Chapter 3 : Desertification

Coordinating Lead Authors: Alisher Mirzabaev (Uzbekistan), Jianguo Wu (China)

Lead Authors: Jason Evans (Australia), Felipe Garcia-Oliva (Mexico), Ismail Abdel Galil Hussein (Egypt), Muhammad Mohsin Iqbal (Pakistan), Joyce Kimutai (Kenya), Tony Knowles (South Africa), Francisco Meza (Chile), Dalila Nedjraoui (Algeria), Fasil Tena (Ethiopia), Murat Türkeş (Turkey), Ranses José Vázquez (Cuba), Mark Weltz (United States of America)

Contributing Authors: Abdul Rasul Awan (Pakistan), Céline Bellard (France), Arden Burrell (Australia), Robyn Hetem (South Africa), Kathleen Hermans (Germany), Margot Hurlbert (Canada), Zaneta Kubik (Poland), Ali Mohammed (Egypt), Katerina Michaelides (Cyprus and United Kingdom), Lindsay Stringer (United Kingdom), Grace Villamor (Philippines)

Review Editors: Mariam Akhtar-Schuster (Germany), Fatima Driouech (Morocco), Mahesh Sankaran (India)

Chapter Scientists: Chuck Chuan Ng (Malaysia), Helen Berga Paulos (Ethiopia)

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3.1. Executive summary

Dryland areas are expected to become more vulnerable to desertification due to increasing number, frequency and intensity of extreme climatic events (*high confidence*) {3.2.1, 3.3.2, 3.6.1, 3.6.2}. Desertification is land degradation in drylands, and the range and the intensity of desertification increased in some dryland areas over the past several decades {3.3.1}. Expansion of drylands, as measured by the aridity index, has already occurred in north-eastern Brazil, southern Argentina, the southwest of the United States, eastern Africa, the Middle East, Central Asia, the Sahel, Zambia and Zimbabwe, some regions of the Mediterranean area, North-Eastern China and sub-Himalayan India during the last three decades as compared to the period of 1951–1980 {3.3.1.2}. However, the CO_2-fertilisation effect is *likely* to be mitigating the expansion of dryland areas in terms of changes in vegetation cover. Biomass productivity-based desertification hotspots currently cover about 10% of the drylands, directly affecting about 277 million people {3.2.1, 3.3}. Future climate changes with increasing frequency, intensity and scales of extreme weather events, for example droughts and heat waves, are expected to further exacerbate the vulnerability and risk of humans and ecosystems to desertification, in particular, drought and/or aridity are projected to increase as a result of 1.5°C to a 2°C global warming (*high confidence*) {3.2.1, 3.3.2, 3.6.1}.

Attribution of desertification to climate variability and change and human activities is context-dependent (*high confidence*). Climate variability and change, particularly through increase both in land surface air temperature and evapotranspiration, and decrease in precipitation, are *likely* to have played a larger role, in interaction with human drivers, in causing desertification than previously estimated for some dryland areas (*medium evidence, medium agreement*) {3.3.2}. The major human drivers of desertification interacting with climate change are expansion of croplands and urban areas, unsustainable land management practices and increased pressure on land from population and income growth (*robust evidence, high agreement*) {3.2.4.2}. Poverty and migration also exacerbate desertification under a changing climate {3.2.4.2} (*limited evidence, medium agreement*).

Desertification exacerbates climate change through several mechanisms such as changes in vegetation cover, surface albedo, sand and dust aerosols and greenhouse gases fluxes (*high confidence*). Through its effect on vegetation and soils, desertification changes the absorption and release of associated greenhouse gases (GHGs) {3.4.3}. The extent of areas in which dryness controls CO_2 exchange (rather than temperature) has increased by 6% since 1948 and is expected to increase by at least another 8% by 2050 if the expansion continues at the same rate. In these areas, net carbon uptake is about 27% lower than in other areas {3.6.2}. Vegetation loss and drying of surface cover due to desertification increases the frequency of dust storms (*high confidence*). Dust particles intercept, reflect and absorb solar radiation in the atmosphere, reducing the heat energy available at the land surface and increasing the temperature of the atmosphere. Depending on the types and amounts of aerosols present, sand and dust storms increase the cloud reflectivity and decrease the chances of precipitation {3.4.1}. Deposition of dust storms on the oceans was found to have a direct effect of cooling, while the indirect effect of dust storms as a source of nutrients for the upper ocean biota is contested {3.4.1.1}.

The interaction of climate change and desertification reduces the provision of dryland ecosystem services and lowers ecosystem health, including loss of biodiversity, affecting food security and human well-being (*high confidence*). Desertification processes, coupled with climate change, are expected to cause reductions in crop and livestock productivity, increases in soil erosion, and soil salinity in dryland areas in Latin America, Caribbean and sub-Saharan Africa by 2055 (*high confidence*) {3.5.1.1}. Desertification has already contributed to the global loss of biodiversity (*medium confidence*) {3.5.1.2}. Wildlife are *likely* to be negatively affected by coupled effects of climate change and desertification. A reduction in the quality and quantity of resources available to herbivores is *likely* to have synergistic consequences for predators, potentially disrupting ecological cascades (*limited evidence, low agreement*) {3.5.1.2}. Future desertification with climate change will bring high risk for the ecosystem services and biodiversity in drylands due to increasing frequency and intensity of droughts, dust storms, and soil erosion. The distribution of areas affected by desertification is also projected to change due to changes in drylands areas following climate change, in particular from a 1.5°C to a 2°C global warming (*low confidence*) {3.6.2}.

Increasing population pressures combined with climate change are *likely* to push dryland populations beyond their resilience thresholds and the limits for their autonomous adaptation, requiring policy interventions aimed at maintaining and strengthening their resilience and adaptive capacities (*robust evidence, medium agreement*). The combination of pressures coming from climate change and desertification contribute, in interaction with other contextual factors, to migration, conflict, poverty, food insecurity, and increased disease burden (*medium confidence*) {3.5.2}. Migration is increasingly used as an adaptation response in the context of environmental change (*medium evidence, high agreement*). However, environmentally-induced migration is complex and its attribution to environmental change should account for multiple drivers of mobility as well as other adaptation measures undertaken by populations exposed to environmental risk (*high confidence*) {3.5.2.4}.

Higher frequency, intensity and scales of dust storms due to climate change-desertification interactions will reduce human wellbeing in drylands and beyond (*high confidence*). Increased dust storm activity because of desertification and climate change has a high potential for negative human health impacts due to associated respiratory and cardiovascular illnesses (*medium evidence, high agreement*) {3.5.2.8}. Higher intensity of sand storms and sand dune movements under climate change also cause damage to transportation and solar energy generating infrastructures (*high confidence*) {3.5.2.9, 3.5.2.10}.

Site-specific technological solutions, based both on new scientific innovations and indigenous and local knowledge, are available to avoid, reduce and reverse desertification, simultaneously contributing to climate change mitigation and adaptation (*high confidence*). Sustainable land management (SLM) practices in drylands contribute to climate change mitigation and adaptation, increase agricultural productivity, and have substantial co-benefits for the attainment of Sustainable Development Goals (*high confidence*) {3.5.2, 3.7.1}. Integrated soil and water conservation measures increase vegetation coverage and density (*medium confidence*). Conservation agriculture contributes to carbon sequestration in dryland areas (*medium confidence*). It also increases climate change adaptation capacities of agricultural households (*high confidence*). The combined use of salt-tolerant crops, improved irrigation practices, chemical remediation measures and appropriate mulch and compost (that are low in salts) is effective in reducing salinity-induced desertification (*medium confidence*). Rangeland management systems such as sustainable grazing approaches and re-vegetation increases rangeland productivity (*medium confidence*). Agroforestry practices generate diverse ecological benefits, including soil and water conservation, increased carbon sequestration, and reduced erosion. Afforestation programs for the creation of windbreaks in the form of "green walls", "green belts", and "green dams" helped to stabilise and reduce sand storms, avert aeolian desertification, and served as carbon sinks {3.7.1, 3.8.2}.

Investments into land restoration and rehabilitation in dryland areas have positive economic returns (*high confidence*). Each dollar invested into land restoration has social returns of 2–5 dollars over a 30-year period globally {3.7.1}. Despite their benefits in addressing desertification, mitigating and adapting to climate change, many SLM practices are not widely adopted due to insecure property rights, lack of access to credit and agricultural advisory services, and insufficient private incentives *(robust evidence, high agreement*) {3.7.1, 3.7.2}.

Indigenous and local knowledge distilled into traditional agroecological practices contributes to enhancing resilience against climate change and combating desertification (*medium confidence*). Dryland populations have historically developed traditional agroecological practices which are well adapted to resource-sparse dryland environments {3.7.1, 3.7.2}. However, there is *robust evidence* documenting losses of traditional agroecological knowledge. Traditional agroecological practices are also increasingly unable to cope with growing demand pressures and environmental changes {3.7.2}. Innovative combinations of indigenous and local knowledge and modern agronomic practices can contribute to overcoming combined challenges of climate change and desertification (*medium evidence, medium confidence*).

Policy frameworks promoting the adoption of sustainable land management solutions contribute to addressing desertification as well as mitigating and adapting to climate change, with significant co-benefits for poverty reduction and food security among dryland populations (*medium confidence*). On-farm and off-farm livelihood diversification strategies increase the resilience of rural agricultural households against extreme weather events, such as droughts, and desertification (*high confidence*). Strengthening collective action is important for addressing desertification causes and impacts, and for adapting to climate change (*medium confidence*). Access to markets, such as those based on new information and communication technologies, raises agricultural profitability and motivates investment into climate change adaptation and SLM (*medium confidence*) {3.7.2, 3.7.3}. Promoting schemes that provide payments for ecosystem services gives additional incentives to land users to adopt SLM practices (*medium evidence, high agreement*) {3.7.3}.

Improving human and institutional capacities and accessibility to information, including to early warning, hydro-meteorological and remote sensing-based earth monitoring systems, and expanded use of digital technologies are high return investments for measuring progress in addressing desertification under changing climate (*low evidence, high agreement*). Effective national, regional and international monitoring and early warning systems help combat desertification and extreme events (*medium confidence*) {3.8.6}. Adoption of land degradation neutrality policies lead to balancing of ecosystem service performance and land improvement (*low evidence, high agreement*). Increasing investments into strengthening research, education and extension services accelerates the achievement of land degradation neutrality targets (*high confidence*). Expanded use of new information and communication technologies, remotely sensed information and of “citizen science” for data collection helps in measuring progress towards achieving the land degradation neutrality target and raising public awareness and participation in sustainable land management (*low evidence, high agreement*) {3.7.2, 3.7.3}.

3.2. The Nature of Desertification

3.2.1. Introduction

Desertification is land degradation in arid, semi-arid, and dry sub-humid areas resulting from many factors, including human activities and climatic variations (UNCCD, 1994; Glossary). Arid, semi-arid, and dry sub-humid areas, together with hyper-arid areas, constitute drylands (UNEP, 1992). Consequently, although land degradation occurs anywhere across the world, it is defined as desertification when it occurs in drylands. Desertification is not limited to only irreversible forms of land degradation, nor is it limited to processes of desert expansion, but is used to represent all forms and levels of land degradation occurring in drylands. In turn, land degradation is a deterioration or persistent decline in land conditions resulting in long-term reduction or loss of the biological productivity of land, its ecological complexity, and/or its human values, caused by direct and/or indirect human-induced processes or impacts, including climate change (Chapter 4; Glossary). Thus, desertification is manifested through the reduced provision of the sum of dryland ecosystem services (Verstraete et al., 2009; Safriel et al., 2005; Scholes, 2009).

The geographic classification of drylands is often based on the aridity index - the ratio of average annual precipitation amount (P) to potential evapotranspiration amount (PET) (Figure 3.1, Glossary). Hyper-arid areas, where the aridity index is below 0.05, are included in drylands (Section 3.8.4), but are excluded from the definition of desertification (UNCCD, 1994). Moreover, aridity is different from drought: aridity is a long-term climatic feature, whereas drought is a temporary climatic event (Maliva and Missimer, 2012). Droughts are not restricted to drylands, but occur both in drylands and humid areas (Wilhite et al., 2014). IPCC (2014) defines drought as "a period of abnormally dry weather long enough to cause a serious hydrological imbalance" (Section 3.8.6; Glossary).

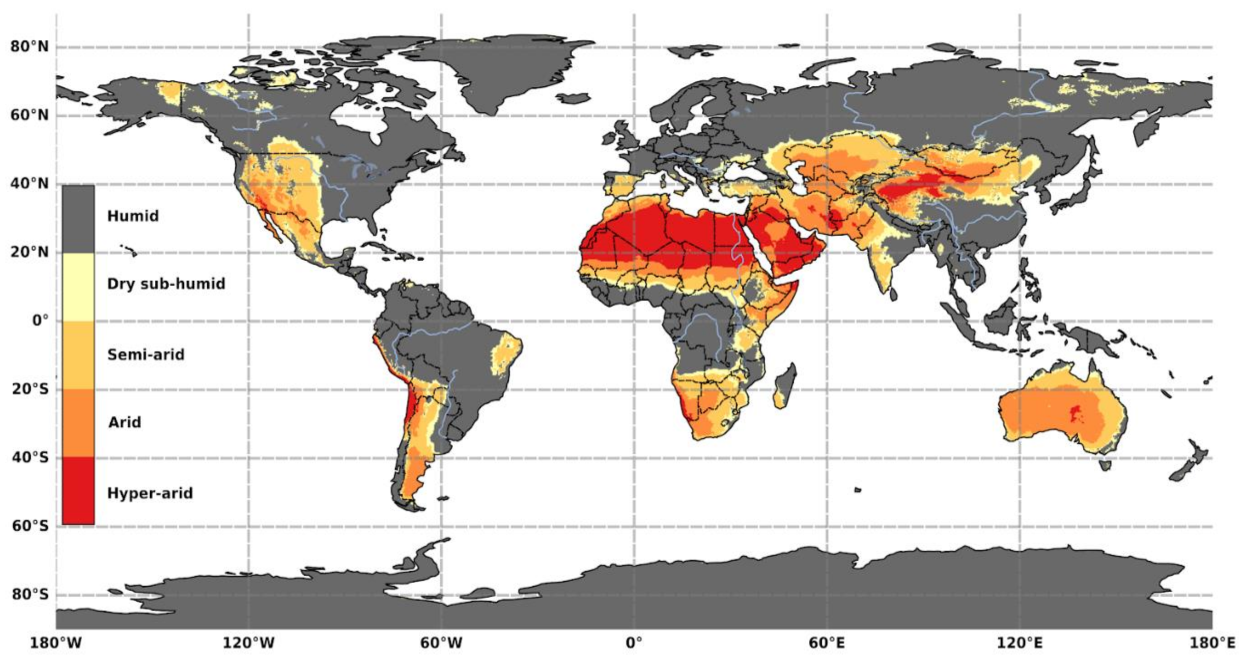

Figure 3.1 Geographical distribution of drylands, delimited based on the Aridity Index. The classification of the aridity index (AI) is: Humid AI>0.65, Dry sub-humid 0.50 < AI < 0.65, Semi-arid 0.20 < AI < 0.50, Arid 0.05 < AI < 0.20, Hyper-arid AI < 0.05. Data: TerraClimate precipitation and potential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

Safriel et al. (2005) earlier estimated that drylands occupy about 41.3% of the Earth's land surface. A more recent estimate suggested that drylands cover about 45.4% of the global land area, the difference being mainly due to improved data, capturing the expansion of drylands towards northern latitudes (Prăvălie, 2016). Although climate change is expected to decrease the aridity index, implying more arid conditions in the future due to increases in potential evaporation, the assumptions that underpin the potential evaporation calculation are not consistent with a changing CO_2 environment (Roderick et al., 2015; Greve et al., 2017). Given that future climate is characterised by significant increases in CO_2, the usefulness of currently applied aridity index thresholds to estimate dryland areas is limited under climate change. If instead of the aridity index, 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.

The majority of dryland areas, approximately 70%, are located in Africa and Asia (Figure 3.2). The biggest land use/cover in drylands, if deserts are excluded, in terms of area 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 (Section 3.8.4). Moreover, FAO (2016) defines grasslands as permanent pastures and meadows used continuously for more than 5 years. In drylands, transhumance often leads to non-permanent pasture systems, thus, some of the areas under "other land" category are also used as non-permanent pastures (Ramankutty et al., 2008; Fetzel et al., 2017; Erb et al., 2016).

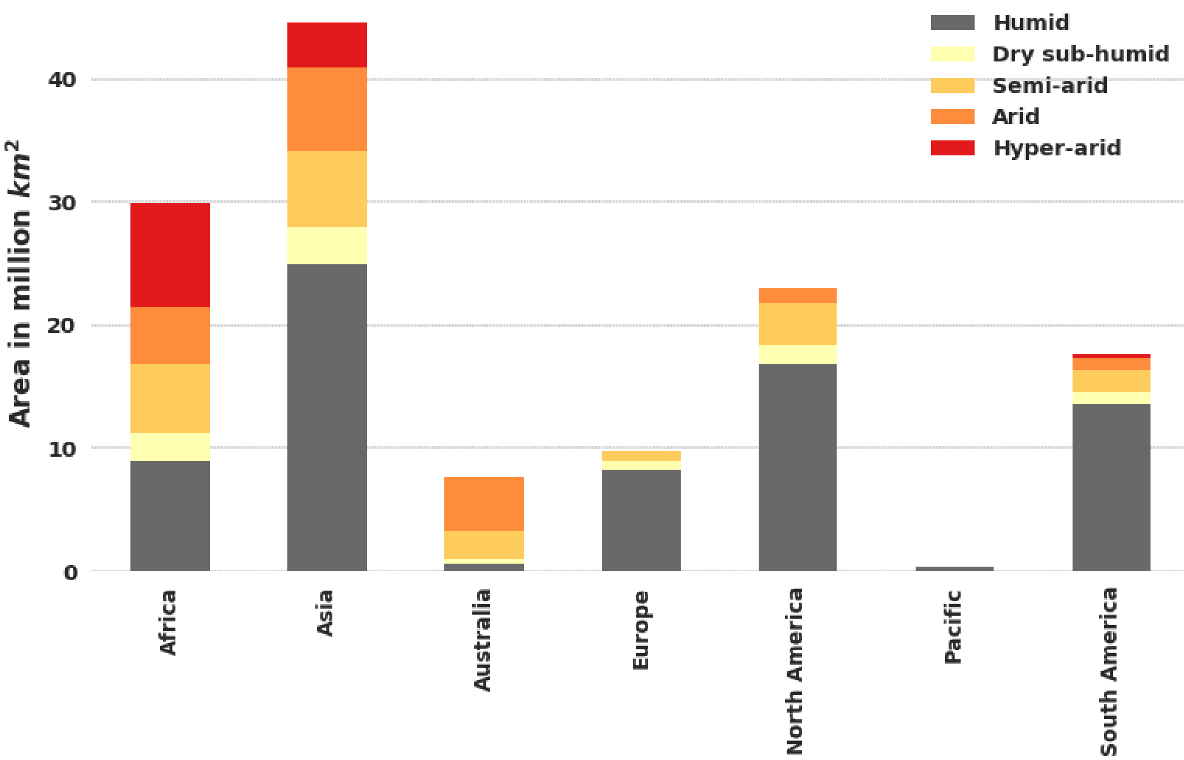

Figure 3.2 Dryland categories across geographic areas. Data: TerraClimate precipitation and potential evapotranspiration (1980-2015) (Abatzoglou et al., 2018)

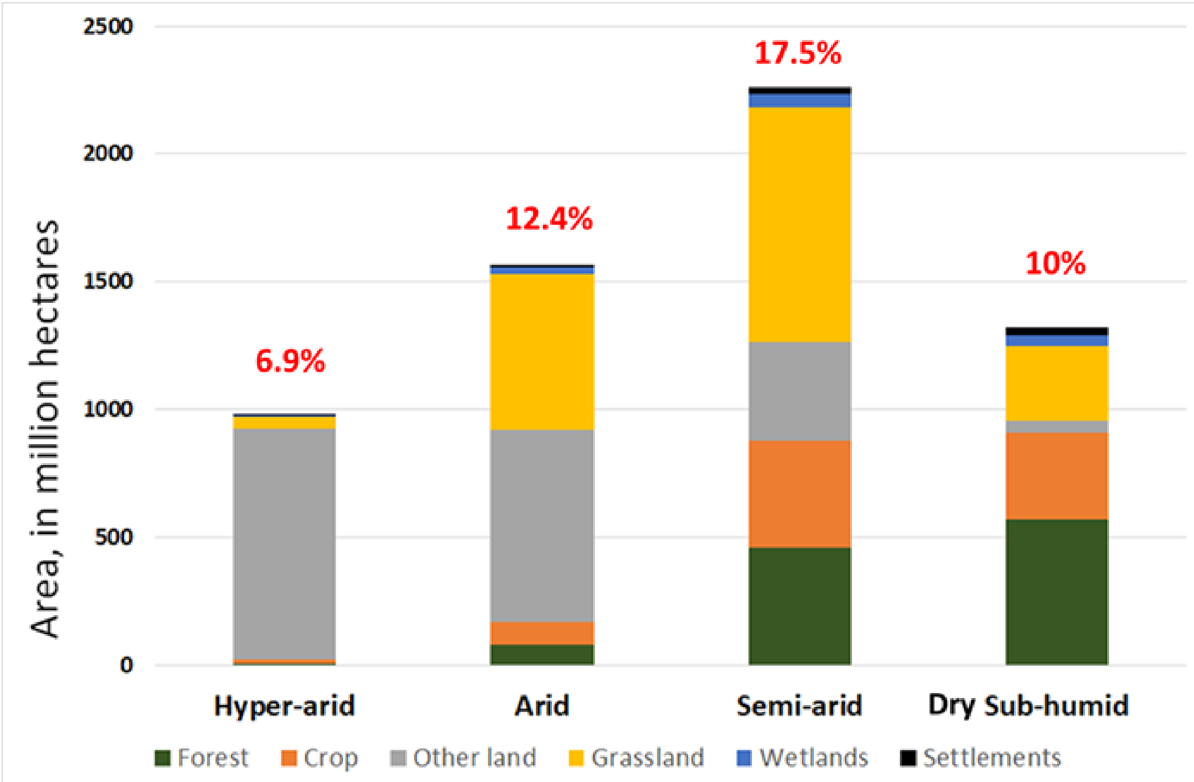

Figure 3.3 Land use and land cover in drylands (in million hectares) and share of each dryland category in global land area (in percentages). Source: FAO (2016) and own calculations for the shares in the global land area

Earlier global assessments of desertification since the 1970s, based on qualitative expert evaluations, estimated the extent of desertification to be 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 a reduction in biomass productivity between 1980s and 2000s (Bai et al., 2008; Le et al., 2016b). The figures by Bai et al. (2008) show that only 28% of the global areas with biomass productivity loss are located in drylands. Analysis of the figures by Le et al. (2016b) show that about 10% of drylands, excluding hyper-arid areas, experienced significant declines in biomass productivity between 1980s and 2000s.

Available assessments of the global extent and severity of desertification are still 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), NDVI (Piao et al., 2005), drought-tolerant plant species (An et al., 2007), grass cover (Bestelmeyer et al., 2013), land productivity dynamics trend (Baskan et al., 2017), ecosystem net primary productivity (Zhou et al., 2015) or environmentally sensitive land area index (Symeonakis et al., 2014). In addition, some synthetic indicators of desertification have also been used to assess desertification extent and desertification process, such as climate, land use, soil, and socioeconomic 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 extent of desertification than previously estimated (Section 3.3).

The present assessment of desertification under changing climate is conceptually structured taking into account that it is the links within coupled social-ecological systems that drive desertification-climate change interactions, at each level from drivers (Section 3.2.4) and feedbacks (Section 3.4), to observed and projected impacts (Sections 3.5 and 3.6), and responses (Section 3.7). Moreover, this assessment highlights that dryland populations are highly vulnerable to desertification and climate change (Section 3.3 and 3.5). However, the evidence does not support the narrative of an inevitable vicious cycle of resource degradation and poverty in drylands due to desertification and climate change (Section 3.2.4.3). On the contrary, 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 3.7.2). However, increasing population pressures combined with climate change can push dryland populations beyond their resilience thresholds and the limits for their autonomous adaptation, requiring policy interventions aimed at maintaining and strengthening their resilience and adaptive capacities. The assessment finds that policies promoting sustainable land management in drylands will contribute to climate change mitigation and adaptation, with substantial co-benefits in terms of sustainable development.

3.2.2. Desertification in previous IPCC and related reports

The Fifth Assessment report (AR5) of the IPCC includes some discussion of desertification. In AR5 Working Group I desertification is mentioned as a forcing agent for the production of atmospheric dust (IPCC, 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, while processes such as soil degradation are identified as increasing the risk of desertification (IPCC, 2014). For Africa, AR5 Working Group II notes "Climate change will amplify existing stress on water availability and on agricultural systems particularly in semi-arid environments (*high confidence*)." 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℃ (IPCC, 2018) pointed out that there is *limited evidence and medium agreement* that the extent of deserts will increase in the coming decades. However, the deserts are expected to become drier and warmer more rapidly than other terrestrial areas (IPCC, 2018). IPCC (2018) assessed as "*low confidence*" that desertification linked to climate change will directly or indirectly influence soil health and productivity due to accelerated soil erosion in drylands. IPCC (2018) also had "*low confidence"* in the projections of future increases in dust storms with higher aridity.

The recent Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) Assessment report on land degradation and restoration (IPBES, 2018) is also of particular relevance. While acknowledging a wide variety of past estimates of the area undergoing degradation such that there is *low agreement* about where degradation is taking place, IPBES (2018) nevertheless concludes that desertification is occurring on all continents and affects more than the total population of the drylands due to effects outside the drylands through migration. They recognise that "at a regional or global scale, distinguishing the impacts of climate change and variability from anthropogenic degradation remains problematic (unresolved)." They also identify that "there is growing concern over the impacts that climate change may have on degradation (inconclusive).” However, this issue is not examined in great detail and is not the focus of IPBES (2018).

The third edition of the World Atlas of Desertification (Cherlet et al., 2018) argues against the idea of deterministically mapping land degradation globally, or its subset - desertification, indicating 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) present global maps showing the convergence of various pressures on land resources. For example, although climate variability, particularly related to droughts, is recognised as a limit to sustainability, Cherlet et al. (2018) do not focus on climate change *per se*. Various sources of pressures on land and limits to sustainability emphasise the interactions within coupled social-ecological systems in driving desertification (Cherlet et al., 2018).

3.2.3. Dryland Populations: Vulnerability and Resilience to Desertification and Climate Change

Drylands are home to approximately 37.5% of the global population (Netherlands Environmental Assessment Agency (PBL), 2017), that is about 2.7 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 (PBL, 2017). The population in drylands is projected to increase about twice as rapidly as those in non-drylands to reach 4 billion people by 2050 (PBL, 2017). This is due to higher population growth rates in drylands.

In terms of the number of people affected by desertification, the earlier estimates by MEA (2005) and Reynolds et al. (2007) indicated that desertification was directly affecting 250 million people and indirectly 1 billion people. Similarly, the data from Le et al. (2016b) show that approximately 277 million people reside in dryland areas which experienced significant loss in biomass productivity between 1982–1984 and 2004–2006.

Dryland populations are highly vulnerable (Glossary) to desertification and climate change (Howe et al., 2013; Huang et al., 2016, 2017; Liu et al., 2016; Thornton et al., 2014; Lawrence et al., 2018), because their livelihoods are predominantly dependent on agriculture; one of the most susceptible sectors 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 3.5.1 and 3.5.2). One key vulnerable group in drylands are pastoral and agropastoral households[1]. It is estimated that there are about 120 million people practicing pastoralism and agropastoralism globally (Rass, 2006), predominantly in drylands, 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

[1]FOOTNOTE: Pastoralists derive more than 50% of their income from livestock and livestock products, whereas agro-pastoralists generate more than 50% of their income from crop production and at least 25% from livestock production (Swift, 1988).

[2]FOOTNOTE: 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).

security threats, 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).

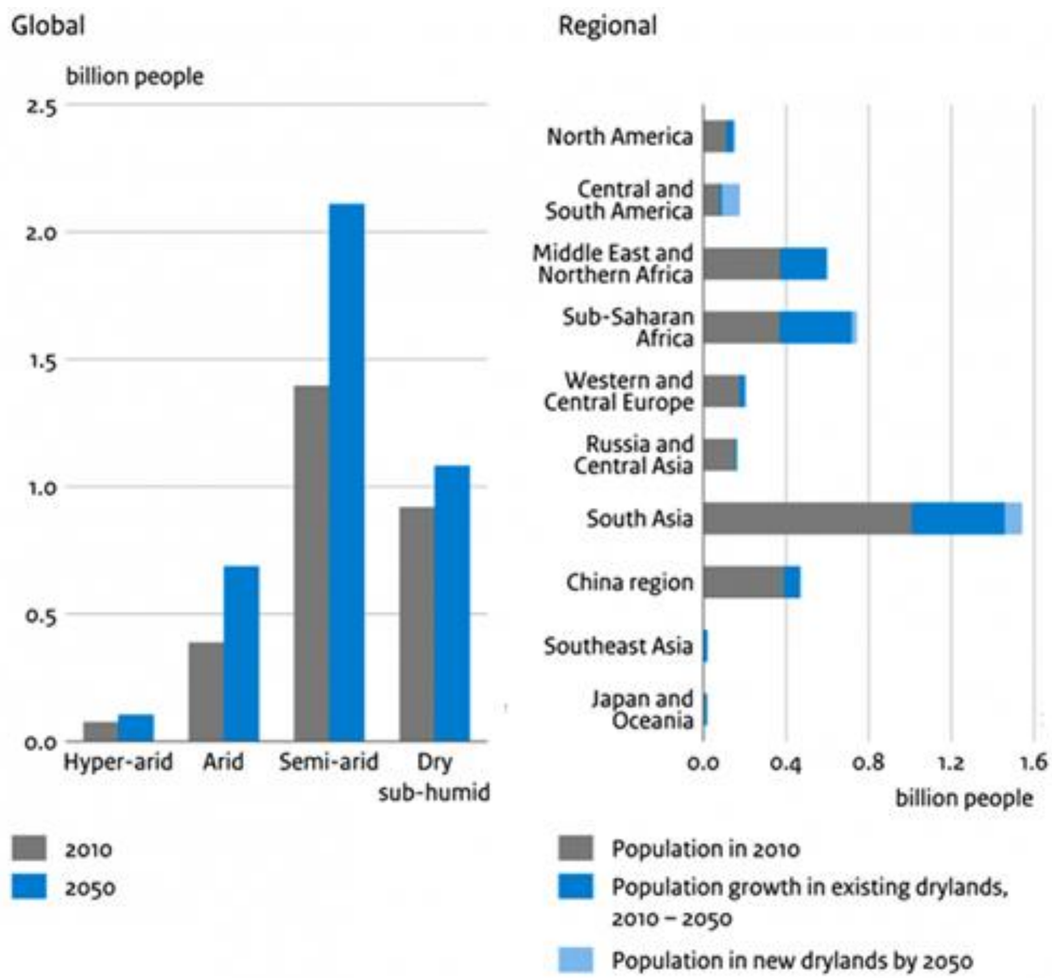

Figure 3.4 Current and projected population in drylands. Source: (PBL, 2017)

There is an increasing concentration of poverty in the dryland areas of Sub-Saharan Africa and India (von Braun and Gatzweiler, 2014; Barbier and Hochard, 2016). Rapid economic growth and poverty reduction in China over the last three decades decreased the absolute global numbers of dryland populations living in poverty. Only between 1981 and 2001, the share of the people living under poverty in China declined from 53% to 8% (Ravallion and Chen, 2007), translating to about 426 million people getting out of poverty. However, the areas affected by desertification remain among the poorest in China (Yang et al., 2008; Liu et al., 2017). 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, but 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 et al., 2010; Alkire and Santos, 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, low agricultural productivity (Sietz et al., 2011; Reynolds et al., 2011; Safriel and Adeel, 2008; Stafford Smith, 2016). However, an up-to-date quantification of poverty in drylands, particularly of multi-dimensional aspects of poverty, and of its sub-national variations, are currently not available. Even in high-income countries, dryland areas depending on agricultural livelihoods represent relatively poorer locations nationally, with lack of 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). Local cultural traditions and patriarchal 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 3.5.2 and 3.7.3).

Despite these environmental, socio-economic and institutional constraints, dryland populations have historically demonstrated remarkable resilience (Glossary), ingenuity and innovations, distilled into indigenous and local knowledge (Glossary) to cope with high climatic variability and sustain livelihoods (Safriel and Adeel, 2008; Davies, 2017; Sections 3.7.1 and 3.7.2). Indigenous and local knowledge has been used for centuries to manage dynamic interactions between local communities and ecosystems in dryland areas. For example, in the Middle East and North Africa (MENA), informal bylaws were enforced by the Bedouin communities for regulating grazing, collection and cutting of herbs and wood, limiting rangeland degradation (Hussein, 2011). Pastoralists in Mongolia developed indigenous classifications of pasture resources which facilitated ecologically optimal grazing practices (Fernandez-Gimenez, 2000) (Section 3.7.2). However, climate change is increasing the exposure of dryland populations to extreme weather events, such as droughts, floods and dust storms, testing their adaptive capacities, potentially beyond historical precedents (Orlowsky and Seneviratne, 2012; Huang et al., 2016). Out of 424.7 million people exposed to droughts in Sub-Saharan Africa in 2010, Cervigni et al. (2016) estimated that about 23% were not able to cope with them, implying that following a drought shock these households' incomes will fall below the poverty line. Policy actions promoting the adoption of sustainable land management (SLM) (Glossary; Chapter 4) 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 attaining other Sustainable Development Goals (Safriel and Adeel, 2008; Schwilch et al., 2014; Cowie et al., 2018; Nkonya et al., 2016a; Stafford Smith et al., 2017; IPBES, 2018; Liniger et al., 2017).

3.2.4. Processes and Drivers of Desertification under Climate Change

3.2.4.1 Processes of Desertification and Their Climatic Drivers

Processes of desertification are mechanisms by which drylands are degraded. Desertification consists of both abiotic and biotic 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 (Racine, 2008; Lal, 2016; IPBES, 2018; UNCCD, 2017). Those which are particularly relevant for this assessment in terms of their links to climate change are: for physical processes - soil erosion by water and wind, and soil structure degradation; for chemical processes - secondary salinisation and nutrient depletion; for biological processes - changes in vegetation cover and composition, including through over/under grazing, deforestation and biodiversity loss (Chapter 4; Glossary).

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 and Lambin, 2004; Reynolds et al., 2007) (Sections 3.2.4.2, 3.2.4.3). By definition, processes and drivers of desertification are similar to the processes and drivers of land degradation. For this reason, they are summarised in Cross-Chapter Table 4.1 in Chapter 4.

Erosion refers to removal of soil by the physical forces of water, wind, or through farming activities such as tillage (Pierson and Williams, 2016). There is a significant potential for climate change to increase global soil erosion by water, as precipitation volumes and intensity are projected to increase (Panthou et al., 2014; Nearing et al., 2015). On the other hand, there is *low evidence* concerning climate change impacts on wind erosion (Cross-Chapter Table 4.1 in Chapter 4; Section 3.8.1).

Saline and sodic soils occur naturally in arid, semiarid and dry sub-humid regions of the world. Climate change or hydrological change can cause soil salinisation due to the increase of the mineralised ground water level. However, secondary salinisation occurs when 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 sea water intrusion are amplified by climate change (Section 4.11.6 in Chapter 4).

A major consequence of desertification is the reduction in soil carbon (C) and transfer of C from soil to the atmosphere (Lal, 2009). Global warming is expected to accelerate soil organic carbon turnover, in some areas leading to soil organic carbon decline (Section 3.4.3; Section 3.6.2).

North Atlantic sea surface temperature (SST) anomalies are positively correlated with Sahel rainfall anomalies (Knight et al., 2006; 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 before present), 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).

Invasive plants contributed to desertification and loss of ecosystem services in many dryland areas in the last century (Section 3.8.3). Extensive woody plant encroachment altered runoff and soil erosion across much of the drylands and significantly contributed to desertification. 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 Great Basin 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 desertification and reductions in forage availability, wildlife habitat, and biodiversity (Pierson et al., 2011, 2013; Miller et al., 2013).

Predicted increases in temperature and the severity of drought events across dryland areas of the world are *likely* to increase chances of wildfire occurrence (Jolly et al. 2015; Williams and Funk 2010; Clarke and Evans 2018). This includes the semiarid and dry sub-humid areas of the world, where 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).

3.2.4.2. Anthropogenic Drivers of Desertification under Climate Change

There are numerous drivers of desertification related to human activities. The literature on these human drivers of desertification is substantial (D'Odorico et al., 2013; Sietz et al., 2011b; Yan and Cai, 2015; Sterk et al., 2016; Varghese and Singh, 2016; to list a few) and there have been several comprehensive reviews and assessments of these drivers very recently (IPBES, 2018; UNCCD, 2017; Nkonya et al., 2016b,d; Cherlet et al., 2018). IPBES (2018) identified cropland expansion, unsustainable land management practices, urban expansion, infrastructure development, and extractive industries as the main drivers of land degradation. IPBES (2018) also found that the ultimate driver of land degradation is high and growing consumption, 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.

Some of the major forms of desertification are related to land use conversions, including transformation of rangelands and woodlands into croplands in order to meet growing food demands (Bestelmeyer et al., 2015; D'Odorico et al., 2013). Climate change is projected to have negative impacts on crop yields across dryland areas (Section 3.5.1; Chapter 5), potentially reducing local production of food and feed. Without research breakthroughs to mitigate these productivity losses through higher agricultural productivity, and reducing food waste and loss, meeting increasing food demands of growing populations will require expansion of cropped areas to more marginal and easily degradable areas (with most prime areas in drylands already being under cultivation), thus intensifying degradation processes (Lambin, 2012; Lambin et al., 2013; Eitelberg et al., 2015). Although local food demands could also be met by importing from other areas, this would mean increasing the pressure on land in other areas (Lambin and Meyfroidt, 2011). The net effects of such global agricultural production shifts on desertification are not known.

Climate change will exacerbate poverty among some categories of dryland populations (Section 3.5.2). Depending on the context, this impact comes through changes in agricultural productivity, agricultural prices and extreme weather events (Hertel and Lobell, 2014; Hallegatte and Rozenberg, 2017). There is *robust evidence and high agreement* that poverty limits both capacities to adapt to climate change and availability of financial resources to invest into sustainable land management (SLM) (Sections 3.6.2, 3.7.2, 3.7.3; Gerber et al., 2014; Way, 2016; Vu et al., 2014).

Another key human driver which will interact with climate change is labour mobility. Although strong impacts of climate change on migration are contested, in some places, it is *likely* to provide an added incentive to migrate (Section 3.5.2.7). 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 re-invested into sustainable land management. Out-migration could allow land consolidation, gradually leading to mechanisation and agricultural intensification (Wang et al., 2014, 2018). On the other hand, it increases 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 for land use expansion to marginal areas(Taylor et al., 2016; Gray and Bilsborrow, 2014). The net effect of these countervailing mechanisms is context-dependent (Qin and Liao, 2016). There is very little literature evaluating these joint effects of climate change, desertification and migration (Chapter 7).

Besides these factors, there are 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 incentive (D'Odorico et al., 2013; Geist and Lambin 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 role in modulating responses for climate change adaptation and mitigation. Section 3.7.3 on policy responses discusses these factors from such a perspective.

3.2.4.3 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 paradigm" 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 into sustainable land management (Safriel and Adeel, 2008; MEA, 2005b). The alternative paradigm is one of "drylands development", which refers to social and technical ingenuity of dryland populations as a driver of dryland sustainability (Reynolds et al., 2007; Safriel and Adeel, 2008; MEA, 2005b). Reynolds et al. (2007) indicate that in drylands, which are a non-equilibrium system, there is a high temporal climatic variability. 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 simultaneous interactions of human and climatic drivers within coupled social-ecological systems. This non-equilibrium nature of drylands led Behnke and Mortimore (2016), and earlier Swift (1996), to conclude that the concept of desertification as irreversible degradation distorts policy and governance in the dryland areas. Mortimore (2016) suggested that instead of externally imposed technical solutions, what is needed is for local populations to adapt to this variable environment which they cannot control.

As demonstrated by the plethora of attribution studies discussed in Section 3.3.2, the quantified evidence on which factors, human or climatic, are more important in influencing the state of drylands is mixed. This is because anthropogenic and climatic drivers interact in complex ways in causing desertification (D'Odorico et al., 2013; Polley et al., 2013; Ravi et al., 2010). However, these biophysical and socio-economic drivers of desertification usually interact in typical patterns (Geist and Lambin, 2004; Scholes, 2009; D'Odorico et al., 2013; Polley et al., 2013; Ravi et al., 2010). The main assumption behind these typical patterns is that there is a limited set of biophysical and socio-economic factors, whose distinct patterns of interactions explain desertification. More recent efforts were focused on more spatially explicit clustering of different patterns of vulnerability in drylands (Sietz et al., 2011b; Kok et al., 2016; Sietz et al., 2017). Despite this progress in identifying dryland vulnerability typologies, the resulting considerable numbers of clusters and archetypes are not always mutually consistent, their translation into more context-specific national or sub-national policies and programs is not yet evident.

3.3. Observations of Desertification and Attribution

3.3.1. 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 Aridity Index (AI) (UNEP, 1992), which is the ratio of precipitation to potential evapotranspiration (Glossary). The AI thresholds for dryland climate classes as defined in Middleton and Thomas (1997) are: hyper-arid AI ≤ 0.05; arid 0.05 < AI ≤ 0.2; semi-arid 0.2 < AI ≤ 0.5; dry sub-humid 0.5 < AI ≤ 0.65. The AI decreased in many parts of the world over the last several decades and based on these constant AI thresholds this has been interpreted as expanding the extent of drylands in some regions (*robust evidence, high agreement*) (Feng and Fu, 2013; Asadi Zarch et al., 2015; Ji et al., 2015; Spinoni et al., 2015; Huang et al., 2016). The expansion of the drylands does not imply desertification by itself, if there is no long-term loss of the biological productivity of drylands, their ecological complexity, and/or their human values.

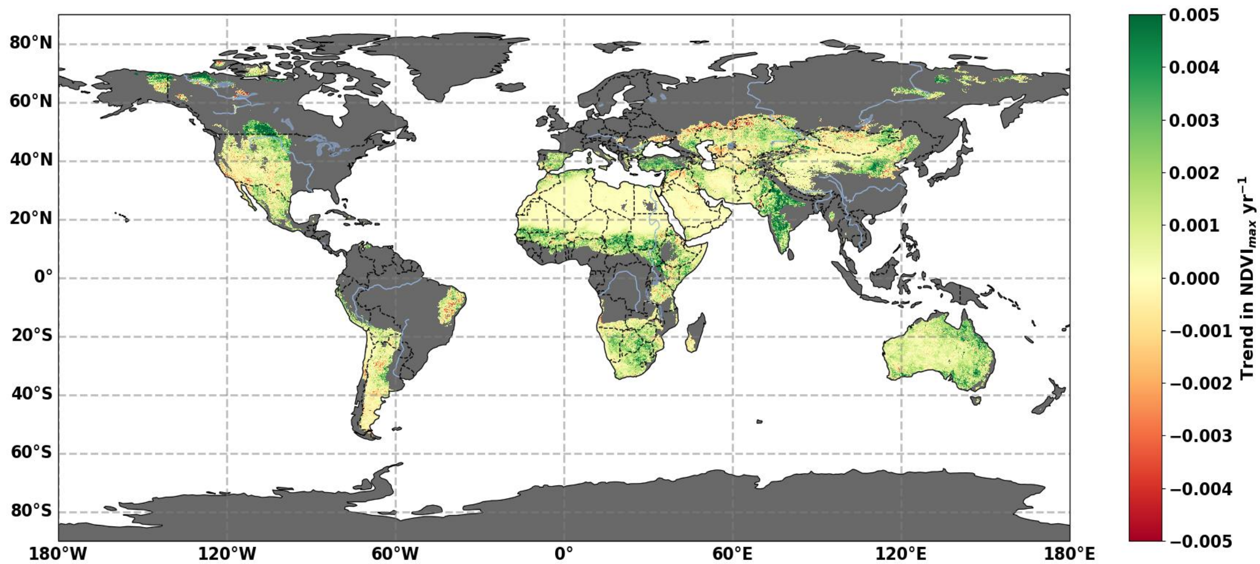

Figure 3.5 Trend in the Annual Maximum NDVI 1982-2015 (GIMMS NDVI3g v1) calculated using the Theil-Sen estimator which is a median based estimator, and is robust to outliers. Non-dryland regions (Aridity Index > 0.65) are masked in grey

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). 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). 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 at least partially explains 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 and, while not agreeing entirely with the aridity index, they also found a tendency toward drier climate types (Feng et al., 2014; Spinoni et al., 2015).

Depending on the definitions applied and methodologies used in evaluation, the status and extent of desertification globally and regionally still show substantial variations (D'Odorico et al., 2013). The four 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; see also Chapter 4).

3.3.1.1. Global Scale

Complex human-environment interactions coupled with biophysical, social, economic and political environments unique to any given location on drylands render desertification difficult to be mapped at a global scale (Cherlet et al., 2018). Early attempts to assess desertification focused on expert knowledge to achieve global coverage rapidly and cost-effectively. **Expert judgement** continues to play an important role because degradation remains a subjective quality whose benchmarks vary among locations (Sonneveld and Dent, 2007). On its initial quantification attempts, GLASOD (Global Assessment of Human-Induced Soil Degradation) estimated nearly 2 billion hectares (22.5% of the global land) had been degraded by early 1990s since mid-20th century. GLASOD was criticised for perceived subjectiveness and exaggeration (Helldén and Tottrup, 2008). Dregne and Chou (1992) found 3 billion ha in 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, using 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 course (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 Normalized Difference Vegetation Index (NDVI) providing a measure of canopy greenness, which is related to the quantity of standing biomass at a given point (Bai et al., 2008; de Jong et al., 2011; Fensholt et al., 2012; Andela et al., 2013; Fensholt et al., 2015; Le et al., 2016b, Figure 3.6). 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 (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 it is land degradation. All studies show a mixture of positive and negative NDVI trends, with positive trends dominating globally. According to this measure there are regions undergoing desertification, however, the drylands are greening on average (Figure 3.5).

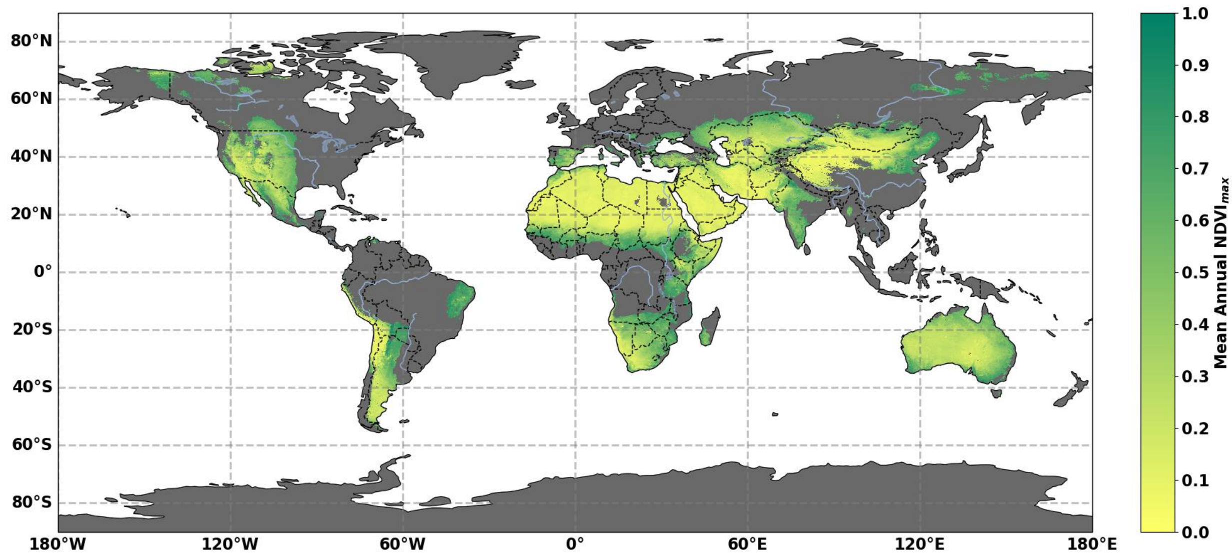

Figure 3.6 Mean Annual Maximum NDVI 1982-2015 (GIMMS NDVI3g v1). Non-dryland regions (Aridity Index > 0.65) are masked in grey

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., 2016b). The NDVI is strongly coupled to precipitation in the drylands and precipitation has high inter-annual variability. This means that the 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 (Verbesselt et al., 2010; de Jong et al., 2013), and methods to explicitly account for the effect of CO_2 fertilisation (Le et al., 2016b).

Using the RESTREND method, Andela et al. (2013) found that human activity contributed to a mixture of improving and degrading regions in the 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 west 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 unlikely to accurately capture the changes that have occurred in the majority of the drylands. Le et al. (2016b) 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. After also accounting for factors such as fertiliser use, Le et al. (2016) reported that about 29% of the global land area contained biomass-based land degradation hotspots.

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. For example, the Enhanced Vegetation Index (EVI) was designed to provide more information when dense vegetation is present and the NDVI signal can saturate. Studies have compared these 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.

Another vegetation index, Vegetation Optical Depth (VOD), has also been available since the 1980s. VOD is based on microwave measurements and is linearly 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 dynamics 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.

A major shortcoming of these studies based on vegetation datasets derived from satellite images 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 effectively 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 satellite have been made (Geerken et al., 2005; Evans and Geerken, 2006; Geerken, 2009; Verbesselt et al., 2010; Verbesselt, et al., 2010; Brandt et al., 2016a,b). These depend on well identified reference NDVI time series for particular vegetation groupings, and can only differentiate vegetation types that have distinct spectral evolution signatures and generally require extensive ground observations for validation. A recent alternative satellite based 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).

Biophysical models use global data sets 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). Terms used to describe marginal agricultural land are abandoned farmland, degraded land, wasteland and idle land. For example, Cai et al. (2011) mapped marginal agricultural land that can be utilised for biofuel production using a biophysical model of agricultural productivity based on spatial descriptions of soil groups, soil productivity, topography, average air temperature and precipitation, combined with expert opinions and global land cover datasets. According to Cai et al. (2011) marginal areas with low-productivity cropping were designated as abandoned, idle, or wasted, while marginal areas with fully utilised for agriculture designated as degraded. Uncertainties associated with this method arise from data limitations and spatial heterogeneities of socioeconomic conditions and agricultural technologies used.

Land productivity is a proxy for above ground Net Primary Productivity. The Land Productivity Dynamics (LPD) dataset shows land's capacity to sustain primary productivity. During the period from 1999 to 2013, primary productivity declines were observed on approximately 37% of the area of Australia, 27% of South America and 22% of Africa (UNCCD, 2017). According to UNCCD (2017), approximately 9% of global land area with more than 50% of cropland and 5% of global rangeland is exposed to between eight and 143 global change issues (GCIs) that trigger land change processes that are relevant to land degradation. According to Cherlet et al. (2018), Africa has more GCIs than any other continent with 76% of the total area having five to seven GCIs. The dominant GCIs are high population density and change, low income levels, fires, high livestock densities and fertiliser deficiencies. For Asia, 65% of low density cropland has between four and six GCIs with dominant GCIs being population density and change, high livestock densities, low income and water stress. Agricultural plains of Bangladesh and Myanmar, for example, are experiencing population pressure resulting in increased irrigation schemes and high livestock densities (Cherlet et al., 2018).

Overall, more efforts are required for improved estimations and mapping of desertified areas, using a combination of rapidly expanding sources of remotely sensed data and ground observations. 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 Sustainable Development Goals (SDGs).

3.3.1.2. Regional Scales

While global scale studies provide information for any region of interest, there are many studies that focused on sub-continental scales often using *in-situ* observations and providing more in-depth analysis and understanding. Regional studies are important and critical because effects of climate change and variability show varied characteristics in different climate regions and time scales. Here we discuss studies relevant for each UNCCD annex region.

3.3.1.2.1 Africa

It is estimated that desertification is affecting 46 of 57 nations in Africa (Prăvălie, 2016). Horn of Africa and parts of northern Africa experienced drying over the last three decades, whereas wetter conditions were experienced in central Africa and the Sahel (Damberg and AghaKouchak, 2014). Desertification in the Sahel has been a significant area of research since the 1970s, which in concert with a large scale drought at that time, culminated in the UN Convention to Combat Desertification in 1994. Significant changes have occurred in the landscapes of the Sahel region of West Africa with cropland areas doubling since 1975, and the settlement area increasing by about 150% (Traore et al., 2014). From satellite and rainfall data, a greening trend in the Sahelian belt has been observed since the 1980s (Huber et al., 2011; Brandt et al., 2015; Rishmawi et al., 2016; Tian et al., 2016; Leroux et al., 2017; Herrmann and Hutchinson, 2005). Greening in southern Africa has been observed too but it is relatively weak compared to other regions in the continent (Helldén and Tottrup, 2008; Fensholt et al., 2012). However, greening can also be accompanied by desertification due to factors such as decreasing species richness, changes in species composition and shrub encroachment (Mbow et al., 2013; Herrmann and Tappan, 2013; Kaptué et al., 2015; Herrmann and Sop, 2016). For example, some of the observed greening in Southern Africa has been associated with shrub encroachment (Saha et al., 2015). Soil loss through run-off is 16 times higher in bare degraded soils of the Sahel than in the sub-humid zones where soils are more structured. Moderate or higher severity degradation over recent decades has been identified in many river basins including the Nile (42%), the Niger (50%), the Senegal (51%), the Volta (67%), the Limpopo (66%) and the Lake Chad (26%) (Thiombiano and Tourino-Soto, 2007). Although many studies demonstrate that there was neither a progressive southwards extension nor large-scale expansion of less productive lands (e.g., Anyamba and Tucker (2005), Thomas and Nigam (2018) found out that the Sahara had expanded by 10% over the 20th century by taking a long-term perspective (Section 3.3.2).

In arid Algerian High Plateaus, desertification due to both climatic and human causes led to the loss of indigenous plant biodiversity and overall loss of vegetation between 1975 and 2006 (Hirche et al., 2011). The greening process for the Sahel region (Helldén and Tottrup, 2008) was not observed in the North African steppes. Ayoub (1998) identified 64 million hectares in Sudan as degraded, with the Central North Kordofan state being most affected. However, the 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).

Based on NDVI residuals computed by Gichenje and Godinho (2018), using annual mean data of the NDVI and soil moisture relationship, Kenya experienced persistent negative trends (browning) over 21.6% of the country, and persistent positive trends (greening) in 8.9% of the country for the period 1992–2015. Grasslands increased by 12,171 km^2, bare land decreased by 9,877 km^2 and forestland decreased by 7,182 km^2 during the same period. Habitat fragmentation, decline in pastoral grazing range, loss of wildlife dispersal areas and increase in livestock population density are considered to be the main drivers for vegetation structure loss in the northern rangelands of Kenya (Otuoma et al., 2009). For instance, in Meru conservancy, open wooded grasslands have decreased by 42% and bushland vegetation increased by 42% since 1980.

In Burkina Faso, Dimobe et al. (2015) estimated that from 1984 to 2001, tree savannahs, bare soils and agricultural lands increased by 17.55%, 18.79% and 21.79%, respectively, while woodland, gallery forest, shrub savannahs and water bodies decreased by 22.02%, 5.03%, 40.08% and 31.2%, respectively. From 2001 to 2013, gallery forests decreased by 14.33%, tree savannahs by 22.30% and shrub savannahs by 5.14%, while agricultural lands increased by 167.87% and woodlands by 3.21%. Desertification occurred at a higher rate in areas bordering Bontioli wildlife reserve compared to the protected and inaccessible areas.

In evaluating hydrological responses of land degradation on the Owena River basin in Nigeria, Aladejana et al. (2018) showed that between 1986 and 2015, 18.56% of the forest cover around the basin was lost of which 16.19% was converted to agricultural land. For the period 1982–2003, Le et al. (2012) found that 8% of the Volta River basin's landmass had been degraded with 65% of the land losing its soil quality and vegetation productivity.

In the Okavango river Basin in Southern Africa, conversion of land towards higher utilisation intensities, unsustainable agricultural practises and overexploitation of the savannah ecosystems have been observed in recent decades (Weinzierl et al. 2016).

3.3.1.2.2 Middle East and Europe

Drylands cover 33.8% of the lands of the Northern Mediterranean countries; approximately 69% of Spain, 66% of Cyprus, and between 16% and 62% in Greece, Portugal, Italy and France (Zdruli, 2011). The estimates from Rubio and Recatalá (2006) show that there are 30 million hectares of semi-arid drylands in the whole Mediterranean region. 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. Damberg and AghaKouchak (2014) found that parts of the Mediterranean region experienced drying over the last three decades, whereas wetter conditions were experienced in parts of eastern Europe. Helldén and Tottrup (2008) observed a greening trend in the Mediterranean between 1982–2003, while Fensholt et al. (2012) also show a dominance of greening in Eastern Europe.

Developed in the framework of the MEDALUS and DESERTLINKS projects, the Environmental Sensitivity Areas (ESA) approach has been used to estimate land vulnerability to desertification in the Mediterranean Europe (e.g., Contador et al., 2009; Salvati and Bajocco, 2011). The process assesses climate, soil, vegetation and land management to arrive at the Environmental Sensitivity index (ESI) (Ferrara et al., 2012). Other indices have also been developed in the European context (Santini et al., 2010; Kairis et al., 2014; Prăvălie et al., 2017). These indices provide guidance on locations where attention to sustainable land use practices is required to avoid possible future desertification. The European Environment Agency (EEA) indicated that 14 Mha, 8% of the territory of the European Union (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 included (Prăvălie et al. 2017; European Court of Auditors 2018).

Turkey is considered highly vulnerable to drought, land degradation and desertification (Türkeş, 1999; Türkeş, 2003). About 60% of Turkey's land area (i.e., of 5.77% semi-arid, 24.75% dry sub-humid and 28.54% moist sub-humid) is characterised with hydro-climatological conditions favourable for desertification (Türkeş, 2013). Consistent with these findings, ÇEMGM (2017) estimated that about half of Turkey's land area (48.6%) is under moderate to high desertification risk.

Desertification has increased substantially in Iran since the 1930s. Despite numerous efforts to rehabilitate degraded areas and combat desertification, it still poses a major threat to agricultural livelihoods in the country (Amiraslani and Dragovich, 2011). Ahmady-Birgani et al. (2017) showed a progressing sand dune movement and subsequent desertification in the Rigboland sand sea area in central Iran.

In north-west Jordan, three quarters of variation in soil erosion was related to topography, while the remaining share was due to wind erosion (Al-Bakri et al., 2016).

3.3.1.2.3 Asia

Prăvălie (2016) found that desertification is currently affecting 38 of 48 countries in Asia. Damberg and AghaKouchak (2014) found that northern India experienced drying over the last three decades. Helldén and Tottrup (2008) highlighted a greening up trend in East Asia between 1982 and 2003. 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., 2015).

Xue et al. (2017) used remote sensing (RS) images from four periods (1975, 1990, 2000, and 2015) to classify the intensity of wind-driven desertified land in north Shanxi in China, and found that desertification experienced three major development stages: slower expansion during 1975–1990 at a rate of 96.58 km^2 yr^{-1}, rapid expansion during 1990–2000, and a reversion during 2000–2015 with a net decrease. Throughout the 18th and 19th centuries, sandy desertification took place on the Mongolian Plateau, north-eastern China, and the Yellow River basin (Lamchin et al., 2016) due to shifts in monsoons and wind activity during the Little Ice Age with a significant increase in aridity observed in the Northern region (Hua et al., 2014).

Central Asian countries are facing a massive environmental catastrophe associated with the drying up of the Aral Sea due to anthropogenic causes (Micklin, 2007). The mean temperatures increased by 0.18°C per decade between 1901 and 2003 in the region (Chen et al., 2009), a rate twice the average over the northern hemisphere (Jones and Moberg, 2003). Precipitation was higher between 1930 and 2009 (Chen et al. 2011; Li et al. 2006). NDVI and gridded high-resolution land data analysis (1984–2013) showed that shrub and sparse vegetation density significantly decreased due to droughts in the Karakum and Kyzylkum Deserts, the Ustyurt Plateau and the wetland delta of the Aral Sea.

Desertification through salinisation is a major concern across the drylands in Asia as it impacts both food and water security and it's highly likely to be exacerbated by climate change (D'Odorico et al., 2013). Examples of 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, a major irrigation agriculture district in northwest China (Zhou et al., 2013), Aral Sea Basin of Central Asia (Cai et al., 2003).

3.3.1.2.4 Australia

Damberg and AghaKouchak (2014) found that wetter conditions were experienced in northern Australia over the last three decades. A widespread greening was identified between 1981 and 2006 over much of Australia, except for eastern Australia where large areas with decreases were present, based on Advanced High Resolution Radiometer (AVHRR) satellite data (Donohue et al., 2009). From 2002 to 2009 much of eastern Australia was affected by drought. For the period 1982–2013, Burrell et al. (2017) also found widespread greening over Australia and greening in 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. Burrell et al. (2017) also applied a RESTREND analysis to account for this precipitation influence finding that for most of the continent precipitation accounted for some of the vegetation increase, with some scattered regions experiencing degradation due to anthropogenic and other causes. The RESTREND methodology was extended to account for the non-monotonic nature of the vegetation change called Time Series Segmentation RESTREND (TSS-RESTREND). It was found that degradation due to anthropogenic and other causes was larger than otherwise predicted particularly near the central west coast, and affected just over 5% of Australia. 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.

3.3.1.2.5 Latin America and the Caribbean

In Latin America and the Caribbean, 25% of the total land area are drylands. Granados-Sánchez et al. (2012) estimated that 516 million hectares in in Latin America are susceptible to desertification, while Morales and Parada (2005) estimated about 378 Mha undergoing severe degradation. South and Central America have a total degraded land area of 300 Mha, with a decreasing trend in net primary productivity, and decline in ecosystem productivity (Zdruli et al., 2010). In Guatemala, the area undergoing desertification is estimated to be up to 12% of the total land area, especially in regions where deforestation is rampant, resulting from the expansion of the agricultural frontier based on subsistence agriculture (Morales and Parada, 2005). In Bolivia, Chile, Ecuador and Peru, between 27% and 43% of the total land area are affected by desertification. Around 77% of the Bolivia's population is living in degraded areas. Morales et al. (2011) showed that 75% of land in Argentina, 8% in Brazil (94% as per Vieira et al. (2015), 34% in Peru is undergoing some form of degradation. 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 degradation in the Patagonian Monte region of Argentina. The Caatinga region of western Brazil is estimated to have experienced widespread desertification with up to 50% of the area being degraded (Leal et al., 2005).

3.3.1.2.6 North America

Damberg and AghaKouchak (2014) found that south-western United States and Texas experienced drying over the last three decades. Using desertification trend risk index (DTRI) based on Landsat images, Becerril-Pina Rocio et al. (2015) showed that semi-arid regions of central and parts of western and southern Querétaro state in Mexico are severely degraded. Desertification in the form of shrub encroachment has been occurring over the last century in the Jornada Basin within the Chihuahuan Desert in New Mexico, USA (Rachal et al., 2012). This encroachment is observed over a fairly wide area of western North American grasslands and seems to spread at a faster rate despite grazing restrictions intended to curb the spread (Yanoff and Muldavin, 2008; Browning and Archer, 2011; Van Auken, 2009). Also, sand dune encroachment has been identified as a cause of desertification in California, USA (Lam et al., 2011). The major river basins of San Joaquin Valley and Colorado River Basin is undergoing salinisation (Qadir et al., 2007).

3.3.2. Attribution of Desertification

Desertification is a result of complex interactions within coupled social-ecological systems. Thus, the relative contribution of climatic, anthropogenic and other factors to desertification will vary depending on specific regional 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 overwhelm the vegetation ability to recover and cause 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. Climate change on the other hand, exemplified by a non-stationary climate, can gradually cause a persistent change in the ecosystem through aridification. Assuming no human influence, this 'natural' climatic version of desertification can take place over longer periods of time as the ecosystem slowly adjusts 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, the RESTREND (residual trend) method analyses the correlation between annual maximum NDVI (or other vegetation index) 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. Using this relationship, the climate component of the NDVI time series can be reconstructed, and the difference between this and the original time series is attributed to anthropogenic and other causes. Evans and Geerken (2004) applied the technique to the Syrian rangelands and found around twice as much area was being degraded by anthropogenic and other factors compared to examining the NDVI trends alone.

The RESTREND method, or minor variations of it, has been applied extensively. Herrmann and Hutchinson (2005) examined the African Sahel from 1982 to 2003. They found that climate was responsible for widespread greening, 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 to 2007 by Huber et al. (2011). Wessels et al. (2007) applied RESTREND to South Africa. They show that RESTREND produced a more accurate identification of degraded land than Rain Use Efficiency. In this case RESTREND identified a smaller area undergoing desertification due to anthropogenic and other non-climate causes compared to the NDVI trends. Li et al. (2012) used RESTREND to identify desertification in Inner Mongolia China. They show significant changes to the locations and extent of human-caused desertification in response to policy changes. Liu et al. (2013b) 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, and suggested that increases in goat density and wildfire occurrence are causal factors in those areas. RESTREND has also been applied in the Sahel (Leroux et al., 2017), Somalia (Omuto et al., 2010), West Africa (Ibrahim et al., 2015), China (Yin et al., 2014), Central Asia (Jiang et al., 2017) and Australia (Burrell et al., 2017). These studies represent the best regional, remote sensing based attribution studies to date, noting that RESTREND has some limitations.

One assumption in RESTREND is that any trend is linear throughout the period examined. That is there are no discontinuities or break points in the trend. To overcome this limitation, Burrell et al. (2017) introduced the Time Series Segmentation-RESTREND (TSS-RESTREND) which allows a breakpoint within the period examined. 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 Earth observation trend shifts in rain-use efficiency (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 or as the result of major droughts. Recent human-induced turning points in ecosystems functioning were uncovered nearby Volgograd (Russia) and around Lake Balkhash (Kazakhstan), respectively, attributed to recultivation, increased salinisation, and increased grazing.

Attribution of vegetation changes to human activity has also been done within modelling frameworks (Figure 3.7). In these methods ecosystem type 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 comparison performed at global scales 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 deposition, explaining 85% and 41% of the trend, respectively (Piao et al., 2015). 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).

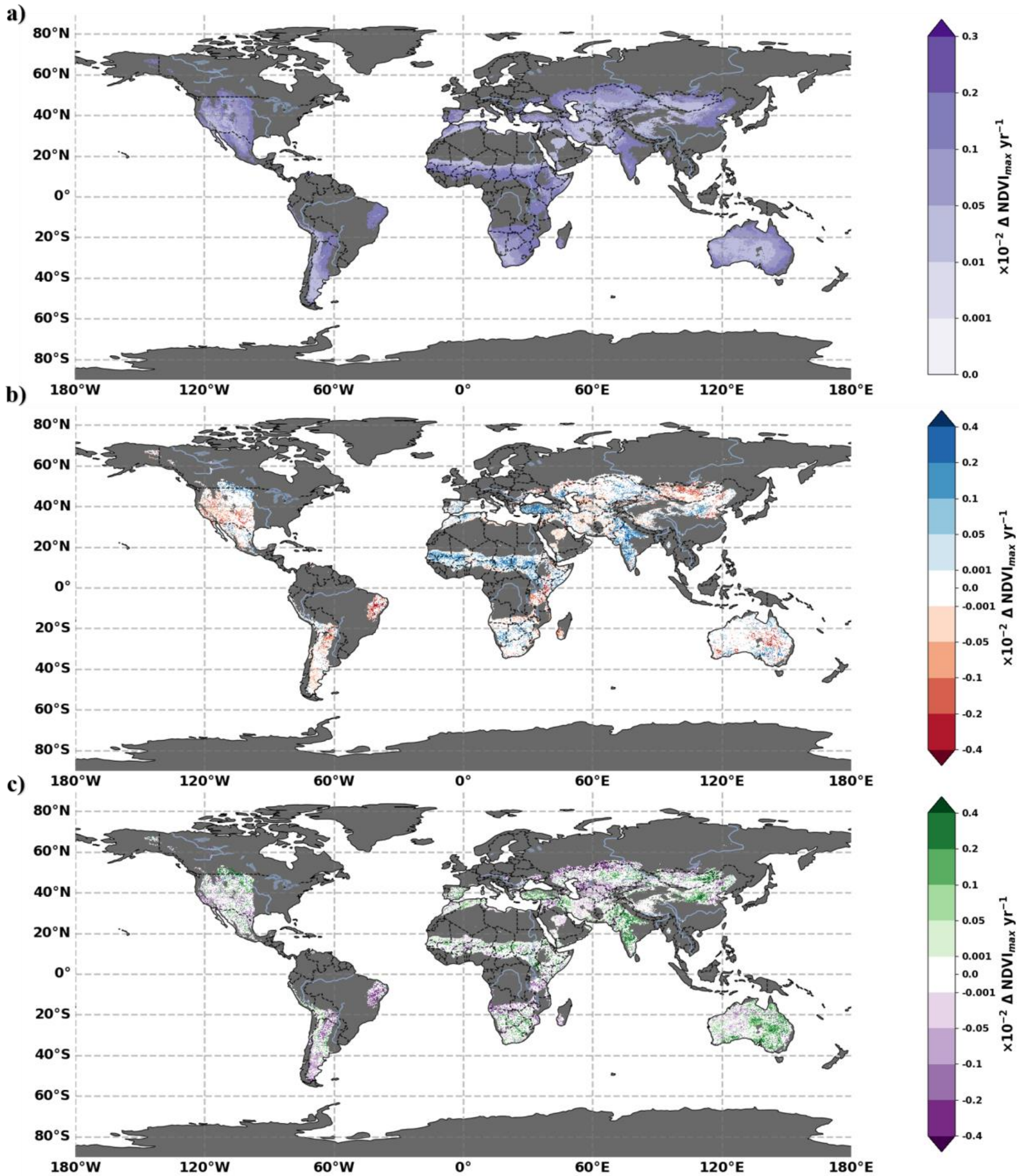

Figure 3.7 The Drivers of Dryland Vegetation Change. The mean annual change in NDVImax between 1982 and 2015 (See Figure 3.7 for total change using GIMMS NDVI3g v1 dataset) attributable to a) CO_2 fertilisation b) climate and c) land use. The change attributable to CO2 fertilisation was calculated using the CO2 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

Probabilistic event attribution (PEA) methodology showed that the dominant influence for droughts in Eastern Africa during 2016–2017 October November December 'short rains' season was the prevailing sea surface temperature patterns (La Niña in that case), although temperature trends indicate that the drought conditions were hotter than it would have been without climate change. There was no detectable trend in rainfall, but small changes in the risk of poor rains linked to climate change could not be excluded (Uhe et al., 2017). A strong warming tendency in the western Indian Ocean has been attributed to increases in greenhouse gasses (*medium confidence*) (Verdin et al., 2005; Williams and Funk, 2011).

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, natural climate cycles (the cold phase of Atlantic Multi-Decadal Oscillation and Pacific Decadal Oscillation) has been attributed to two-thirds of the observed expansion of the Sahara Desert from 1920–2003 (Thomas and Nigam, 2018). Drought is considered the main driver of desertification in Africa (Masih et al., 2014) especially in rangelands. However, other studies suggest that although droughts may contribute to desertification, the underlying causes are human activities, for instance, pressures on land in southern Mali are *likely* to have doubled background dust loads over the Atlantic Ocean since the mid-1960s (Aguirre Salado et al., 2012; Moulin and Chiapello, 2004; Section 3.4.1). Brandt et al. (2016b) found that woody vegetation trends are negatively correlated with human population density while 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 et al., 2008; Liu and Tian, 2010; Zhao et al., 2013, 2006). Evidence shows that socioeconomic factors were dominant in causing desertification in north Shanxi, China, between 1983 and 2012, accounting for about 80% of desertification expansion (Feng et al., 2015). Encroachment of shrubs into the northern Chihuahuan Desert (USA) since the mid-1800s is mainly attributed to overgrazing and nutrient depletion, which impedes successful grass establishment (Kidron and Gutschick, 2017). In Iran, human and climatic factors combined were attributed to severe droughts between 1950 and 2010 (Modarres et al., 2016). Human activities led to rangeland degradation in Pakistan and Mongolia during 2000-2011 (Lei et al., 2011). More equal shares of climatic and human factors were attributed for changes in rangeland improvement and degradation in China (Yang et al., 2016).

This kaleidoscope of local case studies demonstrate how attribution of desertification is still challenging, and this is due to several reasons. Firstly, desertification is caused by a combination of factors that change over time and vary by location. Secondly, in drylands, vegetation responds closely to rainfall fluctuations so the interaction between biomass change and rainfall trends 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.

Rasmussen et al. (2016) studied the generic reasons behind the overall lack of scientific agreement in trends of environmental changes in the Sahel supported by contrasting empirical evidence. The study distinguished between divergences in interpretations emerging from conceptualisations, definitions and choice of indicators, and biases, for example, related to selection of study sites, methodological choices, measurement accuracy, perceptions among interlocutors, and selection of temporal and spatial scales of analysis. 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 are context-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.

3.4. Desertification Feedbacks to Climate

Climate change and desertification have strong mutual interactions, and the land use and land cover changes associated with desertification contribute to climatic changes, whereas changes in precipitation, temperature, wind speed, and their variabilities due to climatic changes constitute factors affecting desertification (Sivakumar, 2007). These climate-desertification interactions require multi-faceted approaches to limit negative impacts on human wellbeing (Adeel et al., 2005; Archer and Tadross, 2009).

While climate change can drive desertification (3.2.4.1), the process of desertification can also alter the local climate providing a feedback. This feedback can lead to either a damping of the desertification process (negative feedback) or an enhancement of the desertification process (positive feedback). 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 are summarised in Figure 3.8.

Drylands are characterised by limited soil moisture availability compared to more humid regions. Thus, the sensible heat accounts for a higher proportion of the surface net radiation than latent heat 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.

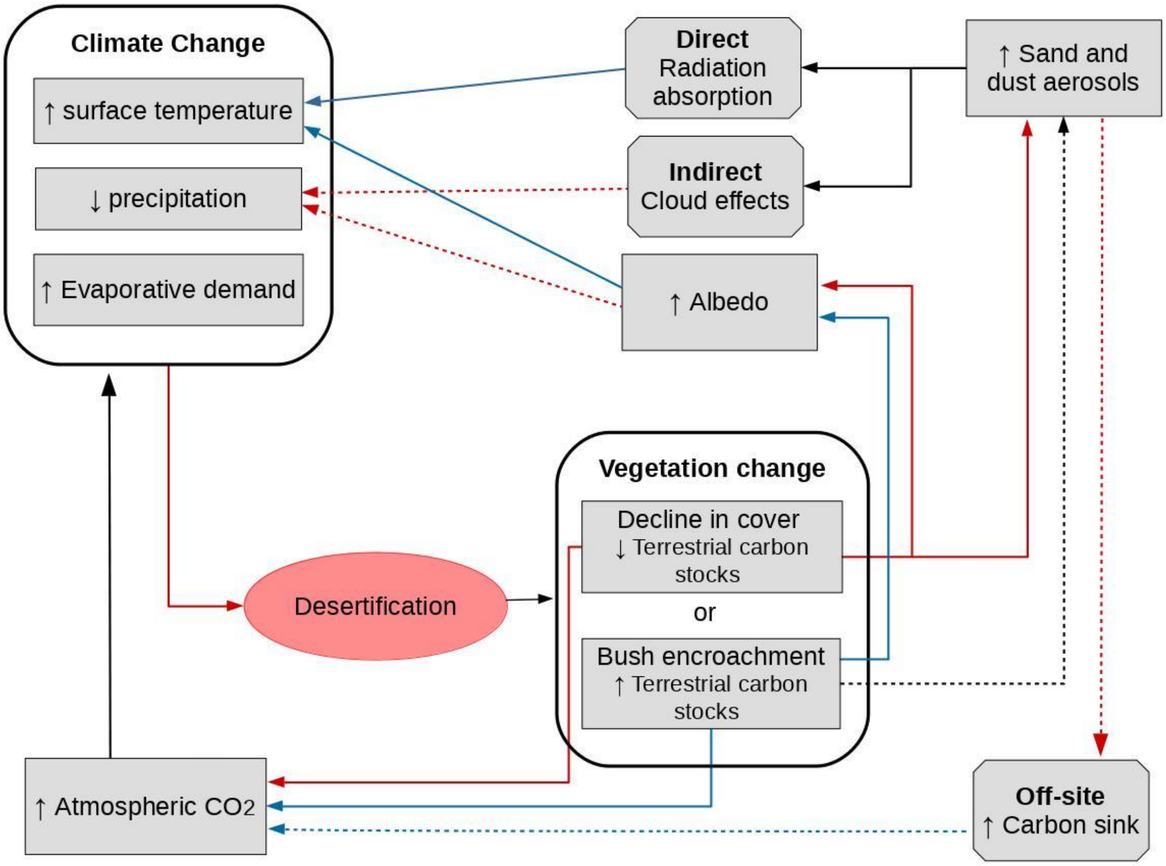

Figure 3.8 Schematic of main pathways through which desertification can feedback on climate. Note: red arrows indicate a positive effect. Blue arrows indicate a negative effect. Black arrows indicate an indeterminate effect (potentially both positive and negative). Solid arrows are direct while dashed arrows are indirect

3.4.1. Sand and Dust Aerosols

Sand and mineral dust are frequently mobilised from sparsely vegetated drylands forming “sand storms” or “dust storms”. The African continent is the most important source of desert dust, nearly 30% of atmospheric suspended dust comes from the Sahara (Gonzalez-Martin et al., 2014; Middleton, 2017). 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 soil crusts have been shown to effectively stabilise dryland soils and thus their loss due to intense land use and/or climate change can be expected to cause an increase in sand and dust storms (Field et al., 2010; Rodriguez-Caballero et al., 2018). These events 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). The heating of the dust layer can cause changes in the relative humidity and atmospheric stability, which can alter 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 rain. 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 in dryland regions (Elmore et al., 2008). This can occur through shallow 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 (e.g. Decker et al., 2013; Lamontagne et al., 2005; Patten et al., 2008). In these locations decreases in groundwater levels have the potential to decrease vegetation cover.

Desertification can decrease the amount of green cover and hence increase the occurrence of sand and dust storms. This would increase the amount of shortwave cooling associated with the direct effect. There is *high 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).

3.4.1.1. Off-site Feedbacks

Aerosols can act as a vehicle for the long-range transport of nutrients to oceans (Okin et al., 2011) and terrestrial land surfaces (Das et al., 2013). In several locations, notably the Atlantic Ocean, 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 dust on the ocean surface has been found to be a cooling effect (Doherty and Evan, 2014; 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. However, while some observational studies support this hypothesis (Lenes et al., 2001; Shaw et al., 2008), others find little or no response in the biological activity (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 cause increases in melt (Painter et al., 2018), impacting a region’s hydrological cycle.

3.4.2. Changes in Surface Albedo

The hypothesis that changing surface albedo in dryland regions will feedback on the local climate has been around since at least Charney et al. (1975). They used a climate model to show that over North Africa, an increase in albedo produced a decrease in available energy at the surface, a decrease in the surface temperature, a shallower planetary boundary layer and a reduction in precipitation. Since lower precipitation was associated with lower soil moisture and an increase in surface albedo, this represents a positive feedback.

More recent modelling work demonstrated this albedo feedback can occur in desert regions worldwide, including those outside the hyper-arid zone (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; Meng et al., 2014b), 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 (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 initial analysis, it was estimated that the change in surface albedo due to the degradation of semi-arid areas over the past few decades has decreased radiative forcing equivalent to approximately 20% of global anthropogenic GHG emissions to date (Rotenberg and Yakir, 2010).

3.4.3. 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 carbon dioxide during growth or releasing carbon during combustion and respiration, or through processes such as enteric fermentation of domestic and wild ruminants that lead to the release of methane and nitrous oxide (Sivakumar, 2007). 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 as well as soil organic carbon pools (Abdalla et al., 2018). While dryland ecosystems are often characterised by open vegetation, it should be noted that not all drylands necessarily 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 tonnes of Carbon per hectare (t C 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 areas with similar rainfall, may have only 5-35 t C 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 carbon dioxide, nitrous oxide and methane emissions from fire. In a similar manner, decreasing productivity may lead to a reduction in 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). For example, within the drylands of Texas, shrub encroachment led to a 32% increase in aboveground carbon stocks over a period of 69 years (3.8 t C ha^{1} to 5.0 t C 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 t C ha^{-1} over a 50–65 year period (1936–2001) (Hudak et al., 2003). In terms of potential changes in soil organic carbon stocks, the effect may be dependent on annual rainfall and soil type. Whereas increasing woody cover generally leads to an increase in soil organic carbon stocks in drylands that have less than 800 mm of annual rainfall, encroachment can lead to a loss of soil carbon in more mesic 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). In addition, increasing woody cover may lead to a decrease and even halt in surface fires and associated GHG emissions. In 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. Whereas 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 remains to be undertaken.

3.5. Impacts of Desertification on Natural and Socio-Economic Systems under Climate Change

3.5.1. Natural and Managed Ecosystems

3.5.1.1. Impacts on Ecosystems and their Services in Drylands

The Millennium Ecosystem Assessment (2005) proposed four classes of ecosystem services: provisioning, regulating, supporting and cultural services. 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). Desertification coupled with climate change negatively impacts provisioning services, particularly food and fodder production (Hopkins and Del Prado, 2007). Zika and Erb (2009) reported a rough estimation of Net Primary Productivity (NPP) losses between 0.8 and 2.0 Pg C yr^{-1} due to dryland degradation, comparing the potential NPP and the NPP calculated for the year 2000. Furthermore, desertification-climate change interactions modify the prevalence of livestock diseases, the composition of plant species and biological diversity (D'Odorico and Bhattachan, 2012; Thornton et al., 2009). Climate change, together with human population growth and global economic integration, is causing the abandonment of cattle rearing in favour of small ruminant husbandry as well as shifts into other forms of land use such as settled irrigated agriculture in East Africa (Homewood et al., 2001). Changes 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). Warm and humid conditions causing heat stress increase livestock mortality (Howden et al., 2008). On the other hand, fewer days with extreme cold temperatures during winters in the temperate zones are associated with lower livestock mortality. In addition, the ecosystem water availability is negatively affected by the combination of drought with increments in temperature at the late 20th and early 21st centuries; for example, (Woodhouse et al., 2010) estimated a reduction from 2-8% of the Colorado river runoff for each 1°C increment of temperature.

Among regulating services, desertification can influence levels of atmospheric carbon dioxide. In drylands, the majority of carbon is stored below ground in the form of biomass and soil organic carbon (SOC) (FAO, 1995). Drivers of soil degradation, mainly by land-use change, 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) and intensive grazing (Sharkhuu et al., 2016). In contrast, if the soil management includes soil conservation practices combined with irrigation, the cropland has a higher SOC content than native shrubland or native pastures, as shown in China (Liu et al., 2011). However, water management must be sustainable to ensure its availability for these results to persist. If restored, the degraded woodlands, grasslands, and deserts of the world could sequester up to 3.5 GtC yr^{-1} over this century (Yang et al., 2016), which is about 37% of 2017 fossil fuel carbon emissions (Le Quéré et al., 2018).

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). Low 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. 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 semiarid 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 and drier in the future with an increasing frequency of extreme drought and high rainfall events (Donat et al., 2016).

When desertification and climate change reduces vegetation cover below 25% (threshold that has a biological significance), this would alter the soil surface, affect the albedo and the water balance (Gonzalez-Martin et al., 2014). In such situations, the dust storms have no more obstacles, increasing the wind erosion. Mineral aerosols have an important influence on the dispersal of soil nutrients and lead to changes in soil characteristics (Peñate et al., 2013). Thereby, the soil formation as a supporting ecosystem service is negatively affected. Moreover, dust storms reduce crop yields by loss of plant tissue caused by sandblasting (resulting loss of plant leaves and hence reduced photosynthetic activity), exposing crop roots, crop seed burial under sand deposits, and leading to losses of nutrients and fertiliser from top soil (Stefanski and Sivakumar 2009). Dust storms also impact crop yields by reducing the quantity of water available for irrigation because it could 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).

3.5.1.2. Impacts on Biodiversity: Plant and Wildlife

3.5.1.2.1. Plant Biodiversity

Over 20% of global plant biodiversity centres are located within drylands (White and Nackoney, 2003). Furthermore, 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, 2005; Reynolds et al., 2007). 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). 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, where only drought-adapted species have by now survived on arid rangelands (Akhter and Arshad, 2006). Similar trends were observed in desert steppes of Mongolia (Khishigbayar et al., 2015).

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, xeromorphistic 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*) (Observatoire du Sahara et du Sahel, 2013).

3.5.1.2.2. Wildlife biodiversity

Dryland ecosystems have high levels of faunal diversity and endemism (Whitford, 2002; Millennium Ecosystem Assessment, 2005). Over 30% of the endemic bird areas are located within these regions, which is also home to 25% of vertebrate species (Millennium Ecosystem Assessment, 2005; Maestre et al., 2012). Yet, many species within drylands are threatened with extinction (Durant et al., 2014; Walther, 2016). Desert animal species have an array of adaptations to conserve body water and are able to withstand remarkably high body temperatures. Yet, perturbations from normal body temperatures are likely to represent a stress to these species (Hetem et al., 2016). 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 can have knock-on consequences for predators and may ultimately disrupt trophic cascades (Rey et al., 2017). Responses to desertification are likely to be species-specific and mechanistic models are not yet able to accurately predict individual species responses to the multitude of factors associated with desertification (Fuller et al., 2016).

3.5.2. Socio-economic Systems

The impacts of climate change-desertification interactions on socio-economic development in drylands are complex. Figure 3.9 schematically represents qualitatively our assessment of the magnitudes and the uncertainties associated with these impacts using the framework of Sustainable Development Goals (SDGs). The impacts of desertification and climate change are difficult to isolate from the effects of other socio-economic, institutional and political factors that simultaneously affect these dimensions of sustainable development (Pradhan et al., 2017). It is *very likely*, however, that climate change will exacerbate the already high vulnerability of dryland populations to desertification, and that the combination of pressures coming from climate change and desertification will amplify, in interaction with other contextual factors, poverty, food and nutritional insecurity, disease burden, lack of access to water and sanitation, and the likelihood of conflict (Sections 3.5.2.1 – 3.5.2.7). 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) represent the strong interactions between desertification-climate change interactions and achieving the targets of SDGs 13 and 15. The following sub-sections present the literature and the assessment which serves as the basis for Figure 3.9.

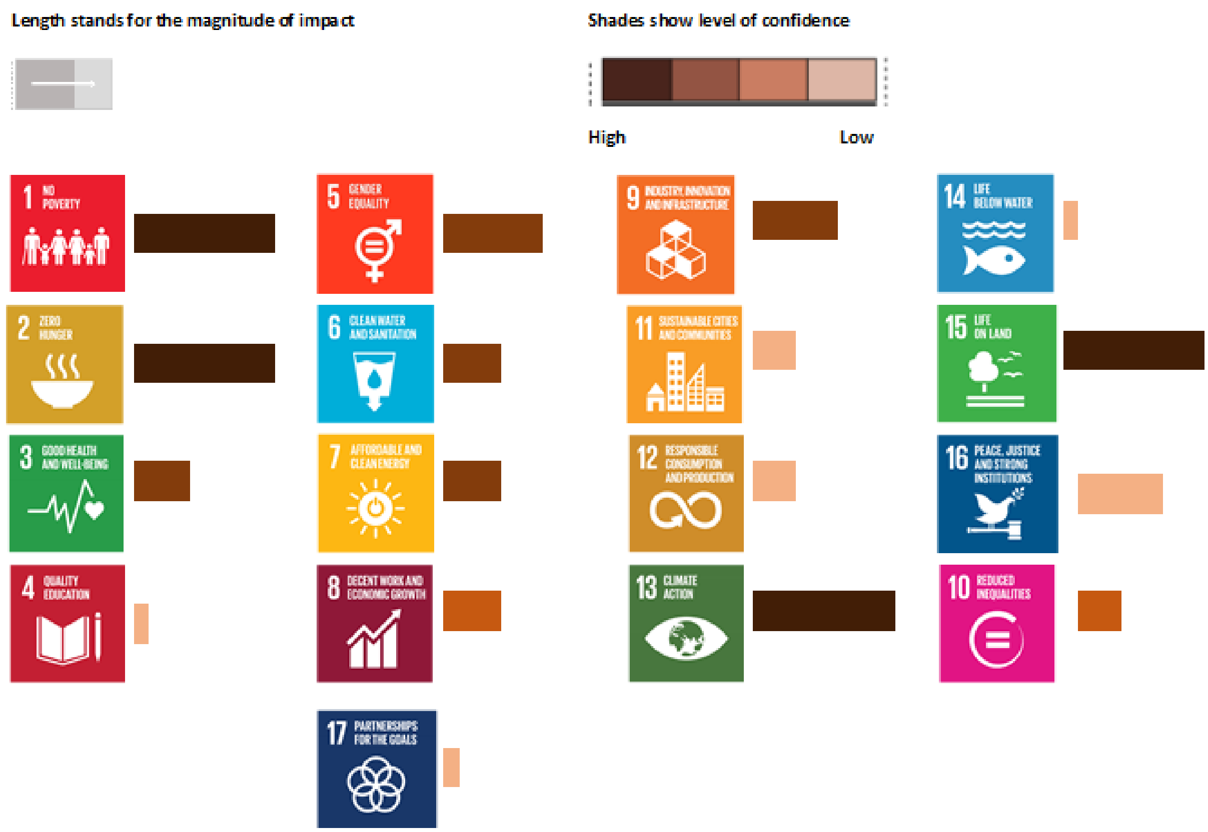

Figure 3.9 Socio-economic impacts of desertification and climate change with the SDG framework

3.5.2.1 Food and Nutritional Insecurity

About 815 million people globally were food insecure in 2016, of whom 62% are in Asia, 30% in Africa and 5% in Latin America and the Caribbean (FAO et al., 2017). Sub-Saharan Africa and South Asia had the highest share of undernourished populations in the world in 2016, with 22.7% and 14.4%, respectively. The drylands of Eastern Africa represent the global hotspot of food insecurity, where 33.9% of the population are undernourished (FAO et al., 2017). The climate change-desertification interactions affect all four dimensions of food security: availability, access, utilisation and stability (for more detailed discussion see Chapter 5). 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); Section 3.5.1; Chapter 5). 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, these numbers are global, not specific to desertification, and are estimated only for three crops. It is not clear to what level of food insecurity these losses translate. Despite a lack of global level estimates of the impacts of desertification on all the dimensions of food security, there is *robust evidence* 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). Negative impacts on crop yields and higher agricultural prices worsen existing food insecurity, especially for net-food buying rural households and urban dwellers. In contrast, there is a limited number of studies quantitatively tracing desertification impacts on stability and utilisation dimensions of food security.

Overall, there is *robust evidence and high agreement* for the high potential negative impact of climate change on agricultural productivity in drylands (Hochman et al., 2017; Mendelsohn, 2008; Nelson et al., 2010; C. Zhao et al., 2017, chapter 5). Although there is a lack of estimates on the aggregate impact of desertification on food security in drylands, local case studies point to significant losses of agricultural productivity and production due to desertification (Kirui, 2016; Moussa et al., 2016; Mythili and Goedecke, 2016; Tun et al., 2015). 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.

3.5.2.2 Poverty

The relationship between desertification and poverty, understood from multidimensional perspectives (Section 3.2.3), is often conceptualised as a vicious cycle, where desertification and poverty cause each other. This is because poor households have a higher dependence on environmental income (Thondhlana and Muchapondwa, 2014), so any degradation of the environmental resource base exacerbates their poverty, in the worst cases trapping them in poverty (Lybbert et al., 2004). Risk-averseness among poor households, in this context, is considered to lead to under-investment into sustainable land management practices (Teklewold and Kohlin, 2011), contributing to desertification and also making them more vulnerable to climate change. It is *very likely* that climate change will have substantial impacts on poverty in drylands (Hallegatte and Rozenberg, 2017; Hertel and Lobell, 2014). The impacts of climate change on poverty vary significantly 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, Philippines and Thailand, because higher prices compensated for productivity losses (Hertel et al., 2010).

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). However, the literature quantifying to what extent desertification contributes to poverty *per se* is thin, that is beyond the impacts of desertification on agricultural productivity and incomes. Moreover, at the global scale, there is little quantified evidence causally linking desertification to poverty, as the related literature remains qualitative or correlational (Barbier and Hochard, 2016). At the local level, on the other hand, there is *medium evidence* quantifying the impacts of desertification on multidimensional poverty. For example, it was found that land degradation decreased agricultural incomes in Ghana by 4.2 billion USD between 2006 and 2015, increasing the national poverty rate in 2015 by 5.4% (Diao and Sarpong, 2011). Land degradation increased the probability of household poverty 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), using a case study from Inner Mongolia in China, indicated that desertification is positively related to growing incomes in the short run, while in the long run higher incomes help in reducing desertification. This relationship corresponds to the Environmental Kuznets Curve, which hypotheses that environmental degradation initially rises and subsequently falls with rising income (Stern, 2017). There is *limited evidence* on the validity of this hypothesis regarding desertification.

3.5.2.3 Pastoral Communities

Pastoral production systems occupy a significant portion of the world (Rass, 2006; Dong, 2016). Due to frequent droughts and conflicts, aggravated by climate change, pastoral households are becoming more food insecure (Gomes, 2006). The Sahelian droughts of the 1960s show an example of how droughts could inflict a heavy toll on livestock resources and crop productivity, resulting in 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 the desertification processes, leading to ecological changes that forced people to use marginal lands and ecosystems. Similarly, the rate of rangeland degradation is increasing nowadays because of environmental changes and overexploitation of the 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 (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); and led to degradation of resources and out-migration of people in search of better livelihoods (Gebeye, 2016; Liao et al., 2015; Yeh, 2009). Restrictions on the mobile lifestyle is reducing the resilient adaptive capacity of pastoralists to natural calamities 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) is threatening pastoral livelihoods. For example, commercial cotton 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). Many pastoralists are having to switch to other forms of land uses such as settled irrigated agriculture, shifting from rearing cattle to rearing small ruminants because of climate change, desertification, human population growth, and as a result of global economy integration (Homewood et al., 2001). The sustainability of pastoral systems is therefore coming into question because of social and political marginalisation of the system (Davies et al., 2016) and also because of the fierce competition it is facing from other livelihood sources such as crop farming (De Haan et al., 2016). Moreover, the impacts of future climate change and desertification could be unprecedented in scale (Galvin, 2008), affecting political stability (Raleigh 2010), resulting in continued displacement and out-migration of pastoral communities into other geographical areas (Bassett and Turner, 2007).

3.5.2.4. 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). Moreover, secondary salinisation in the irrigated drylands often requires annual leaching with considerable amounts of water (Greene et al., 2016; Wichelns and Qadir, 2015). All these processes reduce water availability for other needs. In this context. climate change is *likely* to intensify water scarcity in many dryland areas and increase the severity and frequency of droughts (IPCC, 2013; Section 3.8.5). Higher water scarcity will imply growing use of wastewater effluents for irrigation (Pedrero et al., 2010). The use of untreated wastewater is *likely* to exacerbate desertification processes (Tal, 2016; Singh et al., 2004; Qishlaqi et al., 2008; Hanjra et al., 2012), with negative human health impacts (Faour-Klingbeil and Todd, 2018; Hanjra et al., 2012). Climate change, thus, will amplify the need for integrated land and water management for sustainable development.

3.5.2.5. Gender-differentiated Impacts

Environmental issues such as desertification and impacts of climate change have been increasingly investigated through a gender lens because of the significant differences of men and women (Bose, 2015; Broeckhoven and Cliquet, 2015; Kaijser and Kronsell, 2014; Kiptot et al., 2014; Villamor and van Noordwijk, 2016). These differences emanate from socially structured gender-specific roles and responsibilities, daily activities, access and control over resources, decision making and opportunities that lead men and women to interact differently with natural resources and landscapes. However, it is recognised that women will be impacted more than men by environmental degradation (Arora-Jonsson, 2011; Gurung et al., 2006).

Despite these known differences between men and women, gender issues have been marginally addressed in many conservation efforts which often remain gender-blind particularly in land restoration and rehabilitation efforts. Assessments of the gender dimension of desertification and climate change impacts are still very scarce, particularly at the macro or landscape level, because of two main reasons. First, because the impacts of climate change on men and women and their responses to desertification are context specific (or place-based). For example, in the drylands of Sub-Saharan Africa, the most common significant predicted consequence of climate change is the overall decrease in available water. Women, who are primary natural resource managers and providers of food security in the region, are often expected to fetch water and to collect fuelwood from increasingly remote areas (Mekonnen et al., 2017; Scheurlen, 2015). Whereas, men migrate to nearby towns or other countries for better opportunities, leaving women behind with more responsibilities. And yet, women are usually excluded from local decision making on actions regarding desertification and climate change. On the other hand, the comparative case study from Southeast Asian countries showed that women, due to their increasing productive roles, could become agents of either land degradation or restoration (Catacutan and Villamor, 2016). Second, these 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 (i.e., 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 problem is crucial.

3.5.2.6 Conflicts

Degradation of the natural resource base and ecosystem services in drylands due to climate change-desertification interactions amplify, in interaction with contextual factors, some conflicts in some regions (*medium evidence*, *medium agreement*). The related triggers of conflicts, to which desertification and climate change feed, are higher food prices (Arezki and Bruckner, 2011; Bush, 2010; Bellemare, 2015), droughts (Gleick, 2014; Hsiang et al., 2013; Hsiang and Meng, 2014; Maystadt and Ecker, 2014; Mehta, 2017), competition between pastoralist communities and crop producers for access to land (Abbass, 2014; Benjaminsen et al., 2012; Huho et al., 2011). There is *high confidence* that desertification and climate change do not cause conflict and civil strife by themselves, but add to the overall conflict potential. For example, the likelihood of new civil conflicts throughout the tropics were found to double during El Niño years relative to La Niña years. The El Niño–Southern Oscillation (ENSO) was suggested to have contributed to 21% of all civil conflicts since 1950 (Hsiang et al., 2011). Each one standard deviation increase in temperatures or rainfall was found to increase interpersonal violence by 4% and intergroup conflict by 14% (Hsiang et al., 2013). Similarly, a one-standard deviation increase in dryness was found to raise the likelihood of riots in Sub-Saharan African countries by 8.3% during the 1990–2011 period (Almer et al., 2017). It was also suggested that drought played a direct role in the Syrian conflict (Gleick, 2014), where drought, considered to be the longest and most intense in the last 900 years (Cook et al., 2016), considerably decreased crop yields and displaced hundreds of thousands people (Kelley et al., 2015; Trigo et al., 2010). However, the attribution of this drought in Syria to climate change is challenged (Selby et al., 2017, see also Chapter 5, section 5.6.2.1). On the other hand, droughts and heatwaves were not found to significantly affect the level of regional conflict in East Africa (Owain and Maslin, 2018). The droughts and desertification in the Sahel are *likely* to have 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). Similarly, the role of environmental factors as the key drivers of conflicts were 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. The emerging consensus is that climate change, with its interactions with desertification, is only a contributing factor to some conflicts in some regions (Butler and Kefford, 2018; Gleick, 2014; Kelley et al., 2015; Raleigh, 2010), where there may already be a conflict potential due to other reasons (Adano et al., 2012), for example, ethnic divisions (Schleussner et al., 2016) or widespread availability of small arms (e.g. after the fall of the Idi Amin government in Uganda) (Mkutu, 2007).

3.5.2.7 Migration

Environmentally-induced migration is complex and should account for multiple drivers of mobility as well as other adaptation measures undertaken by populations exposed to environmental risk (*high agreement*). This nuanced view is in stark contrast with the forecasts of large-scale environmental displacements suggested especially by the early literature (Myers, 2002; Myers and Kent, 1995), but reiterated in the recent World Bank report predicting that by 2050, more than 143 million people would be forced to move internally if no climate action is taken (World Bank, 2018). In a similar vein, Missirian and Schlenker (2017) predict that under continued future warming, by the end of the 21st century, the asylum applications to the European Union will increase by 28% up to 188% depending on the climate scenario. However, even though the modelling efforts have greatly improved over the years (Hunter et al., 2015; McLeman, 2011; Sherbinin and Bai, 2018), the estimates 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). Accordingly, the available micro-level evidence suggests that climate-related shocks are one of the drivers of migration (London Government Office for Science and Foresight, 2011; Melde et al., 2017; Adger et al., 2014), but the individual responses to climate risk are more complex than commonly assumed (Gray and Mueller, 2012). For example, despite strong focus on natural disasters, neither flooding (Gray and Mueller, 2012; Mueller et al., 2014) nor earthquakes (Halliday, 2006) induce mobility; but instead, slow-onset changes, especially those provoking crop failures and heat stress, do affect household or individual migration decisions (Gray and Mueller, 2012; 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) and indeed, a substantial body of literature suggests that households engage in local or internal migration as a response to drought (Findlay, 2011; Gray and Mueller, 2012), while international migration decreases with drought in some contexts (Henry et al., 2004), but might increase international mobility in contexts where migration networks are well established (Feng et al., 2010; Nawrotzki and DeWaard, 2016; Nawrotzki et al., 2015; Nawrotzki et al., 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 household's financial capacity to undertake migration (Nawrotzki and Bakhtsiyarava, 2017), especially in the case of the poorest households (Koubi et al., 2016; Kubik and Maurel, 2016; McKenzie and Yang, 2015).

3.5.2.8 Dust Storms and Human Health

The frequency of dust storms is increasing due to land use and climatic changes in some regions of the world (Gu et al., 2010; Indoitu et al., 2015; Rashki et al., 2012; Tan et al., 2012; Türkeş, 2017) (*high confidence*). There is *robust evidence and high agreement* that dust storms have negative impacts on human health (Díaz et al., 2017; Goudarzi et al., 2017; Goudie, 2014; Samoli et al., 2011). In view of growing intensity, frequency and scale of dust storms due to climate change-desertification interactions, these health impacts are *very likely* to increase in the future. 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.

Dust storms transport particulate matter, pollutants and potential allergens that are dangerous for human health over long distances (Goudie and Middleton, 2006; Sprigg, 2016). Particulate matter (PM), i.e. the suspended particles in the air having sizes between 10 micrometer (PM10) and 2.5 micrometer (PM2.5) or less, 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). If 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 reason for 15–50% of all cardiopulmonary deaths. A 10μgm^{-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). A review of impacts of Saharan dust storms on Europe showed no significant association between fine particles (PM2.5) and total or cause- specific daily mortality (Karanasiou et al., 2012). However, a conclusion on the health impact of coarser fractions (PM10 and PM2.5–10) could not be reached (Karanasiou et al., 2012). In Kermanshah, Iran, 92% of morbidity and mortality cases happened during days with PM10 concentrations lower than 150 μg/m^3, with the highest health damage occurring in the range of 100–109 μg/m^3 of PM10 concentrations. The percentage of all-cause deaths attributed to fine particulate matter in Iranian cities affected by Middle Eastern dust storms (MED) were 0.56–5.02%, while the same percentage for non-affected cities were 0.16–4.13% (Hopke et al., 2018). In case of lung cancer deaths, the percentage of deaths attributed to fine particles in MED-affected cities were between 13.74% and 26.47%, that was higher than those for cities with anthropogenic air pollution (Hadei et al., 2017). The Meningococcal Meningitis epidemics occur in the Sahelian region during the dry seasons with dusty conditions (Molesworth et al., 2003). Despite a strong concentration of dust storms in the Sahara region, the Middle East and Central Asia, there is relatively little research on human health impacts of dust storms in these regions.

3.5.2.9 Dust Storms and Impacts on Transport Infrastructure

Sand storms and movement of sand dunes threaten the safety and operation of railway and road infrastructure in arid and hyper-arid areas, and lead to road and airport closures due to reductions in visibility. 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 concentration 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 in Linhai-Ceke line in north-western China and 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 countries of the Arab Gulf Cooperation Council (GCC), or Lamu Port, South Sudan, Ethiopia Transport 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.

3.5.2.10 Dust Storms and Impacts on Energy Infrastructure

There is *robust evidence and high agreement* that dust depositions during dust storms negatively affect the operational potential of solar power generating equipment and can reduce effective electricity distribution in high-voltage transmission lines (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). The extent of the impact depends on numerous factors such as frequency and intensity of dust depositions, dust particle size and morphology, tilt angles of solar power installations, and exposure period (Jiang et al., 2011; Sarver et al., 2013). Direct exposure to desert dust storm can reduce energy generation efficiency of solar panels by 70–80% in one hour (Ghazi et al., 2014), whereas even in relatively dust free areas such as United Kingdom (UK), one month without cleaning of solar panels could reduce their energy generation by 5–6% (Ghazi et al., 2013). This has important implications for climate change mitigation efforts using the expansion of solar 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, thus, imposes additional costs for climate change mitigation through deployment of solar energy generation in dryland areas. Most frequently used solutions to this problem involve physically wiping or washing the surface of solar devices with water. However, these result in additional costs in terms of already scarce water resources and labour (Middleton, 2017).

3.6. Future Projections

3.6.1. 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 and their effect on a few processes or drivers that trigger desertification.

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*) (Zarch et al., 2015; Cook et al., 2014; Dai, 2011; Dominguez et al., 2010; Feng and Fu, 2013; Ficklin et al., 2016; Greve and Seneviratne, 1999; Lin et al., 2015; 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 an approximately 10% increase in hyper-arid areas globally (Zeng and Yoon, 2009). 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 sub-tropical dry zones and hence an expansion in drylands (Scheff and Frierson, 2012). Increases in PET are projected to continue due to climate change (Cook et al., 2014; Fu et al., 2016; Lin et al., 2015; Scheff and Frierson, 2015), decreasing the aridity index (AI), and this was interpreted as an expansion of global dryland areas by approximately 10% by the end of this century under RCP8.5 (Feng and Fu, 2013).

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*). IPCC 1.5°C (IPCC, 2018) report 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. (2015) 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.

The projected increases in the aridity index have *very high confidence*. However, several studies have challenged the use of the AI, and PET more generally, as reliable measures of the amount of moisture available to sustain life in terrestrial ecosystems (Greve et al., 2017; Milly and Dunne, 2016). Work on CO_2 fertilisation suggested that PET is not a good indicator of water use as increasing CO_2 increases plant water use efficiency and hence there is more plant productivity with less evapotranspiration (Lemordant et al., 2018; Swann et al., 2016). Roderick et al. (2015) and Greve et al. (2017) showed that in climate models, evapotranspiration does not increase at the same rate as PET and suggest that the AI is not a good measure of aridity in an environment with changing atmospheric CO_2. Evidence from precipitation, runoff or photosynthetic uptake of CO_2 suggest that a future warmer world will be less arid. This indicates that constant AI thresholds used to define climate classes may not be the right approach in a changing climate.

Climate strongly affects the mechanisms that explain wind erosion driven desertification. Wang et al. (2009) assessed future wind erosion driven desertification in arid and semiarid China using a range of SRES scenarios and HadCM3 simulations. The majority of scenarios showed a decrease in desertification by 2039, increasing thereafter.

Global estimates of the impact of climate change on soil salinisation show that under the IS92a emissions scenario 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 4.11.6 in Chapter 4), and surface hydrology and vegetation that affect wetlands and favour salinisation (Nielsen and Brock, 2009).

3.6.1.1. Future Vulnerability and Risk to Desertification

Following the conceptual framework developed in previous IPCC assessments future risks are assessed by examining changes in exposure (i.e., presence of people, livelihoods and/or ecosystems, see Glossary), changes in vulnerability (predisposition to be adversely affected, see Glossary) and changes in the nature and magnitude of hazards (climate 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). Droughts also increase vulnerability to land degradation in arid zones. An assessment by Carrão et al. (2017) showed an increase in global drought hazards by mid-(2021–2050) and late-century (2071–2099) compared to a baseline (1971–2000) under all RCPs in global Mediterranean ecosystems and the Amazon region. 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. Globally, climate change is predicted to intensify the occurrence and severity of droughts (*medium evidence*, *high agreement*) (Dai, 2013; Sheffield and Wood, 2008; Swann et al., 2016; Wang, 2005; Zhao and Dai, 2015). Ukkola et al. (2018) showed large discrepancies between CMIP5 models for all types of droughts, with only precipitation drought metrics having enough agreement to provide confidence in model projections.

Drylands are characterised by the 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, it is estimated that under RCP4.5 areas of increased vulnerability to climate change and land desertification will largely surpass those that will experience decreased vulnerability (Liu et al., 2016).

3.6.2. Future Projections of Impacts

Future climate change is expected to affect the potential for increased soil erosion. 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% till 2050 (Zare et al., 2016). In northern Australia, a decrease or an increase in rainfall post 2030 will influence the erosion rates and erosion patterns (Serpa 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).

Rodriguez-Caballero et al. (2018) analysed the cover of biological soil crusts under current and future environmental conditions utilizing an environmental niche modelling approach. Their results suggest that biological soil crusts currently cover about16 million km^2 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 being equally relevant in this process. The predicted loss is expected to substantially reduce their contribution to nitrogen cycling and to enhance dust emissions.

Potential dryland expansion implies lower carbon sequestration and higher risk of desertification (Huang et al., 2017), with severe impacts on land usability and threatening food security. At the level of biomes, soil carbon uptake is determined mostly by weather variability. The area of the land surface in which dryness controls CO_2 exchange has risen by 6% 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). 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.

World Bank (2009) projected that, without the carbon fertilisation effect, climate change will reduce the mean yields for 11 major global crops, such as millet, field pea, sugar beet, sweet potato, wheat, rice, maize, soybean, groundnut, sunflower, and rapeseed, by 15% in Sub-Saharan Africa, by 11% in Middle East and North Africa, by 18% in South Asia, and by 6% in Latin America and Caribbean by 2046–2055, compared with 1996–2005. A separate meta-analysis suggested a similar order of 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 (Chapter 5). 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).

Desertification under climate change will threaten biodiversity in drylands (*low confidence*). 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 southern California deserts climate change-driven extreme heat and drought may surpass the survival thresholds of some desert species (Bachelet et al., 2016). In semiarid Mediterranean shrublands in eastern Spain, plant species richness and plant cover are 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 the 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 climate change could cause the loss of 17% of species within shrubland and 8% within hot deserts by 2050 (van Vuuren et al., 2006) (*low confidence*). A study in the semi-arid Chinese Altai Mountains showed that mammal species richness will decline and 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 abiotic controls of desertification (Fischlin et al., 2007). In a 2℃ world, annual water discharge is projected to decline and heatwaves are projected to pose risk to food production by 2070 (Waha et al., 2017). The forecasts for Sub-Saharan Africa point to higher temperatures, increase in the number of heatwaves, and increasing aridity will affect the rain-fed agricultural systems (Serdeczny et al., 2017). A study by Wang et al. (2009) in arid and semiarid 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).

3.7. Responses to Desertification under Climate Change

Increasing population pressures and potentially unprecedented nature of climatic changes could push dryland populations beyond their resilience thresholds, requiring policy and technology interventions aimed at maintaining and strengthening their resilience and adaptive capacities, and preventing them from following development trajectories consistent with the "desertification paradigm". This section of the chapter considers each of these response options, starting from technological innovations and sustainable land management practices, through social responses undertaken by dryland households and communities at micro-level, and finally to broader policy responses.

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 global and local ecosystem benefits and importance to human livelihoods and economies (Thomas, 2008).

A broad suite of on the ground response measures exist to address the syndromes of 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). However, firstly, it is recognised that such actions require financial, institutional and policy support to remain sustainable over the long-term (Stringer et al., 2007). Secondly, actions need to be considered as part of coupled socio-economic systems in the broader context of dryland development and long-term SLM (Reynolds et al., 2007a; Stringer et al., 2017).

A description and assessment of predominant on the ground actions and forms of supporting planning and policy focusing on drylands is made here in Chapter 3. The response section in Chapter 4 considers the broader process of developing SLM together with high-level responses to land degradation. Chapter 6 then considers potential interlinkages in the form of co-benefits, trade-offs and synergies of SLM measures.

3.7.1. Technologies and SLM Practices: on the Ground Actions

A broad range of activities and measures can potentially avoid, reduce and reverse degradation across the dryland areas of the world. Many of these actions are also contribute to climate change adaptation and mitigation goals, with further sustainable development co-benefits. An assessment is made of six activities and measures that have historically been widely considered across the biomes and anthromes of the dryland domain (Figure 3.10). The 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). 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 explored further in Chapters 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.

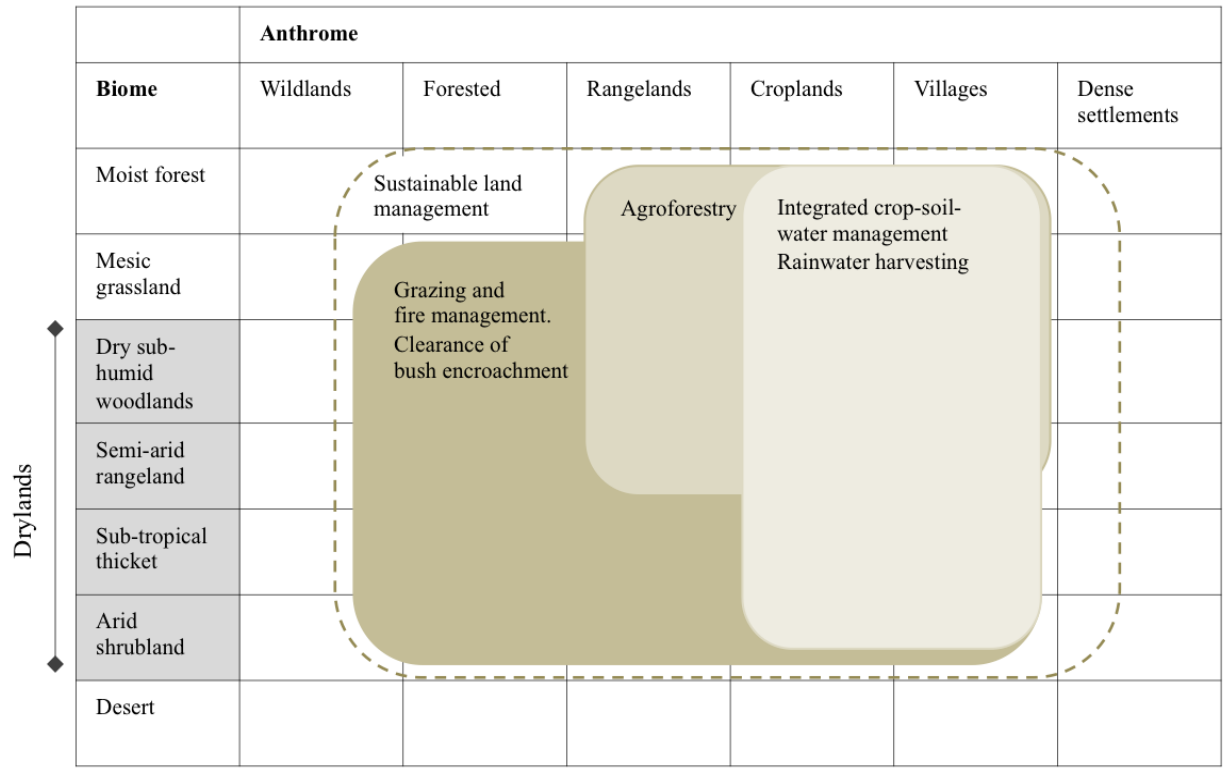

Figure 3.10 The typical distribution of on the ground actions across global biomes and anthromes

3.7.1.1. Integrated Crop-Soil-Water Management

Forms of integrated cropland management have been practiced in drylands for over thousands of years (Knörzer et al., 2009). Actions include planting a diversity of species, reducing tillage, applying organic compost and fertiliser, and maintaining vegetation and mulch cover.

In the contemporary era, these actions have been adopted within the context of SLM and 'conservation agriculture' to address desertification and the impact of climate change. Conservation agriculture provides a range of benefits through potentially mitigating climate change and improving agricultural production, water quality and the conservation of biodiversity (Dumanski et al., 2006). In particular, conservation agriculture helps to conserve and improve the health of topsoil that is predisposed to erosion and is essential for production of crops and further ecosystem services (Derpsch, 2005).

There is *robust evidence and medium agreement* that various cropping methods including, changing crop rotations, intercropping (inter- and intra- row planting of companion crops) and relay cropping (temporally differentiated planting of companion crops) can be used to increase production, increase the diversity of species, maintain cover over a larger fraction of the year, increase soil nitrogen and decrease the abundance of pests (Altieri and Koohafkan, 2008; Tanveer et al., 2017; Wilhelm and Wortmann, 2004; Zhang et al., 2007). For example, intercropping maize and sorghum with *Desmodium* (an insect repellent forage legume) and *Brachiaria* (an insect trapping grass) is being promoted in drylands of east and central Africa as climate-smart and profitable compared to conventional practices (Khan et al., 2014). This practice led to the productivity of maize increasing by two-three fold, while over 80% of stem borers were removed (Khan et al., 2014).

Surface runoff is a principle contributor to the degradation of soil resources, negatively impacting ecosystem services in turn. Semi-permeable stone bunds (often referred to by the French term "digue filtrante") are used in dryland areas to reduce the velocity of runoff and erosion (Stroosnijder, 2003; Taye et al., 2015). Furthermore, adequate soil cover and mulching can reduce runoff (González-Núñe et al., 2004; Mwango et al., 2015) as well as the use of cover crops such as deep and coarse rooted legume plant species (e.g., *Melilotus*, *Lathyrus* and *Linum*) that can reduce the loss of rainfall by up to 17% (Ramos et al., 2015; Yu et al., 2006).

A wide variety of traditional soil and water conservation methods including zai (historically practiced in West Africa), micro basins, earthen bunds and ridges (Nyamadzawo et al., 2013), *fanya juus* infiltration pits (Twomlow et al., 2008) and contour stone bunds (Garrity et al., 2010) are used in response to drought and climate variability. The use of zai (small basins traditionally used to capture surface runoff), together with the application of nitrogen fertiliser, has led up to 2000% and 250% increase in potato and bean yield respectively in Ethiopian highlands. This in turned raised household income by 20-fold (Amede et al., 2011). Integrated use of the various forms of planting pits resulted in an increase in yields of up to 300% increase in South African drylands (Twomlow et al., 2008). Zai in combination with contour stone bunds, the application of manure and tree planting can result in enhanced crop and biomass production. Up to 94% of the cultivated land was rehabilitated in areas where such methods were implemented for an extended period of time (Reij, 2009).

A further method employed to avoid and reverse degradation in croplands is the use of different forms of agroforestry and shelter belts (for an example, see Section 3.8.2). Tree based wind shelters were established in the pastoral regions of Northern China in the late 1970s (Yu et al., 2006). The size of shelter belts was chosen based on local conditions, wind speed and types of plant species used. Mixed species were used in 50–600 m long and 4–30 m wide belts in a perpendicular orientation to wind direction (Wang et al., 2008). Shelter belts reportedly reduced wind speed, reduced soil temperature by up to 40% and increased soil moisture by 30%.

3.7.1.2. Grazing and Fire Management in Drylands

Humankind's use of grazing animals started approximately 8000 years ago (Mignon-Grasteau et al., 2005). In time, animal husbandry has come to form one of the principle land use options in dryland ecosystems, in which the majority of the world's grazing lands and livestock production are located (Safriel et al., 2005). Light to moderate grazing pressure has generally been shown to have minimal impact on integrity of rangelands and the ecosystem services they provide (Papanastasis et al., 2017). When correctly implemented, sustainable grazing and livestock management provides a source of food, fibre, leather, and transportation. However, overgrazing by livestock has been documented as a key driver of desertification in some contexts (D'Odorico et al., 2013; Geist and Lambin, 2004; Havstad et al., 2006; Huang et al., 2007; Manzano and Návar, 2000), with a resulting loss of ecosystem services in terms of a reduction in livestock production, carbon sequestered in soils and the regulation of water flow and erosion. In addition to reducing litter and basal cover, intense livestock pressure can severely impact biological soil crusts, inhibiting the important role they perform in controlling erosion and fixing nutrients (Pointing and Belnap, 2012; Weber et al., 2016).

Sustainable grazing and fire regimes are time and place dependent and must be developed in a context specific manner. They require the systematic monitoring of climate, vegetation and animal health to adjust livestock numbers accordingly. It is also crucial that monitoring be linked to drought contingency planning and timely management actions to avoid desertification (Torell et al., 2010; Stafford Smith and Foran, 1992). Continuous heavy grazing during droughts can result in a loss of basal cover, an increase in woody plants, accelerated soil erosion, increases in exotic invasive weeds and loss of ecosystem services (Archer et al., 2017). Methods to achieve sustainable grazing regimes include adjusting: 1) season of use; 2) duration of use; 3) stocking rate (animals per unit area); 4) type of livestock and balance of grazers and browsers; 5) class of livestock (e.g., yearlings vs bulls for cattle); 6) herding; 7) fencing to control access; 8) access to water; 9) location of mineral block; 10) shade structures; 11) supplemental feeding; 12) predator control and 13) silvo-pastoral practices (Bailey, 2005; Bailey et al., 2008; Fuhlendorf and Engle, 2004; Ganskopp, 2001; Viswanath et al., 2018)**.** Further management practices in the form of implementing appropriate fire regimes, the application of fertiliser, woody plant control, and reseeding can be used to restore and sustainably manage grazing lands (Fuhlendorf and Engle, 2004; Briske et al., 2011).

Within many dryland areas, the concept of Holistic Planned Grazing has been advocated as a process through which to enhance livestock production and restore and sustainably maintain rangeland structure, function and diversity (Savory and Parsons, 1980; Savory, 1983). The concept follows a set of key principles that aim to simulate the effect that herds of indigenous herbivores historically had on land, particularly short-duration, high-intensity grazing with long periods of rest to allow vegetation to recover (Hawkins et al., 2017). Although there are studies that indicate that farmers in certain areas have noted an increase in production and biodiversity (e.g. Stinner et al., 1997), more recent reviews and comparative assessments have questioned its effectiveness and found no difference in principle rangeland indicators when compared to season-long continuous grazing regimes (Carter et al. 2014; Nordborg and Roos 2016; Hawkins et al. 2017).

However, if an ecological tipping point has been exceeded, restoration to a historic state may not be economical or ecologically feasible (D'Odorico et al. 2013). In general, preventing desertification is strongly preferable and more cost-effective than allowing land to degrade and then attempting to restore it (IPBES 2018). For this reason, the Land Degradation Neutrality framework response hierarchy prioritises avoiding and reducing land degradation, before restoration measures: *Avoid* > *Reduce* > *Reverse* (Cowie et al., 2018; Orr et al., 2017).

3.7.1.3. 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), with clearly documented trends in southern Africa (Dougill et al., 2016; Joubert et al., 2008; O'Connor et al., 2014), North America (Archer et al., 2017; Barger et al., 2011) and Australia (Eldridge and Soliveres, 2014). 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 ecosystem degradation and desertification. At intense levels, where a closing woody plant canopy inhibits grass production, bush encroachment can lead to a decrease in stream flow, fodder production and further ecosystem services, resulting in 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 (Eldridge et al., 2011; Eldridge and Soliveres, 2014). In certain areas, an intermediate level of woody cover may be desired where the ability of the landscape to produce fodder for livestock is retained, while the production of wood and associated products increases. This may be particularly important in regions such as southern Africa where 95% of rural households depend on wood fuel from surrounding landscapes as well as livestock production (Shackleton and Shackleton, 2004).

Where bush encroachment has been assessed as a form of degradation, there are often substantial efforts to clear woody species. Early bush clearing efforts can be found in Namibia where a national program is aimed at clearing woody species through mechanical measures (harvesting of trees or clearance by bulldozers and heavy rollers) as well as the application of arboricides (Smit et al., 2015). However, the long-term success of clearance and improved fire and grazing management remains to be evaluated, especially restoration back towards an 'original state' with associated ecosystem services and biodiversity. In particular regions, for example, northern Namibia, the rapid reestablishment of woody seedlings following clearing has raised questions about whether full clearance and restoration is possible (Smit et al., 2015). The underlying drivers of encroachment need to be addressed, in particular overgrazing and fire regimes that may have led to the initial encroachment (Eldridge and Soliveres, 2014). If these primary drivers of woody plant encroachment cannot be addressed, it is likely that the landscape will be invaded once again. In such cases, 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).

Lastly, the cost of clearing and the economic feasibility of generating products such as sawn timber, fencing poles, fuel wood and commercial energy production opportunities remains to be assessed in full. Initial studies in Namibia and South Africa (Stafford-Smith et al., 2017) indicate that there may be good opportunity, but factors such as the cost of transport can substantially influence the financial feasibility of implementation. This remains an area where additional research is required.

3.7.1.4. Rainwater Harvesting

Rainwater harvesting (RWH) provides a means of increasing the amount of water available for agriculture and livelihoods through the capture and storage of runoff and peak flow. It is often highlighted as a practical response to climate change and dryness (i.e., long-term aridity and low seasonal precipitation) as it forms a partial buffer against rainfall variability that is already relatively high within drylands and predicted to become more acute over time (Dile et al., 2013; Vohland and Barry, 2009). For these reasons, RWH is recommended by the UNCCD and is widely implemented by government and non-governmental agencies, often as part of a broader suite of rural development and water, agriculture and land management activities (Dile et al., 2013; Vohland and Barry, 2009; Yosef and Asmamaw, 2015).

One of the primary reasons for implementing RWH is to improve agricultural output and resilience. There is *high agreement* that the implementation of RWH systems leads to an increase in agricultural production in drylands (see reviews by Biazin et al., 2012; Bouma et al., 2016; Dile et al., 2013). A meta-analysis of changes in crop production due to the adoption of RWH techniques across the drylands of Africa and Asia noted an average increase in yields of 78%, ranging from –28% to 468% (Bouma et al., 2016). Of particular relevance to anticipated climate change in drylands, is that across the dataset of 158 assessments, studies in dry years with rainfall below 330 mm had the highest observed yield improvements.

RWH can result in modified landscapes and impact ecosystem functioning at a range of temporal and spatial scales (Vohland and Barry, 2009). For example, at a plot scale, RWH structures may increase available water and enhance agricultural production, biomass accumulation, soil organic carbon and nutrient availability (Singh et al., 2012; Vohland and Barry, 2009; Yosef and Asmamaw, 2015). However, at a catchment scale, it may reduce runoff and important flows to wetlands and downstream urban economies (Meijer et al., 2013).

Yet despite delivering a clear set of benefits, initial adoption and long-term sustainability may be inhibited by a number of economic and institutional reasons. There is a growing focus in recent literature on the governance of RWH systems, including financial, institutional, social, and water and land use planning considerations that impact the sustainability of not only new measures, but those that have been in place for hundreds of years (Bitterman et al., 2016).

Initial adoption is often inhibited by socio-economic status of parties living in drylands (Bahta and Lombard, 2017; Kajisa et al., 2007). Based on broad model of costs and returns, Bouma et al. (2016) illustrated that period required to pay-back the capital investment in RWH is approximately 2 years in Asia and 15 years in African drylands. While remaining viable and delivering a broad suite of benefits, this relatively high upfront cost may explain why some farmers are reluctant to invest.

Aside from potential economic constraints, broader consideration of how RWH incorporated in coupled human-ecological landscapes is required (Dile et al., 2013). The effectiveness of RWH interventions, even those that have been in place for hundreds of years, may be affected by invasive species, urbanisation and a lack of their consideration in broader land use planning (Bitterman et al., 2016). Furthermore, the inappropriate storage of water in warm climes can lead to an increase in water related diseases and associated health issues unless managed correctly (Boelee et al., 2013). Whereas the outcomes of RWH at a plot scale are well understood, its integration into modern emerging dryland landscapes requires further investigation. This includes considering its broader role in assisting communities to adapt to climate change. Although its ability to improve the resilience of dryland agriculture is relatively well known, a broader understanding of its impact on landscape resilience and disaster risk reduction is required.

3.7.1.5 Use of Halophytes for the Revegetation of Saline Lands

Soil salinity can severely limit the growth and productivity of crops (Jan et al., 2017) and lead to a decrease in available arable land. Salinity reduces the productive capacity of soil through numerous morphological, physiological and biochemical processes, which lead to high seed mortality, poor or delayed germination, poor crop stand, stunted growth and reduced yield (Ahmad et al., 2010; Ashraf et al., 2012). Saline land usually becomes loosely structured, powdery and highly erosive. This increases the erosion of soil by wind as well as water through particle-laden runoff (Hameed et al., 2008; Qadir et al., 2009). The salinity of land can increase to an extent where the cultivation of normal crops becomes impossible.

In terms of response options, 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. Adoption of such crops is not new; they were used during the ancient civilisation of Mesopotamia, and they in effect, allow farmers to reclaim farmland (Gul and Khan, 2003).The biomass produced can be used as forage, food, feed, essential oils, biofuel, timber, 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 revegetated (Dagar et al., 2014; Wicke et al., 2013).

In Pakistan, where about 6.2 million hectares of fertile land is affected by salinity, pioneering work on utilising salt tolerant plants for the revegetation of saline lands (Biosaline Agriculture) was done in the early 1970s at the Nuclear Institute for Agriculture (NIAB, 1997). Under this approach, a succession of plants is grown, ranging from highly salt tolerant grasses and woody plants, to salt tolerant crops, and the produced biomass is used for various economical purposes. A number of local and exotic varieties were initially screened for salt tolerance in lab- and greenhouse based studies, and then distributed to further saline areas with similar land and socioeconomic conditions (Ashraf et al., 2010). The selected salt tolerant species included tree species (*Acacia ampliceps, A. 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 (*Gossypium hirsutum*), wheat (*Triticum aestivum*) and *Brassica* spp (Mahmood et al., 2016).

In India and elsewhere, tree species including *Prosopis juliflora, Dalbergia sissoo, Eucalyptus tereticornis* have been used to revegetate 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) has been grown as fodder crop on degraded soils with brackish water.

3.7.1.6 Incentivising Sustainable Land Management and Restoration

The adoption of SLM practices depends on the compatibility of the technology with prevailing socio-economic and bio-physical conditions (Sanz et al., 2017). Globally, it was shown that every dollar invested into restoring degraded lands yields social returns in the range of 2–5 dollars over a 30-year period (Nkonya et al., 2016a). A similar range of returns from land restoration activities were 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). 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 (Sanz et al., 2017; Giger et al., 2015; Schmidt et al., 2017). Furthermore, market failures in the form of lack of access to credit, input and output markets, and insecure land tenure (Section 3.2.3) result in the lack of adoption of SLM technologies (Moussa et al., 2016). Enabling policy frameworks contribute to overcoming these market failures (Section 3.7.3). Measures to expand payments for ecosystem services, or inclusion of subsidies that support SLM adoption in existing agricultural support policies, are *likely* to lead to a higher level of adoption of SLM and land restoration activities (Schiappacasse et al., 2012; Lambin et al., 2014; van Zanten et al., 2014; Reed et al., 2015; Bouma and Wösten, 2016; Section 3.7.3).

3.7.2. Socio-economic Responses

Socio-economic and policy responses are often crucial in enhancing the adoption of SLM practices (Fleskens and Stringer, 2014; Cordingley et al., 2015; Nyanga et al., 2016) and for assisting agricultural households to diversify their sources of income (Shiferaw and Djido, 2016; Barrett et al., 2017). Technology and socio-economic responses are not independent, but are in continuously evolving interaction (Hornbeck, 2012; Liu and Lan, 2015).

3.7.2.1. Socio-economic Responses for Combating Desertification Under Climate Change

Desertification limits the choice of potential climate change mitigation and adaptation response options. Furthermore, many additional factors, for example, a lack of access to markets or insecurity of land tenure, hinder the adoption of SLM. An important consideration is that these factors are largely beyond the control of individuals or local communities and require broader policy interventions (Section 3.7.3). Nevertheless, local collective action and indigenous and local knowledge are still crucial to the ability of households to respond to the combined challenge of climate change and desertification.

The use of indigenous and local knowledge enhances the success of SLM and its ability to address desertification (Altieri and Nicholls, 2017; Engdawork and Hans-Rudolf, 2016). There are abundant examples of how indigenous and local knowledge, also often referred to as agroecological techniques (Altieri, 2018), has 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 3.7.1). Traditional "ndiva" water harvesting system in Tanzania enables 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 North Africa and the Middle East, a similar 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 and environmental changes (Enfors and Gordon, 2008; Engdawork and Hans-Rudolf, 2016). Moreover, there is *robust evidence* documenting the marginalisation or loss of indigenous and local knowledge (Fernández-Giménez and Fillat Estaque, 2012; Hussein, 2011; Kodirekkala, 2017; Moreno-Calles et al., 2012; Dominguez, 2014). In this context, innovative combinations of indigenous and local knowledge and modern management practices can contribute to overcoming the combined challenge of climate change and desertification (Engdawork and Hans-Rudolf, 2016; Guzman et al., 2018).

Collective action is a result of social capital and has the potential to contribute to the sustainable management of common resources and climate change adaptation (Adger, 2003; Engdawork and Hans-Rudolf, 2016; Eriksen and Lind, 2009; Ostrom, 2009; Rodima-Taylor et al., 2012). 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 bylaws 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 (Bonney et al., 2016; Le et al., 2016a; 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 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).

Despite the ingenuity, innovation and collective action of dryland residents, their adoption of SLM practices remains insufficient to address desertification and adapt to climate change due to the highlighted constraints to 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; Nkonya et al., 2016c; Wildemeersch et al., 2015; Sections 3.2.4.2 and 3.7.1.6). 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).

3.7.2.2. 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 (*robust evidence, high agreement*). 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; Varghese and Singh, 2016). Access to non-agricultural employment is especially important for smallholder pastoral households as their small herd sizes make them less resilient to drought (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 evidence, high agreement*). 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 United States in the 1930s (Hornbeck, 2012), and during droughts in Burkina Faso in the 2000s (Barbier et al., 2009) and in Mexico in the 1990s (Nawrotzki et al., 2016). There is *robust evidence* 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) (3.5.2). Barrios et al. (2006) found that urbanisation in Sub-Saharan Africa was partially influenced by climatic factors during the 1950 to 2000 period, in parallel to liberalisation of internal restrictions on labour movements: with 1% reduction in rainfall associated with 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. On the other hand, there is *high agreement* that households with migrant worker members are more resilient against extreme weather events and environmental degradation as 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., 2016).

3.7.3. Policy Responses

Many socio-economic factors shaping individual responses to desertification typically operate at larger scales (Scholes, 2009). 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 in the case with socio-economic responses, policy responses are classified below in two ways: those which seek to combat desertification under changing climate through avoiding, reducing and reversing it (Cowie et al., 2018; Orr et al., 2017); and 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 framework, namely, avoiding, reducing and reversing land degradation (Cowie et al., 2018; Orr et al., 2017).

3.7.3.1. Policy Responses towards Combating Desertification under Climate Change

Policy responses to combat desertification take numerous forms. Below we discuss some major ones consistently highlighted across the literature also in connection with climate change, because these response options were found to strengthen adaptation capacities and to contribute to climate change mitigation. They include improving market access, gender empowerment, expanding access to rural 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 by farming households, help agricultural producers to sell more of their produce at higher prices. Increased profits both motivate and enable them to invest more into sustainable land management. High access to input, output and credit markets is a major determinant for the adoption of sustainable land management practices in a wide number of settings across the drylands (Aw-Hassan et al., 2016; Kirui, 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-climatic interaction, with long-term consequences on the livelihoods and incomes, as was shown for the case of the American Dust Bowl during 1930s (Hornbeck, 2012). Government policies aimed at improving market access usually involve constructing and upgrading rural-urban transportation infrastructures. However, besides infrastructural constraints, providing improved access often involves relieving institutional constraints to market access (Little, 2010).

Gender empowerment. A greater emphasis on understanding gender-specific differences over land-use and land management practices as an entry point is *likely* to make land restoration projects more successful (Broeckhoven and Cliquet, 2015; Carr and Thompson, 2014; Catacutan and Villamor, 2016; Dah-gbeto and Villamor, 2016). This includes taking into account the differences of men and women in processing similar information as well as their perception to risk and uncertainties (Slovic, 1999). 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 programs to assure that their interests are better represented. 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 (Thompson-Hall et al., 2016), bounded rationality for gendered decision making (Villamor and van Noordwijk, 2016), anticipatory learning (Dah-gbeto and Villamor, 2016) and systematic leverage points (Manlosa et al., 2018), which all aim to improve gender equality within agro-ecological landscapes through systems approach.

Expanding access to rural advisory services. Awareness of desertification and associated land degradation problems was found to be a significant factor to incentivising the adoption of sustainable land management practices (Kassie et al., 2015; Nkonya et al., 2015; Nyanga et al., 2016). Agricultural initiatives to improve the adaptive capacities of vulnerable populations were more successful when they were conducted through reorganised social institutions and improved communication (Osbahr et al., 2008). Improved communication and education could be facilitated by wider use of new information and communication technologies (Peters et al., 2015). Investments into education were associated with higher investments into soil conservation measures (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 rural advisory services, and disconnects between advisory 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 programs in the Mediterranean region (Reichardt, 2010). Rural advisory 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; Le et al., 2016a; Vente et al., 2016; Rusike et al., 2006). Moreover, rural advisory 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, and facilitate mutual learning between multiple stakeholders (Morton, 2017).

Strengthening land tenure security. There is *robust evidence* that strengthening land tenure security is a major factor contributing to the adoption of soil conservation measures in croplands (Aw-Hassan et al., 2016; Kirui, 2016; Mythili and Goedecke, 2016; Sow et al., 2016). Moreover, land tenure security leads to more investment in trees (Deininger and Jin, 2006; Etongo et al., 2015). Secure land tenure increased investments into SLM practices in Ghana but it did not affect 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 (Baumgartner, 2017; Dell'Angelo et al., 2017; Aha and Ayitey, 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 driven mainly by climate risks, there is *robust evidence and high agreement* that institutions need to respond with flexible tenure, allowing mobility for pastoralist communities, and not fragmenting their areas of movement (Behnke, 1994; Turner et al., 2016; Holden and Ghebru, 2016; Wario et al., 2016; Liao et al., 2017). More research is needed on the optimal tenure mix, including low-cost land certification, redistribution reforms, market-assisted reforms and gender focused reforms, as well as collective forms of land tenure such as communal land tenure and cooperative land tenure.

Payment for ecosystem services (PES) provide incentives for the land restoration and SLM (Lambin et al., 2014; Reed et al., 2015; Schiappacasse et al., 2012). Several studies illustrate that social cost of desertification are larger that its private cost (Costanza et al., 2014; Nkonya et al., 2016a). Therefore, although SLM can generate positive externalities that are often public goods, individual land custodians underinvest in SLM as they are unable to reap all the benefits. 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. 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 programs in Yunnan, China (He and Sikor, 2015). Yet, when reviewing the performance of PES programs in the tropics, Calvet-Mir et al. (2015), found that they are generally effective in terms of environmental outcomes, despite being sometimes unfair. 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).

Decentralisation of natural resource management. Local institutions often play a vital role in implementing SLM initiatives (Gibson et al., 2005). 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 (Fernández-Giménez 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 programs focused on rangeland management in Dirre, Ethiopia (Abdu and Robinson, 2017). Decentralised governance leads to improved management in forested landscapes (Ostrom and Nagendra, 2006; Dressler et al., 2010). However, when local elites were placed in control, decentralised natural resource management negatively impacted the livelihoods of the poor who were dependent on forest products (Dressler et al., 2010).

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 related to desertification drivers (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 (see Section 3.7.1). There is *high agreement and robust evidence* that such technologies help improve the food security of smallholder dryland farming households (Harris and Orr, 2014, Chapter 6). Strengthening research on desertification is of paramount importance not only to meet Sustainable Development Goals (SDGs) but also effectively manage ecosystems based on solid scientific knowledge. Research is needed on degradation mechanisms and environmental restoration methods, and on unravelling the impact of globalisation on drylands (Bisaro et al., 2011); and ecological ramifications of climate change and its impact on ecosystem services and biodiversity (Ren et al., 2008). Sources of desert dust, the process of their emission and transportation over long distances before redeposition are insufficiently studied (Middleton, 2017). In addition, the impact of desert dust on ecosystems and human and animal health is not fully understood. More investment into research institutes and training the younger generation of researchers helps 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 allows 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 effectiveness of those engaged in assessing and monitoring the phenomenon at various levels (Reed et al. 2011).

Investing into monitoring of desertification and deserts storms. Actions to combat desertification are numerous and diversified, but often lack effectiveness because they are not based on scientific data from long-term ecological and socio-economic observations (Sergeant et al., 2012). Risks related to climate change further emphasise the need for reliable and long-term environmental data (Cornet, 2012; Haase et al., 2018). For monitoring desertification, integration of biophysical (climate, ecological factors, biodiversity) and socio-economic aspects (use of natural resources by local population) provides a basis for better vulnerability prediction and assessment (OSS, 2012; Vogt et al., 2011b). Creation of sandstorm alert systems can help strengthen research on siltation and desert dust (Nickovic et al., 2012). Some well-known examples of previous monitoring observatories and related initiatives include the observation stations of the "Arid Zones" program of UNESCO (1952–1960), the study sites of the International Biological Program (IBP), the observatories of the Long Term Ecological Research Program of the USA, Integrated Carbon Observation System (ICOS), National Ecological, Observatory Network (NEON; USA), Terrestrial Ecosystem Research Network (TERN; Australia), the DESERTLINKS project by the European Union; Cameleo (Changes in Arid Mediterranean Ecosystems on the long term and Earth Observation), ROSELT (Network of Observatories of Long-Term Ecological Monitoring) is a program initiated by the OSS at the level of the countries of the Sahel and North Africa (OSS, 2012) - whose experiences could be evaluated for the design of future successful programs.

Developing modern renewable energy sources. 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 (IEA 2013; Amugune et al. 2017). Use of biomass for energy, mostly fuelwood (especially as charcoal), was associated with deforestation and desertification in some dryland areas (Neufeldt et al., 2015; Iiyama et al., 2014; Mekuria et al., 2018; Zulu, 2010), while in some other areas there was no link between fuelwood collection and desertification (Simon and Peterson 2018; Twine and Holdo 2016; Swemmer et al. 2018). 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 desertification. Transition to renewable energy sources in high-income countries in dryland areas primarily contributes to reducing greenhouse gas 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 transitions to renewable energy are being promoted by governments across drylands (Sen and Ganguly, 2017; Hong et al., 2013; Cancino-Solórzano et al., 2016) even in fossil-fuel rich countries (Stambouli et al., 2012; Vidadili et al., 2017; Farnoosh et al., 2014; Dehkordi et al., 2017), despite important social, political and technical barriers to expanding renewable energy production (Afsharzade et al., 2016; Karatayev et al., 2016; Baker et al., 2014). Improving the social awareness about the benefits of transitioning to renewable energy resources, such as hydro-energy, solar and wind energy contributes to their improved adoption (Aliyu et al., 2017).

3.7.3.2. Policy Responses towards Economic Diversification

Despite policy responses for combating desertification, climate change, population pressures and growing food demands, as well as the need to reduce poverty and strengthen food security, are *very likely* to put strong pressures on the land (Cherlet et al., 2018; see also Chapter 6 and 7). 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 and agricultural commercialisation. Investments into expanding irrigation in dryland areas can help improve labour productivity and boost production and income revenue from agriculture and livestock sectors, the major driving factor being profitability. Barrett et al. (2017) noted 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. However, such a transition did not improve farmers' livelihoods in all cases (Reardon et al., 2009). High value cash crop/animal production is being bolstered by wide scale use of technologies, for example, mechanisation, 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).

Structural transformations in rural economies implies that the development of non-agricultural sectors facilitate 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. Economic transformations also occur by the way of urbanisation, through the shift of underutilised labour in rural areas into gainful employment in urban areas (Jedwab and Vollrath, 2015). The larger share 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).

Furthermore, this period will demand effective ways of managing ecosystems and mitigation of desertification while addressing the demand of cities. 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 getting 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 is also contributing to expedited agricultural commercialisation by providing sustainable market outlet for cash and high value crop 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).

3.8. Hotspots and Case Studies

Desertification has been addressed in drylands over decades using different strategies. Some examples of hotspots and responses to desertification and climate change are presented in this section.

3.8.1. Case Study on Climate Change and Soil Erosion in Drylands

3.8.1.1. Global Status of Soil Erosion and its Main Drivers

Soil erosion is present in nearly all geographic regions. Wind erosion, is a significant process, affecting 41% of the global land area, especially arid and semi-arid areas (Moharana et al., 2016). At a country scale, in the case of Chile, erosion rates reach up to 100 t $ha^{-1}y^{-1}$, having increased substantially over the last 50 years (Ellies, 2000). More than 10% of the country exhibits high erosion rates (greater than 1 t $ha^{-1}yr^{-1}$) (Bonilla et al., 2010). Using the Universal Soil Loss Equation, it has been estimated that soil erosion can be as high as 300 t $ha^{-1}yr^{-1}$ (equivalent to a net loss of 18 mm per year) in Spain (López-Bermúdez, 1990). In Turkey, the amount of sediment recently released through erosion into seas was estimated to be 168 million tonnes per year, which is considerably lower than the 500 million ton per year that was estimated to be lost in the 1970s (ÇEMGM, 2017). The decrease in erosion rates is attributed to an increase in spatial extent of forest land, the rehabilitation of degraded forests, erosion control, prevention of overgrazing, and improvement in irrigation technologies. Despite its global importance, 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). In addition to the loss of soil, erosion has a direct and negative impact on soil nutrients and organic matter, thereby impacting land's productive capacity. Globally, water erosion is estimated to result in the loss of 23–42 MtN and 14.6–26.4 MtP annually (Pierzynski et al., 2017). Soil organic carbon stocks are also affected by erosion as topsoil is lost. Wind erosion results in a loss of fine soil particles (silt and clay), reducing the ability of soil to sequester carbon (Wiesmeier et al., 2015).

3.8.1.2. Observed Trends in Arid Lands

The results of soil erosion models indicate that water erosion is a global phenomenon. At a continental level, in the year 2001, South America was predicted to have the highest rate of soil erosion rate (3.53 Mg $ha^{-1}yr^{-1}$), followed by Africa (3.51 Mg $ha^{-1}yr^{-1}$) and Asia (3.47 Mg $ha^{-1}yr^{-1}$) (Borrelli et al., 2017). Within the United States, average soil erosion rates on all cropland have decreased more than 38% since 1982 due to better soil management practices (Kertis, 2013). The national average annual erosion rate on non-Federal US rangeland is estimated to be 1.41 t $ha^{-1}yr^{-1}$. Over 18% of the non-Federal rangelands might benefit from treatment to reduce soil loss to below 2.24 t $ha^{-1}yr^{-1}$ (Weltz et al., 2014). Within China, rainfall erosivity has shown a positive trend in dryland areas between 1961 and 2012 (Yang and Lu, 2015). Zhang et al. (2015) state that while water erosion area in Xinjiang has decreased by 23.2%, the erosion considered as severe or intense was still increasing. In the case of Chile, there has been a significant increasing trend in soil erosion, while in 1970 only a small fraction of the territory had been affected by erosion, in year 2000, erosion effected one quarter of the country (Mathieu et al., 2007). Soil loss due to erosion is much higher on bare land compared to cultivated land. Nabi et al. (2008) estimated the soil erosion rate from barren lands in Soan river basin in the Potohar region of Pakistan to be 63.41 t $ha^{-1}yr^{-1}$ and that from the cropped land to be 18.76 t $ha^{-1}yr^{-1}$. In Pakistan, the highest rate of erosion was estimated to be 150–160 t $ha^{-1}yr^{-1}$ (Anjum et al., 2010). In the semi-arid regions of Brazil, no major changes in runoff and sediment transport were observed (Santos et al., 2017). In Mediterranean Europe, Guerra et al. (2016) found a reduction of erosion due to greater effectiveness of soil erosion prevention between 2001 and 2013.

3.8.1.3. Climate Change Impacts on Erosion in Arid Lands

There are several erosion mechanisms that will be highly affected by future climate change. Intensification of glacier retreat could increase soil erosion in certain regions (dependent on other variables) as eroded material is transported when ice moves over the underlying bedrock. The same occurs with sea level rise and storm surge intensities that increase soil erosion in coastal areas (Kalhoro et al., 2017). Land use change and deforestation aggravates the effect of climate on erosion (Gutiérrez-Elorza, 2006). Accelerated erosion and sediment transport as a result of deforestation reduces the lifespan of dams, irrigation systems and local infrastructure. A particularly notable example is Warsak dam in Pakistan, built in Khyber Pakhtunkhwa province on the Kabul River in 1960, which was completely filled with sediment in three years. Similarly, the lifespan of other major dams, Tarbela and Mangla, was reduced by more than 10 years (Nabi et al., 2008). Changes in the intensity and seasonal distribution of precipitation, as projected by most climate change scenarios, increase the frequency and intensity of flood events and can intensify erosion processes (*robust evidence; high agreement*) (Molnar, 2001; Nearing et al., 2015; Ziadat and Taimeh, 2013).

3.8.1.4. Successful Restoration and Rehabilitation Examples

3.8.1.4.1 Soil Erosion and Desertification in Algeria

In Algeria, desertification mainly affects the steppes of arid and semi-arid regions where the economy is based on pastoral farming. Algerian steppes are marked by great interannual rainfall variability. Rainfall has declined about 18% to 27% and the dry season has increased by two months in the last century. Associated with this drying, floods are often observed, increasing the vulnerability of soils to erosion (Belala et al., 2018; Hirche et al., 2011).

The population of the steppes has increased substantially, from 1,024,777 inhabitants in 1968 to 7,500,000 in 2008, an average annual growth rate of 2.5% (last census in 2008 by National Statistical Office). The population of livestock, predominantly sheep, has grown exponentially since 1968, leading to severe overgrazing, trampling and soil compaction, which greatly increases the risk of erosion (Nedjraoui and Bédrani, 2008). Prevalent wind erosion is due to prevailing climatic conditions and anthropogenic action that reduces the vegetation cover (Hirche et al., 2018; Le Houérou, 1996). Wind carries away fine particles such as sands and clays and leaves a lag gavel pavement which is unproductive. Water erosion transports soil and nutrients offsite resulting in a loss of soil fertility and water holding capacity.

The national map of soil sensitivity to erosion (Salamani et al., 2012) provides a stark picture of the amount of area at risk of soil erosion in the steppe and pre-Saharan areas of Algeria (Figure 3.11). More than three million hectares of land in the steppe provinces (Naama, El-bayadh, Djelfa, M'sila, Tebessa) experience intense wind activity and are particularly high-risk areas for soil erosion. Nearly 600,000 ha of land in the steppe zone are totally desertified without the possibility of biological recovery.

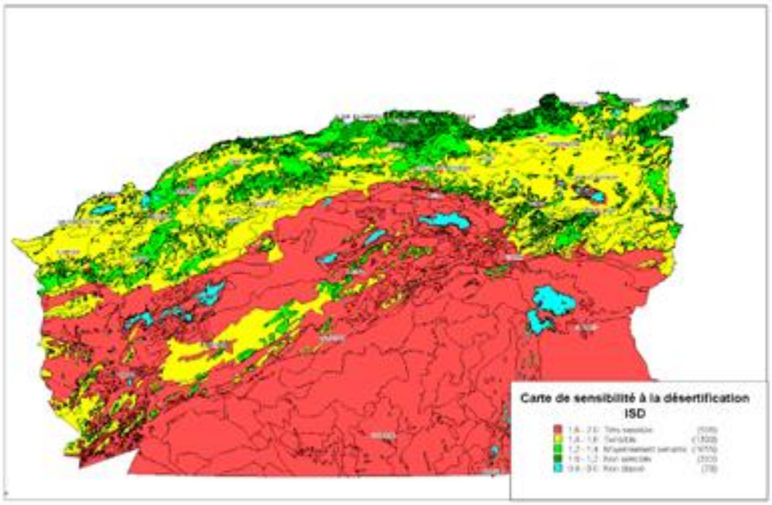

Figure 3.11 Map of sensitivity to desertification of northern Algeria (Salamani et al., 2012)

Combating soil erosion has therefore been one of the priority objectives of the state authorities since the beginning of the 1970s. Many ecological and socio-economic programs have been launched at different times. These programs aim to revitalise degraded areas and improve the management of livestock and natural resources. The Last National Reforestation assessment documented the successful reforestation of 60% of the area (895,260 ha) since 2000. Additional work is planned, 2015–2019, to extend the forested area for the protection and restoration of rangelands from silting.

3.8.1.4.2 No-Till Practices in Central Chile

Over the last few decades there has been an increasing interest in the development of No-Till (also called zero tillage) technologies as a way 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 have positively impacted soil biology with increases in soil organic matter. Early evaluations by Crovetto (1998) showed that No-Till farm (after seven years) had doubled the biological activity indicators of traditional farming and even surpassed those found in pasture (grown for the last 15 years). Besides erosion control, additional benefits are an increase of water holding capacity and reduction in bulk density. The influence of this iconic farm has resulted in the adoption of soil conservation practices and specially No-Till in dryland areas of the Mediterranean climate region of central Chile (Martínez et al., 2011).

3.8.1.4.3 Combating Wind Erosion and Deflation in Turkey: The Greening Desert of Karapınar

The Karapınar district is located on the plains between the Konya and Ereğli districts of Turkey. It is characterised 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 a large drifting dunes that threaten settlements around Karapınar (Groneman, 1968). Such dune movement had begun to affect the Karapınar settlement in 1956 (Kantarcı et al., 2011). Consequently, the Karapınar town and nearby villages faced danger of abandonment due to out-migration in early 1960s (Figure 3.12). The reasons for increasing wind erosion in the Karapınar district can be summarised as follows: sandy material originated from an old lake bed and was mobilised following drying of lake; hot and semi-arid climate conditions along with high seasonality in precipitation and year-to-year variability; overgrazing and use of pasture plants for fuel; excess tillage, particularly the Shock-Disc Plough that degrades soil structure and buries the productive surface horizon; and Karapınar is located in an area with strong prevailing winds.

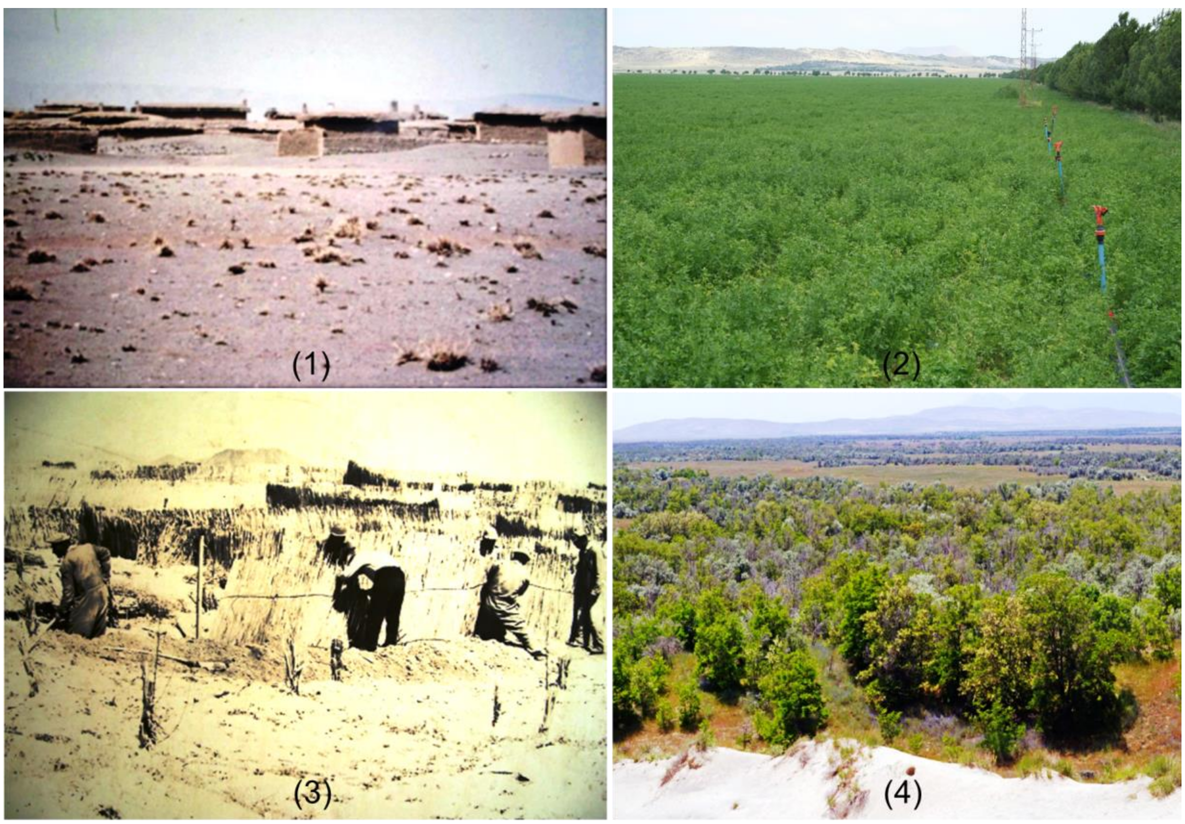

Figure 3.12 (1) A general view of a nearby village of Karapınar town in early 1960s (Çarkaci, 1999). (2) A present view of the Karapınar wind erosion area that has been opened to sustainable and productive agricultural practices for many years (Photograph: Murat Türkeş, 17.06.2013). (3) Construction of Cane Screens in early 1960s in order to decrease speed of the wind and prevent movement of the sand accumulations and dunes, which was one of the physical measures during the prevention and mitigation period (Çarkaci, 1999). (4) A view of present mix vegetation in most of the Karapınar wind erosion area, 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 (Photograph: Murat Türkeş, 17.06.2013)

Restoration and mitigation strategies were initiated in 1959 and today, 4,300 ha of this land has been restored (Akay and Yildirim, 2010) (Figure 3.12-2), using specific measures: (1) Physical measures:

construction of cane screens to decrease wind speed and prevent sand movement (Figure 3.12Figure 3.11-3); (2) 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; (3) 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 3.12-4). 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).

3.8.2 Case Study on Green Walls, Green Dams and Green Belts

In order to combat desertification and to adapt to and mitigate climate change, the measures and actions of the Green Dam, Green Belt or Green Great Wall have been applied in East Asia (e.g., China), Mediterranean area (e.g., Turkey), and Africa (e.g., Algeria, Sahara and the Sahel region).

3.8.2.1. The Experiences of Combating Desertification in China

Arid and semiarid areas of China, including north-eastern, northern and north-western regions, cover an area of more than 1.6 million km^2, with annual rainfall of below 450 mm. Over the past several centuries, more than 60% of areas in arid and semiarid 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 livelihoods of nearly 200 million people, and afforestation programs for combating desertification have been initiated since 1978. China is committed to go beyond the “Land-Degradation Neutrality” objective as indicated by the following programs that have been implemented. The Chinese Government began the Three North’s Forest Shelterbelt program 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 semiarid regions. The project is implemented in three stages (1978–2000, 2001–2020, and 2021–2050). In addition, the Chinese government launched Beijing and Tianjin Sandstorm Source Treatment Project (2001–2010), Returning Farmlands to Forest Project (2003–present), 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; Tao, 2014; Wang et al., 2013b).

The results of the fifth monitoring (2010–2014) showed: (1) Compared with 2009, the area of degraded land decreased by 12,120 km^2 over a five-year period; (2) 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%; (3) Compared with 2009, the amount of wind erosion decreased by 33%, the average annual occurrence of sandstorms decreased by 20.3% in 2014; (4) 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; (5) The sandy area created 5.4 million ha of fruit production with annual output of 4.86 10^{10} kg of fresh and dried fruits, accounting for 33.9% of the national annual output (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 have been accelerated, and the trainings of technicians for farmers and herdsmen have been strengthened. To improve the monitoring capability and technical level of desertification, the monitoring network system has been strengthened, and the popularisation and application of modern technologies are intensified (e.g., information and remote sensing). Special laws on combating desertification have been decreed by the government. The provincial government responsibilities for desertification prevention and controlling objectives and laws have been strictly implemented.

3.8.2.2. The Green Dam in Algeria

After independence, the Algerian government had the reconstruction of forests destroyed by the war and the steppes by desertification among its top priorities (Belaaz, 2003). In 1972, the government invested in the “Green Dam" (“Barrage Vert”) project to stop desertification. 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 and 200 mm. It is a 3 Mha band of plantation running from east to west (Figure 3.13). It is over 1,200 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 population is low (3 to 92 hab km^{-2}), but has a high rate of growth (1.6 to 4.3).

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 experience of previous years led to the launch of integrated management assessments, the improvement of tree and fodder shrub plantations, and the development of water conservation techniques. Reforestation during this period was carried out using multiple species, including fruit trees to increase and diversify sources of income of the population.

The evaluation of the Green Dam from 1972 to 2015 (General Forest Direction) shows that 300,000 ha 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%, depending on the region. Currently, in line with the New Rural Renewal Policy, the government has planned to relaunch the rehabilitation of the Green Dam by incorporating new concepts related to sustainable development, combating desertification, and adaptation to climate change. Through demonstration, the Green Dam has inspired several African nations 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, 2016).

Figure 3.13 Localisation of green Dam in Algeria (Saifi et al., 2015). Note: The green coloured band represents the location of the green dam, the yellow band delineates the national border of Algeria

3.8.2.3. Afforestation and Erosion Control in the Green Belt of Turkey

Turkey has a high level of land degradation and erosion due to its topographical structure, sensitivity of land to erosion, climate and improper agricultural practices including destruction of range and forest lands (Yurtoglu, 2015). A cooperative project "The National Afforestation and Erosion Control Mobilization Action Plan (NAECMAP)" was implemented by Turkey's Ministry of Environment and Forestry (MOEF) (now the Ministry of Agriculture and Forestry) to reduce GHG emissions and increase carbon sequestration. Public institutions, municipalities, non-governmental organisations and community assigned by National Afforestation and Erosion Control Mobilization Law no. 4122 were organised to combat desertification. NAECMAP (2008 to 2012) prescribed coordinated work among public bodies and parties undertaking of afforestation, rehabilitation and erosion control work over an area of 2.3 million ha. The MOEF aimed to implement over an area of 2.16 million ha, with other institutions covering an area of 136,000 ha. The total cost of these activities was estimated to be more than 2.7 billion Turkish Liras.

The NAECMAP had following objectives: Rehabilitation of forests and 10% canopy closure, restoring productivity with minimum cost and effort; Decreasing GHG emissions and increasing carbon sequestration through rehabilitation of infertile forests and afforestation where possible; Restoring intact ecosystems to minimise the adverse impacts of climate change and desertification on livelihoods; Preventing floods and overflows that lead to loss of lives and goods; regulating water run-off in watersheds and improving water quality; Reducing pressure on remaining forests by establishing new forests to meet country's need for wood; Raising public awareness of importance of caring for forests by establishing the planting of saplings as a common tradition practised by citizens every year.

Figure 3.14 Each year of the mobilisation, 300,000 people were employed for seed and seedling production, afforestation, rehabilitation and erosion control work in Turkey. It also addresses a growing need for recreational space in urban areas of Turkey

In 1973, forest covered 20.2 million ha of land and by 2012 an additional 1.5 million ha of land had been forested through the program. During the five years of the NAECMAP, Turkey achieved the afforestation of 210,169 ha; soil protection and afforestation in 315,889 ha; and private afforestation in 49,385 ha (Figure 3.14). Some 1.75 million ha of degraded forest and 37,880 ha of degraded rangeland were rehabilitated. In addition, erosion control and revegetation was done along 8,135 km of highways and 2,262 km of village roads together with that in 27,000 school yards, 1,095 health centres and 9,826 sanctuaries and cemeteries. In the scope of the 'Schools get life' initiative, school orchards are being planted. Green belt afforestation was also achieved around cities (Figure 3.14). Over the five years, 109 million seedlings were distributed to the public free of charge. NAECMAP also provided employment opportunities to the rural population employing 300,000 people for six months.

3.8.2.4. 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. 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 7,775 km from Senegal on the Atlantic coast to Eritrea on the Red Sea coast, with a width of 15 km (Figure 3.15). The wall passes through Djibouti, Eritrea, Ethiopia, Sudan, Chad, Niger, Nigeria, Mali, Burkina Faso and 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 species that are resilient 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.

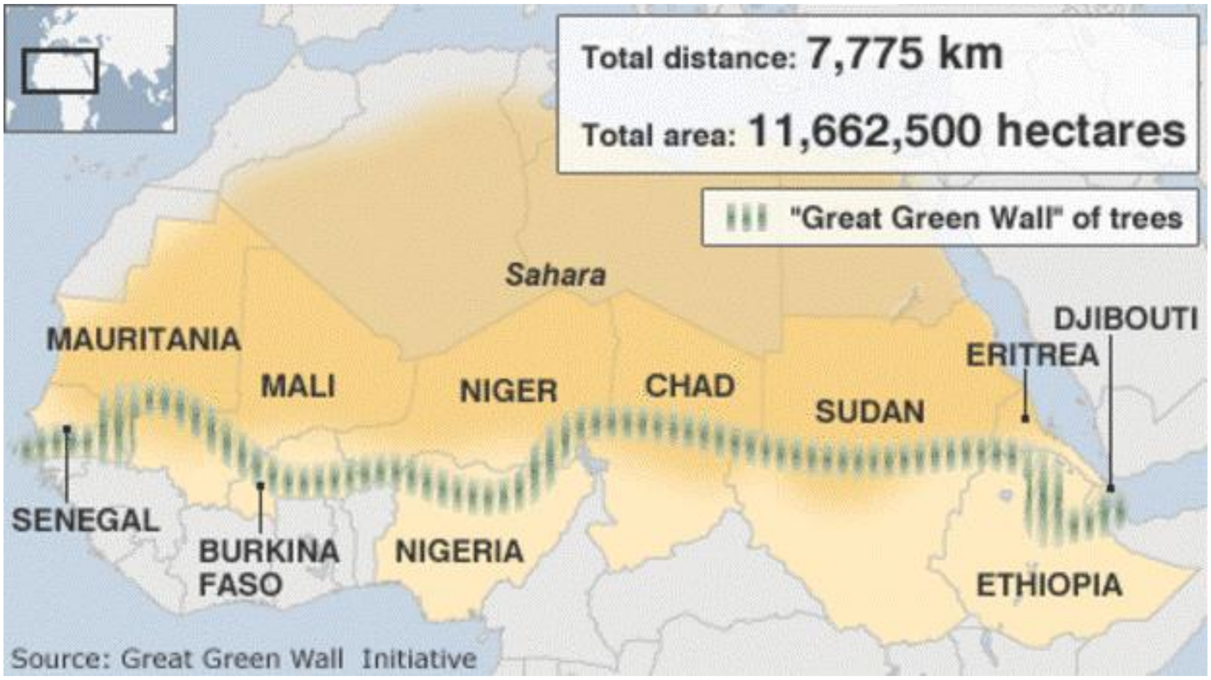

Figure 3.15 The Great Green Wall

The implementation of the initiative has already started in several countries. For example, the FAO's Action Against Desertification project is 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 million hectares can be restored, requiring the restoration of 10 million hectares per year to achieve Land Degradation Neutrality targets by 2030. Despite this early implementation actions on the ground, the achievement of the planned targets is questionable and challenging without significant additional funding.

3.8.3. Case Study on Invasive Plant Species

3.8.3.1. 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; Settele et al., 2014). 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 mesic 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., 2011, 2015; Newton et al., 2011). Moreover, increasing human populations in dryland areas are responsible for creating new invasion opportunities (Safriel and Adeel, 2005).

Current drivers of species introductions include population growth, expanding global trade and travel, land degradation and changes in climate (Richardson et al., 2011; Chytrý et al., 2012; 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 semiarid 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 and Dukes et al. (2011) observed that the invasive plant *Centaurea solstitialis* grew six time larger under elevated atmospheric CO_2 expected in future climate change scenarios.

Climate change is most likely going 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 are 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 in turn promote invasive species, which in turn may lead to further desertification.

Overall, the mean number of invasive species predicted to find suitable climate conditions in dryland areas is anticipated to decrease slightly by 2050 (Lowe et al., 2000). At a regional scale, Bellard et al. (2013) predicted increasing risk in Africa and Asia, with declining risk in Australia (Figure 3.16). This projection does not represent an exhaustive list of invasive alien species occurring in drylands.

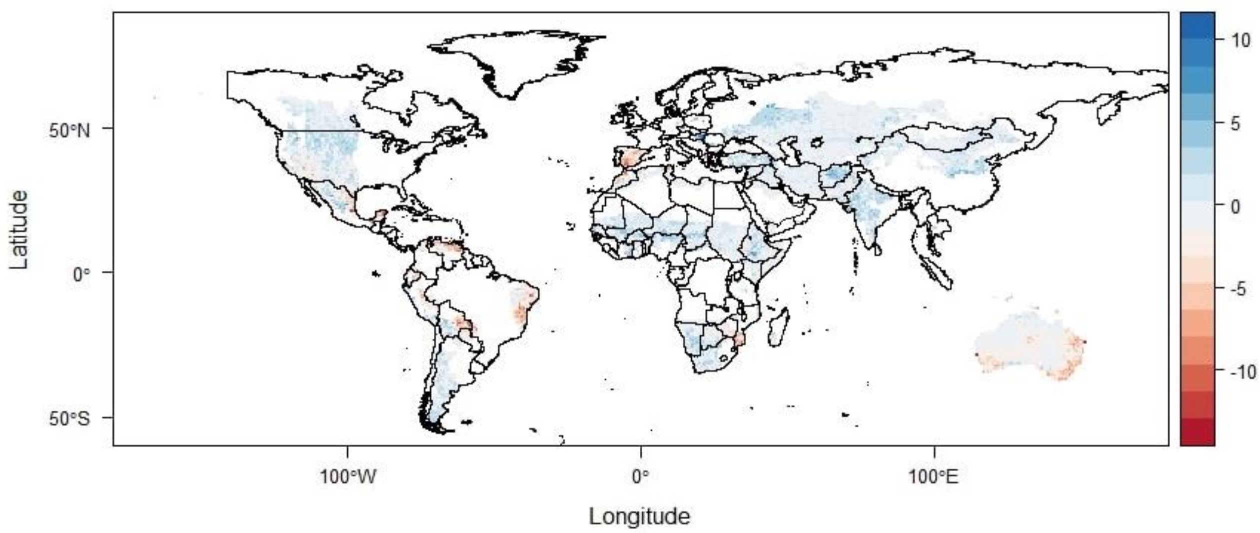

Figure 3.16 Difference between the number of invasive alien species (n=99, from Bellard et al. (2013)) predicted to occur by 2050 (under A1B scenario) and current period "2000"

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

3.8.3.2. Description of the Problem

Ethiopia. The two invasive plants that inflict the heaviest damage are *Parthenium hysterophorus* and *Prosopis juliflora* (Adkins and Shabbir, 2014). It is assumed that *Prosopis juliflora* (mesquite) was introduced in the 1970s and has since spread rapidly. Likewise, a recent study reported that *Parthenium hysterophorus L* has spread into 32 out of 34 districts in the northernmost region of Ethiopia, Tigray (Teka, 2016). The weed is a substantial agricultural and natural resource problem and forms a significant health hazard (Reda, 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).

Mexico. Buffelgrass (*Cenchrus ciliaris* L.) is 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 Sonora desert of Mexico, the distribution of buffelgrass has increased exponentially, covering 1Mha in Sonora State (Castellanos-Villegas et al., 2002). Furthermore, its potential distribution extended to 53% of Sonora State and 12% of semiarid and arid ecosystems in Mexico (Arriaga et al., 2004).

United States. Sagebrush ecosystems have declined from 25 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 United States (Figure 3.17). Cheatgrass infests more than 10 M ha 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 (Pilliod et al., 2017; Balch et al., 2013).

Tamarisk species are shrubs or small trees considered to be among the most aggressively invasive and potentially detrimental exotic plants in the U.S.A. (Kerns et al., 2009; Pearce and Smith, 2007; Nagler et al., 2010). Tamarisk were introduced into the U.S.A. as ornamentals and planted for erosion control and have spread across the western United States and northern Mexico (Pearce and Smith, 2007; Glenn et al., 2012). Tamarisk is now the third most frequently occurring woody riparian plant and the second most abundant species (out of 42 native and non-native species) evaluated along rivers in the western United States (Nagler et al., 2010). Tamarisks concentrates salts on and within underlying soils, utilises large amounts of water as facultative phreatophytes, replaces native riparian vegetation, reduces biodiversity of aquatic macroinvertebrates, provides poor quality habitat for most wildlife, alters decomposition processes, limits recreational opportunities, and changes flood regimes by narrowing river corridors (Sala et al., 1996; Di Tomaso, 1998; Bailey et al., 2001; Glenn et al., 2012; Bedano et al., 2014).

Climate change will alter the distribution and abundance of tamarisk via both direct effects on plants and indirect effects resulting from changes to stream flow, biotic interactions and human activities. Direct effects of warming will shift species distributions northward and upstream (Parmesan, 2006). Changes in the dynamics of highly managed stream flow and ground water regimes along rivers in the western U.S.A. which are already changing as a function of changing climate, will increase niches and opportunities to spread (Barnett et al., 2008). The largest impact on ecosystem services is predicted to be from tamarisks when both indirect and direct effects are accounted for (Ikeda et al., 2014). Habitat suitability model results indicate that 21 % of the northwestern region of the U.S.A supports suitable tamarisk habitat under projected climate changes. Climate change provides opportunity for tamarisk to move into Canada and disrupts its river systems and biodiversity (Pearce and Smith, 2007). Although uncertainty exists regarding future climate change on the rate of spread of invasive species, it is projected that a 2 to 10-fold increase in highly suitable tamarisk habitat will occur by the end of the century in the central region of the U.S.A. and into Canada as the species moves north with changes in temperature (Kerns et al., 2009).

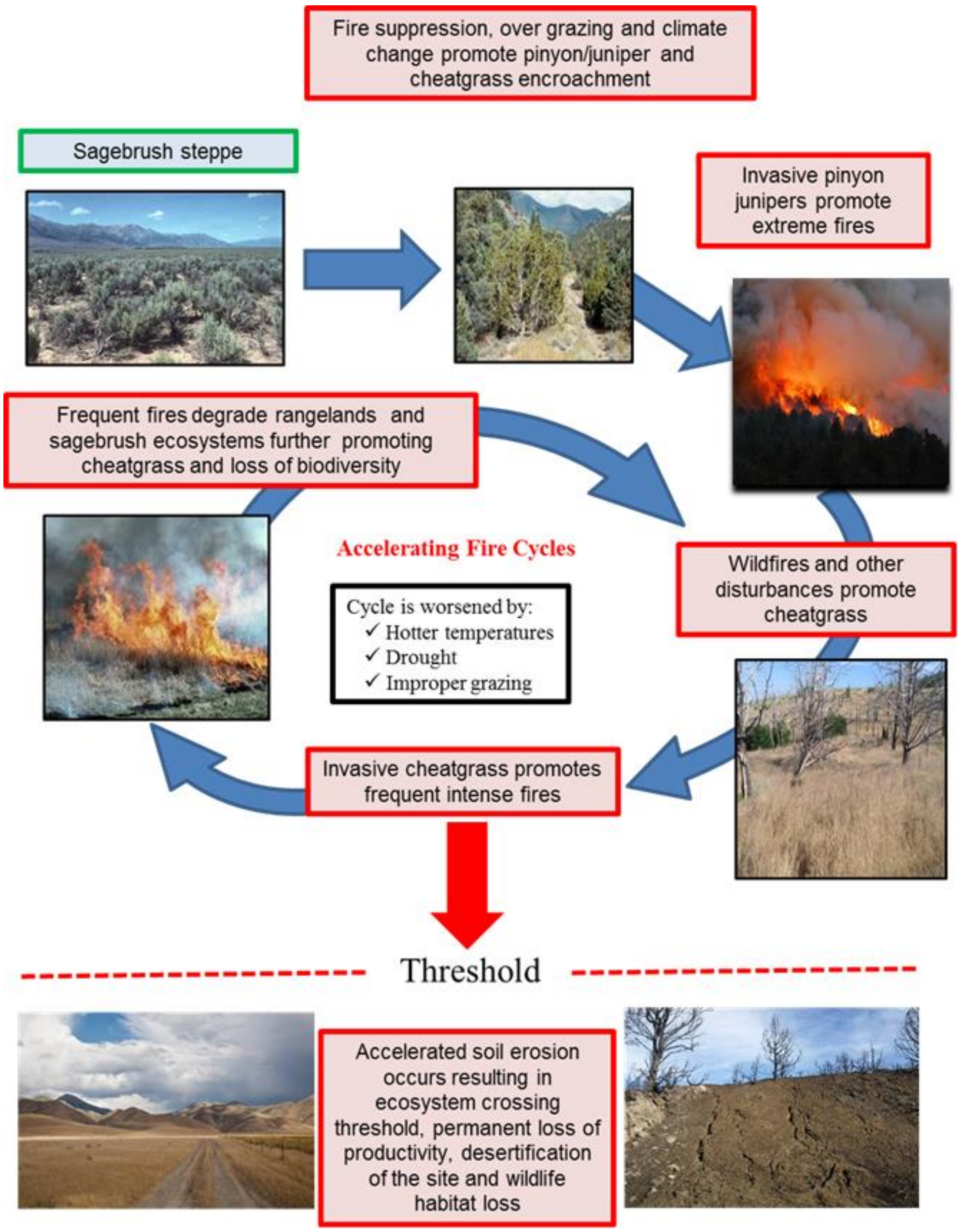

Figure 3.17 Invasive species cycle in the Great Basin region of the western U.S.A. Note: Showing loss of biodiversity and site degradation as a result of interaction between drought, overgrazing promoting open space between sagebrush plants that are infilled by Juniper trees and cheatgrass (*Bromus tectorum*). This promotes fire by increasing fine fuels and latter fuels to ignite the encroaching trees. The site then is colonised by cheatgrass which promotes accelerated soil erosion and permanent loss of productivity. Endemic wildlife species (e.g., sage grouse and pygmy rabbits) habitat is lost and these animals face possible extinction. Source: Mark Weltz, United States Department of Agriculture, Agricultural Research Service

3.8.4.3. Consequences

Ethiopia. *Prosopis,* classified as the highest priority invader in the country, is threatening livestock production and challenging the sustainability of the pastoral systems. 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).

Mexico. Castellanos et al. (2016) reported that soil moisture was lower in the buffelgrass savannah cleared 35 years ago than in the native semiarid 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).

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

3.8.3.4. Interventions and Lessons Learned to Date

Ethiopia. There is neither a comprehensive intervention plan nor a clear institutional mandate to deal with invasive weeds, however, there are fragmented efforts where local communities have tried to clear *Prosopis* by cutting and burning. *Parthenium* was declared a noxious weed in 2001 (Dinwiddie, 2014) and even though the measures were taken to arrest the spread of the invader, they are clearly inadequate. The lessons learned are related to actions that have contributed to current scenario. First, a 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. The second major lesson is the cost of inaction. Research and development organisations did sound the alarm, but the warnings went largely unheeded, resulting in the spread of two invasive plant species that could have been avoided.

Mexico. Incorporation of buffelgrass is considered a good management practice by producers and the government. For this reason, no remedial actions are undertaken.

United States. Attempts to reduce cheatgrass impacts through reseeding 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 unlikely, particularly in relatively low elevation sites with minimal annual precipitation (less than 200 mm per year) (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). Projections of increasing temperature (Abatzoglou and Kolden, 2011), and observed reductions in and earlier melting of snowpack in the Great Basin region (Mote et al., 2005, 2018; Harpold and Brooks, 2018) suggest that there is a need to understand current and past climatic variability as this will drive wildfire and invasions of annual grasses.

3.8.4. Case Study on Oases in Hyperarid Areas

Climate change is projected to have substantial and complex effects on hyper-arid areas around the world (Abatzoglou and Kolden, 2011; Ashkenazy et al., 2012; Bachelet et al., 2016; Guan et al., 2018; Iknayan and Beissinger, 2018). Safriel et al. (2006) found that deserts warmed-up at an average rate of 0.2ºC–0.8ºC per decade between 1976 and 2000. Oases in hyper-arid areas are, thus, at the forefront of climate change and desertification impacts. Oases are isolated areas in deserts with reliable water supply, usually from lakes and springs, allowing for flourishing vegetation (Ling et al., 2013).

Among such oases, Siwa oasis is located the Western Desert of Egypt (Figure 3.18), in the north of the sand dune belt of the Great Sand Sea. The Great Sand Sea is currently expanding to the southwest of the Siwa Oasis causing severe damage to human settlements, roads, irrigation and drainage networks (Abo-Ragab and Zaghloul, 2017). The Siwa Oasis covers 1050 km^2 of area (Abo-Ragab, 2010). There has been a steady increase in population growth since 1970 from 15,000 people to 38,000 people by 2016 (DRC, 2016). Agriculture is the most important economic activity in Siwa, based on the cultivation of dates, olives and alfalfa.

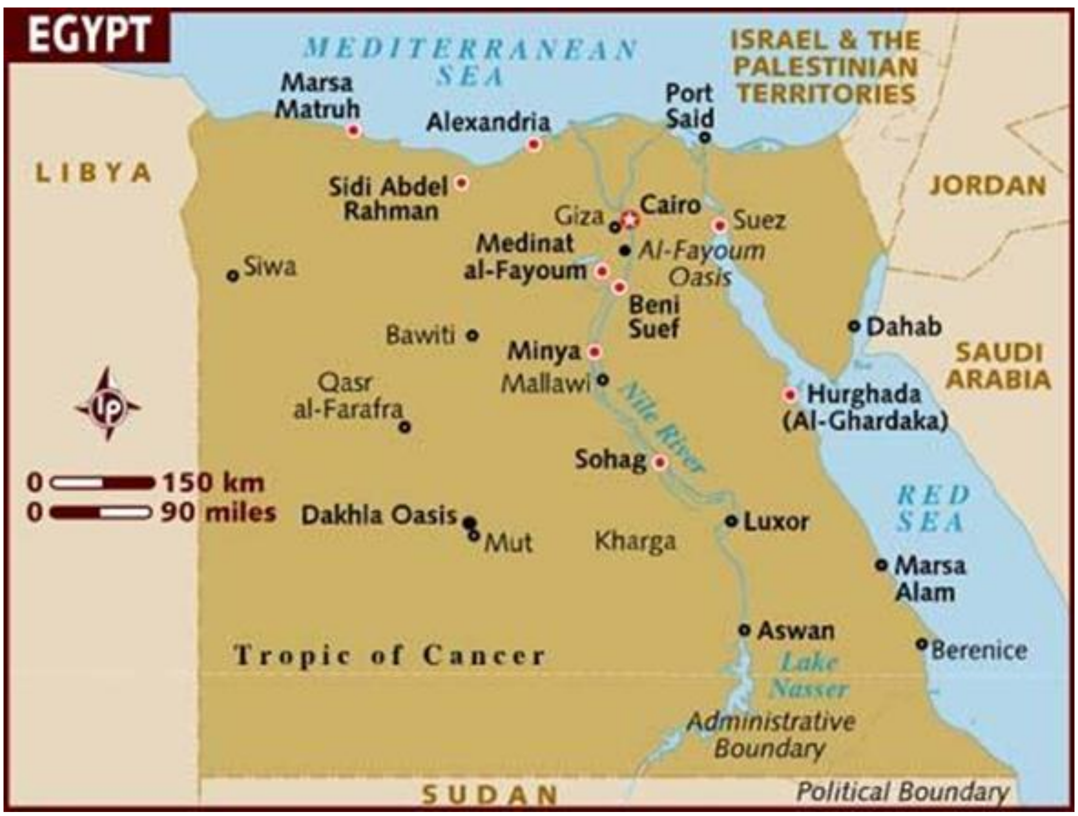

Figure 3.18 Map of Egypt showing the location of Siwa Oasis

The population growth in Siwa is associated with agricultural expansion and land reclamation. The Siwan farmers are turning the surrounding desert into reclaimed land by applying traditional agroecological practices. Yet, agriculture expansion in the Oasis depends on the groundwater that outflows from wells and springs, and the remainder goes to the natural lakes. Moreover, soil salinisation and vegetation loss accelerated since 2000 (Masoud and Koike, 2006). About 85 km^2 of land in Siwa became salinised during 1987–2003, whereas the vegetation loss was observed on 21 km^2 of area (Masoud and Koike, 2006), due to water mismanagement, improper drainage systems and climate warming. In support of that, Gad and Abdel-Baki (2002), Marlet et al. (2009) and Askri et al. (2010) reported that the inefficiency of water use by farmers is the major cause for secondary salinisation (Figure 3.19 and Figure 3.20).

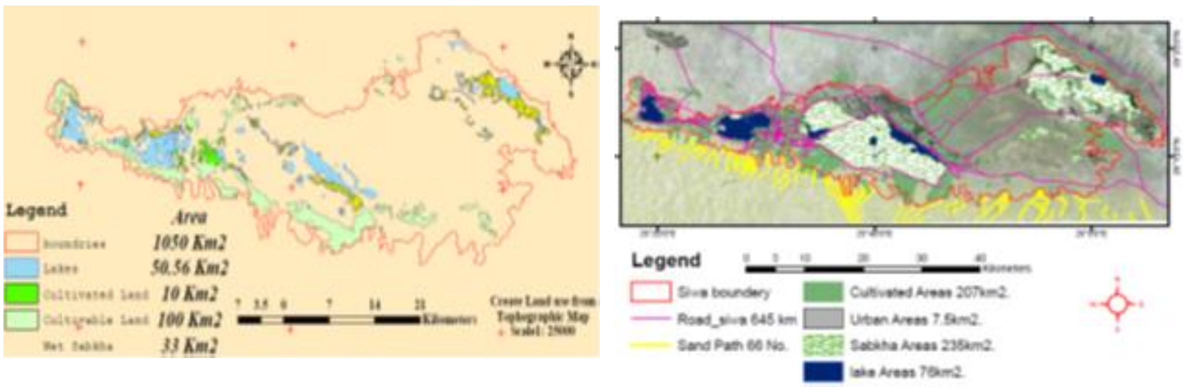

Figure 3.19 Images showing Siwa Oasis in 1929 (left) and 2017 (right). Lakes and wet Sabkha cover 50.6 and 33 km^2 in 1929, and 76 and 235 km^2 in 2017, respectively

Figure 3.20 Increase in water-table in Siwa Oasis

The data from a meteorological station at Siwa Oasis showed that the average increase in temperature from 1990 to 2016 was $+1.33^{\circ}C$ (Figure 3.22) and in wind speed +0.87 mph (Figure 3.23). The increase in these parameters over the years in the Oasis indicates that the evapotranspiration has substantially increased. That accelerated the rate of soil salinisation in the Oasis. Safriel et al. (2006) stated that the rate of increase in temperature and rainfall in the western desert of Egypt was $+0.8^{\circ}C$ and +4% per decade, respectively, in the period of 1976–2000. By using two different global emissions scenarios developed by IPCC (2011), this increase is anticipated to range between +2◦C and +4◦C for temperature and 0% for rainfall per decade in 2071–2100. Along with strong population growth, the cultivated land has increased from 1000 hectares in 1929 to 20,700 hectares in 2017. This increase has been associated by the over-exploitation of the Oasis' groundwater which causes waterlogging for cultivated land and was further exacerbated by the seasonally high evaporation and evapotranspiration rates, intertwined with the improper setup of drainage systems (Masoud and Koike, 2006). These conditions lead to the development of a thick salty layer that hampers agricultural activities (Misak et al., 1997). As a result of the formation of large amounts of salts on the edges of the lakes, salt trade has been recently flourished in the Oasis as a commodity for export to Europe.

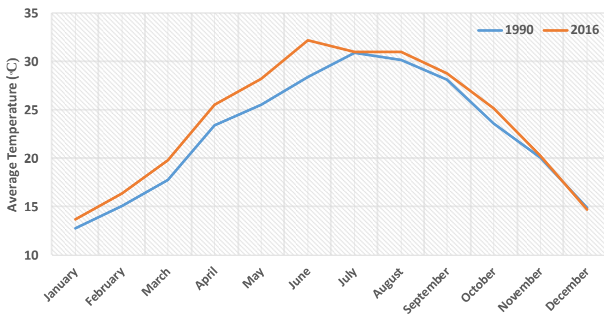

Figure 3.21 Mean monthly temperature at Siwa Oasis in 1990 and 2016 (Data collected from www. geographic.org)

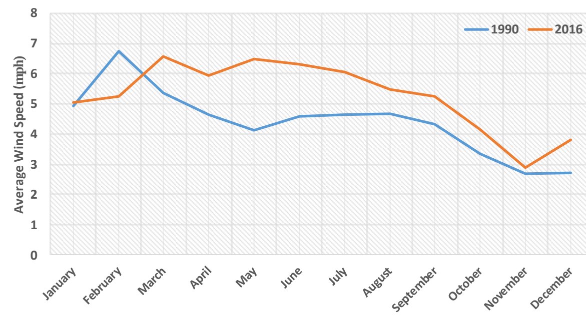

Figure 3.22 Mean monthly wind speed at Siwa Oasis in 1990 and 2016 (Data collected from www. geographic.org)

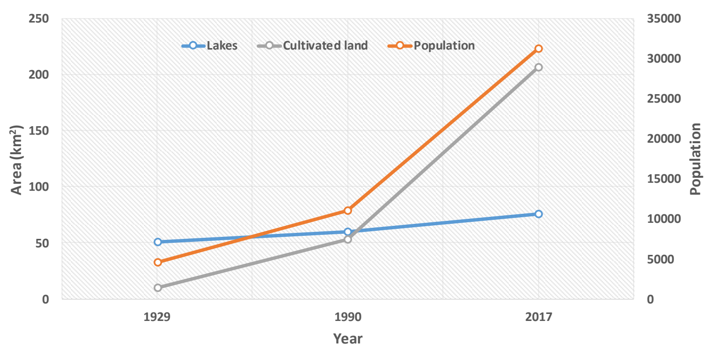

Figure 3.23 The increasing trend in lakes, cultivated land and population in Siwa Oasis over the years. (data collected from Siwa Information Center and satellite images analysis)

Similar interactions of human activities and climatic factors are also observed in other oasis areas around the world. In China, the oases are mainly located in hyper-arid or arid zones of Xingjian, the Hexi corridor belonging to Gansu Province, in Northwest China. Over the past several decades, air temperature and the rainfall increased in the arid region of Northwest China (Chen et al., 2015; Wang et al., 2017), together with fluctuations in evaporation and water flows from glaciers (Chen et al., 2015), and land use changes in different oasis regions. For example, in the oasis along the Keriya River in southern Tarim Basin in Xingjian, cropland areas were expanded, while grassland areas were reduced (Muyibul et al., 2018). In Xingjian Altay Prefecture oasis zone, forests and bare lands, sand fixation area and available water resources decreased, whereas wind erosion increased in the adjacent desert area (Fu et al., 2017). Jinta oasis, a typical agricultural oasis in Hexi corridor, Gansu, China, expanded from 1963 to 2010, and water resource use, land policies, population growth, and climate change all influenced the conversions between oasis and desert regions (Xie et al., 2014).

The attribution of impacts to climate change and human activities vary depending on the location (Song and Zhang, 2015). In the Tarim Basin oasis region, hyper aridity has been the main climate condition since the late Pleistocene, and variations in oasis area was mainly determined by fluctuations on water resources and climatic factors. Increasing population and economic growth caused the expansion of cultivated land and shrinking of native vegetation (Zhang et al., 2003). Following environmental changes over the past years, the ecosystem services in oasis regions in China also changed. In the oasis regions in the Manas River Basin of Xinjiang, China, over the past 60 years, agriculture and animal husbandry production services, and sandstorm and climate adjustment services increased, while the soil preservation and habitats carrying services decreased. The main drivers of these changes were the population growth and unsustainable agricultural activities. The regulation of ecosystem services for sandstorms and water resources were mainly influenced by climate change (Wei et al., 2018).

In the coming years, the inhabitants living in oasis regions will face challenges due to their limited adaptation capacity to global environmental change (Chen et al., 2018). Hence, efforts to increase adaptation capacity to climate change is crucial for sustainable development of oasis regions. This will require addressing the tradeoffs between environmental restoration and farmers' livelihoods (Chen et al., 2018). Managing grasslands and increasing investment into research on eco-agricultural technologies and enhancing protected areas in the mountain and desert areas can contribute to the sustainability of ecosystem services (Fu et al., 2017). Other sustainable land management practices to recover the relatively stable ecological zones between oases and deserts include establishment of straw checkerboards and planting drought-tolerant local natural shrubs, leveling sand dunes and drawing water for irrigation, closing dunes for grass reservation, and developing stable artificial protective forest (Su et al., 2007). Restoring groundwater by enhancing the surface water supply, decreasing groundwater utilisation, particularly sustainable use of the limited water and land resources, are all crucial to the sustainability in oasis regions (Hao et al., 2017). Ultimately, sustainability in oasis regions will require policies integrating the provision of ecosystem services and social and human welfare needs (Wang et al. 2017a).

3.8.5. Desertification Watershed Management: a case study from Ethiopia and Jordan

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 restoration or integrated watershed management (IWM) must address physical, biological and social approaches to achieve sustainable land management objectives (German et al., 2007). The use of indigenous, integrated natural resource conservation measures at watershed scale reportedly dates back thousands years among the indigenous communities of for example the Gedeo, Konso and Borana Oromo in Ethiopia (Chimdesa, 2016). The modern implementation of IWM in Ethiopia dates to the recovery efforts implemented after the droughts of the 1970s and 1980s (Gashaw, 2015).

In Tigray and Amara regions of Ethiopia, a combination of trenches, gabions, stone check-dams, bund stabilisation, recharge pits, and sediment retention ponds combined with gully restoration through enhancing vegetation of resulted in significant reduction in soil loss and allowed for production of pigeon pea and establishment of woodlands with commercial benefits (Mekonen and Tesfahunegn, 2011). Decreases of soil loss between 59% and 89% have been documented in the Enabered watershed (Haregeweyn et al., 2012) and Agula watershed (Fenta et al. 2016), respectively. Encouraged by the positive outcomes farmers agreed to contribute 30–40% free labour for watershed development. The most effective collective investment areas which motivated farmers to do more in protecting and developing their land and environment were access to potable water supply, technologies and inputs; awareness raising functions and governance (Adimassu et al., 2013).

In Abreha-We-Atsibeha, a small village in Tigray region of Ethiopia, the implementation of IWM has over the last 23-years resulted in bare land area coverage declining from 33% in 1991 to 8.6% in 2014. Between 1984 and 2010, shrub land and forestland cover increased by about two-fold with a rate of 54.8 and 19.5 ha yr^{-1} ,respectively (Biedemariam et al., 2017). There was a reduction by about four-fold in bare land 60.2 ha yr^{-1}. The production of cereals such as wheat has increased from 1.8 tons ha^{-1} in 2001 to 2.95 tons ha^{-1} in 2010. The enhancement in soil conditions and water availability allowed farmers to transition from cereal crops to of cash earning from spices and vegetables.

Figure 3.24 An example of how gullies are restored – the fully eroded gully (left) rehabilitated with vegetation (center and right). Source: Alem et al. (2017)

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 200mm of rainfall per year, with some areas receiving less than 100mm (Al-Tabini et al., 2012). Climate analysis has indicated a generally increasing dryness over the West Asia and Middle Eastern region (Zhang et al., 2005; 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 with a similar confidence over the region. Locally increased incidence of extreme events over the Mediterranean region have 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's 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 correspond with a low rainfall interception and infiltration rate, which leads to increased surface runoff and subsequent erosion and gullying, speeding up the drainage of rainwater from the watersheds that 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; Ngigi 2003; Gammoh and Oweis 2011) is being widely applied for rehabilitation of highly degraded rangeland areas in Jordan. Tractor digs out small water harvesting pits on the contour of the slope (Figure 3.24) allowing the retention, infiltration and the local storage of surface runoff in the soil (Oweis, 2017). The micro catchments are planted native shrub seedlings, such as saltbush (*Atriplex halimus*), with enhance survival as a function of increased soil moisture (Figure 3.25) and increased dry matter yields (>300 kg ha^{-1}) that can serve as forage for livestock (Tabieh et al., 2015; Oweis, 2017).

Figure 3.25 Fresh Vallerani micro water harvesting catchment (left) and aerial imaging showing micro water harvesting catchment treatment after planting (middle) and 1 year after treatment (right)

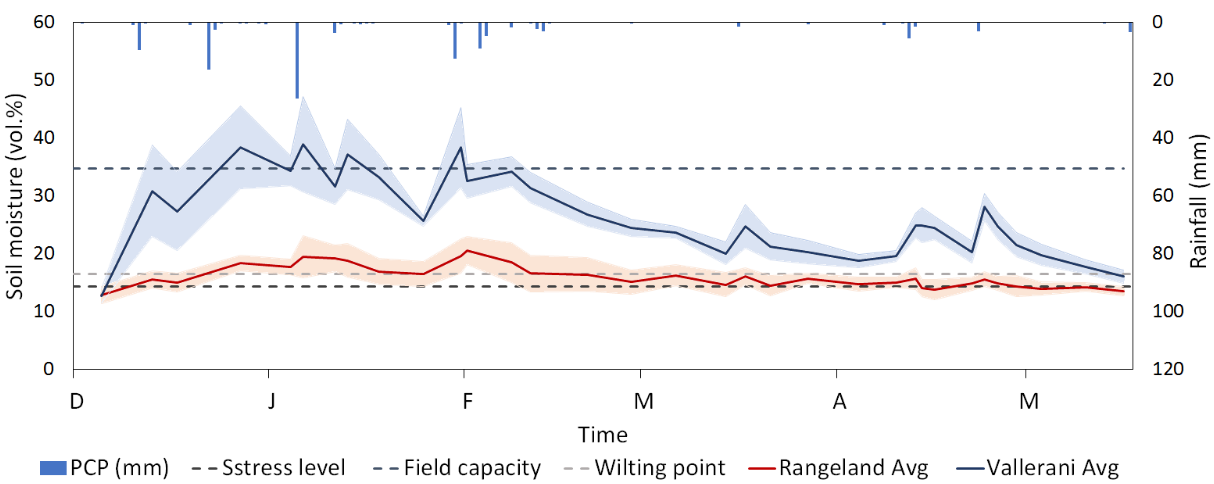

Figure 3.26 Illustration of enhanced soil water retention in the Mechanized Micro Rainwater Harvesting compared to untreated Badia rangelands in Jordan

Simultaneously to MIRWH upland measures, the gully erosion is being treated through intermitted stone plug intervention (Figure 3.27), stabilising the gully beds, increasing soil moisture in proximity of the plugs and 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. Eventually, the recruitment processes and revegetation 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. In-depth understanding of the Badia's rangeland status transition, coupled with sustainable rangeland management, are still subject to further investigation, development and adoption; required to eventually mitigate the ongoing degradation of the Middle Eastern rangeland ecosystems.

Oweis (2017) indicated costs of the fully automated Vallerani technique per hectare was approximately 32 USD. The total cost of the restoration package included the production, planting, and maintenance of the shrub seedlings (USD 11.0 per ha). Tabieh et al. (2015) calculated a benefit cost ratio (BCR) of > 1.5 when revegetation degraded Badia areas through MIRWH and saltbush. However, costs will vary based on the seedling's costs and availability of trained labour.

Figure 3.27 Gully plug development (September 2017) and post rainfall event in March 2018 near Amman, Jordan

Cross-Chapter Box 4: Case Study on Policy Responses to Drought

Contributing authors: Margot Hurlbert (Canada), Muhammad Mohsin Iqbal (Pakistan), Joyce Kimutai (Kenya), Alisher Mirzabaev (Uzbekistan), Fasil Tena (Ethiopia)

Drought is a highly complex natural hazard. It is difficult to precisely identify its start and end. It is slow and gradual. 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). IPCC (2014) defines drought as "a period of abnormally dry weather long enough to cause a serious hydrological imbalance". 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; Wilhite et al., 2014). This recurrent nature of droughts requires pro-actively 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 the societal resilience against droughts (Gerber and Mirzabaev, 2017).

The previous assessment by the IPCC (2014) showed *low confidence* in emerging drought trends at the global scale since 1950, however, IPCC (2014) had a *high confidence* in the increase of frequency and intensity of droughts during the same period in some specific regions of the world, such as the Mediterranean and Western Africa. IPCC (2014) also had *medium confidence* in projected decreases in soil moisture and increases in the frequency and intensity of agricultural droughts in currently dry regions. Surface drying was expected to occur with *high confidence* in the Mediterranean, southwest USA and southern Africa regions by 2100. For Eastern Africa, IPCC (2007) earlier projected a general upward trend in precipitation rates and heavy rainfall events in the twenty-first century, with reduced propensity for drought. However, some recent reports forecast drying over Eastern Africa (Cook and Vizy, 2013).

Droughts are among the costliest of natural hazards. Initial global estimates suggested that the global annual costs of drought equalled 80 billion USD (Carlowicz, 1996), of which around 6–8 billion USD were incurred in the USA (FEMA, 1995). Figures from the European Union showed that the annual damage caused by droughts in the EU was around 7.5 billion Euros in early 2000s (European Comission, 2007). On a sub-national level, Howitt et al. (2015) found that the drought in California in 2015 led to losses equal to 2.7 billion USD. Taylor et al. (2014b) calculated that Uganda lost 237 million USD annually due to droughts between 2005 and 2015. The large-scale drought in Central and Southern Asia during 2000–2002 had also resulted in massive economic costs. In Pakistan, drought negatively affected the agriculture sector and caused its annual GDP growth to decrease from an average of 4.5% (during 1990–2000) to 2.6% during 2000 and 0.07 % during 2001; the GDP growth regained 4.15% in 2002 when drought was over (Anjum et al., 2010). In Uzbekistan, it led to 130 million USD of losses (World Bank, 2005) and about 600,000 people were in need of food aid to the value of 19 million USD (World Bank, 2005).

Usually, these estimates capture only direct and on-site costs of droughts. Droughts have wide-ranging indirect and off-site costs, which are seldom quantified. These indirect effects are both biophysical and socio-economic. Droughts affect not only water quantity, but also water quality (Mosley, 2014). The costs of these water quality impacts are yet to be quantified adequately. Socio-economic indirect impacts of droughts are related to conflict, migration, poverty and short and long-term health consequences due to drought-caused undernutrition (Gray and Mueller, 2012; Johnstone and Mazo, 2011; Linke et al., 2015; Lohmann and Lechtenfeld, 2015). 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; many countries do not do this due to high costs of data collection. However, the opportunities provided by remotely sensed data and novel analytical methods based in big data and artificial intelligence, including use of citizen science for data collection, could help in reducing these gaps. Moreover, it is important to bear in mind that droughts do not cause these indirect socio-economic consequences by themselves, but always together with other economic and institutional factors lowering societal resilience and adaptive capacities. Marginalisation of pastoral communities in dryland areas of Eastern Africa greatly amplifies the impacts of droughts on their livelihoods and food security (Opiyo et al., 2015; Rowhani et al., 2012; Silvestri et al., 2012; Sulieman and Elagib, 2012).

There are three broad (and sometimes overlapping) policy approaches for responding to droughts. Firstly, responding to drought when it occurs by providing direct drought relief is known as crisis management. Gerber and Mirzabaev (2017) suggested that crisis management is also the costliest among policy approaches to droughts because they incentivise the continuation of activities vulnerable to droughts.

The second approach involves development of drought preparedness plans which coordinate the policies for providing relief measures when droughts occur. Clarke and Hill (2013) found that combining resources to respond to droughts at regional level in Sub-Saharan Africa was more effective and cheaper than separate individual country drought relief funding. IFRC (2003) found that providing jobs to drought affected populations in building terraces and check dams helped to strengthen local resilience to future droughts more than providing direct food or cash aid.

The third category of responses to droughts involves drought risk mitigation. Drought risk mitigation is a set of proactive policies aimed at reducing the future impacts of droughts. Drought risk mitigation policies aim to limit the exposure to droughts and to increasing societal resilience to droughts (Vicente-Serrano et al., 2012). For example, 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). Policies also include those addressing livelihood needs including marketing interventions such as destocking, or selling livestock, emergency livestock vaccination, negotiation of exceptional access for grazing to protected areas or commercial ranches (Catley et al., 2009; Morton and Barton, 2002; Abebe et al., 2008).

Reliable, relevant and timely climate and weather information available to and applied by key user groups including farmers, extension officers, policy makers and emergency response units could help monitor drought risks and respond 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). Famine Early Warning System Network (FEWSNet) in East Africa, CILSS/AGRHYMET for West Africa, and Southern African Development Community (SADC) regional early warning unit, for example, have been quite effective in monitoring and forecasting drought events in these regions, as well as an experimental Sub-Saharan drought monitoring and forecasting system (Sheffield et al., 2014). Every US dollar invested into strengthening hydro-meteorological and early warning services in developing countries was found to yield between 4 to 35 USD (Pulwarty and Sivakumar, 2014). Thus far there are weak links with community early warning systems and national and international ones (Wilhite et al., 2014). These indicators have been successfully linked with social media (Tang et al., 2015). There must be care exercised in these instruments not leading to perverse outcomes when linked to some forms of government support (Botterill and Hayes, 2012).

Although previous literature claimed such drought risk mitigation approaches to be much less costly than ex post drought relief, there has not been much research done on quantifying the cost differentials. 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 interregional trade in water could reduce drought costs by 20–30% in the Rio Grande basin, USA. In response to drought some governments have declared emergencies and adopted a system of water rationing while in other jurisdictions water property rights dictate through seniority preference rights who does or does not receive water. 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 responsive timely adjustment to drought (Hurlbert, 2018). Supply side managed water that only provides for proportionate reductions in water delivery, prevents the important adaptation 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 preventing an important adaptation (Hurlbert, 2018). Drought mitigation activities at the macroeconomic level need to be complemented by similar measures at the household and community levels. There is *robust evidence* in the literature that secure land tenure, access to markets, access to agricultural advisory services, and off-farm employment facilitates the adoption of drought mitigation practices by farming households (Alam, 2015; Kusunose and Lybbert, 2014). Programs that provide financial assistance to agricultural producers to build water infrastructure (such as water storage dugouts, pipelines to provide water to livestock) have improved the adaptive capacity of agricultural programs as well as programs that assist producers in planning for environmental risk including drought, soil degradation, pests (Hurlbert, 2018).

All in all, the accumulated evidence shows that it will be increasingly costly to continue with policy responses to droughts based on drought relief measures. The excessive burden of drought relief funding on public budgets have already led to a paradigm shift towards pro-active drought risk mitigation in such countries as USA and Australia. Climate change will only re-enforce the need for pro-active drought risk mitigation approaches, including increased investments into science and research for developing technological and policy options for drought risk mitigation.

3.9. Knowledge Gaps and Key Uncertainties

There are knowledge gaps on the extent of desertification at global and regional scales. Despite numerous related studies, consistent indicators for attributing desertification to climatic and/or human causes are still lacking due to methodological shortcomings. The knowledge of future climate change impacts on specific desertification processes, such as soil erosion, salinisation, nutrient depletion, and vegetation cover and composition change, as well as on dust storms remain limited, especially at the local level. At the global level, the evidence base is not strong and sufficiently granular on how climate change will modify the extent of desertified areas in drylands. Considering the non-equilibrium nature of drylands, with strong influence of climatic variations on the extent of desertification, this is a gap that could be filled within the currently available modelling tools. 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 desertification and climate remains limited. Monitoring desertification to identify the interaction between desertification and climate using Earth observation systems could help fill this gap.

Knowledge gaps persists in the quantification of the impacts of desertification on natural and socio-economic systems. 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. Currently, there is a good understanding of various anthropogenic drivers of desertification. However, the knowledge is lacking on how these drivers will evolve in the future, how they will interact with future climate change, and what would be the effect on desertification.

Despite a lot of studies on separate responses to desertification or to climate change, the knowledge and understanding of the synergies and trade-offs among actions for combating desertification, adapting to and mitigating climate change, and various positive or negative externalities that they will generate in terms of other Sustainable Development Goals is limited. All these aspects are crucial for understanding climate change-desertification interactions and how they will affect people, ecosystems and biodiversity in the future. Filling these gaps requires considerable investments in research and data collection.

Frequently Asked Questions

FAQ 3.1 How do climate change and desertification interact with land use? How can climate change induced desertification be avoided, reduced or reversed?

Climate change and desertification have strong mutual interactions, and the land use and land cover changes associated with desertification contribute to climatic changes, whereas changes in precipitation, temperature, wind speed, and their variabilities due to climatic changes constitute factors affecting desertification. Desertification affects global climate change through the loss of fertile soil and vegetation. In fact, the soil of the drylands contains large amounts of carbon that could enter the atmosphere due to desertification, with important repercussions for the global climate system. The impact of global climate change on desertification is complex and knowledge on the subject is still insufficient. On the one hand, the increase in temperatures can have negative effects by increasing the evaporation of soil water, and some dryland regions will also have reduced rainfall. On the other hand, the increase of carbon dioxide in the atmosphere can enhance the growth of plants. Climate change could translate into an increased risk of aridity and desertification in many areas, although it is difficult to predict the effects of the subsequent loss of biodiversity on desertification.

Sustainable land management (SLM) practices can help avoid, reduce or reverse desertification, mitigate and adapt to climate change. Such SLM practices include conservation agriculture, afforestation and reforestation, crop diversification, planting drought-resilient crop varieties, and many others. Desertification limits choices for such land-based climate change adaptation and mitigation options.

FAQ 3.2 How could land-based options to mitigate climate change affect ecosystem services and biodiversity?

Sustainable land management (SLM) practices which include actions of soil and water conservation in drylands could improve ecosystems services and protect biodiversity. Among provisioning services, conservation agriculture and rangeland management can increase plant biomass, and therefore, the production of food and fibers. Moreover, these practices, as well as, reforestation and afforestation practices can also increase the regulating and supporting services such as soil fertility, water availability and carbon sequestration. SLM practices also support biodiversity through habitat protection and reducing the invasion of alien species. Biodiversity protection results in higher genetic resources, which significantly contributes to human wellbeing through supporting a variety of provisioning ecosystem services.

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