1.1 The nature of desertification

1.1.1 Introduction

In this report, desertification is defined as land degradation in arid, semi-arid, and dry sub-humid areas resulting from many factors, including climatic variations and human activities (United Nations Convention to Combat Desertification (UNCCD) 1994). Land degradation is a negative trend in land condition, caused by direct or indirect human-induced processes including anthropogenic climate change, expressed as long-term reduction or loss of at least one of the following: biological productivity, ecological integrity or value to humans (Section 4.1.3). Arid, semi-arid, and dry sub-humid areas, together with hyper-arid areas, constitute drylands (UNEP 1992), home to about 3 billion people (van der Esch et al. 2017). The difference between desertification and land degradation is not process-based but geographic. Although land degradation can occur anywhere across the world, when it occurs in drylands, it is considered desertification (FAQ 1.3). Desertification is not limited to irreversible forms of land degradation, nor is it equated to desert expansion, but represents all forms and levels of land degradation occurring in drylands.

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The geographic classification of drylands is often based on the aridity index (AI) – the ratio of average annual precipitation amount (P) to potential evapotranspiration amount (PET, see Glossary) (Figure 3.1). Recent estimates, based on AI, suggest that drylands cover about 46.2% (±0.8%) of the global land area (Koutroulis 2019; Prăvălie 2016) (*low confidence*). Hyper-arid areas, where the aridity index is below 0.05, are included in drylands, but are excluded from the definition of desertification (UNCCD 1994). Deserts are valuable ecosystems (UNEP 2006; Safriel 2009) geographically located in drylands and vulnerable to climate change. However, they are not considered prone to desertification. Aridity is a long-term climatic feature characterised by low average precipitation or available water (Gbeckor-Kove 1989; Türkeş 1999). Thus, aridity is different from drought, which is a temporary climatic event (Maliva and Missimer 2012). Moreover, droughts are not restricted to drylands, but occur both in drylands and humid areas (Wilhite et al. 2014). Following the Synthesis Report (SYR) of the IPCC Fifth Assessment Report (AR5), drought is defined here as "a period of abnormally dry weather long enough to cause a serious hydrological imbalance" (Mach et al. 2014) (Cross-Chapter Box 5 in this chapter).

AI is not an accurate proxy for delineating drylands in an increasing CO_2 environment (Section 3.2.1). The suggestion that most of the world has become more arid, since the AI has decreased, is not supported by changes observed in precipitation, evaporation or drought (Sheffield et al. 2012; Greve et al. 2014). While climate change is expected to decrease the AI due to increases in potential evaporation, the assumptions that underpin the potential evaporation calculation are not consistent with a changing CO_2 environment and the effect this has on transpiration rates (Roderick et al. 2015; Milly and Dunne 2016; Greve et al. 2017) (Section 3.2.1). Given that future climate is characterised by significant increases in CO_2, the usefulness of currently applied AI thresholds to estimate dryland areas is limited under climate change. If instead of the AI, other variables such as precipitation, soil moisture, and primary productivity are used to identify dryland areas, there is no clear indication that the extent of drylands will change overall under climate change (Roderick et al. 2015; Greve et al. 2017; Lemordant et al. 2018). Thus, some dryland borders will expand, while some others will contract (*high confidence*).

Approximately 70% of dryland areas are located in Africa and Asia (Figure 3.2). The biggest land use/cover in terms of area in drylands, if deserts are excluded, are grasslands, followed by forests and croplands (Figure 3.3). The category of 'other lands' in Figure 3.3 includes bare soil, ice, rock, and all other land areas that are not included within the other five categories (FAO 2016). Thus, hyper-arid areas contain mostly deserts, with some small exceptions, for example, where grasslands and croplands are cultivated under oasis conditions with irrigation (Section 3.7.4). Moreover, FAO (2016) defines grasslands as permanent pastures and meadows used continuously for more than five years. In drylands, transhumance, i.e. seasonal migratory grazing,

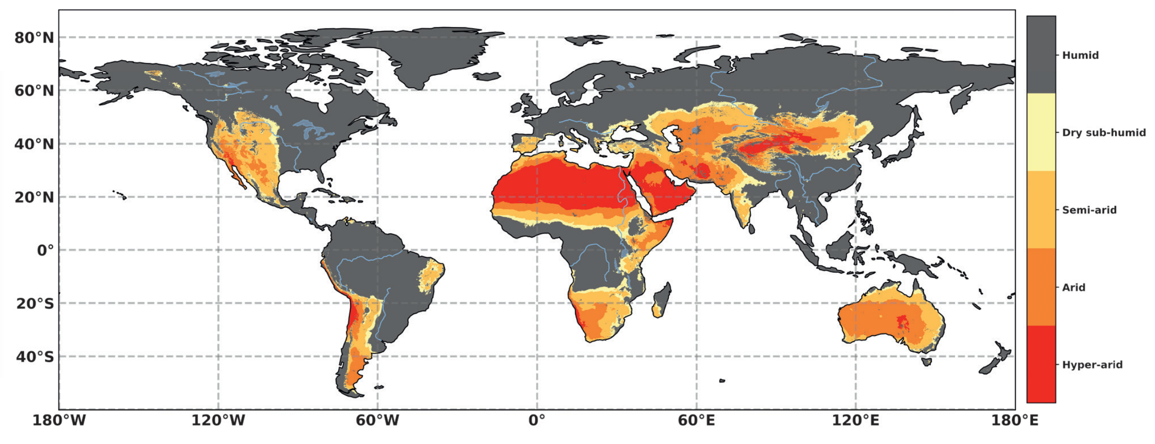

Figure 3.1 | Geographical distribution of drylands, delimited based on the aridity index (AI). The classification of AI is: Humid AI > 0.65, Dry sub-humid 0.50 < AI ≤ 0.65, Semi-arid 0.20 < AI ≤ 0.50, Arid 0.05 < AI ≤ 0.20, Hyper-arid AI < 0.05. Data: TerraClimate precipitation and potential evapotranspiration (1980–2015) (Abatzoglou et al. 2018).

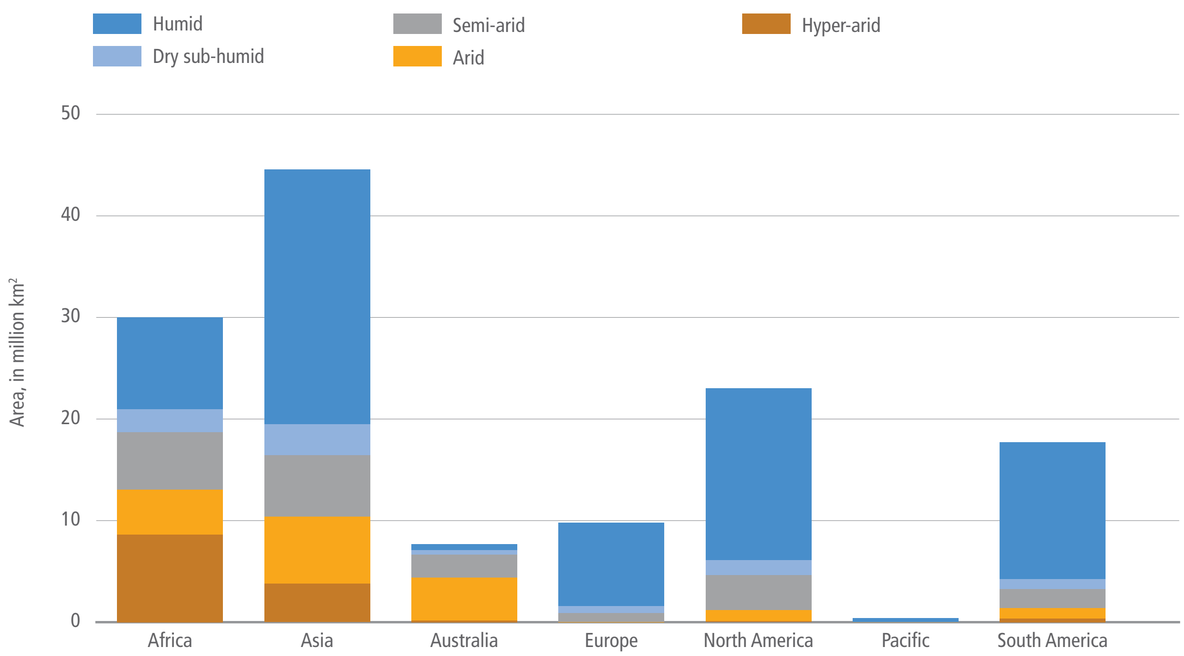

Figure 3.2 | Dryland categories across geographical areas (continents and Pacific region). Data: TerraClimate precipitation and potential evapotranspiration (1980–2015) (Abatzoglou et al. 2018).

often leads to non-permanent pasture systems, thus some of the areas under the 'other land' category are also used as non-permanent pastures (Ramankutty et al. 2008; Fetzel et al. 2017; Erb et al. 2016).

In the earlier global assessments of desertification (since the 1970s), which were based on qualitative expert evaluations, the extent of desertification was found to range between 4% and 70% of the area of drylands (Safriel 2007). More recent estimates, based on remotely sensed data, show that about 24–29% of the global land area experienced reductions in biomass productivity between the 1980s and 2000s (Bai et al. 2008; Le et al. 2016), corresponding to about 9.2% of drylands (±0.5%) experiencing declines in biomass productivity during this period (*low confidence*), mainly due to anthropogenic causes. Both of these studies consider rainfall dynamics, thus, accounting for the effect of droughts. While less than 10% of drylands is undergoing desertification, it is occurring in areas that contain around 20% of dryland population (Klein Goldewijk et al. 2017). In these areas the population has increased from approximately 172 million in 1950 to over 630 million today (Figure 1.1).

Available assessments of the global extent and severity of desertification are relatively crude approximations with considerable uncertainties, for example, due to confounding effects of invasive bush encroachment in some dryland regions. Different indicator sets and approaches have been developed for monitoring and assessment of desertification from national to global scales (Imeson 2012; Sommer et al. 2011; Zucca et al. 2012; Bestelmeyer et al. 2013). Many indicators of desertification only include a single factor or characteristic of desertification, such as the patch size distribution of vegetation (Maestre and Escudero 2009; Kéfi et al. 2010), Normalized Difference Vegetation Index (NDVI) (Piao et al. 2005), drought-tolerant plant species (An et al. 2007), grass cover (Bestelmeyer et al. 2013), land productivity dynamics (Baskan et al. 2017), ecosystem net primary productivity (Zhou et al. 2015) or Environmentally Sensitive Land Area Index (Symeonakis et al. 2016). In addition, some synthetic indicators of desertification have also been used to assess desertification extent and desertification processes, such as climate, land use, soil, and socio-economic parameters (Dharumarajan et al. 2018), or changes in climate, land use, vegetation cover, soil properties and population as the desertification vulnerability index (Salvati et al. 2009). Current data availability and methodological challenges do not allow for accurately and comprehensively mapping desertification at a global scale (Cherlet et al. 2018). However, the emerging partial evidence points to a lower global extent of desertification than previously estimated (*medium confidence*) (Section 3.2).

This assessment examines the socio-ecological links between drivers (Section 3.1) and feedbacks (Section 3.3) that influence desertification–climate change interactions, and then examines associated observed and projected impacts (Sections 3.4 and 3.5) and responses (Section 3.6). Moreover, this assessment highlights that dryland populations are highly vulnerable to desertification and climate change (Sections 3.2 and 3.4). At the same time, dryland populations also have significant past experience and sources of resilience embodied in indigenous and local knowledge and practices in order to successfully adapt to climatic changes and address desertification (Section 3.6). Numerous site-specific technological response options are also available for SLM in drylands that can help increase the resilience of agricultural livelihood systems to climate change (Section 3.6). However, continuing environmental degradation combined with climate change is straining the resilience of dryland populations. Enabling policy responses for SLM and livelihoods

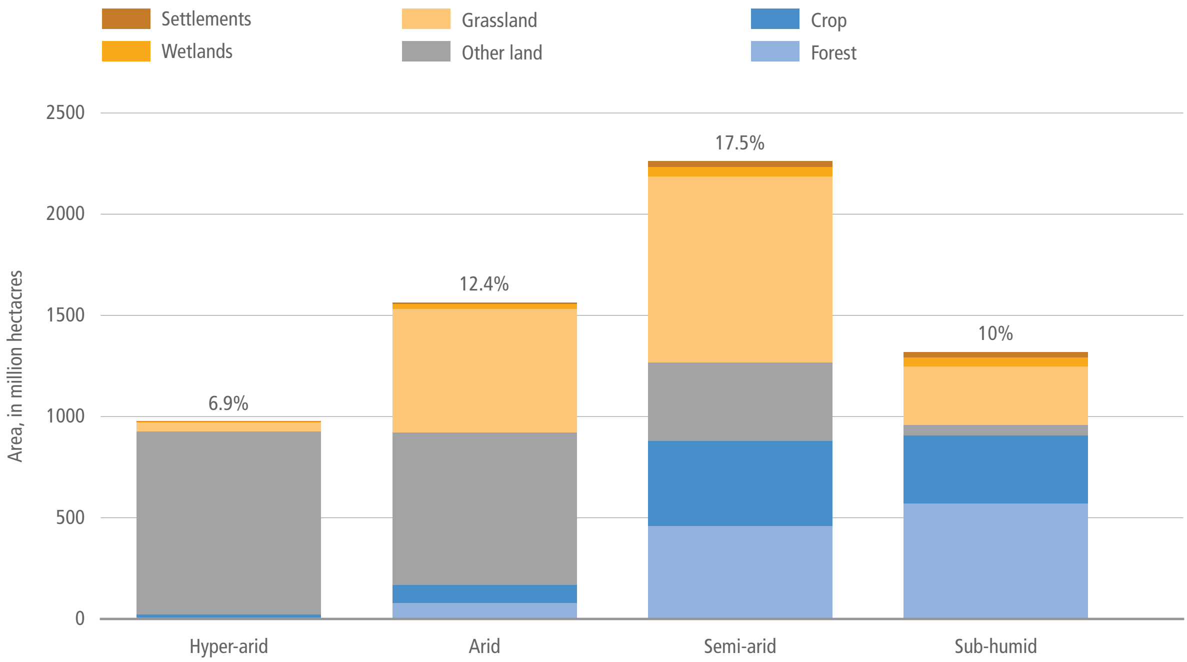

Figure 3.3 | Land use and land cover in drylands and share of each dryland category in global land area. Source: FAO (2016).

diversification can help maintain and strengthen the resilience and adaptive capacities in dryland areas (Section 3.6). The assessment finds that policies promoting SLM in drylands will contribute to climate change adaptation and mitigation, with co-benefits for broader sustainable development (*high confidence*) (Section 3.4).

1.1.2 Desertification in previous IPCC and related reports

The IPCC Fifth Assessment Report (AR5) and Special Report on Global Warming of 1.5°C include a limited discussion of desertification. In AR5 Working Group I desertification is mentioned as a forcing agent for the production of atmospheric dust (Myhre et al. 2013). The same report had *low confidence* in the available projections on the changes in dust loadings due to climate change (Boucher et al. 2013). In AR5 Working Group II desertification is identified as a process that can lead to reductions in crop yields and the resilience of agricultural and pastoral livelihoods (Field et al. 2014; Klein et al. 2015). AR5 Working Group II notes that climate change will amplify water scarcity, with negative impacts on agricultural systems, particularly in semi-arid environments of Africa (*high confidence*), while droughts could exacerbate desertification in southwestern parts of Central Asia (Field et al. 2014). AR5 Working Group III identifies desertification as one of a number of often overlapping issues that must be dealt with when considering governance of mitigation and adaptation (Fleurbaey et al. 2014). The IPCC Special Report on Global Warming of 1.5°C noted that limiting global warming to 1.5°C instead of 2°C is strongly beneficial for land ecosystems and their services (*high confidence*) such as soil conservation, contributing to avoidance of desertification (Hoegh-Guldberg et al. 2018).

The recent Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) Land Degradation and Restoration Assessment report (IPBES 2018a) is also of particular relevance. While acknowledging a wide variety of past estimates of the area undergoing degradation, IPBES (2018a) pointed at their lack of agreement about where degradation is taking place. IPBES (2018a) also recognised the challenges associated with differentiating the impacts of climate variability and change on land degradation from the impacts of human activities at a regional or global scale.

The third edition of the World Atlas of Desertification (Cherlet et al. 2018) indicated that it is not possible to deterministically map the global extent of land degradation – and its subset, desertification – pointing out that the complexity of interactions between social, economic, and environmental systems make land degradation not amenable to mapping at a global scale. Instead, Cherlet et al. (2018) presented global maps highlighting the convergence of various pressures on land resources.

1.1.3 Dryland populations: Vulnerability and resilience

Drylands are home to approximately 38.2% (±0.6%) of the global population (Koutroulis 2019; van der Esch et al. 2017), that is about 3 billion people. The highest number of people live in the drylands of South Asia (Figure 3.4), followed by Sub-Saharan Africa and Latin America (van der Esch et al. 2017). In terms of the number of people affected by desertification, Reynolds et al. (2007) indicated that desertification was directly affecting 250 million people. More recent estimates show that 500 (±120) million people lived in 2015 in those dryland areas which experienced significant loss in biomass

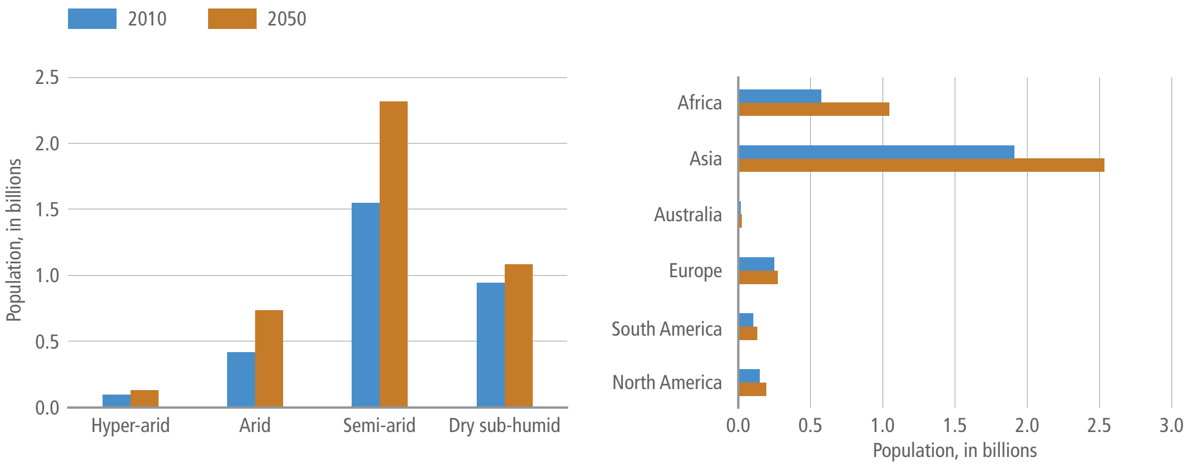

Figure 3.4 | Current and projected population (under SSP2) in drylands, in billions. Source: van der Esch et al. (2017).

productivity between the 1980s and 2000s (Bai et al. 2008; Le et al. 2016). The highest numbers of affected people were in South and East Asia, North Africa and the Middle East (*low confidence*). The population in drylands is projected to increase about twice as rapidly as non-drylands, reaching 4 billion people by 2050 (van der Esch et al. 2017). This is due to higher population growth rates in drylands. About 90% of the population in drylands live in developing countries (UN-EMG 2011).

Dryland populations are highly vulnerable to desertification and climate change because their livelihoods are predominantly dependent on agriculture, one of the sectors most susceptible to climate change (Rosenzweig et al. 2014; Schlenker and Lobell 2010). Climate change is projected to have substantial impacts on all types of agricultural livelihood systems in drylands (CGIAR-RPDS 2014) (Sections 3.4.1 and 3.4.2).

One key vulnerable group in drylands are pastoral and agropastoral households.[1] There are no precise figures about the number of people practicing pastoralism globally. Most estimates range between 100 million and 200 million (Rass 2006; Secretariat of the Convention on Biological Diversity 2010), of whom 30–63 million are nomadic pastoralists (Dong 2016; Carr-Hill 2013).[2] Pastoral production systems represent an adaptation to high seasonal climate variability and low biomass productivity in dryland ecosystems (Varghese and Singh 2016; Krätli and Schareika 2010), which require large areas for livestock grazing through migratory pastoralism (Snorek et al. 2014). Grazing lands across dryland environments are being degraded, and/or being converted to crop production, limiting the opportunities for migratory livestock systems, and leading to conflicts with sedentary crop producers (Abbass 2014; Dimelu et al. 2016). These processes, coupled with ethnic differences, perceived security threats, and misunderstanding of pastoral rationality, have led to increasing marginalisation of pastoral communities and disruption of their economic and cultural structures (Elhadary 2014; Morton 2010). As a result, pastoral communities are not well prepared to deal with increasing weather/climate variability and weather/climate extremes due to changing climate (Dong 2016; López-i-Gelats et al. 2016), and remain amongst the most food insecure groups in the world (FAO 2018).

There is an increasing concentration of poverty in the dryland areas of Sub-Saharan Africa and South Asia (von Braun and Gatzweiler 2014; Barbier and Hochard 2016), where 41% and 12% of the total populations live in extreme poverty, respectively (World Bank 2018). For comparison, the average share of global population living in extreme poverty is about 10% (World Bank 2018). Multidimensional poverty, prevalent in many dryland areas, is a key source of vulnerability (Safriel et al. 2005; Thornton et al. 2014; Fraser et al. 2011; Thomas 2008). Multidimensional poverty incorporates both income-based poverty, and also other dimensions such as poor healthcare services, lack of education, lack of access to water, sanitation and energy, disempowerment, and threat from violence (Bourguignon and Chakravarty 2003; Alkire and Santos 2010, 2014). Contributing elements to this multidimensional poverty in drylands are rapid population growth, fragile institutional environment, lack of infrastructure, geographic isolation and low market access, insecure land tenure systems, and low agricultural productivity (Sietz et al. 2011; Reynolds et al. 2011; Safriel and Adeel 2008; Stafford Smith 2016). Even in high-income countries, those dryland areas that depend on agricultural livelihoods represent relatively poorer locations nationally, with fewer livelihood opportunities, for example in Italy (Salvati 2014). Moreover, in many drylands areas, female-headed households, women and subsistence farmers (both male and female) are more vulnerable to the impacts of desertification and climate change (Nyantakyi-Frimpong and Bezner-Kerr 2015; Sultana 2014; Rahman 2013). Some local cultural traditions and patriarchal

1 Pastoralists derive more than 50% of their income from livestock and livestock products, whereas agropastoralists generate more than 50% of their income from crop production and at least 25% from livestock production (Swift, 1988).

2 The estimates of the number of pastoralists, and especially of nomadic pastoralists, are very uncertain, because often nomadic pastoralists are not fully captured in national surveys and censuses (Carr-Hill, 2013).

relationships were found to contribute to higher vulnerability of women and female-headed households through restrictions on their access to productive resources (Nyantakyi-Frimpong and Bezner-Kerr 2015; Sultana 2014; Rahman 2013) (Sections 3.4.2 and 3.6.3, and Cross-Chapter Box 11 in Chapter 7).

Despite these environmental, socio-economic and institutional constraints, dryland populations have historically demonstrated remarkable resilience, ingenuity and innovations, distilled into ILK to cope with high climatic variability and sustain livelihoods (Safriel and Adeel 2008; Davis 2016; Davies 2017) (Sections 3.6.1 and 3.6.2, and Cross-Chapter Box 13 in Chapter 7). For example, across the Arabian Peninsula and North Africa, informal community by-laws were successfully used for regulating grazing, collection and cutting of herbs and wood, and which limited rangeland degradation (Gari 2006; Hussein 2011). Pastoralists in Mongolia developed indigenous classifications of pasture resources which facilitated ecologically optimal grazing practices (Fernandez-Gimenez 2000) (Section 3.6.2). Currently, however, indigenous and local knowledge and practices are increasingly lost or can no longer cope with growing demands for land-based resources (Dominguez 2014; Fernández-Giménez and Fillat Estaque 2012; Hussein 2011; Kodirekkala 2017; Moreno-Calles et al. 2012) (Section 3.4.2). Unsustainable land management is increasing the risks from droughts, floods and dust storms (Sections 3.4.2 and 3.5). Policy actions promoting the adoption of SLM practices in dryland areas, based on both indigenous and local knowledge and modern science, and expanding alternative livelihood opportunities outside agriculture can contribute to climate change adaptation and mitigation, addressing desertification, with co-benefits for poverty eradication and food security (*high confidence*) (Cowie et al. 2018; Liniger et al. 2017; Safriel and Adeel 2008; Stafford-Smith et al. 2017).

1.1.4 Processes and drivers of desertification under climate change

1.1.4.1 Processes of desertification and their climatic drivers

Processes of desertification are mechanisms by which drylands are degraded. Desertification consists of both biological and non-biological processes. These processes are classified under broad categories of degradation of physical, chemical and biological properties of terrestrial ecosystems. The number of desertification processes is large and they are extensively covered elsewhere (IPBES 2018a; Lal 2016; Racine 2008; UNCCD 2017). Section 4.2.1 and Tables 4.1 and 4.2 in Chapter 4 highlight those which are particularly relevant for this assessment in terms of their links to climate change and land degradation, including desertification.

Drivers of desertification are factors which trigger desertification processes. Initial studies of desertification during the early-to-mid 20th century attributed it entirely to human activities. In one of the influential publications of that time, Lavauden (1927) stated that: "Desertification is purely artificial. It is only the act of the man…" However, such a uni-causal view of desertification was shown to be invalid (Geist et al. 2004; Reynolds et al. 2007) (Sections 3.1.4.2 and 3.1.4.3). Tables 4.1 and 4.2 in Chapter 4 summarise the drivers, linking them to the specific processes of desertification and land degradation under changing climate.

Erosion refers to removal of soil by the physical forces of water, wind, or often caused by farming activities such as tillage (Ginoux et al. 2012). The global estimates of soil erosion differ significantly, depending on scale, study period and method used (García-Ruiz et al. 2015), ranging from approximately 20 Gt yr^{-1} to more than 200 Gt yr^{-1} (Boix-Fayos et al. 2006; FAO 2015). There is a significant potential for climate change to increase soil erosion by water, particularly in those regions where precipitation volumes and intensity are projected to increase (Panthou et al. 2014; Nearing et al. 2015). On the other hand, while it is a dominant form of erosion in areas such as West Asia and the Arabian Peninsula (Prakash et al. 2015; Klingmüller et al. 2016), there is *limited evidence* concerning climate change impacts on wind erosion (Tables 4.1 and 4.2 in Chapter 4, and Section 3.5).

Saline and sodic soils (see Glossary) occur naturally in arid, semi-arid and dry sub-humid regions of the world. Climate change or hydrological change can cause soil salinisation by increasing the mineralised groundwater level. However, secondary salinisation occurs when the concentration of dissolved salts in water and soil is increased by anthropogenic processes, mainly through poorly managed irrigation schemes. The threat of soil and groundwater salinisation induced by sea level rise and seawater intrusion are amplified by climate change (Section 4.9.7).

Global warming is expected to accelerate soil organic carbon (SOC) turnover, since the decomposition of the soil organic matter by microbial activity begins with low soil water availability, but this moisture is insufficient for plant productivity (Austin et al. 2004) (Section 3.4.1.1). SOC is also lost due to soil erosion (Lal 2009); therefore, in some dryland areas leading to SOC decline (Sections 3.3.3 and 3.5.2) and the transfer of carbon (C) from soil to the atmosphere (Lal 2009).

Sea surface temperature (SST) anomalies can drive rainfall changes, with implications for desertification processes. North Atlantic SST anomalies are positively correlated with Sahel rainfall anomalies (Knight et al. 2006; Gonzalez-Martin et al. 2014; Sheen et al. 2017). While the eastern tropical Pacific SST anomalies have a negative correlation with Sahel rainfall (Pomposi et al. 2016), a cooler North Atlantic is related to a drier Sahel, with this relationship enhanced if there is a simultaneous relative warming of the South Atlantic (Hoerling et al. 2006). Huber and Fensholt (2011) explored the relationship between SST anomalies and satellite observed Sahel vegetation dynamics, finding similar relationships but with substantial west–east variations in both the significant SST regions and the vegetation response. Concerning the paleoclimatic evidence on aridification after the early Holocene 'Green Sahara' period (11,000 to 5000 years ago), Tierney et al. (2017) indicate that a cooling of the North Atlantic played a role (Collins et al. 2017; Otto-Bliesner et al. 2014; Niedermeyer et al. 2009) similar to that found in modern observations. Besides these SST relationships, aerosols have also been suggested as a potential driver of the Sahel droughts (Rotstayn and Lohmann 2002; Booth et al. 2012; Ackerley et al. 2011). For eastern Africa, both recent droughts and decadal declines have been linked to human-induced warming in the western Pacific (Funk et al. 2018).

Invasive plants contributed to desertification and loss of ecosystem services in many dryland areas in the last century (*high confidence*) (Section 3.7.3). Extensive woody plant encroachment altered runoff and soil erosion across much of the drylands, because the bare soil between shrubs is very susceptible to water erosion, mainly in high-intensity rainfall events (Manjoro et al. 2012; Pierson et al. 2013; Eldridge et al. 2015). Rising CO_2 levels due to global warming favour more rapid expansion of some invasive plant species in some regions. An example is the Great Basin region in western North America where over 20% of ecosystems have been significantly altered by invasive plants, especially exotic annual grasses and invasive conifers, resulting in loss of biodiversity. This land-cover conversion has resulted in reductions in forage availability, wildlife habitat, and biodiversity (Pierson et al. 2011, 2013; Miller et al. 2013).

The wildfire is a driver of desertification, because it reduces vegetation cover, increases runoff and soil erosion, reduces soil fertility and affects the soil microbial community (Vega et al. 2005; Nyman et al. 2010; Holden et al. 2013; Pourreza et al. 2014; Weber et al. 2014; Liu and Wimberly 2016). Predicted increases in temperature and the severity of drought events across some dryland areas (Section 2.2) can increase chances of wildfire occurrence (*medium confidence*) (Jolly et al. 2015; Williams et al. 2010; Clarke and Evans 2018) (Cross-Chapter Box 3 in Chapter 2). In semi-arid and dry sub-humid areas, fire can have a profound influence on observed vegetation and particularly the relative abundance of grasses to woody plants (Bond et al. 2003; Bond and Keeley 2005; Balch et al. 2013).

While large uncertainty exists concerning trends in droughts globally (AR5) (Section 2.2), examining the drought data by Ziese et al. (2014) for drylands only reveals a large inter-annual variability combined with a trend toward increasing dryland area affected by droughts since the 1950s (Figure 1.1). Thus, over the period 1961–2013, the annual area of drylands in drought has increased, on average, by slightly more than 1% per year, with large inter-annual variability.

1.1.4.2 Anthropogenic drivers of desertification under climate change

The literature on the human drivers of desertification is substantial (e.g., D'Odorico et al. 2013; Sietz et al. 2011; Yan and Cai 2015; Sterk et al. 2016; Varghese and Singh 2016) and there have been several comprehensive reviews and assessments of these drivers very recently (Cherlet et al. 2018; IPBES 2018a; UNCCD 2017). IPBES (2018a) identified cropland expansion, unsustainable land management practices including overgrazing by livestock, urban expansion, infrastructure development, and extractive industries as the main drivers of land degradation. IPBES (2018a) also found that the ultimate driver of land degradation is high and growing consumption of land-based resources, e.g., through deforestation and cropland expansion, escalated by population growth. What is particularly relevant in the context of the present assessment is to evaluate if, how and which human drivers of desertification will be modified by climate change effects.

Growing food demand is driving conversion of forests, rangelands, and woodlands into cropland (Bestelmeyer et al. 2015; D'Odorico et al. 2013). Climate change is projected to reduce crop yields across dryland areas (Sections 3.4.1 and 5.2.2), potentially reducing local production of food and feed. Without research breakthroughs mitigating these productivity losses through higher agricultural productivity, and reducing food waste and loss, meeting the increasing food demands of growing populations will require expansion of cropped areas to more marginal areas (with most prime areas in drylands already being under cultivation) (Lambin 2012; Lambin et al. 2013; Eitelberg et al. 2015; Gutiérrez-Elorza 2006; Kapović Solomun et al. 2018). Borrelli et al. (2017) showed that the primary driver of soil erosion in 2012 was cropland expansion. Although local food demands could also be met by importing from other areas, this would mean increasing the pressure on land in those areas (Lambin and Meyfroidt 2011). The net effects of such global agricultural production shifts on land condition in drylands are not known.

Climate change will exacerbate poverty among some categories of dryland populations (Sections 3.4.2 and 3.5.2). Depending on the context, this impact comes through declines in agricultural productivity, changes in agricultural prices and extreme weather events (Hertel and Lobell 2014; Hallegatte and Rozenberg 2017). There is *high confidence* that poverty limits both capacities to adapt to climate change and availability of financial resources to invest into SLM (Gerber et al. 2014; Way 2016; Vu et al. 2014) (Sections 3.5.2, 3.6.2 and 3.6.3).

Labour mobility is another key human driver that will interact with climate change. Although strong impacts of climate change on migration in dryland areas are disputed, in some places, it is *likely* to provide an added incentive to migrate (Section 3.4.2.7). Out-migration will have several contradictory effects on desertification. On one hand, it reduces an immediate pressure on land if it leads to less dependence on land for livelihoods (Chen et al. 2014; Liu et al. 2016a). Moreover, migrant remittances could be used to fund the adoption of SLM practices. Labour mobility from agriculture to non-agricultural sectors could allow land consolidation, gradually leading to mechanisation and agricultural intensification (Wang et al. 2014, 2018). On the other hand, this can increase the costs of labour-intensive SLM practices due to lower availability of rural agricultural labour and/or higher rural wages. Out-migration increases the pressure on land if higher wages that rural migrants earn in urban centres will lead to their higher food consumption. Moreover, migrant remittances could also be used to fund land-use expansion to marginal areas (Taylor et al. 2016; Gray and Bilsborrow 2014). The net effect of these opposite mechanisms varies from place to place (Qin and Liao 2016). There is very little literature evaluating these joint effects of climate change, desertification and labour mobility (Section 7.3.2).

There are also many other institutional, policy and socio-economic drivers of desertification, such as land tenure insecurity, lack of property rights, lack of access to markets, and to rural advisory services, lack of technical knowledge and skills, agricultural price distortions, agricultural support and subsidies contributing to desertification, and lack of economic incentives for SLM (D'Odorico et al. 2013; Geist et al. 2004; Moussa et al. 2016; Mythili and Goedecke 2016; Sow et al.

2016; Tun et al. 2015; García-Ruiz 2010). There is no evidence that these factors will be materially affected by climate change, however, serving as drivers of unsustainable land management practices, they do play a very important role in modulating responses for climate change adaptation and mitigation (Section 3.6.3).

1.1.4.3 Interaction of drivers: Desertification syndrome versus drylands development paradigm

Two broad narratives have historically emerged to describe responses of dryland populations to environmental degradation. The first is 'desertification syndrome' which describes the vicious cycle of resource degradation and poverty, whereby dryland populations apply unsustainable agricultural practices leading to desertification, and exacerbating their poverty, which then subsequently further limits their capacities to invest in SLM (MEA 2005; Safriel and Adeel 2008). The alternative paradigm is one of 'drylands development', which refers to social and technical ingenuity of dryland populations as a driver of dryland sustainability (MEA 2005; Reynolds et al. 2007; Safriel and Adeel 2008). The major difference between these two frameworks is that the 'drylands development' paradigm recognises that human activities are not the sole and/or most important drivers of desertification, but there are interactions of human and climatic drivers within coupled social-ecological systems (Reynolds et al. 2007). This led Behnke and Mortimore (2016), and earlier Swift (1996), to conclude that the concept of desertification as irreversible degradation distorts policy and governance in dryland areas. Mortimore (2016) suggested that instead of externally imposed technical solutions, what is needed is for populations in dryland areas to adapt to this variable environment which they cannot control. All in all, there is *high confidence* that anthropogenic and climatic drivers interact in complex ways in causing desertification. As discussed in Section 3.2.2, the relative influence of human or climatic drivers on desertification varies from place to place (*high confidence*) (Bestelmeyer et al. 2018; D'Odorico et al. 2013; Geist and Lambin 2004; Kok et al. 2016; Polley et al. 2013; Ravi et al. 2010; Scholes 2009; Sietz et al. 2017; Sietz et al. 2011).

1.2 Observations of desertification

1.2.1 Status and trends of desertification

Current estimates of the extent and severity of desertification vary greatly due to missing and/or unreliable information (Gibbs and Salmon 2015). The multiplicity and complexity of the processes of desertification make its quantification difficult (Prince 2016; Cherlet et al. 2018). The most common definition for the drylands is based on defined thresholds of the AI (Figure 3.1; UNEP 1992). While past studies have used the AI to examine changes in desertification or extent of the drylands (Feng and Fu 2013; Zarch et al. 2015; Ji et al. 2015; Spinoni et al. 2015; Huang et al. 2016; Ramarao et al. 2018), this approach has several key limitations: (i) the AI does not measure desertification, (ii) the impact of changes in climate on the land surface and systems is more complex than assumed by AI, and (iii) the relationship between climate change and changes in vegetation is complex due to the influence of CO_2. Expansion of the drylands does not imply desertification by itself, if there is no long-term loss of at least one of the following: biological productivity, ecological integrity, or value to humans.

The use of the AI to define changing aridity levels and dryland extent in an environment with changing atmospheric CO_2 has been strongly challenged (Roderick et al. 2015; Milly and Dunne 2016; Greve et al. 2017; Liu et al. 2017). The suggestion that most of the world has become more arid, since the AI has decreased, is not supported by changes observed in precipitation, evaporation or drought (*medium confidence*) (Sheffield et al. 2012; Greve et al. 2014). A key issue is the assumption in the calculation of potential evapotranspiration that stomatal conductance remains constant, which is invalid if atmospheric CO_2 changes. Given that atmospheric CO_2 has been increasing over the last century or more, and is projected to continue increasing, this means that AI with constant thresholds (or any other measure that relies on potential evapotranspiration) is not an appropriate way to estimate aridity or dryland extent (Donohue et al. 2013; Roderick et al. 2015; Greve et al. 2017). This issue helps explain the apparent contradiction between the drylands becoming more arid according to the AI and also becoming greener according to satellite observations (Fensholt et al. 2012; Andela et al. 2013) (Figure 3.5). Other climate type classifications based on various combinations of temperature and precipitation (Köppen-Trewartha, Köppen-Geiger) have also been used to examine historical changes in climate zones, finding a tendency toward drier climate types (Feng et al. 2014; Spinoni et al. 2015).

The need to establish a baseline when assessing change in the land area degraded has been extensively discussed in Prince et al. (2018). Desertification is a process, not a state of the system, hence an 'absolute' baseline is not required; however, every study uses a baseline defined by the start of their period of interest.

Depending on the definitions applied and methodologies used in evaluation, the status and extent of desertification globally and regionally still show substantial variations (*high confidence*) (D'Odorico et al. 2013). There is *high confidence* that the range and intensity of desertification has increased in some dryland areas over the past several decades (Sections 3.2.1.1 and 3.2.1.2). The three methodological approaches applied for assessing the extent of desertification: expert judgement, satellite observation of net primary productivity, and use of biophysical models, together provide a relatively holistic assessment but none on its own captures the whole picture (Gibbs and Salmon 2015; Vogt et al. 2011; Prince 2016) (Section 4.2.4).

1.2.1.1 Global scale

Complex human–environment interactions, coupled with biophysical, social, economic and political factors unique to any given location, render desertification difficult to map at a global scale (Cherlet et al. 2018). Early attempts to assess desertification focused on expert knowledge in order to obtain global coverage in a cost-effective manner. **Expert judgement** continues to play an important role because degradation remains a subjective feature whose indicators are different from place to place (Sonneveld and Dent 2007). GLASOD

(Global Assessment of Human-induced Soil Degradation) estimated nearly 2000 million hectares (Mha) (15.3% of the total land area) had been degraded by the early 1990s since the mid-20th century. GLASOD was criticised for perceived subjectiveness and exaggeration (Helldén and Tottrup 2008; Sonneveld and Dent 2007). Dregne and Chou (1992) found 3000 Mha in drylands (i.e. about 50% of drylands) were undergoing degradation. Significant improvements have been made through the efforts of WOCAT (World Overview of Conservation Approaches and Technologies), LADA (Land Degradation Assessment in Drylands) and DESIRE (Desertification Mitigation and Remediation of Land) who jointly developed a mapping tool for participatory expert assessment, with which land experts can estimate current area coverage, type and trends of land degradation (Reed et al. 2011).

A number of studies have used **satellite-based remote sensing** to investigate long-term changes in the vegetation and thus identify parts of the drylands undergoing desertification. Satellite data provides information at the resolution of the sensor, which can be relatively coarse (up to 25 km), and interpretations of the data at sub-pixel levels are challenging. The most widely used remotely sensed vegetation index is the NDVI, providing a measure of canopy greenness that is related to the quantity of standing biomass (Bai et al. 2008; de Jong et al. 2011; Fensholt et al. 2012; Andela et al. 2013; Fensholt et al. 2015; Le et al. 2016) (Figure 3.5). A main challenge associated with NDVI is that although biomass and productivity are closely related in some systems, they can differ widely when looking across land uses and ecosystem types, giving a false positive in some instances (Pattison et al. 2015; Aynekulu et al. 2017). For example, bush encroachment in rangelands and intensive monocropping with high fertiliser application gives an indication of increased productivity in satellite data though these could be considered as land degradation. According to this measure there are regions undergoing desertification, however the drylands are greening on average (Figure 3.6).

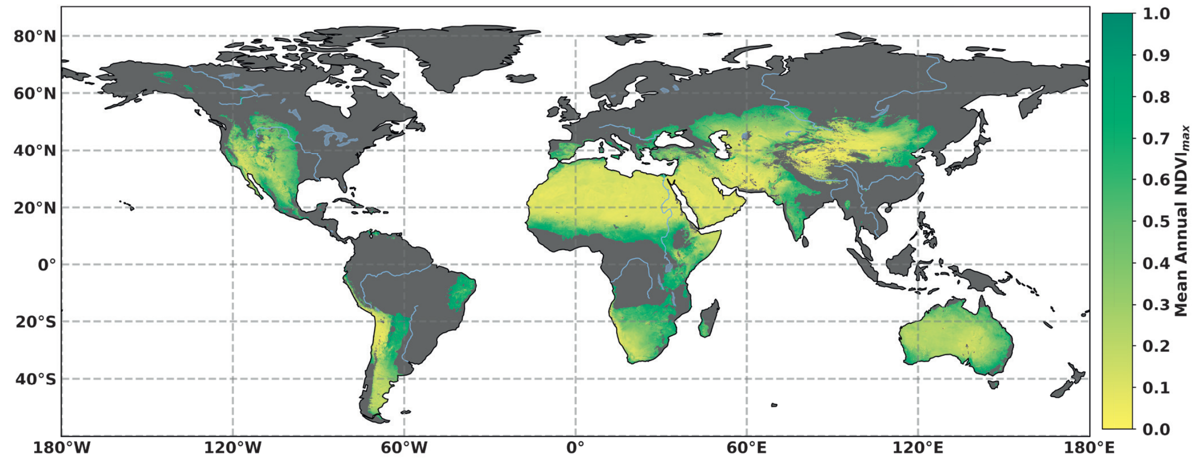

Figure 3.5 | Mean annual maximum NDVI 1982–2015 (Global Inventory Modelling and Mapping Studies NDVI3g v1). Non-dryland regions (aridity index >0.65) are masked in grey.

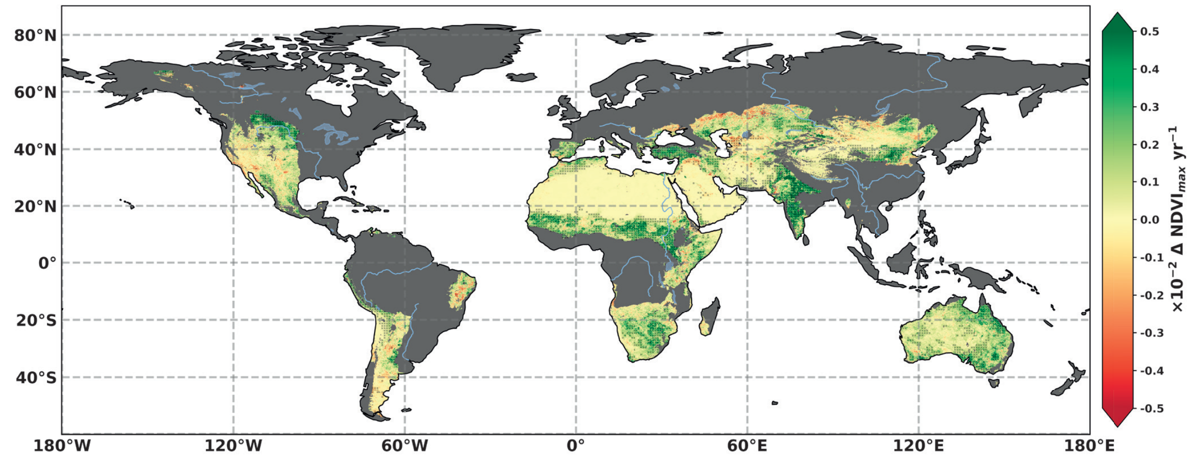

Figure 3.6 | Trend in the annual maximum NDVI 1982–2015 (Global Inventory Modelling and Mapping Studies NDVI3g v1) calculated using the Theil–Sen estimator which is a median based estimator, and is robust to outliers. Non-dryland regions (aridity index >0.65) are masked in grey.

A simple linear trend in NDVI is an unsuitable measure for dryland degradation for several reasons (Wessels et al. 2012; de Jong et al. 2013; Higginbottom and Symeonakis 2014; Le et al. 2016). NDVI is strongly coupled to precipitation in drylands where precipitation has high inter-annual variability. This means that NDVI trend can be dominated by any precipitation trend and is sensitive to wet or dry periods, particularly if they fall near the beginning or end of the time series. Degradation may only occur during part of the time series, while NDVI is stable or even improving during the rest of the time series. This reduces the strength and representativeness of a linear trend. Other factors such as CO_2 fertilisation also influence the NDVI trend. Various techniques have been proposed to address these issues, including the residual trends (RESTREND) method to account for rainfall variability (Evans and Geerken 2004), time-series break point identification methods to find major shifts in the vegetation trends (de Jong et al. 2013; Verbesselt et al. 2010a), and methods to explicitly account for the effect of CO_2 fertilisation (Le et al. 2016).

Using the RESTREND method, Andela et al. (2013) found that human activity contributed to a mixture of improving and degrading regions in drylands. In some locations these regions differed substantially from those identified using the NDVI trend alone, including an increase in the area being desertified in southern Africa and northern Australia, and a decrease in southeast and western Australia and Mongolia. De Jong et al. (2013) examined the NDVI time series for major shifts in vegetation activity and found that 74% of drylands experienced such a shift between 1981 and 2011. This suggests that monotonic linear trends are unsuitable for accurately capturing the changes that have occurred in the majority of the drylands. Le et al. (2016) explicitly accounted for CO_2 fertilisation effect and found that the extent of degraded areas in the world is 3% larger when compared to the linear NDVI trend.

Besides NDVI, there are many vegetation indices derived from satellite data in the optical and infrared wavelengths. Each of these datasets has been derived to overcome some limitation in existing indices. Studies have compared vegetation indices globally (Zhang et al. 2017) and specifically over drylands (Wu 2014). In general, the data from these vegetation indices are available only since around 2000, while NDVI data is available since 1982. With less than 20 years of data, the trend analysis remains problematic with vegetation indices other than NDVI. However, given the various advantages in terms of resolution and other characteristics, these newer vegetation indices will become more useful in the future as more data accumulates.

A major shortcoming of these studies based on vegetation datasets derived from satellite sensors is that they do not account for changes in vegetation composition, thus leading to inaccuracies in the estimation of the extent of degraded areas in drylands. For example, drylands of eastern Africa currently face growing encroachment of invasive plant species, such as *Prosopis juliflora* (Ayanu et al. 2015), which constitutes land degradation since it leads to losses in economic productivity of affected areas but appears as a greening in the satellite data. Another case study in central Senegal found degradation manifested through a reduction in species richness despite satellite observed greening (Herrmann and Tappan 2013). A number of efforts to identify changes in vegetation composition from satellites have been made (Brandt et al. 2016a, b; Evans and Geerken 2006; Geerken 2009; Geerken et al. 2005; Verbesselt et al. 2010a, b). These depend on well-identified reference NDVI time series for particular vegetation groupings, can only differentiate vegetation types that have distinct spectral phenology signatures, and require extensive ground observations for validation. A recent alternative approach to differentiating woody from herbaceous vegetation involves the combined use of optical/infrared-based vegetation indices, indicating greenness, with microwave based Vegetation Optical Depth (VOD) which is sensitive to both woody and leafy vegetation components (Andela et al. 2013; Tian et al. 2017).

Vegetation Optical Depth (VOD) has been available since the 1980s. VOD is based on microwave measurements and is related to total above-ground biomass water content. Unlike NDVI, which is only sensitive to green canopy cover, VOD is also sensitive to water in woody parts of the vegetation and hence provides a view of vegetation changes that can be complementary to NDVI. Liu et al. (2013) used VOD trends to investigate biomass changes and found that VOD was closely related to precipitation changes in drylands. To complement their work with NDVI, Andela et al. (2013) also applied the RESTREND method to VOD. By interpreting NDVI and VOD trends together they were able to differentiate changes to the herbaceous and woody components of the biomass. They reported that many dryland regions are experiencing an increase in the woody fraction often associated with shrub encroachment and suggest that this was aided by CO_2 fertilisation.

Biophysical models use global datasets that describe climate patterns and soil groups, combined with observations of land use, to define classes of potential productivity and map general land degradation (Gibbs and Salmon 2015). All biophysical models have their own set of assumptions and limitations that contribute to their overall uncertainty, including: model structure; spatial scale; data requirements (with associated errors); spatial heterogeneities of socio-economic conditions; and agricultural technologies used. Models have been used to estimate the vegetation productivity potential of land (Cai et al. 2011) and to understand the causes of observed vegetation changes. Zhu et al. (2016) used an ensemble of ecosystem models to investigate causes of vegetation changes from 1982–2009, using a factorial simulation approach. They found CO_2 fertilisation to be the dominant effect globally, though climate and land-cover change were the dominant effects in various dryland locations. Borrelli et al. (2017) modelled that about 6.1% of the global land area experienced very high soil erosion rates (exceeding 10 Mg $ha^{-1} yr^{-1}$) in 2012, particularly in South America, Africa, and Asia.

Overall, improved estimation and mapping of areas undergoing desertification are needed. This requires a combination of rapidly expanding sources of remotely sensed data, ground observations and new modelling approaches. This is a critical gap, especially in the context of measuring progress towards achieving the land degradation-neutrality target by 2030 in the framework of SDGs.

1.2.1.2 Regional scale

While global-scale studies provide information for any region, there are numerous studies that focus on sub-continental scales, providing more in-depth analysis and understanding. Regional and local studies are important to detect location-specific trends in desertification and heterogeneous influences of climate change on desertification. However, these regional and local studies use a wide variety of methodologies, making direct comparisons difficult. For details of the methodologies applied by each study refer to the individual papers.

Africa

It is estimated that 46 of the 54 countries in Africa are vulnerable to desertification, with some already affected (Prăvălie 2016). Moderate or higher severity degradation over recent decades has been identified in many river basins including the Nile (42% of area), Niger (50%), Senegal (51%), Volta (67%), Limpopo (66%) and Lake Chad (26%) (Thiombiano and Tourino-Soto 2007).

The Horn of Africa is getting drier (Damberg and AghaKouchak 2014; Marshall et al. 2012) exacerbating the desertification already occurring (Oroda 2001). The observed decline in vegetation cover is diminishing ecosystem services (Pricope et al. 2013). Based on NDVI residuals, Kenya experienced persistent negative (positive) trends over 21.6% (8.9%) of the country, for the period 1992–2015 (Gichenje and Godinho 2018). Fragmentation of habitats, reduction in the range of livestock grazing, and higher stocking rates are considered to be the main drivers for vegetation structure loss in the rangelands of Kenya (Kihiu 2016; Otuoma et al. 2009).

Despite desertification in the Sahel being a major concern since the 1970s, wetting and greening conditions have been observed in this region over the last three decades (Anyamba and Tucker 2005; Huber et al. 2011; Brandt et al. 2015; Rishmawi et al. 2016; Tian et al. 2016; Leroux et al. 2017; Herrmann et al. 2005; Damberg and AghaKouchak 2014). Cropland areas in the Sahel region of West Africa have doubled since 1975, with settlement area increasing by about 150% (Traore et al. 2014). Thomas and Nigam (2018) found that the Sahara expanded by 10% over the 20th century based on annual rainfall. In Burkina Faso, Dimobe et al. (2015) estimated that from 1984 to 2013, bare soils and agricultural lands increased by 18.8% and 89.7%, respectively, while woodland, gallery forest, tree savannahs, shrub savannahs and water bodies decreased by 18.8%, 19.4%, 4.8%, 45.2% and 31.2%, respectively. In Fakara region in Niger, a 5% annual reduction in herbaceous yield between 1994 and 2006 was largely explained by changes in land use, grazing pressure and soil fertility (Hiernaux et al. 2009). Aladejana et al. (2018) found that between 1986 and 2015, 18.6% of the forest cover around the Owena River basin was lost. For the period 1982–2003, Le et al. (2012) found that 8% of the Volta River basin's landmass had been degraded, with this increasing to 65% after accounting for the effects of CO_2 (and NOx) fertilisation.

Greening has also been observed in parts of southern Africa but it is relatively weak compared to other regions of the continent (Helldén and Tottrup 2008; Fensholt et al. 2012). However, greening can be accompanied by desertification when factors such as decreasing species richness, changes in species composition and shrub encroachment are observed (Smith et al. 2013; Herrmann and Tappan 2013; Kaptué et al. 2015; Herrmann and Sop 2016; Saha et al. 2015) (Sections 3.1.4 and 3.7.3). In the Okavango river Basin in southern Africa, conversion of land towards higher utilisation intensities, unsustainable agricultural practises and overexploitation of the savanna ecosystems have been observed in recent decades (Weinzierl et al. 2016).

In the arid Algerian High Plateaus, desertification due to both climatic and human causes led to the loss of indigenous plant biodiversity between 1975 and 2006 (Hirche et al. 2011). Ayoub (1998) identified 64 Mha in Sudan as degraded, with the Central North Kordofan state being most affected. However, reforestation measures in the last decade sustained by improved rainfall conditions have led to low-medium regrowth conditions in about 20% of the area (Dawelbait and Morari 2012). In Morocco, areas affected by desertification were predominantly on plains with high population and livestock pressure (del Barrio et al. 2016; Kouba et al. 2018; Lahlaoi et al. 2017). The annual costs of soil degradation were estimated at about 1% of Gross Domestic Product (GDP) in Algeria and Egypt, and about 0.5% in Morocco and Tunisia (Réquier-Desjardins and Bied-Charreton 2006).

Asia

Prăvălie (2016) found that desertification is currently affecting 38 of 48 countries in Asia. The changes in drylands in Asia over the period 1982–2011 were mixed, with some areas experiencing vegetation improvement while others showed reduced vegetation (Miao et al. 2015a). Major river basins undergoing salinisation include: Indo-Gangetic Basin in India (Lal and Stewart 2012), Indus Basin in Pakistan (Aslam and Prathapar 2006), Yellow River Basin in China (Chengrui and Dregne 2001), Yinchuan Plain in China (Zhou et al. 2013), Aral Sea Basin of Central Asia (Cai et al. 2003; Pankova 2016; Qadir et al. 2009).

Helldén and Tottrup (2008) highlighted a greening trend in East Asia between 1982 and 2003. Over the past several decades, air temperature and the rainfall increased in the arid and hyper-arid region of Northwest China (Chen et al. 2015; Wang et al. 2017). Within China, rainfall erosivity has shown a positive trend in dryland areas between 1961 and 2012 (Yang and Lu 2015). While water erosion area in Xinjiang, China, has decreased by 23.2%, erosion considered as severe or intense was still increasing (Zhang et al. 2015). Xue et al. (2017) used remote sensing data covering 1975 to 2015 to show that wind-driven desertified land in northern Shanxi in China had expanded until 2000, before contracting again. Li et al. (2012) used satellite data to identify desertification in Inner Mongolia, China and found a link between policy changes and the locations and extent of human-caused desertification. Several oasis regions in China have seen increases in cropland area, while forests, grasslands and available water resources have decreased (Fu et al. 2017; Muyibul et al. 2018; Xie et al. 2014). Between 1990 and 2011 15.3% of Hogno Khaan nature reserve in central Mongolia was subjected to desertification (Lamchin et al. 2016). Using satellite data Liu et al. (2013) found the area of Mongolia undergoing non-climatic desertification was associated with increases in goat density and wildfire occurrence.

In Central Asia, drying up of the Aral Sea is continuing to have negative impacts on regional microclimate and human health (Issanova and Abuduwaili 2017; Lioubimtseva 2015; Micklin 2016; Xi and Sokolik 2015). Half of the region's irrigated lands, especially in the Amudarya and Syrdarya river basins, were affected by secondary salinisation (Qadir et al. 2009). Le et al. (2016) showed that about 57% of croplands in Kazakhstan and about 20% of croplands in Kyrgyzstan had reductions in their vegetation productivity between 1982 and 2006. Chen et al. (2019) indicated that about 58% of the grasslands in the region had reductions in their vegetation productivity between 1999 and 2015. Anthropogenic factors were the main driver of this loss in Turkmenistan and Uzbekistan, while the role of human drivers was smaller than that of climate-related factors in Tajikistan and Kyrgyzstan (Chen et al. 2019). The total costs of land degradation in Central Asia were estimated to equal about 6 billion USD annually (Mirzabaev et al. 2016).

Damberg and AghaKouchak (2014) found that parts of South Asia experienced drying over the last three decades. More than 75% of the area of northern, western and southern Afghanistan is affected by overgrazing and deforestation (UNEP-GEF 2008). Desertification is a serious problem in Pakistan with a wide range of human and natural causes (Irshad et al. 2007; Lal 2018). Similarly, desertification affects parts of India (Kundu et al. 2017; Dharumarajan et al. 2018; Christian et al. 2018). Using satellite data to map various desertification processes, Ajai et al. (2009) found that 81.4 Mha were subject to various processes of desertification in India in 2005, while salinisation affected 6.73 Mha in the country (Singh 2009).

Saudi Arabia is highly vulnerable to desertification (Ministry of Energy Industry and Mineral Resources 2016), with this vulnerability expected to increase in the north-western parts of the country in the coming decades. Yahiya (2012) found that Jazan, south-western Saudi Arabia, lost about 46% of its vegetation cover from 1987 to 2002. Droughts and frequent dust storms were shown to impose adverse impacts over Saudi Arabia, especially under global warming and future climate change (Hasanean et al. 2015). In north-west Jordan, 18% of the area was prone to severe to very severe desertification (Al-Bakri et al. 2016). Large parts of the Syrian drylands have been identified as undergoing desertification (Evans and Geerken 2004; Geerken and Ilaiwi 2004). Moridnejad et al. (2015) identified newly desertified regions in the Middle East based on dust sources, finding that these regions accounted for 39% of all detected dust source points. Desertification has increased substantially in Iran since the 1930s. Despite numerous efforts to rehabilitate degraded areas, it still poses a major threat to agricultural livelihoods in the country (Amiraslani and Dragovich 2011).

Australia

Damberg and AghaKouchak (2014) found that wetter conditions were experienced in northern Australia over the last three decades with widespread greening observed between 1981 and 2006 over much of Australia, except for eastern Australia where large areas were affected by droughts from 2002 to 2009 based on Advanced High Resolution Radiometer (AVHRR) satellite data (Donohue et al. 2009). For the period 1982–2013, Burrell et al. (2017) also found widespread greening over Australia including eastern Australia over the post-drought period. This dramatic change in the trend found for eastern Australia emphasises the dominant role played by precipitation in the drylands. Degradation due to anthropogenic activities and other causes affects over 5% of Australia, particularly near the central west coast. Jackson and Prince (2016) used a local NPP scaling approach applied with MODIS derived vegetation data to quantify degradation in a dryland watershed in Northern Australia from 2000 to 2013. They estimated that 20% of the watershed was degraded. Salinisation has also been found to be degrading parts of the Murray-Darling Basin in Australia (Rengasamy 2006). Eldridge and Soliveres (2014) examined areas undergoing woody encroachment in eastern Australia and found that rather than degrading the landscape, the shrubs often enhanced ecosystem services.

Europe

Drylands cover 33.8% of northern Mediterranean countries: approximately 69% of Spain, 66% of Cyprus, and between 16% and 62% in Greece, Portugal, Italy and France (Zdruli 2011). The European Environment Agency (EEA) indicated that 14 Mha, that is 8% of the territory of the European Union (mostly in Bulgaria, Cyprus, Greece, Italy, Romania, Spain and Portugal), had a 'very high' and 'high sensitivity' to desertification (European Court of Auditors 2018). This figure increases to 40 Mha (23% of the EU territory) if 'moderately' sensitive areas are included (Prăvălie et al. 2017; European Court of Auditors 2018). Desertification in the region is driven by irrigation developments and encroachment of cultivation on rangelands (Safriel 2009) caused by population growth, agricultural policies, and markets. According to a recent assessment report (ECA 2018), Europe is increasingly affected by desertification leading to significant consequences on land use, particularly in Portugal, Spain, Italy, Greece, Malta, Cyprus, Bulgaria and Romania. Using the Universal Soil Loss Equation, it was estimated that soil erosion can be as high as 300 t ha^{-1} yr^{-1} (equivalent to a net loss of 18 mm yr^{-1}) in Spain (López-Bermúdez 1990). For the badlands region in south-east Spain, however, it was shown that biological soil crusts effectively prevent soil erosion (Lázaro et al. 2008). In Mediterranean Europe, Guerra et al. (2016) found a reduction of erosion due to greater effectiveness of soil erosion prevention between 2001 and 2013. Helldén and Tottrup (2008) observed a greening trend in the Mediterranean between 1982 and 2003, while Fensholt et al. (2012) also show a dominance of greening in Eastern Europe.

In Russia, at the beginning of the 2000s, about 7% of the total area (that is, approximately 130 Mha) was threatened by desertification (Gunin and Pankova 2004; Kust et al. 2011). Turkey is considered highly vulnerable to drought, land degradation and desertification (Türkeş 1999, 2003). About 60% of Turkey's land area is characterised with hydro-climatological conditions favourable for desertification (Türkeş 2013). ÇEMGM (2017) estimated that about half of Turkey's land area (48.6%) is prone to moderate-to-high desertification.

North America

Drylands cover approximately 60% of Mexico. According to Pontifes et al. (2018), 3.5% of the area was converted from natural vegetation to agriculture and human settlements between 2002 and 2011. The region is highly vulnerable to desertification due to frequent droughts and floods (Méndez and Magaña 2010; Stahle et al. 2009; Becerril-Pina Rocio et al. 2015).

For the period 2000–2011 the overall difference between potential and actual NPP in different land capability classes in the south-western United States was 11.8% (Noojipady et al. 2015); reductions in grassland-savannah and livestock grazing area and forests were the highest. Bush encroachment is observed over a fairly wide area of grasslands in the western United States, including Jornada Basin within the Chihuahuan Desert, and is spreading at a fast rate despite grazing restrictions intended to curb the spread (Yanoff and Muldavin 2008; Browning and Archer 2011; Van Auken 2009; Rachal et al. 2012). In comparing sand dune migration patterns and rates between 1995 and 2014, Potter and Weigand (2016) established that the area covered by stable dune surfaces, and sand removal zones, decreased, while sand accumulation zones increased from 15.4 to 25.5 km^2 for Palen Dunes in the Southern California desert, while movement of Kelso Dunes is less clear (Lam et al. 2011). Within the United States, average soil erosion rates on all croplands decreased by about 38% between 1982 and 2003 due to better soil management practices (Kertis 2003).

Central and South America

Morales et al. (2011) indicated that desertification costs between 8% and 14% of gross agricultural product in many Central and South American countries. Parts of the dry Chaco and Caldenal regions in Argentina have undergone widespread degradation over the last century (Verón et al. 2017; Fernández et al. 2009). Bisigato and Laphitz (2009) identified overgrazing as a cause of desertification in the Patagonian Monte region of Argentina. Vieira et al. (2015) found that 94% of northeast Brazilian drylands were susceptible to desertification. It is estimated that up to 50% of the area was being degraded due to frequent prolonged droughts and clearing of forests for agriculture. This land-use change threatens the extinction of around 28 native species (Leal et al. 2005). In Central Chile, dryland forest and shrubland area was reduced by 1.7% and 0.7%, respectively, between 1975 and 2008 (Schulz et al. 2010).

1.2.2 Attribution of desertification

Desertification is a result of complex interactions within coupled social-ecological systems. Thus, the relative contributions of climatic, anthropogenic and other drivers of desertification vary depending on specific socio-economic and ecological contexts. The high natural climate variability in dryland regions is a major cause of vegetation changes but does not necessarily imply degradation. Drought is not degradation as the land productivity may return entirely once the drought ends (Kassas 1995). However, if droughts increase in frequency, intensity and/or duration they may overwhelm the vegetation's ability to recover (ecosystem resilience, Prince et al. 2018), causing degradation. Assuming a stationary climate and no human influence, rainfall variability results in fluctuations in vegetation dynamics which can be considered temporary, as the ecosystem tends to recover with rainfall, and desertification does not occur (Ellis 1995; Vetter 2005; von Wehrden et al. 2012). Climate change on the other hand, exemplified by a non-stationary climate, can gradually cause a persistent change in the ecosystem through aridification and CO_2 changes. Assuming no human influence, this 'natural' climatic version of desertification may take place rapidly, especially when thresholds are reached (Prince et al. 2018), or over longer periods of time as the ecosystems slowly adjust to a new climatic norm through progressive changes in the plant community composition. Accounting for this climatic variability is required before attributions to other causes of desertification can be made.

For attributing vegetation changes to climate versus other causes, rain use efficiency (RUE – the change in net primary productivity (NPP) per unit of precipitation) and its variations in time have been used (Prince et al. 1998). Global applications of RUE trends to attribute degradation to climate or other (largely human) causes have been performed by Bai et al. (2008) and Le et al. (2016) (Section 3.2.1.1). The RESTREND (residual trend) method analyses the correlation between annual maximum NDVI (or other vegetation index as a proxy for NPP) and precipitation by testing accumulation and lag periods for the precipitation (Evans and Geerken 2004). The identified relationship with the highest correlation represents the maximum amount of vegetation variability that can be explained by the precipitation, and corresponding RUE values can be calculated. Using this relationship, the climate component of the NDVI time series can be reconstructed, and the difference between this and the original time series (the residual) is attributed to anthropogenic and other causes.

The RESTREND method, or minor variations of it, have been applied extensively. Herrmann and Hutchinson (2005) concluded that climate was the dominant causative factor for widespread greening in the Sahel region from 1982–2003, and anthropogenic and other factors were mostly producing land improvements or no change. However, pockets of desertification were identified in Nigeria and Sudan. Similar results were also found from 1982–2007 by Huber et al. (2011). Wessels et al. (2007) applied RESTREND to South Africa and showed that RESTREND produced a more accurate identification of degraded land than RUE alone. RESTREND identified a smaller area undergoing desertification due to non-climate causes compared to the NDVI trends. Liu et al. (2013) extended the climate component of RESTREND to include temperature and applied this to VOD observations of the cold drylands of Mongolia. They found the area undergoing desertification due to non-climatic causes is much smaller than the area with negative VOD trends. RESTREND has also been applied in several other studies to the Sahel (Leroux et al. 2017), Somalia (Omuto et al. 2010), West Africa (Ibrahim et al. 2015), China (Li et al. 2012; Yin et al. 2014), Central Asia (Jiang et al. 2017), Australia (Burrell et al. 2017) and globally (Andela et al. 2013). In each of these studies the extent to which desertification can be attributed to climate versus other causes varies across the landscape.

3

These studies represent the best regional, remote-sensing based attribution studies to date, noting that RESTREND and RUE have some limitations (Higginbottom and Symeonakis 2014). Vegetation growth (NPP) changes slowly compared to rainfall variations and may be sensitive to rainfall over extended periods (years), depending on vegetation type. Detection of lags and the use of weighted antecedent rainfall can partially address this problem, though most studies do not do this. The method addresses changes since the start of the time series; it cannot identify whether an area is already degraded at the start time. It is assumed that climate, particularly rainfall, is a principal factor in vegetation change which may not be true in more humid regions.

Another assumption in RESTREND is that any trend is linear throughout the period examined. That is, there are no discontinuities (break points) in the trend. Browning et al. (2017) have shown that break points in NDVI time series reflect vegetation changes based on long-term field sites. To overcome this limitation, Burrell et al. (2017) introduced the Time Series Segmentation-RESTREND (TSS-RESTREND) which allows a breakpoint or turning point within the period examined (Figure 3.7). Using TSS-RESTREND over Australia they identified more than double the degrading area than could be identified with a standard RESTREND analysis. The occurrence and drivers of abrupt change (turning points) in ecosystem functioning were also examined by Horion et al. (2016) over the semi-arid Northern Eurasian agricultural frontier. They combined trend shifts in RUE, field data and expert knowledge, to map environmental hotspots of change and attribute them to climate and human activities. One-third of the area showed significant change in RUE, mainly occurring around the fall of the Soviet Union (1991) or as the result of major droughts. Recent human-induced turning points in ecosystem functioning were uncovered around Volgograd (Russia) and around Lake Balkhash (Kazakhstan), attributed to recultivation, increased salinisation, and increased grazing.

Attribution of vegetation changes to human activity has also been done within modelling frameworks. In these methods ecosystem models are used to simulate potential natural vegetation dynamics, and this is compared to the observed state. The difference is attributed to human activities. Applied to the Sahel region during the period of 1982–2002, it showed that people had a minor influence on vegetation changes (Seaquist et al. 2009). Similar model/observation comparisons performed globally found that CO_2 fertilisation was the strongest forcing at global scales, with climate having regionally varying effects (Mao et al. 2013; Zhu et al. 2016). Land-use/land-cover change was a dominant forcing in localised areas. The use of this method to examine vegetation changes in China (1982–2009) attributed most of the greening trend to CO_2 fertilisation and nitrogen (N) deposition (Piao et al. 2015). However in some parts of northern and western China, which includes large areas of drylands, Piao et al. (2015) found climate changes could be the dominant forcing. In the northern extratropical land surface, the observed greening was consistent with increases in greenhouse gases (notably CO_2) and the related climate change, and not consistent with a natural climate that does not include anthropogenic increase in greenhouse gases (Mao et al. 2016). While many studies found widespread influence of CO_2 fertilisation, it is not ubiquitous; for example, Lévesque et al. (2014) found little response to CO_2 fertilisation in some tree species in Switzerland/northern Italy.

Using multiple extreme-event attribution methodologies, Uhe et al. (2018) shows that the dominant influence for droughts in eastern Africa during the October–December 'short rains' season is the prevailing tropical SST patterns, although temperature trends mean that the current drought conditions are hotter than they would have been without climate change. Similarly, Funk et al. (2019) found that 2017 March–June East African drought was influenced by Western Pacific SST, with high SST conditions attributed to climate change.

There are numerous local case studies on attribution of desertification, which use different periods, focus on different land uses and covers, and consider different desertification processes. For example, two-thirds of the observed expansion of the Sahara Desert from 1920–2003 has been attributed to natural climate cycles (the cold phase of Atlantic Multi-Decadal Oscillation and Pacific Decadal Oscillation) (Thomas and Nigam 2018). Some studies consider drought to be the main driver of desertification in Africa (e.g., Masih et al. 2014). However, other studies suggest that although droughts may contribute to desertification, the underlying causes are human activities (Kouba et al. 2018). Brandt et al. (2016a) found that woody vegetation trends are negatively correlated with human population density. Changes in land use, water pumping and flow diversion have enhanced drying of wetlands and salinisation of freshwater aquifers in Israel (Inbar 2007). The dryland territory of China has been found to be very sensitive to both climatic variations and land-use/land-cover changes (Fu et al. 2000; Liu and Tian 2010; Zhao et al. 2013, 2006). Feng et al. (2015) shows that socio-economic factors were dominant in causing desertification in north Shanxi, China, between 1983 and 2012, accounting for about 80% of desertification expansion. Successful grass establishment has been impeded by overgrazing and nutrient depletion leading to the encroachment of shrubs into the northern Chihuahuan Desert (USA) since the mid-19th century (Kidron and Gutschick 2017). Human activities led to rangeland degradation in Pakistan and Mongolia during 2000–2011 (Lei et al. 2011). More equal shares of climatic (temperature and precipitation trends) and human factors were attributed for changes in rangeland condition in China (Yang et al. 2016).

This kaleidoscope of local case studies demonstrates how attribution of desertification is still challenging for several reasons. Firstly, desertification is caused by an interaction of different drivers which vary in space and time. Secondly, in drylands, vegetation reacts closely to changes in rainfall so the effect of rainfall changes on biomass needs to be 'removed' before attributing desertification to human activities. Thirdly, human activities and climatic drivers impact vegetation/ecosystem changes at different rates. Finally, desertification manifests as a gradual change in ecosystem composition and structure (e.g., woody shrub invasion into grasslands). Although initiated at a limited location, ecosystem change may propagate throughout an extensive area via a series of feedback mechanisms. This complicates the attribution of desertification to human and climatic causes, as the process can develop independently once started.

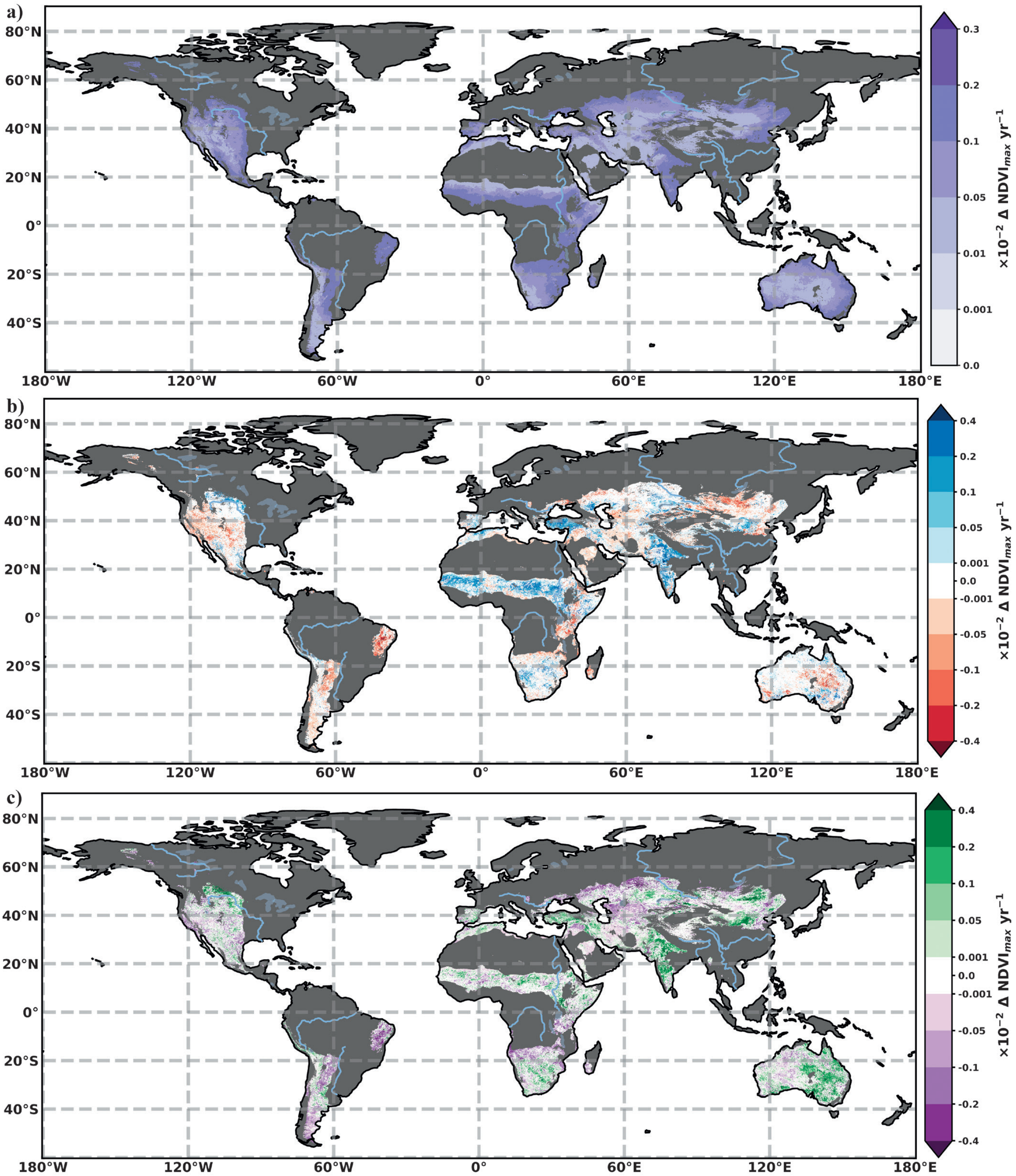

Figure 3.7 | The drivers of dryland vegetation change. The mean annual change in NDVImax between 1982 and 2015 (see Figure 3.6 for total change using Global Inventory Modelling and Mapping Studies NDVI3g v1 dataset) attributable to **(a)** CO_2 fertilisation **(b)** climate and **(c)** land use. The change attributable to CO_2 fertilisation was calculated using the CO_2 fertilisation relationship described in Franks et al. 2013. The Time Series Segmented Residual Trends (TSS-RESTREND) method (Burrell et al. 2017) applied to the CO_2-adjusted NDVI was used to separate Climate and Land Use. A multi-climate dataset ensemble was used to reduce the impact of dataset errors (Burrell et al. 2018). Non-dryland regions (aridity index >0.65) are masked in dark grey. Areas where the change did not meet the multi-run ensemble significance criteria, or are smaller than the error in the sensors (±0.00001) are masked in white.

Rasmussen et al. (2016) studied the reasons behind the overall lack of scientific agreement in trends of environmental changes in the Sahel, including their causes. The study indicated that these are due to differences in conceptualisations and choice of indicators, biases in study site selection, differences in methods, varying measurement accuracy, differences in time and spatial scales. High-resolution, multi-sensor airborne platforms provide a way to address some of these issues (Asner et al. 2012).

The major conclusion of this section is that, with all the shortcomings of individual case studies, relative roles of climatic and human drivers of desertification are location-specific and evolve over time (*high confidence*). Biophysical research on attribution and socio-economic research on drivers of land degradation have long studied the same topic, but in parallel, with little interdisciplinary integration. Interdisciplinary work to identify typical patterns, or typologies, of such interactions of biophysical and human drivers of desertification (not only of dryland vulnerability), and their relative shares, done globally in comparable ways, will help in the formulation of better informed policies to address desertification and achieve land degradation neutrality.

1.3 Desertification feedbacks to climate

While climate change can drive desertification (Section 3.1.4.1), the process of desertification can also alter the local climate, providing a feedback (Sivakumar 2007). These feedbacks can alter the carbon cycle, and hence the level of atmospheric CO_2 and its related global climate change, or they can alter the surface energy and water budgets, directly impacting the local climate. While these feedbacks occur in all climate zones (Chapter 2), here we focus on their effects in dryland regions and assess the literature concerning the major desertification feedbacks to climate. The main feedback pathways discussed throughout Section 3.3 are summarised in Figure 3.8.

Drylands are characterised by limited soil moisture compared to humid regions. Thus, the sensible heat (heat that causes the atmospheric temperature to rise) accounts for more of the surface net radiation than latent heat (evaporation) in these regions (Wang and Dickinson 2013). This tight coupling between the surface energy balance and the soil moisture in semi-arid and dry sub-humid zones makes these regions susceptible to land–atmosphere feedback loops that can amplify changes to the water cycle (Seneviratne et al. 2010). Changes to the land surface caused by desertification can change the surface energy budget, altering the soil moisture and triggering these feedbacks.

1.3.1 Sand and dust aerosols

Sand and mineral dust are frequently mobilised from sparsely vegetated drylands forming 'sand storms' or 'dust storms' (UNEP et al. 2016). The African continent is the most important source of desert dust; perhaps 50% of atmospheric dust comes from the Sahara (Middleton 2017). Ginoux et al. (2012) estimated that 25% of global dust emissions have anthropogenic origins, often in drylands. These events can play an important role in the local energy balance. Through reducing vegetation cover and drying the surface conditions, desertification can increase the frequency of these events. Biological or structural soil crusts have been shown to effectively stabilise dryland soils. Thus their loss due to intense land use and/or climate change can be expected to cause an increase in sand and dust storms (*high confidence*) (Rajot et al. 2003; Field et al. 2010; Rodriguez-Caballero et al. 2018). These sand and dust aerosols impact the regional climate in several ways (Choobari et al. 2014). The direct effect is the interception, reflection and absorption of solar radiation in the atmosphere, reducing the energy available at the land surface and increasing the temperature of the atmosphere in layers with sand and dust present (Kaufman et al. 2002; Middleton 2017; Kok et al. 2018). The heating of the dust layer can alter the relative humidity and atmospheric stability, which can change cloud lifetimes and water content. This has been referred to as the semi-direct effect (Huang et al. 2017). Aerosols also have an indirect effect on climate through their role as cloud condensation nuclei, changing cloud radiative properties as well as the evolution and development of precipitation (Kaufman et al. 2002). While these indirect effects are more variable than the direct effects, depending on the types and amounts of aerosols present, the general tendency is toward an increase in the number, but a reduction in the size of cloud droplets, increasing the cloud reflectivity and decreasing the chances of precipitation. These effects are referred to as aerosol-radiation and aerosol–cloud interactions (Boucher et al. 2013).

There is *high confidence* that there is a negative relationship between vegetation green-up and the occurrence of dust storms (Engelstaedter et al. 2003; Fan et al. 2015; Yu et al. 2015; Zou and Zhai 2004). Changes in groundwater can affect vegetation and the generation of atmospheric dust (Elmore et al. 2008). This can occur through groundwater processes such as the vertical movement of salt to the surface causing salinisation, supply of near-surface soil moisture, and sustenance of groundwater dependent vegetation. Groundwater dependent ecosystems have been identified in many dryland regions around the world (Decker et al. 2013; Lamontagne et al. 2005; Patten et al. 2008). In these locations declining groundwater levels can decrease vegetation cover. Cook et al. (2009) found that dust aerosols intensified the 'Dust Bowl' drought in North America during the 1930s.

By decreasing the amount of green cover and hence increasing the occurrence of sand and dust storms, desertification will increase the amount of shortwave cooling associated with the direct effect (*high confidence*). There is *medium confidence* that the semi-direct and indirect effects of this dust would tend to decrease precipitation and hence provide a positive feedback to desertification (Huang et al. 2009; Konare et al. 2008; Rosenfeld et al. 2001; Solmon et al. 2012; Zhao et al. 2015). However, the combined effect of dust has also been found to increase precipitation in some areas (Islam and Almazroui 2012; Lau et al. 2009; Sun et al. 2012). The overall combined effect of dust aerosols on desertification remains uncertain with *low agreement* between studies that find positive (Huang et al. 2014), negative (Miller et al. 2004) or no feedback on desertification (Zhao et al. 2015).

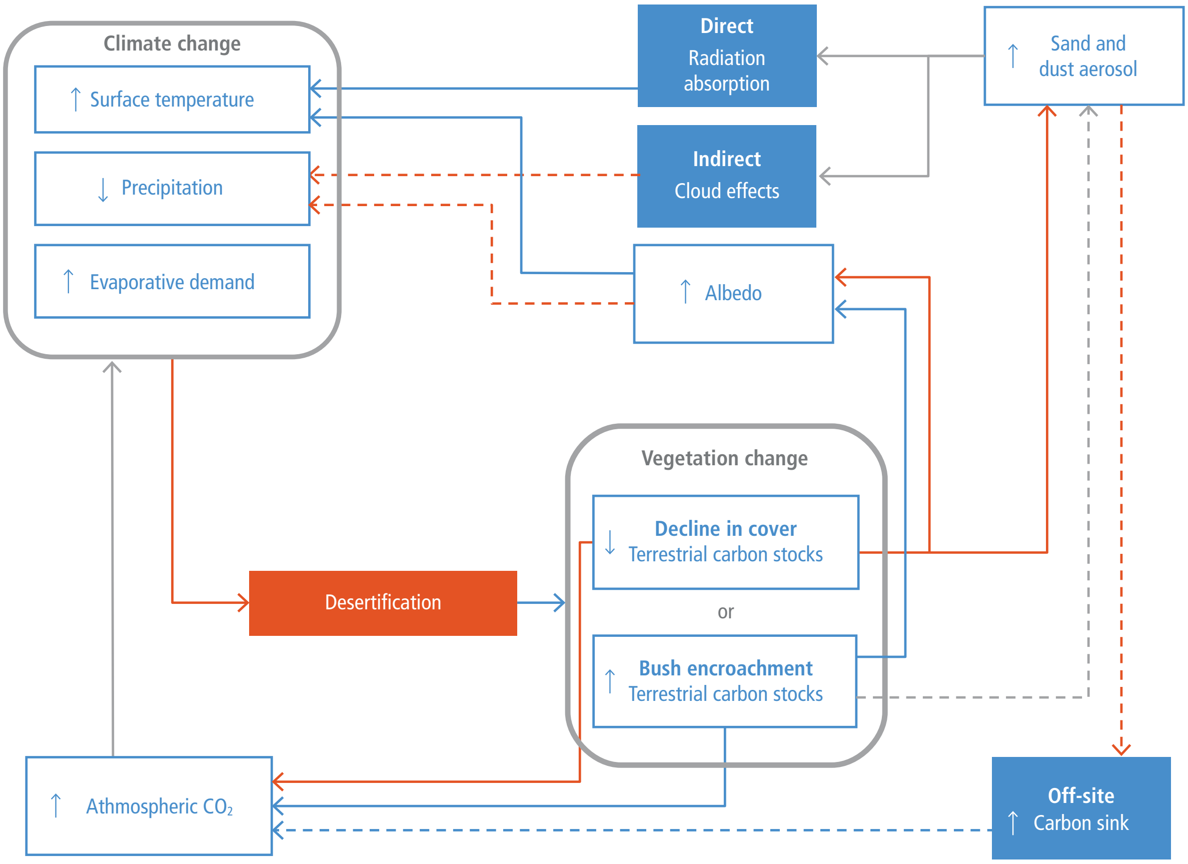

Figure 3.8 | Schematic of main pathways through which desertification can feed back on climate, as discussed in Section 3.4. Note: Red arrows indicate a positive effect. Blue arrows indicate a negative effect. Grey arrows indicate an indeterminate effect (potentially both positive and negative). Solid arrows are direct while dashed arrows are indirect.

1.3.1.1 Off-site feedbacks

Aerosols can act as a vehicle for the long-range transport of nutrients to oceans (Jickells et al. 2005; Okin et al. 2011) and terrestrial land surfaces (Das et al. 2013). In several locations, notably the Atlantic Ocean, the west of northern Africa, and the Pacific Ocean east of northern China, a considerable amount of mineral dust aerosols, sourced from nearby drylands, reaches the oceans. It was estimated that 60% of dust transported off Africa is deposited in the Atlantic Ocean (Kaufman et al. 2005), while 50% of the dust generated in Asia reaches the Pacific Ocean or further (Uno et al. 2009; Zhang et al. 1997). The Sahara is also a major source of dust for the Mediterranean basin (Varga et al. 2014). The direct effect of atmospheric dust over the ocean was found to be a cooling of the ocean surface (*limited evidence*, *high agreement*) (Evan and Mukhopadhyay 2010; Evan et al. 2009) with the tropical North Atlantic mixed layer cooling by over 1°C (Evan et al. 2009).

It has been suggested that dust may act as a source of nutrients for the upper ocean biota, enhancing the biological activity and related carbon sink (*medium evidence*, *low agreement*) (Lenes et al. 2001; Shaw et al. 2008; Neuer et al. 2004). The overall response depends on the environmental controls on the ocean biota, the type of aerosols including their chemical constituents, and the chemical environment in which they dissolve (Boyd et al. 2010).

Dust deposited on snow can increase the amount of absorbed solar radiation leading to more rapid melting (Painter et al. 2018), impacting a region's hydrological cycle (*high confidence*). Dust deposition on snow and ice has been found in many regions of the globe (e.g., Painter et al. 2018; Kaspari et al. 2014; Qian et al. 2015; Painter et al. 2013), however quantification of the effect globally and estimation of future changes in the extent of this effect remain knowledge gaps.

1.3.2 Changes in surface albedo

Increasing surface albedo in dryland regions will impact the local climate, decreasing surface temperature and precipitation, and provide a positive feedback on the albedo (*high confidence*) (Charney et al. 1975). This albedo feedback can occur in desert regions worldwide (Zeng and Yoon 2009). Similar albedo feedbacks have also been found in regional studies over the Middle East (Zaitchik et al. 2007), Australia (Evans et al. 2017; Meng et al. 2014a, b), South America (Lee and Berbery 2012) and the USA (Zaitchik et al. 2013).

Recent work has also found albedo in dryland regions can be associated with soil surface communities of lichens, mosses and cyanobacteria (Rodriguez-Caballero et al. 2018). These communities compose the soil crust in these ecosystems and due to the sparse vegetation cover, directly influence the albedo. These communities are sensitive to climate changes, with field experiments indicating albedo changes greater than 30% are possible. Thus, changes in these communities could trigger surface albedo feedback processes (*limited evidence*, *high agreement*) (Rutherford et al. 2017).

A further pertinent feedback relationship exists between changes in land-cover, albedo, carbon stocks and associated GHG emissions, particularly in drylands with low levels of cloud cover. One of the first studies to focus on the subject was Rotenberg and Yakir (2010), who used the concept of 'radiative forcing' to compare the relative climatic effect of a change in albedo with a change in atmospheric GHGs due to the presence of forest within drylands. Based on this analysis, it was estimated that the change in surface albedo due to the degradation of semi-arid areas has decreased radiative forcing in these areas by an amount equivalent to approximately 20% of global anthropogenic GHG emissions between 1970 and 2005.

1.3.3 Changes in vegetation and greenhouse gas fluxes

Terrestrial ecosystems have the ability to alter atmospheric GHGs through a number of processes (Schlesinger et al. 1990). This may be through a change in plant and soil carbon stocks, either sequestering atmospheric CO_2 during growth or releasing carbon during combustion and respiration, or through processes such as enteric fermentation of domestic and wild ruminants that leads to the release of methane and nitrous oxide (Sivakumar 2007). It is estimated that 241–470 GtC is stored in dryland soils (top 1 m) (Lal 2004; Plaza et al. 2018). When evaluating the effect of desertification, the net balance of all the processes and associated GHG fluxes needs to be considered.

Desertification usually leads to a loss in productivity and a decline in above – and below-ground carbon stocks (Abril et al. 2005; Asner et al. 2003). Drivers such as overgrazing lead to a decrease in both plant and SOC pools (Abdalla et al. 2018). While dryland ecosystems are often characterised by open vegetation, not all drylands have low biomass and carbon stocks in an intact state (Lechmere-Oertel et al. 2005; Maestre et al. 2012). Vegetation types such as the subtropical thicket of South Africa have over 70 tC ha^{-1} in an intact state, greater than 60% of which is released into the atmosphere during degradation through overgrazing (Lechmere-Oertel et al. 2005; Powell 2009). In comparison, semi-arid grasslands and savannahs with similar rainfall, may have only 5–35 tC ha^{-1} (Scholes and Walker 1993; Woomer et al. 2004).

At the same time, it is expected that a decline in plant productivity may lead to a decrease in fuel loads and a reduction in CO_2, nitrous oxide and methane emissions from fire. In a similar manner, decreasing productivity may lead to a reduction in numbers of ruminant animals that in turn would decrease methane emissions. Few studies have focussed on changes in these sources of emissions due to desertification and it remains a field that requires further research.

In comparison to desertification through the suppression of primary production, the process of woody plant encroachment can result in significantly different climatic feedbacks. Increasing woody plant cover in open rangeland ecosystems leads to an increase in woody carbon stocks both above – and below-ground (Asner et al. 2003; Hughes et al. 2006; Petrie et al. 2015; Li et al. 2016). Within the drylands of Texas, USA, shrub encroachment led to a 32% increase in aboveground carbon stocks over a period of 69 years (3.8 tC ha^{-1} to 5.0 tC ha^{-1}) (Asner et al. 2003). Encroachment by taller woody species can lead to significantly higher observed biomass and carbon stocks. For example, encroachment by *Dichrostachys cinerea* and several Vachellia species in the sub-humid savannahs of north-west South Africa led to an increase of 31–46 tC ha^{-1} over a 50–65 year period (1936–2001) (Hudak et al. 2003). In terms of potential changes in SOC stocks, the effect may be dependent on annual rainfall and soil type. Woody cover generally leads to an increase in SOC stocks in drylands that have less than 800 mm of annual rainfall, while encroachment can lead to a loss of soil carbon in more humid ecosystems (Barger et al. 2011; Jackson et al. 2002).

The suppression of the grass layer through the process of woody encroachment may lead to a decrease in carbon stocks within this relatively small carbon pool (Magandana 2016). Conversely, increasing woody cover may lead to a decrease and even halt in surface fires and associated GHG emissions. In their analysis of drivers of fire in southern Africa, Archibald et al. (2009) note that there is a potential threshold around 40% canopy cover, above which surface grass fires are rare. Whilst there have been a number of studies on changes in carbon stocks due to desertification in North America, southern Africa and Australia, a global assessment of the net change in carbon stocks – as well as fire and ruminant GHG emissions due to woody plant encroachment – has not been done yet.

1.4 Desertification impacts on natural and socio-economic systems under climate change

1.4.1 Impacts on natural and managed ecosystems

1.4.1.1 Impacts on ecosystems and their services in drylands

The Millenium Ecosystem Assessement (2005) proposed four classes of ecosystem services: provisioning, regulating, supporting and cultural services (Cross-Chapter Box 8 in Chapter 6). These ecosystem services in drylands are vulnerable to the impacts of climate change due to high variability in temperature, precipitation and soil fertility (Enfors and Gordon 2008; Mortimore 2005). There is *high confidence* that desertification processes such as soil erosion, secondary salinisation, and overgrazing have negatively impacted provisioning ecosystem services in drylands, particularly food and fodder production (Majeed and Muhammad 2019; Mirzabaev et al. 2016; Qadir et al. 2009; Van Loo et al. 2017; Tokbergenova et al. 2018) (Section 3.4.2.2). Zika and Erb (2009) reported an estimation of NPP losses between 0.8 and 2.0 GtC yr^{-1} due to desertification, comparing the potential NPP and the NPP calculated for the year 2000. In terms of climatic factors, although climatic changes between 1976 and 2016 were found to be favourable for crop yields overall in Russia (Ivanov et al. 2018), yield decreases of up to 40–60% in dryland areas were caused by severe and extensive droughts (Ivanov et al. 2018). Increase in temperature can have a direct impact on animals in the form of increased physiological stress (Rojas-Downing et al. 2017), increased water requirements for drinking and cooling, a decrease in the production of milk, meat and eggs, increased stress during conception and reproduction (Nardone et al. 2010) or an increase in seasonal diseases and epidemics (Thornton et al. 2009; Nardone et al. 2010). Furthermore, changes in temperature can indirectly impact livestock through reducing the productivity and quality of feed crops and forages (Thornton et al. 2009; Polley et al. 2013). On the other hand, fewer days with extreme cold temperatures during winter in the temperate zones are associated with lower livestock mortality. The future projection of impacts on ecosystems is presented in Section 3.5.2.

Over-extraction is leading to groundwater depletion in many dryland areas (*high confidence*) (Mudd 2000; Mays 2013; Mahmod and Watanabe 2014; Jolly et al. 2008). Globally, groundwater reserves have been reduced since 1900, with the highest rate of estimated reductions of 145 km^3 yr^{-1} between 2000 and 2008 (Konikow 2011). Some arid lands are very vulnerable to groundwater reductions, because the current natural recharge rates are lower than during the previous wetter periods (e.g., the Atacama Desert, and Nubian aquifer system in Africa) (Squeo et al. 2006; Mahmod and Watanabe 2014; Herrera et al. 2018).

Among regulating services, desertification can influence levels of atmospheric CO_2. In drylands, the majority of carbon is stored below ground in the form of biomass and SOC (FAO 1995) (Section 3.3.3). Land-use changes often lead to reductions in SOC and organic matter inputs into soil (Albaladejo et al. 2013; Almagro et al. 2010; Hoffmann et al. 2012; Lavee et al. 1998; Rey et al. 2011), increasing soil salinity and soil erosion (Lavee et al. 1998; Martinez-Mena et al. 2008). In addition to the loss of soil, erosion reduces soil nutrients and organic matter, thereby impacting land's productive capacity. To illustrate, soil erosion by water is estimated to result in the loss of 23–42 Mt of nitrogen and 14.6–26.4 Mt of phosphorus from soils globally each year (Pierzynski et al. 2017).

Precipitation, by affecting soil moisture content, is considered to be the principal determinant of the capacity of drylands to sequester carbon (Fay et al. 2008; Hao et al. 2008; Mi et al. 2015; Serrano-Ortiz et al. 2015; Vargas et al. 2012; Sharkhuu et al. 2016). Lower annual rainfall resulted in the release of carbon into the atmosphere for a number of sites located in Mongolia, China and North America (Biederman et al. 2017; Chen et al. 2009; Fay et al. 2008; Hao et al. 2008; Mi et al. 2015; Sharkhuu et al. 2016). Low soil water availability promotes soil microbial respiration, yet there is insufficient moisture to stimulate plant productivity (Austin et al. 2004), resulting in net carbon emissions at an ecosystem level. Under even drier conditions, photodegradation of vegetation biomass may often constitute an additional loss of carbon from an ecosystem (Rutledge et al. 2010). In contrast, years of good rainfall in drylands resulted in the sequestration of carbon (Biederman et al. 2017; Chen et al. 2009; Hao et al. 2008). In an exceptionally rainy year (2011) in the southern hemisphere, the semi-arid ecosystems of this region contributed 51% of the global net carbon sink (Poulter et al. 2014). These results suggest that arid ecosystems could be an important global carbon sink, depending on soil water availability (*medium evidence*, *high agreement*). However, drylands are generally predicted to become warmer with an increasing frequency of extreme drought and high rainfall events (Donat et al. 2016).

When desertification reduces vegetation cover, this alters the soil surface, affecting the albedo and the water balance (Gonzalez-Martin et al. 2014) (Section 3.3). In such situations, erosive winds have no more obstacles, which favours the occurrence of wind erosion and dust storms. Mineral aerosols have an important influence on the dispersal of soil nutrients and lead to changes in soil characteristics (Goudie and Middleton 2001; Middleton 2017). Thereby, the soil formation as a supporting ecosystem service is negatively affected (Section 3.3.1). Soil erosion by wind results in a loss of fine soil particles (silt and clay), reducing the ability of soil to sequester carbon (Wiesmeier et al. 2015). Moreover, dust storms reduce crop yields by loss of plant tissue caused by sandblasting (resulting in loss of plant leaves and hence reduced photosynthetic activity (Field et al. 2010), exposing crop roots, crop seed burial under sand deposits, and leading to losses of nutrients and fertiliser from topsoil (Stefanski and Sivakumar 2009)). Dust storms also impact crop yields by reducing the quantity of water available for irrigation; they can decrease the storage capacity of reservoirs by siltation, and block conveyance canals (Middleton 2017; Middleton and Kang 2017; Stefanski and Sivakumar 2009). Livestock productivity is reduced by injuries caused by dust storms (Stefanski and Sivakumar 2009). Additionally, dust storms favour the dispersion of microbial and plant species, which can make local endemic species vulnerable to extinction and promote the invasion of plant and microbial species (Asem and Roy 2010; Womack et al. 2010). Dust storms increase microbial species in remote sites (*high confidence*) (Kellogg et al. 2004; Prospero et al. 2005; Griffin et al. 2006; Schlesinger et al. 2006; Griffin 2007; De Deckker et al. 2008; Jeon et al. 2011; Abed et al. 2012; Favet et al. 2013; Woo et al. 2013; Pointing and Belnap 2014).

1.4.1.2 Impacts on biodiversity: Plant and wildlife

Plant biodiversity

Over 20% of global plant biodiversity centres are located within drylands (White and Nackoney 2003). Plant species located within these areas are characterised by high genetic diversity within populations (Martínez-Palacios et al. 1999). The plant species within these ecosystems are often highly threatened by climate change and desertification (Millennium Ecosystem Assessment 2005b; Maestre et al. 2012). Increasing aridity exacerbates the risk of extinction of some plant species, especially those that are already threatened due to small populations or restricted habitats (Gitay et al. 2002). Desertification, including through land-use change, already contributed to the loss of biodiversity across drylands (*medium confidence*) (Newbold et al. 2015; Wilting et al. 2017). For example, species richness decreased from 234 species in 1978 to 95 in 2011 following long periods of drought and human driven degradation on the steppe land of south-western Algeria (Observatoire du Sahara et du Sahel 2013). Similarly, drought and overgrazing led to loss of biodiversity in Pakistan to the point that only drought-adapted species can now survive on the arid rangelands (Akhter and Arshad 2006). Similar trends were observed in desert steppes of Mongolia (Khishigbayar et al. 2015). In contrast, the increase in annual moistening of southern European Russia from the late 1980s to the beginning of the 21st century caused the restoration of steppe vegetation, even under conditions of strong anthropogenic pressure (Ivanov et al. 2018). The seed banks of annual species can often survive over the long term, germinating in wet years, suggesting that these species could be resilient to some aspects of climate change (Vetter et al. 2005). Yet, Hiernaux and Houérou (2006) showed that overgrazing in the Sahel tended to decrease the seed bank of annuals, which could make them vulnerable to climate change over time. Perennial species, considered as the structuring element of the ecosystem, are usually less affected as they have deeper roots, xeromorphic properties and physiological mechanisms that increase drought tolerance (Le Houérou 1996). However, in North Africa, long-term monitoring (1978–2014) has shown that important plant perennial species have also disappeared due to drought (*Stipa tenacissima* and *Artemisia herba alba*) (Hirche et al. 2018; Observatoire du Sahara et du Sahel 2013). The aridisation of the climate in the south of Eastern Siberia led to the advance of the steppes to the north and to the corresponding migration of steppe mammal species between 1976 and 2016 (Ivanov et al. 2018). The future projection of impacts on plant biodiversity is presented in Section 3.5.2.

Wildlife biodiversity

Dryland ecosystems have high levels of faunal diversity and endemism (MEA 2005; Whitford 2002). Over 30% of the endemic bird areas are located within these regions, which is also home to 25% of vertebrate species (Maestre et al. 2012; MEA 2005). Yet, many species within drylands are threatened with extinction (Durant et al. 2014; Walther 2016). Habitat degradation and desertification are generally associated with biodiversity loss (Ceballos et al. 2010; Tang et al. 2018; Newbold et al. 2015). The 'grazing value' of land declines with both a reduction in vegetation cover and shrub encroachment, with the former being more detrimental to native vertebrates (Parsons et al. 2017). Conversely, shrub encroachment may buffer desertification by increasing resource and microclimate availability, resulting in an increase in vertebrate species abundance and richness observed in the shrub-encroached arid grasslands of North America (Whitford 1997) and Australia (Parsons et al. 2017). However, compared to historically resilient drylands, these encroached habitats and their new species assemblages may be more sensitive to droughts, which may become more prevalent with climate change (Schooley et al. 2018). Mammals and birds may be particularly sensitive to droughts because they rely on evaporative cooling to maintain their body temperatures within an optimal range (Hetem et al. 2016) and risk lethal dehydration in water limited environments (Albright et al. 2017). The direct effects of reduced rainfall and water availability are *likely* to be exacerbated by the indirect effects of desertification through a reduction in primary productivity. A reduction in the quality and quantity of resources available to herbivores due to desertification under changing climate can have knock-on consequences for predators and may ultimately disrupt trophic cascades (*limited evidence*, *low agreement)* (Rey et al. 2017; Walther 2010). Reduced resource availability may also compromise immune response to novel pathogens, with increased pathogen dispersal associated with dust storms (Zinabu et al. 2018). Responses to desertification are species-specific and mechanistic models are not yet able to accurately predict individual species' responses to the many factors associated with desertification (Fuller et al. 2016).

1.4.2 Impacts on socio-economic systems

Combined impacts of desertification and climate change on socio-economic development in drylands are complex. Figure 3.9 schematically represents our qualitative assessment of the magnitudes and the uncertainties associated with these impacts on attainment of the SDGs in dryland areas (UN 2015). The impacts of desertification and climate change are difficult to isolate from the effects of other socio-economic, institutional and political factors (Pradhan et al. 2017). However, there is *high confidence* that climate change will exacerbate the vulnerability of dryland populations to desertification, and that the combination of pressures coming from climate change and desertification will diminish opportunities for reducing poverty, enhancing food and nutritional security, empowering women, reducing disease burden, and improving access to water and sanitation. Desertification is embedded in SDG 15 (Target 15.3) and climate change is under SDG 13. The *high confidence* and high magnitude impacts depicted for these SDGs (Figure 3.9) indicate that the interactions between desertification and climate change strongly affect the achievement of the targets of SDGs 13 and 15.3, pointing at the need for the coordination of policy actions on land degradation neutrality and mitigation and adaptation to climate change. The following subsections present the literature and assessments which serve as the basis for Figure 3.9.

1.4.2.1 Impacts on poverty

Climate change has a high potential to contribute to poverty particularly through the risks coming from extreme weather events (Olsson et al. 2014). However, the evidence rigourously attributing changes in observed poverty to climate change impacts is currently not available. On the other hand, most of the research on links between poverty and desertification (or more broadly, land degradation) focused on whether or not poverty is a cause of land degradation (Gerber et al. 2014; Vu et al. 2014; Way 2016) (Section 4.7.1). The literature measuring the extent to which desertification contributed to poverty globally is lacking: the related literature remains qualitative or correlational (Barbier and Hochard 2016). At the local level, on the other hand, there is *limited evidence* and *high agreement* that desertification increased multidimensional poverty. For example, Diao and Sarpong (2011) estimated that land degradation lowered agricultural incomes in Ghana by 4.2 billion USD between 2006 and 2015, increasing the national poverty rate by 5.4% in 2015. Land degradation increased the probability of households becoming poor

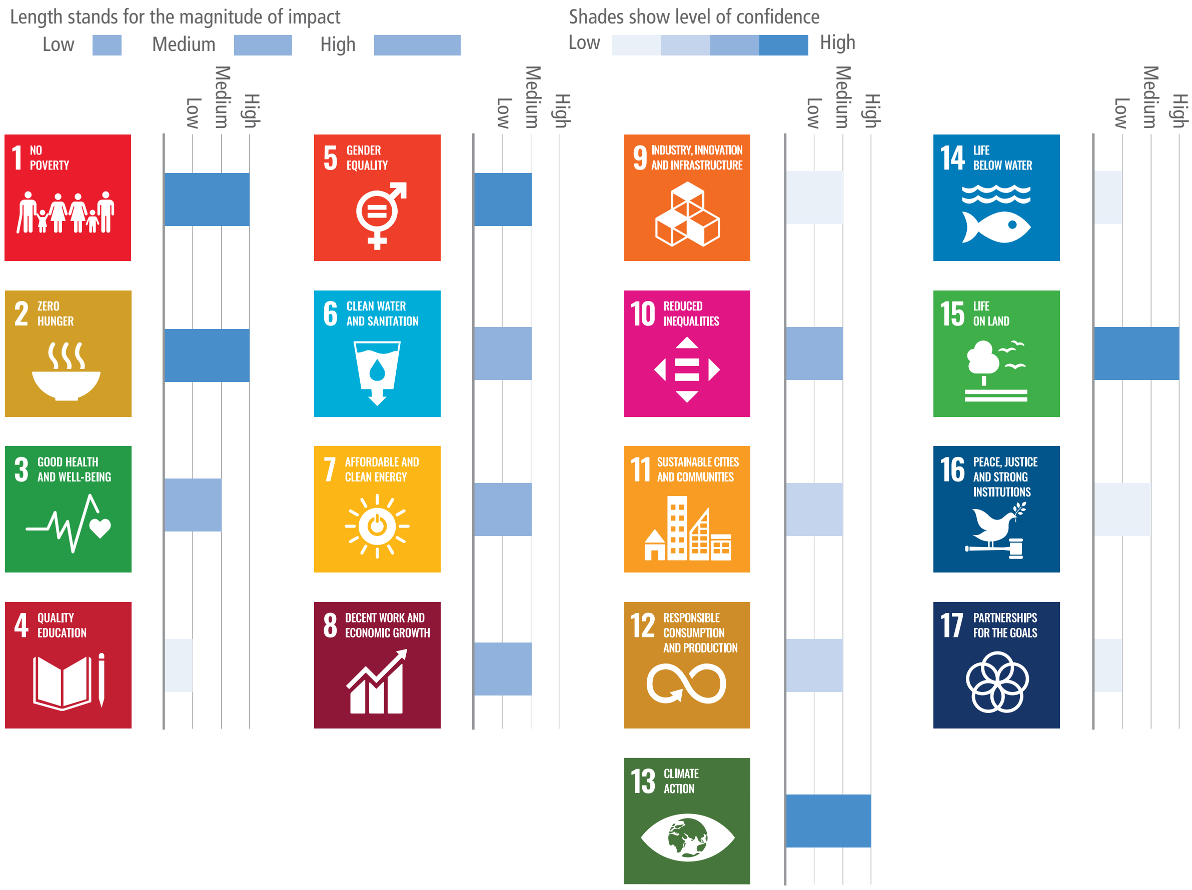

Figure 3.9 | Socio-economic impacts of desertification and climate change with the SDG framework.

by 35% in Malawi and 48% in Tanzania (Kirui 2016). Desertification in China was found to have resulted in substantial losses in income, food production and jobs (Jiang et al. 2014). On the other hand, Ge et al. (2015) indicated that desertification was positively associated with growing incomes in Inner Mongolia in China in the short run since no costs were incurred for SLM, while in the long run higher incomes allowed allocation of more investments to reduce desertification. This relationship corresponds to the Environmental Kuznets Curve, which posits that environmental degradation initially rises and subsequently falls with rising income (e.g., Stern 2017). There is *limited evidence* on the validity of this hypothesis regarding desertification.

1.4.2.2 Impacts on food and nutritional insecurity

About 821 million people globally were food insecure in 2017, of whom 63% in Asia, 31% in Africa and 5% in Latin America and the Caribbean (FAO et al. 2018). The global number of food insecure people rose by 37 million since 2014. Changing climate variability, combined with a lack of climate resilience, was suggested as a key driver of this increase (FAO et al. 2018). Sub-Saharan Africa, East Africa and South Asia had the highest share of undernourished populations in the world in 2017, with 28.8%, 31.4% and 33.7% respectively (FAO et al. 2018). The major mechanism through which climate change and desertification affect food security is through their impacts on agricultural productivity. There is *robust evidence* pointing to negative impacts of climate change on crop yields in dryland areas (*high agreement*) (Hochman et al. 2017; Nelson et al. 2010; Zhao et al. 2017) (Sections 3.4.1, 5.2.2 and 4.7.2). There is also *robust evidence* and *high agreement* on the losses in agricultural productivity and incomes due to desertification (Kirui 2016; Moussa et al. 2016; Mythili and Goedecke 2016; Tun et al. 2015). Nkonya et al. (2016a) estimated that cultivating wheat, maize, and rice with unsustainable land management practices is currently resulting in global losses of 56.6 billion USD annually, with another 8.7 billion USD of annual losses due to lower livestock productivity caused by rangeland degradation. However, the extent to which these losses affected food insecurity in dryland areas is not known. Lower crop yields and higher agricultural prices worsen existing food insecurity, especially for net food-buying rural households and urban dwellers. Climate change and desertification are not the sole drivers of food insecurity, but especially in the areas with high dependence on agriculture, they are among the main contributors.

1.4.2.3 Impacts on human health through dust storms

The frequency and intensity of dust storms are increasing due to land-use and land-cover changes and climate-related factors (Section 2.4) particularly in some regions of the world such as the Arabian Peninsula (Jish Prakash et al. 2015; Yu et al. 2015; Gherboudj et al. 2017; Notaro et al. 2013; Yu et al. 2013; Alobaidi et al. 2017; Maghrabi et al. 2011; Almazroui et al. 2018) and broader Middle East (Rashki et al. 2012; Türkeş 2017; Namdari et al. 2018) as well as Central Asia (Indoitu et al. 2015; Xi and Sokolik 2015), with growing negative impacts on human health (*high confidence*) (Díaz et al. 2017; Goudarzi et al. 2017; Goudie 2014; Samoli et al. 2011). Dust storms transport particulate matter, pollutants, pathogens and potential allergens that are dangerous for human health over long distances (Goudie and Middleton 2006; Sprigg 2016). Particulate matter (PM; that is, the suspended particles in the air of up to 10 micrometres (PM10) or less in size), have damaging effects on human health (Díaz et al. 2017; Goudarzi et al. 2017; Goudie 2014; Samoli et al. 2011). The health effects of dust storms are largest in areas in the immediate vicinity of their origin, primarily the Sahara Desert, followed by Central and eastern Asia, the Middle East and Australia (Zhang et al. 2016), however, there is *robust evidence* showing that the negative health effects of dust storms reach a much wider area (Bennett et al. 2006; Díaz et al. 2017; Kashima et al. 2016; Lee et al. 2014; Samoli et al. 2011; Zhang et al. 2016). The primary health effects of dust storms include damage to the respiratory and cardiovascular systems (Goudie 2013). Dust particles with a diameter smaller than 2.5μm were associated with global cardiopulmonary mortality of about 402,000 people in 2005, with 3.47 million years of life lost in that single year (Giannadaki et al. 2014). Although globally only 1.8% of cardiopulmonary deaths were caused by dust storms, in the countries of the Sahara region, Middle East, South and East Asia, dust storms were suggested to be the cause of 15–50% of all cardiopulmonary deaths (Giannadaki et al. 2014). A 10 μgm^{-3} increase in PM10 dust particles was associated with mean increases in non-accidental mortality from 0.33% to 0.51% across different calendar seasons in China, Japan and South Korea (Kim et al. 2017). The percentage of all-cause deaths attributed to fine particulate matter in Iranian cities affected by Middle Eastern dust storms (MED) was 0.56–5.02%, while the same percentage for non-affected cities was 0.16–4.13% (Hopke et al. 2018). Epidemics of meningococcal meningitis occur in the Sahelian region during the dry seasons with dusty conditions (Agier et al. 2012; Molesworth et al. 2003). Despite a strong concentration of dust storms in the Sahel, North Africa, the Middle East and Central Asia, there is relatively little research on human health impacts of dust storms in these regions. More research on health impacts and related costs of dust storms, as well as on public health response measures, can help in mitigating these health impacts.

1.4.2.4 Impacts on gender equality

Environmental issues such as desertification and impacts of climate change have been increasingly investigated through a gender lens (Bose (n.d.); Broeckhoven and Cliquet 2015; Kaijser and Kronsell 2014; Kiptot et al. 2014; Villamor and van Noordwijk 2016). There is *medium evidence* and *high agreement* that women will be impacted more than men by environmental degradation (Arora-Jonsson 2011; Gurung et al. 2006) (Cross-Chapter Box 11 in Chapter 7). Socially structured gender-specific roles and responsibilities, daily activities, access and control over resources, decision-making and opportunities lead men and women to interact differently with natural resources and landscapes. For example, water scarcity affected women more than men in rural Ghana as they had to spend more time in fetching water, which has implications on time allocations for other activities (Ahmed et al. 2016). Despite the evidence pointing to differentiated impact of environmental degradation on women and men, gender issues have been marginally addressed in many land restoration and rehabilitation efforts, which often remain gender-blind. Although there is *robust evidence* on the location-specific impacts of climate change and desertification on gender equality, there is *limited evidence* on the gender-related impacts of land restoration and rehabilitation activities. Women are usually excluded from local decision-making on actions regarding desertification and climate change. Socially constructed gender-specific roles and responsibilities are not static because they are shaped by other factors such as wealth, age, ethnicity and formal education (Kaijser and Kronsell 2014; Villamor et al. 2014). Hence, women's and men's environmental knowledge and priorities for restoration often differ (Sijapati Basnett et al. 2017). In some areas where sustainable land options (e.g., agroforestry) are being promoted, women were not able to participate due to culturally embedded asymmetries in power relations between men and women (Catacutan and Villamor 2016). Nonetheless women, particularly in the rural areas, remain heavily involved in securing food for their households. Food security for them is associated with land productivity and women's contribution to address desertification is crucial.

1.4.2.5 Impacts on water scarcity and use

Reduced water retention capacity of degraded soils amplifies floods (de la Paix et al. 2011), reinforces degradation processes through soil erosion, and reduces annual intake of water to aquifers, exacerbating existing water scarcities (Le Roux et al. 2017; Cano et al. 2018). Reduced vegetation cover and more intense dust storms were found to intensify droughts (Cook et al. 2009). Moreover, secondary salinisation in the irrigated drylands often requires leaching with considerable amounts of water (Greene et al. 2016; Wichelns and Qadir 2015). Thus, different types of soil degradation increase water scarcity both through lower water quantity and quality (Liu et al. 2017; Liu et al. 2016c). All these processes reduce water availability for other needs. In this context, climate change will further intensify water scarcity in some dryland areas and increase the frequency of droughts (*medium confidence*) (IPCC 2013; Zheng et al. 2018) (Section 2.2). Higher water scarcity may imply growing use of wastewater effluents for irrigation (Pedrero et al. 2010). The use of untreated wastewater exacerbates soil degradation processes (Tal 2016; Singh et al. 2004; Qishlaqi et al. 2008; Hanjra et al. 2012), in addition to negative human health impacts (Faour-Klingbeil and Todd 2018; Hanjra et al. 2012). Climate change will thus amplify the need for integrated land and water management for sustainable development.

1.4.2.6 Impacts on energy infrastructure through dust storms

Desertification leads to conditions that favour the production of dust storms (*high confidence*) (Section 3.3.1). There is *robust evidence* and *high agreement* that dust storms negatively affect the operational potential of solar and wind power harvesting equipment through dust deposition, reduced reach of solar radiation and increasing blade-surface roughness, and can also reduce effective electricity distribution in high-voltage transmission lines (Zidane et al. 2016; Costa et al. 2016; Lopez-Garcia et al. 2016; Maliszewski et al. 2012; Mani and Pillai 2010; Mejia and Kleissl 2013; Mejia et al. 2014; Middleton 2017; Sarver et al. 2013; Kaufman et al. 2002; Kok et al. 2018). Direct exposure to desert dust storm can reduce energy generation efficiency of solar panels by 70–80% in one hour (Ghazi et al. 2014). (Saidan et al. 2016) indicated that in the conditions of Baghdad, Iraq, one month's exposure to weather reduced the efficiency of solar modules by 18.74% due to dust deposition. In the Atacama desert, Chile, one month's exposure reduced thin-film solar module performance by 3.7–4.8% (Fuentealba et al. 2015). This has important implications for climate change mitigation efforts using the expansion of solar and wind energy generation in dryland areas for substituting fossil fuels. Abundant access to solar energy in many dryland areas makes them high-potential locations for the installation of solar energy generating infrastructure. Increasing desertification, resulting in higher frequency and intensity of dust storms imposes additional costs for climate change mitigation through deployment of solar and wind energy harvesting facilities in dryland areas. Most frequently used solutions to this problem involve physically wiping or washing the surface of solar devices with water. These result in additional costs and excessive use of already scarce water resources and labour (Middleton 2017). The use of special coatings on the surface of solar panels can help prevent the deposition of dusts (Costa et al. 2016; Costa et al. 2018; Gholami et al. 2017).

1.4.2.7 Impacts on transport infrastructure through dust storms and sand movement

Dust storms and movement of sand dunes often threaten the safety and operation of railway and road infrastructure in arid and hyper-arid areas, and can lead to road and airport closures due to reductions in visibility. For example, the dust storm on 10 March 2009 over Riyadh was assessed to be the strongest in the previous two decades in Saudi Arabia, causing limited visibility, airport shutdown and damages to infrastructure and environment across the city (Maghrabi et al. 2011). There are numerous historical examples of how moving sand dunes led to the forced decommissioning of early railway lines built in Sudan, Algeria, Namibia and Saudi Arabia in the late 19th and early 20th century (Bruno et al. 2018). Currently, the highest concentrations of railways vulnerable to sand movements are located in north-western China, Middle East and North Africa (Bruno et al. 2018; Cheng and Xue 2014). In China, sand dune movements are periodically disrupting the railway transport on the Linhai–Ceke line in north-western China and on the Lanzhou–Xinjiang High-speed Railway in western China, with considerable clean-up and maintenance costs (Bruno et al. 2018; Zhang et al. 2010). There are large-scale plans for expansion of railway networks in arid areas of China, Central Asia, North Africa, the Middle East, and eastern Africa. For example, The Belt and Road Initiative promoted by China, the Gulf Railway project by the Cooperation Council for the Arab States of the Gulf or Lamu Port–South Sudan–Ethiopia Transport (LAPSSET) Corridor in Eastern Africa. These investments have long-term return and operation periods. Their construction and associated engineering solutions will therefore benefit from careful consideration of potential desertification and climate change effects on sand storms and dune movements.

1.4.2.8 Impacts on conflicts

There is *low confidence* in climate change and desertification leading to violent conflicts. There is *medium evidence* and *low agreement* that climate change and desertification contribute to already existing conflict potentials (Herrero 2006; von Uexkull et al. 2016; Theisen 2017; Olsson 2017; Wischnath and Buhaug 2014) (Section 4.7.3). To illustrate, Hsiang et al. (2013) found that each one standard deviation increase in temperature or rainfall was found to increase interpersonal violence by 4% and intergroup conflict by 14% (Hsiang et al. 2013). However, this conclusion was disputed by Buhaug et al. (2014), who found no evidence linking climate variability to violent conflict after replicating Hsiang et al. (2013) by studying only violent conflicts. Almer et al. (2017) found that a one standard deviation increase in dryness raised the likelihood of riots in Sub-Saharan African countries by 8.3% during the 1990–2011 period. On the other hand, Owain and Maslin (2018) found that droughts and heatwaves were not significantly affecting the level of regional conflict in East Africa. Similarly, it was suggested that droughts and desertification in the Sahel played a relatively minor role in the conflicts in the Sahel in the 1980s, with the major reasons for the conflicts during this period being political, especially the marginalisation of pastoralists (Benjaminsen 2016), corruption and rent-seeking (Benjaminsen et al. 2012). Moreover, the role of environmental factors as the key drivers of conflicts was questioned in the case of Sudan (Verhoeven 2011) and Syria (De Châtel 2014). Selection bias, when the literature focuses on the same few regions where conflicts occurred and relates them to climate change, is a major shortcoming, as it ignores other cases where conflicts did not occur (Adams et al. 2018) despite degradation of the natural resource base and extreme weather events.

1.4.2.9 Impacts on migration

Environmentally induced migration is complex and accounts for multiple drivers of mobility as well as other adaptation measures undertaken by populations exposed to environmental risk (*high confidence*). There is *medium evidence* and *low agreement* that climate change impacts migration. The World Bank (2018) predicted that 143 million people would be forced to move internally by 2050 if no climate action is taken. Focusing on asylum seekers alone, rather than the total number of migrants, Missirian and Schlenker (2017) predict that asylum applications to the European Union will increase from 28% (98,000 additional asylum applications per year) up to 188% (660,000 additional applications per year) depending on the climate scenario by 2100. While the modelling efforts have greatly improved over the years (Hunter et al. 2015; McLeman 2011; Sherbinin and Bai 2018) and in particular, these recent estimates provide an important insight into potential future developments, the quantitative projections are still based on the number of people exposed to risk rather than the number of people who would actually engage in migration as a response to this risk (Gemenne 2011; McLeman 2013) and they do not take into account individual agency in migration decision nor adaptive capacities of individuals (Hartmann 2010; Kniveton et al. 2011; Piguet 2010) (see Section 3.6.2 discussing migration as a response to desertification). Accordingly, the available micro-level evidence suggests that climate-related shocks are one of the many drivers of migration (Adger et al. 2014; London Government Office for Science and Foresight 2011; Melde et al. 2017), but the individual responses to climate risk are more complex than commonly assumed (Gray and Mueller 2012a). For example, despite strong focus on natural disasters, neither flooding (Gray and Mueller 2012b; Mueller et al. 2014) nor earthquakes (Halliday 2006) were found to induce long-term migration; but instead, slow-onset changes, especially those provoking crop failures and heat stress, could affect household or individual migration decisions (Gray and Mueller 2012a; Missirian and Schlenker 2017; Mueller et al. 2014). Out-migration from drought-prone areas has received particular attention (de Sherbinin et al. 2012; Ezra and Kiros 2001). A substantial body of literature suggests that households engage in local or internal migration as a response to drought (Findlay 2011; Gray and Mueller 2012a), while international migration decreases with drought in some contexts (Henry et al. 2004), but might increase in contexts where migration networks are well established (Feng et al. 2010; Nawrotzki and DeWaard 2016; Nawrotzki et al. 2015, 2016). Similarly, the evidence is not conclusive with respect to the effect of environmental drivers, in particular desertification, on mobility. While it has not consistently entailed out-migration in the case of Ecuadorian Andes (Gray 2009, 2010), environmental and land degradation increased mobility in Kenya and Nepal (Gray 2011; Massey et al. 2010), but marginally decreased mobility in Uganda (Gray 2011). These results suggest that in some contexts, environmental shocks actually undermine households' financial capacity to undertake migration (Nawrotzki and Bakhtsiyarava 2017), especially in the case of the poorest households (Barbier and Hochard 2018; Koubi et al. 2016; Kubik and Maurel 2016; McKenzie and Yang 2015). Adding to the complexity, migration, especially to frontier areas, by increasing pressure on land and natural resources, might itself contribute to environmental degradation at the destination (Hugo 2008; IPBES 2018a; McLeman 2017). The consequences of migration can also be salient in the case of migration to urban or peri-urban areas; indeed, environmentally induced migration can add to urbanisation (Section 3.6.2.2), often exacerbating problems related to poor infrastructure and unemployment.

1.4.2.10 Impacts on pastoral communities

Pastoral production systems occupy a significant portion of the world (Rass 2006; Dong 2016). Food insecurity among pastoral households is often high (Gomes 2006) (Section 3.1.3). The Sahelian droughts of the 1970s–1980s provided an example of how droughts could affect livestock resources and crop productivity, contributing to hunger, out-migration and suffering for millions of pastoralists (Hein and De Ridder 2006; Molua and Lambi 2007). During these Sahelian droughts low and erratic rainfall exacerbated desertification processes, leading to ecological changes that forced people to use marginal lands and ecosystems. Similarly, the rate of rangeland degradation is now increasing because of environmental changes and overexploitation of resources (Kassahun et al. 2008; Vetter 2005). Desertification coupled with climate change is negatively affecting livestock feed and grazing species (Hopkins and Del Prado 2007), changing the composition in favour of species with low forage quality, ultimately reducing livestock productivity (D'Odorico et al. 2013; Dibari et al. 2016) and increasing livestock disease prevalence (Thornton et al. 2009). There is *robust evidence* and *high agreement* that weak adaptive capacity, coupled with negative effects from other climate-related factors, are predisposing pastoralists to increased poverty from desertification and climate change globally (López-i-Gelats et al. 2016; Giannini et al. 2008; IPCC 2007). On the other hand, misguided policies such as enforced sedentarisation, and in certain cases protected area delineation (fencing), which restrict livestock mobility have hampered optimal use of grazing land resources (Du 2012). Such policies have led to degradation of resources and out-migration of people in search of better livelihoods (Gebeye 2016; Liao et al. 2015). Restrictions on the mobile lifestyle are reducing the resilient adaptive capacity of pastoralists to natural hazards including extreme and variable weather conditions, drought and climate change (Schilling et al. 2014). Furthermore, the exacerbation of the desertification phenomenon due to agricultural intensification (D'Odorico et al. 2013) and land fragmentation caused by encroachment of agriculture into rangelands (Otuoma et al. 2009; Behnke and Kerven 2013) is threatening pastoral livelihoods. For example, commercial cotton (*Gossypium hirsutum*) production is crowding out pastoral systems in Benin (Tamou et al. 2018). Food shortages and the urgency to produce enough crop for public consumption are leading to the encroachment of agriculture into productive rangelands and those converted rangelands are frequently prime lands used by pastoralists to produce feed and graze their livestock during dry years (Dodd 1994). The sustainability of pastoral systems is therefore coming into question because of social and political marginalisation of those systems (Davies et al. 2016) and also because of the fierce competition they are facing from other livelihood sources such as crop farming (Haan et al. 2016).

1.5 Future projections

1.5.1 Future projections of desertification

Assessing the impact of climate change on future desertification is difficult as several environmental and anthropogenic variables interact to determine its dynamics. The majority of modelling studies regarding the future evolution of desertification rely on the analysis of specific climate change scenarios and Global Climate Models (GCMs) and their effect on a few processes or drivers that trigger desertification (Cross-Chapter Box 1 in Chapter 1).

With regards to climate impacts, the analysis of global and regional climate models concludes that under all representative concentration pathways (RCPs) potential evapotranspiration (PET) would increase worldwide as a consequence of increasing surface temperatures and surface water vapour deficit (Sherwood and Fu 2014). Consequently, there would be associated changes in aridity indices that depend on this variable (*high agreement*, *robust evidence*) (Cook et al. 2014a;

Dai 2011; Dominguez et al. 2010; Feng and Fu 2013; Ficklin et al. 2016; Fu et al. 2016; Greve and Seneviratne 1999; Koutroulis 2019; Scheff and Frierson 2015). Due to the large increase in PET and decrease in precipitation over some subtropical land areas, aridity index will decrease in some drylands (Zhao and Dai 2015), with one model estimating approximately 10% increase in hyper-arid areas globally (Zeng and Yoon 2009). Increases in PET are projected to continue due to climate change (Cook et al. 2014a; Fu et al. 2016; Lin et al. 2015; Scheff and Frierson 2015). However, as noted in Sections 3.1.1 and 3.2.1, these PET calculations use assumptions that are not valid in an environment with changing CO_2. Evidence from precipitation, runoff or photosynthetic uptake of CO_2 suggest that a future warmer world will be less arid (Roderick et al. 2015). Observations in recent decades indicate that the Hadley cell has expanded poleward in both hemispheres (Fu et al. 2006; Hu and Fu 2007; Johanson et al. 2009; Seidel and Randel 2007), and under all RCPs would continue expanding (Johanson et al. 2009; Lu et al. 2007). This expansion leads to the poleward extension of subtropical dry zones and hence an expansion in drylands on the poleward edge (Scheff and Frierson 2012). Overall, this suggests that while aridity will increase in some places (*high confidence*), there is insufficient evidence to suggest a global change in dryland aridity (*medium confidence*).

Regional modelling studies confirm the outcomes of Global Climate Models (Africa: Terink et al. 2013; China: Yin et al. 2015; Brazil: Marengo and Bernasconi 2015; Cook et al. 2012; Greece: Nastos et al. 2013; Italy: Coppola and Giorgi 2009). According to the IPCC AR5 (IPCC 2013), decreases in soil moisture are detected in the Mediterranean, southwest USA and southern African regions. This is in line with alterations in the Hadley circulation and higher surface temperatures. This surface drying will continue to the end of this century under the RCP8.5 scenario (*high confidence*). Ramarao et al. (2015) showed that a future climate projection based on RCP4.5 scenario indicated the possibility for detecting the summer-time soil drying signal over the Indian region during the 21st century in response to climate change. The IPCC Special Report on Global Warming of 1.5°C (SR15) (Chapter 3; Hoegh-Guldberg et al. 2018) concluded with '*medium confidence*' that global warming by more than 1.5°C increases considerably the risk of aridity for the Mediterranean area and southern Africa. Miao et al. (2015b) showed an acceleration of desertification trends under the RCP8.5 scenario in the middle and northern part of Central Asia and some parts of north-western China. It is also useful to consider the effects of the dynamic–thermodynamical feedback of the climate. Schewe and Levermann (2017) show increases of up to 300% in the Central Sahel rainfall by the end of the century due to an expansion of the West African monsoon. Warming could trigger an intensification of monsoonal precipitation due to increases in ocean moisture availability.

The impacts of climate change on dust storm activity are not yet comprehensively studied and represent an important knowledge gap. Currently, GCMs are unable to capture recent observed dust emission and transport (Evan 2018; Evan et al. 2014), limiting confidence in future projections. Literature suggests that climate change decreases wind erosion/dust emission overall, with regional variation (*low confidence*). Mahowald et al. (2006) and Mahowald (2007) found that climate change led to a decrease in desert dust source areas globally using CMIP3 GCMs. Wang et al. (2009) found a decrease in sand dune movement by 2039 (increasing thereafter) when assessing future wind-erosion-driven desertification in arid and semi-arid China using a range of SRES scenarios and HadCM3 simulations. Dust activity in the Southern Great Plains in the USA was projected to increase, while in the Northern Great Plains it was projected to decrease under RCP8.5 climate change scenario (Pu and Ginoux 2017). Evan et al. (2016) project a decrease in African dust emission associated with a slowdown of the tropical circulation in the high CO_2 RCP8.5 scenario.

Global estimates of the impact of climate change on soil salinisation show that under the IS92a emissions scenario (a scenario prepared in 1992 that contains 'business as usual' assumptions) (Leggett et al. 1992) the area at risk of salinisation would increase in the future (*limited evidence*, *high agreement*) (Schofield and Kirkby 2003). Climate change has an influence on soil salinisation that induces further land degradation through several mechanisms that vary in their level of complexity. However, only a few examples can be found to illustrate this range of impacts, including the effect of groundwater table depletion (Rengasamy 2006) and irrigation management (Sivakumar 2007), salt migration in coastal aquifers with decreasing water tables (Sherif and Singh 1999) (Section 4.10.7), and surface hydrology and vegetation that affect wetlands and favour salinisation (Nielsen and Brock 2009).

1.5.1.1 Future vulnerability and risk of desertification

Following the conceptual framework developed in the Special Report on extreme events (SREX) (IPCC 2012), future risks are assessed by examining changes in exposure (that is, presence of people; livelihoods; species or ecosystems; environmental functions, service, and resources; infrastructure; or economic, social or cultural assets; see Glossary), changes in vulnerability (that is, propensity or predisposition to be adversely affected; see Glossary) and changes in the nature and magnitude of hazards (that is, potential occurrence of a natural or human-induced physical event that causes damage; see Glossary). Climate change is expected to further exacerbate the vulnerability of dryland ecosystems to desertification by increasing PET globally (Sherwood and Fu 2014). Temperature increases between 2°C and 4°C are projected in drylands by the end of the 21st century under RCP4.5 and RCP8.5 scenarios, respectively (IPCC 2013). An assessment by Carrão et al. 2017 showed an increase in drought hazards by late-century (2071–2099) compared to a baseline (1971–2000) under high RCPs in drylands around the Mediterranean, south-eastern Africa, and southern Australia. In Latin America, Morales et al. (2011) indicated that areas affected by drought will increase significantly by 2100 under SRES scenarios A2 and B2. The countries expected to be affected include Guatemala, El Salvador, Honduras and Nicaragua. In CMIP5 scenarios, Mediterranean types of climate are projected to become drier (Alessandri et al. 2014; Polade et al. 2017), with the equatorward margins being potentially replaced by arid climate types (Alessandri et al. 2014). Globally, climate change is predicted to intensify the occurrence and severity of droughts (*medium confidence*) (Dai 2013; Sheffield and Wood 2008; Swann et al. 2016; Wang 2005; Zhao and Dai 2015; Carrão et al. 2017; Naumann et al. 2018) (Section 2.2). Ukkola et al. (2018)

showed large discrepancies between CMIP5 models for all types of droughts, limiting the confidence that can be assigned to projections of drought.

Drylands are characterised by high climatic variability. Climate impacts on desertification are not only defined by projected trends in mean temperature and precipitation values but are also strongly dependent on changes in climate variability and extremes (Reyer et al. 2013). The responses of ecosystems depend on diverse vegetation types. Drier ecosystems are more sensitive to changes in precipitation and temperature (Li et al. 2018; Seddon et al. 2016; You et al. 2018), increasing vulnerability to desertification. It has also been reported that areas with high variability in precipitation tend to have lower livestock densities and that those societies that have a strong dependence on livestock that graze natural forage are especially affected (Sloat et al. 2018). Social vulnerability in drylands increases as a consequence of climate change that threatens the viability of pastoral food systems (Dougill et al. 2010; López-i-Gelats et al. 2016). Social drivers can also play an important role with regards to future vulnerability (Máñez Costa et al. 2011). In the arid region of north-western China, Liu et al. (2016b) estimated that under RCP4.5 areas of increased vulnerability to climate change and desertification will surpass those with decreased vulnerability.

Using an ensemble of global climate, integrated assessment and impact models, Byers et al. (2018) investigated 14 impact indicators at different levels of global mean temperature change and socio-economic development. The indicators cover water, energy and land sectors. Of particular relevance to desertification are the water (e.g., water stress, drought intensity) and the land (e.g., habitat degradation) indicators. Under shared socio-economic pathway SSP2 ('Middle of the Road') at 1.5°C, 2°C and 3°C of global warming, the numbers of dryland populations exposed (vulnerable) to various impacts related to water, energy and land sectors (e.g., water stress, drought intensity, habitat degradation) are projected to reach 951 (178) million, 1152 (220) million and 1285 (277) million, respectively. While at global warming of 2°C, under SSP1 ('Sustainability'), the exposed (vulnerable) dryland population is 974 (35) million, and under SSP3 ('Fragmented World') it is 1267 (522) million. Steady increases in the exposed and vulnerable populations are seen for increasing global mean temperatures. However much larger differences are seen in the vulnerable population under different SSPs. Around half the vulnerable population is in South Asia, followed by Central Asia, West Africa and East Asia.

1.5.2 Future projections of impacts

Future climate change is expected to increase the potential for increased soil erosion by water in dryland areas (*medium confidence*). Yang et al. (2003) use a Revised Universal Soil Loss Equation (RUSLE) model to study global soil erosion under historical, present and future conditions of both cropland and climate. Soil erosion potential has increased by about 17%, and climate change will increase this further in the future. In northern Iran, under the SRES A2 emission scenario the mean erosion potential is projected to grow by 45%, comparing the period 1991–2010 with 2031–2050 (Zare et al. 2016). A strong decrease in precipitation for almost all parts of Turkey was projected for the period 2021–2050 compared to 1971–2000 using Regional Climate Model, RegCM4.4 of the International Centre for Theoretical Physics (ICTP) under RCP4.5 and RCP8.5 scenarios (Türkeş et al. 2019). The projected changes in precipitation distribution can lead to more extreme precipitation events and prolonged droughts, increasing Turkey's vulnerability to soil erosion. In Portugal, a study comparing wet and dry catchments under A1B and B1 emission scenarios showed an increase in erosion in dry catchments (Serpa et al. 2015). In Morocco an increase in sediment load is projected as a consequence of reduced precipitation (Simonneaux et al. 2015). WGII AR5 concluded the impact of increases in heavy rainfall and temperature on soil erosion will be modulated by soil management practices, rainfall seasonality and land cover (Jiménez Cisneros et al. 2014). Ravi et al. (2010) predicted an increase in hydrologic and aeolian soil erosion processes as a consequence of droughts in drylands. However, there are some studies that indicate that soil erosion will be reduced in Spain (Zabaleta et al. 2013), Greece (Nerantzaki et al. 2015) and Australia (Klik and Eitzinger 2010), while others project changes in erosion as a consequence of the expansion of croplands (Borrelli et al. 2017).

Potential dryland expansion implies lower carbon sequestration and higher risk of desertification (Huang et al. 2017), with severe impacts on land usability and threats to food security. At the level of biomes (global-scale zones, generally defined by the type of plant life that they support in response to average rainfall and temperature patterns; see Glossary), soil carbon uptake is determined mostly by weather variability. The area of the land in which dryness controls CO_2 exchange has risen by 6% since 1948 and is projected to expand by at least another 8% by 2050. In these regions net carbon uptake is about 27% lower than elsewhere (Yi et al. 2014). Potential losses of soil carbon are projected to range from 9% to 12% of the total carbon stock in the 0–20 cm layer of soils in southern European Russia by end of this century (Ivanov et al. 2018).

Desertification under climate change will threaten biodiversity in drylands (*medium confidence*). Rodriguez-Caballero et al. (2018) analysed the cover of biological soil crusts under current and future environmental conditions utilising an environmental niche modelling approach. Their results suggest that biological soil crusts currently cover approximately 1600 Mha in drylands. Under RCP scenarios 2.6 to 8.5, 25–40% of this cover will be lost by 2070 with climate and land use contributing equally. The predicted loss is expected to substantially reduce the contribution of biological soil crusts to nitrogen cycling (6.7–9.9 TgN yr^{-1}) and carbon cycling (0.16–0.24 PgC yr^{-1}) (Rodriguez-Caballero et al. 2018). A study in Colorado Plateau, USA showed that changes in climate in drylands may damage the biocrust communities by promoting rapid mortality of foundational species (Rutherford et al. 2017), while in the Southern California deserts climate change-driven extreme heat and drought may surpass the survival thresholds of some desert species (Bachelet et al. 2016). In semi-arid Mediterranean shrublands in eastern Spain, plant species richness and plant cover could be reduced by climate change and soil erosion (García-Fayos and Bochet 2009). The main drivers of species extinctions are land-use change, habitat pollution, over-exploitation, and species invasion, while climate change is indirectly linked to species extinctions (Settele et al. 2014). Malcolm et al. (2006) found that more than 2000 plant species located within dryland biodiversity hotspots could become extinct within 100 years, starting 2004 (within the Cape Floristic Region, Mediterranean Basin and southwest Australia). Furthermore, it is suggested that land use and climate change could cause the loss of 17% of species within shrublands and 8% within hot deserts by 2050 (*low confidence*) (van Vuuren et al. 2006). A study in the semi-arid Chinese Altai Mountains showed that mammal species richness will decline, rates of species turnover will increase, and more than 50% of their current ranges will be lost (Ye et al. 2018).

Changing climate and land use have resulted in higher aridity and more droughts in some drylands, with the rising role of precipitation, wind and evaporation on desertification (Fischlin et al. 2007). In a 2°C world, annual water discharge is projected to decline, and heatwaves are projected to pose risk to food production by 2070 (Waha et al. 2017). However, Betts et al. (2018) found a mixed response of water availability (runoff) in dryland catchments to global temperature increases from 1.5°C to 2°C. The forecasts for Sub-Saharan Africa suggest that higher temperatures, increase in the number of heatwaves, and increasing aridity, will affect the rainfed agricultural systems (Serdeczny et al. 2017). A study by Wang et al. (2009) in arid and semi-arid China showed decreased livestock productivity and grain yields from 2040 to 2099, threatening food security. In Central Asia, projections indicate a decrease in crop yields, and negative impacts of prolonged heat waves on population health (Reyer et al. 2017) (Section 3.7.2). World Bank (2009) projected that, without the carbon fertilisation effect, climate change will reduce the mean yields for 11 major global crops – millet, field pea, sugar beet, sweet potato, wheat, rice, maize, soybean, groundnut, sunflower and rapeseed – by 15% in Sub-Saharan Africa, 11% in Middle East and North Africa, 18% in South Asia, and 6% in Latin America and the Caribbean by 2046–2055, compared to 1996–2005. A separate meta-analysis suggested a similar reduction in yields in Africa and South Asia due to climate change by 2050 (Knox et al. 2012). Schlenker and Lobell (2010) estimated that in sub-Saharan Africa, crop production may be reduced by 17–22% due to climate change by 2050. At the local level, climate change impacts on crop yields vary by location (Section 5.2.2). Negative impacts of climate change on agricultural productivity contribute to higher food prices. The imbalance between supply and demand for agricultural products is projected to increase agricultural prices in the range of 31% for rice, to 100% for maize by 2050 (Nelson et al. 2010), and cereal prices in the range between a 32% increase and a 16% decrease by 2030 (Hertel et al. 2010). In southern European Russia, it is projected that the yields of grain crops will decline by 5–10% by 2050 due to the higher intensity and coverage of droughts (Ivanov et al. 2018).

Climate change can have strong impacts on poverty in drylands (*medium confidence*) (Hallegatte and Rozenberg 2017; Hertel and Lobell 2014). Globally, Hallegatte et al. (2015) project that without rapid and inclusive progress on eradicating multidimensional poverty, climate change could increase the number of the people living in poverty by between 35 million and 122 million people by 2030. Although these numbers are global and not specific to drylands, the highest impacts in terms of the share of the national populations being affected are projected to be in the drylands areas of the Sahel region, eastern Africa and South Asia (Stephane Hallegatte et al. 2015). The impacts of climate change on poverty vary depending on whether the household is a net agricultural buyer or seller. Modelling results showed that poverty rates would increase by about one-third among the urban households and non-agricultural self-employed in Malawi, Uganda, Zambia and Bangladesh due to high agricultural prices and low agricultural productivity under climate change (Hertel et al. 2010). On the contrary, modelled poverty rates fell substantially among agricultural households in Chile, Indonesia, the Philippines and Thailand, because higher prices compensated for productivity losses (Hertel et al. 2010).

1.6 Responses to desertification under climate change

Achieving sustainable development of dryland livelihoods requires avoiding dryland degradation through SLM and restoring and rehabilitating the degraded drylands due to their potential wealth of ecosystem benefits and importance to human livelihoods and economies (Thomas 2008). A broad suite of on-the-ground response measures exists to address desertification (Scholes 2009), be it in the form of improved fire and grazing management, the control of erosion; integrated crop, soil and water management, among others (Liniger and Critchley 2007; Scholes 2009). These actions are part of the broader context of dryland development and long-term SLM within coupled socio-economic systems (Reynolds et al. 2007; Stringer et al. 2017; Webb et al. 2017). Many of these response options correspond to those grouped under 'land transitions' in the IPCC Special Report on Global Warming of 1.5°C (Coninck et al. 2018) (Table 6.4). It is therefore recognised that such actions require financial, institutional and policy support for their wide-scale adoption and sustainability over time (Sections 3.6.3, 4.8.5 and 6.4.4).

1.6.1 SLM technologies and practices: On-the-ground actions

A broad range of activities and measures can help avoid, reduce and reverse degradation across the dryland areas of the world. Many of these actions also contribute to climate change adaptation and mitigation, with further sustainable development co-benefits for poverty eradication and food security (*high confidence*) (Section 6.3). As preventing desertification is strongly preferable and more cost-effective than allowing land to degrade and then attempting to restore it (IPBES 2018b; Webb et al. 2013), there is a growing emphasis on avoiding and reducing land degradation, following the Land Degradation Neutrality framework (Cowie et al. 2018; Orr et al. 2017) (Section 4.8.5).

An assessment is made of six activities and measures practicable across the biomes and anthromes of the dryland domain (Figure 3.10). This suite of actions is not exhaustive, but rather a set of activities that are particularly pertinent to global dryland ecosystems. They are not necessarily exclusive to drylands and are often implemented across a range of biomes and anthromes (Figure 3.10;

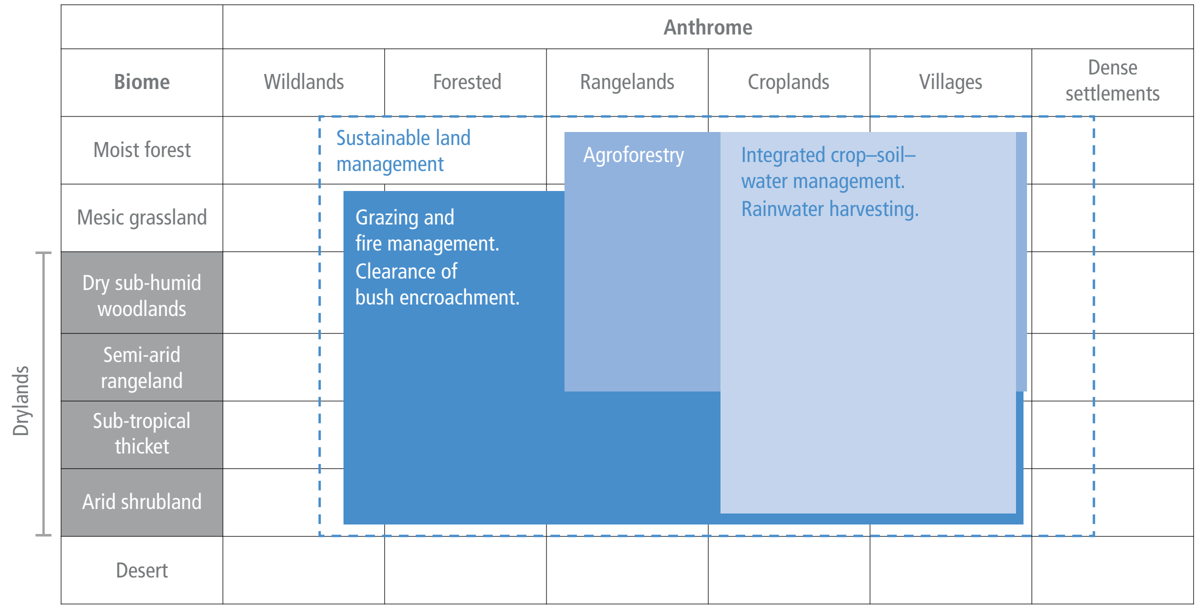

Figure 3.10 | The typical distribution of on-the-ground actions across global biomes and anthromes.

for afforestation, see Section 3.7.2, Cross-Chapter Box 2 in Chapter 1, and Chapter 4 (Section 4.8.3)). The use of anthromes as a structuring element for response options is based on the essential role of interactions between social and ecological systems in driving desertification within coupled socio-ecological systems (Cherlet et al. 2018). The concept of the anthromes is defined in the Glossary and explored further in Chapters 1, 4 and 6.

The assessment of each action is twofold: firstly, to assess the ability of each action to address desertification and enhance climate change resilience, and secondly, to assess the potential impact of future climate change on the effectiveness of each action.

1.6.1.1 Integrated crop–soil–water management

Forms of integrated cropland management have been practiced in drylands for thousands of years (Knörzer et al. 2009). Actions include planting a diversity of species including drought-resilient ecologically appropriate plants, reducing tillage, applying organic compost and fertiliser, adopting different forms of irrigation and maintaining vegetation and mulch cover. In the contemporary era, several of these actions have been adopted in response to climate change.

In terms of climate change *adaptation*, the resilience of agriculture to the impacts of climate change is strongly influenced by the underlying health and stability of soils as well as improvements in crop varieties, irrigation efficiency and supplemental irrigation, for example, through rainwater harvesting (*medium evidence, high agreement*) (Altieri et al. 2015; Amundson et al. 2015; Derpsch et al. 2010; Lal 1997; de Vries et al. 2012). Desertification often leads to a reduction in ground cover that in turn results in accelerated water and wind erosion and an associated loss of fertile topsoil that can greatly reduce the resilience of agriculture to climate change (*medium evidence, high agreement*) (Touré et al. 2019; Amundson et al. 2015; Borrelli et al. 2017; Pierre et al. 2017). Amadou et al. (2011) note that even a minimal cover of crop residues (100 kg ha^{-1}) can substantially decrease wind erosion.

Compared to conventional (flood or furrow) irrigation, drip irrigation methods are more efficient in supplying water to the plant root zone, resulting in lower water requirements and enhanced water use efficiency (*robust evidence*, *high agreement*) (Ibragimov et al. 2007; Narayanamoorthy 2010; Niaz et al. 2009). For example, in the rainfed area of Fetehjang, Pakistan, the adoption of drip methods reduced water usage by 67–68% during the production of tomato, cucumber and bell peppers, resulting in a 68–79% improvement in water use efficiency compared to previous furrow irrigation (Niaz et al. 2009). In India, drip irrigation reduced the amount of water consumed in the production of sugarcane by 44%, grapes by 37%, bananas by 29% and cotton by 45%, while enhancing yields by up to 29% (Narayanamoorthy 2010). Similarly, in Uzbekistan, drip irrigation increased the yield of cotton by 10–19% while reducing water requirements by 18–42% (Ibragimov et al. 2007).

A prominent response that addresses soil loss, health and cover is altering cropping methods. The adoption of intercropping (inter – and intra-row planting of companion crops) and relay cropping (temporally differentiated planting of companion crops) maintains soil cover over a larger fraction of the year, leading to an increase in production, soil nitrogen, species diversity and a decrease in pest abundance (*robust evidence*, *medium agreement*) (Altieri and Koohafkan 2008; Tanveer et al. 2017; Wilhelm and Wortmann 2004). For example, intercropping maize and sorghum with *Desmodium* (an insect repellent forage legume) and *Brachiaria* (an insect trapping grass), which is being promoted in drylands of East Africa, led to a two-to-three-fold increase in maize production and an 80% decrease in stem boring insects (Khan et al. 2014). In addition to changes in cropping methods, forms of agroforestry and shelterbelts are often used to reduce erosion and improve soil conditions (Section 3.7.2). For example, the use of tree belts of mixed species in northern China led to a reduction of surface wind speed and an associated reduction in soil temperature of up to 40% and an increase in soil moisture of up to 30% (Wang et al. 2008).

A further measure that can be of increasing importance under climate change is rainwater harvesting (RWH), including traditional *zai* (small basins used to capture surface runoff), earthen bunds and ridges (Nyamadzawo et al. 2013), *fanya juus* infiltration pits (Nyagumbo et al. 2019), contour stone bunds (Garrity et al. 2010) and semi-permeable stone bunds (often referred to by the French term *digue filtrante*) (Taye et al. 2015). RWH increases the amount of water available for agriculture and livelihoods through the capture and storage of runoff, while at the same time reducing the intensity of peak flows following high-intensity rainfall events. It is therefore often highlighted as a practical response to dryness (i.e., long-term aridity and low seasonal precipitation) and rainfall variability, both of which are projected to become more acute over time in some dryland areas (Dile et al. 2013; Vohland and Barry 2009). For example, for drainage in Wadi Al-Lith, Saudi Arabia, the use of rainwater harvesting was suggested as a key climate change adaptation action (Almazroui et al. 2017). There is *robust evidence* and *high agreement* that the implementation of RWH systems leads to an increase in agricultural production in drylands (Biazin et al. 2012; Bouma and Wösten 2016; Dile et al. 2013). A global meta-analysis of changes in crop production due to the adoption of RWH techniques noted an average increase in yields of 78%, ranging from –28% to 468% (Bouma and Wösten 2016). Of particular relevance to climate change in drylands is that the relative impact of RWH on agricultural production generally increases with increasing dryness. Relative yield improvements due to the adoption of RWH were significantly higher in years with less than 330 mm rainfall, compared to years with more than 330 mm (Bouma and Wösten 2016). Despite delivering a clear set of benefits, there are some issues that need to be considered. The impact of RWH may vary at different temporal and spatial scales (Vohland and Barry 2009). At a plot scale, RWH structures may increase available water and enhance agricultural production, SOC and nutrient availability, yet at a catchment scale, they may reduce runoff to downstream uses (Meijer et al. 2013; Singh et al. 2012; Vohland and Barry 2009; Yosef and Asmamaw 2015). Inappropriate storage of water in warm climes can lead to an increase in water related diseases unless managed correctly, for example, schistosomiasis and malaria (Boelee et al. 2013).

Integrated crop–soil–water management may also deliver climate change *mitigation* benefits through avoiding, reducing and reversing the loss of SOC (Table 6.5). Approximately 20–30 Pg of SOC have been released into the atmosphere through desertification processes, for example, deforestation, overgrazing and conventional tillage (Lal 2004). Activities, such as those associated with conservation agriculture (minimising tillage, crop rotation, maintaining organic cover and planting a diversity of species), reduce erosion, improve water use efficiency and primary production, increase inflow of organic material and enhance SOC over time, contributing to climate change mitigation and adaptation (*high confidence*) (Plaza-Bonilla et al. 2015; Lal 2015; Srinivasa Rao et al. 2015; Sombrero and de Benito 2010). Conservation agriculture practices also lead to increases in SOC (*medium confidence*). However, sustained carbon sequestration is dependent on net primary productivity and on the availability of crop-residues that may be relatively limited and often consumed by livestock or used elsewhere in dryland contexts (Cheesman et al. 2016; Plaza-Bonilla et al. 2015). For this reason, expected rates of carbon sequestration following changes in agricultural practices in drylands are relatively low (0.04–0.4 tC ha^{-1}) and it may take a protracted period of time, even several decades, for carbon stocks to recover if lost (*medium confidence*) (Farage et al. 2007; Hoyle et al. 2013; Lal 2004). This long recovery period enforces the rationale for prioritising the avoidance and reduction of land degradation and loss of C, in addition to restoration activities.

1.6.1.2 Grazing and fire management in drylands

Rangeland management systems such as sustainable grazing approaches and re-vegetation increase rangeland productivity (*high confidence*) (Table 6.5). Open grassland, savannah and woodland are home to the majority of world's livestock production (Safriel et al. 2005). Within these drylands areas, prevailing grazing and fire regimes play an important role in shaping the relative abundance of trees versus grasses (Scholes and Archer 1997; Staver et al. 2011; Stevens et al. 2017), as well as the health of the grass layer in terms of primary production, species richness and basal cover (the propotion of the plant that is in the soil) (Plaza-Bonilla et al. 2015; Short et al. 2003). This in turn influences levels of soil erosion, soil nutrients, secondary production and additional ecosystem services (Divinsky et al. 2017; Pellegrini et al. 2017). A further set of drivers, including soil type, annual rainfall and changes in atmospheric CO_2 may also define observed rangeland structure and composition (Devine et al. 2017; Donohue et al. 2013), but the two principal factors that pastoralists can manage are grazing and fire, by altering their frequency, type and intensity.

The impact of grazing and fire regimes on biodiversity, soil nutrients, primary production and further ecosystem services is not constant and varies between locations (Divinsky et al. 2017; Fleischner 1994; van Oijen et al. 2018). Trade-offs may therefore need to be considered to ensure that rangeland diversity and production are resilient to climate change (Plaza-Bonilla et al. 2015; van Oijen et al. 2018). In certain locations, even light to moderate grazing has led to a significant decrease in the occurrence of particular species, especially forbs (O'Connor et al. 2011; Scott-shaw and Morris 2015). In other locations, species richness is only significantly impacted by heavy grazing and is able to withstand light to moderate grazing (Divinsky et al. 2017). A context specific evaluation of how grazing and fire impact particular species may therefore be required to ensure the persistence of target species over time (Marty 2005). A similar trade-off may need to be considered between soil carbon sequestration and livestock production. As noted by Plaza-Bonilla et al. (2015) increasing grazing pressure has been found to increase SOC stocks in some locations, and decrease them in others. Where it has led to a decrease in soil carbon stocks, for example in Mongolia (Han et al. 2008) and Ethiopia (Bikila et al. 2016), trade-offs between carbon sequestration and the value of livestock to local livelihoods need be considered.

Although certain herbaceous species may be unable to tolerate grazing pressure, a complete lack of grazing or fire may not be desired in terms of ecosystems health. It can lead to a decrease in basal cover and the accumulation of moribund, unpalatable biomass that inhibits primary production (Manson et al. 2007; Scholes 2009). The utilisation of the grass sward through light to moderate grazing stimulates the growth of biomass and basal cover, and allows water services to be sustained over time (Papanastasis et al. 2017; Scholes 2009). Even moderate to heavy grazing in periods of higher rainfall may be sustainable, but constant heavy grazing during dry periods, and especially droughts, can lead to a reduction in basal cover, SOC, biological soil crusts, ecosystem services and an accelerated erosion (*high agreement*, *robust evidence*) (Archer et al. 2017; Conant and Paustian 2003; D'Odorico et al. 2013; Geist and Lambin 2004; Havstad et al. 2006; Huang et al. 2007; Manzano and Návar 2000; Pointing and Belnap 2012; Weber et al. 2016). For this reason, the inclusion of drought forecasts and contingency planning in grazing and fire management programmes is crucial to avoid desertification (Smith and Foran 1992; Torell et al. 2010). It is an important component of avoiding and reducing early degradation. Although grasslands systems may be relatively resilient and can often recover from a moderately degraded state (Khishigbayar et al. 2015; Porensky et al. 2016), if a tipping point has been exceeded, restoration to a historic state may not be economical or ecologically feasible (D'Odorico et al. 2013).

Together with livestock management (Table 6.5), the use of fire is an integral part of rangeland management, which can be applied to remove moribund and unpalatable forage, exotic weeds and woody species (Archer et al. 2017). Fire has less of an effect on SOC and soil nutrients in comparison to grazing (Abril et al. 2005), yet elevated fire frequency has been observed to lead to a decrease in soil carbon and nitrogen (Abril et al. 2005; Bikila et al. 2016; Bird et al. 2000; Pellegrini et al. 2017). Although the impact of climate change on fire frequency and intensity may not be clear due to its differing impact on fuel accumulation, suitable weather conditions and sources of ignition (Abatzoglou et al. 2018; Littell et al. 2018; Moritz et al. 2012), there is an increasing use of prescribed fire to address several global change phenomena, for example, the spread of invasive species and bush encroachment, as well as the threat of intense runaway fires (Fernandes et al. 2013; McCaw 2013; van Wilgen et al. 2010). Cross-Chapter Box 3 in Chapter 2 provides a further review of the interaction between fire and climate change.

There is often much emphasis on reducing and reversing the degradation of rangelands due to the wealth of benefits they provide, especially in the context of assisting dryland communities to adapt to climate change (Webb et al. 2017; Woollen et al. 2016). The emerging concept of ecosystem-based adaptation has highlighted the broad range of important ecosystem services that healthy rangelands can provide in a resilient manner to local residents and downstream economies (Kloos and Renaud 2016; Reid et al. 2018). In terms of climate change mitigation, the contribution of rangelands, woodland and sub-humid dry forest (e.g., Miombo woodland in south-central Africa) is often undervalued due to relatively low carbon stocks per hectare. Yet due to their sheer extent, the amount of carbon sequestered in these ecosystems is substantial and can make a valuable contribution to climate change mitigation (Lal 2004; Pelletier et al. 2018).

1.6.1.3 Clearance of bush encroachment

The encroachment of open grassland and savannah ecosystems by woody species has occurred for at least the past 100 years (Archer et al. 2017; O'Connor et al. 2014; Schooley et al. 2018). Dependent on the type and intensity of encroachment, it may lead to a net loss of ecosystem services and be viewed as a form of desertification (Dougill et al. 2016; O'Connor et al. 2014). However, there are circumstances where bush encroachment may lead to a net increase in ecosystem services, especially at intermediate levels of encroachment, where the ability of the landscape to produce fodder for livestock is retained, while the production of wood and associated products increases (Eldridge et al. 2011; Eldridge and Soliveres 2014). This may be particularly important in regions such as southern Africa and India where over 65% of rural households depend on fuelwood from surrounding landscapes as well as livestock production (Komala and Prasad 2016; Makonese et al. 2017; Shackleton and Shackleton 2004).

This variable relationship between the level of encroachment, carbon stocks, biodiversity, provision of water and pastoral value (Eldridge and Soliveres 2014) can present a conundrum to policymakers, especially when considering the goals of three Rio Conventions: UNFCCC, UNCCD and UNCBD. Clearing intense bush encroachment may improve species diversity, rangeland productivity, the provision of water and decrease desertification, thereby contributing to the goals of the UNCBD and UNCCD as well as the adaptation aims of the UNFCCC. However, it would lead to the release of biomass carbon stocks into the atmosphere and potentially conflict with the mitigation aims of the UNFCCC.

For example, Smit et al. (2015) observed an average increase in above-ground woody carbon stocks of 44 tC ha^{-1} in savannahs in northern Namibia. However, since bush encroachment significantly inhibited livestock production, there are often substantial efforts to clear woody species (Stafford-Smith et al. 2017). Namibia has a national programme, currently in its early stages, aimed at clearing woody species through mechanical measures (harvesting of trees) as well as the application of arboricides (Smit et al. 2015). However, the long-term success of clearance and subsequent improved fire and grazing management remains to be evaluated, especially restoration back towards an 'original open grassland state'. For example, in northern Namibia, the rapid reestablishment of woody seedlings has raised questions about whether full clearance and restoration is possible (Smit et al. 2015). In arid landscapes, the potential impact of elevated atmospheric CO_2 (Donohue et al. 2013; Kgope et al. 2010) and opportunity to implement high-intensity fires that remove woody species and maintain rangelands in an open state has been questioned (Bond and Midgley 2000). If these drivers of woody plant encroachment cannot be addressed, a new form of 'emerging ecosystem' (Milton 2003) may need to be explored that includes both improved livestock and fire management as well as the utilisation of biomass as a long-term commodity and source of revenue (Smit et al. 2015). Initial studies in Namibia and South Africa (Stafford-Smith et al. 2017) indicate that there may be good opportunity to produce sawn timber, fencing poles, fuelwood and commercial energy, but factors such as the cost of transport can substantially influence the financial feasibility of implementation.

The benefit of proactive management that prevents land from being degraded (altering grazing systems or treating bush encroachment at early stages before degradation has been initiated) is more cost-effective in the long term and adds more resistance to climate change than treating lands after degradation has occurred (Webb et al. 2013; Weltz and Spaeth 2012). The challenge is getting producers to alter their management paradigm from short-term objectives to long-term objectives.

1.6.1.4 Combating sand and dust storms through sand dune stabilisation

Dust and sand storms have a considerable impact on natural and human systems (Sections 3.4.1 and 3.4.2). Application of sand dune stabilisation techniques contributes to reducing sand and dust storms (*high confidence*). Using a number of methods, sand dune stabilisation aims to avoid and reduce the occurrence of dust and sand storms (Mainguet and Dumay 2011). Mechanical techniques include building palisades to prevent the movement of sand and reduce sand deposits on infrastructure. Chemical methods include the use of calcium bentonite or using silica gel to fix mobile sand (Aboushook et al. 2012; Rammal and Jubair 2015). Biological methods include the use of mulch to stabilise surfaces (Sebaa et al. 2015; Yu et al. 2004) and establishing permanent plant cover using pasture species that improve grazing at the same time (Abdelkebir and Ferchichi 2015; Zhang et al. 2015) (Section 3.7.1.3). When the dune is stabilised, woody perennials are introduced that are selected according to climatic and ecological conditions (FAO 2011). For example, such re-vegetation processes have been implemented on the shifting dunes of the Tengger Desert in northern China leading to the stabilisation of sand and the sequestration of up to 10 tC ha^{-1} over a period of 55 years (Yang et al. 2014).

1.6.1.5 Use of halophytes for the re-vegetation of saline lands

Soil salinity and sodicity can severely limit the growth and productivity of crops (Jan et al. 2017) and lead to a decrease in available arable land. Leaching and drainage provides a possible solution, but can be prohibitively expensive. An alternative, more economical option, is the growth of halophytes (plants that are adapted to grow under highly saline conditions) that allow saline land to be used in a productive manner (Qadir et al. 2000). The biomass produced can be used as forage, food, feed, essential oils, biofuel, timber, or fuelwood (Chughtai et al. 2015; Mahmood et al. 2016; Sharma et al. 2016). A further co-benefit is the opportunity to mitigate climate change through the enhancement of terrestrial carbon stocks as land is re-vegetated (Dagar et al. 2014; Wicke et al. 2013). The combined use of salt-tolerant crops, improved irrigation practices, chemical remediation measures and appropriate mulch and compost is effective in reducing the impact of secondary salinisation (*medium confidence*).

In Pakistan, where about 6.2 Mha of agricultural land is affected by salinity, pioneering work on utilising salt-tolerant plants for the re-vegetation of saline lands (biosaline agriculture) was done in the early 1970s (NIAB 1997). A number of local and exotic varieties were initially screened for salt tolerance in lab – and greenhouse-based studies, and then distributed to similar saline areas (Ashraf et al. 2010). These included tree species (*Acacia ampliceps, Acacia nilotica, Eucalyptus camaldulensis, Prosopis juliflora, Azadirachta indica*) (Awan and Mahmood 2017), forage plants (*Leptochloa fusca*, *Sporobolus arabicus*, *Brachiaria mutica*, *Echinochloa* sp., *Sesbania* and *Atriplex* spp.) and crop species including varieties of barley (*Hordeum vulgare*), cotton, wheat (*Triticum aestivum*) and *Brassica* spp. (Mahmood et al. 2016) as well as fruit crops in the form of date palm (*Phoenix dactylifera*) that has high salt tolerance with no visible adverse effects on seedlings (Yaish and Kumar 2015; Al-Mulla et al. 2013; Alrasbi et al. 2010). Pomegranate (*Punica granatum L.*) is another fruit crop of moderate to high salt tolerance. Through regulating growth form and nutrient balancing, it can maintain water content, chlorophyll fluorescence and enzyme activity at normal levels (Ibrahim 2016; Okhovatian-Ardakani et al. 2010).

In India and elsewhere, tree species including *Prosopis juliflora, Dalbergia sissoo*, and *Eucalyptus tereticornis* have been used to re-vegetate saline land. Certain biofuel crops in the form of *Ricinus communis* (Abideen et al. 2014), *Euphorbia antisyphilitica* (Dagar et al. 2014), *Karelinia caspia* (Akinshina et al. 2016) and *Salicornia* spp. (Sanandiya and Siddhanta 2014) are grown in saline areas, and *Panicum turgidum* (Koyro et al. 2013) and *Leptochloa fusca* (Akhter et al. 2003) have been grown as fodder crop on degraded soils with brackish water. In China, intense efforts are being made on the use of halophytes (Sakai et al. 2012; Wang et al. 2018). These examples reveal that there is great scope for saline areas to be used in a productive manner through the utilisation of halophytes. The most productive species often have yields equivalent to conventional crops, at salinity levels matching even that of seawater.

1.6.2 Socio-economic responses

Socio-economic and policy responses are often crucial in enhancing the adoption of SLM practices (Cordingley et al. 2015; Fleskens and Stringer 2014; Nyanga et al. 2016) and for assisting agricultural households to diversify their sources of income (Barrett et al. 2017; Shiferaw and Djido 2016). Technology and socio-economic responses are not independent, but continuously interact.

1.6.2.1 Socio-economic responses for combating desertification under climate change

Desertification limits the choice of potential climate change mitigation and adaptation response options by reducing climate change adaptive capacities. Furthermore, many additional factors, for example, a lack of access to markets or insecurity of land tenure, hinder the adoption of SLM. These factors are largely beyond the control of individuals or local communities and require broader policy interventions (Section 3.6.3). Nevertheless, local collective action and ILK are still crucial to the ability of households to respond to the combined challenge of climate change and desertification. Raising awareness, capacity building and development to promote collective action and indigenous and local knowledge contribute to avoiding, reducing and reversing desertification under changing climate.

The use of indigenous and local knowledge enhances the success of SLM and its ability to address desertification (Altieri and Nicholls 2017; Engdawork and Bork 2016). Using indigenous and local knowledge for combating desertification could contribute to climate change adaptation strategies (Belfer et al. 2017; Codjoe et al. 2014; Etchart 2017; Speranza et al. 2010; Makondo and Thomas 2018; Maldonado et al. 2016; Nyong et al. 2007). There are abundant examples of how indigenous and local knowledge, which are an important part of broader agroecological knowledge (Altieri 2018), have allowed livelihood systems in drylands to be maintained despite environmental constraints. An example is the numerous traditional water harvesting techniques that are used across the drylands to adapt to dry spells and climate change. These include creating planting pits (*zai*, *ngoro*) and micro-basins, contouring hill slopes and terracing (Biazin et al. 2012) (Section 3.6.1). Traditional *ndiva* water harvesting systems in Tanzania enable the capture of runoff water from highland areas to downstream community-managed micro-dams for subsequent farm delivery through small-scale canal networks (Enfors and Gordon 2008). A further example are pastoralist communities located in drylands who have developed numerous methods to sustainably manage rangelands. Pastoralist communities in Morocco developed the *agdal* system of seasonally alternating use of rangelands to limit overgrazing (Dominguez 2014) as well as to manage forests in the Moroccan High Atlas Mountains (Auclair et al. 2011). Across the Arabian Peninsula and North Africa, a rotational grazing system, *hema*, was historically practiced by the Bedouin communities (Hussein 2011; Louhaichi and Tastad 2010). The Beni-Amer herders in the Horn of Africa have developed complex livestock breeding and selection systems (Fre 2018). Although well adapted to resource-sparse dryland environments, traditional practices are currently not able to cope with increased demand for food and environmental changes (Enfors and Gordon 2008; Engdawork and Bork 2016). Moreover, there is *robust evidence* documenting the marginalisation or loss of indigenous and local knowledge (Dominguez 2014; Fernández-Giménez and Fillat Estaque 2012; Hussein 2011; Kodirekkala 2017; Moreno-Calles et al. 2012). Combined use of indigenous and local knowledge and new SLM technologies can contribute to raising resilience to the challenges of climate change and desertification (*high confidence*) (Engdawork and Bork 2016; Guzman et al. 2018).

Collective action has the potential to contribute to SLM and climate change adaptation (*medium confidence*) (Adger 2003; Engdawork and Bork 2016; Eriksen and Lind 2009; Ostrom 2009; Rodima-Taylor et al. 2012). Collective action is a result of social capital. Social capital is divided into structural and cognitive forms: structural corresponding to strong networks (including outside one's immediate community); and cognitive encompassing mutual trust and cooperation within communities (van Rijn et al. 2012; Woolcock and Narayan 2000). Social capital is more important for economic growth in settings with weak formal institutions, and less so in those with strong enforcement of formal institutions (Ahlerup et al. 2009). There are cases throughout the drylands showing that community by-laws and collective action successfully limited land degradation and facilitated SLM (Ajayi et al. 2016; Infante 2017; Kassie et al. 2013; Nyangena 2008; Willy and Holm-Müller 2013; Wossen et al. 2015). However, there are also cases when they did not improve SLM where they were not strictly enforced (Teshome et al. 2016). Collective action for implementing responses to dryland degradation is often hindered by local asymmetric power relations and 'elite capture' (Kihiu 2016; Stringer et al. 2007). This illustrates that different levels and types of social capital result in different levels of collective action. In a sample of East, West and southern African countries, structural social capital in the form of access to networks outside one's own community was suggested to stimulate the adoption of agricultural innovations, whereas cognitive social capital, associated with inward-looking community norms of trust and cooperation, was found to have a negative relationship with the adoption of agricultural innovations (van Rijn et al. 2012). The latter is indirectly corroborated by observations of the impact of community-based rangeland management organisations in Mongolia. Although levels of cognitive social capital did not differ between them, communities with strong links to outside networks were able to apply more innovative rangeland management practices in comparison to communities without such links (Ulambayar et al. 2017).

Farmer-led innovations. Agricultural households are not just passive adopters of externally developed technologies, but are active experimenters and innovators (Reij and Waters-Bayer 2001; Tambo and Wünscher 2015; Waters-Bayer et al. 2009). SLM technologies co-generated through direct participation of agricultural households have higher chances of being accepted by them (*medium confidence*) (Bonney et al. 2016; Vente et al. 2016). Usually farmer-driven innovations are more frugal and better adapted to their resource scarcities than externally introduced technologies (Gupta et al. 2016). Farmer-to-farmer sharing of their own innovations and mutual learning positively contribute to higher technology adoption rates (Dey et al. 2017). This innovative ability can be given a new dynamism by combining it with emerging external technologies. For example, emerging low-cost phone applications ('apps') that are linked to soil and water monitoring sensors can provide farmers with previously inaccessible information and guidance (Cornell et al. 2013; Herrick et al. 2017; McKinley et al. 2017; Steger et al. 2017).

Currently, the adoption of SLM practices remains insufficient to address desertification and contribute to climate change adaptation and mitigation more extensively. This is due to the constraints on the use of indigenous and local knowledge and collective action, as well as economic and institutional barriers for SLM adoption (Banadda 2010; Cordingley et al. 2015; Lokonon and Mbaye 2018; Mulinge et al. 2016; Wildemeersch et al. 2015) (Section 3.1.4.2; 3.6.3). Sustainable development of drylands under these socio-economic and environmental (climate change, desertification) conditions will also depend on the ability of dryland agricultural households to diversify their livelihoods sources (Boserup 1965; Safriel and Adeel 2008).

1.6.2.2 Socio-economic responses for economic diversification

Livelihood diversification through non-farm employment increases the resilience of rural households against desertification and extreme weather events by diversifying their income and consumption (*high confidence*). Moreover, it can provide the funds to invest into SLM (Belay et al. 2017; Bryan et al. 2009; Dumenu and Obeng 2016; Salik et al. 2017; Shiferaw et al. 2009). Access to non-agricultural employment is especially important for poorer pastoral households as their small herd sizes make them less resilient to drought (Fratkin 2013; Lybbert et al. 2004). However, access to alternative opportunities is limited in the rural areas of many developing countries, especially for women and marginalised groups who lack education and social networks (Reardon et al. 2008).

Migration is frequently used as an adaptation strategy to environmental change (*medium confidence*). Migration is a form of livelihood diversification and a potential response option to desertification and increasing risk to agricultural livelihoods under climate change (Walther et al. 2002). Migration can be short-term (e.g., seasonal) or long-term, internal within a country or international. There is *medium evidence* showing rural households responding to desertification and droughts through all forms of migration, for example: during the Dust Bowl in the USA in the 1930s (Hornbeck 2012); during droughts in Burkina Faso in the 2000s (Barbier et al. 2009); in Mexico in the 1990s (Nawrotzki et al. 2016); and by the Aymara people of the semi-arid Tarapacá region in Chile between 1820 and 1970, responding to declines in rainfall and growing demands for labour outside the region (Lima et al. 2016). There is *robust evidence* and *high agreement* showing that migration decisions are influenced by a complex set of different factors, with desertification and climate change playing relatively lesser roles (Liehr et al. 2016) (Section 3.4.2). Barrios et al. (2006) found that urbanisation in Sub-Saharan Africa was partially influenced by climatic factors during the 1950–2000 period, in parallel to liberalisation of internal restrictions on labour movements: each 1% reduction in rainfall was associated with a 0.45% increase in urbanisation. This migration favoured more industrially diverse urban areas in Sub-Saharan Africa (Henderson et al. 2017), because they offer more diverse employment opportunities and higher wages. Similar trends were also observed in Iran in response to water scarcity (Madani et al. 2016).

However, migration involves some initial investments. For this reason, reductions in agricultural incomes due to climate change or desertification have the potential to decrease out-migration among the poorest agricultural households, who become less able to afford migration (Cattaneo and Peri 2016), thus increasing social inequalities. There is *medium evidence* and *high agreement* that households with migrant worker members are more resilient against extreme weather events and environmental degradation compared to non-migrant households, who are more dependent on agricultural income (Liehr et al. 2016; Salik et al. 2017; Sikder and Higgins 2017). Remittances from migrant household members potentially contribute to SLM adoptions, however, substantial out-migration was also found to constrain the implementation of labour-intensive land management practices (Chen et al. 2014; Liu et al. 2016a).

1.6.3 Policy responses

The adoption of SLM practices depends on the compatibility of the technology with prevailing socio-economic and biophysical conditions (Sanz et al. 2017). Globally, it was shown that every USD invested into restoring degraded lands yields social returns, including both provisioning and non-provisioning ecosystem services, in the range of 3–6 USD over a 30-year period (Nkonya et al. 2016a). A similar range of returns from land restoration activities was found in Central Asia (Mirzabaev et al. 2016), Ethiopia (Gebreselassie et al. 2016), India (Mythili and Goedecke 2016), Kenya (Mulinge et al. 2016), Niger (Moussa et al. 2016) and Senegal (Sow et al. 2016) (*medium confidence*). Despite these relatively high returns, there is *robust evidence* that the adoption of SLM practices remains low (Cordingley et al. 2015; Giger et al. 2015; Lokonon and Mbaye 2018). Part of the reason for these low adoption rates is that the major share of the returns from SLM are social benefits, namely in the form of non-provisioning ecosystem services (Nkonya et al. 2016a). The adoption of SLM technologies does not always provide implementers with immediate private benefits (Schmidt et al. 2017). High initial investment costs, institutional and governance constraints and a lack of access to technologies and equipment may inhibit their adoption further (Giger et al. 2015; Sanz et al. 2017; Schmidt et al. 2017). However, not all SLM practices have high upfront costs. Analysing the World Overview of Conservation Approaches and Technologies (WOCAT) database, a globally acknowledged reference database for SLM, Giger et al. (2015) found that the upfront costs of SLM technologies ranged from about 20 USD to 5000 USD, with the median cost being around 500 USD. Many SLM technologies are profitable within 3 to 10 years (*medium confidence*) (Djanibekov and Khamzina 2016; Giger et al. 2015; Moussa et al. 2016; Sow et al. 2016). About 73% of 363 SLM technologies evaluated were reported to become profitable within three years, while 97% were profitable within 10 years (Giger et al. 2015). Similarly, it was shown that social returns from investments in restoring degraded lands will exceed their costs within six years in many settings across drylands (Nkonya et al. 2016a). However, even with affordable upfront costs, market failures – in the form of lack of access to credit, input and output markets, and insecure land tenure (Section 3.1.3) – result in the lack of adoption of SLM technologies (Moussa et al. 2016). Payments for ecosystem services, subsidies for SLM, and encouragement of community collective action can lead to a higher level of adoption of SLM and land restoration activities (*medium confidence*) (Bouma and Wösten 2016; Lambin et al. 2014; Reed et al. 2015; Schiappacasse et al. 2012; van Zanten et al. 2014) (Section 3.6.3). Enabling the policy responses discussed in this section will contribute to overcoming these market failures.

Many socio-economic factors shaping individual responses to desertification typically operate at larger scales. Individual households and communities do not exercise control over these factors, such as land tenure insecurity, lack of property rights, lack of access to markets, availability of rural advisory services, and agricultural price distortions. These factors are shaped by national government policies and international markets. As is the case with socio-economic responses, policy responses are classified below in two ways: those which seek to combat desertification under changing climate; and

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those which seek to provide alternative livelihood sources through economic diversification. These options are mutually complementary and contribute to all the three hierarchical elements of the Land Degradation Neutrality (LDN) framework, namely, avoiding, reducing and reversing land degradation (Cowie et al. 2018; Orr et al. 2017) (Sections 4.8.5 and 7.4.5, and Table 7.2). An enabling policy environment is a critical element for the achievement of LDN (Chasek et al. 2019). Implementation of LDN policies can contribute to climate change adaptation and mitigation (*high confidence*) (Sections 3.6.1 and 3.7.2).

1.6.3.1 Policy responses towards combating desertification under climate change

Policy responses to combat desertification take numerous forms (Marques et al. 2016). Below we discuss major policy responses consistently highlighted in the literature in connection with SLM and climate change, because these response options were found to strengthen adaptation capacities and to contribute to climate change mitigation. They include improving market access, empowering women, expanding access to agricultural advisory services, strengthening land tenure security, payments for ecosystem services, decentralised natural resource management, investing into research and monitoring of desertification and dust storms, and investing into modern renewable energy sources.

Policies aiming at improving market access, that is the ability to access output and input markets at lower costs, help farmers and livestock producers earn more profit from their produce. Increased profits both motivate and enable them to invest more in SLM. Higher access to input, output and credit markets was consistently found as a major factor in the adoption of SLM practices in a wide number of settings across the drylands (*medium confidence*) (Aw-Hassan et al. 2016; Gebreselassie et al. 2016; Mythili and Goedecke 2016; Nkonya and Anderson 2015; Sow et al. 2016). Lack of access to credit limits adjustments and agricultural responses to the impacts of desertification under changing climate, with long-term consequences for the livelihoods and incomes, as was shown during the North American Dust Bowl of the 1930s (Hornbeck 2012). Government policies aimed at improving market access usually involve constructing and upgrading rural–urban transportation infrastructure and agricultural value chains, such as investments into construction of local markets, abattoirs and cold storage warehouses, as well as post-harvest processing facilities (McPeak et al. 2006). However, besides infrastructural constraints, providing improved access often involves relieving institutional constraints to market access (Little 2010), such as improved coordination of cross-border food safety and veterinary regulations (Ait Hou et al. 2015; Keiichiro et al. 2015; McPeak et al. 2006; Unnevehr 2015), and availability and access to market information systems (Bobojonov et al. 2016; Christy et al. 2014; Nakasone et al. 2014).

Women's empowerment. A greater emphasis on understanding gender-specific differences over land use and land management practices as an entry point can make land restoration projects more successful (*medium confidence*) (Broeckhoven and Cliquet 2015; Carr and Thompson 2014; Catacutan and Villamor 2016; Dah-gbeto and Villamor 2016). In relation to representation and authority to make decisions in land management and governance, women's participation remains lacking particularly in the dryland regions. Thus, ensuring women's rights means accepting women as equal members of the community and citizens of the state (Nelson et al. 2015). This includes equitable access of women to resources (including extension services), networks, and markets. In areas where socio-cultural norms and practices devalue women and undermine their participation, actions for empowering women will require changes in customary norms, recognition of women's (land) rights in government policies, and programmes to assure that their interests are better represented (Section 1.4.2 and Cross-Chapter Box 11 in Chapter 7). In addition, several novel concepts are recently applied for an in-depth understanding of gender in relation to science–policy interface. Among these are the concepts of intersectionality, that is, how social dimensions of identity and gender are bound up in systems of power and social institutions (Thompson-Hall et al. 2016), bounded rationality for gendered decision-making, related to incomplete information interacting with limits to human cognition leading to judgement errors or objectively poor decision making (Villamor and van Noordwijk 2016), anticipatory learning for preparing for possible contingencies and consideration of long-term alternatives (Dah-gbeto and Villamor 2016) and systematic leverage points for interventions that produce, mark, and entrench gender inequality within communities (Manlosa et al. 2018), which all aim to improve gender equality within agroecological landscapes through a systems approach.

Education and expanding access to agricultural services. Providing access to information about SLM practices facilitates their adoption (*medium confidence*) (Kassie et al. 2015; Nkonya et al. 2015; Nyanga et al. 2016). Moreover, improving the knowledge of climate change, capacity building and development in rural areas can help strengthen climate change adaptive capacities (Berman et al. 2012; Chen et al. 2018; Descheemaeker et al. 2018; Popp et al. 2009; Tambo 2016; Yaro et al. 2015). Agricultural initiatives to improve the adaptive capacities of vulnerable populations were more successful when they were conducted through reorganised social institutions and improved communication, for example, in Mozambique (Osbahr et al. 2008). Improved communication and education could be facilitated by wider use of new information and communication technologies (ICTs) (Peters et al. 2015). Investments into education were associated with higher adoption of soil conservation measures, for example, in Tanzania (Tenge et al. 2004). Bryan et al. (2009) found that access to information was the prominent facilitator of climate change adaptation in Ethiopia. However, resource constraints of agricultural services, and disconnects between agricultural policy and climate policy can hinder the dissemination of climate-smart agricultural technologies (Morton 2017). Lack of knowledge was also found to be a significant barrier to implementation of soil rehabilitation programmes in the Mediterranean region (Reichardt 2010). Agricultural services will be able to facilitate SLM best when they also serve as platforms for sharing indigenous and local knowledge and farmer innovations (Mapfumo et al. 2016). Participatory research initiatives conducted jointly with farmers have higher chances of resulting in technology adoption (Bonney et al. 2016; Rusike et al. 2006; Vente et al. 2016). Moreover, rural advisory

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services are often more successful in disseminating technological innovations when they adopt commodity/value chain approaches, remain open to engagement in input supply, make use of new opportunities presented by ICTs, facilitate mutual learning between multiple stakeholders (Morton 2017), and organise science and SLM information in a location-specific manner for use in education and extension (Bestelmeyer et al. 2017).

Strengthening land tenure security. Strengthening land tenure security is a major factor contributing to the adoption of soil conservation measures in croplands (*high confidence*) (Bambio and Bouayad Agha 2018; Higgins et al. 2018; Holden and Ghebru 2016; Paltasingh 2018; Rao et al. 2016; Robinson et al. 2018), thus contributing to climate change adaptation and mitigation. Moreover, land tenure security can lead to more investment in trees (Deininger and Jin 2006; Etongo et al. 2015). Land tenure recognition policies were found to lead to higher agricultural productivity and incomes, although with inter-regional variations, requiring an improved understanding of overlapping formal and informal land tenure rights (Lawry et al. 2017). For example, secure land tenure increased investments into SLM practices in Ghana, but without affecting farm productivity (Abdulai et al. 2011). Secure land tenure, especially for communally managed lands, helps reduce arbitrary appropriations of land for large-scale commercial farms (Aha and Ayitey 2017; Baumgartner 2017; Dell'Angelo et al. 2017). In contrast, privatisation of rangeland tenures in Botswana and Kenya led to the loss of communal grazing lands and actually increased rangeland degradation (Basupi et al. 2017; Kihiu 2016) as pastoralists needed to graze livestock on now smaller communal pastures. Since food insecurity in drylands is strongly affected by climate risks, there is *robust evidence* and *high agreement* that resilience to climate risks is higher with flexible tenure for allowing mobility for pastoralist communities, and not fragmenting their areas of movement (Behnke 1994; Holden and Ghebru 2016; Liao et al. 2017; Turner et al. 2016; Wario et al. 2016). More research is needed on the optimal tenure mix, including low-cost land certification, redistribution reforms, market-assisted reforms and gender-responsive reforms, as well as collective forms of land tenure such as communal land tenure and cooperative land tenure (see Section 7.6.5 for a broader discussion of land tenure security under climate change).

Payment for ecosystem services (PES) provides incentives for land restoration and SLM (*medium confidence*) (Lambin et al. 2014; Li et al. 2018; Reed et al. 2015; Schiappacasse et al. 2012). Several studies illustrate that the social costs of desertification are larger than its private cost (Costanza et al. 2014; Nkonya et al. 2016a). Therefore, although SLM can generate public goods in the form of provisioning ecosystem services, individual land custodians underinvest in SLM as they are unable to reap these benefits fully. Payment for ecosystem services provides a mechanism through which some of these benefits can be transferred to land users, thereby stimulating further investment in SLM. The effectiveness of PES schemes depends on land tenure security and appropriate design, taking into account specific local conditions (Börner et al. 2017). However, PES has not worked well in countries with fragile institutions (Karsenty and Ongolo 2012). Equity and justice in distributing the payments for ecosystem services were found to be key for the success of the PES programmes in Yunnan, China (He and Sikor 2015). Yet, when reviewing the performance of PES programmes in the tropics, Calvet-Mir et al. (2015), found that they are generally effective in terms of environmental outcomes, despite being sometimes unfair in terms of payment distribution. It is suggested that the implementation of PES will be improved through decentralised approaches giving local communities a larger role in the decision-making process (He and Lang 2015).

Empowering local communities for decentralised natural resource management. Local institutions often play a vital role in implementing SLM initiatives and climate change adaptation measures (*high confidence*) (Gibson et al. 2005; Smucker et al. 2015). Pastoralists involved in community-based natural resource management in Mongolia had greater capacity to adapt to extreme winter frosts, resulting in less damage to their livestock (Fernandez-Gimenez et al. 2015). Decreasing the power and role of traditional community institutions, due to top-down public policies, resulted in lower success rates in community-based programmes focused on rangeland management in Dirre, Ethiopia (Abdu and Robinson 2017). Decentralised governance was found to lead to improved management in forested landscapes (Dressler et al. 2010; Ostrom and Nagendra 2006). However, there are also cases when local elites were placed in control and this decentralised natural resource management negatively impacted the livelihoods of the poorer and marginalised community members due to reduced access to natural resources (Andersson and Ostrom 2008; Cullman 2015; Dressler et al. 2010).

The success of decentralised natural resource management initiatives depends on increased participation and empowerment of a diverse set of community members, not only local leaders and elites, in the design and management of local resource management institutions (Kadirbeyoglu and Özertan 2015; Umutoni et al. 2016), while considering the interactions between actors and institutions at different levels of governance (Andersson and Ostrom 2008; Carlisle and Gruby 2017; McCord et al. 2017). An example of such programmes where local communities played a major role in land restoration and rehabilitation activities is the cooperative project on The National Afforestation and Erosion Control Mobilization Action Plan in Turkey, initiated by the Turkish Ministry of Agriculture and Forestry (Çalişkan and Boydak 2017), with the investment of 1.8 billion USD between 2008 and 2012. The project mobilised local communities in cooperation with public institutions, municipalities, and non-governmental organisations, to implement afforestation, rehabilitation and erosion control measures, resulting in the afforestation and reforestation of 1.5 Mha (Yurtoglu 2015). Moreover, some 1.75 Mha of degraded forest and 37,880 ha of degraded rangelands were rehabilitated. Finally, the project provided employment opportunities for 300,000 rural residents for six months every year, combining land restoration and rehabilitation activities with measures to promote socio-economic development in rural areas (Çalişkan and Boydak 2017).

Investing in research and development. Desertification has received substantial research attention over recent decades (Turner et al. 2007). There is also a growing research interest on climate change adaptation and mitigation interventions that help address desertification (Grainger 2009). Agricultural research on SLM practices has generated a significant number of new innovations and technologies that increase crop yields without degrading the land, while contributing to climate change adaptation and mitigation (Section 3.6.1). There is *robust evidence* that such technologies help improve the food security of smallholder dryland farming households (Harris and Orr 2014) (Section 6.3.5). Strengthening research on desertification is of high importance not only to meet SDGs but also to manage ecosystems effectively, based on solid scientific knowledge. More investment in research institutes and training the younger generation of researchers is needed for addressing the combined challenges of desertification and climate change (Akhtar-Schuster et al. 2011; Verstraete et al. 2011). This includes improved knowledge management systems that allow stakeholders to work in a coordinated manner by enhancing timely, targeted and contextualised information sharing (Chasek et al. 2011). Knowledge and flow of knowledge on desertification is currently highly fragmented, constraining the effectiveness of those engaged in assessing and monitoring the phenomenon at various levels (Reed et al. 2011). Improved knowledge and data exchange and sharing increase the effectiveness of efforts to address desertification (*high confidence*).

Developing modern renewable energy sources. Transitioning to renewable energy resources contributes to reducing desertification by lowering reliance on traditional biomass in dryland regions (*medium confidence*). This can also have socioeconomic and health benefits, especially for women and children (*high confidence*). Populations in most developing countries continue to rely on traditional biomass, including fuelwood, crop straws and livestock manure, for a major share of their energy needs, with the highest dependence in Sub-Saharan Africa (Amugune et al. 2017; IEA 2013). Use of biomass for energy, mostly fuelwood (especially as charcoal), was associated with deforestation in some dryland areas (Iiyama et al. 2014; Mekuria et al. 2018; Neufeldt et al. 2015; Zulu 2010), while in some other areas there was no link between fuelwood collection and deforestation (Simon and Peterson 2018; Swemmer et al. 2018; Twine and Holdo 2016). Moreover, the use of traditional biomass as a source of energy was found to have negative health effects through indoor air pollution (de la Sota et al. 2018; Lim and Seow 2012), while also being associated with lower female labour force participation (Burke and Dundas 2015). Jiang et al. (2014) indicated that providing improved access to alternative energy sources such as solar energy and biogas could help reduce the use of fuelwood in south-western China, thus alleviating the spread of rocky desertification. The conversion of degraded lands into cultivation of biofuel crops will affect soil carbon dynamics (Albanito et al. 2016; Nair et al. 2011) (Cross-Chapter Box 7 in Chapter 6). The use of biogas slurry as soil amendment or fertiliser can increase soil carbon (Galvez et al. 2012; Negash et al. 2017). Large-scale installation of wind and solar farms in the Sahara Desert was projected to create a positive climate feedback through increased surface friction and reduced albedo, doubling precipitation over the neighbouring Sahel region with resulting increases in vegetation (Li et al. 2018). Transition to renewable energy sources in high-income countries in dryland areas primarily contributes to reducing GHG emissions and mitigating climate change, with some other co-benefits such as diversification of energy sources (Bang 2010), while the impacts on desertification are less evident. The use of renewable energy has been proposed as an important mitigation option in dryland areas as well (El-Fadel et al. 2003). Transitions to renewable energy are being promoted by governments across drylands (Cancino-Solórzano et al. 2016; Hong et al. 2013; Sen and Ganguly 2017) including in fossil-fuel rich countries (Farnoosh et al. 2014; Dehkordi et al. 2017; Stambouli et al. 2012; Vidadili et al. 2017), despite important social, political and technical barriers to expanding renewable energy production (Afsharzade et al. 2016; Baker et al. 2014; Elum and Momodu 2017; Karatayev et al. 2016). Improving social awareness about the benefits of transitioning to renewable energy resources, and access to hydro-energy, solar and wind energy contributes to their improved adoption (Aliyu et al. 2017; Katikiro 2016).

Developing and strengthening climate services relevant for desertification. Climate services provide climate, drought and desertification-related information in a way that assists decision-making by individuals and organisations. Monitoring desertification, and integrating biogeophysical (climate, soil, ecological factors, biodiversity) and socio-economic (use of natural resources by local population) issues provide a basis for better vulnerability prediction and assessment (OSS, 2012; Vogt et al. 2011). Examples of relevant services include: drought monitoring and early warning systems, often implemented by national climate and meteorological services but also encompassing regional and global systems (Pozzi et al. 2013); and the Sand and Dust Storm Warning Advisory and Assessment System (SDS-WAS), created by WMO in 2007, in partnership with the World Health Organization (WHO) and the United Nations Environment Program (UNEP). Currently, there is also a lack of ecological monitoring in arid and semi-arid regions to study surface winds, dust and sand storms, and their impacts on ecosystems and human health (Bergametti et al. 2018; Marticorena et al. 2010). Reliable and timely climate services, relevant to desertification, can aid the development of appropriate adaptation and mitigation options, reducing the impact of desertification under changing climate on human and natural systems (*high confidence*) (Beegum et al. 2016; Beegum et al. 2018; Cornet 2012; Haase et al. 2018; Sergeant et al. 2012).

1.6.3.2 Policy responses supporting economic diversification

Despite policy responses for combating desertification, other factors will put strong pressures on the land, including climate change and growing food demands, as well as the need to reduce poverty and strengthen food security (Cherlet et al. 2018) (Sections 6.1.4 and 7.2.2). Sustainable development of drylands and their resilience to combined challenges of desertification and climate change will thus also depend on the ability of governments to promote policies for economic diversification within agriculture and in non-agricultural sectors in order make dryland areas less vulnerable to desertification and climate change.

Investing into irrigation. Investments into expanding irrigation in dryland areas can help increase the resilience of agricultural production to climate change, improve labour productivity and boost production and income revenue from agriculture and livestock sectors (Geerts and Raes 2009; Olayide et al. 2016; Oweis and Hachum 2006). This is particularly true for Sub-Saharan Africa, where currently only 6% of the cultivated areas are irrigated (Nkonya et al. 2016b). While renewable groundwater resources could help increase the share of irrigated land to 20.5–48.6% of croplands in the region (Altchenko and Villholth 2015). On the other hand, over-extraction of groundwaters, mainly for irrigating crops, is becoming an important environmental problem in many dryland areas (Cherlet et al. 2018), requiring careful design and planning of irrigation expansion schemes and use of water-efficient irrigation methods (Bjornlund et al. 2017; Woodhouse et al. 2017). For example, in Saudi Arabia, improving the efficiency of water management, for example through the development of aquifers, water recycling and rainwater harvesting, is part of a suite of policy actions to combat desertification (Bazza, et al. 2018; Kingdom of Saudi Arabia 2016). The expansion of irrigation to riverine areas, crucial for dry season grazing of livestock, needs to consider the income from pastoral activities, which is not always lower than income from irrigated crop production (Behnke and Kerven 2013). Irrigation development could be combined with the deployment of clean-energy technologies in economically viable ways (Chandel et al. 2015). For example, solar-powered drip irrigation was found to increase household agricultural incomes in Benin (Burney et al. 2010). The sustainability of irrigation schemes based on solar-powered extraction of groundwaters depends on measures to avoid over-abstraction of groundwater resources and associated negative environmental impacts (Closas and Rap 2017).

Expanding agricultural commercialisation. Faster poverty rate reduction and economic growth enhancement is realised when countries transition into the production of non-staple, high-value commodities and manage to build a robust agro-industry sector (Barrett et al. 2017). Ogutu and Qaim (2019) found that agricultural commercialisation increased incomes and decreased multidimensional poverty in Kenya. Similar findings were earlier reported by Muriithi and Matz (2015) for commercialisation of vegetables in Kenya. Commercialisation of rice production was found to have increased smallholder welfare in Nigeria (Awotide et al. 2016). Agricultural commercialisation contributed to improved household food security in Malawi, Tanzania and Uganda (Carletto et al. 2017). However, such a transition did not improve farmers' livelihoods in all cases (Reardon et al. 2009). High-value cash crop/animal production can be bolstered by wide-scale use of technologies, for example, mechanisation, application of inorganic fertilisers, crop protection and animal health products. Market oriented crop/animal production facilitates social and economic progress, with labour increasingly shifting out of agriculture into non-agricultural sectors (Cour 2001). Modernised farming, improved access to inputs, credit and technologies enhances competitiveness in local and international markets (Reardon et al. 2009).

Facilitating structural transformations in rural economies implies that the development of non-agricultural sectors encourages the movement of labour from land-based livelihoods, vulnerable to desertification and climate change, to non-agricultural activities (Haggblade et al. 2010). The movement of labour from agriculture to non-agricultural sectors is determined by relative labour productivities in these sectors (Shiferaw and Djido 2016). Given already high underemployment in the farm sector, increasing labour productivity in the non-farm sector was found as the main driver of labour movements from farm sector to non-farm sector (Shiferaw and Djido 2016). More investments into education can facilitate this process (Headey et al. 2014). However, in some contexts, such as pastoralist communities in Xinjiang, China, income diversification was not found to improve the welfare of pastoral households (Liao et al. 2015). Economic transformations also occur through urbanisation, involving the shift of labour from rural areas into gainful employment in urban areas (Jedwab and Vollrath 2015). The majority of world population will be living in urban centres in the 21st century and this will require innovative means of agricultural production with minimum ecological footprint and less dependence on fossil fuels (Revi and Rosenzweig 2013), while addressing the demand of cities (see Section 4.9.1 for discussion on urban green infrastructure). Although there is some evidence of urbanisation leading to the loss of indigenous and local ecological knowledge, however, indigenous and local knowledge systems are constantly evolving, and are also being integrated into urban environments (Júnior et al. 2016; Reyes-García et al. 2013; van Andel and Carvalheiro 2013). Urban areas are attracting an increasing number of rural residents across the developing world (Angel et al. 2011; Cour 2001; Dahiya 2012). Urban development contributes to expedited agricultural commercialisation by providing market outlet for cash crops, high-value crops, and livestock products. At the same time, urbanisation also poses numerous challenges in the form of rapid urban sprawl and pressures on infrastructure and public services, unemployment and associated social risks, which have considerable implications on climate change adaptive capacities (Bulkeley 2013; Garschagen and Romero-Lankao 2015).

Cross-Chapter Box 5 | Policy responses to drought

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Drought is a highly complex natural hazard (for floods, see Box 7.2). It is difficult to precisely identify its start and end. It is usually slow and gradual (Wilhite and Pulwarty 2017), but sometimes can evolve rapidly (Ford and Labosier 2017; Mo and Lettenmaier 2015). It is context-dependent, but its impacts are diffuse, both direct and indirect, short-term and long-term (Few and Tebboth 2018; Wilhite and Pulwarty 2017). Following the Synthesis Report (SYR) of the IPCC Fifth Assessment Report (AR5), drought is defined here as "a period of abnormally dry weather long enough to cause a serious hydrological imbalance" (Mach et al. 2014). Although drought is considered abnormal relative to the water availability under the mean climatic characteristics, it is also a recurrent element of any climate, not only in drylands, but also in humid areas (Cook et al. 2014b; Seneviratne and Ciais 2017; Spinoni et al. 2019; Türkeş 1999; Wilhite et al. 2014). Climate change is projected to increase the intensity or frequency of droughts in some regions across the world (for a detailed assessment see Section 2.2, and IPCC Special Report on Global Warming of 1.5°C (Hoegh-Guldberg et al. 2018)). Droughts often amplify the effects of unsustainable land management practices, especially in drylands, leading to land degradation (Cook et al. 2009; Hornbeck 2012). Especially in the context of climate change, the recurrent nature of droughts requires proactively planned policy instruments both to be well-prepared to respond to droughts when they occur and also undertake *ex ante* actions to mitigate their impacts by strengthening societal resilience against droughts (Gerber and Mirzabaev 2017).

Droughts are among the costliest of natural hazards (*robust evidence*, *high agreement*). According to the International Disaster Database (EM-DAT), droughts affected more than 1.1 billion people between 1994 and 2013, with the recorded global economic damage of 787 billion USD (CRED 2015), corresponding to an average of 41.4 billion USD per year. Drought losses in the agricultural sector alone in developing countries were estimated to equal 29 billion USD between 2005 and 2015 (FAO 2018). Usually, these estimates capture only direct and on-site costs of droughts. However, droughts have also wide-ranging indirect and off-site impacts, which are seldom quantified. These indirect impacts are both biophysical and socio-economic, with poor households and communities being particularly exposed to them (Winsemius et al. 2018). Droughts affect not only water quantity, but also water quality (Mosley 2014). The costs of these water quality impacts are yet to be adequately quantified. Socio-economic indirect impacts of droughts are related to food insecurity, poverty, lowered health and displacement (Gray and Mueller 2012; Johnstone and Mazo 2011; Linke et al. 2015; Lohmann and Lechtenfeld 2015; Maystadt and Ecker 2014; Yusa et al. 2015) (Section 3.4.2.9 and Box 5.5), which are difficult to quantify comprehensively. Research is required for developing methodologies that could allow for more comprehensive assessment of these indirect drought costs. Such methodologies require the collection of highly granular data, which is currently lacking in many countries due to high costs of data collection. However, the opportunities provided by remotely sensed data and novel analytical methods based on big data and artificial intelligence, including use of citizen science for data collection, could help in reducing these gaps.

There are three broad (and sometimes overlapping) policy approaches for responding to droughts (Section 7.4.8). These approaches are often pursued simultaneously by many governments. Firstly, responding to drought when it occurs by providing direct drought relief, known as crisis management. Crisis management is also the costliest among policy approaches to droughts because it often incentivises the continuation of activities vulnerable to droughts (Botterill and Hayes 2012; Gerber and Mirzabaev 2017).

The second approach involves development of drought preparedness plans, which coordinate the policies for providing relief measures when droughts occur. For example, combining resources to respond to droughts at regional level in Sub-Saharan Africa was found to be more cost-effective than separate individual country drought relief funding (Clarke and Hill 2013). Effective drought preparedness plans require well-coordinated and integrated government actions – a key lesson learnt from 2015 to 2017 during drought response in Cape Town, South Africa (Visser 2018). Reliable, relevant and timely climate and weather information helps respond to droughts appropriately (Sivakumar and Ndiang'ui 2007). Improved knowledge and integration of weather and climate information can be achieved by strengthening drought early warning systems at different scales (Verbist et al. 2016). Every USD invested into strengthening hydro-meteorological and early warning services in developing countries was found to yield between 4 and 35 USD (Hallegatte 2012). Improved access and coverage by drought insurance, including index insurance, can help alleviate the impacts of droughts on livelihoods (Guerrero-Baena et al. 2019; Kath et al. 2019; Osgood et al. 2018; Ruiz et al. 2015; Tadesse et al. 2015).

The third category of responses to droughts involves drought risk mitigation. Drought risk mitigation is a set of proactive measures, policies and management activities aimed at reducing the future impacts of droughts (Vicente-Serrano et al. 2012). For example, *Cross-Chapter Box 5 (continued)*

policies aimed at improving water use efficiency in different sectors of the economy, especially in agriculture and industry, or public advocacy campaigns raising societal awareness and bringing about behavioural change to reduce wasteful water consumption in the residential sector are among such drought risk mitigation policies (Tsakiris 2017). Public outreach and monitoring of communicable diseases, air and water quality were found to be useful for reducing health impacts of droughts (Yusa et al. 2015). The evidence from household responses to drought in Cape Town, South Africa, between 2015 and 2017, suggests that media coverage and social media could play a decisive role in changing water consumption behaviour, even more so than official water consumption restrictions (Booysen et al. 2019). Drought risk mitigation approaches are less costly than providing drought relief after the occurrence of droughts. To illustrate, Harou et al. (2010) found that establishment of water markets in California considerably reduced drought costs. Application of water saving technologies reduced drought costs in Iran by 282 million USD (Salami et al. 2009). Booker et al. (2005) calculated that inter-regional trade in water could reduce drought costs by 20–30% in the Rio Grande basin, USA. Increasing rainfall variability under climate change can make the forms of index insurance based on rainfall less efficient (Kath et al. 2019). A number of diverse water property instruments, including instruments allowing water transfer, together with the technological and institutional ability to adjust water allocation, can improve timely adjustment to droughts (Hurlbert 2018). Supply-side water management, providing for proportionate reductions in water delivery, prevents the important climate change adaptation option of managing water according to need or demand (Hurlbert and Mussetta 2016). Exclusive use of a water market to govern water allocation similarly prevents the recognition of the human right to water at times of drought (Hurlbert 2018). Policies aiming to secure land tenure, and to expand access to markets, agricultural advisory services and effective climate services, as well as to create off-farm employment opportunities, can facilitate the adoption of drought risk mitigation practices (Alam 2015; Kusunose and Lybbert 2014), increasing resilience to climate change (Section 3.6.3), while also contributing to SLM (Sections 3.6.3 and 4.8.1, and Table 5.7).

The excessive burden of drought relief funding on public budgets is already leading to a paradigm shift towards proactive drought risk mitigation instead of reactive drought relief measures (Verner et al. 2018; Wilhite 2016). Climate change will reinforce the need for such proactive drought risk mitigation approaches. Policies for drought risk mitigation that are already needed now will be even more relevant under higher warming levels (Jerneck and Olsson 2008; McLeman 2013; Wilhite et al. 2014). Overall, there is *high confidence* that responding to droughts through *ex post* drought relief measures is less efficient compared to *ex ante* investments into drought risk mitigation, particularly under climate change.

1.6.4 Limits to adaptation, maladaptation, and barriers for mitigation

Chapter 16 in the IPCC Fifth Assessment Report (AR5) (Klein et al. 2015) discusses the existence of soft and hard limits to adaptation, highlighting that values and perspectives of involved agents are relevant to identify limits (Sections 4.8.5.1 and 7.4.9). In that sense, adaptation limits vary from place to place and are difficult to generalise (Barnett et al. 2015; Dow et al. 2013; Klein et al. 2015). Currently, there is a lack of knowledge on adaptation limits and potential maladaptation to combined effects of climate change and desertification (see Section 4.8.6 for discussion on resilience, thresholds, and irreversible land degradation, also relevant for desertification). However, the potential for residual risks (those risks which remain after adaptation efforts were taken, irrespective of whether they are tolerable or not, tolerability being a subjective concept) and maladaptive outcomes is high (*high confidence*). Some examples of residual risks are illustrated below in this section. Although SLM measures can help lessen the effects of droughts, they cannot fully prevent water stress in crops and resulting lower yields (Eekhout and de Vente 2019). Moreover, although in many cases SLM measures can help reduce and reverse desertification, there would still be short-term losses in land productivity. Irreversible forms of land degradation (for example, loss of topsoil, severe gully erosion) can lead to the complete loss of land productivity. Even when solutions are available, their costs could be prohibitive, presenting the limits to adaptation (Dixon et al. 2013). If warming in dryland areas surpasses human thermal physiological thresholds (Klein et al. 2015; Waha et al. 2013), adaptation could eventually fail (Kamali et al. 2018). Catastrophic shifts in ecosystem functions and services (for example coastal erosion (Chen et al. 2015; Schneider and Kéfi 2016) (Section 4.9.8)) and economic factors can also result in adaptation failure (Evans et al. 2015). Despite the availability of numerous options that contribute to combating desertification, climate change adaptation and mitigation, there are also chances of maladaptive actions (*medium confidence*) (see Glossary). Some activities favouring agricultural intensification in dryland areas can become maladaptive due to their negative impacts on the environment (*medium confidence*). Agricultural expansion to meet food demands can come through deforestation and consequent diminution of carbon sinks (Godfray and Garnett 2014; Stringer et al. 2012). Agricultural insurance programmes encouraging higher agricultural productivity and measures for agricultural intensification can result in detrimental environmental outcomes in some settings (Guodaar et al. 2019; Müller et al. 2017) (Table 6.12). Development of more drought-tolerant crop varieties is considered as a strategy for adaptation to shortening rainy seasons, but this can also lead to a loss of local varieties (Al Hamndou and Requier-Desjardins

2008). Livelihood diversification to collecting and selling firewood and charcoal production can exacerbate deforestation (Antwi-Agyei et al. 2018). Avoiding maladaptive outcomes can often contribute both to reducing the risks from climate change and combating desertification (Antwi-Agyei et al. 2018). Avoiding, reducing and reversing desertification would enhance soil fertility, increase carbon storage in soils and biomass, thus reducing carbon emissions from soils to the atmosphere (Section 3.7.2 and Cross-Chapter Box 2 in Chapter 1). In specific locations, there may be barriers for some of these activities. For example, afforestation and reforestation programmes can contribute to reducing sand storms and increasing carbon sinks in dryland regions (Chu et al. 2019) (Sections 3.6.1 and 3.7.2). However, implementing agroforestry measures in arid locations can be constrained by lack of water (Apuri et al. 2018), leading to a trade-off between soil carbon sequestration and other water uses (Cao et al. 2018). Thus, even when solutions are available, social, economic and institutional constraints could post barriers to their implementation (*medium confidence*).

1.7 Hotspots and case studies

The challenges of desertification and climate change in dryland areas across the world often have very location-specific characteristics. The five case studies in this section present rich experiences and lessons learnt on: (i) soil erosion, (ii) afforestation and reforestation through ‘green walls’, (iii) invasive plant species, (iv) oases in hyper-arid areas, and (v) integrated watershed management. Although it is impossible to cover all hotspots of desertification and on-the-ground actions from all dryland areas, these case studies present a more focused assessment of these five issues, which emerged as salient in the group discussions and several rounds of review of this chapter. The choice of these case studies was also motivated by the desire to capture a wide diversity of dryland settings.

1.7.1 Climate change and soil erosion

1.7.1.1 Soil erosion under changing climate in drylands

Soil erosion is a major form of desertification occurring in varying degrees in all dryland areas across the world (Section 3.2), with negative effects on dryland ecosystems (Section 3.4). Climate change is projected to increase soil erosion potential in some dryland areas through more frequent heavy rainfall events and rainfall variability (see Section 3.5.2 for a more detailed assessment) (Achite and Ouillon 2007; Megnounif and Ghenim 2016; Vachtman et al. 2013; Zhang and Nearing 2005). There are numerous soil conservation measures that can help reduce soil erosion (Section 3.6.1). Such soil management measures include afforestation and reforestation activities, rehabilitation of degraded forests, erosion control measures, prevention of overgrazing, diversification of crop rotations, and improvement in irrigation techniques, especially in sloping areas (Anache et al. 2018; ÇEMGM 2017; Li and Fang 2016; Poesen 2018; Ziadat and Taimeh 2013). Effective measures for soil conservation can also use spatial patterns of plant cover to reduce sediment connectivity, and the relationships between hillslopes and sediment transfer in eroded channels (García-Ruiz et al. 2017). The following three examples present lessons learnt from the soil erosion problems and measures to address them in different settings of Chile, Turkey and the Central Asian countries.

1.7.1.2 No-till practices for reducing soil erosion in central Chile

Soil erosion by water is an important problem in Chile. National assessments conducted in 1979, which examined 46% of the continental surface of the country, concluded that very high levels of soil erosion affected 36% of the territory. The degree of soil erosion increases from south to north. The leading locations in Chile are the region of Coquimbo with 84% of eroded soils (Lat. 29°S, semi-arid climate), the region of Valparaíso with 57% of eroded soils (Lat. 33°S, Mediterranean climate) and the region of O’Higgins with 37% of eroded soils (Lat. 34°S, Mediterranean climate). The most important drivers of soil erosion are soil, slope, climate erosivity (i.e., precipitation, intensity, duration and frequency) due to a highly concentrated rainy season, and vegetation structure and cover. In the region of Coquimbo, goat and sheep overgrazing have aggravated the situation (CIREN 2010). Erosion rates reach up to 100 t ha^{-1} annually, having increased substantially over the last 50 years (Ellies 2000). About 10.4% of central Chile exhibits high erosion rates (greater than 1.1 t ha^{-1} annually) (Bonilla et al. 2010).

Over the last few decades there has been an increasing interest in the development of no-till (also called zero tillage) technologies to minimise soil disturbance, reduce the combustion of fossil fuels and increase soil organic matter. No-till, in conjunction with the adoption of strategic cover crops, has positively impacted soil biology with increases in soil organic matter. Early evaluations by Crovetto, (1998) showed that no-till application (after seven years) had doubled the biological activity indicators compared to traditional farming and even surpassed those found in pasture (grown for the previous 15 years). Besides erosion control, additional benefits are an increase of water-holding capacity and reduction in bulk density. Currently, the above no-till farm experiment has lasted for 40 years and continues to report benefits to soil health and sustainable production (Reicosky and Crovetto 2014). The influence of this iconic farm has resulted in the adoption of soil conservation practices – and especially no-till – in dryland areas of the Mediterranean climate region of central Chile (Martínez et al. 2011). Currently, it has been estimated that the area under no-till farming in Chile varies between 0.13 and 0.2 Mha (Acevedo and Silva 2003).

1.7.1.3 Combating wind erosion and deflation in Turkey: The greening desert of Karapınar

In Turkey, the amount of sediment recently released through erosion into seas was estimated to be 168 Mt yr^{-1}, which is considerably lower than the 500 Mt yr^{-1} that was estimated to be lost in the 1970s. The decrease in erosion rates is attributed to an increase in spatial extent of forests, rehabilitation of degraded forests, erosion control, prevention of overgrazing, and improvement in irrigation technologies. Soil conservation measures conducted in the Karapınar district, Turkey, exemplify these activities. The district is characterised

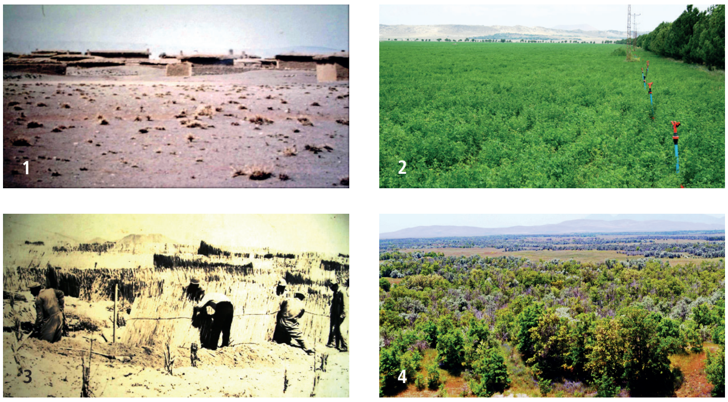

Figure 3.11 | (1) A general view of a nearby village of Karapınar town in the early 1960s (Çarkaci 1999). **(2)** A view of the Karapınar wind erosion area in 2013 (Photo: Murat Türkeş, 17 June 2019). **(3)** Construction of cane screens in the early 1960s in order to decrease wind speed and prevent movement of the sand accumulations and dunes; this was one of the physical measures during the prevention and mitigation period (Çarkaci 1999). **(4)** A view of mixed vegetation, which now covers most of the Karapınar wind erosion area in 2013, the main tree species of which were selected for afforestation with respect to their resistance to the arid continental climate conditions along with a warm/hot temperature regime over the district (Photo: Murat Türkeş, 17 June 2013).

by a semi-arid climate and annual average precipitation of 250–300 mm (Türkeş 2003; Türkeş and Tatlı 2011). In areas where vegetation was overgrazed or inappropriately tilled, the surface soil horizon was removed through erosion processes resulting in the creation of large drifting dunes that threatened settlements around Karapınar (Groneman 1968). Such dune movement had begun to affect the Karapınar settlement in 1956 (Kantarcı et al. 2011). Consequently, by the early 1960s, Karapınar town and nearby villages were confronted with the danger of abandonment due to out-migration in the early 1960s (Figure 3.11(1)). The reasons for increasing wind erosion in the Karapınar district can be summarised as follows: sandy material was mobilised following drying of the lake; hot and semi-arid climate conditions; overgrazing and use of pasture plants for fuel; excessive tillage; and strong prevailing winds.

Restoration and mitigation strategies were initiated in 1959, and today 4300 ha of land have been restored (Akay and Yildirim 2010) (Figure 3.11 (2)), using specific measures: (i) physical measures: construction of cane screens to decrease wind speed and prevent sand movement (Figure 3.11(3)); (ii) restoration of cover: increasing grass cover between screens using seeds collected from local pastures or the cultivation of rye (*Secale* sp.) and wheat grass (*Agropyron elongatum*) that are known to grow in arid and hot conditions; and (iii) afforestation: saplings obtained from nursery gardens were planted and grown between these screens. Main tree species selected were oleaster (*Eleagnus* sp.), acacia (*Robinia pseudeaccacia*), ash (*Fraxinus* sp.), elm (*Ulmus* sp.) and maple (*Acer* sp.) (Figure 3.11 (4)). Economic growth occurred after controlling erosion and new tree nurseries have been established with modern irrigation. Potential negative consequences through the excessive use of water can be mitigated through engagement with local stakeholders and transdisciplinary learning processes, as well as by restoring the traditional land uses in the semi-arid Konya closed basin (Akça et al. 2016).

1.7.1.4 Soil erosion in Central Asia under changing climate

Soil erosion is widely acknowledged to be a major form of degradation of Central Asian drylands, affecting a considerable share of croplands and rangelands. However, up-to-date information on the actual extent of eroded soils at the regional or country level is not available. The estimates compiled by Pender et al. (2009), based on the Central Asian Countries Initiative for Land Management (CACILM), indicate that about 0.8 Mha of the irrigated croplands were subject to high degree of soil erosion in Uzbekistan. In Turkmenistan, soil erosion was indicated to be occurring in about 0.7 Mha of irrigated land. In Kyrgyzstan, out of 1 Mha of irrigated land in the foothill zones, 0.76 Mha were subject to soil erosion by water, leading to losses in crop yields of 20–60% in these eroded soils. About 0.65 Mha of arable land were prone to soil erosion by wind (Mavlyanova et al. 2017). Soil erosion is widespread in rainfed and irrigated areas in Kazakhstan (Saparov 2014). About 5 Mha of rainfed croplands were subject to high levels of soil erosion (Pender et al. 2009). Soil erosion by water was indicated to be a major concern in sloping areas in Tajikistan (Pender et al. 2009).

The major causes of soil erosion in Central Asia are related to human factors, primarily excessive water use in irrigated areas (Gupta et al. 2009), deep ploughing and lack of maintenance of vegetative cover in rainfed areas (Suleimenov et al. 2014), and overgrazing in rangelands (Mirzabaev et al. 2016). Lack of good maintenance of watering infrastructure for migratory livestock grazing, and fragmentation of livestock herds led to overgrazing near villages, increasing the soil erosion by wind (Alimaev et al. 2008). Overgrazing in the rangeland areas of the region (e.g., particularly in Kyzylkum) contributes to dust storms, coming primarily from the Ustyurt Plateau, desertified areas of Amudarya and Syrdarya rivers' deltas, the dried seabed of the Aral Sea (now called Aralkum), and the Caspian Sea (Issanova and Abuduwaili 2017; Xi and Sokolik 2015). Xi and Sokolik (2015) estimated that total dust emissions in Central Asia were 255.6 Mt in 2001, representing 10–17% of the global total.

Central Asia is one of the regions highly exposed to climate change, with warming levels projected to be higher than the global mean (Hoegh-Guldberg et al. 2018), leading to more heat extremes (Reyer et al. 2017). There is no clear trend in precipitation extremes, with some potential for moderate rise in occurrence of droughts. The diminution of glaciers is projected to continue in the Pamir and Tian Shan mountain ranges, a major source of surface waters along with seasonal snowmelt. Glacier melting will increase the hazards from moraine-dammed glacial lakes and spring floods (Reyer et al. 2017). Increased intensity of spring floods creates favourable conditions for higher soil erosion by water, especially in the sloping areas in Kyrgyzstan and Tajikistan. The continuation of some of the current unsustainable cropland and rangeland management practices may lead to elevated rates of soil erosion, particularly in those parts of the region where climate change projections point to increases in floods (Kyrgyzstan, Tajikistan) or increases in droughts (Turkmenistan, Uzbekistan) (Hijioka et al. 2014). Increasing water use to compensate for higher evapotranspiration due to rising temperatures and heat waves could increase soil erosion by water in the irrigated zones, especially in sloping areas and crop fields with uneven land levelling (Bekchanov et al. 2010). The desiccation of the Aral Sea resulted in a hotter and drier regional microclimate, adding to the growing wind erosion in adjacent deltaic areas and deserts (Kust 1999).

There are numerous sustainable land and water management practices available in the region for reducing soil erosion (Abdullaev et al. 2007; Gupta et al. 2009; Kust et al. 2014; Nurbekov et al. 2016). These include: improved land levelling and more efficient irrigation methods such as drip, sprinkler and alternate furrow irrigation (Gupta et al. 2009); conservation agriculture practices, including no-till methods and maintenance of crop residues as mulch in the rainfed and irrigated areas (Kienzler et al. 2012; Pulatov et al. 2012); rotational grazing; institutional arrangements for pooling livestock for long-distance mobile grazing; reconstruction of watering infrastructure along the livestock migratory routes (Han et al. 2016; Mirzabaev et al. 2016); afforesting degraded marginal lands (Djanibekov and Khamzina 2016; Khamzina et al. 2009; Khamzina et al. 2016); integrated water resource management (Dukhovny et al. 2013; Kazbekov et al. 2009); and planting salt – and drought-tolerant halophytic plants as windbreaks in sandy rangelands (Akinshina et al. 2016; Qadir et al. 2009; Toderich et al. 2009; Toderich et al. 2008), and potentially the dried seabed of the former Aral Sea (Breckle 2013). The adoption of enabling policies, such as those discussed in Section 3.6.3, can facilitate the adoption of these sustainable land and water management practices in Central Asia (*high confidence*) (Aw-Hassan et al. 2016; Bekchanov et al. 2016; Bobojonov et al. 2013; Djanibekov et al. 2016; Hamidov et al. 2016; Mirzabaev et al. 2016).

1.7.2 Green walls and green dams

This case study evaluates the experiences of measures and actions implemented to combat soil erosion, decrease dust storms, and to adapt to and mitigate climate change under the Green Wall and Green Dam programmes in East Asia (e.g., China) and Africa (e.g., Algeria, Sahara and the Sahel region). These measures have also been implemented in other countries, such as Mongolia (Do and Kang 2014; Lin et al. 2009), Turkey (Yurtoglu 2015; Çalişkan and Boydak 2017) and Iran (Amiraslani and Dragovich 2011), and are increasingly considered as part of many national and international initiatives to combat desertification (Goffner et al. 2019) (Cross-Chapter Box 2 in Chapter 1). Afforestation and reforestation programmes can contribute to reducing sand storms and increasing carbon sinks in dryland regions (*high confidence*). On the other hand, green wall and green dam programmes also decrease the albedo and hence increase the surface absorption of radiation, increasing the surface temperature. The net effect will largely depend on the balance between these and will vary from place to place depending on many factors.

1.7.2.1 The experiences of combating desertification in China

Arid and semi-arid areas of China, including north-eastern, northern and north-western regions, cover an area of more than 509 Mha, with annual rainfall of below 450 mm. Over the past several centuries, more than 60% of the areas in arid and semi-arid regions were used as pastoral and agricultural lands. The coupled impacts of past climate change and human activity have caused desertification and dust storms to become a serious problem in the region (Xu et al. 2010). In 1958, the Chinese government recognised that desertification and dust storms jeopardised the livelihoods of nearly 200 million people, and afforestation programmes for combating desertification have been initiated since 1978. China is committed to go beyond the Land Degradation Neutrality objective, as indicated by the following programmes that have been implemented. The Chinese Government began the Three North's Forest Shelterbelt programme in Northeast China, North China, and Northwest China, with the goal to combat desertification and to control dust storms by improving forest cover in arid and semi-arid regions. The project is implemented in three stages (1978–2000, 2001–2020 and 2021–2050). In addition, the Chinese government launched the Beijing and Tianjin Sandstorm Source Treatment Project (2001–2010), Returning Farmlands to Forest Project (2003–present), and the Returning Grazing Land to Grassland Project (2003–present) to combat desertification, and for adaptation and mitigation of climate change (State Forestry Administration of China 2015; Wang 2014; Wang et al. 2013).

The results of the fifth monitoring period (2010–2014) showed: (i) compared with 2009, the area of degraded land decreased by 12,120 km^2 over a five-year period; (ii) in 2014, the average coverage of vegetation in the sand area was 18.33%, an increase of 0.7% compared with 17.63% in 2009, and the carbon sequestration increased by 8.5%; (iii) compared with 2009, the amount of wind erosion decreased by 33%, the average annual occurrence of sandstorms decreased by 20.3% in 2014; (iv) as of 2014, 203,700 km^2 of degraded land were effectively managed, accounting for 38.4% of the 530,000 km^2 of manageable desertified land; (v) the restoration of degraded land has created an annual output of 53.63 Mt of fresh and dried fruits, accounting for 33.9% of the total national annual output of fresh and dried fruits (State Forestry Administration of China 2015). This has become an important pillar for economic development and a high priority for peasants as a method to eradicate poverty (State Forestry Administration of China 2015).

Stable investment mechanisms for combating desertification have been established along with tax relief policies and financial support policies for guiding the country in its fight against desertification. The investments in scientific and technological innovation for combating desertification have been improved, the technologies for vegetation restoration under drought conditions have been developed, the popularisation and application of new technologies has been accelerated, and the training of technicians to assist farmers and herdsmen has been strengthened. To improve the monitoring capability and technical level of desertification studies, the monitoring network system has been strengthened, and the popularisation and application of modern technologies have been intensified (e.g., information technology and remote sensing) (Wu et al. 2015). Special laws on combating desertification have been decreed by the government. The provincial government's responsibilities for desertification prevention and controlling objectives and laws have been strictly implemented.

Many studies showed that these projects generally played an active role in combating desertification and fighting against dust storms in China over the past several decades (*high confidence*) (Cao et al. 2018; State Forestry Administration of China 2015; Wang et al. 2013; Wang et al. 2014; Yang et al. 2013). At the beginning of the projects, some problems appeared in some places due to lack of enough knowledge and experience (*low confidence*) (Jiang 2016; Wang et al. 2010). For example, some tree species selected were not well suited to local soil and climatic conditions (Zhu et al. 2007), and there was inadequate consideration of the limitation of the amount of available water on the carrying capacity of trees in some arid regions (Dai 2011; Feng et al. 2016) (Section 3.6.4). In addition, at the beginning of the projects, there was an inadequate consideration of the effects of climate change on combating desertification (Feng et al. 2015; Tan and Li 2015). Indeed, climate change and human activities over past years have influenced the desertification and dust storm control effects in China (Feng et al. 2015; Wang et al. 2009; Tan and Li 2015), and future climate change will bring new challenges for combating desertification in China (Wang et al. 2017; Yin et al. 2015; Xu et al. 2019). In particular, the desertification risk in China will be enhanced at 2°C compared to 1.5°C global temperature rise (Ma et al. 2018). Adapting desertification control to climate change involves: improving the adaptation capacity to climate change for afforestation and grassland management by executing SLM practices; optimising the agricultural and animal husbandry structure; and using big data to meet the water resources regulation (Zhang and Huisingh 2018). In particular, improving scientific and technological supports in desertification control is crucial for adaptation to climate change and combating desertification, including protecting vegetation in desertification-prone lands by planting indigenous plant species, facilitating natural restoration of vegetation to conserve biodiversity, employing artificial rain or snow, water-saving irrigation and water storage technologies (Jin et al. 2014; Yang et al. 2013).

1.7.2.2 The Green Dam in Algeria

After independence in 1962, the Algerian government initiated measures to replant forests destroyed by the war, and the steppes affected by desertification, among its top priorities (Belaaz 2003). In 1972, the government invested in the Green Dam (*Barrage vert*) project. This was the first significant experiment to combat desertification, influence the local climate and decrease the aridity by restoring a barrier of trees. The Green Dam extends across arid and semi-arid zones between the isohyets 300 mm and 200 mm. It is a 3 Mha band of plantation running from east to west (Figure 3.12). It is over 1200 km long (from the Algerian–Moroccan border to the Algerian–Tunisian border) and has an average width of about 20 km. The soils in the area are shallow, low in organic matter and susceptible to erosion. The main objectives of the project were to conserve natural resources, improve the living conditions of local residents and avoid their exodus to urban areas. During the first four decades (1970–2000) the success rate was low (42%) due to lack of participation by the local population and the choice of species (Bensaid 1995).

The Green Dam did not have the desired effects. Despite tree-planting efforts, desertification intensified on the steppes, especially in south-western Algeria, due to the prolonged drought during the 1980s. Rainfall declined in the range from 18% to 27%, and the dry season has increased by two months in the last century (Belala et al. 2018). Livestock numbers in the Green Dam regions, mainly sheep, grew exponentially, leading to severe overgrazing, causing trampling and soil compaction, which greatly increased the risk of erosion. Wind erosion, very prevalent in the region, is due to climatic conditions and the strong anthropogenic action that reduced the vegetation cover. The action of the wind carries fine particles such as sands and clays and leaves on the soil surface a lag-gravel pavement, which is unproductive. Water erosion is largely due to torrential rains in the form of severe thunderstorms that disintegrate the bare soil surface from raindrop impact (Achite et al. 2016). The detached soil and nutrients are transported offsite via runoff, resulting in loss of fertility and water holding capacity. The risk of and severity of water erosion is a function of human land-use activities that increase soil loss through removal of vegetative cover. The National Soil Sensitivity to Erosion Map (Salamani et al. 2012) shows that more than 3 Mha of land in the steppe provinces are currently experiencing intense wind activity (Houyou et al. 2016) and that these areas are at particular risk of soil erosion. Mostephaoui et al. (2013), estimates that each year there is a loss of 7 t ha^{-1} of soils due to erosion. Nearly 0.6 Mha of land in the steppe zone are fully degraded without the possibility of biological recovery.

To combat the effects of erosion and desertification, the government has planned to relaunch the rehabilitation of the Green Dam by incorporating new concepts related to sustainable development, and adaptation to climate change. The experience of previous years has led to integrated rangeland management, improved tree and fodder shrub plantations and the development of water conservation techniques. Reforestation is carried out using several species, including fruit trees, to increase and diversify the sources of income for the population.

The evaluation of the Green Dam from 1972 to 2015 (Merdas et al. 2015) shows that 0.3 Mha of forest plantation have been planted, which represents 10% of the project area. Estimates of the success rate of reforestation vary considerably between 30% and 75%,

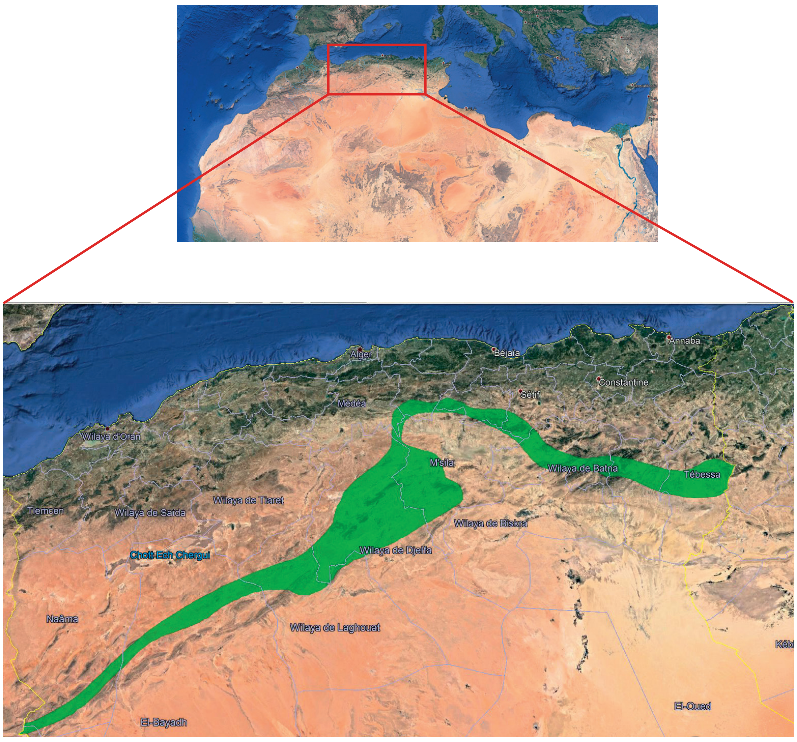

Figure 3.12 | Location of the Green Dam in Algeria (Saifi et al. 2015). Note: The green coloured band represents the location of the Green Dam.

depending on the region. Through demonstration, the Green Dam has inspired several African nations to work together to build a Great Green Wall to combat land degradation, mitigate climate change effects, loss of biodiversity and poverty in a region that stretches from Senegal to Djibouti (Sahara and Sahel Observatory (OSS) 2016) (Section 3.7.2.3).

1.7.2.3 The Great Green Wall of the Sahara and the Sahel Initiative

The Great Green Wall is an initiative of the Heads of State and Government of the Sahelo-Saharan countries to mitigate and adapt to climate change, and to improve the food security of the Sahel and Saharan peoples (Sacande 2018; Mbow 2017). Launched in 2007, this regional project aims to restore Africa's degraded arid landscapes, reduce the loss of biodiversity and support local communities to sustainable use of forests and rangelands. The Great Green Wall focuses on establishing plantations and neighbouring projects, covering a distance of 7775 km from Senegal on the Atlantic coast to Eritrea on the Red Sea coast, with a width of 15 km (Figure 3.13). The wall passes through Djibouti, Eritrea, Ethiopia, Sudan, Chad, Niger, Nigeria, Mali, Burkina Faso, Mauritania and Senegal.

The choice of woody and herbaceous species that will be used to restore degraded ecosystems is based on biophysical and socio-economic criteria, including socio-economic value (food, pastoral, commercial, energetic, medicinal, cultural); ecological importance (carbon sequestration, soil cover, water infiltration); and resilience to climate change and variability. The Pan-African Agency of the Great Green Wall (PAGGW) was created in 2010 under the auspices of the African Union and CEN-SAD to manage the project. The initiative is implemented at the level of each country by a national structure. A monitoring and evaluation system has been defined, allowing nations to measure outcomes and to propose the necessary adjustments.

In the past, reforestation programmes in the arid regions of the Sahel and North Africa that have been undertaken to stop desertification were poorly studied and cost a lot of money without significant success (Benjaminsen and Hiernaux 2019). Today, countries have changed their strategies and opted for rural development projects that can be more easily funded. Examples of scalable practices for land restoration include managing water bodies for livestock and crop production, and promoting fodder trees to reduce runoff (Mbow 2017).

The implementation of the initiative has already started in several countries. For example, the FAO's Action Against Desertification project was restoring 18,000 hectares of land in 2018 through planting native tree species in Burkina Faso, Ethiopia, The Gambia, Niger, Nigeria and Senegal (Sacande 2018). Berrahmouni et al. (2016) estimated that 166 Mha can be restored in the Sahel, requiring the restoration of 10 Mha per year to achieve Land Degradation Neutrality targets by 2030. Despite these early implementation actions on the ground, the achievement of the planned targets is questionable, and will be challenging without significant additional funding.

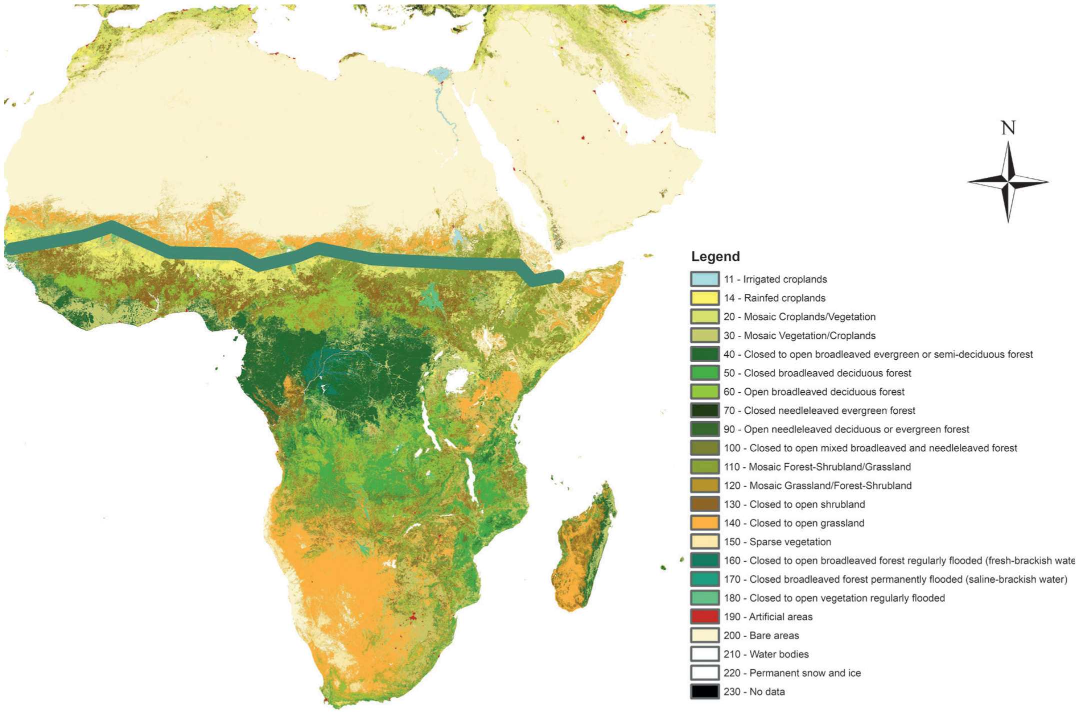

Figure 3.13 | The Great Green Wall of the Sahara and the Sahel. Source for the data layer: This dataset is an extract from the GlobCover 2009 land cover map, covering Africa and the Arabian Peninsula. The GlobCover 2009 land cover map is derived by an automatic and regionally tuned classification of a time series of global MERIS (MEdium Resolution Imaging Spectrometer) FR mosaics for the year 2009. The global land cover map counts 22 land cover classes defined with the United Nations (UN) Land Cover Classification System (LCCS).

1.7.3 Invasive plant species

1.7.3.1 Introduction

The spread of invasive plants can be exacerbated by climate change (Bradley et al. 2010; Davis et al. 2000). In general, it is expected that the distribution of invasive plant species with high tolerance to drought or high temperatures may increase under most climate change scenarios (*medium* to *high confidence*) (Bradley et al. 2010; Settele et al. 2014; Scasta et al. 2015). Invasive plants are considered a major risk to native biodiversity and can disturb the nutrient dynamics and water balance in affected ecosystems (Ehrenfeld 2003). Compared to more humid regions, the number of species that succeed in invading dryland areas is low (Bradley et al. 2012), yet they have a considerable impact on biodiversity and ecosystem services (Le Maitre et al. 2015, 2011; Newton et al. 2011). Moreover, human activities in dryland areas are responsible for creating new invasion opportunities (Safriel et al. 2005).

Current drivers of species introductions include expanding global trade and travel, land degradation and changes in climate (Chytrý et al. 2012; Richardson et al. 2011; Seebens et al. 2018). For example, Davis et al. (2000) suggests that high rainfall variability promotes the success of alien plant species – as reported for semi-arid grasslands and Mediterranean-type ecosystems (Cassidy et al. 2004; Reynolds et al. 2004; Sala et al. 2006). Furthermore, Panda et al. (2018) demonstrated that many invasive species could withstand elevated temperature and moisture scarcity caused by climate change. Dukes et al. (2011) observed that the invasive plant yellow-star thistle (*Centaurea solstitialis)* grew six time larger under the elevated atmospheric CO_2 expected in future climate change scenarios.

Climate change is *likely* to aggravate the problem as existing species continue to spread unabated and other species develop invasive characteristics (Hellmann et al. 2008). Although the effects of climate change on invasive species distributions have been relatively well explored, the greater impact on ecosystems is less well understood (Bradley et al. 2010; Eldridge et al. 2011).

Due to the time lag between the initial release of invasive species and their impact, the consequence of invasions is not immediately detected and may only be noticed centuries after introduction (Rouget et al. 2016). Climate change and invading species may act in concert (Bellard et al. 2013; Hellmann et al. 2008; Seebens et al. 2015). For example, invasion often changes the size and structure of fuel loads, which can lead to an increase in the frequency and intensity of fire (Evans et al. 2015). In areas where the climate is becoming warmer, an increase in the likelihood of suitable weather conditions for fire may promote invasive species, which in turn may lead to further desertification. Conversely, fire may promote plant invasions via several mechanisms (by reducing cover of competing vegetation, destroying native vegetation and clearing a path for invasive plants or creating favourable soil conditions) (Brooks et al. 2004; Grace et al. 2001; Keeley and Brennan 2012).

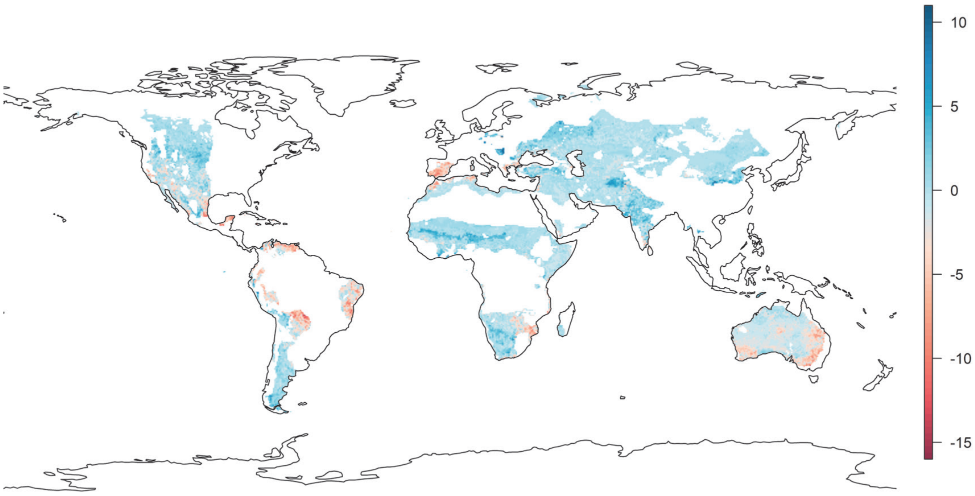

Figure 3.14 | Difference between the number of invasive alien species (n=99, from Bellard et al. (2013)) predicted to occur by 2050 (under A1B scenario) and current period '2000' within the dryland areas.

3

At a regional scale, Bellard et al. (2013) predicted increasing risk in Africa and Asia, with declining risk in Australia (Figure 3.14). This projection does not represent an exhaustive list of invasive alien species occurring in drylands.

A set of four case studies in Ethiopia, Mexico, the USA and Pakistan is presented below to describe the nuanced nature of invading plant species, their impact on drylands and their relationship with climate change.

1.7.3.2 Ethiopia

The two invasive plants that inflict the heaviest damage to ecosystems, especially biodiversity, are the annual herbaceous weed, *Parthenium hysterophorus* (*Asteraceae*) also known as Congress weed; and the tree species, *Prosopis juliflora* (*Fabaceae*) also called Mesquite, both originating from the southwestern United States to Central/South America (Adkins and Shabbir 2014). *Prosopis* was introduced in the 1970s and has since spread rapidly. *Prosopis,* classified as the highest priority invader in Ethiopia, is threatening livestock production and challenging the sustainability of the pastoral systems. *Parthenium* is believed to have been introduced along with relief aid during the debilitating droughts of the early 1980s, and a recent study reported that it has spread into 32 out of 34 districts in Tigray, the northernmost region of Ethiopia (Teka 2016). A study by Etana et al. (2011) indicated that Parthenium caused a 69% decline in the density of herbaceous species in Awash National Park within a few years of introduction. In the presence of Parthenium, the growth and development of crops is suppressed due to its allelopathic properties. McConnachie et al. (2011) estimated a 28% crop loss across the country, including a 40–90% reduction in sorghum yield in eastern Ethiopia alone (Tamado et al. 2002). The weed is a substantial agricultural and natural resource problem and constitutes a significant health hazard (Fasil 2011). Parthenium causes acute allergic respiratory problems, skin dermatitis, and reportedly mutagenicity both in humans and livestock (Mekonnen 2017; Patel 2011). The eastern belt of Africa – including Ethiopia – presents a very suitable habitat, and the weed is expected to spread further in the region in the future (Mainali et al. 2015).

There is neither a comprehensive intervention plan nor a clear institutional mandate to deal with invasive weeds, however, there are fragmented efforts involving local communities even though they are clearly inadequate. The lessons learned, related to actions that have contributed to the current scenario, are several. First, lack of coordination and awareness – mesquite was introduced by development agencies as a drought-tolerant shade tree with little consideration of its invasive nature. If research and development institutions had been aware, a containment strategy could have been implemented early on. The second major lesson is the cost of inaction. When research and development organisations did sound the alarm, the warnings went largely unheeded, resulting in the spread and buildup of two of the worst invasive plant species in the world (Fasil 2011).

1.7.3.3 Mexico

Buffelgrass (*Cenchrus ciliaris* L.), a native species from southern Asia and East Africa, was introduced into Texas and northern Mexico in the 1930s and 1940s, as it is highly productive in drought conditions (Cox et al. 1988; Rao et al. 1996). In the Sonoran desert of Mexico, the distribution of buffelgrass has increased exponentially, covering 1 Mha in Sonora State (Castellanos-Villegas et al. 2002). Furthermore, its potential distribution extended to 53% of Sonora State and 12% of semi-arid and arid ecosystems in Mexico (Arriaga et al. 2004). Buffelgrass has also been reported as an aggressive invader in Australia and the USA, resulting in altered fire cycles that enhance further spread of this plant and disrupt ecosystem processes (Marshall et al. 2012; Miller et al. 2010; Schlesinger et al. 2013).

Castellanos et al. (2016) reported that soil moisture was lower in the buffelgrass savannah cleared 35 years ago than in the native semi-arid shrubland, mainly during the summer. The ecohydrological changes induced by buffelgrass can therefore displace native plant species over the long term. Invasion by buffelgrass can also affect landscape productivity, as it is not as productive as native vegetation (Franklin and Molina-Freaner 2010). Incorporation of buffelgrass is considered a good management practice by producers and the government. For this reason, no remedial actions are undertaken.

1.7.3.4 United States of America

Sagebrush ecosystems have declined from 25 Mha to 13 Mha since the late 1800s (Miller et al. 2011). A major cause is the introduction of non-native cheatgrass (*Bromus tectorum*), which is the most prolific invasive plant in the USA. Cheatgrass infests more than 10 Mha in the Great Basin and is expanding every year (Balch et al. 2013). It provides a fine-textured fuel that increases the intensity, frequency and spatial extent of fire (Balch et al. 2013). Historically, wildfire frequency was 60 to 110 years in Wyoming big sagebrush communities and has increased to five years following the introduction of cheatgrass (Balch et al. 2013; Pilliod et al. 2017).

The conversion of the sagebrush steppe biome to annual grassland with higher fire frequencies has severely impacted livestock producers, as grazing is not possible for a minimum of two years after fire. Furthermore, cheatgrass and wildfires reduce critical habitat for wildlife and negatively impact species richness and abundance – for example, the greater sage-grouse (*Centocercus urophasianus*) and pygmy rabbit (*Brachylagus idahoensis*) which are on the verge of being listed for federal protection (Crawford et al. 2004; Larrucea and Brussard 2008; Lockyer et al. 2015).

Attempts to reduce cheatgrass impacts through reseeding of both native and adapted introduced species have occurred for more than 60 years (Hull and Stewart 1949) with little success. Following fire, cheatgrass becomes dominant and recovery of native shrubs and grasses is improbable, particularly in relatively low-elevation sites with minimal annual precipitation (less than 200 mm yr^{-1}) (Davies et al. 2012; Taylor et al. 2014). Current rehabilitation efforts emphasise the use of native and non-native perennial grasses, forbs and shrubs (Bureau of Land Management 2005). Recent literature suggests that these treatments are not consistently effective at displacing cheatgrass populations or re-establishing sage-grouse habitat, with success varying with elevation and precipitation (Arkle et al. 2014; Knutson et al. 2014). Proper post-fire grazing rest, season-of-use, stocking rates, and subsequent management are essential to restore resilient sagebrush ecosystems before they cross a threshold and become an annual grassland (Chambers et al. 2014; Miller et al. 2011; Pellant et al. 2004). Biological soil crust protection may be an effective measure to reduce cheatgrass germination, as biocrust disturbance has been shown to be a key factor promoting germination of non-native grasses (Hernandez and Sandquist 2011). Projections of increasing temperature (Abatzoglou and Kolden 2011), and observed reductions in and earlier melting of snowpack in the Great Basin region (Harpold and Brooks 2018; Mote et al. 2005) suggest that there is a need to understand current and past climatic variability as this will drive wildfire variability and invasions of annual grasses.

1.7.3.5 Pakistan

The alien plants invading local vegetation in Pakistan include *Brossentia papyrifera* (found in Islamabad Capital territory), *Parthenium hysterophorus* (found in Punjab and Khyber Pakhtunkhwa provinces), *Prosopis juliflora* (found all over Pakistan), *Eucalyptus camaldulensis* (found in Punjab and Sindh provinces), *Salvinia* (aquatic plant widely distributed in water bodies in Sindh), *Cannabis sativa* (found in Islamabad Capital Territory), *Lantana camara* and *Xanthium strumarium* (found in upper Punjab and Khyber Pakhtunkhwa provinces) (Khan et al. 2010; Qureshi et al. 2014). Most of these plants were introduced by the Forest Department decades ago for filling the gap between demand and supply of timber, fuelwood and fodder. These non-native plants have some uses but their disadvantages outweigh their benefits (Marwat et al. 2010; Rashid et al. 2014).

Besides being a source of biological pollution and a threat to biodiversity and habitat loss, the alien plants reduce the land value and cause huge losses to agricultural communities (Rashid et al. 2014). *Brossentia papyrifera,* commonly known as Paper Mulberry, is the root cause of inhalant pollen allergy for the residents of lush green Islamabad during spring. From February to April, the pollen allergy is at its peak, with symptoms of severe persistent coughing, difficulty in breathing, and wheezing. The pollen count, although variable at different times and days, can be as high as 55,000 m^{-3}. Early symptoms of the allergy include sneezing, itching in the eyes and skin, and blocked nose. With changing climate, the onset of disease is getting earlier, and pollen count is estimated to cross 55,000 m^{-3} (Rashid et al. 2014). About 45% of allergic patients in the twin cities of Islamabad and Rawalpindi showed positive sensitivity to the pollens (Marwat et al. 2010). Millions of rupees have been spent by the Capital Development Authority on pruning and cutting of Paper Mulberry trees but because of its regeneration capacity growth is regained rapidly (Rashid et al. 2014). Among other invading plants, *Prosopis juliflora* has allelopathic properties, and *Eucalyptus* is known to transpire huge amounts of water and deplete the soil of its nutrient elements (Qureshi et al. 2014).

Although a Biodiversity Action Plan exists in Pakistan, it is not implemented in letter or spirit. The Quarantine Department focuses only on pests and pathogens but takes no notice of plant and animal species being imported. Also, there is no provision for checking the possible impacts of imported species on the environment (Rashid et al. 2014) or for carrying out bioassays of active allelopathic compounds of alien plants.

1.7.4 Oases in hyper-arid areas in the Arabian Peninsula and northern Africa

Oases are isolated areas with reliable water supply from lakes and springs, located in hyper-arid and arid zones (Figure 3.15). Oasis agriculture has long been the only viable crop production system throughout the hot and arid regions of the Arabian Peninsula and North Africa. Oases in hyper-arid climates are usually subject to water shortage as evapotranspiration exceeds rainfall. This often causes salinisation of soils. While many oases have persisted for several thousand years, many others have been abandoned, often in response to changes in climate or hydrologic conditions (Jones et al. 2019), providing testimony to societies' vulnerability to climatic shifts and raising concerns about similarly severe effects of anthropogenic climate change (Jones et al. 2019).

On the Arabian Peninsula and in North Africa, climate change is projected to have substantial and complex effects on oasis areas (Abatzoglou and Kolden 2011; Ashkenazy et al. 2012; Bachelet et al. 2016; Guan et al. 2018; Iknayan and Beissinger 2018; Ling et al. 2013). To illustrate, by the 2050s, the oases in southern Tunisia are expected to be affected by hydrological and thermal changes, with an average temperature increase of 2.7°C, a 29% decrease in precipitation and a 14% increase in evapotranspiration rate (Ministry of Agriculture and Water Resources of Tunisia and GIZ 2007). In Morocco, declining aquifer recharge is expected to impact the water supply of the Figuig oasis (Jilali 2014), as well as for the Draa Valley (Karmaoui et al. 2016). Saudi Arabia is expected to experience a 1.8°C–4.1°C increase in temperatures by 2050, which is forecast to raise agricultural water demand by 5–15% in order to maintain production levels equal to those of 2011 (Chowdhury and Al-Zahrani 2013). The increase of temperatures and variable pattern of rainfall over the central, north and south-western regions of Saudi Arabia may pose challenges for sustainable water resource management (Tarawneh and Chowdhury 2018). Moreover, future climate scenarios are expected to increase the frequency of floods and flash floods, such as in the coastal areas along the central parts of the Red Sea and the south-southwestern areas of Saudi Arabia (Almazroui et al. 2017).

While many oases are cultivated with very heat-tolerant crops such as date palms, even such crops eventually have declines in their productivity when temperatures exceed certain thresholds or hot conditions prevail for extended periods. Projections so far do not indicate severe losses in land suitability for date palm for the Arabian Peninsula (Aldababseh et al. 2018; Shabani et al. 2015). It is unclear, however, how reliable the climate response parameters in the underlying models are, and actual responses may differ substantially.

Figure 3.15 | Oases across the Arabian Peninsula and North Africa (alphabetically by country). **(a)** Masayrat ar Ruwajah oasis, Ad Dakhiliyah Governorate, Oman (Photo: Eike Lüdeling). **(b)** Tasselmanet oasis, Ouarzazate Province, Morocco (Photo: Abdellatif Khattabi). **(c)** Al-Ahsa oasis, Al-Ahsa Governarate, Saudi Arabia (Photo: Shijan Kaakkara). **(d)** Zarat oasis, Governorate of Gabes, Tunisia (Photo: Hamda Aloui). The use rights for (a), (b) and (d) were granted by copyright holders; (c) is licensed under the Creative Commons Attribution 2.0 Generic license.

Date palms are routinely assumed to be able to endure very high temperatures, but recent transcriptomic and metabolomic evidence suggests that heat stress reactions already occur at 35°C (Safronov et al. 2017), which is not exceptionally warm for many oases in the region. Given current assumptions about the heat-tolerance of date palm, however, adverse effects are expected to be small (Aldababseh et al. 2018; Shabani et al. 2015). For some other perennial oasis crops, impacts of temperature increases are already apparent. Between 2004/2005 and 2012/2013, high-mountain oases of Al Jabal Al Akhdar in Oman lost almost all fruit and nut trees of temperate-zone origin, with the abundance of peaches, apricots, grapes, figs, pears, apples, and plums dropping by between 86% and 100% (Al-Kalbani et al. 2016). This implies that that the local climate may not remain suitable for species that depend on cool winters to break their dormancy period (Luedeling et al. 2009). A similar impact is very probable in Tunisia and Morocco, as well as in other oasis locations in the Arabian Peninsula and North Africa (Benmoussa et al. 2007). All these studies expect strong decreases in winter chill, raising concerns that many currently well-established species will no longer be viable in locations where they are grown today. The risk of detrimental chill shortfalls is expected to increase gradually, slowly diminishing the economic prospects to produce such species. Without adequate adaptation actions, the consequences of this development for many traditional oasis settlements and other plantations of similar species could be highly negative.

At the same time, population growth and agricultural expansion in many oasis settlements are leading to substantial increases in water demand for human consumption (Al-Kalbani et al. 2014). For example, a large unmet water demand has been projected for future scenarios in the valley of Seybouse in East Algeria (Aoun-Sebaiti et al. 2014), and similar conclusions were drawn for Wadi El Natrun in Egypt (Switzman et al. 2018). Modelling studies have indicated long-term decline in available water and increasing risk of water shortages – for example, for oases in Morocco (Johannsen et al. 2016; Karmaoui et al. 2016), the Dakhla oasis in Egypt's Western Desert (Sefelnasr et al. 2014) and for the large Upper Mega Aquifer of the Arabian Peninsula (Siebert et al. 2016). Mainly due to the risk of water shortages, Souissi et al. (2018) classified almost half of all farmers in Tunisia as non-resilient to climate change, especially those relying on tree crops, which limit opportunities for short-term adaptation actions.

The maintenance of the oasis systems and the safeguarding of their population's livelihoods are currently threatened by continuous water degradation, increasing soil salinisation, and soil contamination (Besser et al. 2017). Waterlogging and salinisation of soils due to rising saline groundwater tables coupled with inefficient drainage systems have become common to all continental oases in Tunisia, most of which are concentrated around saline depressions, known locally as *chotts* (Ben Hassine et al. 2013). Similar processes of salinisation are also occurring in the oasis areas of Egypt due to agricultural expansion, excessive use of water for irrigation and deficiency of the drainage systems (Abo-Ragab 2010; Masoud and Koike 2006). A prime example for this is Siwa oasis (Figure 3.16), a depression extending over 1050 km^2 in the north-western desert of Egypt in the north of the sand dune belt of the Great Sand Sea (Abo-Ragab and Zaghloul 2017). Siwa oasis has been recognised as a Globally Important Agricultural Heritage Site (GIAHS) by the FAO for being an *in situ* repository of plant genetic resources, especially of uniquely adapted varieties of date palm, olive and secondary crops that are highly esteemed for their quality and continue to play a significant role in rural livelihoods and diets (FAO 2016).

The population growth in Siwa is leading rapid agricultural expansion and land reclamation. The Siwan farmers are converting the surrounding desert into reclaimed land by applying their old inherited traditional practices. Yet, agricultural expansion in the oasis mainly depends on non-renewable groundwaters. Soil salinisation and vegetation loss have been accelerating since 2000 due to water mismanagement and improper drainage systems (Masoud and Koike 2006). Between 1990 and 2008, the cultivated area increased from 53 to 88 km^2, lakes from 60 to 76 km^2, *sabkhas* (salt flats) from 335 to 470 km^2, and the urban area from 6 to 10 km^2 (Abo-Ragab 2010). The problem of rising groundwater tables was exacerbated by climatic changes (Askri et al. 2010; Gad and Abdel-Baki 2002; Marlet et al. 2009).

Water supply is *likely* to become even scarcer for oasis agriculture under changing climate in the future than it is today, and viable solutions are difficult to find. While some authors stress the possibility to use desalinated water for irrigation (Aldababseh et al. 2018), the economics of such options, especially given the high evapotranspiration rates in the Arabian Peninsula and North Africa, are debatable. Many oases are located far from water sources that are suitable for desalination, adding further to feasibility constraints. Most authors therefore stress the need to limit water use (Sefelnasr et al. 2014), for example, by raising irrigation efficiency (Switzman et al. 2018), reducing agricultural areas (Johannsen et al. 2016) or imposing water use restrictions (Odhiambo 2017), and to carefully monitor desertification (King and Thomas 2014). Whether adoption of crops with low water demand, such as sorghum (*Sorghum bicolor* (L.) Moench) or jojoba (*Simmondsia chinensis* (Link) C. K. Schneid.) (Aldababseh et al. 2018), can be a viable option for some oases remains to be seen, but given their relatively low profit margins compared to currently grown oasis crops, there are reasons to doubt the economic feasibility of such proposals. While it is currently unclear to what extent oasis agriculture can be maintained in hot locations of the region, cooler sites offer potential for shifting towards new species and cultivars, especially for tree crops, which have particular climatic needs across seasons. Resilient options can be identified, but procedures to match tree species and cultivars with site climate need to be improved to facilitate effective adaptation.

There is *high confidence* that many oases of North Africa and the Arabian Peninsula are vulnerable to climate change. While the impacts of recent climate change are difficult to separate from the consequences of other change processes, it is *likely* that water resources have already declined in many places and the suitability of the local climate for many crops, especially perennial crops, has already decreased. This decline of water resources and thermal suitability of oasis locations for traditional crops is *very likely* to continue throughout the 21st century. In the coming years, the people living in oasis regions across the world will face challenges due to increasing impacts of global environmental change (Chen et al. 2018). Hence, efforts to increase their adaptive capacity to

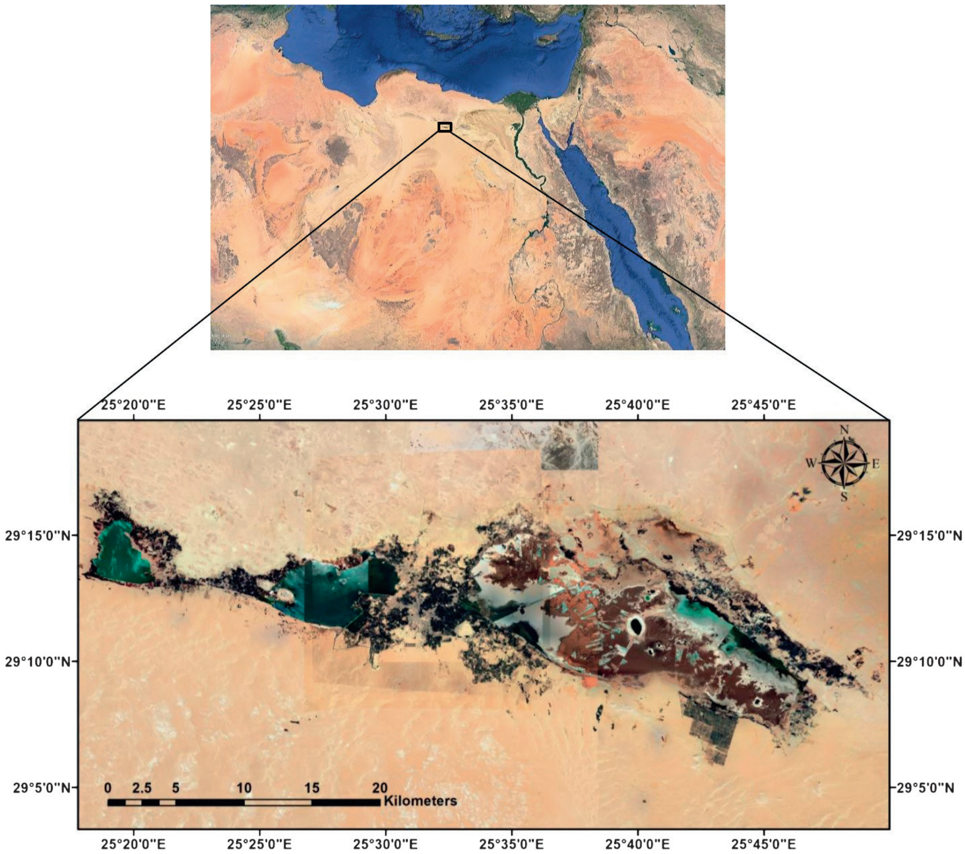

Figure 3.16 | Satellite image of the Siwa Oasis, Egypt. Source: Google Maps.

climate change can facilitate the sustainable development of oasis regions globally. In particular this wil mean addressing the trade-offs between environmental restoration and agricultural livelihoods (Chen et al. 2018). Ultimately, sustainability in oasis regions will depend on policies integrating the provision of ecosystem services and social and human welfare needs (Wang et al. 2017).

1.7.5 Integrated watershed management

Desertification has resulted in significant loss of ecosystem processes and services, as described in detail in this chapter. The techniques and processes to restore degraded watersheds are not linear and integrated watershed management (IWM) must address physical, biological and social approaches to achieve SLM objectives (German et al. 2007).

1.7.5.1 Jordan

Population growth, migration into Jordan and changes in climate have resulted in desertification of the Jordan Badia region. The Badia region covers more than 80% of the country's area and receives less than 200 mm of rainfall per year, with some areas receiving less than 100 mm (Al-Tabini et al. 2012). Climate analysis has indicated a generally increasing dryness over the West Asia and Middle East region (AlSarmi and Washington 2011; Tanarhte et al. 2015), with reduction in average annual rainfall in Jordan's Badia area (De Pauw et al. 2015). The incidence of extreme rainfall events has not declined over the region. Locally increased incidence of extreme events over the Mediterranean region has been proposed (Giannakopoulos et al. 2009).

The practice of intensive and localised livestock herding, in combination with deep ploughing and unproductive barley agriculture, are the main drivers of severe land degradation and depletion of the rangeland natural resources. This affected both the quantity and the diversity of vegetation as native plants with a high nutrition value were replaced with invasive species with low palatability and nutritional content (Abu-Zanat et al. 2004). The sparsely covered and crusted soils in Jordan's Badia area have a low rainfall interception and infiltration rate, which leads to increased surface runoff and subsequent erosion and gullying, speeding up

Figure 3.17 | (a) Newly prepared micro water harvesting catchment, using the Vallerani system. **(b)** Aerial imaging showing micro water harvesting catchment treatment after planting **(c)** one year after treatment. Source: Stefan Strohmeier.

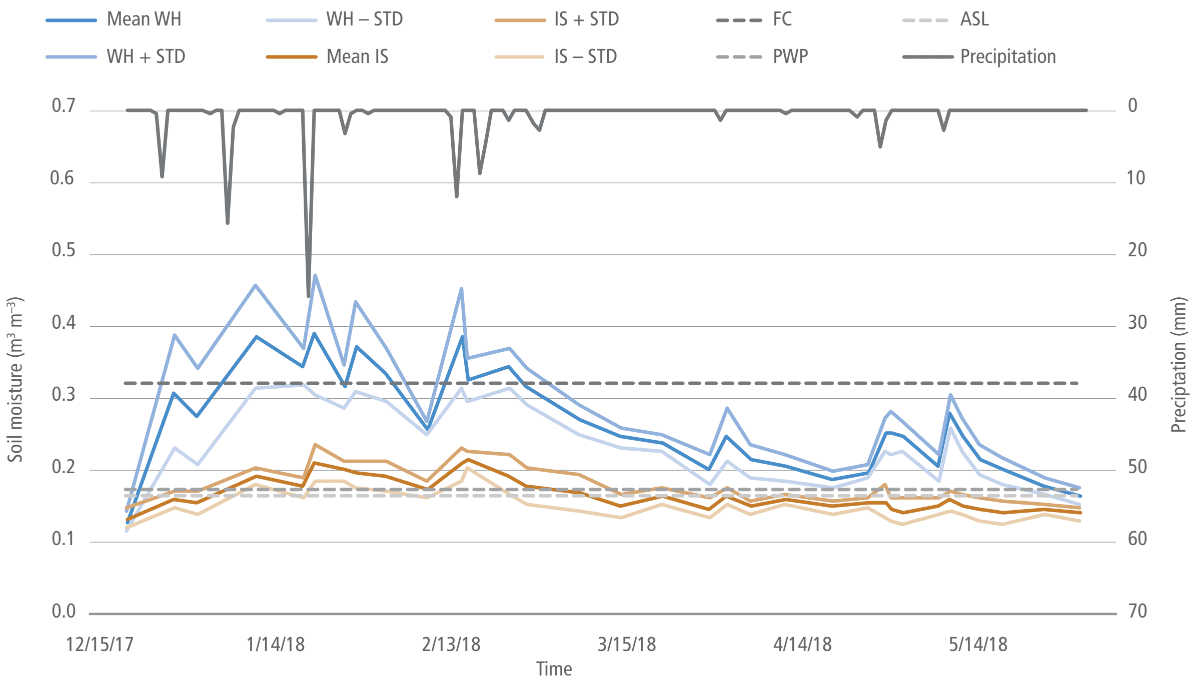

Figure 3.18 | Illustration of enhanced soil water retention in the Mechanized Micro Rainwater Harvesting compared to untreated Badia rangelands in Jordan, showing precipitation (PCP), sustained stress level resulting in decreased production, field capacity and wilting point for available soil moisture, and then measured soil moisture content between the two treatments (degraded rangeland and the restored rangeland with the Vallerani plough).

the drainage of rainwater from the watersheds, which can result in downstream flooding in Amman, Jordan (Oweis 2017).

To restore the desertified Badia an IWM plan was developed using hillslope-implemented water harvesting micro catchments as a targeted restoration approach (Tabieh et al. 2015). Mechanized Micro Rainwater Harvesting (MIRWH) technology using the 'Vallerani plough' (Antinori and Vallerani 1994; Gammoh and Oweis 2011; Ngigi 2003) is being widely applied for rehabilitation of highly degraded rangeland areas in Jordan. A tractor digs out small water harvesting pits on the contour of the slope (Figure 3.17) allowing the retention, infiltration and local storage of surface runoff in the soil (Oweis 2017). The micro catchments are planted with native shrub seedlings, such as saltbush (*Atriplex halimus*), with enhanced survival as a function of increased soil moisture (Figure 3.18) and increased dry matter yields (>300 kg ha^{-1}) that can serve as forage for livestock (Oweis 2017; Tabieh et al. 2015).

Simultaneously to MIRWH upland measures, the gully erosion is being treated through intermittent stone plug intervention (Figure 3.19), stabilising the gully beds, increasing soil moisture in proximity of the plugs, dissipating the surface runoff's energy, and mitigating further back-cutting erosion and quick drainage of water. Eventually, the treated gully areas silt up and dense vegetation cover can re-establish. In addition, grazing management practices are implemented to increase the longevity of the treatment. Ultimately, the recruitment processes and re-vegetation shall control the watershed's hydrological regime through rainfall interception, surface runoff deceleration and filtration, combined with the less erodible and enhanced infiltration characteristics of the rehabilitated soils.

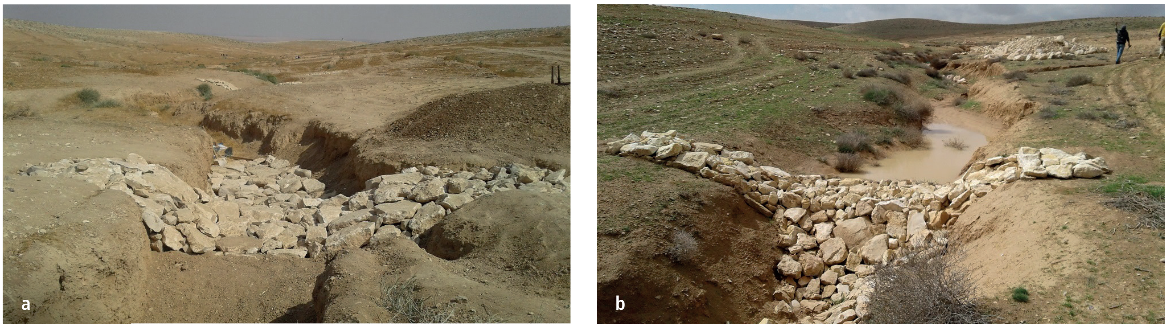

Figure 3.19 | (a) Gully plug development in September 2017. (b) Post-rainfall event (March 2018). Near Amman, Jordan. Source: Stefan Strohmeier.

In-depth understanding of the Badia's rangeland status transition, coupled with sustainable rangeland management, are still subject to further investigation, development and adoption; a combination of all three is required to mitigate the ongoing degradation of the Middle Eastern rangeland ecosystems.

Oweis (2017) indicated that the cost of the fully automated Vallerani technique was approximately 32 USD ha^{-1}. The total cost of the restoration package included the production, planting and maintenance of the shrub seedlings (11 USD ha^{-1}). Tabieh et al. (2015) calculated a benefit-cost ratio (BCR) of above 1.5 for re-vegetation of degraded Badia areas through MIRWH and saltbush. However, costs vary based on the seedling's costs and availability of trained labour.

Water harvesting is not a recent scientific advancement. Water harvesting is known to have been developed during the Bronze Age and was widely practiced in the Negev Desert during the Byzantine time period (1300–1600 years ago) (Fried et al. 2018; Stavi et al. 2017). Through construction of various structures made of packed clay and stone, water was either held on site in half-circular dam structures (*hafir*) that faced up-slope to capture runoff, or on terraces that slowed water allowing it to infiltrate and to be stored in the soil profile. Numerous other systems were designed to capture water in below-ground cisterns to be used later to provide water to livestock or for domestic use. Other water harvesting techniques divert runoff from hillslopes or *wadis* and spread the water in a systematic manner across *playas* and the toe-slope of a hillslope. These systems allow production of crops in areas with 100 mm of average annual precipitation by harvesting an additional 300+ mm of water (Beckers et al. 2013). Water harvesting is a proven technology to mitigate or adapt to climate change where precipitation may be reduced, and allow for small-scale crop and livestock production to continue supporting local needs.

1.7.5.2 India

The second great challenge after the Green Revolution in India was the low productivity in the rain-fed and semi-arid regions where land degradation and drought were serious concerns. In response to this challenge IWM projects were implemented over large areas in semi-arid biomes over the past few decades. IWM was meant to become a key factor in meeting a range of social development goals in many semi-arid rainfed agrarian landscapes in India (Bouma et al. 2007; Kerr et al. 2002). Over the years, watershed development has become the fulcrum of rural development, and has the potential to achieve the twin objectives of ecosystem restoration and livelihood assurance in the drylands of India (Joy et al. 2004).

Many reports indicate significant improvements in mitigation of drought impacts, raising crops and fodder, livestock productivity, expanding the availability of drinking water and increasing incomes as a result of IWM (Rao 2000), but in some cases overall the positive impact of the programme has been questioned and, except in a few cases, the performance has not lived up to expectations (Joy et al. 2004; JM Kerr et al. 2002). Comparisons of catchments with and without IWM projects using remotely sensed data have sometimes shown no significant enhancement of biomass, in part due to methodological challenges of space for time comparisons (Bhalla et al. 2013). The factors contributing to the successful cases were found to include effective participation of stakeholders in management (Rao 2000; Ratna Reddy et al. 2004).

Attribution of success in soil and water conservation measures was confounded by inadequate monitoring of rainfall variability and lack of catchment hydrologic indicators (Bhalla et al. 2013). Social and economic trade-offs included bias of benefits to downstream crop producers at the expense of pastoralists, women and upstream communities. This biased distribution of IWM benefits could potentially be addressed by compensation for environmental services between communities (Kerr et al. 2002). The successes in some areas also led to increased demand for water, especially groundwater, since there has been no corresponding social regulation of water use after improvement in water regime (Samuel et al. 2007). Policies and management did not ensure water allocation to sectors with the highest social and economic benefits (Batchelor et al. 2003). Limited field evidence of the positive impacts of rainwater harvesting at the local scale is available, but there are several potential negative impacts at the watershed scale (Glendenning et al. 2012). Furthermore, watershed projects are known to have led to more water scarcity, and higher expectations for irrigation water supply, further exacerbating water scarcity (Bharucha et al. 2014).

In summary, the mixed performance of IWM projects has been linked to several factors. These include: inequity in the distribution of benefits (Kerr et al. 2002); focus on institutional aspects rather than application of appropriate watershed techniques and functional aspects of watershed restoration (Joy et al. 2006; Vaidyanathan 2006); mismatch between scales of focus and those that are optimal for catchment processes (Kerr 2007); inconsistencies in criteria used to select watersheds for IWM projects (Bhalla et al. 2011); and in a few cases additional costs and inefficiencies of local non-governmental organisations (Chandrasekhar et al. 2006; Deshpande 2008). Enabling policy responses for improvement of IWM performance include: a greater emphasis on ecological restoration rather than civil engineering; sharper focus on sustainability of livelihoods than just conservation; adoption of 'water justice' as a normative goal and minimising externalities on non-stakeholder communities; rigorous independent biophysical monitoring, with feedback mechanisms and integration with larger schemes for food and ecological security, and maintenance of environmental flows for downstream areas (Bharucha et al. 2014; Calder et al. 2008; Joy et al. 2006). Successful adaptation of IWM to achieve land degradation neutrality would largely depend on how IWM creatively engages with dynamics of large-scale land use and hydrology under a changing climate, involvement of livelihoods and rural incomes in ecological restoration, regulation of groundwater use, and changing aspirations of rural population (*robust evidence*, *high agreement*) (O'Brien et al. 2004; Samuel et al. 2007; Samuel and Joy 2018).

1.7.5.3 Limpopo River Basin

Covering an area of 412,938 km^2, the Limpopo River basin spans parts of Botswana, South Africa, Zimbabwe and Mozambique, eventually entering into the Mozambique Channel. It has been selected as a case study as it provides a clear illustration of the combined effect of desertification and climate change, and why IWM may be a crucial component of reducing exposure to climate change. It is predominantly a semi-arid area with an average annual rainfall of 400 mm (Mosase and Ahiablame 2018). Rainfall is both highly seasonal and variable, with the prominent impact of the El Niño/La Niña phenomena and the Southern Oscillation leading to severe droughts (Jury 2016). It is also exposed to tropical cyclones that sweep in from the Mozambique Channel often leading to extensive casualties and the destruction of infrastructure (Christie and Hanlon 2001). Furthermore, there is good agreement across climate models that the region is going to become warmer and drier, with a change in the frequency of floods and droughts (Engelbrecht et al. 2011; Zhu and Ringler 2012). Seasonality is predicted to increase, which in turn may increase the frequency of flood events in an area that is already susceptible to flooding (Spaliviero et al. 2014).

A clear need exists to both address exposure to flood events as well as predicted decreases in water availability, which are already acute. Without the additional impact of climate change, the basin is rapidly reaching a point where all available water has been allocated to users (Kahinda et al. 2016; Zhu and Ringler 2012). The urgency of the situation was identified several decades ago (FAO 2004), with the countries of the basin recognising that responses are required at several levels, both in terms of system governance and the need to address land degradation.

Recent reviews of the governance and implementation of IWM within the basin recognise that an integrated approach is needed and that a robust institutional, legal, political, operational, technical and support environment is crucial (Alba et al. 2016; Gbetibouo et al. 2010; Machethe et al. 2004; Spaliviero et al. 2011; van der Zaag and Savenije 1999). Within the scope of emerging lessons, two principal ones emerge. The first is capacity and resource constraints at most levels. Limited capacity within Limpopo Watercourse Commission (LIMCOM) and national water management authorities constrains the implementation of IWM planning processes (Kahinda et al. 2016; Spaliviero et al. 2011). Whereas strategy development is often relatively well-funded and resourced through donor funding, long-term implementation is often limited due to competing priorities. The second is adequate representation of all parties in the process in order to address existing inequalities and ensure full integration of water management. For example, within Mozambique, significant strides have been made towards the decentralisation of river basin governance and IWM. Despite good progress, Alba et al. (2016) found that the newly implemented system may enforce existing inequalities as not all stakeholders, particularly smallholder farmers, are adequately represented in emerging water management structures and are often inhibited by financial and institutional constraints. Recognising economic and socio-political inequalities, and explicitly considering them to ensure the representation of all participants, can increase the chances of successful IWM implementation.

1.8 Knowledge gaps and key uncertainties

- Desertification has been studied for decades and different drivers of desertification have been described, classified, and are generally understood (e.g., overgrazing by livestock or salinisation from inappropriate irrigation) (D'Odorico et al. 2013). However, there are knowledge gaps on the extent and severity of desertification at global, regional, and local scales (Zhang and Huisingh 2018; Zucca et al. 2012). Overall, improved estimation and mapping of areas undergoing desertification is needed. This requires a combination of rapidly expanding sources of remotely sensed data, ground observations and new modelling approaches. This is a critical gap, especially in the context of measuring progress towards achieving the Land Degradation Neutrality target by 2030 in the framework of SDGs.

- Despite numerous relevant studies, consistent indicators for attributing desertification to climatic and/or human causes are still lacking due to methodological shortcomings.

- Climate change impacts on dust and sand storm activity remain a critical gap. In addition, the impacts of dust and sand storms on human welfare, ecosystems, crop productivity and animal health are not measured, particularly in the highly affected regions such as the Sahel, North Africa, the Middle East and Central Asia. Dust deposition on snow and ice has been found in many regions of the globe (e.g., Painter et al. 2018; Kaspari et al. 2014; Qian et al. 2015; Painter et al. 2013), however, the quantification of the effect globally, and estimation of future changes in the extent of this effect, remain knowledge gaps.

- Future projections of combined impacts of desertification and climate change on ecosystem services, fauna and flora, are lacking, even though this topic is of considerable social importance. Available information is mostly on separate, individual impacts of either (mostly) climate change or desertification. Responses to desertification are species-specific and mechanistic models are not yet able to accurately predict individual species responses to the many factors associated with desertification under changing climate.

- Previous studies have focused on the general characteristics of past and current desertification feedbacks to the climate system. However, the information on the future interactions between climate and desertification (beyond changes in the aridity index) are lacking. The knowledge of future climate change impacts on such desertification processes as soil erosion, salinisation, and nutrient depletion remains limited both at the global and at the local levels.

- Further research to develop the technologies and innovations needed to combat desertification is required, but it is also important to gain a better understanding of the reasons for the observed poor adoption of available innovations, to improve adoption rates.

- Desertification under changing climate has a high potential to increase poverty, particularly through the risks coming from extreme weather events (Olsson et al. 2014). However, the evidence rigorously attributing changes in observed poverty to climate change impacts is currently not available.

- The knowledge on the limits to adaptation to the combined effects of climate change and desertification is insufficient. This is an important gap since the potential for residual risks and maladaptive outcomes is high.

- Filling these gaps involves considerable investments in research and data collection. Using Earth observation systems in a standardised approach could help fill some of these gaps. This would increase data comparability and reduce uncertainty in approaches and costs. Systematically collected data would provide far greater insights than incomparable fragmented data.

Frequently Asked Questions

FAQ 3.1 | How does climate change affect desertification?

Desertification is land degradation in drylands. Climate change and desertification have strong interactions. Desertification affects climate change through loss of fertile soil and vegetation. Soils contain large amounts of carbon, some of which could be released to the atmosphere due to desertification, with important repercussions for the global climate system. The impacts of climate change on desertification are complex and knowledge on the subject is still insufficient. On the one hand, some dryland regions will receive less rainfall and increases in temperatures can reduce soil moisture, harming plant growth. On the other hand, the increase of CO_2 in the atmosphere can enhance plant growth if there are enough water and soil nutrients available.

FAQ 3.2 | How can climate change induced desertification be avoided, reduced or reversed?

Managing land sustainably can help avoid, reduce or reverse desertification, and contribute to climate change mitigation and adaptation. Such sustainable land management practices include reducing soil tillage and maintaining plant residues to keep soils covered, planting trees on degraded lands, growing a wider variety of crops, applying efficient irrigation methods, improving rangeland grazing by livestock and many others.

FAQ 3.3 | How do sustainable land management practices affect ecosystem services and biodiversity?

Sustainable land management practices help improve ecosystems services and protect biodiversity. For example, conservation agriculture and better rangeland management can increase the production of food and fibres. Planting trees on degraded lands can improve soil fertility and fix carbon in soils. Sustainable land management practices also support biodiversity through habitat protection. Biodiversity protection allows for the safeguarding of precious genetic resources, thus contributing to human well-being.