetation health and productivity, and vegetation management intensity and use, are the most
7 common indicators in the vegetation condition of forest and woodlands (Stocking et al. 2001; Wiesmair
8 et al. 2017; Ghazoul and Chazdon 2017).

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

22 Useful indicators of degradation and improvement include changes in ecological processes and
23 disturbance regimes that regulate the flow of energy and materials and that control ecosystem dynamics
24 under a climate change scenario. Proxies of dynamics include spatial and temporal turnover of species
25 and habitats within ecosystems (Ghazoul et al. 2015; Bahamondez and Thompson 2016). Indicators in
26 agricultural lands include crop yield decreases and difficulty in maintaining yields (Stocking et al.
27 2001). Indicators of landscape degradation/improvement in fragmented forest landscapes include the
28 extent, size, and distribution of remaining forest fragments, an increase in edge habitat, and loss of
29 connectivity and ecological memory (Zahawi et al. 2015; Pardini et al. 2010).

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

37

38 **4.5 Status and current trends of land degradation**

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

42 **4.5.1 Land degradation**

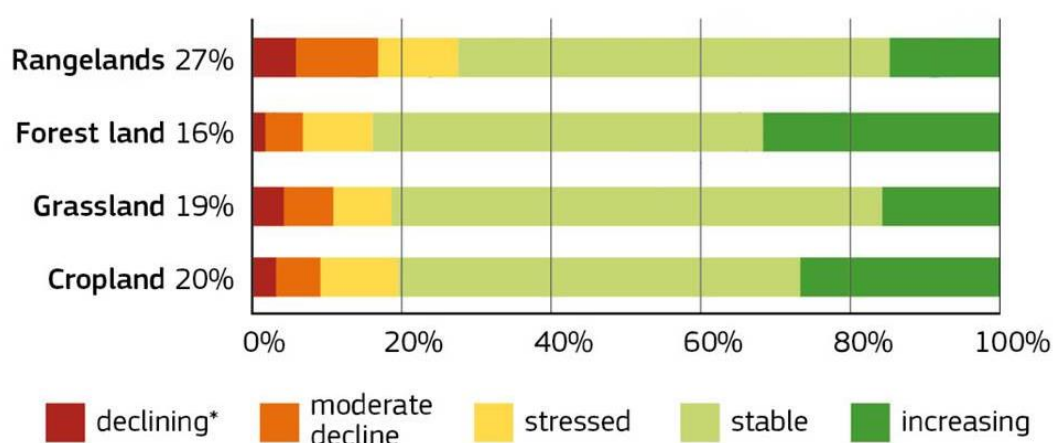
43 There are no reliable global maps of the extent and severity of land degradation (Gibbs and Salmon
44 2015), not because land degradation is not a severe problem. The reasons are both conceptual, i.e., how

1 is land degradation defined (over what time period for example) and methodological (how can it be
 2 measured). Even if there is a strong consensus that land degradation is a reduction in productivity of
 3 the land or soil, there are diverging views regarding the spatial and temporal scales at which land
 4 degradation occurs, and of course how this can be quantified and mapped. Proceeding from the
 5 definition in this report, there are also diverging views concerning ecological integrity (4.7.5) and the
 6 value to humans. A comprehensive treatment of the conceptual discussion about land degradation is
 7 provided by the recent IPBES report on land degradation (IPBES 2018).

8 A review of different attempts to map global land degradation, based on expert opinion, satellite
 9 observations, biophysical models and a data base of abandoned agricultural lands, suggested that
 10 between <1 billion ha to 6 billion ha have been degraded globally (*low agreement, limited evidence*)
 11 (Gibbs and Salmon, 2015). In a recent attempt to map soil erosion globally, Borelli et al. (2017) used a
 12 soil erosion model (RUSLE) and suggested that soil erosion is mainly caused in areas of crop land
 13 expansion, particularly in sub-Saharan Africa, south America and Southeast Asia. One widely used
 14 global assessment of land degradation used trends in NDVI as a proxy for land degradation and
 15 improvement during the period 1983 to 2006 (Bai et al. 2008b,c) with an update to 2011 (Bai et al.
 16 2015). These studies indicated that between 22% and 24% of the global land area was subject to a
 17 downward trend, while about 16% showed an increasing trend. The study also suggested, contrary to
 18 earlier assessments (Middleton and Thomas 1997), that drylands were not among the most affected
 19 regions.

20 Several publications have analysed the global/regional distribution of biological productivity loss/gain
 21 based time series of remotely sensed vegetation index data, such as NDVI over several decades
 22 (Barbosa et al. 2015; Madhu et al. 2015).

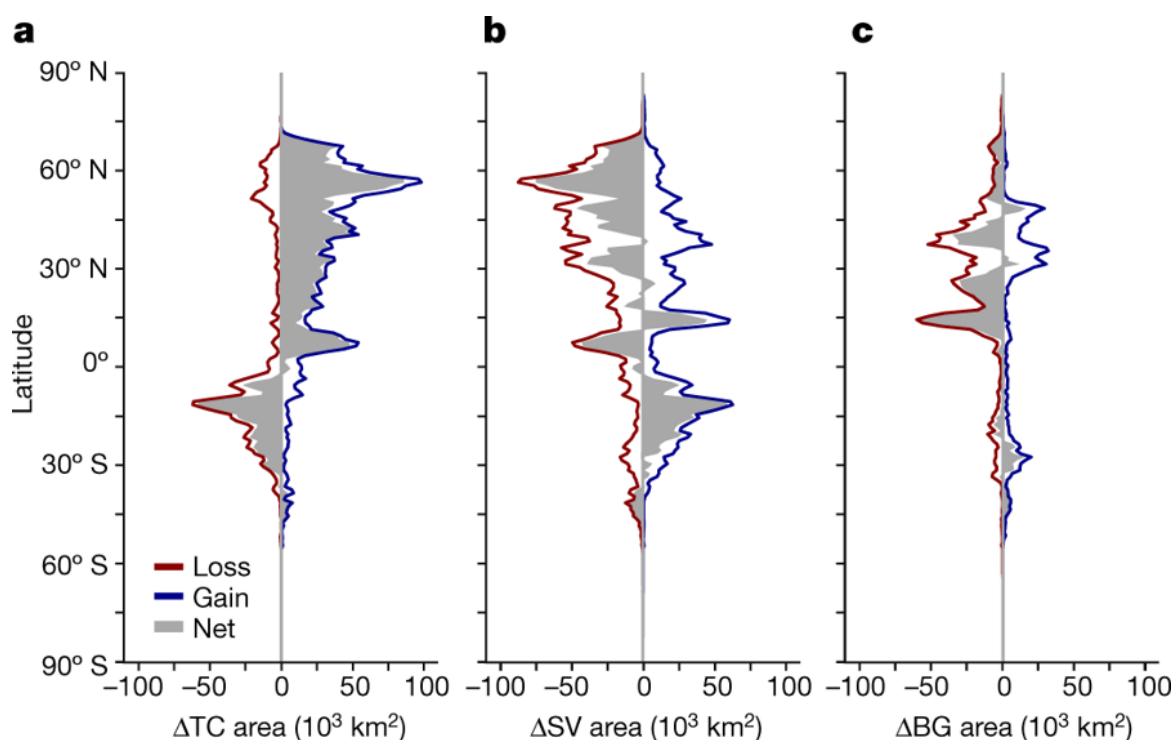
23 The World Atlas of Desertification comprises a global map of land productivity, which is one useful
 24 proxy for land degradation. In summary it shows that over the period 1999–2013 about 20% of the
 25 global ice-free land area shows signs of declining or unstable productivity whereas about 20% shows
 26 increasing productivity (JRC 2018). The same report also summarised the productivity trends by land
 27 categories and found that most forest land showed increasing trends while rangelands had more
 28 declining trends than increasing trends (Figure 4.4). These productivity assessments, however, do not
 29 distinguish between trends due to climate change and trends due to other factors.



32 **Figure 4.4 Proportional global land productivity trends by land cover/land use class. (Cropland includes**
 33 **arable land, permanent crops and mixed classes with over 50% crops; Grassland includes natural**
 34 **grassland and managed pasture land; Rangelands include shrub land, herbaceous and sparsely vegetated**
 35 **areas; Forestland includes all forest categories and mixed classes with tree cover greater than 40%)**

1 The increasing trend of productivity is to some extent consistent with findings from analysing high
 2 resolution satellite data (Landsat) suggesting that (i) global tree cover increased by 2.24 million km²
 3 between 1982 and 2016 (corresponding to +7.1%) but differentially with a net loss in the tropics and a
 4 net gain at higher latitudes, and (ii) the fraction of bare ground decreased by 1.16 million km²
 5 (corresponding to -3.1%), mainly in agricultural regions of Asia (Song et al. 2018), see Figure 4.5. The
 6 changes detected from 1982 to 2016 were primarily linked to direct human action, such as land use
 7 changes (about 60% of the observed changes) but also to indirect effects, such as human induced climate
 8 change (about 40% of the observed changes). The climate induced effects were clearly discernable in
 9 some regions, such as forest decline in the US Northwest due to increasing pest infestation and
 10 increasing fire frequency, warming induced vegetation increase in the Arctic region, general greening
 11 in the Sahel probably as a result of increasing rainfall and atmospheric CO₂, and advancing treelines in
 12 mountain regions (Song et al. 2018).

13



14

15 **Figure 4.5 Diagrams showing latitudinal profiles of land cover change over the period 1982 to 2016 based**
 16 **on analysis of timeseries of Landsat imagery: a, Tree canopy cover change (ΔTC). b, Short vegetation**
 17 **cover change (ΔSV). c, Bare ground cover change (ΔBG). Area statistics were calculated for every 1° of**
 18 **latitude (Song et al. 2018)**

19 An assessment of the global severity of soil erosion in agriculture, based on a large number of published
 20 scientific studies around the world, indicated that the global net median rate of soil formation (i.e.,
 21 formation minus erosion) is about 0.004 mm yr⁻¹ compared with the median net rate of soil loss in
 22 agricultural fields, 1.52 mm yr⁻¹ in tilled fields and 0.065 mm yr⁻¹ in no-till fields (Montgomery 2007a).
 23 This means that the rate of soil erosion from agricultural fields is in between 360 and 16 times the
 24 natural rate of soil formation (*medium agreement, limited evidence*). Climate change, mainly through
 25 the intensification of rainfall, will further increase this rate unless land management is improved (*high*
 26 *agreement, medium evidence*).

27 Global soils contain about 2500 Gt of carbon, of which 1550 is organic carbon, which is 3.3 times more
 28 carbon than the atmosphere and 4.5 times more than what is held in the vegetation of the world (Lal
 29 2004). When natural ecosystems are cultivated they lose rapidly much of the stored carbon. Estimates

1 of the magnitude of loss varies (Jackson et al. 2017; Murty et al. 2002; Guo and Gifford 2002) but is
2 larger in tropical regions, about 75%, than in cool climates, about 40-60%, (Murty et al. 2002; Lal
3 2003; Crews et al. 2018; Soussana et al. 2006). The amount of soil carbon lost explicitly due to land
4 degradation is not known. While conventional cultivation of cereals is generally a carbon source of 100
5 – 250 ton C km² yr⁻¹ (Schmidt et al. 2012) perennial grain crops can be a carbon sink of 30 – 50 ton C
6 km² yr⁻¹ (de Oliveira et al. 2018) to 150 ton C km² yr⁻¹ in sugar cane (La Scala Júnior et al. 2012).
7 Exactly how much carbon can be sequestered through reducing land degradation is highly contextual
8 and depends on the degree of soil carbon depletion, yet the potential is globally significant (Schmidt et
9 al. 2012).

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

14 In summary, there are no consistent maps and estimates of the location, extent and severity of land
15 degradation. Proxy estimates based on satellite data provide one important information source, but
16 attribution of the observed changes in productivity to climate change, human activities, or other drivers
17 is not possible. The different attempts to map the extent of global land degradation suggest that about a
18 quarter of the ice free land area is subject to some form of land degradation (*medium agreement, limited
19 evidence*). Attempts to estimate the severity of land degradation through soil erosion estimates suggest
20 that soil erosion is a serious form of land degradation in croplands closely associated with unsustainable
21 land management in combination with climatic parameters, some of which are subject to climate change
22 (*high agreement, limited evidence*). Climate change is one among several causal factors in the status
23 and current trends of land degradation (*high agreement, limited evidence*).

24 **4.5.2 Forest degradation**

25 The lack of a consistent definition of forest degradation also affects the ability to establish estimates of
26 the rates or impacts of forest degradation because the drivers of degradation are not clearly defined
27 (Sasaki and Putz 2009). Based on empirical data provided by 46 countries, the drivers for deforestation
28 (due to commercial agriculture) and forest degradation (due to timber extraction and logging) are similar
29 in Africa, Asia and Latin America (Hosonuma et al. 2012). Keenan et al. (2015) and Sloan and Sayer
30 (2015) studied the 2015 Forest Resources Assessment of the FAO and found that the total forest area
31 from 1990 to 2015 declined by 3%, an estimate that is not supported by independent remote sensing
32 analyses of changes in global land cover (Song et al. 2018). The loss rate in tropical forest areas from
33 2010 to 2015 is 5.5 M ha per year. The global natural forest area also declined from 3961 Mha to 3721
34 Mha during the period 1990 to 2015. Similarly, the forests of sub-Saharan Africa accounts for 10-20%
35 of the global plant carbon and the Congo basin forests have a carbon stock varying from 50 to 150 tons
36 ha⁻¹ (de Wasseige et al. 2015). Based on 547 sampling sites spread over humid forest areas of central
37 African countries, the annual rate of deforestation in the Congo basin is estimated around 0.09% during
38 1999 to 2000 along with a net degradation of 0.05% (Ernst et al. 2013). Due to land cover changes over
39 Southeast Asia, a majority of natural forests are converted to savanna and grasslands and changed to
40 managed land cover (Miettinen et al. 2014). The carbon stocks were less than two thirds of their natural
41 conditions mainly due to logging (Putz et al. 2008). The carbon emissions in the tropical forests of
42 Indonesia and Malaysia are estimated as 6 and 27.5 Tg C per year during 2005 (Pearson et al. 2014).
43 However, there is an increase in the planted forest. It is reported by (FAO 2010) that the total
44 deforestation during the period 1990 to 2010 in Brazil is about 55.3 million ha, in Venezuela about 5.8
45 million ha and in Mexico about 5.5 million ha. This deforestation is mainly to convert natural forests
46 and shrubs into pastures. This overuse of forest land might be due to the extensive livestock grazing,
47 illegal crop planting etc. (Willaarts et al. 2014).

1 The intensification of soil degradation due to climate change is one of the major concerns (IPCC,
2 2014B). The increase in intensive mechanized agriculture in the Brazilian Amazon resulted in the
3 conversion of more than 540,000 ha from forest to cropland during 2001 to 2004. These deforestation
4 rates have profound implications on carbon fluxes and forest fragmentations (Morton et al. 2006). In
5 Indonesia, new policies (as part of REDD++) to reduce the greenhouse gas emissions from deforestation
6 lead to increase in site level deforestation rates viz. a. viz 17-127%, 44-129% and 3.1-11.1% for oil
7 palm, timber and logging extracts (Busch et al. 2015). Globally, deforestation and other land cover
8 changes contribute to a 53 to 58% difference between current and potential biomass stocks (Erb et al.
9 2018). However, the contribution to carbon emissions (of degradation) is uncertain, with estimates
10 varying from 10% (Houghton and Nassikas 2018) to nearly 70% of carbon losses (Baccini et al. 2017),
11 although these two estimates are not strictly comparable. The “10%” estimate refers to emissions from
12 land-use change, while the “70%” estimate refers to all changes in biomass (from both losses from land-
13 use change and gains from environmental factors). Baccini et al. (2017) found that degradation within
14 forests accounted for 69% of the losses of forest biomass for the entire tropics. Studies of Lapola et al.
15 (2014) on the Brazilian land use system reported that deforestation cannot be coupled with the
16 agricultural expansion since mid of 2000s: as deforestation decreased agricultural land use and cattle
17 herd size increased. The Brazilian Amazon has undergone high rates of deforestation and about 18% of
18 its vegetal cover was affected by deforestation due to the infrastructure development, agriculture and
19 pasture (Ometto et al. 2016). Pearson et al. (2017) defined degradation as “a direct, human-induced
20 decrease in carbon stocks in forests resulting from a loss of canopy cover that is insufficient to be classed
21 as deforestation” and estimated rates of gross emissions for 74 developing countries from changes in
22 canopy density. They estimated annual gross emissions of 2.1 billion tons of carbon dioxide, of which
23 53% were derived from timber harvest, 30% from wood fuel harvest and 17% from forest fire (Pearson
24 et al. 2017). Estimating gross emissions only, creates a distorted representation of human impacts on
25 forest carbon cycles. While there is no doubt that in most countries the impacts of forest harvest for
26 timber and fuel wood and land–use change (deforestation) contribute to gross carbon emissions, it is
27 also necessary to quantify net emissions, i.e. the balance of gross emissions and gross removals of
28 carbon from the atmosphere through forest regrowth (Chazdon et al. 2016a).

29 Current efforts to reduce atmospheric CO₂ concentrations can be supported by reductions in forest-
30 related carbon emissions and increases in sinks, which requires that the net impact of forest management
31 on the atmosphere be evaluated (Griscom et al. 2017). Forest management and the use of wood products
32 in GHG mitigation strategies result in changes in forest ecosystem C stocks, changes in harvested wood
33 product (HWP) C stocks, and changes in emissions resulting from the use of wood products and forest
34 biomass that substitute for other emissions-intensive materials such as concrete, steel and fossil fuels
35 (Nabuurs et al. 2007b; Lemprière et al. 2013; Kurz et al. 2016). The net impact of these changes on
36 GHG emissions and removals, relative to a scenario without forest mitigation actions needs to be
37 quantified, (e.g. (Werner et al. 2010; Smyth et al. 2014; Xu et al. 2018b)). Therefore, reductions in
38 forest ecosystem C stocks are an incomplete estimator of the impacts of forest management on the
39 atmosphere (Nabuurs et al. 2007b; Lemprière et al. 2013; Kurz et al. 2016).

40 Assessments of forest degradation based on remote sensing of changes in canopy density or land cover,
41 (e.g., (Hansen et al. 2013; Pearson et al. 2017)) quantify changes in aboveground biomass C stocks and
42 require additional assumptions or model-based analyses to also quantify the impacts on the other carbon
43 stocks defined by the IPCC, including belowground biomass, litter, woody debris and soil carbon.
44 Depending on the type of disturbance, changes in aboveground biomass may lead to decreases or
45 increases in other carbon pools, for example, windthrow may result in losses in aboveground biomass
46 that are (initially) off-set by corresponding increases in dead organic matter carbon pools, while
47 deforestation will reduce all ecosystem carbon pools.

1 Impacts of deforestation and forest degradation, including forest management have resulted in carbon
2 stock reductions (-21 to 38%) in global forests relative to the hypothetical natural forest conditions (Erb
3 et al. 2018). A regional study of Erb et al. (2008) over Austria found that carbon released due to
4 expansion of crop land and fossil fuel burning increased carbon in the atmosphere during the period
5 1830 to 2000. Studies of Gingrich et al. (2015) on the long-term trends in land use over nine European
6 countries (Albania, Austria, Denmark, Germany, Italy, the Netherlands, Romania, Sweden and the
7 United Kingdom) show the increase in forest land and reduction in crop land and grazing land from the
8 19th century to the early 20th century. However, the extent to which human activities have reduced the
9 productive capacity of forest lands is poorly understood. Biomass Production Efficiency (BPE) was
10 significantly higher in managed forests compared to natural forests (and it was also higher in managed
11 compared to natural grasslands) (Campioli et al. 2015). A larger proportion of Net Primary Production
12 in managed forests is allocated to biomass carbon storage, but lower allocation to fine roots is
13 hypothesized to reduce soil C stocks in the long-term (Noormets et al. 2015).

14 As economies evolve, the patterns of land use and carbon stock changes associated with human
15 expansion into forested areas often include a period of rapid decline of forest area and carbon stocks,
16 recognition of the need for forest conservation and rehabilitation, and a transition to more sustainable
17 land management that is often associated with increasing carbon stocks, (e.g. Birdsey et al. 2006).
18 Developed and developing countries around the world are in various stages of forest transition (Kauppi
19 et al. 2018). Thus, opportunities exist for sustainable forest management to contribute to atmospheric
20 carbon targets through avoidance of deforestation and degradation, forest conservation, forest
21 restoration and enhancements of carbon stocks in forests and harvested wood products(Griscom et al.
22 2017).

23

24 **4.6 Projections of land degradation in a changing climate**

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

34 Even if climate change is exacerbating many land degradation processes (*high to very high confidence*),
35 prediction of future land degradation is challenging because land management practices determine to a
36 very large extent the state of the land. Scenarios of climate change in combination with land degradation
37 models can provide very useful knowledge on what kind and extent of land management is necessary
38 in order to avoid, reduce and prevent land degradation.

39 **4.6.1 Direct impacts on land degradation**

40 There are two main levels of uncertainty in assessing the risks of future climate change induced land
41 degradation. The first level, where uncertainties are comparatively low, is the changes of the degrading
42 agent, such as erosive power of precipitation, heat stress from increasing temperature extremes (HÜVE
43 et al. 2011), and water stress from droughts. The second level of uncertainties, and where the
44 uncertainties are much larger, relates to the vegetation changes as a result of changes in rainfall,
45 temperature, and increasing level of CO₂. The protective function of vegetation is crucial for erosion
46 (Mullan et al. 2012; García-Ruiz et al. 2015).

1 Changes in rainfall patterns, such as distribution in time and space, and intensification of rainfall events
2 will increase the risk of land degradation, both in terms of likelihood and consequences (*high*
3 *agreement, medium evidence*). Climate induced vegetation changes will pose increasing risk of land
4 degradation in some areas (where vegetation cover will decline) (*medium confidence*). Changes of
5 hydrological regimes as a combined result of human land/water use and climate change will seriously
6 impact floodplains and delta areas (*very high confidence*). Erosion of coastal areas as a result of sea
7 level rise will increase worldwide (*very high confidence*). In hurricane prone areas (such as the
8 Caribbean, Southeast Asia, and the Bay of Bengal) the combination of sea level rise and more intense
9 hurricanes, and sometimes also land subsidence, will pose a serious risk to people and livelihoods (*very*
10 *high confidence*), in some cases even exceeding limits to adaptation.

11 **4.6.1.1 Changes in erosion risk due to precipitation changes**

12 The hydrological cycle is intensifying with increasing warming of the atmosphere. The intensification
13 means that the number of heavy rainfall events is increasing while the total number of rainfall events
14 tends to decrease (Trenberth 2011; Li and Fang 2016; Kendon et al. 2014; Guerreiro et al. 2018; Burt
15 et al. 2016; Westra et al. 2014) (*high agreement, robust evidence*).

16 Modelling of changes in land degradation as a result of climate change alone is hard because of the
17 importance of local contextual factors. As shown above, actual erosion rate is extremely dependent on
18 local conditions, primarily vegetation cover and topography (García-Ruiz et al. 2015). Nevertheless,
19 modelling of soil erosion risks has advanced substantially in recent decades and such studies are
20 indicative of future changes in the risk of soil erosion while actual erosion rates will still primarily be
21 determined by land management. In a review article, Li & Fang (Li and Fang 2016) summarised 205
22 representative modelling studies around the world where erosion models had been used in combination
23 with down-scaled climate models to assess future (between 2030 to 2100) erosion rates. Almost all of
24 the sites had current soil loss rates above 1 t ha^{-1} (often assumed to be the upper limit for acceptable soil
25 erosion) and 136 out of 205 studies predicted increased soil erosion rates. The percentage increase in
26 erosion rates varied between 1.2% to as much as over 1600%, whereas 49 out of 205 studies projected
27 more than 50% increase.

28 Mesoscale convective systems (MCS), typically thunder storms, have increased markedly in recent 3-
29 4 decades in the USA and Australia and they are projected to increase substantially (Prein et al. 2017).
30 Using a climate model with the ability to represent MCS, Prein and colleagues were able to predict
31 future increases in frequency, intensity, and size of such weather systems. Findings include the 30%
32 decrease in number of MCS of $<40\text{ mm h}^{-1}$, but a sharp increase of 380% in the number of extreme
33 precipitation events of $>90\text{ mm h}^{-1}$ over the North American continent. The combined effect of
34 increasing precipitation intensity and increasing size of the weather systems implies that the total
35 amount of precipitation from these weather systems is expected to increase by up to 80% (Prein et al.
36 2017), which will substantially increase the risk of land degradation in terms of landslides, extreme
37 erosion events, flashfloods etc.

38 Using a comparative approach Serpa and colleagues (2015) studied two Mediterranean catchments (one
39 dry and one humid) using a spatially explicit hydrological model (SWAT) in combination with land use
40 and climate scenarios for 2071-2100. Climate change projections showed, on the one hand, decreased
41 rainfall and streamflow for both catchments whereas sediment export decreased only for the humid
42 catchment; projected land use change, from traditional to more profitable, on the other hand resulted in
43 increase in streamflow. The combined effect of climate and land use change resulted in reduced
44 sediment export for the humid catchment (-29% for A1B; -22% for B1) and increased sediment export
45 for the dry catchment (+222% for A1B; +5% for B1). Similar methods have been used elsewhere, also
46 showing the dominant effect of land use/land cover for runoff and soil erosion (Neupane and Kumar
47 2015).

1 A study of future erosion rates in Northern Ireland, using a spatially explicit erosion model in
2 combination with downscaled climate projections (with and without sub-daily rainfall intensity
3 changes), showed that erosion rates without land management changes would decrease by 2020s, 2050s
4 and 2100s irrespective of changes in intensity, mainly as a result of a general decline in rainfall (Mullan
5 et al. 2012). When land management scenarios were added to the modelling, the erosion rates started
6 to vary dramatically for all three time periods, ranging from a decrease of 100% for no-till land use, to
7 an increase of 3621% for row crops under annual tillage and sub-days intensity changes (Mullan et al.
8 2012). Again, it shows how crucial land management is for addressing soil erosion, and the important
9 role of rainfall intensity changes.

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

13 **4.6.1.2 Climate induced vegetation changes, implications for land degradation**

14 Forests influence the storage and flow of water in watersheds (Eisenbies et al. 2007, Sheil and
15 Murdiyarso, 2009; Ellisson et al 2012 & 2017; Creed and van Noordwijk 2018; Sheil 2018) and are
16 therefore important for regulating how climate change will impact landscapes. Hence the ways in which
17 forests are impacted by climate change and other dynamics, such as forest management and natural
18 disturbances, are keys to understanding future impacts on land degradation. Generally, removal of trees
19 through harvesting or forest death (Anderegg et al. 2012) will reduce transpiration and hence increase
20 the runoff during the growing season. Management induced soil disturbance (such as skid trails and
21 roads) will affect water flow routing to rivers and streams (Zhang et al. 2017; Luo et al. 2018; Eisenbies
22 et al. 2007).

23 Climate change affects forests in both positive and negative ways (Trumbore et al. 2015) and there will
24 be regional and temporal differences in vegetation responses (Hember et al. 2017; Midgley and Bond
25 2015). In high latitudes, a warmer climate will increase water use efficiency and extend the growing
26 seasons, while increasing levels of atmospheric CO₂ will potentially increase vigor and growth (Norby
27 et al. 2010). Increasing forest productivity has been observed in most of Fennoscandia, Siberia and the
28 northern reaches of North America as a response to a warming trend (Gauthier et al. 2015), which is
29 predicted to continue until 2030 in most of Siberia (FAO 2012), in contrast to North America where a
30 decrease is predicted (Price et al. 2013; Girardin et al. 2016; Beck et al. 2011). The climatic conditions
31 in high latitudes are changing a magnitude faster than the ability of forests to adapt with detrimental,
32 yet unpredictable, consequences (Gauthier et al. 2015). There is an increasing carbon storage in old-
33 growth forests the pan-tropical area (Baker et al. 2004; Lewis et al. 2009).

34 Negative impacts dominate, however, and have already been documented (Lewis et al. 2004 ; Bonan et
35 al. 2008; Beck et al. 2011) and are predicted to increase (Miles et al. 2004 ; Allen et al. 2010; Gauthier
36 et al. 2015; Girardin et al. 2016; Trumbore et al. 2015). Several authors have emphasised a concern that
37 tree mortality will increase due to climate induced physiological stress as well as interactions between
38 physiological stress and other stressors, such as insect outbreaks and wildfires (Anderegg et al. 2012;
39 Sturrock et al. 2011; Bentz et al. 2010; McDowell et al. 2011). Extreme events such as extreme heat
40 and drought, storms, and floods also pose increased risks to forests in both high and low latitude forests
41 (Lindner et al. 2010; Mokria et al. 2015). Forests are subject to increasing frequency and intensity of
42 wildfires which is projected to increase substantially with continued climate change (see Cross-Chapter
43 Box 3: Fire and climate change, Chapter 2). In the tropics, interaction between climate change, CO₂ and
44 fire are susceptible to lead to abrupt shifts between woodland and grassland dominated states in the
45 future (Shanahan et al. 2016). Several types of forest disturbance can be cause by fire in the tropics and
46 beyond (Ferry Slik et al. 2002) (Dale et al. 2001). Such fire generally affects the ecosystem services
47 expected from this forest (Page et al. 2002).

1 Water balance of at least partly forested landscapes is to a large extent controlled by forest ecosystems
2 (Sheil and Murdiyarso 2009; Pokam et al. 2014). This includes surface runoff, as determined by
3 evaporation and transpiration and soil conditions, and water flow routing (Eisenbies et al. 2007). Water
4 use efficiency (i.e., the ratio of water loss to biomass gain) is increasing with increased CO₂ levels
5 (Keenan et al. 2013), hence transpiration is predicted to decrease which in turn will increase surface
6 runoff (Schlesinger and Jasechko 2014). Surface runoff is an important agent in soil erosion.

7 Rangelands are projected to change in complex ways due to climate change. Increasing levels of
8 atmospheric CO₂ stimulate directly plant growth and can potentially compensate negative effects from
9 drying by increasing rain use efficiency. But the positive effect of increasing CO₂ will be mediated by
10 other environmental conditions, primarily water availability but also nutrient cycling, fire regimes and
11 invasive species. Studies over the North American rangelands suggest for example that warmer and
12 dryer climatic conditions will reduce NPP in the southern Great Plains, the Southwest, and northern
13 Mexico, but warmer and wetter conditions will increase NPP in the northern Plains and southern Canada
14 (Polley et al. 2013).

15 Forests influence the storage and flow of water in watersheds (Eisenbies et al. 2007) and are therefore
16 important for regulating how climate change will impact landscapes. Hence the ways in which forests
17 are impacted by climate change and other dynamics, such as forest management and natural
18 disturbances, are keys to understanding future impacts on land degradation. Generally, removal of trees
19 through harvesting or forest death (Anderegg et al. 2012) will reduce transpiration and hence increase
20 the runoff during the growing season. Management induced soil disturbance (such as skid trails and
21 roads) will affect water flow routing to rivers and streams (Zhang et al. 2017; Luo et al. 2018; Eisenbies
22 et al. 2007).

23 **4.6.1.3 Changes of hydrological regimes**

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

26 Changes in river runoff between different climate scenarios can be studied through hydrological models
27 in combination with climate scenarios and models. Comparisons of different hydrological models and
28 different climate models suggest that the variability in the runoff results is considerably larger between
29 climate models than the variability among hydrological models (Gosling et al. 2011; Teng et al. 2012).
30 Such studies in Australia predict a drier future for the Southeastern part of the continent and that the
31 hydrological response of reduced rainfall is amplified (Teng et al. 2012). In West Africa, a meta-study
32 of 19 hydrological simulations of climate change impacts on river runoff showed mixed results
33 concluding that even if there are theoretical reasons for climate change impacts on runoff in the region,
34 available data and models do not yet capture these changes (Roudier et al. 2014).

35 **4.6.1.4 Coastal erosion**

36 Coastal erosion is expected to increase dramatically by sea level rise and in some areas in combination
37 with increasing intensity of hurricanes/typhoons (highlighted in Section 4.11.5 Hurricane induced
38 coastal erosion). Coastal regions are also characterised by high population density, particularly in Asia
39 (Bangladesh, China, India, Indonesia, Vietnam) whereas the highest population increase of coastal
40 regions is projected in Africa (East Africa, Egypt, and West Africa) (Neumann et al. 2015).

41 Despite the uncertainty related to the responses of the large ice sheets of Greenland and west Antarctica,
42 climate change-induced sea level rise is largely accepted and represents one of the biggest threats faced
43 by coastal communities and ecosystems (Nicholls et al. 2011; Cazenave and Cozannet 2014). With
44 significant socio-economic effects, the physical impacts of projected sea level rise, notably coastal
45 erosion, have received considerable scientific attention (Nicholls et al. 2011; Rahmstorf 2010). A
46 relationship has been found, valid for different regions of the world, between the rates of relative sea-

1 level rise and coastal erosion or recession (Meeder and Parkinson 2018; Shearman et al. 2013; Savard
2 et al. 2009; Yates et al. 2013).

3 Rates of coastal erosion or recession will increase due to rising sea levels and in some regions also in
4 combination with increasing oceans waves (McInnes et al. 2011; Mori et al. 2010), lack or absence of
5 sea-ice (Savard et al. 2009) and changing hurricane paths (Tamarin-Brodsky and Kaspi 2017). The
6 respective role of the different climate factors in the coastal erosion process will vary spatially. Some
7 studies have shown that the role of sea level rise on the coastal erosion process can be less important
8 than other climate factors, like wave heights, changes in the frequency of the storms, and the cryogenic
9 processes (Ruggiero 2013; Savard et al. 2009). Therefore, in order to have a complete picture of the
10 potential effects of sea level rise on rates of coastal erosion, it is crucial to consider the combined effects
11 of the aforementioned climate controls and also the lithostratigraphy of the coast under study.

12 Many large coastal deltas are subject to the additional stress of shrinking deltas as a consequence of the
13 combined effect of reduced sediment loads from rivers due to damming and water use, and land
14 subsidence resulting from extraction of ground water or natural gas, and aquaculture (Higgins et al.
15 2013; Tessler et al. 2016; Minderhoud et al. 2017; Tessler et al. 2015; Brown and Nicholls 2015). In
16 some cases the rate of subsidence can outpace the rate of sea level rise by one magnitude of order
17 (Minderhoud et al. 2017) or even two (Higgins et al. 2013).

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

22 **4.6.2 Indirect impacts on land degradation**

23 Indirect impacts of climate change on land degradation are difficult to quantify because of the many
24 conflating factors. The causes of land use change are complex and multifaceted, combining physical,
25 biological and socioeconomic drivers (Lambin et al. 2001; Lambin and Meyfroidt 2011). One such
26 driver of land use change is the degradation of agricultural land, which can result in a negative cycle of
27 natural land being converted to agricultural land to sustain production levels. The intensive management
28 of agricultural land can lead to a loss of soil function, negatively impacting the many ecosystem services
29 provided by soils including maintenance of water quality and soil carbon sequestration (Smith et al.
30 2016a). The degradation of soil quality is of particular concern in tropical regions, where it results in a
31 loss of productive potential of the land, affecting regional food security and driving conversion of non-
32 agricultural land, such as forestry, to agriculture (Lambin et al. 2003). Climate change will exacerbate
33 these negative cycles unless sustainable land managed practices are implemented.

34 Climate change impacts on agricultural productivity will have implications for the intensity of land use
35 and hence exacerbate the risk of increasing land degradation. There will be both localised effects (i.e.,
36 climate change impacts on productivity affecting land use in the same region) and teleconnections (i.e.,
37 climate change impacts and land use change are spatially and temporally separate) (Wicke et al. 2012;
38 Pielke et al. 2007). Crop modelling studies suggest that the productivity of major crops will decline as
39 a result of climate change, particularly from increasing warming (Schlenker and Roberts 2009;
40 Rosenzweig et al. 2014). Grain yield of rice declined 10% for each 1°C increase in night-time
41 temperature during the dry season (Peng et al. 2004), wheat yields are expected to decline by 6% for
42 each 1°C increase (Asseng et al. 2015), while maize and soy bean yields are expected to decline by 6%
43 for each day above 30°C.

44 If global temperature increases beyond 3°C it will have negative yield impacts on all crops (Porter et
45 al. 2014) which in combination with a doubling of demands by 2050 (Tilman et al. 2011), and increasing

1 competition for land from the expansion of bioenergy (Schleussner et al. 2016) will exert strong
2 pressure on agricultural lands and food security.

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

7

8 **4.7 Impacts of bioenergy provision on land degradation**

9 **4.7.1 Provision of bioenergy and land-based negative emission technologies (NETs)**

10 In addition to the traditional land use drivers (e.g. population growth, agricultural expansion, forest
11 management), two new drivers will interact to increase competition for land throughout this century:
12 climate change induced biome shifts and the potential large-scale implementation of land-based
13 negative emissions technologies. Climate change impacts will cause significant shifts in biomes by
14 affecting temperature and water regimes reducing water supply and biomass productivity in some areas
15 and increasing them in others (Gerten et al. 2013; Gonzalez et al. 2010; Tang et al. 2014).

16 These biome shifts will require various adaptive responses, such as changes in agricultural crop types,
17 tree species and management systems, abandonment of land areas no longer suitable for their current
18 uses, and expansion of land use into previously unmanaged lands. Such changes in land use can result
19 in deforestation, conversion of natural to managed forests and land degradation. Early expressions of
20 biome shifts include increased plant stress, species maladaptation, and increased rates of tree mortality
21 (Allen et al. 2010).

22 In some/most scenarios compatible with stabilisation at 2°C, land-based negative emission technologies
23 such as afforestation, reforestation, intensified land management, and bioenergy crops will need to be
24 implemented on very large land areas (Schleussner et al. 2016; Smith et al. 2016b; Mander et al. 2017).
25 Even larger land areas are required in scenarios aimed at keeping average global temperature increases
26 to below 1.5 °C (IPCC 2018). The Summary for Policymakers of the IPCC SR 1.5 states that “Model
27 pathways that limit global warming to 1.5°C with no or limited overshoot project the conversion of 0.5–
28 8 million km² of pasture and 0–5 million km² of non-pasture agricultural land for food and feed crops
29 into 1–7 million km² for energy crops and a 1 million km² reduction to 10 million km² increase in forests
30 by 2050 relative to 2010” and further states that “[t]he implementation of land-based mitigation options
31 would require overcoming socio-economic, institutional, technological, financing and environmental
32 barriers that differ across regions” (IPCC 2018).

33 The requirement for NETs varies widely between different scenarios for meeting the targets of the Paris
34 Agreement, ranging from 450–1000 GtCO₂ accumulated during the 21st century for 1.5°C and 0–900
35 GtCO₂ for 2°C (Rogelj et al. 2015; Dooley and Kartha 2018). Estimates of the biophysical possibilities
36 of such amounts indicate a range of 370–480 Gt CO₂. Even this range must be considered optimistic
37 from a social and ethical point of view (Cox et al. 2018a; Faran and Olsson 2018; Gough and Vaughan
38 2015). Common to all analyses of land requirements for NET is the very large range of estimates, that
39 can span more than one order of magnitude. This wide range reflects the large differences among the
40 pathways, availability of land in various productivity classes, types of NET implemented, uncertainties
41 in computer models, and social and economic barriers to implementation (Fuss et al. 2018; Nemet et al.
42 2018; Minx et al. 2018).

4.7.2 Potential risks of land-based NET impacts on land degradation

Many drivers and processes can degrade lands (see Section 4.3, Table 4.1 for references). The large-scale implementation of high intensity energy crops, or other activities that may involve non-sustainable land management practices, can contribute to increases in the area of degraded lands. Intensive land management can result in nutrient depletion, over fertilization and soil acidification, salinization (from irrigation without adequate drainage), wet ecosystems drying (from increased evapotranspiration), as well as novel erosion and compaction processes (from high impact biomass harvesting disturbances) and other land degradation processes described in more detail in Section 4.3.

While risks of land degradation from the establishment of energy plantations can be anticipated, the area ultimately affected is presently unknown because the geographic location, local incentives and regulations, extent and plantation types and the management practices that will be implemented are unknown. Other NETs such as reforestation and afforestation are at lower risk to contributing to land degradation and may in fact reverse degradation (see Section 4.7.3).

Global models used in the analyses of mitigation pathways are unable to consider site-specific details. While model projections identify potential areas for NET implementation (Heck et al. 2018), the interaction with climate change induced biome shifts, available land and its vulnerability to degradation are unknown. It is therefore currently not possible to project the area at risk for degradation from the implementation of land-based NETs. However, if land management is changed to implement large-scale bioenergy plantations, there is a definitive risk that some land areas will be adversely affected.

4.7.3 Potential contributions of land-based NETs to land restoration

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

Estimates of the global area of degraded land cover a range of 1,000 to 6,000 Mha (Gibbs and Salmon 2015). Large additional areas are classified as marginal lands and may also be suitable for the implementation of land-based NETs. The yield per hectare of marginal and degraded lands is lower than on fertile lands, and if NETs will be implemented on marginal and degraded lands this will increase the area demand and costs per unit area of achieving negative emissions (Fritsche et al. 2017). Selection of lands suitable for NETs must be considered carefully to reduce conflicts with existing users, to assess the possible trade-offs in biodiversity contributions of the original and the NET land uses, to quantify the impacts on water budgets, and to ensure sustainability of the NET land use.

Land conditions prior to the implementation of NET have strong impacts on the greenhouse gas benefits (Harper et al. 2018). Afforestation and reforestation on marginal or degraded lands can lead to increases in C stocks in biomass, dead organic matter and soil C pools, increased carbon sinks (Amichev et al. 2012), and a number of co-benefits, including increases in biodiversity and improvements of other ecosystem services. Afforestation and reforestation on native grasslands can result in reduced soil carbon stocks, although these are typically more than compensated by increases in biomass and dead organic matter C stocks (Bárcena et al. 2014; Li et al. 2012; Ovalle-Rivera et al. 2015; Shi et al. 2013) and may cause changes in biodiversity (Li et al. 2012) (see also 4.5.1: Large scale forest cover expansion, what can be learned in context of the SRCCL).

Changes in land use can affect surface reflectance, water balances and emissions of volatile organic compounds and thus the non-GHG impacts on the climate system (Anderson et al. 2011; Bala et al. 2007; Betts 2000; Betts et al. 2007), (see Section 4.8 for further details). Some of these impacts reinforce

1 the GHG mitigation benefits, while others off-set the benefits, with strong local (slope, aspect) and
2 regional (boreal vs. tropical biomes) differences in the outcomes (Li et al. 2015). Adverse effects on
3 albedo from afforestation with evergreen conifers in boreal zones can be reduced through planting of
4 broadleaf deciduous species (Astrup et al. 2018; Cai et al. 2011a; Anderson et al. 2011).

5 Implementation of land-based NET will require site-specific local knowledge, matching of species with
6 the local land, water balance, nutrient and climatic conditions, and ongoing monitoring and, where
7 necessary, adaptation of land management to ensure sustainability.

8 Just as NET initiatives can contribute to land restoration objectives, international initiatives to restore
9 lands, such as the Bonn Challenge and the New York Declaration on Forests, and interventions
10 undertaken for Land Degradation Neutrality, can contribute to NET objectives. Such synergies may
11 increase the financial resources available to meet multiple objectives.

12 **4.7.4 Traditional biomass provision and land degradation**

13 More than half (around 55%) of all the wood harvested worldwide is for fuelwood, providing about 9%
14 of global primary energy (Bailis et al. 2015; Masera et al. 2015). The utilisation of traditional biomass
15 to respond to energy needs for heating and cooking generally in an inefficient way (often in open fires
16 or very inefficient indoor stoves) is still the main uses of bioenergy (Chum et al. 2011; REN21 2018).
17 About 2.8 billion people (38% of the global population) without clean cooking facilities (REN21 2018)
18 depend mainly on traditional biomass. In the developing countries of Sub-Saharan Africa, bioenergy
19 still constitutes a high proportion of the national energy, particularly for domestic uses in both urban
20 and rural areas (Cerutti et al. 2015). The predictor for global bioenergy energy will be not only
21 population increase but also changes in household numbers and size (Knight and Rosa 2012). Bio-
22 energy demand is one of the main drivers of the global forest lost particularly in many tropical countries
23 (Arnold et al. 2003). 27–34% of wood fuel harvested is unsustainable with around one quarter of a
24 billion people living in wood fuel depletion hotspots in South Asia and East Africa (Bailis et al. 2015).
25 In South East Asia, for example, without incentives to reduce deforestation and forest degradation,
26 standing wood biomass will be reduced (Sasaki et al. 2009). According to Kraxner et al. (2013), under
27 a high global demand for bio-energy by mid-term century, biomass will to a large extent be sourced
28 from conversion of unmanaged forest to managed forest, from new fast growing short rotation
29 plantations, intensification and optimisation of land use (Kraxner et al. 2013). More importantly, to
30 reverse deforestation, the potential of bio-energy from degraded land can be from 150 to 190 EJ yr⁻¹
31 (Nijsen et al. 2012).

32 Distributed renewables for energy access (DREA), such an improved stove coupled with other Energies
33 efficiencies improvement along the value chain of the traditional biomass appeared as solution to
34 provide energy in a sustainable way (Hofstad et al. 2009; Creutzig et al. 2015; REN21 2018). If 100
35 million improved stoves were to be successfully utilised, this would reduce the global carbon foot print
36 of traditional wood (1.0–1.2 Gt of CO₂ representing 1.9–2.3% of the global emission) by 11 to 17%
37 (Bailis et al. 2015). To turn efforts to reduce deforestation and forest degradation into realities, and thus
38 reduce/reverse the trend of degradation, big political will and transformational changes will be required
39 (Arnold et al. 2003). In Sub-Saharan Africa, around three quarters of countries mention wood fuel in
40 their NDC but fail to identify the transformational process to make fuelwood a sustainable energy source
41 compatible with improved forest management (Amugune et al. 2017). The traditional biomass sector in
42 Africa is generally in a certain form of informality and illegality (Chum et al. 2011) (Schure et al. 2013,
43 2014, 2015; Porter et al. 2014)(Smith et al. 2015). The five main policy interventions useful to transform
44 the sector include: cooking efficiency; fuel substitution; production efficiency; controlling harvest; and
45 plantations (Hofstad et al. 2009). The political will need to be backstopped by a good set of criteria and
46 indicator for sustainable wood fuel (Stupak et al. 2011) and by application of a reasonable price of the
47 Biomass (Lauri et al. 2014).

1 Fuelwood remains a primary source of energy for a large fraction of the global population leading to
2 unsustainable use of vegetation resources, forest degradation and deforestation. Enhanced forest
3 protection, improved forest management, and provision of alternate cooking and heating devices to
4 reduce traditional use of fuelwood and charcoal can reduce pressure on forests with additional co-
5 benefits such as improved human health.

6 **4.7.5 Reducing deforestation and forest degradation**

7 Improved stewardship of forests through reduction or avoidance of deforestation and forest degradation,
8 and enhancement of forest carbon stocks can all contribute to land-based natural climate solutions
9 (Griscom et al. 2017). Estimates of annual emissions from tropical deforestation and forest degradation
10 range from 0.5 to 3.5 Pg C yr⁻¹ (Baccini et al. 2017; Houghton et al. 2012; Mitchard 2018), indicating
11 the large potential to reduce annual emissions from the land sector. Miles and co-authors (2015) in the
12 preparation of the Paris Agreement developed what they call technical mitigation potential of tropical
13 forest. Revised baseline estimates of forest extent for Africa may result in downward adjustments of
14 deforestation and degradation emission estimates (Aleman et al. 2018). Emissions from forest
15 degradation in non-Annex I countries have declined marginally from 1.1 GtCO₂ yr⁻¹ in 2001-2010 to 1
16 GtCO₂ yr⁻¹ in 2011-2015, but the relative emissions from degradation compared to deforestation have
17 increased from a quarter to a third (Federici et al. 2015).

18 Natural regeneration of second-growth forests contributes significant and annually increasing carbon
19 sinks to the global carbon budget (Chazdon and Uriarte 2016). In Latin America, Chazdon et al. (2016)
20 estimated that in 2008, second-growth forests (1 to 60 years old) covered 2.4 million km² of land (28.1%
21 of the total study area). Over 40 years, these lands can potentially accumulate a total aboveground
22 carbon stock of 8.48 Pg C (petagrams of carbon) in aboveground biomass via low-cost natural
23 regeneration or assisted regeneration, corresponding to a total CO₂ sequestration of 31.09 Pg CO₂. And
24 while aboveground biomass carbon stocks are estimated to be declining in the tropics, they are
25 increasing globally due to increasing stocks in temperate and boreal forests (Liu et al. 2015b), consistent
26 with the observations of a land sector carbon sink (Le Quéré et al. 2013; Keenan et al. 2017). From
27 1982 to 2016, Song et al. (2018) noticed that of all land changes, 60% are associated with direct human
28 activities and 40% with indirect drivers such as climate change; and that land use change exhibits
29 regional dominance, including tropical deforestation and agricultural expansion, temperate reforestation
30 or afforestation, cropland intensification and urbanisation.

31 Moving from technical potential (Miles et al. 2015) to real reduction of deforestation and degradation
32 required some transformational changes. A recent studies (Korhonen-Kurki et al. 2018) reveals that
33 tropical countries are making some transformational changes to tackle deforestation and forest
34 degradation. But this transformation is facilitated by two enabling institutional configurations: (1) the
35 presence of already initiated policy change; and (2) scarcity of forest resources combined with an
36 absence of any effective forestry framework and policies. These authors found that the presence of
37 powerful transformational coalitions combined with strong ownership and leadership, and performance-
38 based funding, can both work as a strong incentive for achieving REDD+ goals.

39 Implementing schemes such as REDD+ and various projects related to the voluntary carbon market is
40 often regarded as a no-regrets investment (Seymour and Angelsen 2012) but the social and ecological
41 implications must be carefully considered. From a livelihoods and poverty point of view, the IPCC Fifth
42 Assessment Report showed mixed results from both types of initiatives (Olsson et al. 2014). In terms
43 of ecological integrity of tropical forests, the strong focus on carbon storage is problematic since it often
44 leaves out other aspects of forests ecosystems, such as biodiversity – and particularly fauna (Panfil and
45 Harvey 2016; Peres et al. 2016; Hinsley et al. 2015). Another concern of schemes such as REDD+ is
46 the issue of leakage, i.e. when interventions to reduce deforestation or degradation at one site displace

1 pressures and increase emissions elsewhere (Atmadja and Verchot 2012; Phelps et al. 2010; Lund et al.
2 2017; Balooni and Lund 2014).

3 **4.7.6 Sustainable forest management and negative emissions**

4 While avoiding deforestation and forest degradation may help meet short term goals, sustainable forest
5 management aimed at providing timber, fiber, biomass and non-timber resources can provide long-term
6 livelihood for communities, can reduce the risk of forest conversion to non-forest uses (settlement,
7 crops, etc.), and can maintain land productivity, thus reducing the risks of land degradation (Putz et al.
8 2012; Gideon Neba et al. 2014; Sufo Kankeu et al. 2016; Nitcheu Tchiadje et al. 2016; Rossi et al.
9 2017).

10 Developing sustainable forest management strategies aimed at contributing towards negative emissions
11 throughout this century requires an understanding of forest management impacts on ecosystem carbon
12 stocks (including soils), carbon sinks, carbon fluxes in harvested wood, carbon storage in harvested
13 wood products and the emission reductions achieved through the use of wood products (Nabuurs et al.
14 2007a; Lemprière et al. 2013; Kurz et al. 2016). Transitions from natural to managed forest landscapes
15 can involve a reduction in forest carbon stocks, the magnitude of which depends on the harvest rotation
16 length relative to the frequency and intensity of natural disturbances (Harmon et al. 2009; Kurz et al.
17 1998), but can increase the forest carbon sink, and generate climate benefits through increasing the
18 stock of wood products and displacing GHG-intensive building materials (Werner et al. 2010; Smyth
19 et al. 2014).

20 Forest growth rates, net primary productivity, and net ecosystem productivity are age-dependent with
21 maximum rates of carbon removal from the atmosphere occurring in young to medium aged forests and
22 declining thereafter (Tang et al. 2014). In boreal forest ecosystem, measurements of carbon stocks and
23 carbon fluxes indicate that old growth stands are typically small carbon sinks or carbon sources (Gao
24 et al. 2018; Taylor et al. 2014; Hadden and Grelle 2016). Age-dependent increases in forest carbon
25 stocks and declines in forest carbon sinks mean that landscapes with older forests have accumulated
26 more carbon but their sink strength is diminishing, while landscapes with younger forests contain less
27 carbon but they are removing CO₂ from the atmosphere at a much higher rate (Volkova et al. 2017).
28 This fundamental trade-off between maximising forest C stocks and maximising ecosystem C sinks is
29 at the origin of ongoing debates about optimum management strategies to achieve negative emissions
30 (Keith et al. 2014; Kurz et al. 2016; Lundmark et al. 2014).

31 With increasing forest age, carbon sinks in forests will diminish until harvest or natural disturbances
32 such as wildfire remove biomass carbon or release it to the atmosphere. Sustainable forest management,
33 including harvest and forest regeneration, can help maintain active carbon sinks by maintaining a forest
34 age-class distribution that includes a share of young, actively growing stands (Volkova et al. 2018).
35 The use of the harvested carbon in either long-lived wood products (e.g. for construction), short-lived
36 wood products (e.g., pulp and paper), or biofuels affects the net carbon balance of the forest sector
37 (Lemprière et al. 2013). The use of these wood products can further contribute to GHG emission
38 reduction goals by avoiding the emissions from the products that have been displaced (Nabuurs et al.
39 2007a; Lemprière et al. 2013). In 2007 the IPCC concluded that “[i]n the long term, a sustainable forest
40 management strategy aimed at maintaining or increasing forest carbon stocks, while producing an
41 annual sustained yield of timber, fibre or energy from the forest, will generate the largest sustained
42 mitigation benefit” (Nabuurs et al. 2007a). Initial landscape conditions, in particular the age-class
43 distribution and therefore C stocks of the landscape strongly affect the mitigation potential of forest
44 management options (Kilpeläinen et al. 2017). Landscapes with predominantly mature forests may
45 experience larger reductions in carbon stocks during the transition to managed landscapes (Harmon et
46 al. 2009; Kurz et al. 1998) while in landscapes with predominantly young or recently disturbed forests
47 sustainable forest management can enhance carbon stocks.

1 Sustainable forest management, including the intensification of carbon-focussed management
2 strategies, can make long-term contributions towards negative emissions if the sustainability of
3 management is assured through appropriate governance, monitoring and enforcement.

4.8 Impacts of land degradation on climate systems

6 While Chapter 2 has its focus on land cover changes and their impacts on the climate system, this
7 chapter focuses on the influences of individual land degradation processes on climate (see cross chapter
8 Table 4.1) which may or may not take place in association to land cover changes. The effects of land
9 degradation on CO₂ and other greenhouse gases as well as those on surface albedo and other physical
10 controls of the global radiative balance are discussed.

11 Globally, degradation process affecting cultivated systems are likely to have their dominant impacts on
12 climate through the release CH₄ and N₂O (*medium to high confidence*), whereas degradation processes
13 affecting natural and seminatural systems such as deforestation, increasing wild fires and permafrost
14 thawing display, in most cases, their highest warming impact through the net release of CO₂ (*medium
15 to high confidence*). Many degradation processes introduce additional physical effects such as albedo
16 changes and aerosol release that can be significant and often opposed to those of greenhouse gases
17 (*medium confidence*). The interacting effects call for more integrative climate impact assessments that
18 go beyond greenhouse gas balances.

4.8.1 Impacts on net CO₂ emissions

20 Land degradation processes with direct impact on soil and terrestrial vegetation have great relevance in
21 terms of CO₂ exchange with the atmosphere given the magnitude and activity of these reservoirs in the
22 global C cycle. As the most widespread form of soil degradation, erosion detaches the surface soil
23 material which typically hosts the highest organic C stocks, favoring the mineralisation and release as
24 CO₂. Precise estimation of the CO₂ released from eroded lands is challenged by the fact that only a
25 fraction of the detached C is eventually lost to the atmosphere. Laboratory experiments suggest that
26 more than 20% of the organic C of transported material can be released, leading to an estimated global
27 net source of 1 Pg C yr⁻¹ from land erosion (Lal 2003). It is important to acknowledge that a substantial
28 fraction of the eroded material may preserve its organic C load in field conditions. Moreover, C
29 sequestration may be favored through the burial of both the deposited material and the surface of its
30 hosting soil at the deposition location (Quinton et al. 2010). The cascading effects of erosion on other
31 environmental processes at the affected sites are more likely to cause net CO₂ emissions through their
32 indirect influence on soil fertility and the balance of organic C inputs and outputs, interacting with other
33 non-erosive soil degradation processes such as nutrient depletion, compaction and salinisation, which
34 can lead to the same net C effects (see Table 4.1) (van de Koppel et al. 1997).

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

41 Cultivation and agricultural management of cultivated land are relevant in terms of global CO₂ land-
42 atmosphere exchange. Besides the initial pulse of CO₂ emissions associated with the onset of cultivation
43 and associated vegetation clearing (see Chapter 2), agricultural management practices can increase or
44 reduce C losses to the atmosphere. Although global croplands are considered to be at relatively neutral
45 stage in the current decade (Houghton et al. 2012), this results from a highly uncertain balance between

1 coexisting net losses and gains. Degradation losses of soil and biomass carbon appear to be compensated
2 by gains from soil protection and restoration practices such as conservation tillage and nutrient
3 replenishment favoring organic matter build-up. Cover crops, increasingly used to improve soils, have
4 the potential to sequester 0.12 Pg C yr⁻¹ on global croplands with a saturation time of more than 150
5 years (Poeplau and Don 2015). It is important to highlight that the direct C-sequestration benefits of
6 no-till practices (i.e. tillage elimination favoring crop residue retention in the soil surface) which were
7 seen with high optimism at the onset of this technology appear uncertain after recent assessments (Van
8 den Bygaart 2016). Among soil fertility restoration practices, lime application for acidity correction,
9 increasingly important in tropical regions, can generate a significant net CO₂ source in some soils
10 (Bernoux et al. 2003, Alemu et al 2017).

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

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

29 **4.8.2 Impacts on non-CO₂ greenhouse gases**

30 Agricultural land represents a dominant source of non-CO₂ greenhouse gases. The expansion of rice
31 cultivation (increasing CH₄ sources), ruminant stocks and manure disposal (increasing CH₄ and N₂O
32 fluxes) and nitrogen over-fertilisation combined with soil acidification (increasing N₂O fluxes) are
33 introducing the major impacts (*medium to high confidence*) and their associated emissions appear to
34 be exacerbated by global warming (*medium to high confidence*) (Oertel et al. 2016; Brandt et al. 2018a).

35 As the major sources of global N₂O emissions, over-fertilisation and manure disposal are not only
36 increasing in-situ sources but also stimulating those along the pathway of dissolved inorganic N
37 transport all the way from draining waters to the ocean (*high confidence*). Current budgets of
38 anthropogenically fixed nitrogen on the Earth System (Tian et al. 2015; Schaefer et al. 2016; Wang et
39 al. 2017) suggest that N₂O release from terrestrial soils and wetlands accounts for 10-15% of the inputs,
40 yet many further release fluxes along the hydrological pathway remain uncertain, with emissions from
41 oceanic “dead-zones” being a major aspect of concern (Schlesinger 2009).

42 Hydrological degradations processes, which are typically manifested at the landscape scale, include
43 both drying (as in drained wetlands or lowlands) and wetting trends (as in waterlogged and flooded
44 plains). Drying of wetlands reduces CH₄ emissions (Turetsky et al. 2014) but favors pulses of organic
45 matter mineralisation linked to high N₂O release (Morse and Bernhardt 2013; Norton et al. 2011). The
46 net warming balance of these two effects is not resolved and may be strongly variable across different
47 types of wetlands. In the case of flooding of non-wetland soils, a suppression of CO₂ release is typically

1 out-compensated in terms of net greenhouse impact by enhanced CH₄ fluxes, that stem from
2 anaerobiosis but are aided by the direct effect of extreme wetting on the solubilisation and transport of
3 organic substrates (McNicol and Silver 2014). Both wetlands rewetting/restoration and artificial
4 creation can produce pulses of CH₄ release (Altor and Mitsch 2006; Fenner et al. 2011).

5 **4.8.3 Physical impacts**

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

12 In the case of forest degradation or deforestation, the albedo impacts are highly dependent on the
13 latitudinal/climatic belt to which they belong. In boreal forests the removal or degradation of the tree
14 cover increases albedo (net cooling effect, high confidence) as the reflective snow cover becomes
15 exposed, which can overwhelm the net radiative effect of the associated C release to the atmosphere
16 (Davin et al. 2010). On the other hand, progressive greening of boreal and temperate forests has
17 contributed to net albedo declines (Planque et al. 2017; Li et al. 2018). In the tundra, shrub
18 encroachment leads to the opposite effect as the emergence of plant structures above the snow cover
19 level reduce winter-time albedo (Sturm 2005).

20 The extent to which albedo shifts can compensate carbon storage shifts at the global level has not been
21 estimated. A significant but partial compensation takes place in temperate and subtropical dry
22 ecosystems in which radiation levels are higher and C stocks smaller compared to their more humid
23 counterparts (*medium confidence*). In cleared dry woodlands half of the net global warming effect of
24 net C release has been compensated by albedo decline (Houspanossian et al. 2013), whereas in
25 afforested dry rangelands albedo declines cancelled one fifth of the net C sequestration (Rotenberg and
26 Yakir 2010). Other important cases in which albedo effects impose a partial compensation of C
27 exchanges are the vegetation shifts associated to wild fires, as shown for the savannahs, shrublands and
28 grasslands of sub-Saharan Africa (Dintwe et al. 2017). Besides the net global effects discussed above,
29 albedo shifts can play a significant role on local climate, as exemplified by the effect of no-till
30 agriculture reducing local heat extremes in European landscapes (Davin et al. 2014) and the effects of
31 woody encroachment causing precipitation rises in the North American Great Plains (Ge and Zou 2013).

32 Beyond the albedo effects presented above, other physical impacts of land degradation on the
33 atmosphere can contribute to global and regional climate change. Of particular global relevance are the
34 cooling effects of dust emissions (*medium confidence*). Anthropogenic emission of mineral particles
35 from degrading land appear to have a similar radiative impact than all other anthropogenic aerosols
36 (Sokolik and Toon 1996). Dust emissions may explain regional climate anomalies through reinforcing
37 feedbacks, as suggested for the amplification of the intensity, extent and duration of the low
38 precipitation anomaly of the North American “Dust Bowl” in the 1930s (Cook et al. 2009). The complex
39 physical effects of deforestation, as explored through modeling, converge into general net regional
40 precipitation declines, tropical temperature increases and boreal temperature declines, while net global
41 effects are less certain (Perugini et al. 2017).

43 **4.9 Impacts of climate-related land degradation on people and nature**

44 Unravelling the impacts of climate-related land degradation on people and nature is highly complex.
45 This complexity is due to the interplay of multiple social, political, cultural, and economic factors which

1 shape the ways in which social-ecological systems respond (Morton 2007). Consequently, attribution
2 of the impacts of *climate-related land degradation* on people and nature is rarely undertaken within the
3 literature, with climate often not distinguished from any other driver of land degradation. Climate is
4 nevertheless frequently noted as a risk multiplier for land degradation and is one of many stressors
5 people live with, respond to and adapt to in their daily lives (Reid and Vogel 2006). Climate change is
6 considered to exacerbate land degradation and potentially accelerate it due to heat stress, drought,
7 changes to evapotranspiration rates and biodiversity, as well as resulting from changes to environmental
8 conditions that allow new pests and diseases to thrive (Reed and Stringer 2016). In general terms, the
9 climate (and climate change) can increase human and ecological communities' exposure and sensitivity
10 to land degradation, with land degradation then leaving livelihoods more sensitive to the impacts of
11 climate change and extreme climatic events. If human and ecological communities are exposed and
12 sensitive and cannot adapt, they can be considered vulnerable, while if they are exposed, sensitive and
13 can adapt, they may be considered resilient.

14 **4.9.1 Poverty, livelihood and vulnerability impacts of climate related land degradation**

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

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

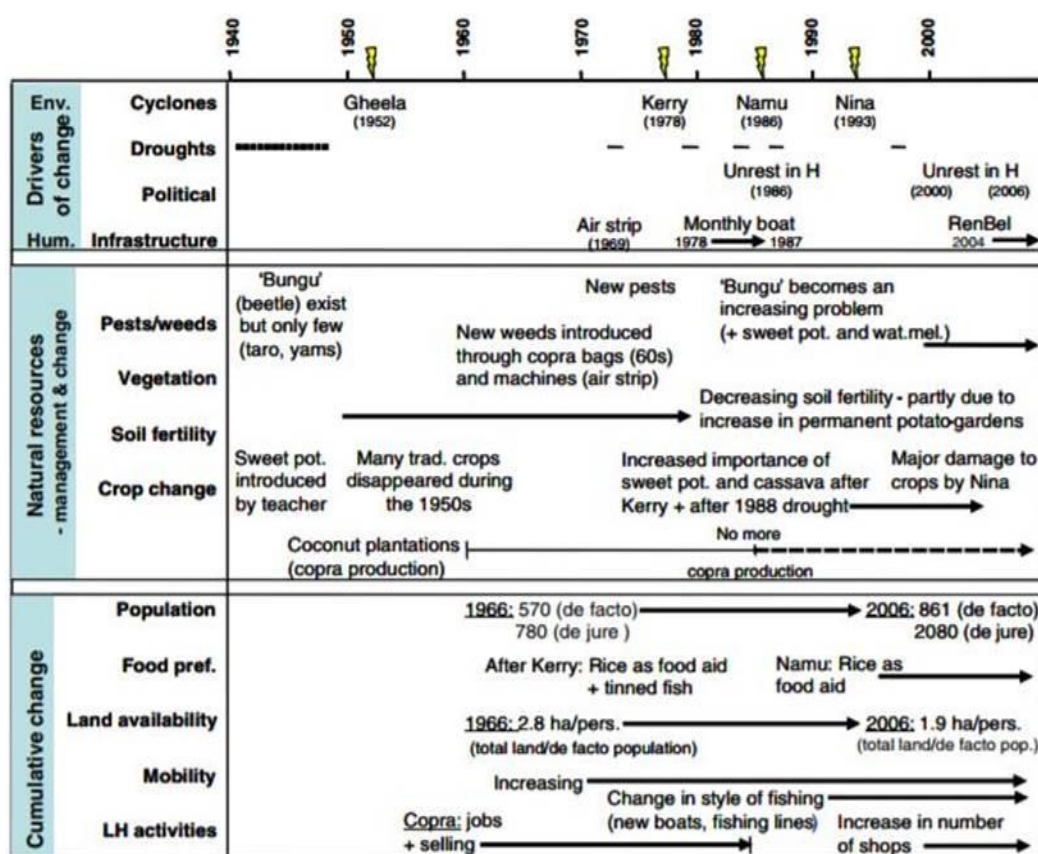
34 Climate and land degradation can both affect poverty and vulnerability through their threat multiplier
35 effect. Often the geographical locations experiencing degradation are the same locations that are directly
36 affected by poverty and vulnerability and are also affected by extreme events linked to climate change
37 and variability. Altieri et al (Altieri and Nicholls 2017) however note that due to the globalised nature
38 of markets and consumption systems, the impacts of changes in crop yields linked to climate related
39 land degradation (manifest as lower yields) will be far reaching, beyond the sites and livelihoods
40 experiencing degradation. Despite these teleconnections, farmers living in poverty in developing
41 countries will be especially vulnerable due to their exposure, dependence on the environment for income
42 and limited options to engage in other ways to make a living (Rosenzweig and Hillel 1998).

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

1 4.8); and conflict (see Section 4.8), as different groups within society compete for scarce resources,
 2 sometimes through non-peaceful actions. Both migration and conflict can lead to land use changes
 3 elsewhere that further fuel climate change through increased emissions as a result of, for example, land
 4 use change.

5 Some authors (Okpara et al. 2016a,b) note that a vulnerability lens can be a useful way to understand
 6 the impacts of environmental and social changes taking place, allowing a focus on livelihood systems
 7 and the interlinkages between poverty, vulnerability, livelihoods and environmental changes. A large
 8 body of literature is focused on understanding the livelihood and poverty impacts of degradation
 9 through a focus on subsistence agriculture, where farms are small, under traditional or informal tenure
 10 and where exposure to environmental (including climate) risks is high (Morton 2007). In these
 11 situations, the poor lack access to assets (financial, social, human, natural and physical) and in the
 12 absence of appropriate institutional supports and social protection, this leaves them sensitive and unable
 13 to adapt, so a vicious cycle of poverty and degradation can ensue.

14 Methodologically, timeline approaches can be useful in unpacking the links between climate, land
 15 degradation and other livelihood impacts over specific temporal scales. For example, research in
 16 Bellona, in the Solomon Islands in the south Pacific (Reenberg et al. 2008) has examined event-driven
 17 impacts on livelihoods, taking into account weather events as one of many drivers of land degradation
 18 and links to broader land use and land cover changes that have taken place (Figure 4.6).



19
 20 **Figure 4.6 Coupled human-environment timelines of Bellona, 1940s to 2006. Source: Group interviews**
 21 **with selected households from the general HH-questionnaire (Reenberg et al. 2008)**

22 In identifying ways in which these interlinkages can be addressed, (Scherr 2000) observes that key
 23 actions that can jointly address poverty and environmental improvement often seek to increase access
 24 to natural resources, enhance the productivity of the natural resource assets of the poor, and to engage
 25 stakeholders in addressing public natural resource management issues. In this regard, it is increasingly
 26 recognised that those suffering from and vulnerable to land degradation and poverty need to have a

1 voice and play a role in the development of solutions, especially where the natural resources and
2 livelihood activities they depend on are further threatened by climate change.

3 **4.9.2 Food Security**

4 How and where we grow food compared to where and when we need to consume it is at the crux of
5 issues surrounding land degradation, climate change and food security, especially because more than
6 75% of the global land surface (excluding Antarctica) faces rain-fed crop production constraints
7 (Fischer et al. 2009). Taken separately, knowledge on land degradation processes and human-induced
8 climate change has attained a great level of maturity. However, their combined effects on food security,
9 notably food supply, remain underappreciated (Webb et al. 2017). Just a few studies have shown how
10 the interactive effects of the aforementioned challenging, interrelated phenomena can impact crop
11 productivity and hence food security and quality (Karami et al. 2009; Allen et al. 2001; Högy and
12 Fangmeier 2008). Along with socio-economic drivers climate change accelerates land degradation due
13 to its influence on land use systems (Millennium Assessment 2005; UNCCD 2017), potentially leading
14 to a decline in agri-food systems, particularly on the supply side. Increases in temperature and changes
15 in precipitation patterns are expected to have impacts on soil quality, including nutrient availability and
16 assimilation (St.Clair and Lynch 2010). Those climate-related changes are expected to have net negative
17 impacts on agricultural productivity, particularly in tropical regions. Anticipated supply side issues
18 linked to land and climate relate to biocapacity factors (including e.g. whether there is enough water to
19 support agriculture); production factors (e.g. chemical pollution of soil and water resources or lack of
20 soil nutrients) and distribution issues (e.g. decreased availability of and/or accessibility to the necessary
21 diversity of quality food where and when it is needed) (Stringer et al. 2011). Climate sensitive transport
22 infrastructure is also problematic for food security, and can lead to increased food waste, while poor
23 siting of roads and transport links can lead to soil erosion and forest loss, further feeding back into
24 climate change.

25 Over the past decades, crop models have been useful tools for assessing and understanding of climate
26 change impacts on crop productivity and food security (White et al. 2011; Rosenzweig et al. 2014). Yet,
27 the interactive effects of soil parameters and climate change on crop yields and food security remain
28 limited, with few studies assessing how they play out in different economic and climate settings (e.g.
29 Sundström et al. 2014). Similarly, there have been few meta-analyses focusing on the adaptive capacity
30 of land use practices such as conservation agriculture in light of climate stress (see e.g. (Steward et al.
31 2018). To be sustainable, any initiative aiming at addressing food security – encompassing supply,
32 diversity and quality - must take into consideration the interactive effects between climate and land
33 degradation in a context of other socio-economic stressors. Such socio-economic factors are especially
34 important if we look at demand side issues too, which include lack of purchasing power, competition
35 in appropriation of supplies and changes to per capita food consumption (Stringer et al. 2011). Lack of
36 food security, combined with lack of livelihood options, is often an important manifestation of
37 vulnerability, and can act as a key trigger for people to migrate. In this way, migration becomes an
38 adaptation strategy.

39 **4.9.3 Impacts of climate related land degradation on migration and conflict**

40 Land degradation may trigger competition for scarce natural resources potentially leading to migration
41 and/or conflict. However, linkages between land degradation and migration occur within a larger
42 context of multi-scale interaction of environmental and non-environmental drivers and processes,
43 including resettlement projects, search for education and/or income, land shortage, political turmoil,
44 and family-related reasons (McLeman 2017). Consequently, impacts of land degradation on migration
45 are largely context-specific and vary across the globe.

1 One of the global ecosystems that is particularly prone to climate-related land degradation is drylands,
2 which is mainly because of its fragile ecological conditions (UNCCD 2017). A significant share of
3 existing studies on land degradation and migration – particularly in drylands – focuses on the effect of
4 rainfall variability and drought and shows how migration serves as adaptation strategy (Piguet et al.
5 2018; McLeman 2017). For example, in the dry Ethiopian highlands severe soil erosion and forest
6 degradation is a major environmental stressor which is amplified by re-occurring droughts, with
7 migration being an important adaptation strategy (Morrissey 2013). In the humid tropics, land
8 degradation, mainly as a consequence of deforestation, has been a reported reason for people leaving
9 their homes during the Amazonian colonisation (Hecht and B. 1983) but was also observed more
10 recently, for example in Guatemala (López-Carr 2012). In contrast, in Equator migration increased with
11 land quality, likely because increased land production was invested in costly forms of migration (Gray
12 and Bilsborrow 2013). These mixed results illustrate the complex, non-linear relationship of land
13 degradation-migration linkages and suggest explaining land degradation-migration linkages requires
14 considering a broad socio-ecological embedding.

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

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

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

47 Land degradation-conflict linkages can be bi-directional. Research suggests that households
48 experiencing natural resource degradation often, although not always, engage in migration for securing

1 livelihoods (Kreamer 2012), which potentially triggers land degradation at the destination leading to
2 conflict there (Kassa et al. 2017). While this indeed holds true for some cases it may not for others given
3 the complexity of processes, contexts and drivers. Where conflict and violence do ensure, it is often as
4 a result of a lack of appreciation for the cultural practices of others.

5 **4.9.4 Impacts of climate related land degradation on cultural practices**

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

12 Studies on resolving the long-term and structural problems created by climate- related land degradation
13 have revealed that the loss of cultural identity and heritage is a significant impact in a variety of contexts.
14 Bush encroachment in Kalahari rangelands in southwest Botswana has adversely affected the forage
15 availability and production of cattle, which play a central role in Batswana culture (Reed and Stringer
16 2016; Reed 2005). In Swaziland, droughts followed by heavy rainfall events cause soil erosion and loss
17 of soil fertility and this land degradation is considered a threat to Swazi culture, in which soil has cultural
18 and spiritual value (Stringer and Reed 2007). The indicators for cultural impacts of climate-related land
19 degradation vary regionally due to differences in climate, culture, soil and ecology in different parts of
20 the world. In Australia the increasing frequency of drought and its effect on productivity of land has
21 increased the levels of farmer suicide, particularly among men (Alston 2011) but ultimately affecting
22 whole communities. Some positive trends have nevertheless emerged from the situation: recognition of
23 the importance of managing land degradation and the need to adapt to climate change has galvanised
24 community action by local land care groups, undertaking for example, revegetation projects,
25 counteracting to some extent the negative social impacts of climate related land degradation.

26 The nature of people's relationship with land is also determined by gender norms. Where land-based
27 livelihoods are concerned, typically men have privileged control over the benefits from production (for
28 example in terms of commercialisation), whilst women are responsible for household reproduction and
29 ensuring food security, although the socially constructed norms and behaviors expected by men and
30 women differ from place to place (Carr 2008; Doss 2002; Gladwin et al. 2001; Bryceson 1995; Dixon
31 1982). This is both reflected in, and reinforced by, gender differences in land tenure that are pervasive
32 in the development world (e.g. (Agarwal 2003). However, it is not just gender but also other social
33 identifiers that play a role – such that in Malawi age was found to be as important as gender in
34 determining access to land (Carr and Thompson 2014). An intersectional approach to analysis is thus
35 now preferred to a binary dichotomy between men and women, recognising that it is the intersection of
36 many social identifiers that creates the norms that govern practices and, thus, in turn the effects of land
37 degradation (Djoudi et al. 2016; Thompson-Hall et al. 2016).

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

1 **4.9.5 Impacts of climate related land degradation on nature**

2 Other feedbacks between climate related land degradation on nature relate to amplified impacts of
3 extreme events on the environment. For example, mangroves, which are found in 123 countries around
4 the world, are being degraded at a phenomenal rate, with 20-35% of global mangrove area lost since
5 1980 (Polidoro et al. 2010). This forest degradation leaves coastlines largely unprotected as mangroves
6 act as ‘first line’ defences. As the frequency of hurricanes, storms and typhoons increases with climate
7 change, greater areas of land are threatened by storm surges. In turn, this can exacerbate degradation
8 issues such as saline intrusion and erosion.

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

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

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

37

38 **4.10 Addressing/targeting land degradation in the context of climate change**

39 Land users and managers have addressed land degradation in many ways since time immemorial.
40 Before the advent of modern sources of nutrients it was imperative for farmers to maintain and improve
41 soil fertility through the prevention of runoff and erosion, and management of nutrients through
42 vegetation residues and manure. Many ancient farming systems were sustainable for hundreds and even
43 thousands of years, such as raised field agriculture in Mexico (Crews and Gliessman 1991), tropical
44 forest gardens in SE Asia and Central America (Ross 2011; Torquebiau 1992; Turner and Sabloff 2012),
45 terraced agriculture in Central America, SE Asia and the Mediterranean basin (Turner and Sabloff 2012;
46 Preti and Romano 2014), and integrated rice-fish cultivation in East Asia (Frei and Becker 2005).

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

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

17 Sustainable Land Management (SLM) is a unifying framework for addressing land degradation and can
18 be defined as ‘a system of technologies and/or planning that aims to integrate ecological with socio-
19 economic and political principles in the management of land for agricultural and other purposes to
20 achieve intra- and intergenerational equity’ (Hurni 2000). It is a comprehensive approach comprising
21 technologies combined with social, economic and political enabling conditions (Nkonya et al. 2011). It
22 is important to stress that farming systems are often shot through with local/traditional knowledge. The
23 power of SLM in small-scale diverse farming was demonstrated effectively in Nicaragua after the
24 severe hurricane Mitch in 1998 (Holt-Giménez 2002). Pairwise analysis of 880 fields with and without
25 implementation of SLM practices showed that the SLM fields systematically fared better than the fields
26 without SLM in terms of more topsoil remaining, higher field moisture, more vegetation, less erosion
27 and lower economic losses after the hurricane. Furthermore the difference between fields with and
28 without SLM increased with increasing levels of storm intensity, increasing slope gradient, and
29 increasing age of SLM (Holt-Giménez 2002).

30 When addressing land degradation through SLM and other approaches, it is important to consider how
31 these approaches feedback onto the atmosphere and impact climate change. Table 4.2 shows how some
32 of the most important land degradation issues are linked to their potential solutions. This table provides
33 a link between the comprehensive lists of land degradation processes (Table 4.1) and land management
34 solutions (Table 4.2).

1 **Table 4.2 Table linking key land degradation issues (how they impact climate change and what the key**
 2 **drivers are) with potential solutions and how these solutions interact with climate change**

Key LD issue	Its impact on CC	Drivers			potential land management options (Table 6.3)	Response options' impact on climate change					Comments
		human driver	direct climate driver	indirect climate change driver		M	A	D	L	F	
degradation of agricultural soils					increase soil organic matter, no till, perennial crops, erosion control, agroforestry	↑	↑	↑	↑	↑	Dietary change and reduction of food waste would have a profound effect
overgrazing					controlled grazing, rangeland management	↑	↑	↑	↑	↑	
firewood and charcoal production					clean cooking	↑	↑	↑	↑	↑	important health co-benefits, particularly for women and children.
increasing fire frequency and intensity					thinning, fire management	↑	↑	↑	↑		
degradation of tropical peat soils					Peatland restoration, erosion control	↑	↑		↑	↓	regulating the use of peat soils would have a profound effect
melting of perma-frost					None						
coastal erosion					wetland and coastal restoration, mangrove conservation	↑	↑		↑	↑	in the long term land use planning, including planned retraction, is essential
bush encroachment					grazingland management, fire management	↓	↑	↑	↑	↓	

3

warming trend		grazing pressure		emission of GHG	
extreme temperatures		bio-energy		emission of GHG	
drying trend		bio-fuel		emission of GHG	
extreme rainfall		farming practices		albedo increase	
shifting rains		expansion of farming		albedo decrease	
hurricanes		forest clearing		carbon sink	
sea level rise		wood fuel			

4

1 **4.10.1 Actions on the ground to address land degradation**

2 **4.10.1.1 Local and indigenous knowledge**

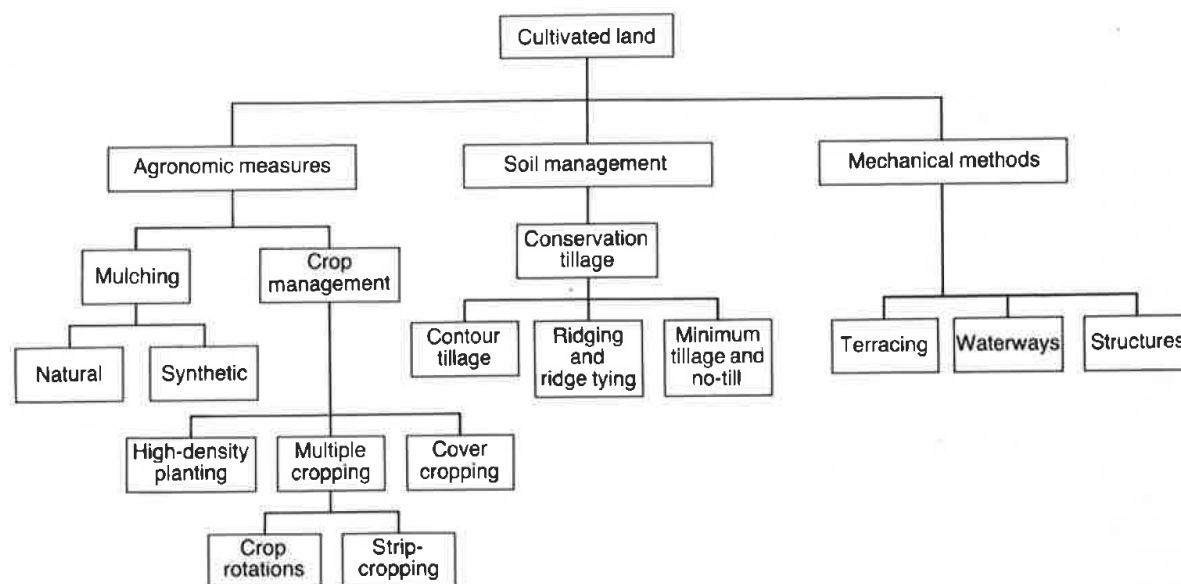
3 Responses to land degradation generally take the form of agronomic, vegetative, structural and
4 management measures (Schwilch et al. 2011). These include inter alia mulching, contour ploughing, pit
5 planting, intercropping, agroforestry, hedging, grass strips, bunds, terraces, dams, area closures, shifting
6 cultivation, rotational grazing and so on. Measures may be combined to reinforce benefits to land
7 quality, as well as improving carbon sequestration that supports climate change mitigation. Some
8 measures offer adaptation options and other co-benefits, especially if e.g. agroforestry involves planting
9 fruit trees that can support food security in the face of climate change impacts (Reed, M.S.; Stringer
10 2016).

11 In practice, responses are anchored both in scientific research, as well as local, indigenous and
12 traditional knowledge and know-how. For example, studies in the Philippines (Camacho et al. 2016)
13 (Camacho et al., 2016) examine how traditional integrated watershed management by indigenous
14 people sustain regulating services vital to agricultural productivity, while delivering co-benefits in the
15 form of biodiversity and ecosystem resilience at a landscape scale. Although responses can be site
16 specific and sustainable at a local scale, the multi-scale interplay of drivers and pressures can
17 nevertheless cause practices that have been sustainable for centuries to become less so. Siahaya et al
18 (2016) explore the traditional knowledge that has informed rice cultivation in the uplands of East
19 Borneo, grounded in sophisticated shifting cultivation methods (*gilir balik*) which have been passed on
20 for generations (more than 200 years) in order to maintain local food production. *Gilir balik* involves
21 temporary cultivation of plots, after which, abandonment takes place as the land user moves to another
22 plot, leaving the natural (forest) vegetation to return. This approach is considered sustainable if it has
23 the support of other subsistence strategies, adapts to and integrates with the local context, and if the
24 carrying capacity of the system is not surpassed (Siahaya et al. 2016). Often *gilir balik* cultivation
25 involves intercropping of rice with bananas, cassava and other food crops. Once the abandoned plot has
26 been left to recover such that soil fertility is restored, clearance takes place again and the plot is reused
27 for cultivation. Rice cultivation in this way plays an important role in forest management, with several
28 different types of succession forest being found in the study area of Siahaya et al (2016). Nevertheless,
29 interplay of these practices with other pressures (large-scale land acquisitions for oil palm plantation,
30 logging and mining), risk their future sustainability. Use of fire is critical in processes of land clearance,
31 so there are also trade-offs for climate change mitigation which have been sparsely assessed.

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

45 **4.10.1.2 Soil conservation technologies**

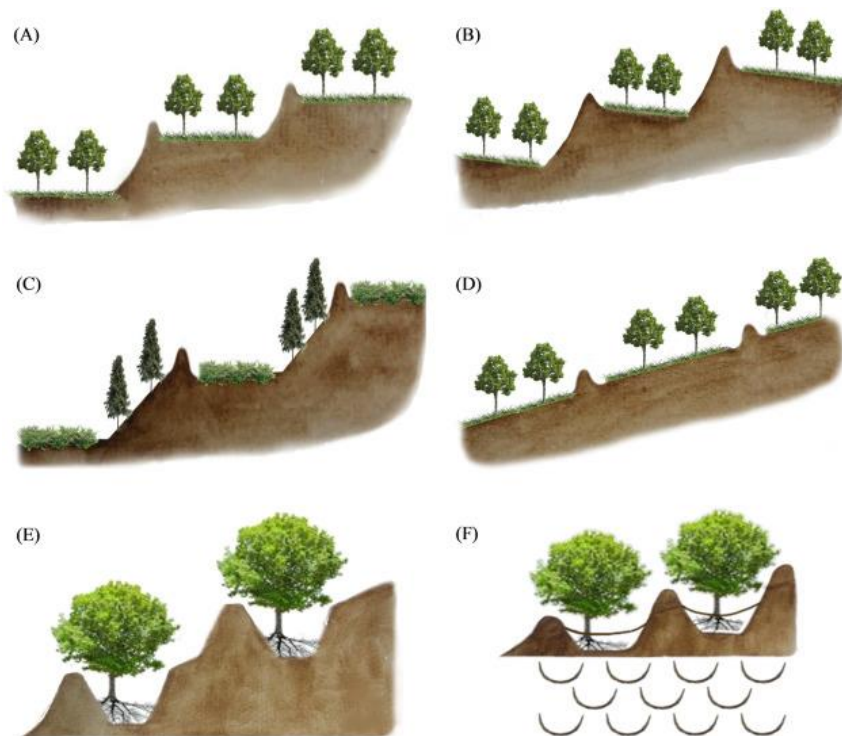
46 Technologies for SLM on agricultural land include soil conservation which can be systematised as in
47 Figure 4.7.



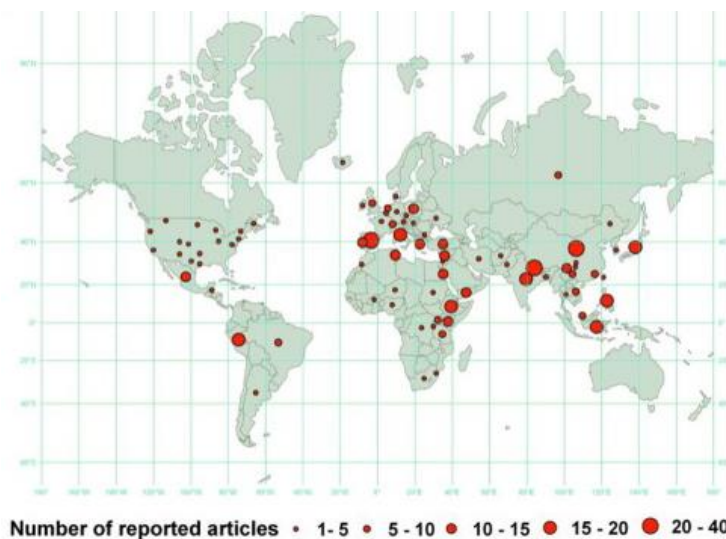
1
2 **Figure 4.7 Technologies for soil conservation in cultivated lands (Morgan 2005)**

3 There are important differences in terms of labour and capital requirements for different technologies,
 4 and also implications for land tenure arrangements. Agronomic measures require generally little extra
 5 capital input and comprise activities repeated annually, hence no particular implication for land tenure
 6 arrangements. Mechanical methods require substantial upfront investments in terms of capital and
 7 labour, resulting in long lasting structural change requiring more secure land tenure arrangements.
 8 Agroforestry is a particularly important strategy for SLM in the context of climate change because the
 9 large potential to sequester carbon in plants and soil.

10 In hilly and mountainous terrain terracing is an ancient but still practiced soil conservation method
 11 worldwide (Preti and Romano 2014) in climatic zones from arid to humid tropics (Balbo 2017), Figure
 12 4.8 and 4.9. By reducing the slope gradient of hillsides terraces provide flat surfaces and deep, loose
 13 soils that increase infiltration, reduce erosion and thus sediment transport. They also decrease the
 14 hydrological connectivity and thus reduce hillside runoff (Preti et al. 2018; Wei et al. 2016; Arnáez et
 15 al. 2015; Chen et al. 2017). In terms of climate change, terraces are a form of adaptation which helps
 16 both in cases where rainfall is increasing or intensifying, by reducing slope gradient and the
 17 hydrological connectivity, and where rainfall is decreasing, by increasing infiltration and reducing
 18 runoff (*high agreement, robust evidence*). There are several challenges, however, to continued
 19 maintenance and construction of new terraces, such as the high costs in terms of labour and/or capital
 20 (Arnáez et al. 2015) and disappearing local knowledge for maintaining and constructing new terraces
 21 (Chen et al. 2017).



1
2 **Figure 4.8** Examples of terracing types based on differences in structure, A: level terraces; B: zig
3 terraces; C: slope separated terraces; D: slope terraces. (Chen et al. 2017)



6
7 **Figure 4.9** Distribution of terraced agriculture based on systematic analysis of published scientific articles
8 (Wei et al. 2016)

9 **4.10.1.3 Agroforestry**

10 Agroforestry is defined as a collective name for land-use systems in which woody perennials (trees,
11 shrubs, etc.) are grown in association with herbaceous plants (crops, pastures) and/or livestock in a
12 spatial arrangement, a rotation or both, and in which there are both ecological and economic interactions
13 between the tree and non-tree components of the system (Young, 1995, p. 11). At least since the 1980s
14 agroforestry has been widely touted as an ideal land management practice in areas vulnerable to climate
15 variations and subject to soil erosion. Agroforestry holds the promise of improving of soil and climatic

1 conditions while generating income from wood energy, timber, and non-timber products – sometimes
2 presented as a synergy of adaptation and mitigation of climate change (Mbow et al. 2014).

3 Through a combination of forestry with agricultural crops and/or livestock, agroforestry systems can
4 provide additional ecosystem services when compared with monoculture crop systems (Waldron et al.
5 2017). Agroforestry can enable sustainable intensification by allowing continuous production on the
6 same unit of land with higher productivity without the need to use shifting agriculture systems to
7 maintain crop yields (Nath et al. 2016). This is especially relevant where there is a regional requirement
8 to find a balance between the demand for increased agricultural production and the protection of
9 adjacent natural ecosystems such as primary and secondary forests (Mbow et al. 2014). For example,
10 the use of agroforestry for crops such as coffee and cocoa are increasingly promoted as offering a route
11 to sustainable farming with important climate change adaptation and mitigation co-benefits (Sonwa et
12 al. 2001; Kroeger et al. 2017). Reported co-benefits of agroforestry in cocoa production include
13 increased carbon sequestration in soils and biomass, improved water and nutrient use efficiency and the
14 creation of a favourable micro-climate for crop production (Sonwa et al. 2017). Importantly, the
15 maintenance of soil fertility using agroforestry has the potential to reduce the occurrence of shifting
16 cocoa-agriculture which results deforestation (Gockowski and Sonwa 2011). However, positive
17 interactions within these systems can be ecosystem and/or species specific, and co-benefits such as
18 increased resilience to extreme climate events, or improved soil fertility are not always observed (Blaser
19 et al. 2017; Abdulai et al. 2018). These contrasting outcomes indicate the importance of field scale
20 research programs to inform agroforestry system design, species selection and management practices
21 (Sonwa et al. 2014) .

22 Despite the many proven benefits adoption of agroforestry has been (s)low (Toth et al. 2017; National
23 Research Centre for Agroforestry et al. 1999; Pattanayak et al. 2003; Jerneck and Olsson 2014). Reasons
24 for the (s)low adoption are many, but many reasons are related to the perception of risks and the time
25 lag between adoption and realisation of benefits (Pattanayak et al. 2003; Mercer 2004).

26 An important question for agroforestry is whether it supports poverty alleviation, or if it favours
27 comparatively affluent households. Experiences from India suggest that the overall adoption is (s)low,
28 18% of the area suitable for agroforestry in a village in South India, but among the relatively rich
29 households who adopted agroforestry, 97% of them were still practicing agroforestry after 6-8 years
30 and some had expanded their operations (Brockington et al. 2016). Similar results were obtained in
31 Western Kenya, that food secure households were much more likely to adopt agroforestry than food
32 insecure households (Jerneck and Olsson 2013, 2014). Other experiences from Western Kenya illustrate
33 the difficulties of having a continued engagement of communities in agroforestry (Noordin et al. 2001).

34 ***4.10.1.4 Crop-livestock interaction as an approach to manage land degradation***

35 The integration of crop and livestock production into “mixed farming” has been promoted among
36 smallholders and others in developing countries, particularly the sub-humid zone of Africa, since the
37 late 1980s, after the mixed farming model came to be seen as outdated in most industrialised countries
38 (Scoones and Wolmer 2002). Mixed farming was seen as an improvement in terms of both efficiency
39 and sustainability (Pritchard et al. 1992) and its promotion driven by the perceived need to increase
40 food production while not compromising environmental sustainability, in a context of increasing human
41 and livestock populations, and increasing food demands (Scoones and Wolmer 2002). It was also driven
42 by views of mobile pastoralism as environmentally damaging. Crop-livestock integration under this
43 model was seen as founded on three pillars; improved use of manure for crop fertility management;
44 expanded use of animal traction; and promotion of cultivated fodder crops (Scoones and Wolmer 2002),
45 as well as indirectly through increased cash liquidity through livestock sales and improved household
46 nutrition (Powell et al. 1998). Integration was seen as natural, progressive and inevitable, but also to
47 be assisted by policy, including formal research and extension (Ramisch et al. 2002).

1 More recent research gives a much more differentiated picture of crop livestock interactions, which can
2 take place either within a single farm household, or between crop and livestock producers, in which
3 case they will be mediated by formal and informal institutions governing the allocation of land, labour
4 and capital, with the interactions evolving through multiple place-specific pathways (Scoones and
5 Wolmer 2002; Ramisch et al. 2002). Developing understandings of pastoralism, mobility and
6 communal tenure of grazing lands (for example (Ellis 1994; Behnke 1994) lessened the imperative to
7 sedentarise pastoral production for ecological reasons and strengthened criticisms of the mixed farming
8 model as a privileged development strategy.

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

17 **4.10.1.5 Forestry based activities (afforestation, reforestation, and changing forest management)**

18 China has the largest afforested area in the world. Afforestation not only contributes to increased carbon
19 storage but also alters local albedo and turbulent energy fluxes, which offers feedback on the local and
20 regional climate. Afforestation decreases daytime land surface temperature (LST), because of enhanced
21 evapotranspiration, and increases night-time LST. This night-time warming tends to offset daytime
22 cooling in dry regions. These results suggest it is necessary to carefully consider where to plant trees to
23 achieve potential climatic benefits in future afforestation projects (Peng et al. 2014).

24 The influence of forest cover was confirmed by comparing observed runoff trends with those resulting
25 from a hydro-climatic model that does not take into account land use changes. Divergence between both
26 trends revealed that forest cover can account for about 40% of the observed decrease in annual runoff
27 (Buendia et al. 2016). Bárcena and co-authors (2014) concluded that significant SOC sequestration in
28 Northern Europe occurs after afforestation of croplands and not grasslands, and changes are small
29 within a 30-year perspective. Successful programmes of large scale afforestation activities in South
30 Korea and China are discussed in-depth a special case study (Section 4.11)

31 The huge economic and social changes happening in Eastern Europe in the 1990s and 2000s have
32 generated a clear trend towards afforestation of lands that were formerly cultivated. The SOC changes
33 from grassland to forest use in these areas (50 t ha^{-1}) represented a doubling in stocks in less than
34 40 years, and their impact on a global scale could represent an unexpected C sink. (Menichetti et al.
35 2017).

36 **4.10.2 Higher-level responses to land degradation**

37 At the World Summit on Sustainable Development (WSSD) in September 2002, land degradation was
38 highlighted as one of the major global environmental and sustainable development challenges of the
39 21st century. There was a thrust on developing projects in related areas and many projects were initiated
40 by the Food and Agricultural Organisation (FAO) and Global Environmental Facility (GEF) with the
41 objective to define methodologies for assessments, data collection and implementation.

42 International efforts to achieve sustainable development heavily influence national laws and policies.
43 For example, international conventions, such as the United Nations Convention to Combat
44 Desertification (UNCCD), have spurred efforts to prevent or mitigate land degradation in dry areas. The
45 clean development mechanism and other global sustainable development efforts have also catalyzed a

1 rapid rise in the global carbon trade. The international community should take advantage of its influence
2 by creating a global initiative to prevent degradation of the world's land and soil.

3 Sustainable land management (SLM) is proposed as a unifying theme for current global efforts on
4 combating desertification, climate change and loss of biodiversity and thereby benefits the other
5 conventions, the UNFCCC and CBD (Thomas 2008). Launched at the Rio Earth Summit in 1992, the
6 UNCCD only became effective in December 1996, with 50 signatory countries having ratified the
7 convention. The UNCCD outlines all countries for mainstreaming land degradation in development
8 processes. It emphasizes promoting sustainable land use that attempts to link poverty reduction on one
9 hand and environmental protection on the other. In this context and UNCCD debates on land
10 degradation neutrality (LDN), there is increasing awareness in both policy and research arenas to
11 identify the interrelationships between actions that successfully address LDN, as well as facilitate land-
12 based adaptation to climate change, climate change mitigation and sustainable development.

13 Land degradation neutrality (LDN) was introduced by the UNCCD at Rio +20, and adopted at UNCCD
14 COP12 (UNCCD 2016a). LDN is defined as "a state whereby the amount and quality of land resources
15 necessary to support ecosystem functions and services and enhance food security remain stable or
16 increase within specified temporal and spatial scales and ecosystems". Pursuit of LDN requires effort
17 to avoid further net loss of the land-based natural capital relative to a reference state, or baseline. LDN
18 encourages a dual-pronged effort involving sustainable land management to reduce the risk of land
19 degradation, combined with efforts in land restoration and rehabilitation, to maintain or enhance land-
20 based natural capital, and its associated ecosystem services (UNCCD/Science-Policy-Interface 2016;
21 Singh et al. 2012; Cowie, A.; Woolf, A.D.; Gaunt, J.; Brandão, M.; de la Rosa 2015) Planning for LDN
22 involves projecting the likely cumulative impacts of land use and land management decisions, then
23 counterbalancing anticipated losses with measures to achieve equivalent gains, within individual land
24 types (where land type is defined by land potential). Under LDN framework developed by UNCCD,
25 three primary indicators are used to assess whether LDN is achieved by 2030: land cover change, net
26 primary productivity and soil organic carbon (Ellbehri et al. 2017). Achieving LDN therefore requires
27 integrated landscape management that seeks to optimize land use to meet multiple objectives
28 (ecosystem health, food security, human well-being). The response hierarchy of Avoid > Reduce >
29 Reverse land degradation articulates the priorities in planning LDN interventions. LDN provides the
30 impetus for widespread adoption of SLM and efforts to restore or rehabilitate land. Through its focus
31 LDN ultimately provides tremendous potential for mitigation of and adaptation to climate change by
32 halting and reversing land degradation and transforming land from a carbon source to a sink. Soils have
33 a capacity to sequester around 1–3 billion tons of CO₂ yr⁻¹ and land as a whole can sequester 7–11
34 billion tons of CO₂ yr⁻¹. There are strong linkages that the concept of LDN has with the Nationally
35 Determined Contributions (NDCs) of many countries with linkages to national climate plans. LDN is
36 also closely related to many SDGs in the areas of poverty, food security, environmental protection and
37 sustainable use of natural resources (UNCCD 2016b).

38 The UNFCCC has the National Adaptation Plans that enables countries to medium- and long-term
39 adaptation needs. It is a continuous, progressive and iterative process which follows a country-driven,
40 gender-sensitive, participatory and fully transparent approach. These programs have the potential of
41 assisting the promotion of SLM. It is realized that the root causes of vulnerability to multiple stresses
42 in land with successful adaptation in these plans will not be realised until holistic solutions to land
43 management are explored. SLM will help address root causes of low productivity, land degradation,
44 loss of income generating capacity as well as contribute to the amelioration of the adverse effects of
45 climate change.

46 The "4 per 1000" initiative (<https://www.4p1000.org/>) aims at capturing CO₂ from the atmosphere
47 through changes to agricultural and forestry practices at a rate that would increase the carbon content
48 of soils by 0.4% per year. If global soil carbon content increases at this rate in the top 30–40 cm, the

1 annual increase in atmospheric CO₂ would be stopped (Dignac et al. 2017). This is an illustration of
2 how extremely important soils are for addressing climate change.

3 In June 2014, 17 SDGs were proposed along with a set of 169 targets. Goal 15 is of direct relevance to
4 land degradation with the objective to protect restore and promote sustainable use of terrestrial
5 ecosystems, sustainably manage forests and combat desertification and halt and reverse land
6 degradation and halt biodiversity loss. For other global concerns SLM will increase productivity and
7 income thereby helping to achieve the SDGs. Examples where progress has been made in reversing
8 land degradation include reforestation of degraded lands and improved soil and water conservation
9 drawing on local knowledge. Target 15.3 specifically addresses land degradation neutrality. There are
10 other goals that also explicitly mention the focus on land, including goal 2, on ending hunger and
11 achieving food security, and goals 3, 7, 11 and 12. Further goals from a cross-cutting nature include 1,
12 6 and 13. It is to be seen how these interconnections are dealt with in practice.

13 Market based mechanisms like the Clean Development Mechanism (CDM) and the Voluntary Carbon
14 Market (VCM) provide platforms for carbon sinks. Implications on local land use and food security
15 have been raised as a concern and need to be assessed (Edstedt and Carton 2018; Olsson et al. 2014).
16 Besides REDD+ projects are being planned and implemented primarily targeting countries with high
17 forest cover and high deforestation rates. Some parameters of incentivizing emissions reduction, quality
18 of forest governance, conservation priorities, local rights and tenure frameworks, and sub-national
19 project potential are being looked into with often very mixed results (Newton et al. 2016; Gebara and
20 Agrawal 2017), see further Section 4.10.4.

21 **4.10.2.1 Limits to adaptation**

22 SLM can be deployed as a powerful adaptation strategy in most instances of climate change impacts on
23 natural and social systems, yet there are limits to adaptation. Exceeding adaptation limits will trigger
24 escalating losses or require undesirable transformational change, such as forced migration. The rate of
25 change in relation to the rate of possible adaptation is crucial (Dow et al. 2013). How limits to adaptation
26 are defined and how they can be measured is contextual and contested. Limits must be assessed in
27 relation to the ultimate goals of adaptation, which is subject to diverse and differential values (Dow et
28 al. 2013; Adger et al. 2009). A particularly sensitive issue is whether migration is accepted as adaptation
29 or not (Black et al. 2011; Tacoli 2009). In the context of land degradation potential limits to adaptation
30 exist if land degradation becomes so severe and irreversible that livelihoods cannot be maintained, and
31 if migration is either not acceptable or possible. Examples are coastal erosion where land disappears,
32 thawing of permafrost, desiccation, and extreme forms of soil erosion (e.g., gully erosion leading to
33 badlands (Poesen et al. 2003)).

34 **4.10.3 Resilience and tipping points**

35 Resilience refers to the capacity of a system, such as a farming system, to absorb disturbance (e.g.,
36 drought, conflict, market collapse), and recover, to retain the same identity. It can be described as
37 “coping capacity”. The disturbance may be a shock - sudden events such as a flood or disease epidemic
38 – or it may be a trend that develops slowly, like a drought or market shift. The shocks and trends
39 anticipated to occur due to climate change are expected to exacerbate risk of land degradation.
40 Therefore, assessing and enhancing resilience to climate change is a critical component of designing
41 sustainable land management strategies.

42 Resilience as an analytical lens is particularly strong in ecology and related research on natural resource
43 management (Folke et al. 2010; Quinlan et al. 2016) while in the social sciences the relevance
44 appropriateness of resilience for studying social and ecological interactions is contested (Cote and
45 Nightingale 2012; Olsson et al. 2015; Cretney 2014; Béné et al. 2012; Joseph 2013).

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

9 Applying resilience concepts involves viewing the land as a component of an interlinked social-
10 ecological system; identifying key relationships that determine system function and vulnerabilities of
11 the system; identifying thresholds or tipping points beyond which the system may transition to an
12 undesirable state; and devising management strategies to steer away from thresholds of potential
13 concern, thus facilitating healthy systems and sustainable production.

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

25 In managing land degradation, it is important to assess the resilience of the existing system, and the
26 proposed management interventions. If the existing system is in an undesirable state or considered
27 unviable under expected climate trends, it may be desirable to promote adaptation or even
28 transformation to a different system that is more resilient to future changes. For example, in an irrigation
29 district where water shortages are predicted, measures could be implemented to improve water use
30 efficiency, for example by establishing drip irrigation systems for water delivery, but transformation to
31 pastoralism or mixed dryland cropping/livestock production may be more sustainable in the longer term,
32 at least for part of the area. Application of sustainable land management practices especially those
33 focussed on ecological functions (e.g., agroecology, ecosystem-based approaches, regenerative
34 agriculture) can be effective in building resilience of agro-ecosystems (Henry et al. 2018). The essential
35 features of a resilience approach to management of land degradation under climate change are described
36 by (O’Connell et al. 2016; Simonsen et al. 2014).

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

42 **4.10.4 Barriers to implementation**

43 There is a growing recognition that addressing barriers and designing solutions to complex
44 environmental problems, such as land degradation, requires awareness of the larger system into which
45 the problems and solutions are embedded (Laniak et al. 2013). An ecosystem approach to SLM based
46 on understanding of the processes of land degradation has been recommended that can separate multiple
47 drivers, pressures and impacts (Kassam et al. 2013), but large uncertainty in model projections of future

1 climate, and associated ecosystem processes (IPCC 2013a) pose additional challenges to the
2 implementation of SLM. As discussed earlier in this chapter, many SLM practices, including both
3 technologies and approaches, are available that can increase yields and contribute to closing the yield
4 gap between actual and potential crop or pasture yield, while also enhancing resilience to climate change
5 (Yengoh and Ardö 2014; WOCAT). However, there are often systemic barriers to adoption and scaling
6 up of SLM practices, especially in developing countries.

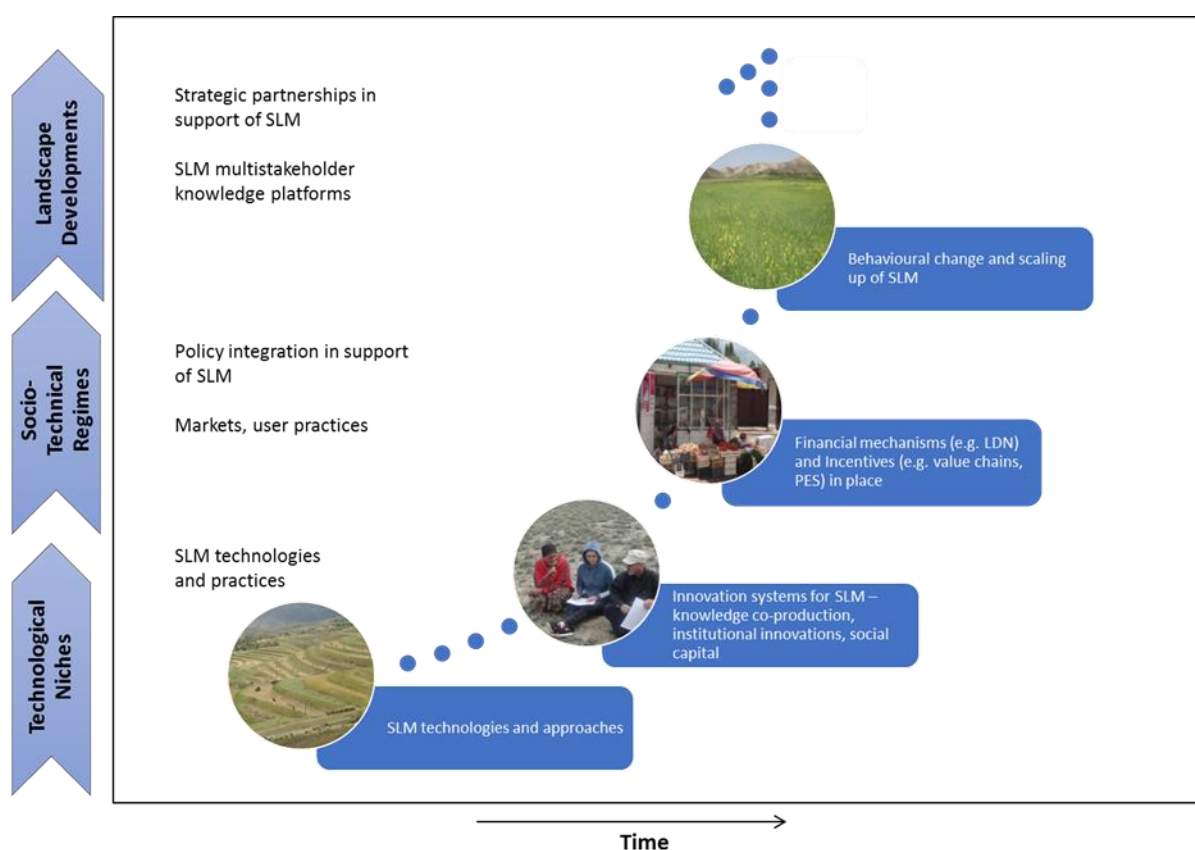
7 Uitto (2016) identified areas of contribution that the GEF, the financial mechanism of the UNCCD,
8 UNFCCC and other multilateral environmental agreements, can address to solve global environmental
9 problems. This includes removal of barriers related to knowledge and information; strategies for
10 implementation of technologies and approaches; and institutional capacity. Strengthening these areas
11 would drive transformational change leading to broader adoption of sustainable environmental practices
12 and behavioural change. Detailed analysis of strategies, methods and approaches to scale up SLM have
13 been undertaken for GEF programs in Africa, China and globally (Tengberg and Valencia 2018; Liniger
14 et al. 2011; Tengberg et al. 2016). A number of interconnected barriers and bottlenecks to the scaling
15 up of SLM have been identified in this context and are related to:

- 16 • Limited access to knowledge and information, including new SLM technologies and problem-
17 solving capacities;
- 18 • Weak enabling environment, including the policy, institutional and legal framework for SLM, and
19 land tenure and property rights;
- 20 • Inadequate learning and adaptive knowledge management in the project cycle, including
21 monitoring and evaluation of impacts; and
- 22 • Limited access to finance for scaling up, including public and private funding, innovative business
23 models for SLM technologies and financial mechanisms and incentives, such as payments for
24 ecosystem services, insurance and micro-credit schemes.

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

43 A study on SLM in the Kenyan highlands using transition theory concluded that barriers to adoption of
44 SLM included high poverty levels, a low input-low output farming system with limited potential to
45 generate income, diminishing land sizes and low involvement of the youth in farming activities.
46 Coupled with a poor coordination of government policies for agriculture and forestry, these barriers
47 created negative feedbacks in the SLM transition process. Scaling up of SLM technologies would
48 require collaboration of diverse stakeholders across multiple scales, a more supportive policy

1 environment and substantial resource mobilization (Mutoko et al. 2014). Tengberg and Valencia (2018)
 2 situated the findings from a review of the GEF integrated natural resources management portfolio of
 3 projects in the transition theory framework. They concluded that to remove barriers to SLM, an
 4 agricultural innovations systems approach that supports co-production of knowledge with multiple
 5 stakeholders, institutional innovations, a focus on value chains and strengthening of social capital to
 6 facilitate shared learning and collaboration could accelerate the scaling up of sustainable technologies
 7 and practices from the niche to the landscape level. Policy integration and establishment of financial
 8 mechanisms and incentives could contribute to overcoming barriers to a regime shift. The new SLM
 9 regime could in turn be stabilised and sustained at the landscape level by multistakeholder knowledge
 10 platforms and strategic partnerships. However, transitions to more sustainable regimes and practices are
 11 often challenged by lock-in mechanisms in the current system (Lawhon and Murphy 2012), such as
 12 economies of scale, investments already made in equipment, infrastructure and competencies, lobbying,
 13 shared beliefs, and practices, which could hamper wider adoption of SLM.



15
 16 **Figure 4.10 The transition from SLM niche adoption to regime shift and landscape development (figure**
 17 **draws inspiration from (Geels 2002)). Adapted from (Tengberg and Valencia 2018)**

18 Adaptive, multi-level and participatory governance of social-ecological systems (Figure 4.10) is
 19 considered important for regime shifts and transitions to take place (Wieczorek 2018) and essential to
 20 secure the capacity of environmental assets to support societal development over longer time periods
 21 (Folke et al. 2005). There is also recognition that effective environmental policies and programs need
 22 to be informed by a comprehensive understanding of the biophysical, social, and economic components
 23 and processes of a system, their complex interactions, and how they respond to different changes (Kelly
 24 (Letcher) et al. 2013). But blueprint policies will not work due to the wide diversity of rules and informal
 25 institutions used across sectors and regions of the world, especially in traditional societies (Ostrom
 26 2009).

1 The most effective way of removing barriers to funding of SLM has been mainstreaming of SLM
 2 objectives and priorities into relevant policy and development frameworks and combining SLM best
 3 practices with economic incentives for land users. As the short-term costs for establishing and
 4 maintaining SLM measures are generally high and constitute a barrier to adoption, land users need to
 5 be compensated for generation of longer-term public goods, such as ecosystem services. Cost-benefit
 6 analysis should therefore be conducted on all SLM interventions (Liniger et al. 2011; Mirzabaev et al.
 7 2016; Tengberg et al. 2016) (Liniger et al. 2011; Nkonya et al. 2016; Tengberg et al. 2016). Alternative
 8 ways of improving adoption and scaling up SLM are emerging through increased focus on participatory
 9 governance, development of new SLM business models and innovative funding schemes, such as LDN
 10 and carbon finance, with the potential to improve local livelihoods, sequester carbon and enhance the
 11 resilience to climate change.

12

13 **4.11 Hotspots and case-studies**

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

19 **Table 4.3 Synthesis of how the case studies interact with climate change and a broader set of co-benefits**
 20 *(Table under development)*

examples of responses (4.11)	readiness	rate of change	mitigation benefits and potential	adaptation benefits	co-benefits
Urban green infrastructure	now				health
Agro-forestry	now				agricultural diversification
Biochar	soon				
Perennial grains	8-10 years				agricultural diversification, reduced costs
Reforestation	now				
Management of peat soils	now				
Forest fire management	now				

drought resistance	carbon sink	reduced emission	reduced flood risk	reduced heat stress	slow	medium	fast

21

22

1 **4.11.1 The role of urban green infrastructure in land degradation, climate change** 2 **adaptation and mitigation**

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

17 There is now a large body of research and application demonstrating the importance of retaining urban
18 green infrastructure (UGI) for the delivery of multiple ecosystem services (Wentworth J 2017;
19 Puigdefábregas 1998) as an important tool to mitigate and adapt to climate change. UGI can be defined
20 as all green elements within a city, including but not limited to retained indigenous ecosystems, parks,
21 public greenspaces, green corridors, street trees, urban forests, urban agriculture, green roofs/walls and
22 private domestic gardens (Tzoulas et al. 2007). The definition is usually extended to include 'blue'
23 infrastructure, such as rivers, lakes, bioswales and other water drainage features.

24 Through retaining existing vegetation and ecosystems, revegetating previous developed land or
25 integrating vegetation into buildings in the form of green walls and roofs, UGI can play a direct role in
26 mitigating climate change through carbon sequestration. However, compared to overall carbon
27 emissions from cities, effects are likely to be small. Given that UGI necessarily involves the retention
28 and management of non-sealed surfaces, co-benefits for land degradation (e.g., improved water
29 retention, carbon storage and vegetation productivity) will also be apparent. Although not currently a
30 priority, its role in mitigating land degradation could be substantial. For instance, appropriately
31 managed innovative urban agricultural production systems, such as vertical farms, could have the
32 potential to both meet some of the food needs of cities and reduce the production (and therefore
33 degradation) pressure on agricultural land in rural areas, although thus far this is unproven (for a recent
34 review Wilhelm and Smith 2018).

35 The importance of UGI as part of a climate change adaptation approach has received greater attention
36 and application (Gill et al. 2007; Fryd et al. 2011) (Demuzere et al. 2014) (Sussams et al. 2015). The
37 EU's Adapting to Climate Change White Paper emphasises the "crucial role in adaptation in providing
38 essential resources for social and economic purposes under extreme climate conditions" (CEC 2009
39 p.5) Increasing vegetation cover, planting street trees and maintaining/expanding public parks reduces
40 temperatures (Cavan et al. 2014) (Di Leo et al. 2016) (Fryd et al. 2011) (Zölch et al. 2016); while the
41 use of green walls and roofs can also reduce energy use in buildings (eg, Coma et al. 2017). Similarly
42 natural flood management and ecosystem based approaches of providing space for water, renaturalising
43 rivers and reducing surface run-off through the presence of impermeable surfaces and vegetated features
44 (including walls and roofs) can manage flood risks, impacts and vulnerability (Cavan et al. 2014)
45 (Munang et al. 2013). Access to UGI in times of environmental stresses and shock can provide safety
46 nets for people and can, therefore, be an important adaption mechanism, both to climate change
47 (Potschin et al. 2016) and land degradation.

1 Most examples of UGI implementation as a climate change adaptation strategy have centered on its role
2 in water management for flood risk reduction. The importance for land degradation is either not stated,
3 or not prioritised. In Beira, Mozambique, the government is using UGI to mitigate against increased
4 flood risks predicted to occur under climate change and urbanisation, which will be done by improving
5 the natural water capacity of the Chiveve River. As part of the UGI approach, mangrove habitats have
6 been restored and future phases include developing new multi-functional urban green spaces along the
7 river (World Bank 2016). The retention of green spaces within the city will have the added benefit of
8 halting further degradation in those areas. Elsewhere, planning mechanisms promote the retention and
9 expansion of green areas within cities to ensure ecosystem service delivery, which directly halts land
10 degradation, but are largely viewed and justified in the context of climate change adaptation and
11 mitigation. For instance, the Landscape Programme in Berlin includes five plans, one of which covers
12 adapting to climate change through the recognition of the role of UGI (Green Surge 2016). Major
13 climate related challenges facing Durban, South Africa, include sea level rise, urban heat island, water
14 runoff and conservation (Roberts and O'Donoghue 2013). Now considered a global leader in climate
15 adaptation planning (Roberts 2010). Durban's Climate Change Adaptation plan includes the retention
16 and maintenance of natural ecosystems in particular those which are important for mitigating flooding,
17 coastal erosion, water pollution, wetland siltation and climate change (eThekweni Municipal Council
18 2014).

19 Urban green infrastructures can deliver several ecosystem services, such as improved water infiltration,
20 flood protection, and reduced heat stress while mitigating climate change through carbon storage in
21 vegetation and soils.

22 **4.11.2 Perennial Grains and Soil Organic Carbon**

23 Agriculture has been responsible for greater land degradation than any other human activity
24 (Montgomery 2007a). The severe ecological perturbation that is inherent in the conversion of native
25 perennial vegetation to annual crops, and the subsequent high frequency of perturbation required to
26 maintain annual crops results in at least four forms of soil degradation that will likely be exacerbated
27 by the effects of climate change (Crews et al. 2016). First, soil erosion is a very serious consequence of
28 annual cropping with median losses exceeding rates of formation by 1-2 orders of magnitude in
29 conventionally plowed agroecosystems, and by a factor of four when conservation tillage is employed
30 (Montgomery 2007b). More severe storm intensity associated with climate change is expected to cause
31 even greater losses to wind and water erosion (Nearing et al. 2004b). Secondly, the periods of time in
32 which live roots are reduced or altogether absent from soils in annual cropping systems allow for
33 substantial losses of nitrogen from fertilized croplands, averaging 50% globally (Ladha et al. 2005).
34 This low retention of nitrogen is also expected to worsen with more intense weather events (Bowles et
35 al. 2018). A third impact of annual cropping is the degradation of soil structure caused by tillage, which
36 can reduce infiltration of precipitation, and increase surface runoff. It is predicted that the percentage
37 of precipitation that infiltrates into agricultural soils will decrease further under climate change
38 scenarios (Basche and DeLonge 2017; Wuest et al. 2006). The fourth form of soil degradation that
39 results from annual cropping is the reduction of soil organic matter (SOM), a topic of particular
40 relevance to climate change mitigation and adaptation.

41 Undegraded cropland soils can theoretically hold far more SOM (which is 58% carbon) than they
42 currently do (Soussana et al. 2006). We know this deficiency because, with few exceptions,
43 comparisons between cropland soils and those of proximate mature native ecosystems show a 20-70%
44 decline in soil carbon attributable to agricultural practices. What happens when native ecosystems are
45 converted to agriculture that induces such significant losses of SOM? Wind and water erosion
46 commonly results in preferential removal of light organic matter fractions that can accumulate on or
47 near the soil surface (Lal 2003). In addition to the effects of erosion, the fundamental practices of
48 growing annual food and fiber crops alters both inputs and outputs of organic matter from most

1 agroecosystems resulting in net reductions in soil carbon equilibria (Soussana et al. 2006; McLauchlan
2 2006; Crews et al. 2016). Native vegetation of almost all terrestrial ecosystems is dominated by
3 perennial plants, and the belowground carbon allocation of these perennials is a key variable in
4 determining formation rates of stable soil organic carbon (SOC) (Jastrow et al. 2007; Schmidt et al.
5 2011). When perennial vegetation is replaced by annual crops, inputs of root-associated carbon (roots,
6 exudates, mycorrhizae) decline substantially. For example, perennial grassland species allocate around
7 67% of productivity to roots, whereas annual crops allocate between 13-30% (Saugier 2001; Johnson
8 et al. 2006).

9 At the same time inputs of SOC are reduced in annual cropping systems, losses are increased because
10 of tillage, compared to native perennial vegetation. Tillage breaks apart stable soil aggregates, which,
11 among other functions, are thought to inhibit soil bacteria, fungi and other microbes from consuming
12 and decomposing soil organic matter (Grandy and Neff 2008). Aggregates limit microbial attack by
13 restricting physical access to the organic compounds as well as reducing oxygen availability. When soil
14 aggregates are broken open with tillage in the conversion of native ecosystems to agriculture, microbial
15 consumption of SOC and subsequent respiration of CO₂ increase dramatically, reducing soil carbon
16 stocks (Grandy and Robertson 2006; Grandy and Neff 2008).

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

26 There are two basic strategies that plant breeders and geneticists are using to develop new perennial
27 grain crop species. The first involves making wide hybrid crosses between existing elite lines of annual
28 crops, such as wheat, sorghum and rice, with related wild perennial species in order to introgress
29 perennialism into the genome of the annual (Cox et al. 2018b; Huang et al. 2018; Hayes et al. 2018).
30 The other approach is de novo domestication of wild perennial species that have crop-like traits of
31 interest (DeHaan et al. 2016; DeHaan and Van Tassel 2014). New perennial crop species undergoing
32 de novo domestication include intermediate wheatgrass, a relative of wheat that produces grain
33 marketed as Kernza® (DeHaan et al. 2018; Cattani and Asselin 2018) and *Silphium integrifolium*, an
34 oilseed crop in the sunflower family (Van Tassel et al. 2017). Other perennial grain crops receiving
35 attention include pigeon pea, barley, buckwheat and maize (Batello et al. 2014; Chen et al. 2018) and a
36 number of legume species (Schlautman et al. 2018).

37 In a perennial agroecosystem, the biogeochemical controls on SOC accumulation shift dramatically,
38 and begin to resemble the controls that govern native ecosystems (Crews et al. 2016). When erosion is
39 reduced or halted, and crop allocation to roots increases by 100-200%, and when soil aggregates are not
40 disturbed thus reducing microbial respiration, SOC levels are expected to increase (Crews and Rumsey
41 2017). Deep roots growing year-round are also effective at increasing nitrogen retention (Culman et al.
42 2013). Substantial increases in SOC have been measured where croplands that had historically been
43 planted to annual grains were converted to perennial grasses, such as in the Conservation Reserve
44 Program (CRP) of the U.S., or in plantings of second generation perennial biofuel crops. Two studies
45 have assessed carbon accumulation in soils when croplands were converted to the perennial grain
46 Kernza. In one, researchers found no differences in soil labile (permanganate-oxidizable) C after 4
47 years of cropping to perennial Kernza versus annual wheat in a sandy textured soil. Given that coarse
48 textured soils do not offer the same physicochemical protection against microbial attack as many finer

1 textured soils, these results are not surprising, but these results do underscore how variable rates of
2 carbon accumulation can be (Jastrow et al. 2007). In the second study, researchers assessed the carbon
3 balance of a Kernza field in Kansas USA over 4.5 years using eddy covariance observations (de Oliveira
4 et al. 2018). They found the net C accumulation rate of about $1500 \text{ g C m}^{-2} \text{ yr}^{-1}$ in the first year of the
5 study corresponding to the biomass of Kernza increasing, to about $300 \text{ g C m}^{-2} \text{ yr}^{-1}$ in the final year where
6 CO_2 respiration losses from the decomposition of roots and soil organic matter approached new carbon
7 inputs from photosynthesis. Based on measurements of soil carbon accumulation in restored grasslands
8 in this part of USA, the net carbon accumulation in stable organic matter under a perennial grain crop
9 might be expected to sequester on the order of $30\text{-}50 \text{ g C m}^{-2} \text{ yr}^{-1}$ (Post and Kwon 2000). Sugar cane, a
10 highly productive perennial, has been shown to accumulate a mean of $187 \text{ g C m}^{-2} \text{ yr}^{-1}$ in Brazil (La
11 Scala Júnior et al. 2012).

12 Reduced soil erosion, increased nitrogen retention, greater water uptake efficiency and enhanced carbon
13 sequestration represent improved ecosystem functions made possible in part by deep and extensive root
14 systems of perennial crops (Figure 4.11).



15
16 **Figure 4.11 Comparison of root systems between the newly domesticated intermediate wheatgrass (left)**
17 **and annual wheat (right). Photo and copyright: Jim Richardson**

18
19 When compared to annual grains like wheat, single species stands of deep rooted perennial grains such
20 as Kernza would be expected to reduce soil erosion, increase nitrogen retention, achieve greater water
21 uptake efficiency and enhance carbon sequestration (Crews et al. 2018) (Figure 4.11). An even higher
22 degree of ecosystem services should at least theoretically be achieved by strategically combining
23 different functional groups of crops such as a grain and a nitrogen-fixing legume (Soussana and Lemaire
24 2014). Not only is there evidence from plant diversity experiments that communities with higher
25 species richness appear to sustain higher concentrations of soil organic carbon (Hungate et al. 2017;
26 Sprunger and Robertson 2018; Chen et al. 2018b), but other valuable ecosystem services such as pest
27 suppression, lower greenhouse gas emissions, and greater nutrient retention may be enhanced (Schnitzer
28 et al. 2011; Culman et al. 2013).

1 Similar to perennial forage crops such as alfalfa, perennial grain crops are expected to have a definite
2 productive life span, most likely in the range of 3-10 years. A key area of research on perennial grains
3 cropping systems is to minimise losses of soil organic carbon during conversion of one stand of
4 perennial grains to another. Work in the upper Midwest USA has demonstrated large net increases in
5 CO₂ and N₂O emissions for several years following initial cultivation of a perennial grassland and
6 conversion to annual crops (Grandy and Robertson 2006). Large emissions are to be expected in such
7 a conversion given the disruption of soil aggregates caused by tillage and the replacement of perennial
8 vegetation with annuals. Recent work suggests that no-till conversion of a mature perennial grassland
9 to another perennial crop will experience several years of high net CO₂ emissions as decomposition of
10 copious crop residues exceeds ecosystem uptake of carbon by the new crop (Abraha et al. 2018).

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

15 **4.11.3 Reversing Land Degradation, examples of large scale restoration and** 16 **rehabilitation of degraded lands (South Korea and China)**

17 **4.11.3.1 Korea Case Study on Reforestation Success**

18 In the first half of the 20th century, forests in the Republic of South Korea were severely degraded and
19 deforested during foreign occupations and the Korean War. Unsustainable harvest for timber and fuel
20 wood resulted in severely degraded landscapes, heavy soil erosion and large areas denuded of vegetation
21 cover. Recognising that Korea's economic health would depend on a healthy environment, Korea
22 established a national forest service (1967) and embarked on the first phase of a 10-year reforestation
23 program in 1973 (Forest Development Program), which was followed by subsequent reforestation
24 programs that ended in 1987, after 2.4 Mha of forests were restored, see Figure 4.12.

25 As a consequence of reforestation, forest volume increased from 11.3 m³ ha⁻¹ in 1973 to 125.6 m³ ha⁻¹
26 in 2010 and 150.2 m³ ha⁻¹ in 2016 (Korea Forest Service 2017). Increases in forest volume had
27 significant co-benefits such as increasing water yield by 43% and reducing soil losses by 87% from
28 1971 to 2010 (Kim et al. 2017).

29 The forest carbon density in Korea has increased from 5–7 Mg C ha⁻¹ in the period 1955–1973 to more
30 than 30 Mg C ha⁻¹ in the late 1990s (Choi et al. 2002). Estimates of C uptake rates in the late 1990s
31 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.
32 2014), lower than other estimates because reforestation programs did not start until 1973. NEP in South
33 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
34 Tg C yr⁻¹ in the 2000s, showing a gradual decline as average forest age increased (Cui et al. 2014). The
35 estimated past and projected future increase in the carbon content of Korea's forest area during 1992-
36 2034 was 11.8 Tg C yr⁻¹ (Kim et al. 2016).



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Figure 4.12 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)

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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 by reducing demand for firewood 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 planting more native species and encouraging natural regeneration (Kim and Zsuffa 1994; Lee et al. 2015). Avoiding monocultures in reforestation programs can reduce susceptibility to pests (Kim and Zsuffa 1994). Other important factors in the success of the reforestation program were that private landowners were heavily involved in initial efforts (both corporate entities and smallholders) and that the reforestation program was made part of the national economic development program (Lamb 2014). In summary, the reforestation program was a comprehensive technical and social initiative that restored forest ecosystems and enhanced the economic performance of rural regions (Kim et al. 2017).

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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).

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The net present value and the benefit-cost ratio of the reforestation program were USD 54.3 billion and 5.84 billion 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).

1 **4.11.3.2 China Case Study on Reforestation Success**

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

19 GFGP became the most ambitious of China’s ecological restoration efforts with over USD 45 billion
20 devoted to its implementation since 1990 (Kolinjivadi and Sunderland 2012; Liqiao and Changjin,
21 2007) The program involves the conversion of farmland on slopes of 15-25° or greater to forest or
22 grassland (Bennett 2008). The pilot program started in three provinces –Sichuan, Shaanxi, and Gansu
23 – in 1999 (Liu and Diamond 2008). After initial success, it was extended to 17 provinces by 2000 and
24 finally to 25 provinces by 2002 (Figure 4.13). The GFGP covers all provinces in western China (Figure
25 4.13), which is 80% of the total area with soil erosion (4360 Mha), including the headwaters of the
26 Yangtze and Yellow rivers (Liu et al. 2008).

27 NFPP and GFGP have dramatically improved China’s land conditions and ecosystem services, and thus
28 have mitigated the unprecedented land degradation in China (Liu et al. 2013; Liu et al 2002; Long et al.
29 2006; Xu et al. 2006). NFPP protected 107 Mha forest area and increased forest area by 10 Mha between
30 2000 and 2010. For the second phase (2011–2020), the NFPP plans to increase forest cover by 5.2 Mha,
31 capture 416 million tons of carbon, provide 648,500 forestry jobs, further reduce land degradation, and
32 enhance biodiversity (Liu, J. Z. Ouyang, W. Yang, W. Xu 2013). During 2000–2007, sediment
33 concentration in the Yellow River had declined by 38%. In the Yellow River basin, it was estimated
34 that surface runoffs would be reduced by 450 million m³ from 2000 to 2020, which is equivalent to
35 0.76% of the total surface water resources (Jia et al. 2006). GFGP had cumulatively increased vegetative
36 cover by 25 Mha, with 8.8 Mha of cropland being converted to forest and grassland, 14.3 Mha barren
37 land being afforested, and 2.0 million ha of forest regeneration from mountain closure. Forest cover
38 within the GFGP region has increased 2% during the first 8 years (Liu et al. 2008). In Guizhou Province,
39 GFGP plots had 35–53% less loss of phosphorus than non-GFGP plots (Liu et al. 2002). In Wuqi County
40 of Shaanxi Province, the Chaigou Watershed had 48% and 55% higher soil moisture and moisture-
41 holding capacity in GFGP plots than in non-GFGP plots, respectively (Liu et al. 2002). According to
42 reports on China’s first national ecosystem assessment (2000–2010), for carbon sequestration and soil
43 retention, coefficients for the GTGP targeting forest restoration and NFCP are positive and statistically
44 significant. For sand fixation, GTGP targeting grassland restoration is positive and statistically
45 significant.

46 NFPP-related activities received a total commitment of 93.7 billion yuan between 1998 and 2009. Most
47 of the money was used to offset economic losses of forest enterprises caused by the transformation from
48 logging to tree plantations and forest management (Liu et al. 2008). By 2009, the cumulative total

1 investment through the NFPP and FGFP exceeded USD 50 billion and directly involved more than 120
 2 million farmers in 32 million households in the FGFP alone (Liu, J. Z. Ouyang, W. Yang, W. Xu 2013).
 3 All programs reduce or reverse land degradation and improve human well-being. Thus, a coupled
 4 human and natural systems perspective (Liu et al. 2008) would be helpful to understand the complexity
 5 of policies and their impacts, and to establish long-term management mechanisms to improve the
 6 livelihood of participants in these programs and other land management policies in both China and
 7 many other parts of the world.



8
 9 **Figure 4.13 Current distribution of the National Forest Protection Program (NFPP) and the Grain for**
 10 **Green Program (GTGP) in China (Liu et al. 2008)**

11 **4.11.4 Degradation and management of peat soils**

12 The important functions of peatland ecosystems in the global carbon cycle and in climate regulation has
 13 been recognised for decades (Gorham 1991) and their degradation caused directly or indirectly by
 14 anthropogenic activities has a significant climatic impact (Haddaway et al. 2014; IPCC 2014b; Evans
 15 et al. 2016) together with critical feedback processes (Loisel and Yu 2013; Mäkiranta et al. 2018).

16 Globally, peatlands cover 3% of the Earth's land area (~420 Mha) (Xu et al. 2018a) and store 26-44%
 17 of estimated global soil organic carbon (Moore 2002). They are most abundant in high northern
 18 latitudes, covering large areas in North America, Russia and Europe where they store approximately
 19 500 Gt of carbon (Loisel and Yu 2013; Xu et al. 2018). At lower latitudes, the largest areas of tropical
 20 peatlands are located in Indonesia, the Congo Basin and the Amazon Basin in the form of peat swamp
 21 forests (Gumbricht et al. 2017; Page et al. 2011). Tropical peatlands are estimated to cover about
 22 400,000 km² and store an additional 50 to 100 Gt C, primarily in Southeast Asia (Yu 2012; Page et al.
 23 2011). There are considerable uncertainties about peatland spatial extent, distribution, status and,
 24 therefore, their contribution to global greenhouse gas (GHG) emissions (Webster et al. 2018).

1 It is estimated that while 80-85% of the global peatland areas is still largely in a natural state, they are
2 such carbon-dense ecosystems that degraded peatlands (0.3% of the terrestrial land) are responsible for
3 a disproportional 5% of global anthropogenic carbon dioxide (CO₂) emissions (Joosten 2009), that is
4 an annual addition of 0.9-3 Gt of CO₂ to the atmosphere (Dommain et al. 2012; IPCC 2014c).

5 **4.11.4.1 Tropical peat soils**

6 Hotspots of peatland degradation have spread to the tropics in the twenty-first century (Lilleskov et al.
7 2018; Page and Baird 2016; Hooijer et al. 2010) have estimated that 47% of the total peat area in
8 Southeast Asia had been deforested by 2006.

9 Recognising these constraints, Gumbricht et al. (2017) developed a novel method for mapping tropical
10 wetlands and peatlands using a hybrid expert system approach with hydrological modelling, time series
11 analysis of soil moisture phenology from optical satellite data, and hydro-geomorphology from
12 topographic data. This approach yielded surprisingly high areas and volumes of tropical peatlands (1.7
13 Mkm² and 7,268 km³), which were more than threefold higher than previous estimates. The new map
14 suggests that South America rather than Asia harbours the largest tropical peatland area and volume
15 (around 44% for both), partly related to some previously unaccounted deep deposits, but mainly to the
16 prediction of the existence of extensive shallow peat in the Amazon Basin. The model needs further
17 validation, but recent data from Central Africa (Dargie et al. 2017), the Western Amazon (Lähteenoja
18 et al. 2012), and the Central Amazon (Lähteenoja et al. 2013) show that tropical peat deposits are much
19 more widespread than previously thought.

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

29 The climate impacts of land use change and degradation in tropical peatlands have primarily been
30 quantified in Southeast Asia, where drainage and conversion to plantation crops is the dominant
31 transition. The CO₂ sink is lost when peatlands are drained and converted into a net source to the
32 atmosphere. Oil palm is the most widespread plantation crop and on average it emits 11 MgC/ha/yr;
33 Acacia plantations for pulpwood are the second most widespread plantation crop and emit 20 MgC ha⁻¹
34 yr⁻¹ (Drösler et al. 2013). Other land uses typically emit less than 10 MgC ha⁻¹ yr⁻¹. Total emissions
35 from peatland drainage in the region are estimated to be between 0.03 and 0.2 PgC yr⁻¹ (Houghton and
36 Nassikas 2017; Frohling et al. 2011). Land use change also affects the fluxes of N₂O and CH₄.
37 Undisturbed tropical peatlands emit about 0.8 Tg CH₄ yr⁻¹ and 0.002 TgN₂O yr⁻¹, while disturbed
38 peatlands emit 0.1 Tg CH₄ yr⁻¹ and 0.2 Tg yr⁻¹N₂O (Frohling et al. 2011). These N₂O emissions are likely
39 low as new findings show that emissions from fertilised oil palm can exceed 20 kg N₂O–N ha⁻¹ yr⁻¹
40 (Oktarita et al. 2017).

41 Fire emissions from tropical peatlands are only a serious issue in Southeast Asia, where they are
42 responsible for 173 (18–1110) TgC yr⁻¹ (van der Werf et al. 2017). Much of the variability is linked
43 with the El Niño Southern Oscillation, which produces drought conditions in this region. Anomalously
44 active fire seasons have also been observed in non-drought years and this has been attributed to the
45 increasing effect of high temperatures that dry vegetation out during short dry spells in otherwise normal
46 rainfall years (Fernandes et al. 2017; Gaveau et al. 2014). These fires have significant societal impacts.
47 Koplitz et al. (2016), for example, used smoke exposure to estimate that the 2015 fires caused over

1 100,000 additional deaths across Indonesia, Malaysia and Singapore and that this event was more than
2 twice as deadly as the 2006 El Niño event.

3 Peatland degradation in other parts of the world differ from Asia. In Africa large peat deposits like those
4 found in the Cuvette Centrale in the Congo Basin or in the Okavango inland delta, the principle threat
5 is changing rainfall regimes due to climate variability and change (Weinzierl et al. 2016; Dargie et al.
6 2017). Expansion of agriculture is not yet a major factor in these regions. In the Western Amazon,
7 extraction of non-timber forest products like the fruits of *Mauritia flexuosa* and Suri worms are major
8 sources of degradation that lead to losses of carbon stocks (Hergoualc'h et al. 2017).

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

20 **4.11.4.2 Temperate and boreal peat soils**

21 Land-use change has led to the loss of 15% of global peatland habitats over the last century (Barthelmes
22 2016). It has been estimated that 65,000 km² or 10% of the maximum peatland area occurring during
23 the Holocene in Europe (Joosten 2009) does not exist anymore as peatlands and 44% of the remaining
24 European peatlands are deemed 'degraded' (Joosten, H., Tanneberger 2017). Moreover, large areas of
25 specific peatland types, such as fens have been entirely 'lost' or greatly reduced in thickness due to peat
26 wastage (Lamers et al., 2015), Figure 4.14.



27
28 **Figure 4.14 A hag top is a free-stranding remaining part of the original bog surface around which the**
29 **peat has eroded away due to past turf cutting and over-grazing along with natural climate erosion**
30 **(Mount Errigal, Co. Donegal, Ireland, Photo Dr Florence Renou-Wilson)**

31 Degradation of peatlands has been on-going for centuries due to economic development in increasingly
32 populated areas and was associated with land use conversion, intensification or other water-related
33 management (Joosten, H., Tanneberger 2017; Page and Baird 2016). The main drivers of the

1 acceleration of peatland degradation in the twentieth century were associated with drainage for
2 agriculture, peat extraction and afforestation and related activities (burning, over-grazing, fertilisation)
3 with a variable scale and severity of impact depending on existing resources in the various countries
4 (Lamers et al. 2015; Landry, J., Rochefort 2012; Renou-Wilson 2018). These drivers have been
5 supplemented by more recent utilisation, e.g. urban development, wind farm construction (Smith et al.
6 2012), hydro-electric development, tar sands mining and recreational structures and activities (Joosten,
7 H., Tanneberger 2017). Anthropogenic pressures are now affecting peatlands in previously
8 geographically isolated areas where the properties and benefits of these undegraded peatlands (in terms
9 of the carbon store for example or support for local livelihood) are only slowly being revealed (Dargie
10 et al. 2017; Lawson et al. 2015).

11 The required drainage for peatland conversion and utilisation has direct and indirect effects that include
12 (1) altered hydrology (decrease in soil moisture, water storage capacity and water quality); (2) peat
13 oxidation and shrinkage; (3) peat subsidence and compression; (4) peat erosion; (5) loss of biodiversity
14 and natural resources for local population and (6) loss of cultural and archaeo-environmental
15 information. In many cases, peatland exploitation has resulted in long-term, irreversible degradation
16 (Illnicki, P., Zeitz 2003) where oxidation of the drained peat over many decades led to subsidence.
17 Subsidence rates varied from a few millimetres to as much as 3 cm yr⁻¹, increasing with higher
18 temperatures (Kasimir-Klemedtsson et al. 1997). This leads to further changes in local environmental
19 and hydrological condition, such that further secondary disturbances are impending (flooding, salt water
20 intrusion, abandonment) or alternative land use (usually reverting to wet-land production such as rice
21 or conservation) required (Drexler et al. 2009).

22 Any degradation processes affecting peat soils lead to altered GHG balances and drained peatlands have
23 been identified as GHG emissions hotspots (Davidson and Janssens 2006; Waddington et al. 2002;
24 Wilson et al. 2015). Lowering of the water table leads to direct and indirect increased CO₂ and N₂O
25 emissions to the atmosphere with emission rates dependent on a range of factors, including the
26 groundwater level, peat moisture, nutrient content, peat temperature and vegetation communities. In
27 addition, drainage significantly increases erosion thus removing stored carbon into streams (Armstrong
28 et al. 2010). Dissolved Organic Carbon (DOC) is accompanied by Particulate Organic Carbon (POC
29 especially where vegetation cover is absent as in peat extraction) (Wilson et al. 2011). In both cases,
30 the organic carbon ultimately returned to the atmosphere as CO₂ via re-mineralisation (Bonn, A., Allott,
31 T., Evans, M., Joosten, H., Stoneman 2016). This leads to a colouration in drinking water and negative
32 socio-economic impacts that include health-related issues associated with the cleaning of the water
33 (O'Driscoll et al. 2018). Furthermore, as pristine bogs often act as natural water filter, once drained,
34 stored pollutant metals are also exported to nearby water bodies (Bain et al. 2011).

35 With a very low bulk density, peat is very susceptible to erosion, especially in wet and sloping
36 landscapes where water flow may exert substantial erosive force (Evans and Warburton 2008; Evans et
37 al. 2006). The predicted impacts of extreme events such as high rainfall storm events are likely to be
38 not only impactful for peat soils with for example low latitudes blanket bogs being at most risks of
39 fluvial erosion under climate-change scenarios (Havlik et al. 2014; Li et al. 2017), but also with negative
40 consequences on the downstream habitats (Gallego-Sala et al., 2016). Where the peat is devoid of
41 vegetation, it is more susceptible to wind, rain, frost and summer desiccation leading to further erosion
42 (Holden et al. 2007).

43 While palaeo-ecological studies of peatlands have shown that peat (and C) accumulation as well as the
44 vegetation and the hydrology of peatlands, have all been altered by past climate change (Charman et
45 al., 2013)(Kołaczek et al. 2018), human-related climate change is a major threat to global peatlands,
46 both in terms of the provision of ecosystem (Parish et al. 2008; Page and Baird 2016) and future
47 potential peat formation (Belyea and Malmer 2004). The climatic envelope for bogs (nutrient poor
48 peatlands) especially is predicted to be severely reduced by the end of the 21st century (Coll et al. 2014)

1 with considerable risks for some regional peatlands, e.g., central Europe (Essl et al. 2012; National
2 Climate Commission 2009; Havlik et al. 2014).

3 The diverse nature of peatlands (both across the globe but also small-scale heterogeneity) and their self-
4 regulation mechanism make it all the more difficult to assess the degradation effect of climate change
5 precisely. The projected changes are likely to affect the extent, distribution and nature of their
6 biogeochemical functions. The changes to peatland functioning arising from climate change can be
7 direct, via increased summer water deficit (lowering water table and impairing primary productivity
8 (Reichstein et al. 2013), increased air/soil temperatures (affecting both CO₂ and CH₄ exchange
9 (Bragazza et al. 2016; Glatzel et al. 2006), increased cloud cover in boreal zone (Nijp et al. 2015) or
10 increased frequency of high precipitation events (Li et al. 2017) and wildfires (Turetsky et al. 2015)
11 (Turetsky et al., 2015), but also indirect via changes in agricultural practices, fire regimes, coastal
12 flooding and population displacements (Henman and Poulter 2008; Veber et al. 2018). While rain-fed
13 nutrient poor bogs are likely to be particular at risk (Gallego-Sala and Colin Prentice 2013), there is a
14 major concern for all peatlands as increasing anthropogenic pressure on water resources and reduced
15 water availability will aggravate the problem (Parish et al. 2008). The scale and rate of degradation will
16 vary depending on the type of peatland and the existing pressures at a particular location (Lindsay
17 2010).

18 Separating the impacts of climate change from anthropogenic activities is not easy since they result
19 from similar hydrological disturbances (Kołaczek et al. 2018). The synergistic effect of climate change
20 and intentional changes (e.g. drainage) can result in severe degradation as drained peat soils (especially
21 those used for peat extraction and agriculture) are projected to become even greater hotspots of CO₂
22 emissions under climate warming scenarios (Renou-Wilson and Wilson 2018). This faster drying and
23 oxidation of the peat is critical in terms of the climatic feedback (Limpens et al. 2008). For example,
24 when drought is combined with drained peatlands where the vegetation has been disturbed and human
25 access has increased, a higher risk and incidence of fire ensue. Recent peatland fires in Southeast Asia,
26 Russia and also in more oceanic areas, such as Scandinavia and Ireland/UK, are increasing in extent,
27 frequency, even during non-drought years (Gaveau et al. 2014) serious implication for climate, pollution
28 and the health of local communities (Page and Hooijer 2016; Turetsky et al. 2015).

29 **4.11.4.3 Management and restoration of peat soils**

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

45 In higher latitudes where degraded peatlands in most cases have been drained, the most effective option
46 to protect these large organic carbon stocks and mitigate climate change is to carry out hydrological
47 manipulations that aim to increase soil moisture and surface wetness (Schils et al 2008), Figure 4.15.
48 Long-term GHG monitoring has demonstrated that rewetting and restoration noticeably reduce

1 emissions in comparison to degraded drained sites (Wilson et al. 2016; IPCC 2014b) and returned the
2 carbon sink function when key species were re-established (Nugent et al. 2018) although they might
3 not yet be as resilient as their intact counterparts (Wilson et al. 2016). Blocking ditches help raise the
4 water table by retarding the discharge to surface runoff (Holden et al. 2017). Several studies have
5 demonstrated the co-benefits of rewetting specific degraded peatlands in order to maximise biodiversity
6 provision and climate change mitigation (Parry et al. 2014; Ramchunder et al. 2012; Renou-Wilson et
7 al. 2018) or to return other ecosystem services such as improvement of water storage and quality
8 (Martin-Ortega et al. 2014) with beneficial consequences for human well-being (Bonn et al. 2016; Parry
9 et al. 2014). On the ground, farmers and land owners may not prioritise climatic benefits per se but may
10 be willing to prioritise management options that halt land degradation and desertification (Joosten et
11 al., 2012)(FAO 2017) or for economic benefits e.g. production of biomass on wet peat also known as
12 paludiculture (Barthelmes, 2016) realising critical synergies (Günther et al., 2015). Finally, care should
13 be taken however not to create environments (e.g., flooded degraded peatlands) that would cause higher
14 CH₄ emissions (Hahn et al. 2015) (Kasimir et al. 2018) (Hahn et al., 2015; Kasimir et al., 2018).



15

16 **Figure 4.15 Near-surface water table at a rewetted (previously drained) raised bog (Moyarwood, Co.**
17 **Galway, Ireland, Photo Dr David Wilson)**

18 Even if peatlands cover only a very small area (3% of the global land area) they contain one third of all
19 soil carbon. This makes peatland conservation and restoration a key priority for climate policy (*very*
20 *high confidence*), but progress so far is meagre particularly in the tropics.

21 Because natural peatlands (i.e. hydrologically intact with vegetation cover) are more resilient to climate
22 changes, their conservation will help reduce severe climatic feedback risks. For degraded peatlands,
23 rewetting and restoration is essential to any strategy that seeks to mitigate the possibility of enhanced
24 physical degradation of wide area of peatlands across the globe with the major effects that this would
25 entail on C cycling, biodiversity and water quality.

1 **4.11.5 Hurricane induced land degradation**

2 Despite tropical cyclones are normal disturbances that natural ecosystems have been affected by and
3 recovered from for millennia, the characteristics of this natural disturbances have changed or will
4 change in a warming climate (Knutson et al. 2010). Large amplitude fluctuations in the frequency and
5 intensity complicate both the detection and attribution of tropical cyclones to climate change. Yet, the
6 intensity and frequency of high-intensity hurricanes have increased and are expected to increase further
7 due to global climate change (Knutson et al. 2010; Bender et al. 2010; Vecchi et al. 2008; Bhatia et al.
8 2018; Tu et al. 2018) (*medium agreement, robust evidence*). Tropical cyclone paths are also shifting
9 towards the poles increasing the area subject to tropical cyclones (Sharmila and Walsh 2018) Climate
10 change alone will affect the hydrology of individual wetland ecosystems mostly through changes in
11 precipitation and temperature regimes with great global variability (Erwin 2009). Over the last seven
12 decades, the speed at which tropical cyclones move has decreased significantly as expected from theory,
13 exacerbating the damage on local communities from increasing rainfall amounts (Kossin 2018).
14 Additionally, the sea level rise can negatively affect the capacity of the wetlands to prevent salinisation
15 of freshwater aquifer and flooding in coastal areas, by negatively impacting the peat accretion in coastal
16 wetlands (Glaser et al. 2012). Tropical cyclones will accelerate changes in coastal forest structure and
17 composition. The heterogeneity of land degradation at coasts that are affected by tropical cyclones can
18 be further enhanced by the interaction of its components (for example, rainfall, wind speed, and
19 direction) with topographic and biological factors (for example, species susceptibility)(Luke et al.
20 2016).

21 Wetlands not only provide food, water and shelter for fish, birds and other wildlife, but also provide
22 important ecosystem services such as water quality improvement, flood abatement and carbon
23 sequestration (Meng et al. 2017). Additionally, coastal wetland regions are under serious threat and
24 have been suffering from severe degradation. Coastal wetlands function as valuable, self-maintaining
25 “horizontal levees” for storm protection, and also provide a host of other ecosystem services that vertical
26 levees do not.

27 Despite its importance coastal wetlands are listed amongst the most heavily damaged of natural
28 ecosystems worldwide. Starting in the 1990s, wetland restoration and re-creation became a “hotspot”
29 in the ecological research fields (Zedler 2000). The coastal wetland restoration and preservation is an
30 extremely cost-effective strategy for society. In the US were estimated to provide USD 23.2 billion
31 yr⁻¹ in storm protection services (Costanza et al. 2008). Additionally, carbon stocks of mangroves only,
32 average $283 \pm 193 \text{ Mg C}_{\text{org}} \text{ ha}^{-1}$. The global carbon stock is estimated to be 9.5 Pg C_{org} (Atwood et al.
33 2017). Thus, the maintenance of this wetlands is critical to prevent coastal degradation. Floodplains,
34 mangroves, seagrasses, saltmarshes, arctic wetlands, peatlands, freshwater marshes and forests are very
35 diverse habitats, with different stressors and hence different management and restoration techniques are
36 needed.

37 The Sundarban, one of the world’s largest coastal wetlands, covers about one million hectares between
38 Bangladesh (approximately 60%) and India (approximately 40%). Large areas of the Sundarban
39 mangroves have been converted into paddy fields over the past two centuries and more recently into
40 shrimp farms. The surroundings of the Sundarban mangroves are some of the most densely populated
41 areas in the world. More than half of the population is impoverished on the Indian side and depend
42 heavily on the goods and services that the forests provide (Ghosh et al. 2015).

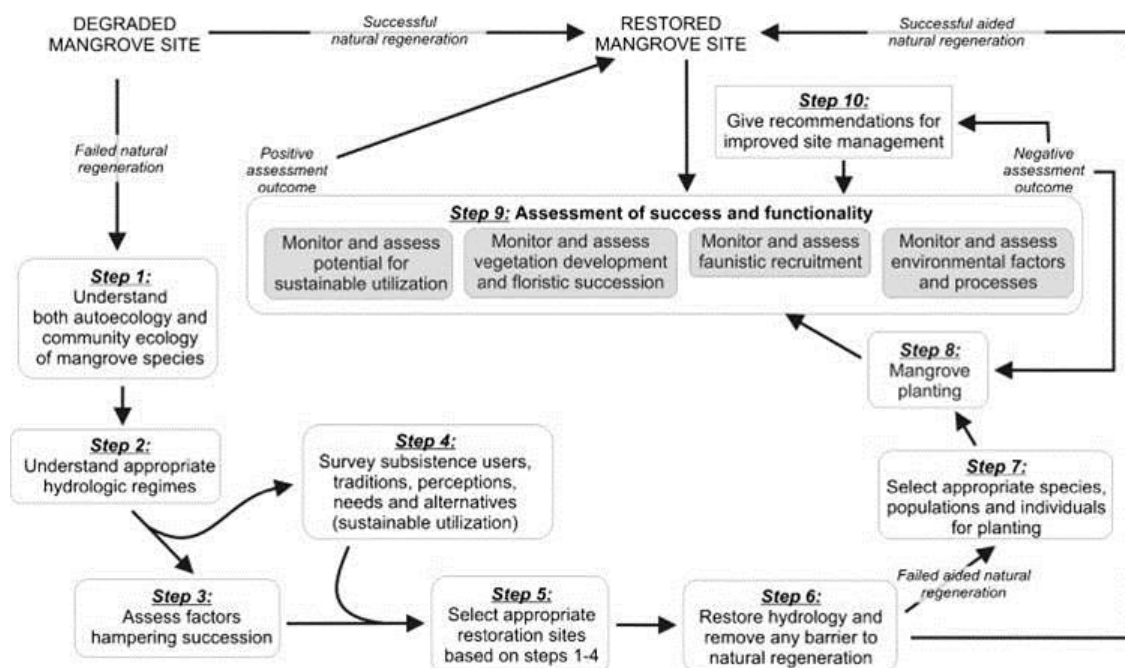
43 In 2009 the cyclone Aila caused incremental stresses on the socioeconomic conditions of the Sundarban
44 coastal communities through rendering huge areas of land unproductive for a long time (Abdullah et al.
45 2016). The impact of Aila was wide spread throughout the Sundarbans mangroves showing changes
46 between pre- and post-cyclonic period of 20-50% in the enhanced vegetation index (Dutta et al. 2015).
47 Although the magnitude of the effects of the Sundarban mangroves derived from climate change is not
48 yet defined (Payo et al. 2016; Loucks et al. 2010; Gopal and Chauhan 2006; Ghosh et al. 2015;

1 Chaudhuri et al. 2015). There is a *high agreement* that the joint effect of climate change and land
 2 degradation will be very negative for the area, strongly affecting the environmental services provided
 3 by these forests, including the extinction of large mammal species (Loucks et al. 2010). This changes
 4 in vegetation are mainly due to inundation and erosion (Payo et al. 2016).

5 The existing mangrove management strategy in the Sundarban is a combination of conservation through
 6 legislative policy, community awareness and sustainable exploitation of forest resources through
 7 cooperative management (Chaudhuri et al. 2015). As a result, mangrove areas are slightly increasing
 8 because of aggradation, plantation, and regrowth on the Sundarban area in the last decades. Despite of
 9 this aggradation, mangrove forest in the Sundarbans decreased by 1.2%, since the 2.5% of forest loss,
 10 due to conversion to agriculture, urban development, shrimp ponds, over harvesting of fruits, and
 11 garbage disposal in Mumbai, India (Giri et al. 2015).

12 There is a *high agreement* with *medium evidence* that the success of wetland restoration depends
 13 mainly on the flow of the water through the system and the degree to which re-flooding occurs, the
 14 disturbance regimes, and the control of invasive species (Burlakova et al. 2009; López-Rosas et al.
 15 2013). The implementation of the Ecological Mangrove Rehabilitation (EMR) protocol (López-
 16 Portillo et al. 2017) that includes monitoring and reporting tasks (Figure 4.16), has been probe to
 17 deliver successful rehabilitation of wetland ecosystem services (Figure 4.17).

18



19

20 **Figure 4.16 Decision tree showing recommended steps and tasks to restore a mangrove wetland based on**
 21 **original site conditions (From Bosire et al. (2008))**

22



1

2 **Figure 4.17 Hydrological restoration implemented in mangrove wetlands in Pichavaram, Tamil Nadu,**
3 **India, showing original main and feeder channels excavated circa 1996. (a): March 3, 2003; (b): January**
4 **29, 2016 (Source: Google Earth Pro; image area: 55.5 ha; eye altitude 881 m; Latitude: 11°25'59.86" N,**
5 **longitude: 79°47'28.89" E at the center of the images (López-Portillo et al. 2017)**

6 **4.11.6 Saltwater intrusion**

7 Current environmental changes, including climate change, have caused sea level to rise worldwide,
8 particularly in the tropical and subtropical regions. Combined with scarcity of water in river channels,
9 such rises have been instrumental in the intrusion of highly saline seawater inland posing a threat to
10 coastal areas and an emerging challenge to land managers and policy makers. Assessing the extent of
11 salinisation due to sea water intrusion at a global scale nevertheless remains challenging. (Wicke et al.
12 2011) suggest that across the world, approximately 1.1 Gha of land is affected by salt, with 14% of this
13 categorised as forest, wetland or some other form of protected area. Seawater intrusion is generally
14 caused by: i) increased tidal activity including storm surges, hurricanes, sea storms, due to changing
15 climate, ii) heavy groundwater extraction or land use changes as a result of changes in precipitation,
16 and droughts/floods, iii) coastal erosion as a result of destruction of mangrove forests and wetlands and
17 iv) construction of vast irrigation canals and drainage networks leading to low river discharge in the

1 deltaic region.

2 The Indus delta, located in the south-eastern coast of Pakistan near Karachi in the North Arabian sea,
3 is one of the six largest estuaries in the world spanning over an area of 600,000 ha. The Indus delta is a
4 clear example of seawater intrusion and land degradation due to local as well as up country climatic
5 and environmental conditions (Rasul et al. 2012) Abbas et al., 2017). The salinisation and waterlogging
6 in the upcountry areas including provinces of Punjab and Sindh is, however, caused by from the
7 irrigation network and over-irrigation (Waheed-uz-Zaman, 2000).

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

13 Over 480,000 ha of fertile land is now affected by sea water intrusion, wherein eight coastal
14 subdivisions of the districts of Badin and Thatta are mostly affected (Chandio, N. H.; M. M. Anwar,
15 M.M. & Chandio 2011). A very high intrusion rate of 0.179 ± 0.0315 km yr⁻¹, based on the analysis of
16 satellite data, was observed in the Indus delta during the past 10 years (2004–2015) (Kalhoro, N.A.; He,
17 Z.; Xu, D.; Faiz, M.; Yafei, L.V.; . Sohoo 2016). The area of agricultural crops under cultivation has
18 been declining with economic losses of millions of USD (IUCN 2003). Crop yields have reduced due
19 to soil salinity, in some places failing entirely. Soil salinity varies seasonally, depending largely on the
20 river discharge; during the wet season (August 2014), salinity (0.18 mg/L) reached 24 km upstream
21 while during the dry season (May 2013), it reached 84 km upstream (Kalhoro, N.A.; He, Z.; Xu, D.;
22 Faiz, M.; Yafei, L.V.; . Sohoo 2016). The freshwater aquifers have also been contaminated with sea
23 water rendering them unfit for drinking or irrigation purposes. Lack of clean drinking water and
24 sanitation causes widespread diseases, of which diarrhoea is most common (IUCN 2003).

25 Lake Urmia in northwest Iran, the second largest saltwater lake in the world and the habitat for endemic
26 Iranian brine shrimp, *Artemia urmiana*, has also been affected by salty water intrusion (Figure 4.18).
27 During a 17-year period between 1998 and 2014, human disruption and years of dam building have
28 affected the natural flow of freshwater as well as salty sea water in the surrounding area of Lake Urmia
29 (Figure 4.18). Water quality has also been adversely affected with salinity fluctuating over time, but in
30 recent years reaching a maximum of 340 g/L. This has rendered the underground water unfit for
31 drinking and agricultural purposes and risky to human health and livelihoods. Adverse impacts of global
32 climate change as well as direct human impacts have caused changes in land use, overuse of
33 underground water resources and construction of dams over rivers which resulted in drying-up of lake
34 in large part. This condition created sand, dust and salt storms in the region which affected many sectors
35 including agriculture, water resources, rangelands, forests, health, etc. and generally provided
36 desertification conditions around the lake (Karbassi et al. 2010; Marjani and Jamali 2014; Shadkam et
37 al. 2016).

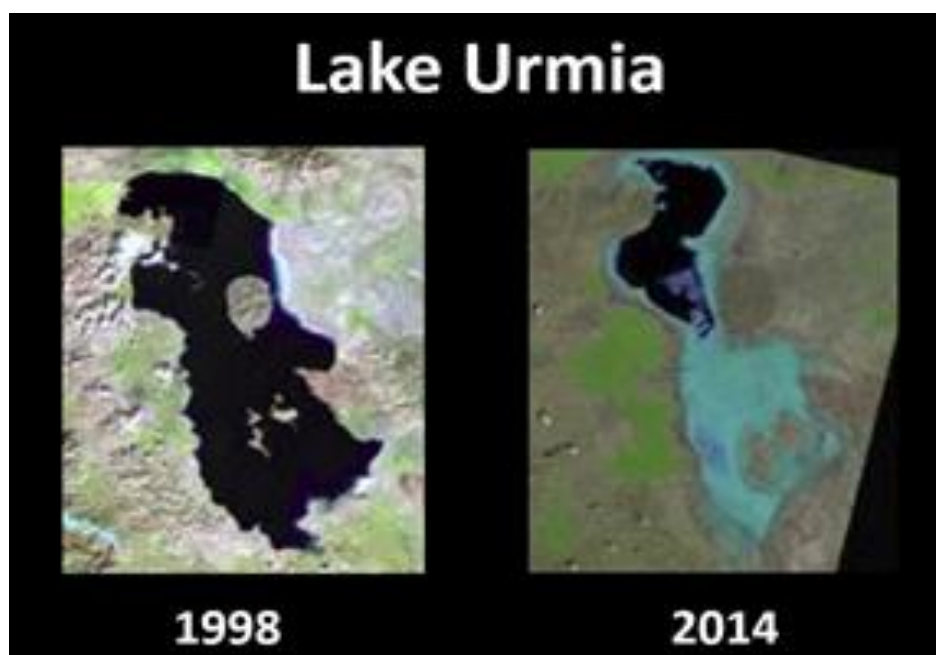


Figure 4.18 Urmia Lake condition in years 1998 and 2014. The dark blue colour shows water area of lake and the light blue colour shows dried-up parts

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

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

In some other countries, intrusion of seawater is exacerbated by destruction of mangrove forests. Mangroves are important coastal ecosystems that provide spawning bed for fish, timber for building, livelihood to dependent communities, act as barriers against coastal erosion, storm surges, tropical cyclones and tsunamis (Kalhor, N.A.; He, Z.; D. Xu.; I.; Muhammad, A.F.; Sohoo 2017) and are among the most carbon-rich stocks on Earth (Atwood et al. 2017). They nevertheless face a variety of threats: climatic (storm surges, tidal activities, high temperatures) and human (coastal developments, pollution, deforestation, conversion to aquaculture, rice culture, oil palm plantation), leading to declines in their areas. In Pakistan, using remote sensing (RS), the mangrove forest cover in the Indus delta has decreased from 260,000 ha in 1980s to 160,000 ha in 1990 (Chandio, N. H.; M. M. Anwar, M.M. & Chandio 2011). Based on remotely sensed 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 1997 (Etemadi et al. 2016). Myanmar has the highest rate (about 1% yr⁻¹) of mangrove deforestation in the world (Atwood et al. 2017). Regarding global loss of carbon stored in the mangrove soil due to deforestation, four countries exhibited high levels of loss: Indonesia (3,410 Gg CO₂ yr⁻¹), Malaysia (1,288 GgCO₂ yr⁻¹), US (206 Gg

1 CO₂ yr⁻¹) and Brazil (186 GgCO₂ yr⁻¹). Only in Bangladesh and Guinea Bissau there was no decline in
2 the mangrove area from 2000 to 2012 (Atwood et al. 2017).

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

9 **4.11.7 Biochar**

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

14 **4.11.7.1 Role of biochar in climate change mitigation**

15 Biochar is relatively resistant to decomposition compared with fresh organic matter or compost, so
16 represents a long-term C store (*very high confidence*). Biochars produced at higher temperature (>
17 450°C) and from woody material have greater stability than those produced at lower temperature (300-
18 450°C), and from manures (*very high confidence*) (Singh et al. 2012; Wang et al. 2016). Biochar
19 stability is influenced by soil properties: biochar carbon can be further stabilised by interaction with
20 clay minerals and native soil organic matter (*medium evidence*) (Fang et al. 2015). Biochar stability is
21 estimated to range from decades to thousands of years, for different biochars in different applications
22 (Singh et al., 2015; Wang et al., 2016). Biochar stability decreases as ambient temperature increases
23 (*limited evidence*) (Fang et al. 2017).

24 Biochar can enhance soil carbon stocks through “negative priming”, in which rhizodeposits are
25 stabilised through sorption of labile C on biochar, and formation of biochar-organo-mineral complexes
26 (Weng et al. 2015, 2017, 2018; Wang et al. 2016). Conversely, some studies show increased turnover
27 of native soil carbon (“positive priming”) due to enhanced soil microbial activity induced by biochar.
28 In clayey soils, positive priming is minor and short-lived compared to negative priming effects, which
29 dominate in the medium to long-term (Singh and Cowie 2014; Wang et al. 2016). Negative priming has
30 been observed particularly in in loamy grassland soil (Ventura et al. 2015) clay-dominated soils,
31 whereas positive priming is reported in sandy soils (Wang et al. 2016) and those with low C content
32 (Ding et al. 2018). Thus, biochar carbon stability is greatest in clay soils in temperate environments;
33 biochar stimulates negative priming and therefore builds soil carbon in clay soils especially in temperate
34 environments but can stimulate loss of native soil carbon in sandy, low carbon soils and at higher
35 ambient temperatures.

36 Biochar can provide additional climate change mitigation by decreasing nitrous oxide (N₂O) emissions
37 from soil, due in part to decreased substrate availability for denitrifying organisms, related to the molar
38 H/C ratio of the biochar (Cayuela et al. 2015). However, this impact varies widely: meta-analyses found
39 an average decrease in N₂O emissions from soil of 30-54%, (Cayuela et al. 2015) (Moore 2002) ,
40 though another study found an average of 0% (Verhoeven et al. 2017). Biochar can reduce methane
41 emissions from flooded soils, such as rice paddies, but also reduces methane uptake by dryland soils
42 though the latter effect is small in absolute terms (Jeffery et al. 2016).

43 Additional climate benefits of biochar can arise through reduced N fertiliser requirements, due to
44 reduced losses of N through leaching and/or volatilization (Singh, Hatton, Balwant, & Cowie, 2010);
45 increased yields of crop, forage, vegetable and tree species (Biederman and Stanley Harpole 2013),
46 particularly in sandy soils and acidic tropical soils (Simon et al. 2017) ; avoided GHG emissions from

1 manure that would otherwise be stockpiled, crop residues that would be burned or processing residues
2 that would be landfilled; and reduced GHG emissions from compost (Agyarko-Mintah et al. 2017) Wu
3 et al., 2017).

4 Climate benefits of biochar could be substantially reduced through reduction in albedo if biochar is
5 surface-applied at high rates to light-coloured soils (Genesio et al. 2012; Bozzi et al. 2015; Woolf et al.
6 2010), or if black carbon dust is released (Genesio et al. 2016). Pelletising or granulating biochar, and
7 applying below the soil surface or incorporating into the soil, minimizes the release of black carbon
8 dust and reduces the effect on albedo (Woolf et al. 2010).

9 Biochar is a potential “negative emissions” technology: the thermochemical conversion of biomass to
10 biochar slows mineralisation of the biomass, delivering long term C storage; gases released during
11 pyrolysis can be combusted for heat or power, displacing fossil energy sources, and could be captured
12 and sequestered if linked with infrastructure for carbon capture and storage (Smith 2016). Studies of
13 the life cycle climate change impacts of biochar systems generally show emissions reduction in the
14 range 0.4 -1.2Mg CO₂e Mg⁻¹ (dry) feedstock (Cowie, A.; Woolf, A.D.; Gaunt, J.; Brandão, M.; de la
15 Rosa 2015). Use of biomass for biochar can deliver greater benefits than use for bioenergy, if applied
16 in a context where it delivers agronomic benefits and/or reduces non-CO₂ GHG emissions (Woolf et
17 al., 2010; Woolf et al, 2018). A global analysis of technical potential, in which biomass supply
18 constraints were applied to protect against food insecurity, loss of habitat and land degradation,
19 estimated technical potential abatement of 3.7 - 6.6Pg CO₂e yr⁻¹ (including 2.6-4.6 GtCO₂e yr⁻¹ carbon
20 stabilization), with theoretical potential to reduce total emissions over the course of a century by 240 –
21 475 Pg CO₂e (Woolf et al. 2010). Fuss et al. 2018 propose a range of 0.5-2 GtCO₂e as the sustainable
22 potential for negative emissions through biochar.

23 **4.11.7.2 Role of biochar in management of land degradation**

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

- 28 • Improved nutrient use efficiency: biochars can reduce leaching of nitrate and ammonium (eg
29 (Haider et al. 2017) and increase availability of phosphorus (P) in soils with high P fixation
30 capacity (Liu et al. 2018b), potentially reducing N and P fertiliser requirements.
- 31 • Management of heavy metals and organic pollutants: application of biochar can substantially
32 reduce bioavailability of toxic elements (O’Connor et al., 2018; Peng ; Deng, ; Peng, & Yue,
33 2018), by reducing availability, through immobilisation due to increased pH and redox effects
34 (Rizwan et al. 2016) and adsorption on biochar surfaces (Zhang et al. 2013) thus providing a
35 means of remediating contaminated soils, and enabling their utilisation for food production.
- 36 • Stimulation of beneficial soil organisms, including earthworms and mycorrhizal fungi (Thies
37 et al. 2015).
- 38 • Improved porosity and water holding capacity (Quin et al. 2014), particularly in sandy soils
39 (Omondi et al. 2016), enhancing microbial function during drought (Paetsch et al. 2018).

40

41 Biochar systems can deliver a range of other co-benefits including destruction of pathogens and weed
42 propagules, avoidance of landfill, improved handling and transport of wastes such as sewage sludge,
43 management of biomass residues such as environmental weeds and urban greenwaste, reduction of
44 odors and management of nutrients from intensive livestock facilities, reduction in environmental N
45 pollution and protection of waterways. As a compost additive, biochar can reduce leaching and
46 volatilization of nutrients, increasing nutrient retention, through absorption and adsorption processes
47 (Joseph et al. 2018).

1 While there are many studies reporting positive responses, some studies have found negative or zero
2 impacts on soil properties or plant response. Pyrolysis of biomass leads to losses of volatile nutrients,
3 especially N. While availability of N and P in biochar is lower in biochar than in fresh biomass (Xu et
4 al. 2016) the impact of biochar on plant uptake is determined by the interactions between biochar, soil
5 minerals and activity of microorganisms (e.g. (Vanek and Lehmann 2015); (Nguyen et al. 2017). To
6 avoid negative responses it is important to select biochar formulations to address known soil constraints,
7 and to apply biochar prior to planting (Nguyen et al., 2017). Nutrient enrichment improves the
8 performance of biochar from low nutrient feedstocks (Joseph et al. 2013). While there are many reports
9 of biochar reducing disease or pest incidence, there are also reports of nil or negative effects (Bonanomi
10 et al. 2015). Biochar may induce systemic disease resistance (e.g., Elad et al. 2011)), though (Viger et
11 al. 2015) reported down-regulation of plant defence genes, suggesting increased susceptibility to insect
12 and pathogen attack. Disease suppression where biochar is applied is associated with increased
13 microbial diversity and metabolic potential of the rhizosphere microbiome (Kolton et al. 2017).
14 Differences in properties related to feedstock (Bonanomi et al. 2018) and differential response to
15 biochar dose, with lower rates more effective (Frenkel et al. 2017) contribute to variable disease
16 responses.

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

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

31 Key message: Application of biochar to soil can improve soil chemical, physical and biological
32 attributes, enhancing productivity and resilience to climate change, while also delivering climate change
33 mitigation through carbon sequestration and reduction in GHG emissions (*medium agreement, robust
34 evidence*). However, responses to biochar depend on biochar properties, in turn dependent on feedstock
35 and biochar production conditions, and the soil and crop to which it is applied. Negative or nil results
36 have been recorded. Agronomic and methane reduction benefits appear greatest in tropical regions,
37 while carbon stabilisation is greater in temperate regions. Biochar is most effective when applied in low
38 volumes to the most responsive soils and when properties are matched to the specific soil constraints
39 and plant needs. Biochar is thus a practice that has potential to address land degradation and climate
40 change simultaneously, while also supporting sustainable development. The potential of biochar is
41 limited by the availability of biomass for its production. Biochar production and use requires regulation
42 and standardisation to manage risks (*strong agreement*).

43 **4.11.8 Avoiding coastal degradation induced by maladaptation to sea-level rise on small** 44 **islands**

45 Coastal degradation—for example, beach erosion, coastal squeeze, and coastal biodiversity loss—as a
46 result of rising sea levels is a major concern for low lying coasts and small islands (*high confidence*).
47 The contribution of climate change to increased coastal degradation has been well documented in AR5

1 (Nurse et al. 2014; Wong et al. 2014) and is further discussed in Section 4.6.1.4 as well as in the IPCC
2 Special Report on the Ocean and Cryosphere in a Changing Climate (SROCC). However, coastal
3 degradation can also be indirectly induced by climate change as the result of adaptation measures that
4 involve changes to the coastal environment, for example, coastal protection measures against increased
5 flooding and erosion due to sea level rise and storm surges transforming the natural coast to a ‘stabilised’
6 coastline (Cooper and Pile 2014; French 2001). Every kind of adaptation response option is context-
7 dependent, and, in fact, sea walls play an important role for adaptation in many places. Nonetheless,
8 there are observed cases where the construction of sea walls can be considered ‘maladaptation’ (Barnett
9 and O’Neill 2010; Magnan et al. 2016) by leading to increased coastal degradation, such as in the case
10 of small islands, where due to limitations of space coastal retreat is less of an option than in continental
11 coastal zones. There is emerging literature on the implementation of alternative coastal protection
12 measures and mechanisms on small islands to avoid coastal degradation induced by sea walls (e.g.,
13 Mycoo and Chadwick 2012; Sovacool 2012).

14 In many cases, increased rates of coastal erosion due to the construction of sea walls are the result of
15 the negligence of local coastal morphological dynamics and natural variability as well as the interplay
16 of environmental and anthropogenic drivers of coastal change (*medium evidence, high agreement*). Sea
17 walls in response to coastal erosion may be ill-suited for extreme wave heights under cyclone impacts
18 and can lead to coastal degradation by keeping overflowing sea water from flowing back into the sea,
19 and therefore affect the coastal vegetation through saltwater intrusion, as observed in Tuvalu
20 (Government of Tuvalu 2006; Wairiu 2017). Similarly, in Kiribati, poor construction of sea walls has
21 resulted in increased erosion and inundation of reclaimed land (Donner 2012; Donner and Webber
22 2014). In the Comoros and Tuvalu, sea walls have been constructed from climate change adaptation
23 funds and ‘often by international development organisations seeking to leave tangible evidence of their
24 investments’ (Marino and Lazrus 2015, p. 344). In these cases, they have even increased coastal erosion,
25 due to poor planning and the negligence of other causes of coastal degradation, such as sand mining
26 (Marino and Lazrus 2015; Betzold and Mohamed 2017; Ratter et al. 2016). On the Bahamas, the
27 installation of sea walls as a response to coastal erosion in areas with high wave action has led to the
28 contrary effect and even increased sand loss in those areas (Sealey 2006). The reduction of natural
29 buffer zones—i.e., beaches and dunes—due to vertical structures, such as sea walls, increased the
30 impacts of tropical cyclones on Reunion Island (Duvat et al. 2016). Such a process of ‘coastal squeeze’
31 (Pontee 2013) also results in the reduction of intertidal habitat zones, such as wetlands and marshes
32 (Linham and Nicholls 2010). Coastal degradation resulting from the construction of sea walls, however,
33 is not only observed in Small Island Developing States (SIDS), as described above, but also on islands
34 in the Global North, for example, the North Atlantic (Muir et al. 2014; Young et al. 2014; Cooper and
35 Pile 2014; Bush 2004).

36 The adverse effects of coastal protection measures may be avoided by the consideration of local social-
37 ecological dynamics, including the critical studying of diverse drivers of ongoing shoreline changes,
38 and the according implementation of locally adequate coastal protection options (French 2001; Duvat
39 2013). Critical elements for avoiding maladaptation include profound knowledge of local tidal regimes,
40 availability of relative sea level rise scenarios and projections for extreme water levels. Moreover, the
41 downdrift effects of sea walls need to be considered, since undefended coasts may be exposed to
42 increased erosion (Linham and Nicholls 2010). In some cases, it may be possible to keep intact and
43 restore natural buffer zones as an alternative to the construction of hard engineering solutions.
44 Otherwise, changes in land use, building codes, or even coastal realignment can be an option in order
45 to protect and avoid the loss of the buffer function of beaches (Duvat et al. 2016; Cooper and Pile 2014).
46 Examples of Barbados show that combinations of hard and soft coastal protection approaches can be
47 sustainable and reduce the risk of coastal ecosystem degradation while keeping the desired level of
48 protection for coastal users (Mycoo and Chadwick 2012). Nature-based solutions and approaches such
49 as ‘building with nature’ (Slobbe et al. 2013) may allow for more sustainable coastal protection

1 mechanisms and avoid coastal degradation. Examples from the Maldives, several Pacific islands and
2 the North Atlantic show the importance of the involvement of local communities in coastal adaptation
3 projects, considering local skills, capacities, as well as demographic and socio-political dynamics, in
4 order to ensure the proper monitoring and maintenance of coastal adaptation measures (Sovacool 2012;
5 Muir et al. 2014; Young et al. 2014; Buggy and McNamara 2016; Petzold 2016).

7 **4.12 Knowledge gaps and key uncertainties**

8 The co-benefits of improved land management, such as mitigation of climate change, increased climate
9 resilience of agriculture, and impacts on rural areas/societies are well known in theory but there is a
10 lack of a coherent and systematic global inventory.

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

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

16 Attribution is a challenge because a complex web of causality rather than simple cause-effect
17 relationships. Also diverging views on land degradation in relation to other challenges is hampering
18 such efforts.

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

21 Land management actions that successfully reverse land degradation, lead to sustainable land
22 management and maintenance of ecosystem services need to be better documented and monitored.

23 Synergies between climate mitigation and adaptation strategies are poorly understood, yet will be
24 essential to avoid, reduce and reverse land degradation.

26 **Frequently Asked Questions**

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

28 Climate change, land degradation, and land use are linked in a complex web of causality. One important
29 impact of climate change on land degradation is that increasing global temperatures intensify the
30 hydrological cycle resulting in more intense rainfall which is an important driver of soil erosion. This
31 means that sustainable land management (SLM) becomes even more important with climate change.
32 Land-use change in the form of clearing of forest for rangeland and cropland (e.g., for provision of bio-
33 fuels) is a major source of greenhouse gas emission from both above and below ground. Many SLM
34 practices (e.g., agroforestry, shifting to no-till agriculture, or perennial crops) increase carbon content
35 of soil and vegetation cover and hence provide both local and immediate adaptation benefits combined
36 with global mitigation benefits in the long term, while providing many social and economic co-benefits.
37 Avoiding, reducing and reversing land degradation has a large potential to mitigate climate change and
38 help communities to adapt to climate change.

40 **FAQ 4.2 How does climate change affect land-related ecosystem services?**

41 Climate change will affect land-related ecosystem services (see Glossary for definition of ecosystem
42 services) globally and both directly and indirectly. The direct impacts range from subtle reductions or

1 enhancements of specific services, such as biological productivity, resulting from changes in
2 temperature, temperature variability or rainfall, to complete disruption and elimination of services.
3 Disruptions of ecosystem services can occur where climate change causes transitions from one biome
4 to another, e.g., forest to grassland as a result of changes in water balance or natural disturbance regimes.
5 Climate change can also alter the mix of land-related ecosystem services. While the net impacts are
6 specific to ecosystem types, ecosystem services and time, there is an asymmetry of risk such that overall
7 impacts of climate change are expected to reduce ecosystem services. Indirect impacts of climate change
8 on land-related ecosystem services include those that result from changes in human behaviour,
9 including potential large-scale implementation of afforestation, reforestation or other changes in land
10 management, which can have positive or negative outcomes on ecosystem services.

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