Chapter 3 : Desertification

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Table of Contents

3.1. Executive summary

Dryland areas are expected to expand and become more vulnerable to desertification under climate change due to increasing aridity (*high confidence*). Desertification hotspots currently cover about 10% of the drylands, directly affecting about 277 million people (3.2.1, 3.3). The expansion of drylands is likely to have already occurred in north-eastern Brazil, southern Argentina, the Sahel, Zambia and Zimbabwe, 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). The relative attribution of desertification to climate variability and change and human activities is context-dependent. However, climate variability and change are likely to have played a bigger role in causing desertification, at least in some regions, than previously understood (3.3.2). The growing frequency, intensity and scales of extreme events under climate change are expected to further exacerbate the vulnerability to desertification (*high confidence*) (3.2.1, 3.3.2, 3.6.1). The major human drivers of desertification interacting with climate change are expansion of croplands, poverty and migration (3.2.3.2) (*medium evidence, medium agreement*).

Desertification affects climate change through several mechanisms, with both cooling and warming effects (*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 has increased by 6% and is expected to increase by at least another 8% by 2050. In these areas net carbon uptake is about 27% lower than in others (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 energy available at the land surface and increasing the temperature of the atmosphere. Depending on the types and amounts of aerosols present, dust storms increase the cloud reflectivity and decrease the chances of rain (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 through serving 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 leads to losses in biodiversity (*high confidence*). Desertification processes coupled with climate change are expected to cause reductions in crop and livestock productivity, increases in soil erosion, soil salinity, and in livestock diseases prevalence (*high confidence*) (3.5.1.1), case studies on soil erosion, on invasive plant species, and on salinisation due to sea water intrusion). Desertification has already contributed to the global loss of biodiversity (*medium confidence*) (3.5.1.2). Wildlife are likely to be stressed by coupled effects of climate change and desertification. A reduction in the quality and quantity of resources available to herbivores is likely to have knock-on consequences for predators, potentially disrupting trophic cascades (*low confidence*) (3.5.1.2).

Climate change is expected to exacerbate the already high vulnerability to desertification among dryland populations (*high confidence*). 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). There is growing evidence that migration is used as an adaptation strategy in the context of environmental change (*medium evidence, high agreement*). However, 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 confidence*) (3.5.2.4). Higher frequency, intensity and scales of dust storms due to climate change-desertification interactions further increase the human disease burden due to associated respiratory and cardiovascular illnesses (*medium evidence, high* *agreement*) (3.5.2.3). Sand storms and sand dune movements also cause damage to infrastructure (*high confidence*) (3.5.2.5).

Dryland populations have shown significant ingenuity in successfully adapting to past climate variability and addressing desertification. Site-specific technological solutions, based both on new scientific innovations and traditional knowledge, are available to address 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). Investments into land restoration and rehabilitation have positive economic returns (3.7.1). Integrated soil and water conservation measures were shown to increase coverage density, with positive economic and social effects (*medium confidence*). Conservation agriculture contributes to carbon sequestration in dryland cropped areas (*low confidence*). It also increases climate change adaptation capacities of agricultural households (*high confidence*). Combined use of salt-tolerant crops, improved irrigation practices, and chemical remediation measures were effective in reducing salinity-induced desertification (*medium confidence*). Rangeland management systems such as sustainable grazing approaches and re-vegetation helped to increase rangeland productivity (*medium confidence*). Agroforestry practices were shown to generate diverse ecological and economic 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 and avert aeolian desertification (3.7.1, case study on “Green Walls”). Despite their benefits in terms of addressing desertification, mitigating and adapting to climate change, these 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).

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 (*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*). Improving 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 allows internalisation of some of the social benefits of SLM helping accelerated adoption of 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 satellite-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*) (case study on policy responses to drought). 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 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). Arid, semi-arid, and dry sub-humid areas, together with hyper-arid areas, constitute drylands (UNEP, 1992). Desertification is, thus, a persistent negative trend in land condition causing long term reduction or loss of the biological productivity of drylands, their ecological complexity, and/or their human values (see also Chapter 4, and Glossary), 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 based on the aridity index - the ratio of average annual precipitation amount (P) to potential evapotranspiration amount (PET) (Figure 3.1, also see Glossary). The above definition of desertification, embodied in the (UNCCD, 1994), excludes hyper-arid areas, where the aridity index is below 0.05.

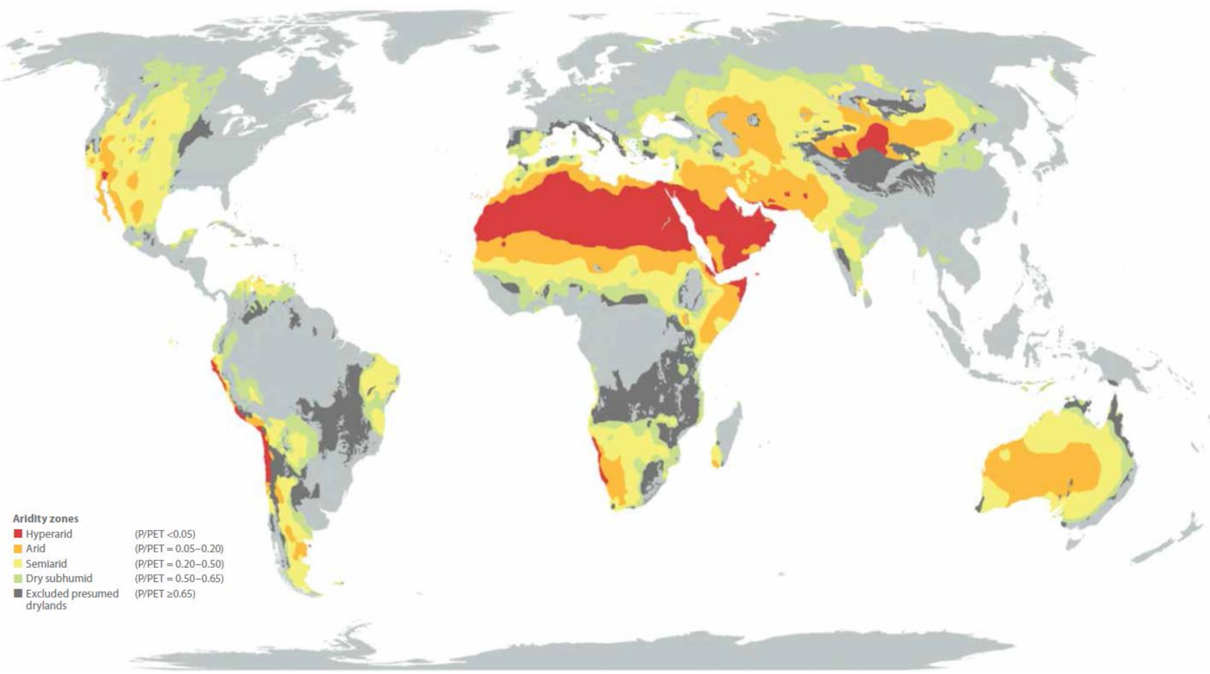

Figure 3.1 Geographical distribution of drylands

Source: (UNEP-WCMC, 2007)

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 coming mainly due to improved data capturing the expansion of drylands towards northern latitudes (Prăvălie, 2016). Excluding deserts, the biggest land use/cover in drylands in terms of area are grasslands, followed by forests and croplands (Figure 3.2).

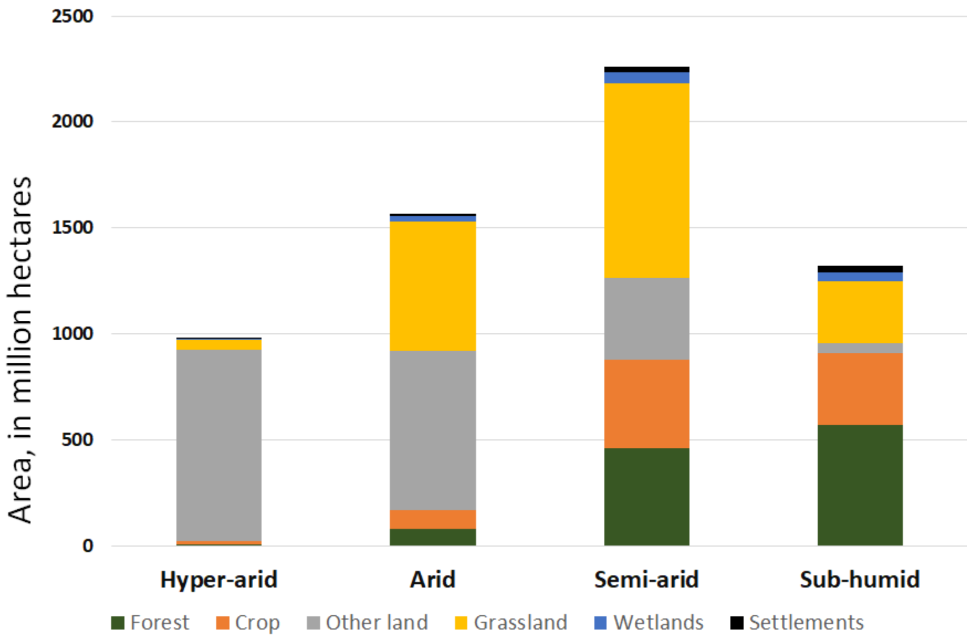

Figure 3.2 Land use and land cover in drylands, in million hectares

Source: FAO (2016)

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, found that about 24%-29% of the global land area experienced a negative change in vegetation (Bai et al., 2008; Le et al., 2016). The figures by Bai et al. (2008) show that only 28% of the areas with vegetation loss are located in drylands. The analysis of the figures by Le et al. (2016) show that about 10% of drylands contain desertification hotspots. 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. However, the emerging evidence points to lower global extent of desertification than previously estimated (see section 3.3 for detailed discussion).

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

Drylands are home to about 37.5% of the global population (PBL, 2017), i.e. about 2.7 billion people. The highest number of people live in the drylands of South Asia (Figure 3.3), followed by Sub-Saharan Africa and Latin America (PBL, 2017). The population growth rates in drylands are projected to increase about twice as rapidly as those in non-drylands, to reach 4 billion people by 2050 (PBL, 2017). In terms of the number of people affected by desertification, the earlier estimates by MEA (2005) and Reynolds et al. (2007) had indicated that desertification is directly affecting 250 million people and indirectly 1 billion people. Le et al. (2016) and IPBES (2018) found that global land degradation is affecting 3.2 billion people. Using the data from Le et al. (2016), it can be calculated that about 277 million people reside in dryland areas which experienced significant vegetation loss between 1982-84 and 2004-06.

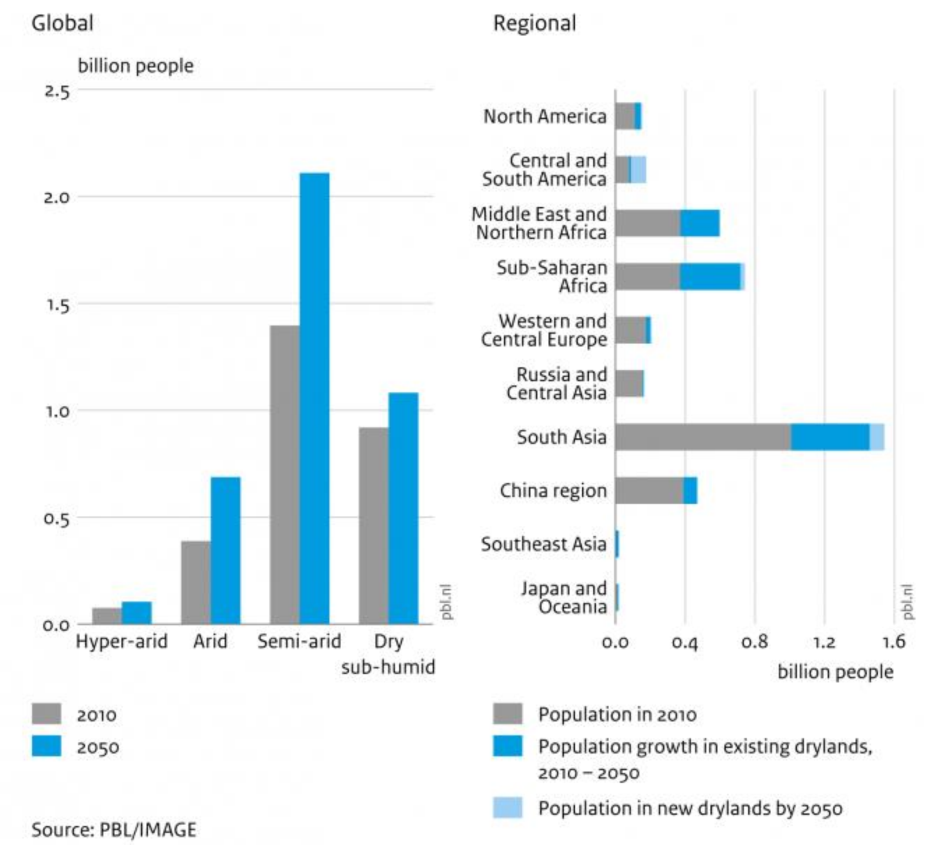

Figure 3.3 Current and projected population in drylands

PBL (2017)

Dryland populations are highly vulnerable (see Glossary for definition of vulnerability) to desertification and climate change (Howe et al., 2013; Huang et al., 2016, 2017; Liu et al., 2016b; 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). Payne (2013) suggested that about 800 million poor people depend on dryland agriculture for their incomes and food security. Climate change is projected to have substantial impacts on all types of agricultural livelihood systems in drylands, including pastoral, agropastoral, rainfed, tree-based and irrigated (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), which require large areas for livestock grazing through migratory pastoralism (Snorek et al., 2014). Grazing lands across

[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 mostly speculative, because often nomadic pastoralists are not fully captured in national surveys and censuses (Carr-Hill, 2013).

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). This has led to increasing marginalisation of pastoral communities and disruption of their economic and cultural structures (Elhadary, 2014). As a result, pastoral communities are not well prepared to deal with increasing weather variability and weather extremes due to changing climate (Dong, 2016; López-i-Gelats et al., 2016).

Globally, it was found that 42% of the very poor live in degrading lands as compared to only 15% of non-poor (Nachtergaele et al., 2010). Payne (2013) claimed that about 16% of dryland populations lived in chronic poverty[3]. Rapid economic growth and poverty reduction in China over the last three decades decreased the absolute global numbers of dryland populations living in poverty (Ravallion and Chen, 2007). However, even in China, Liu et al. (2017) showed that desertified areas have higher levels of poverty and unemployment than the rest of the country. 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). 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 represent relatively poorer locations nationally, with lack of livelihood opportunities, e.g. in Italy (Salvati, 2014). Moreover, within drylands, female-headed households, women and subsistence farmers are more vulnerable to the impacts of desertification and climate change. 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) (section 3.5.2).

Despite these environmental, socio-economic and institutional constraints, dryland populations have historically demonstrated remarkable resilience (see Glossary), ingenuity and innovations, distilled into traditional knowledge[4] and agricultural practices, to cope with high climatic variability and sustain livelihoods (Safriel and Adeel, 2008; Davies, 2017; sections 3.7.1 and 3.7.2). Traditional knowledge has been used over centuries to manage dynamic interactions between the local communities and the ecosystem 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 facilitate ecologically optimal grazing practices (Fernandez-Gimenez, 2000) (section 3.7.2). However, climate change will increase 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

[3]Footnote: Chronically poor are those who are poor over several years, often their entire lives, with their children also frequently being poor (CPRC, 2008).

[4] Footnote: The World Intellectual Property Organization (WIPO) defines traditional knowledge as “knowledge, know-how, skills and practices that are developed, sustained and passed on from generation to generation within a community, often forming part of its cultural or spiritual identity”, http://www.wipo.int/tk/en/tk/

Seneviratne, 2012; Huang et al., 2016). Out of 424.7 million people exposed to droughts in Sub-Saharan Africa, Cervigni et al., (2016) estimated that about 23% were not able to cope with them in 2010. Policy actions promoting the adoption of sustainable land management (SLM) (see Glossary, Chapter 4) practices in dryland areas, based on both traditional 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, 2008a; Cowie et al., 2018; Nkonya et al. 2016a; Stafford-Smith, et al. 2017; IPBES 2018).

3.2.3. Processes and Drivers of Desertification under Climate Change

3.2.3.1 Processes of Desertification and Their Climatic Drivers

Processes of desertification are the direct mechanisms by which drylands are degraded. Desertification consists of both abiotic and biotic processes. These processes are classified under broad categories of physical, chemical and biological degradation. 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; for chemical processes - secondary salinisation and nutrient depletion; for biophysical processes - changes in vegetation cover and composition, including through over/under grazing, deforestation and biodiversity loss (**Error! Reference source not found.**, see also Chapter 4 and Glossary for definitions).

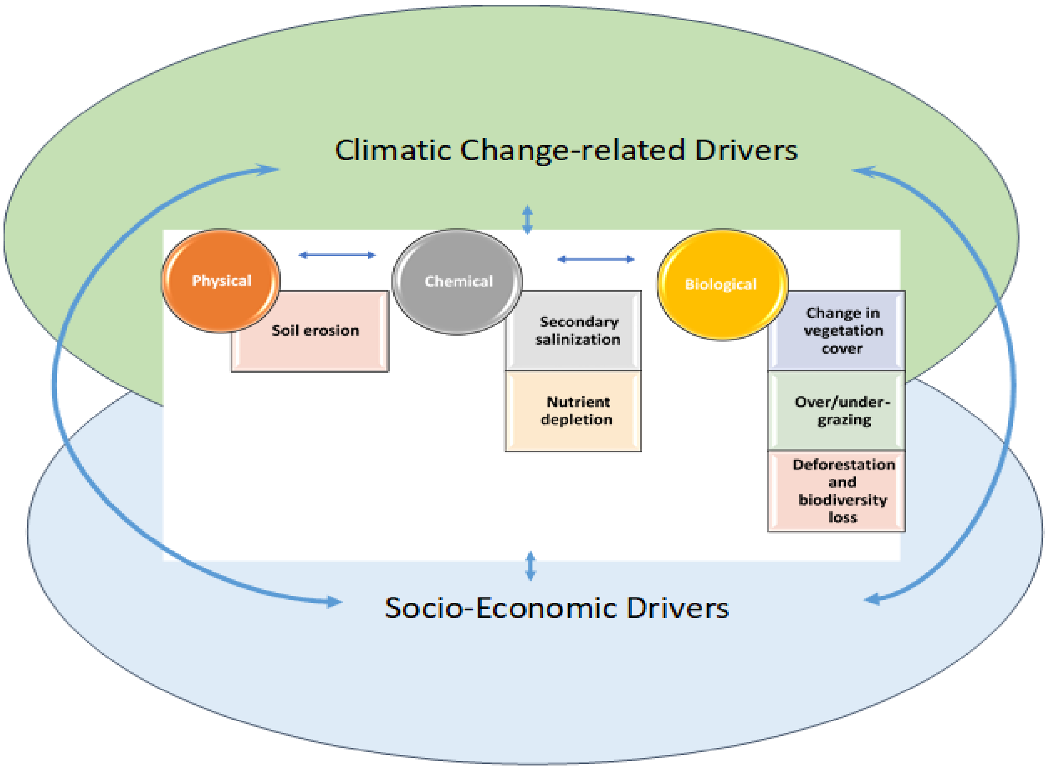

Figure 3.4 Interaction of climate change-related and human drivers of desertification

Drivers of desertification are factors which cause 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, 2004a; Reynolds et al., 2007b). Figure 3.4 presents a schematic framework of climate change-related and socio-economic drivers of desertification. Most of the processes and drivers of desertification are similar with the processes and drivers of land degradation in general. For this reason, they are summarised in Cross-chapter Box 3.1 (see also Chapter 4).

Cross-chapter Box 3.1. Processes and Drivers of Land Degradation, including Desertification

(Placeholder: will be completed together with Chapter 4)

Process	Focus	Proximate Drivers	Influence OF Climate Change	Influence ON Climate Change	References for CC->LD	References for LD->CC	General Reviews of the process
Wind erosion	Soil	Cultivation with poor cover, overgrazing, deforestation/vegetation clearing, dryland farming, Larger plot sizes, vegetation shifts - Documented reversal with vegetation restoration projects (Guo et al. 2017 - Remote Sensing)	Altered wind/drough patterns. Yet, no strong trends for combined humidity/wind speed/available energy assessments (Sheffield et al. 2012). Land use/cover more important than climate in US plains (Nordstrom & Hotta 2004). No clear long term trend on wind climate driving erosion in Sweden (Barring et al. 2003 - Catena), Climate chenge induced vegetation change enhance wind erosion (Munson et al. 2011)	Radiative cooling by aerosols (Tegen et al. 1996). Enhanced weathering + Ocean and Land fertilization + SOC burial (Quinton et al. 2010)	Sheffield et al. 2012 - Nature, Barring et al. 2003 - Catena), Munson et al. 2011 - PNAS	Tegen et al. 1996 - Nature, Quinton et al. 2010 - Nature Geosciences	Nordstrom & Hotta 2004 - Geoderma, Sterk 2003 - LDD, Earth et al., 2015
Water erosion	Soil/Water	Cultivation with poor cover, overgrazing, deforestation/vegetation clearing, poor tillage practices - More indirect: fires, vegetation shifts	Increasing rainfall intensity, drought and vegetation shifts (i.e. forest to woodland with climate change (Allen & Breshears 1998). Rainfall amount increases are amplified by erosion and run-off (Nearing et al. 2004). Increasing fire activity raises erosion (Shakesby 2011), Permaforst melting causing erosion (Jorgenson & Osterkamp 2005), Climate change effects mediated by biomass production appear more important than those through rainfall in modeling study (Pruski and Nearing 2002)	C emissions may be significant globally (approx 1 PgC/yr) Lal 2003	Allen & Breshears 1998 - PNAS, Nearing et al. 2004 - J Soil and Water Conservation, Shakesby 2011 - Earth Science Reviews, Jorgenson & Osterkamp 2005 - Can J Forest Research, Pruski and Nearing 2002 - WRR	Lal 2003 - Environment	Pesen et al. 2003 - Catena (Gully erosion),
Compaction/Hardening	Soil	Land use coversion,machinery overuse, intensive grazing , poor tillage/grazing management (e.g. under wet or waterlogged conditions)	Indirect, reduced SOC due to higher temperatures - More frequent wet/waterlogged periods	Complex effects on GHG emissions. Poor aereation has ambiguous effects on N2O emissions (Ball 2013).		Ball 2013 - Eur J Soil Science	Hamza & Anderson 2005 - S&TR
Salinization	Soil / Water	Irrigation with poor leaching+drainage, Deforestation	Sea level rise, Water balance shifts, Rise of salts after evaporation under drought	Reduced methane emissions with high sulfate			
Nutrient depletion	Soil	Insufficient replensihment of harvested nutrients		Reduced SOC, C release from soils - NO SYNTHESIS WORK FOUND			
Acidification	Soil	High cation depletion, Fertilization, Acid rain	--	Inorganic soil C release?			
Sodification	Soil / Water	Poor water management	Water balance shifts (precipitation/potential evapotranspiration shifts)	Effects of high alkalinity on GHG release??			
Organic matter decline	Soil	Cultivation, reduced plant input, higher decomposition	Raising temperature accelerating SOC turnover (Knorr et al 2002), warming explaining widespread SOC decline in UK (Bellamy et al. 2005), Accelereated global soil respiration due to warming (Bond-Lamberty & Thompson 2010)	Cultivation release of C through global cultivation expansion (Houghton 2003). Reversal due to land abandonement in Russia (Kurganova et al. 2014)	Knorr et al 2002 - Nature, Bellamy et al. 2005 - Nature, Bond-Lamberthy & Thompson 2010 - Nature	Houghton 2003 - Tellus, Kurganova et al. 2014 - Global Change Biology	Conant et al 2011 - Global Change Biology (response of SOC decomposition to raising temperature)
Toxicity	Soil	High cation depletion, Fertilization	--	--			
Waterlogging of dry systems	Water	Deforestation, Irrigation with poor drainage	Water balance shifts, rainfall increase, vegetation changes (forest to cropland-grassland)	CH4 (Fenner et al. 2011), Artificial riparian wetlands release CH4 (Altor and Mitsch 2006)		Hydrobiologia, Altor and Mitsch 2006 - Ecological Engineering, Hahn-Schofl et al.	
Drying of continental waters/wetland/lowland systems	Water	Upstream or Groundwater water consumption, intentional drainage, trampling/overgrazing, droughts	Extended drought causing vegetation dieback and soil degradation (McKee et al. 2004)	C stock reduction / C release in drying montane medows (Norton et al. 2011), N2O release from dried wetlands (Morse & Bernhardt 2013)	Mc Kee et al. 2004 - Global Ecology and Biogeography	Norton et al. 2011 - Ecosystems, Morse and Bernhardt 2013 - Soil Biol & Bioch	
Flooding	Water	Deforestation, Increasing impervious surface	Sea level raise, Water balance shifts, Increasing flood frequency (Milly et al 2002), Increased rainfall extremes in UK (Pall et al. 2011), Rapid climate change causes more floods now and in palerecords (Knox 2000)	CH4 and CO2 release from flooded soils. Boreal soils under reservoirs (Oelbermann & Schiff 2010). Abundant literature on paddy rice and methane.	Milly et al. 2002 - Nature, Pall et al. 2011 (attribution of extremes to CC), Knox 2000 - Quaternary Science Reviews	Oelbermann & Schiff 2010 - Ecoscience	
Eutrophication of continental waters	Water	Excess fertilization, Poor management of livestock/human sewage	Only indirectly (e.g. warming favouring N losses in the land) or interactively (warming x eutrophication compensating effects - Audet et al. 2017), Warming and algal blooms (Paerl & Huisman 2008)	Complex interaction between warming and eutrophication modulating GHG release in shallow lakes (Audet et al. 2017). High potential release of GHG with EUTROPH in humic lakes (Fenner & Freeman 2013). Also in reservoirs (Hattunen et al. 2003) - Vast literature on constructed wetlands for nutrient treatment and GHG emissions (e.g. Gui et al. 2007)	Audet et al. 2017, Paerl & Huisman 2008 - Science	Audet et al. 2017 - Freshwater Biology, Fenner and Freeman 2013 - Global Change Biology. Gui et al. 2007 - Water Science and Technology. Hattunen et al. 2003 - Chemosphere	
Woody encroachment	Plant	Overgrazing, Altered fire regimes	CO2 rise (Van Auken 2009, Wigley et al. 2010)	C sequestration interactive with climate gradients (Knapp et al 2008, Sala & Maestre 2014), increased terrestrial carbon stock decreases atmospheric CO2 (Hughes et al. 2006)	Van Auken 2009 - J Env Management, Wigley et al. 2010 - Global Change Biology	Knapp et al. 2008 - Global Change Biology, Sala & Maestre 2014 - J Ecology, Hughes et al. 20006 - Global Change Biology	
Valued species loss	Plant/Animal	Selective grazing, logging, and fire	Multiple	Multiple			
Invasions	Plant/Animal	Intentional and unintentional species introductions	Multiple	Multiple			
Insect outbreaks	Plant/Animal	Poor pest management practices	Multiple	Multiple			
Fire regime shift	Plant/Soil	Biomass accumulation, intentional burning, fire suppression policies, invasion by invasive species	Warming, Intensification of dry/wet alternative spells	massive C release in tropical peat fires (Page et al. 2002)		Page et al. 2002 - Nature	Certini 2005 - Oecologia (effects on soils)

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 (Nearing et al., 2015). On the other hand, there is low evidence about climate change impacts on wind erosion (Box 3.1, case study on soil erosion). Saline and sodic soils occur naturally in arid, semiarid and dry sub-humid regions of the world. Salinisation is the process whereby the concentration of dissolved salts in water and soil is increased by anthropomorphic processes. The threat of soil and groundwater salinisation induced by sea level rise and sea water intrusion are amplified by climate change (see case study on salinisation due to sea water intrusion). A major consequence of desertification is the reduction in soil carbon (C) and transfer of C from soil to the atmosphere. Global warming is expected to accelerate soil organic carbon turnover, in some areas leading to soil organic carbon decline (Box 3.1).

Warming of the tropical oceans was suggested to be responsible for the Sahel droughts and related desertification in the region (Giannini et al., 2008; Hoerling et al., 2006; Kerr, 2003; Knight et al., 2006). Similarly, the present recovery of precipitation in the Sahel region was related to the internal variability of water temperature in the Atlantic Ocean (Ting et al., 2009).

Invasive plants contributed to rapid desertification and loss of ecosystem services in the last century. Extensive woody plant encroachment altered runoff and soil erosion across much of the drylands and significantly contributed to desertification (case study on invasive plant species). 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 native conifers resulting in loss of biodiversity (Figure 3.5). This land conversion has resulted in desertification and reductions in forage availability, wildlife habitat, and biodiversity (Pierson et al., 2011, 2013; Miller et al., 2013).

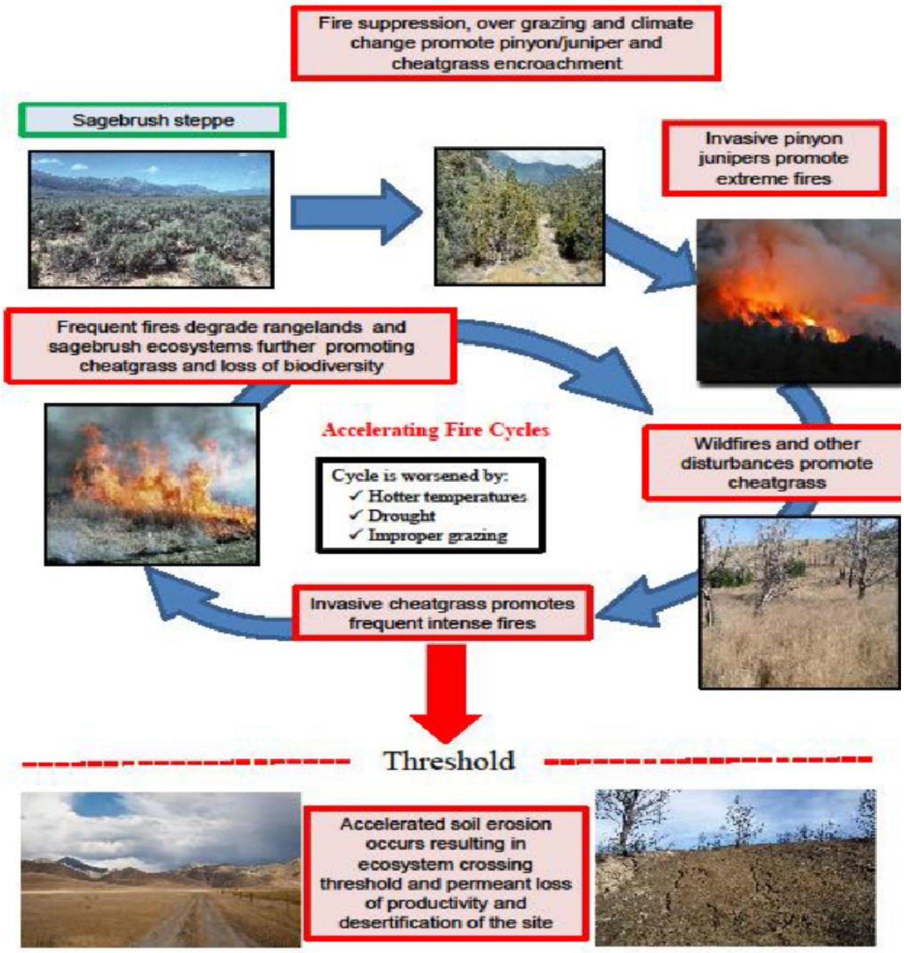

Figure 3.5 Invasive species cycle in the Great Basin region of the western U.S.A.

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. 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.

Predicted decreases in rainfall and increasing temperature across dryland areas of the world are likely to increase chances of wildfire occurrence (Jolly et al., 2015; Williams et al., 2001). 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).

3.2.3.2. 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 adequately prodigious (D'Odorico et al., 2013; Sietz et al., 2011; Yan and Cai, 2015; Sterk et al., 2016; Varghese and Singh, 2016; Mirzabaev et al., 2015; to list a few of recent ones) and there have been several comprehensive reviews and assessments of these drivers very recently (IPBES, 2018; UNCCD, 2017; Nkonya et al., 2016c). IPBES (2018) has identified cropland expansion, unsustainable land management practices, urban expansion, infrastructure development, and extractive industries as the main direct 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., 2013b). 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. Without research breakthroughs to mitigate these productivity losses, 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 be 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 is likely to exacerbate poverty among some categories of dryland populations (see 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). Poverty limits both capacities to adapt to climate change and availability of financial resources to invest into sustainable land management (SLM) (*robust evidence, high agreement*) (Gerber et al., 2014; Way, 2016; Vu et al., 2014).

Another key human driver which is likely to 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 (see section 3.5.2.4). Outmigration 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. On the other, it increases the costs of labour-intensive SLM practices due to lower availability of rural agricultural labour and/or higher rural wages. Outmigration increases the pressure on land if higher wages that rural migrants earn in urban centres will lead to their higher food consumption. At the same time, the remittances from higher earnings could be re-invested into sustainable land management. The net effect of these countervailing mechanisms is context-dependent. There is very little literature evaluating these joint effects of climate change, desertification and migration.

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, agricultural price distortions, centralised management of natural resources (d'Odorico et al., 2013; Geist and Lambin, 2004; Moussa et al., 2016; Mythili and Goedecke, 2016; Sow et al., 2016; Tun et al.,2014). However, 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 on these factors from such a perspective.

3.2.3.4 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. 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, 2008b; MEA, 2005). Reynolds et al. (2007) indicate that drylands being 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. This non-equilibrium nature of drylands led Behnke and Mortimore (2016) to conclude that the concept of desertification as irreversible degradation distorts policy and governance in the dryland areas. Mortimore (2016) suggests 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., 2013a; 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., 2013a; 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., 2011; 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). The most common definition for the drylands is based on the aridity index (UNEP, 1992). Aridity increased in many parts of the world over the last several decades, expanding the extent of drylands in some regions (*high agreement, robust evidence*) (Feng and Fu, 2013; Asadi Zarch et al., 2015; Ji et al., 2015; Spinoni et al., 2015;

Huang et al., 2016). 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). 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.

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 opinion, satellite observation of net primary productivity, use of biophysical models, and inventory of abandoned land, together provide a relatively holistic assessment but none on its own captures the whole picture (Gibbs and Salmon, 2015; Vogt et al., 2011).

3.3.1.1. Global Scale (here and in the following sections on regional scales and attribution, the information will be synthesised into graphic/tabular formats)

Early attempts to assess land degradation focused on expert knowledge to achieve global coverage rapidly and cost-effectively. Expert opinion continues to play an important role, because degradation remains a subjective quality whose benchmarks vary among locations (Sonneveld and Dent, 2007). On their initial quantification attempts, GLASOD (Global Assessment of Human-Induced Soil Degradation) estimated nearly 2 billion ha (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 three 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., 2011a).

A number of studies have used satellite-based remote sensing of vegetation related indices to investigate long-term changes in the vegetation and thus identify parts of the drylands undergoing desertification. The most widely used remotely sensed vegetation index is the Normalized Difference Vegetation Index (NDVI) which provides a measure of canopy greenness (Bai et al., 2008b; de Jong et al., 2011; Fensholt et al., 2012; Andela et al., 2013; Le et al., 2016b). All studies show a mixture of positive and negative NDVI trends, with positive trends dominating globally. By this measure there are regions undergoing desertification, however, the drylands are improving on average.

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 those parts of the time series, while it 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 1981-2011. 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. (2016) explicitly accounted for CO_2 fertilisation effect and found that the extent of degraded areas in the world is 3% larger when compared to the linear NDVI trend. 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.

Another vegetation index, Vegetation Optical Depth (VOD), is also 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. (2013a) used VOD trends to investigate biomass changes globally and found that VOD was closely related to precipitation changes in drylands. To complement their work with NDVI, Andela et al. (2013) also applied the RESTREND method to VOD. By interpreting NDVI and VOD trends together they were able to differentiate changes to the herbaceous and woody components of the biomass. They reported that many dryland regions are experiencing an increase in the woody fraction often associated with shrub encroachment and suggest that this was aided by CO_2 fertilisation.

Biophysical models utilise global data sets that describe climate patterns and soil types, combined with observations of land use, to define classes of potential productivity and map general land degradation (Gibbs and Salmon, 2015). For example, Cai et al. (2011) mapped marginal productivity using a biophysical model of agricultural productivity based on spatial descriptions of soil type, topography, average air temperature and precipitation, combined with expert opinions and global land cover data sets. Marginal areas with low-productivity cropping were designated as abandoned, idle, or wasted, while marginal areas with full cropping were designated as degraded. Improvement to capture and classify broader contexts of degradation is influenced by the quality of the data used for calibration and the model suitability.

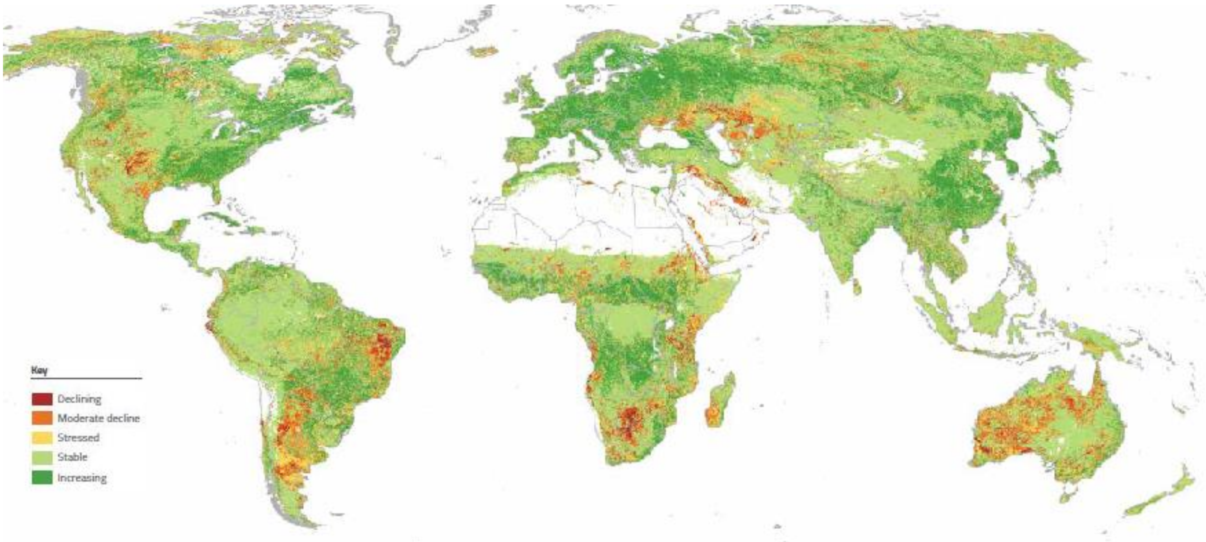

Figure 3.6 Global Land Productivity Dynamics between 1999 and 2013

Source: UNCCD (2017)

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 this report approximately 9% of global land area with more than 50% of cropland and 5% of global rangeland suffers from 8 to 143 issues that trigger land change processes that are relevant to land degradation. Figure 3.6 shows areas of declining, moderately declining, stressed, stable and increasing land productivity.

3.3.1.2. Regional Scales [more assessment towards improved regional balance will be made in the second order draft]

The area of drylands increased globally during the last three decades as compared to the period of 1951-1980, especially in north-eastern Brazil, southern Argentina, the Sahel, Zambia and Zimbabwe, the Mediterranean area, North-Eastern China and sub-Himalayan India (Spinoni et al., 2015). Damberg and AghaKouchak (2014) found that south-western United States, Texas, parts of the Amazon, the Horn of Africa, northern India, and parts of the Mediterranean region, experienced drying over the last three decades, whereas wetter conditions were experienced in central Africa, Thailand, Taiwan, Central America, northern Australia, and parts of eastern Europe. Using Satellite and rainfall data, Helldén and Tottrup (2008) highlighted a 'greening up' trend in the Sahel belt, Mediterranean and East Asia between 1982-2003 with greening in South Africa and South America being weak compared to other parts of the world. Prăvălie (2016) found that in Africa and Asia, which have the most extensive drylands, desertification is currently affecting 46 African states and 38 Asian states. Desertification through salinisation is a major concern in river basins across the drylands as it impacts both food and water security and may be exacerbated by climate change (D'Odorico et al., 2013). Examples of major river basins undergoing salinisation include: San Joaquin Valley and Colorado River Basin in the United States (Qadir et al., 2007), Indo-Gangetic Basin in India Lal and Steward, 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, 2013), Aral Sea Basin of Central Asia (Cai et al., 2003), Euphrates Basin in Syria and Iraq (Sarraf, 2004), Joran River Valley (Ammari et al., 2013) and the Murray-Darling Basin in Australia (Rengasamy, 2006).

In Africa, the sub-humid zones of the Congo, Zambezi, Nile, Niger and Chad river basins were subject to an increasing severity and extent of soil erosion during the recent decades. Moreover, the soil loss through run-off is sixteen times higher in bare degraded soils of the Sahel than in the sub-humid zones where soils are more structured (Lamourdia and Ignacio, 2007). Taking a longer-term perspective, Thomas and Nigam (2018) found that the Sahara Desert has expanded by 10% over the 20th century (also see 3.3.2).

In Asia, according to Miao et al. (2015), the changes in drylands over the period 1982–2011 were mixed, with some areas experiencing vegetation improvement while others showed reduced vegetation. Forests demonstrated more resilience than grasslands, shrubs and sparse vegetation which were extremely sensitive to short-term climatic variations and human activities (Jiang et al., 2017). Xue et al. (2017) adopted remote sensing (RS) images from four periods (1975, 1990, 2000, and 2015) to classify the intensity of aeolian desertified land (ADL) in north Shanxi in China, and found that ADL 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). Li et al. (2015) showed that there was a "wet-west and dry-east" anomaly pattern over the drylands in northern China, which agree with the results from previous studies through analyses of daily rain gauge records in this region (Gong et al., 2004; Zhai et al., 2005). 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). The precipitation generally increased during 1930 to 2009 (Chen et al., 2011), which was supported by the evidence of increasing moisture from tree-ring reconstructed Palmer Drought Severity Index (Li et al., 2006). NDVI and gridded high-resolution land data analysis (1984–2013) showed that vegetation significantly decreased for shrubs and sparse vegetation due to droughts in the Karakum and Kyzylkum Deserts, the Ustyurt Plateau and the wetland delta of the Aral Sea.

In Australia, 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 the Photosynthetically Active Radiation absorbed by vegetation derived from AVHRR satellite data (Donohue et al., 2009). However, from 2002 to 2009 much of eastern Australia was affected by drought. Burrell et al. (2017), using the AVHRR-derived NDVI for the period 1982-2013, also found widespread greening over Australia, though now, post-drought, eastern Australia is also dominated by greening. 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. Burrell et al. (2017) went further by extending the RESTREND methodology to account for the non-monotonic nature of the vegetation change which they 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 Burrell et al. (2017).

In Latin America and Caribbean, Morales et al. (2011) indicated a range of figures from 75% of the land affected by some level of land degradation in Argentina, 8% in Brazil, 34% in Perú. Nearly 24% of the regional area is considered arid where irrigation is fundamental to sustain agricultural productivity and shows some degree of land degradation. The estimates of soil degradation by different processes show that south and central America have a total degraded land area of 3 Mkm^2, with a decreasing trend in net primary productivity, and decline in ecosystem productivity (Zdruli et al., 2010). Becerril-Pina et al. (2015) used a desertification trend risk index (DTRI) based on Landsat images and estimated that semi-arid regions of Central Mexico have experienced positive trends and are at high risk.

In the Middle East and North Africa, three quarters of variation in soil erosion was explained by topography, while the remaining share was due to wind erosion in Jordan (Al-Bakri et al., 2016). 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. The "greening-up" process for the Sahel region (Helldén and Tottrup, 2008) was not observed in the North African steppes (Hirche et al., 2011). In Central North Kordofan state in Sudan, about 120,000 km^2 of land were estimated to be desertified in early 2000s. However, the reforestation measures in the last decade sustained by improved rainfall conditions, have led to low-medium regrowth conditions in about 20% of that area (Dawelbait and Morari, 2012). Drought conditions increased during the period 1980 to – 2014 in Israel increasing the risks of wildfires (Turco et al., 2017). Desertification also increased substantially in Iran starting from the 1930s, and despite numerous efforts to rehabilitate degraded areas and limit the further spread of 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.

The 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. 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). 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.

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).

3.3.2. Attribution of Desertification

Desertification is a result of complex interactions within coupled human-natural systems (Figure 3.4. 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 such as the land productivity may return entirely once the drought ends (Kassas, 1995). 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, were 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. 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 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 these 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).

Attribution of vegetation changes to human activity has also been done within modelling frameworks. 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).

Probabilistic event attribution (PEA) methodology showed that the dominant influence for droughts in Eastern Africa during 2016-2017 was the 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 (Peter et al., 2017) A strong warming tendency in the western Indian Ocean was attributed to increases in greenhouse gasses (*medium confidence*) (Verdin et al., 2005; Funk and Williams, 2010).

There are numerous local case studies on attribution of case studies, 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) were attributed to two-thirds of the observed expansion of the Sahara Desert from 1920-2003 (Thomas and Nigam, 2018). Drought was suggested as the main driver of desertification in Africa (Masih et al., 2014). However, some other studies suggested that although droughts contribute to desertification, the underlying causes of desertification are human activities, for example, 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). Brandt et al. (2016) found that woody vegetation trends are negatively correlated with human population density. Changes in land use, water pumping and flow diversion have enhanced drying of wetlands and salinisation of freshwater aquifers in Israel (Inbar, 2007). The dryland territory of China is very sensitive to both climatic variations and land use and land cover changes (Fu et al., 2000; Liu et al., 2008; Liu and Tian, 2010; Zhao et al., 2013, 2006). In the Yulin region in China between 1986 and 1995 desertification by human activities is the dominant influence, while the role of changing climatic conditions is quite minimal (Wang et al., 2012). There is also evidence that socioeconomic factors are 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 Ararat plains in Armenia, the irrigated meadow-brown soils have experienced a reduction in humus content and absorbed basis is being observed in the arable layer of soils (Ghazaryan et al., 2016). In Iran, human and climatic factors combined in leading to more severe droughts between 1950 and 2010 (Modarres et al., 2016). Greening of some rangeland areas in Uzbekistan between 2000 and 2011 was attributed to climatic variations. 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 may 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.

The major conclusion is that, with all the shortcomings of individual case studies, relative roles of climatic and human drivers are context-specific and evolve over time. However, this has been known for a long time now. 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 (3.2.3.2), 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 (see chapter 2) here we focus on their effects in dryland regions. The main feedback pathways discussed are summarised in Figure 3.7.

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 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.

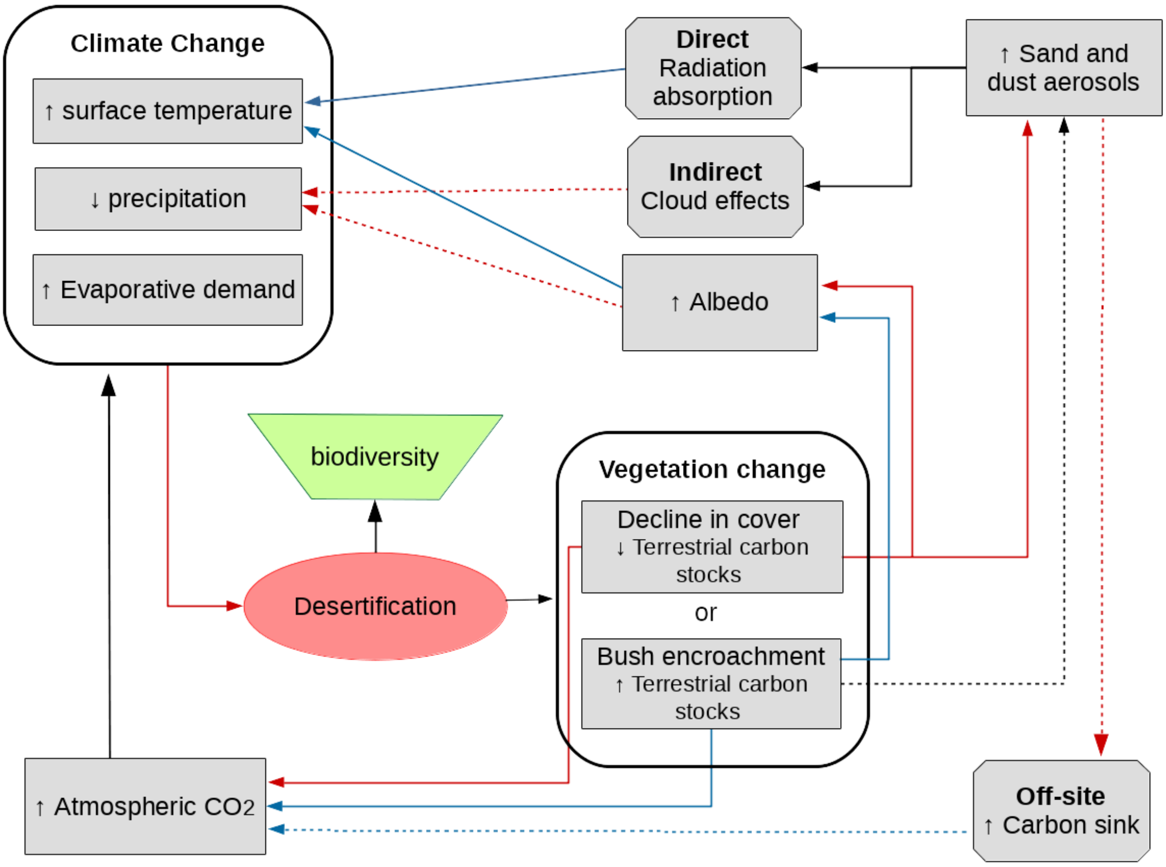

Figure 3.7 Schematic of main pathways through which desertification can feedback on climate

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”. These events can play an important role in the local energy balance. By reducing vegetation cover and drying the surface conditions, desertification can increase the frequency of these events. These events impact the regional climate in several ways. 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). 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 a negative relationship between vegetation green-up and the occurrence of dust storms (Engelstaedter et al., 2003; Fan et al., 2015; Zou and Zhai, 2004). 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. 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, 2014; 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 studies finding 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 direct effect of dust on the ocean surface has been found to be a cooling effect (Doherty and Evan, 2014; Evan et al., 2009, 2010).

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).

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 (Meng et al., 2014b,a; Evans et al., 2017), 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. 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).

In comparison to other climate zones, the change in albedo is generally considerably larger in drylands, particularly those located between 20-30°N and 20-30°S. Within these latitudes, descending dry air of the Hadley circulation leads to the observed aridity of drylands, and in addition, reduces cloud cover. In these types of locations, the change in albedo following the reduction of vegetation can be of greater significance (Bird et al., 2008).

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., 1990a). 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 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, low vegetation, it should be noted that not all drylands necessarily have low biomass and carbon stocks in an intact state. Vegetation types such as the subtropical thicket of South Africa have over 70 tCha^{-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).

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 tCha^{1} to 5.0 tCha^{-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 tCha^{-1} over a 50-70 year period (Hudak et al., 2003). In terms of potential changes in soil organic carbon stocks, the effect may be dependent on annual rainfall. Whereas increasing woody cover generally leads to an increase in soil organic carbon stocks in drylands that have less than 800mm 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 Nature's Contributions to People in Drylands

The Millennium Ecosystem Assessment (2005) proposed four classes of ecosystem services: provisioning, regulating, supporting and cultural services. The ecosystem services in drylands are vulnerable to the impacts of climate change due to higher variability in precipitation and low soil fertility (Enfors and Gordon, 2008; Mortimore, 2005).

Desertification coupled with climate change negatively impacts the provisioning of services, particularly food and fodder production (Hopkins and Del, Prado 2007). Furthermore, it modifies 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.

There is a robust evidence that climate change and desertification have negative impacts on crop yields in drylands, in some cases substantially so. World Bank (2009) projected that, without the carbon fertilisation effect, climate change will reduce the mean yields for 11 major global crops 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 (also see Chapter 5).

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 soil organic carbon (SOC) (FAO, 1995). Drivers of soil degradation and a loss of SOC include a reduction in organic matter input into soil (Albaladejo et al., 2013; Hoffmann et al., 2012; Lavee et al., 1998; Martinez-Mena et al., 2008; Rey et al., 2011), increasing soil salinity and soil erosion (Lavee et al., 1998) and intensive grazing (Sharkhuu et al., 2016). In contrast, if the soil management includes soil conservation practices combined with irrigation, the cropland had a higher SOC content than native shrubland or native pastures, as shown in China (Liu et al., 2011). However, the 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).

Precipitation rather than changes in temperature is considered to be the principle determinant of the capacity of drylands to sequester carbon (Fay et al., 2008; Hao et al., 2008; Vargas et al., 2012; Mi et al., 2015; Serrano-Ortiz et al., 2015; 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; Hao et al., 2008; Fay 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. 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).

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. 2007b). 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).

Globally, it is estimated that between 7%-24% of the plant species will be extinct within 100 years due to climate change (Malcolm et al., 2006; van Vuuren et al., 2006). Malcolm et al. (2006) found that more than 2000 plant species located within dryland biodiversity hotspots could become extinct (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) A global survey of 224 dryland ecosystems across 16 countries identified the preservation of perennial plant biodiversity as imperative to reducing the impact of climate change and desertification (Maestre et al., 2012).

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, 1996b). 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 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 may 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).

The opening of artificial waterholes may provide a short-term solution to a reduction in surface water availability, but such management interventions may have unintended consequences. A consequence is the piosphere that forms around artificial waterholes where increased intensity of animal activity often reduces habitat integrity by reducing habitat heterogeneity, biodiversity and ecosystem resilience. The use of artificial waterholes is also species specific and may disadvantage water independent species (Gaylard et al., 2003). Even if the provision of artificial water reduces mortality of water-dependent species in the short-term, artificially maintained population numbers result in increased herbivory pressure, particularly by large herbivores (Wato et al., 2016), and may ultimately increase the risk of starvation for a multitude of species. Furthermore, congregation around a fixed resource may increase disease transmission (Nunn et al., 2014). These findings have led to the closure of a number of artificial water holes, particularly in ecosystems where natural water sources exist. Careful ecological assessments help in planning additional waterholes.

3.5.2. Socio-economic Systems

The impacts of climate change-desertification interactions on socio-economic development in drylands are complex. Figure 3.8 schematically represents these impacts at first (inner circle) and second (outer circle) levels, adopting the framework of Sustainable Development Goals (SDGs). The first level impacts on poverty, and food and nutritional security are mediated through changes in agricultural productivity (see also section 3.5.1 and Chapter 5). Those on human health and infrastructure come from dust storms and floods (see section 3.4 and 3.6) (that are separated here from human health impacts through malnutrition and hunger, discussed in Chapter 5). The impact on conflict is through competition for land and water resources whose availability is affected by desertification and climate change. These first level socio-economic impacts are the main focus of this section. Naturally, this is a simplification of a myriad of relationships between and within SDGs and climate change-desertification interactions. For example, climate change also affects poverty through non-agricultural mechanisms such as an increase in the frequency, scale and intensity of floods and hurricanes, as well as changes in vector-borne diseases. The choice to focus on the agricultural productivity link is dictated by a desire to focus on climate change-desertification interactions per se, rather than comprehensive impacts of climate change on poverty. Similar reasoning applies to the other mechanisms of socio-economic impacts of climate change and desertification considered here.

The second level effects on gender inequality, decent work and economic growth, access to affordable energy, quality of education, reduced inequalities, responsible consumption and production, and sustainable cities and communities are discussed only briefly through their links to those SDGs which are considered as directly impacted, but are subject of focus in terms of their links to climate change-desertification in 3.7.3 on policy responses. The impacts of desertification and climate change even at what is depicted here as first level, are extremely intricate and 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). There is robust evidence however, that climate change is expected to exacerbate the already high vulnerability of dryland populations to desertification, and that the combination of pressures coming from climate change and desertification contribute to amplifying, in interaction with other contextual factors, poverty, food and nutritional insecurity, disease burden and the likelihood of conflict.

Figure 3.8 Socio-economic impacts of desertification and climate change with the SDG framework

3.5.2.1 Food and Nutritional Insecurity

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 (see 3.5.1, Chapter 5). It is also clear that desertification reduces agricultural productivity, food availability and access (see also 3.5.2.2). 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 (Tun et al., 2015; Kirui, 2016; Moussa et al., 2016; Mythili and Goedecke, 2016).

Negative impacts of climate change on agricultural productivity contribute to higher food prices, especially since food demand will grow due to growing population and incomes. The misbalance between supply and demand for agricultural products will likely increase agricultural prices in the range of 31.2% for rice to 100.7% for maize by 2050 (Nelson et al., 2010), and cereal prices in the range of a 32% increase to a 16% decrease by 2030 (Hertel et al., 2010). 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. About 815 million people globally were estimated to be food insecure in 2016, of whom 508 million are in Asia, 243 million in Africa and 43 million 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).

Overall, there is robust evidence for the high potential negative impact of climate change on agricultural productivity in drylands. 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. 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 (see section 3.2.2), is often conceptualised as one 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. There is growing evidence on climate change impacts on poverty (Hallegatte and Rozenberg, 2017). Under scenarios of low agricultural productivity due to climate change leading to high agricultural prices, poverty rates were found to increase by about one-third among the urban households and non-agricultural self-employed in Malawi, Uganda, Zambia, and Bangladesh, whereas poverty rates fell substantially among agricultural households in Chile, Indonesia, Philippines and Thailand, because higher prices compensated for productivity losses (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. On the other hand, most of the research on links between poverty and desertification (or more broadly, land degradation) focused on whether or not poverty is a cause of land degradation (Gerber et al., 2014; Vu et al., 2014; Way, 2016). However, the literature quantifying to what extent desertification contributes to poverty per se is surprisingly thin, i.e. beyond the impacts of desertification on agricultural productivity and incomes. Moreover, at the global scale, there is low quantified evidence causally linking desertification to poverty, as the related literature remains qualitative or correlational (Barbier and Hochard, 2016; Nachtergaele et al., 2010). 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 during 2010 to 2011 (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 hypothesises 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 Dust Storms and Human Health

The frequency of dust storms is increasing due to land use and climatic changes (Gu et al., 2010; Indoitu et al., 2015; Rashki et al., 2012; Tan et al., 2012). 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 (Goudie, 2014; Goudarzi et al., 2017; Díaz et al., 2017; 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 (Zhang et al., 2016; Díaz et al., 2017; Samoli et al., 2011; Lee et al., 2014; Kashima et al., 2016; Bennett et al., 2006).

The primary health effects of dust storms include damage to the respiratory system (including asthma, tracheitis, pneumonia, allergic rhinitis and silicosis) and the cardiovascular system (including stroke, conjunctivitis, Meningococcal Meningitis, valley fever, skin irritations), and others (Goudie, 2013). Dust particles with a diameter smaller than 2.5 μm 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 10 μgm^{-3} increase in PM10 dust particles was associated with mean increases in non-accidental mortality from 0.33% to 0.51% across different calendar seasons in China, Japan and South Korea (Kim et al., 2017). 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). Molesworth et al. (2003) showed that Meningococcal Meningitis epidemics occur throughout Africa during the dry with dusty conditions. 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.

In summary, there is *high agreement* on the negative health effects of dust storms on human health. In view of growing intensity, frequency and scale of dust storms due to climate change-desertification interactions, these health damages 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.

3.5.2.4 Conflicts and Migration

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, 2014a; Hsiang et al., 2013; Maystadt and Ecker, 2014), competition between pastoralist communities and crop producers for access to land (Abbass, 2014; Huho et al., 2011; Benjaminsen et al., 2012). There is *high agreement* that desertification and climate change do not cause conflict and civil strife by themselves, but add to the overall conflict potential. For example, the probability 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 change in climate toward warmer temperatures or more extreme rainfall was found to increase the frequency of interpersonal violence by 4% and intergroup conflict by 14% (Hsiang et al. 2013). It was also suggested that water scarcity and drought played a direct role in the Syrian conflict (Gleick, 2014). On the other hand, climate variations as recorded by the Palmer Drought Severity Index (PDSI) and global temperature records 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 were suggested 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 (Gleick, 2014b; Kelley et al., 2015; Butler and Kefford, 2018; Raleigh, 2010), where there may already be a conflict potential due to other reasons, for example, ethnic divisions (Schleussner et al., 2016).

There is growing evidence that migration is used as an adaptation strategy in the context of environmental change (*medium evidence, high agreement*). However, the narrative of 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 et al., 1995; Myers, 2002), 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 (Sherbinin and Bai, 2018; Hunter et al., 2015; McLeman, 2011), 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 (McLeman, 2013; Gemenne, 2011) and they do not take into account individual agency in migration decision nor adaptive capacities of individuals (Kniveton et al., 2011; Hartmann, 2010; Piguet, 2010). Accordingly, the available micro-level evidence suggests that climate-related shocks are one of the drivers of migration (Melde et al., 2017; Adger et al., 2014; London Government Office for Science and Foresight, 2011), 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 (Mueller et al., 2014; Gray and Mueller, 2012) 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

(Missirian and Schlenker, 2017; Mueller et al., 2014; Gray and Mueller, 2012). 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 (Gray and Mueller, 2012; Findlay, 2011), 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 (Nawrotzki et al., 2016; Nawrotzki and DeWaard, 2016; Nawrotzki et al., 2015; Feng et al., 2010). 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 2010, 2009) 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 (Kubik and Maurel, 2016; Koubi et al., 2016; McKenzie and Yang, 2015).

3.5.2.6 Damages to Transport Infrastructures

Sand storms and movement of sand dunes threaten the safety and operation of railway and road infrastructures in arid and hyper-arid areas, and lead to road and airports 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 (Cheng and Xue, 2014; Bruno et al., 2018). In China, sand dune movements are periodically disrupting the railway transport in Linhai-Ceke line in north-western China, 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 and Middle East, for example, "The Belt and Road Initiative" promoted by China, or the Gulf Railway project by the countries of the Arab Gulf Cooperation Council (GCC). 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.

3.6. Future Projections

3.6.1. Future Projections of Desertification

(the information here will be synthesised into graphical/tabular illustrations during the next phase)

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 studies regarding the future evolution of desertification rely on the analysis of specific climate change scenarios or 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 potential evapotranspiration will increase worldwide as a consequence of increasing surface temperatures and surface water vapour deficit (Sherwood and Fu, 2014a). Consequently, there will be associated changes in aridity indices that depend on this variable (*high agreement, robust evidence*) (Dai, 2011; Dominguez et al., 2010; Ficklin et al., 2016; Feng and Fu, 2013; Greve and Seneviratne, 2015; Asadi Zarch et al., 2017; Lin et al., 2015; Cook et al., 2014b; Scheff and Frierson, 2015a). Due to the large potential increase in evapotranspiration and decrease in precipitation over some subtropical land areas, a drying trend is expected to occur over a portion of drylands (Zhao and Dai, 2015), with one model estimating an approximately 10% increase in the world's desert areas (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; Seidel and Randel, 2007; Johanson et al., 2009), and will continue expanding due to global warming (Lu et al., 2007; Johanson et al., 2009). This expansion leads to the poleward extension of sub-tropical dry zones and hence an increase in drylands (Scheff and Frierson, 2012).

Increases in potential evapotranspiration and associated dryland expansion, are projected to continue due to climate change (Cook et al., 2014b; Lin et al., 2015; Scheff and Frierson, 2015b; Fu et al., 2016), expanding global dryland areas by approximately 10% by the end of this century (Feng and Fu, 2013). (Huang et al. 2016a) claim that the evaluation based on the Fifth Coupled Model Intercomparison Project may have underestimated the expansion of drylands and therefore the risk of expanding desertification could be even higher. Regional studies confirm the outcomes of Global Climate Models (Terink et al., 2013; Yin et al., 2015; Marengo and Bernasconi, 2015; Cook et al., 2012; Nastos et al., 2013; Coppola and Giorgi, 2009). UN Ecosoc (2007) projects an annual increase of 5% to 8% of arid and semi-arid lands in Africa.

The projected increases in the aridity index have *very high confidence*. However, several studies have challenged the use of the aridity index, and potential evapotranspiration more generally, as reliable measures of the amount of moisture available to sustain life in terrestrial ecosystems. Work on CO_2 fertilisation has suggested that potential evapotranspiration may not be 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 (Swann et al., 2016). Roderick et al. (2015) show that in climate models, evapotranspiration does not increase at the same rate as potential evapotranspiration and suggest that the aridity index (AI) is not a good measure of aridity. 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 zones may not be the right approach in a changing climate.

Miao et al. (2015b) use a linear regression between observed NDVI and growing season precipitation and temperature to determine the climate dependence of the vegetation. They then estimate future vegetation using projections of precipitation and temperature from the CMIP5 ensemble. The study showed an acceleration of desertification trends under the RCP8.5 scenario in the middle and northern part of Middle Asia, north western China except Xinjiang and the Mongolian Plateau.

Climate strongly affects the mechanisms that explain aeolian desertification. Wang et al. (2009) assessed future aeolian 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 the area at risk of salinisation will 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), and surface hydrology and vegetation that affect wetlands and favour salinisation (Nielsen and Brock, 2009).

3.5.1.1. *Future Vulnerability and Risk to Desertification*

Climate change is expected to further exacerbate the vulnerability of dryland ecosystems to desertification by increasing potential evapotranspiration globally (Sherwood and Fu, 2014b) leading to increasing aridity in locations which would not experience similar increases in precipitation. Surface drying was found to be highly likely in the Mediterranean, southwest USA and southern African regions by the end of the 21st century (Wgi et al., 2016). Additional warming of between 2°C and 4°C is projected in drylands by the end of the 21st century under RCP4.5 and RCP8.5 scenarios, respectively (IPCC, 2013).Droughts are extreme meteorological events that 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 in 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*) (Sheffield and Wood, 2008; Wang, 2005; Dai, 2013; Swann et al., 2016; Zhao and Dai, 2015). Although 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 their 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 climate variability patterns and how these may change in the future. The responses of ecosystems depend on diverse vegetation types. Drier ecosystems are more sensitive to changes in precipitation and temperature (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). 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 areas of increased vulnerability to climate change and land desertification will largely surpass those that will experience improvements (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, the mean erosion potential will increase 45% from the estimated 104.52 $t.ha^{-1}$ yr^{-1} in the baseline period 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) In Narmada river basin, India, climate change is projected increase the sediment load by up to 60% from the current erosion rate by 2080. Values are also influenced by slope and degree of clay, SOC loss will increase especially in steeper slopes, sandy soils and fallow lands (Mondal et al., 2017).

Dryland expansion will lead to reduced carbon sequestration and enhanced regional warming, increasing the risk of desertification (Huang et al., 2017), with severe impacts on land usability in Africa and threatening its capacity to provide food security. At biome-scale, terrestrial carbon uptake is controlled mainly by weather variability. The area of the land surface in which dryness controls

CO_2 exchange has increased by 6% and is expected to increase by at least another 8% by 2050. In these regions net carbon uptake is about 27% lower than others (Yi et al., 2014). Huang et al. (2016a) found that accelerating expansion of drylands under RCPs4.5 and 8.5 will result in a substantial change on drylands surface (covering up to 50% of total land surface). Carbon sequestration will be reduced, enhancing regional warming over humid regions, which will result in higher risk of land degradation and desertification.

3.7. Responses to Desertification under Climate Change

The "drylands development paradigm" refers to social and technical ingenuity of dryland populations as a driver of dryland sustainability (Reynolds et al., 2007b; Safriel and Adeel, 2008). Technical and social ingenuity involves adapting mainly agricultural livelihood activities to new environmental conditions through both technological and organisational innovations and changes. However, increasing population pressures and potentially unprecedented nature of climatic changes could push dryland populations beyond their resilience thresholds, beyond the limits for their autonomous adaptation, requiring policy 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 largely undertaken by dryland households and communities at micro-level, and finally to broader policy responses.

Avoiding the "desertification paradigm" and 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). Agriculture is likely to remain the principle source of food production and the foundation of livelihoods across the drylands. In addition, drylands provide vital ecosystem services to downstream urban economies and industry.

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. 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).

An initial 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 degradation. Chapter 6 then considers emerging potential interlinkages in the form of co-benefits, side effects and synergies that may arise through implementation.

3.7.1. Technologies and SLM Practices: on the Ground Actions

A broad range of activities and measures exist that can potentially avoid, reduce and reverse degradation across the dryland areas of the world. Many of these actions are also climate-smart, contributing to climate change adaptation and mitigation, with sustainable development co-benefits. An assessment is made of five actions that have historically been widely considered within the dryland domain, summarised within a matrix of corresponding biomes and anthromes (Figure 3.9). The suite of actions is by no means 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, including more mesic areas (Figure 3.9).

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.

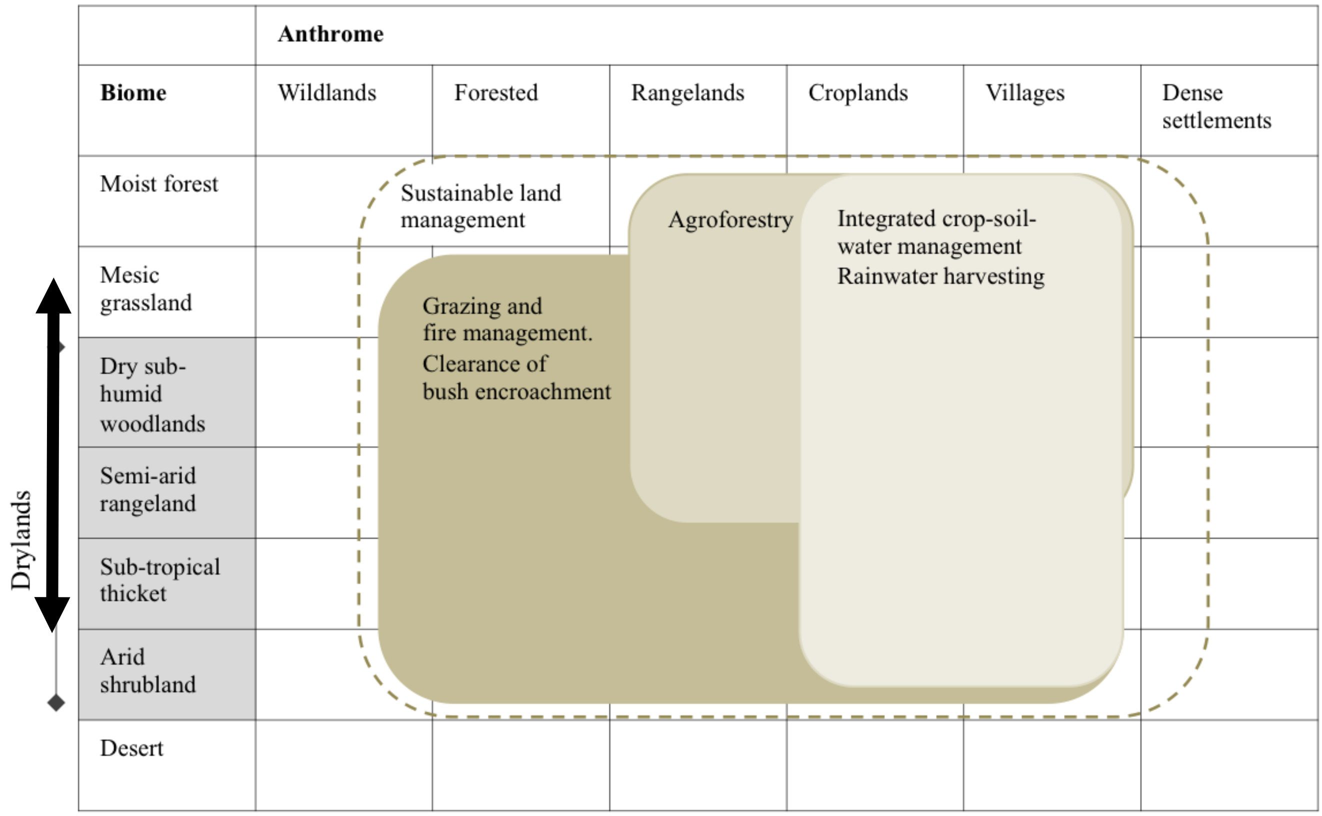

Figure 3.9 The typical distribution of on the ground actions across global biomes and anthromes

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 ‘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).

A number of forms of intercropping and relay cropping 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 (Wilhelm and Wortmann, 2004; Zhang et al., 2007; Altieri and Koohafkan, 2008; Tanveer et al., 2017). For example, intercropping involving maize, desmodium forage legume (insect repellant intercrop) and brachiaria grass (as insect trap plant) has shown merit in drylands of east and central Africa as climate smart and profitable compared to conventional practices (Khan et al., 2014). Productivity of maize increased by two-three fold, while over 80% of stem borers were removed.

Farmers in the drylands usually use location specific farming practices well adapted to soil and environmental conditions including mixed cropping, crop rotations in combination with drought resistant crop cultivars and water management techniques. Traditional farming practices include rotation of maize-bean in Northern Tanzania, millet-watermelons in West Sudan, and Millet and Senegalia (*Senegalia senegal*) in Kordofan area of Sudan integrated with water harvesting, terracing, and lifesaving irrigation from ponds and wells.

Surface runoff is a major factor contributing to degradation of soil resources with a negative impact on ecosystem services. Although it is known that adequate canopy cover or mulching by itself can reduce runoff (González-Núñez et al., 2004), using deep and coarse rooted legume plant species such as *Melilotus*, *Lathyrus* and *Linum* as cover/inter crops can result in up to 17% reduction in rainfall loss (Yu et al., 2006).

A further action employed in croplands to avoid and reverse degradation is the use of different forms of agroforestry and shelter belts (also see case study on “green walls”). Tree based wind shelters were established in the pastoral regions of Northern China in the late 1970s (Yu et al., 2006). Site-specific size of shelter belts was recommended based on local conditions, wind speed and types of plant species used. Mixed species were used in 50–600 m size and 4–30 m wide belts in perpendicular orientation to direction of wind (Wang et al., 2008). Shelter belts reportedly reduce wind speed, reduce soil temperature by up to 40% and increase soil moisture by 30%.

A further prominent problem within dryland agriculture is increasing salinisation that at present covers approximately 995 million hectares in the arid and semiarid regions. Saline areas are rapidly increasing as a result of a lack of water, erosion and poor management practices. In turn, it reduces the productive capacity of soil leading to high seed mortality, poor germination and reduced yield. Salinity often increases to the extent that cultivation of normal crops becomes impossible. Potential response measures including leaching and drainage, however this can be prohibitively expensive. Halophytes are therefore often used. Their adoption is not new; they were used during the ancient civilisation of Mesopotamia, but it allows farmers to reclaim farmland in a profitable manner.

3.7.1.2. Grazing and Fire Management

Humankind’s use of grazing animals began approximately 8000 years ago (Mignon-Grasteau et al., 2005) and has continued to be a dominant land use in arid, semi-arid, and dry sub-humid ecosystems. Dryland rangelands support the majority of the world’s grazing lands and livestock production (Safriel et al. 2005). Light and moderate grazing pressure has generally been shown to have minimal impact on ecosystem services (Papanastasis et al., 2017). However, overgrazing by domestic livestock has been documented as a contributing mechanism of desertification (Manzano and Návar, 2000; Geist and Lambin, 2004; Havstad et al., 2006; Huang et al., 2007; D’Odorico et al., 2013) with resulting loss of ecosystem services. When correctly implemented sustainable grazing can be achieved, providing a major source of food, fibre, leather, transportation, and serving as an engine of economic growth. The principles of sustainable grazing management require that systematic monitoring of climate, vegetation and animal health be implemented to adjust livestock numbers to current climatic conditions and vegetation production.

Tools and techniques of sustainable grazing management are specific in time and place and must be developed for each location of interest. They require the land manager adjust when and where they graze based on climatic conditions (cumulative precipitation and temperature) and pasture conditions. Continuous heavy grazing during droughts can result in loss of grasses and basal cover, increases in woody plants, accelerated soil erosion, increases in exotic invasive weeds and permanent loss of ecosystem services (Archer et al., 2017). Management tools to achieve sustainable grazing include adjusting the following components of a grazing system: 1) season of use; 2) duration of use; 3) stocking rate (animals per unit area); 4) type of livestock (sheep, goat, cattle, horse, donkey, camel); 5) class of livestock (e.g., yearlings vs bulls for cattle); 6) using herders to move animals; 7) fencing to control access; 8) water development and access to water; 9) location of salt/mineral block; 10) shade structures; 11) supplemental feeding; and 12) predator control (Bailey, 2005; Ganskopp, 2001; Fuhlendorf and Engle, 2004; Bailey et al. 2008)**.**

Rangeland improvement practices such as prescribed fire, fertilisation, woody plant control, and reseeding can be used to alter species diversity and abundance and forage quality and quantity to help distribute livestock within a pasture and assist in achieving more uniform and sustained use of the site (Fuhlendorf and Engle, 2004; Briske et al., 2011). Exclusion of use by livestock can assist in restoration of an impacted site if ecological thresholds have not been crossed and favourable climatic conditions occur. However, if ecological thresholds have been crossed then restoration to the historic plant community may not be economical or ecologically feasible (D'Odorico et al., 2013).

Monitoring of utilisation of the forage resource is essential to ensure long-term sustainability. Various qualitative and quantitative monitoring systems have been developed and can be implemented to track utilisation. These monitoring systems provide critical information to managers and can assist them in determining when and where livestock should graze. Monitoring systems can be used to indicate if processes related to desertification have been initiated (for example, an increase in bare patch size and connectivity or an increase in woody plants). It is critical that monitoring protocols be linked to drought contingency planning and timely management actions regarding stocking rates to avoid desertification (Stafford et al., 1992; Torell et al., 2010). Thus, allowing managers to intervene and prevent the site from crossing an ecological tipping point where desertification may not be reversed (D'Odorico et al., 2013). Monitoring systems can include a variety of tools, such as satellites, smartphone applications, utilisation cages, and photographic and observational transects. Skills to implement various monitoring systems vary and need to be adjusted for the area of concern and skill level of the land manager.

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 to 200 years (O'Connor et al., 2014; Archer et al., 2017). Woody encroachment has occurred worldwide with clearly documented trends in southern Africa (Joubert et al., 2008; O'Connor et al., 2014; Dougill et al., 2016), North America (Archer et al., 2017; Barger et al., 2011) and Australia (Eldridge and Soliveres, 2014). It is often viewed as a form of ecosystem degradation and desertification as it can lead to a net loss in the provision of ecosystem services from an area (O'Connor et al., 2014; Dougill et al., 2016). However, it should be noted that this is not universal. As noted by Eldridge et al. (2011b) and Eldridge and Soliveres (2014), there may be circumstances where bush encroachment may lead to a net increase in ecosystem services, especially at intermediate levels of encroachment. In certain areas, an intermediate level of woody cover may be desired where the ability of the landscape to produce fodder for livestock may increase, while some production of wood and associated products may be retained. This may be particularly important in regions such as southern Africa where 95% of rural households depend on wood fuel from surrounding landscapes (Shackleton and Shackleton, 2004).

Where bush encroachment has been assessed as a form of degradation, there are often substantial efforts to clear encroaching woody species. Early implementation of bush clearing efforts can be found in Namibia where a national program has 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). In addition to the clearance of woody cover, 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 are not addressed, it is likely that the landscape will be invaded once again.

The long-term success of woody biomass clearance and improved fire and grazing management remains to be evaluated, especially restoration back towards an 'original state' with an associated full suite of ecosystem services and biodiversity. In particular regions, for example, northern Namibia, the rapid reestablishment of woody seedling following clearing have raised questions about whether full clearance and restoration is possible (Smit et al., 2015). Rather should a new form of "emerging ecosystem' (Milton, 2003) be explored that includes both improved livestock and fire management as well as the utilisation of biomass as a new 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

In situ rainwater harvesting (RWH) provides a means of increasing the amount of water available for agriculture and livelihoods. It is often highlighted as a practical response to climate change as it forms a partial buffer against rainfall variability, which is already relatively high within drylands and predicted to become more acute over time (Vohland and Barry, 2009; Dile et al., 2013). 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 (Vohland and Barry, 2009; Dile et al., 2013; Yosef and Asmamaw, 2015).

One of the primary reasons for implementing RWH is to improve agricultural output and resilience. There is good agreement that the implementation RWH systems lead to an increase in agricultural production in drylands (see reviews by Biazin et al., 2012; Dile et al., 2013; Bouma et al., 2016). 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 in 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 330mm 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 (Vohland and Barry, 2009; Singh et al., 2012; 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 drivers. 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 (Kajisa et al., 2007; Bahta and Lombard, 2017). 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 effected 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 landscapes within the dryland domain requires further investigation. This includes considering its broader role in assisting human kind to adapt to climate change. Although its ability to improve the resilience of dryland agriculture is relative well known, a broader understanding of its impact on landscape resilience and disaster risk reduction is required.

3.7.1.5 Incentivising Sustainable Land Management

The adoption of SLM practices depends on their profitability and good fit with local resource availabilities. It was shown that, globally, every dollar invested into restoring degraded lands yields returns in the range of 2-5 USD over a 6-year period (Nkonya et al., 2016). Similar ranges 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., 2016a), Niger (Moussa et al. 2016) and Senegal (Sow et al., 2016). Despite these relatively high returns, the adoption of SLM practices remains low (Cordingley 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 represents social benefits, namely in the form of non-provisioning ecosystem services (Nkonya et al., 2016), which individual land users cannot internalise. Measures to expand schemes for payments for ecosystem services will contribute to higher adoption of land restoration activities (see section 3.7.3). Moreover, market failures in the form of lack of access to credit, input and output markets, insecure land tenure (section 3.2.3) result in the lack of adoption of SLM technologies which are profitable even at the private level (Moussa et al., 2016), requiring enabling policy frameworks to overcome these market failures (section 3.7.3).

3.7.2. Socio-economic Responses

Technological responses (section 3.7.1) represent only one facet of responses to climate change-desertification interactions. Socio-economic responses undertaken by dryland households and communities complement technological responses in two ways. Firstly, they serve to enable the adoption of SLM practices. Secondly, they help dryland agricultural households diversify to non-agricultural activities, reducing their dependence on land for incomes. These two response mechanisms are not isolated, but are in continuously evolving interactions: successes or failures of SLM adoptions influence household decisions for livelihood diversification. Similarly, diversification of livelihood sources affects socio-economic responses on land use and management.

3.7.2.1. Socio-economic Responses Towards Combatting Desertification under Climate Change

There is rigorous evidence that planting different crops and crop varieties, changing planting dates, modifying input use (especially irrigation), and changes in the portfolio of agricultural activities serve as major mechanisms that agricultural households use to adapt to increasing weather variability and climate change (Anwar et al., 2013; Bryan et al., 2009; Bryan et al., 2013; Burke and Emerick, 2016; IPCC, 2014; Waha et al., 2013). Degradation of drylands limits choices for such climate change adaptation options in agriculture. On the other hand, adoption of soil conservation measures, reforestation and afforestation, water harvesting, grazing and fire management (section 3.7.1) contribute both to land degradation-neutrality and climate change adaptation and mitigation (Campbell et al., 2014; Cowie et al., 2018; Lipper et al., 2014; Mbow et al.,2014). There are factors, such as lack of access to markets and land tenure security, which hinder or facilitate SLM adoptions, but are largely beyond the control of individual households and communities and require enabling policy interventions (section 3.6.3), however, traditional knowledge, collective action and own innovativeness are the major sources of resourcefulness through which agricultural households in drylands can respond to the coupled challenges of climate change and desertification.

Use of traditional knowledge for SLM contributes to enhancing resilience against climate change and combatting desertification (Altieri and Nicholls, 2017; Engdawork and Hans-Rudolf, 2016). There are abundant examples of how traditional knowledge has allowed livelihood systems in drylands to be maintained despite environmental constraints. Numerous traditional water harvesting techniques exist across the drylands, such as those based on digging of planting pits ("zai", "ngoro"), contouring of hill slopes, terracing and micro-basins (Biazin et al., 2012). For example, 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). In contrast to the narrative of "the tragedy of commons", pastoralist communities in drylands came up with a considerable number of approaches for sustainable rangeland governance. Pastoralist communities in Morocco developed "agdal" system of alternating seasonal use of rangelands to limit overgrazing (Dominguez, 2014). 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). Although well adapted to resource-sparse dryland environments, traditional practices are currently not able to cope with increased demand pressures and environmental changes (Enfors and Gordon, 2008; Engdawork and Hans-Rudolf, 2016). Moreover, an increasing number of studies document losses of traditional knowledge (Fernández-Giménez and Fillat Estaque, 2012; Hussein, 2011; Kodirekkala, 2017; Moreno-Calles et al., 2012), or its marginalisation (Dominguez, 2014). In this context, innovative combinations of traditional knowledge and modern agronomic practices contribute to overcoming combined challenges 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 pool resources and climate change adaptation (Adger, 2003; Engdawork and Bork, 2014; Eriksen and Lind, 2009; Ostrom, 2009; Rodima-Taylor et al., 2012). Social capital is divided into structural and cognitive forms, structural corresponding to strong involvement in various 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 adoption (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 result in improved soil and water management when they were not strictly enforced (Teshome et al. 2016). High enforcement costs are especially detrimental for collective action in communities with higher shares of conditional co-operators (Rustagi et al., 2010), i.e. those who adjust their behaviour depending on how other members of the community are acting. 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 points out that different levels and types of social capital result in different levels of collective action. 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 in a sample of countries from Eastern, Western and Southern Africa (van Rijn et al. 2012). The latter is indirectly corroborated also by the impact of community-based rangeland management organisations in Mongolia, with strong links to outside networks, who were more effective applying innovative rangeland management practices than in communities without joint organisations for rangeland management, although the levels of cognitive social capital did not differ between them (Ulambayar et al., 2017).

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). Historical innovations by earlier generations of dryland farmers and pastoralists has resulted in what is now called as traditional knowledge. 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 farmer innovativeness can be given a new dynamism by combining it with new information and communication technologies, development and dissemination of low cost smart-phone connected soil and water monitoring sensors and corresponding mobile phone applications to bring together farmer knowledge and modern science within citizen science initiatives (Cornell et al., 2013; Herrick et al., 2017; McKinley et al., 2017; Steger et al., 2017). SLM technologies co-generated through direct participation of agricultural households are more likely to be accepted by them (Bonney et al., 2016; Le et al., 2016; Vente et al., 2016).

Despite the ingenuity, innovation and collective action by dryland agricultural populations, their adoption of SLM practices remains low (Banadda, 2010; Cordingley et al., 2015; Lokonon and Mbaye, 2018; Mulinge et al., 2016; Nkonya et al., 2016; Wildemeersch et al., 2015), due to the highlighted constraints to the use of traditional knowledge and collective action. 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 (Safriel and Adeel 2008b; Boserup, 1965).

3.7.2.2. Socio-economic Responses towards Economic Diversification

Livelihood diversification through non-farm employment increases the resilience of rural agricultural households against desertification and extreme weather events, such as frequent droughts (see Glossary), by smoothing their income and consumption, by providing the funds to re-invest into agricultural adaptations to changing climate and adopting SLM practices (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 the smallholder pastoral households who are vulnerable to poverty traps, because their small herd sizes make them less resilient against droughts (Lybbert et al., 2004). Access to non-farm jobs, however, is limited in rural areas of the developing countries, especially for women and marginalised groups with lack of education and social networks (Reardon et al., 2008).

Migration is a form of livelihood diversification and a potential response option to desertification and increasing risk context for agricultural livelihoods under changing climate (Walther et al., 2002). However, there is a limited evidence if households actually respond specifically to desertification and climate change through migration. 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). Earlier, 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. However, migration involves some initial investments. For this reason, reductions in agricultural incomes due to climate change or desertification have the potential to reduce outmigration among the poorest agricultural households who become less able to afford migration (Cattaneo and Peri, 2016), thus increasing inequalities. On the other hand, there is a *high agreement* that households with migrant worker members are more resilient against extreme weather events and environmental degradation as compared to non-migrant households 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, massive 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

(*Towards the next stage, policy responses discussed here will be structured using the SDGs framework in coordination with other chapters. The purpose of such structuring will be to assess the impacts of the discussed policy responses not only on desertification (with specific focus on land degradation neutrality target)-climate change, but also their synergies and trade-offs for achieving other SDGs*)

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. As in the case with socio-economic responses, policy responses towards desertification can be classified in two ways: those which seek to combat desertification through avoiding, addressing and reversing it; and those which seek to provide alternative livelihood sources through economic diversification. These options are mutually complementary.

3.7.3.1. Policy Responses towards Combatting 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, instituting mechanisms for payments for ecosystem services, decentralised natural resource management, investing into research and development, and investing into renewable energy sources.

Policies aiming at improving market access, i.e. 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 market access was suggested to be a major determinant for the adoption of sustainable land management practices in a wide number of settings across the drylands (Nkonya and Anderson, 2015; Sow et al., 2016; Aw-Hassan, et al. 2016; Mythili and Goedecke, 2016; Kirui, 2016). In contrast to these findings, earlier Scherr and Hazell (1994), and more recently, Mirzabaev et al. (2016) found that high market access was associated with higher opportunity cost of labour, especially in lower-income locations, making households less likely to adopt labour-intensive SLM practices. This is because high market access areas are those closer to urban centres. Higher urban wages also raise the wages in neighbouring rural areas. It is likely that both of these mechanisms play out simultaneously, with the net effect depending on the interaction of market access with other location characteristics. Overall, the effect of higher market access on land management is only a part of its potential impacts. Others include improved access of rural populations to urban public services, such as health care and education, and thus has a positive effect. Government policies aimed at improving market access usually involve constructing and upgrading rural-urban transportation infrastructures.

Gender Empowerment. In dryland areas women usually have lower access than men to resources which limits their resilience and adaptive capacities to sustainably manage their land. Environmental issues such as desertification and impacts of climate change have been increasingly investigated through a gender equity lens (Bose, 2015; Broeckhoven and Cliquet, 2015; Kaijser and Kronsell, 2014; Kiptot et al., 2014; Villamor et al., 2014). 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 generally recognised that women who are already poor and marginalised are most likely to be impacted disproportionally (Arora-Jonsson, 2011; IFAD, 2006). Gender issues related to SLM are only marginally addressed in conservation efforts and government policies. A greater emphasis on understanding gender-specific differences over land use and land management practices contributes to the success of land restoration projects and initiatives (Catacutan and Villamor, 2016; Broeckhoven and Cliquet, 2015). For example, Fisher and Carr (2015) show that improving access to agricultural land, information and to credit is necessary for improving the adoption of drought-tolerant maize varieties by female-headed households in Uganda.

Moreover, the studies of the gender dimension of land degradation and climate change impacts are still very scarce particularly at the macro or landscape level assessment because of two main reasons. First, the impacts of climate change to men and women and their responses to desertification are context specific. For example, in the drylands of Sub-Saharan region, the common significant predicted consequences of climate change are the overall decrease in available water. Women of this region who are primary natural resource managers and providers of food security 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 countries for better opportunities, leaving women with more responsibility. Yet, women are often routinely absent from local decision making processes on how to address impacts of climate change. In contrast, a study of Southeast Asia countries showed that women, due to their increasingly productive roles, could become agents of either land degradation or restoration (Catacutan and Villamor, 2016). Additionally, these socially constructed gender-specific roles and responsibilities are not static as they are shaped by other factors such as wealth, age, ethnicity, and formal education (Kaijser and Kronsell, 2014; Villamor et al., 2014). Hence, women 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 (Catacutan and Villamor, 2016). Nonetheless, women particularly in the rural areas, remain heavily involved in securing food for their households and their role in addressing desertification is crucial. One aspect where the gender dimension should be considered is the restoration efforts of degraded lands.

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). Similarly, 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. 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 traditional knowledge and farmer innovations. Participatory research indicatives conducted jointly with farmers are likely to result in higher rates of technology adoption.

Strengthening land tenure security. The preponderance of evidence suggests that strengthening land tenure is a major factor contributing to the adoption of soil conservation measures in croplands (Sow et al., 2016; Aw-Hassan et al., 2016; Mythili and Goedecke, 2016; Kirui, 2016). Moreover, land tenure security led to investment in trees that enhanced households' land transfer capacities in Ethiopia (Deininger and Jin, 2006). On the other hand, although more secure land tenure led to more investments into SLM practices in Ghana, it did not affect farm productivity (Abdulai et al., 2011). In contrast, privatisation of rangeland tenures in Botswana and Kenya lead to the loss of communal grazing lands and actually increased rangeland degradation (Basupi et al., 2017; Kihiu, 2016a) as pastoralists needed to graze livestock on now smaller communal pastures. Holden and Ghebru (2016) indicated that since food insecurity in drylands is driven mainly by climate risks, institutions need to respond with flexible tenure, allowing mobility for pastoralist communities, and not fragmenting their areas of movement. More research is needed on the optimal tenure mix, including low-cost land certification, redistribution reforms, market-assisted reforms and gender focused reforms.

Payments for ecosystem services (PES) provide incentives for land restoration and rehabilitation activities (Schiappacasse et al., 2012; Lambin et al., 2014; Reed et al., 2015). Several studies showed that the costs of land degradation, including in drylands, in terms of non-provisioning ecosystem services are larger than in terms of marketable provisioning services such as food products (Costanza et al., 2014; Nkonya et al., 2016c). For this reason, SLM generates positive externalities, which are public goods. Individual land users cannot internalise all the benefits from investments into SLM. In such settings, there will be under-investment into SLM. Payments for ecosystem services can transfer some of these benefits to land users, stimulating more investments into SLM. One of such community-operated PES schemes was established in the Tarangire National Park in Tanzania involving the limitation of agricultural cultivation near the Park (Nelson et al., 2010). The South African PES scheme on Working for Water proposed an innovative way of clearing invasive plants to restore land ecosystem services through tendering these tasks to the unemployed applicants, thus directly contributing to their livelihoods (Turpie et al., 2008). PES schemes did not work 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). Reviewing the performance of PES programs in tropics, Calvet-Mir et al. (2015), on the other hand, found that they are in the majority environmentally effective, despite being unfair. The evaluation of the effectiveness, equity and poverty-reducing effects of PES schemes will benefit from quantitatively more rigorous and comparable approaches. The implementation of PES schemes will be improved if they are implemented through decentralised approaches giving local communities a bigger role in the decision making process (He and Lang, 2015).

Decentralisation of Natural Resource Management. Local institutions often play a vital role in modulating the outcomes of SLM initiatives (Gibson et al., 2005). Pastoralists involved in community-based natural resource management in Mongolia had higher capacity for successfully adapting to extreme winter frosts with less damage to their livestock herds (Fernandez-Gimenez et al., 2015). Decreasing power and role of traditional community institutions resulted in lower success rates of community-based programs for rangeland management in Dirre, Ethiopia (Abdu and Robinson, 2017). Ostrom and Nagendra (2006) showed that decentralised governance leads to improved management of forests. Similarly, Dressler et al. (2010) found that the shift from centralised control of forests back to communities improved degraded forests. However, when local elites were placed in control, it resulted in negative impacts on the livelihoods of forest product-dependent poor (Dressler et al., 2010).

Investing in Research and Development. Desertification has received substantial research attention over recent decades (Seely and Wöhl, 2004). Agricultural research on SLM practices helped to generate a significant number of new innovations and technologies that help to increase crop yields without degrading the land (see section 3.7.1). Such technologies can help improve the food security of smallholder dryland farming households, but at the same time, may not be sufficient by themselves to lift smallholders out of poverty (Harris and Orr, 2014). Strengthening research on desertification is of paramount importance not only to meet SDGs but also effectively monitor, manage and develop ecosystems based on solid scientific knowledge. If research is going to contribute significantly to abating the impact of desertification, substantial investment is required to improve science-policy-public dialogue and communication (Torres et al., 2015). Comprehensive research is required to develop a clear understanding, supported by quantified data and evidence, on the current economic development trends including agricultural commercialisation (e.g. expansion of irrigated commercial agriculture), industrialisation and urbanisation and their role in exacerbating desertification phenomenon and to come up with recommendations that would help to change the tide in favour of sustainable development. More research is needed on drought and climate change mitigation and minimising the impacts of desertification based on traditional knowledge and appropriate to local circumstances. Research needs to work more 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). Policy research and discourse analysis need to be strongly supported to understand how desertification phenomenon is perceived, how cause and effect relationships are understood by governments and different sectors of the society and how solutions can be formulated (Aster, 2004). As the drylands strive to grow economically, so will the demand for energy, and research should come up with solutions that promote energy conservation and energy efficiency (Zeng et al., 2008), clean development mechanisms (Zomer et al., 2008) and rational utilisation of natural resources. One critical constraint common to all drylands is institutional capacity limitation. There is a lack of professionals in dryland, desertification and conservation research. There is so much to be desired in terms of putting in place adequate numbers of research institutes and getting existing ones up to standard in terms of equipment, facilities and staffing (Pimm et al., 2001). Significant effort is required to build database and information repositories and facilitate seamless linkages and networking between research, conservation and higher learning institutions to promote multidisciplinary approaches to the problem and synergy/complementarity of plans and programs. National action plans on desertification and land degradation give due emphasis to research, science and technology, however, in the developing world where accessing information and the capacity is lacking to convert information into actions, the plans often remain on paper, hardly implemented. A means should be sought to strengthen the implementation capacity of developing countries. Investing in research and development per se would come to no avail if it is not backed up by a robust knowledge management system. Improved knowledge management systems 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 highly fragmented at present, constraining effectiveness of those engaged in assessing and monitoring the phenomenon at various levels (Reed et al., 2011b). At national level, countries develop NAPs, however due to capacity limitations there is so much to be desired in terms of putting in place an effective knowledge exchange and monitoring system (Akhtar-Schuster et al., 2011). With regard to various global assessments there has been very little progress in realising institutionalised and long term knowledge management systems despite the repeated calls to strengthen cooperation and collaboration. There is no reference centre (a coordination centre where each agency retains its own data and information in an accepted format) for reports, complicating access to assessment and monitoring information and database which in turn leads to redundancies and task overlaps (Chasek et al., 2011). The contribution of such bodies would have been more laudable and focussed if they had been better coordinated.

Developing Renewable Energy Sources. Most developing countries continue to rely on traditional biomass for the majority of their energy sources. Even in rapidly emerging economies, such as China, the share of traditional energy in the total energy consumption remains very high, especially in rural areas (Zhou et al., 2008). In Sub-Saharan Africa, the dependence on traditional biomass is the highest in the world (Amugune et al., 2017). Use of biomass for energy, mostly fuelwood, crop straws and livestock manure, often leads to deforestation and desertification in drylands. Moreover, usually women and girls spend considerable amounts of time collecting fuelwood. 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. Stambouli et al. (2012) indicates that Algeria, despite being rich in conventional fossil fuel resources, is also expanding its renewable energy production in order to reduce the negative externalities of fossil fuel use on the environment. However, many countries (e.g. Nigeria and Iran) face important social, political and technical barriers to expanding their renewable energy production. High cost of fossil fuels (compared to rural incomes) resulted in increased deforestation and other biomass consumption in Iran (Afsharzade et al., 2016). However, improved access to alternative technologies would require building-up of technical capacities and skills. 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 aiming at combatting desertification, population pressures and growing food demands, as well as the need to reduce poverty and strengthen food security, is likely to put strong pressures on the land. Sustainable development of drylands under their current socio-economic and environmental (climate change-desertification) conditions will also depend on the ability of governments to promote policies for economic diversification both with agriculture and away from agriculture.

In this regard, Barrett et al. (2017) asserts that Africa is well on its way in the economic transformation process, even though such a statement might be considered a gross generalisation by others. African countries are taking measures to promote commercialisation in the agriculture sector, including the implementation of integrated agro-industrial parks (e.g. Ethiopia), specialised economic zones (e.g. Nigeria) and agro-poles (e.g. Senegal) to strengthen the agro-processing sector and spur rural industrialisation. This is a step in the right direction considering the declining revenue many African countries were experiencing following the fall of primary commodity prices.

In parallel, efforts are underway to improve labour productivity and boost production and income revenue from agriculture and livestock sectors, the major driving factor being profitability. Case in point, the scaling up of mechanised irrigation using motorised pumps, which has enabled year-round cropping on Oasis farms in Maghreb countries of North Africa (De Haas, 2001), and a shift from production of staples to high value commodities (e.g. cash crops) in the Sahel (Cour, 2001). The relatively fast growth registered in agriculture is paving the way for agribusiness development (Tschirley et al., 2015). 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 as experienced in many Asian countries; even though such a transition may not have been reflected in terms of positive changes on farmers livelihoods in all cases, for example, China (Reardon et al., 2009). However, it is reasonable to assume that there are many countries that are showing impressive gains, but also some that are struggling (Collier, 2008). 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 ushers in social and economic progress with labour increasingly shifting out of agriculture into more productive sectors (Cour, 2001). Although it has yet to occur more broadly across the continent, some sub-Saharan countries, for example, Rwanda and Ethiopia, are reported to have registered initial success in terms of concurrently developing the agriculture and industry sectors.

Structural transformation in rural economies presupposes the critical role of output and livelihood diversification following productivity growth and improved market systems, where the non-agriculture sector facilitates movement of household into making a living from modern service and industry sectors away from farming (Haggblade et al., 2010). Farmers are being obliged to supply an assortment of agricultural products responding to rapidly changing food habits in the growing urban centres (Hussein and Nelson, 1998). The pursuit of diversification has led to a drop in on-farm employment and to the quadrupling of GDP in West Africa (Cour, 2001). In terms of output diversification, livestock seems to provide ample opportunities within the system itself or in complement to it (Brockhaus et al. 2013). Farm households substantially improved income from farming to hedge against environmental shocks by earning additional revenue from engaging in livestock enterprises such as small ruminant (goat, sheep) production, employing intensive and advanced husbandry and marketing practices. This is so especially for women in the pastoral systems.

Structural changes are happening in the form of increased rural-urban interactions, fostering trade and investment through regional economic integration and integration into global value chains (Cour, 2001). Modernised farming, improved access to inputs, credit and technologies is enhancing competitiveness in local and international markets (Mortimore and Adams, 1999). Export oriented development is increasingly being accepted as a viable strategy for development, and promotion of exports is particularly expected to be beneficial to poor farmers due to the relatively high sales income that can be obtained (Leichenko and O'Brien, 2002).

One of the recognised drivers of successful economic transformation is believed to be urbanisation, through the shift of underutilised labour in rural into gainful employment in urban areas (Jedwab and Vollrath, 2015). The larger share of world population will be living in urban centres in the early decades of 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. Many argue that urbanisation has to be promoted as a viable strategy for getting people off the farm, as it is cities that are attracting an overwhelming number of rural people across the developing world because of opportunities including jobs (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.

3.8. Hotspots and Case Studies

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 virtually every geographic region representing one of the most important environmental problems in agricultural, rangeland, and forestry systems. Wind erosion, for instance, is the most important process in arid and semi-arid regions of the world, affecting 41% of global land area (Moharana et al., 2016). In the case of Chile, erosion rates reach up to 100 tons of soil per hectare per year, affecting 25% of the surface, having increased over the last 50 years (Ellies, 2000). Using the Universal Soil Loss Equation (USLE) in Spain, it has been estimated that soil erosion can be as high as 300 tons per hectare per year (equivalent to a net loss of 18 mm per year) (López-Bermúdez, 1990). In spite of the relevance and worldwide coverage, the estimates of soil erosion from different methods and sources show a great degree of variability 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} (FAO, 2015).

Soil erosion has direct, negative effects for global agriculture. One of the most important effects is the loss of soil fertility, which for the case of water erosion has been estimated to be equivalent to 23-42 MtN and 14.6-26.4 MtP annually (Pierzynski et al., 2017). Carbon storage capacity is also affected by soil erosion. Wind erosion results in a loss of fine soil particles (silt and clay), reducing the ability to sequester carbon in the soil organic carbon pool (Wiesmeier et al., 2015).

3.8.1.2. Observed Trends in Arid Lands

Because of its negative impacts on agriculture, soil erosion has been the focus of numerous research initiatives that monitor its development, understand the main factors and processes that determine its rate, and evaluate management and policy interventions to reduce its spread. Being a phenomenon that depends on several variables, there are no consistent observable trends in recent years in drylands (*robust evidence, little agreement*). However, according to (FAO, 2015), rates of soil erosion, although still high in extensive areas of cropland and rangeland, have been substantially reduced.

Based on the 2003 Annual Natural Resources Inventory data, average soil erosion rates on all US cropland have decreased more than 38% since 1982 due to better soil management practices (Kertis, 2003). National average annual erosion rate on non-Federal US rangeland is estimated to be 1.41 $tha^{-1}yr^{-1}$. Over 29.2 x 106 ha or 18% of the non-Federal rangelands might benefit from treatment to reduce soil loss to below 2.24 $tha^{-1}yr^{-1}$ (Weltz et al., 2014). Rainfall erosivity has shown a positive trend in dryland areas of China 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 intensity 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). The soil loss by erosion is much higher from bare lands compared to cropped lands. Using remote sensing and GIS techniques, Nabi et al. (2008) estimated the soil erosion rate from the barren lands in Soan river basin in the Potohar region of Pakistan to be 6341 $tonnes/km^2\ yr^{-1}$ and that from the cropped land 1876 $tkm^{-2}yr^{-1}$. In Pakistan, the highest rate of erosion was estimated to be 150-160 $tha^{-1}yr^{-1}$ (Anjum et al., 2010). In the semi-arid region 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 will further exacerbate soil erosion. 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. The worst example is Warsak dam in Pakistan, built in Khyber Pakhtunkhwa province on the Kabul river in 1960, which was completely choked with sediments 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 predicted by most of the climate change scenarios, affect the flood frequency distribution and can intensify erosion processes (*robust evidence; high agreement*) (Molnar, 2001; Ziadat and Taimeh, 2013; Nearing et al., 2015).

3.8.1.4. Successful Restoration and Rehabilitation Examples

3.8.1.4.1 *Soils 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 a great interannual rainfall variability. Rainfall has declined about 18% to 27% and the dry season has increased by two months in the last century. The national map of soil sensitivity to erosion provides a stark picture of the amount of area at risk of soil erosion in the steppe and pre-Saharan areas of Algeria. More than three million hectares of land in the steppe provinces (Naama, El-bayadh, Djelfa, M'sila, Tebessa) are now experiencing intense wind activity and are particularly high-risk areas for soil erosion. Since independence, population has increased at a growth rate of 3.5%. Livestock in these regions, whose predominant component is sheep, have grown exponentially since 1968, leading to severe overgrazing, causing trampling and soil compaction, which greatly increased the risk of erosion (Nedjraoui and Bédrani, 2008). Wind erosion, very prevalent in this region, is due to climatic conditions and the strong anthropogenic action that reduces the vegetation cover. The action of the wind carries fine particles such as sands and clays and leaves on the soil surface a lag gavel pavement which is unproductive. Nearly 600,000 ha of land in the steppe zone are totally desertified without the possibility of biological recovery. Water erosion is largely due to torrential rains in the form of severe thunderstorms that disintegrate the bare soil surface from raindrop impact. The detached soil and nutrients are transported offsite via runoff resulting in loss of fertility and water holding capacity (Figure 3.10). The risk of and severity of water erosion is a function of human land use activities that increase soil loss through removal of vegetative cover.

(1) (2)

Figure 3.10 Effects of wind (1) and water erosion (2) on steppe soils in Algeria

Combating soil erosion has been one of the priority objectives of the state authorities since the beginning of the 1970s. Long-term monitoring (1974-2017) was implemented to document desertification, its causes, its consequences, and success of rehabilitation programs (Slimani et al., 2010; Hourizi et al., 2017). 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 and for the protection and restoration of rangelands.

3.8.1.4.2 No-Till Practices in Central Chile

Over the last decades there has been an increasing interest in the development of No-Till technologies as a way to minimise soil damage, reduce energy usage and increase soil organic matter accumulation. Motivated by a strong desire to work with nature and to prevent soil erosion, early pioneer Carlos Crovetto led a significant revolution in soil conservation, by adopting No-Till practices in the early 1970s. The Chequen Farm is located in south-central Chile. Main crops are wheat, corn, soybean, lupines, oats, canola, and sunflower. Forestry systems (*Pinus radiata*) were established in the extreme gullies (Reicosky and Crovetto, 2014). 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 (seven years) had doubled the biological activity indicators of traditional farming and even surpassed those found in pasture (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 (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 over the plateaus and plains between Konya and Ereğli districts of Turkey. It is characterised by a semi-arid steppe climate of the continental Central Anatolia Region and annual average precipitation of about 250-300 mm. Historically the area was a mixture of dryland cropping and sheep pasture. In areas where the vegetation was denuded by overgrazing the thin surface soil horizon was removed through erosion processes resulting in wind deflation and creation of a large continental drifting dune threatening the settlements around Karapınar. Consequently, the Karapınar town and nearby villages faced the danger of abandonment due to out-migration in early 1960s. The reasons for increasing wind erosion and deflation in the Karapınar district can be summarised as follows: the sandy material originated from an old lake bed and was mobilised following drying of the lake; the hot and semi-arid continental climate conditions along with the high precipitation seasonality and year-to-year variability; excess 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 on strong prevailing wind course. Reclamation work initiated to rehabilitate the area included a systematic approach whereby wind screens were placed on the dunes to reduce wind speed, the area between the rows of wind screens were planted with grass, after establishment of grass shrubs and trees were planted and protected from grazing and harvested for fuel. Dryland farming is practiced on the flat lands between the dunes by banding the rows of crops perpendicular to the prevailing wind to reduce the fetch. After controlling erosion, the economic efficiency of the region was enhanced and new tree nurseries have been established with modern irrigation.

3.8.2. Case Study on Salinisation due to Sea Water Intrusion

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

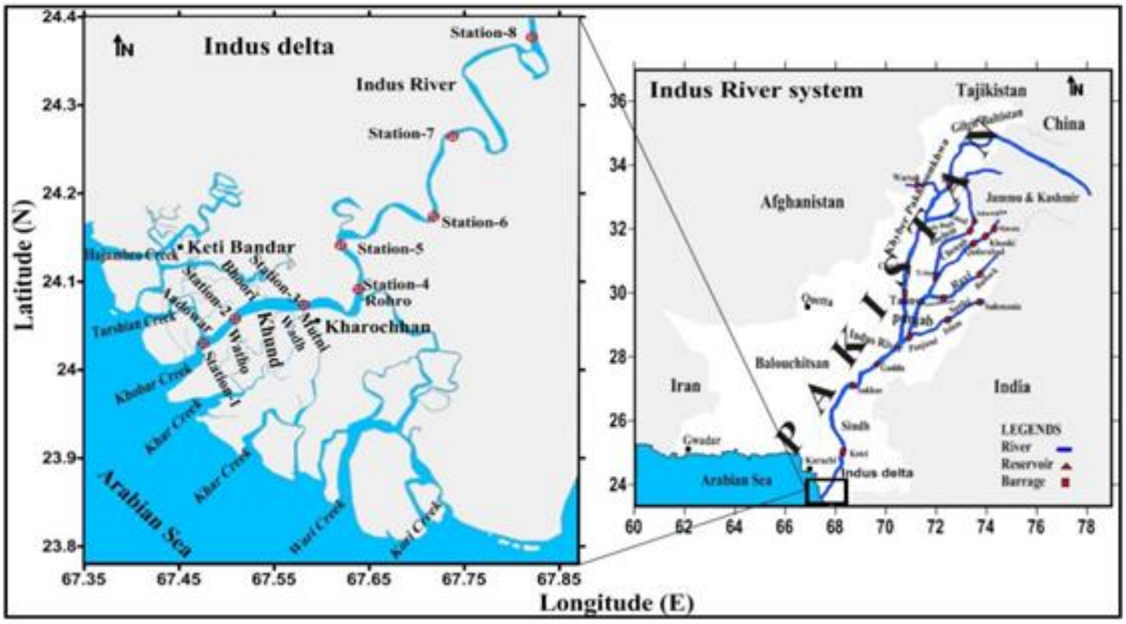

Figure 3.11 Indus delta with a network of major creeks and tributaries (Kalhoro et al. 2017)

The Indus delta, located in the south-eastern coast of Pakistan near Karachi in the North Arabian sea, is one of the six largest estuaries in the world spanning over an area of 600,000 ha (Figure 3.11). The Indus delta is a clear example of seawater intrusion and land degradation due to local as well as up country climatic and environmental conditions (Rasul et al., 2012).

Such degradation takes the form of high soil salinity, inundation and waterlogging, erosion and freshwater contamination. The inter-annual variability of precipitation with flooding conditions in some years and drought conditions in others has caused variable river flows and sediment runoffs below Kotri barrage (about 200km upstream of the Indus delta) (Figure 3.12). This has affected hydrological processes in the lower reaches of the river and the delta, contributing to the degradation (Rasul et al., 2012).

Figure 3.12 A view of the dry Indus river at Kotri barrage, about 200 km upstream of Indus delta (Kalhoro et al. 2017)

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

Lake Urmia in northwest Iran, the second largest saltwater lake in the world and the habitat for endemic Iranian brine shrimp, *Artemia urmiana*, has also been affected by sea water ingression. During a 17 year period between 1998 and 2014, human disruption and years of dam building have attracted the natural flow of sweet water as well as salty sea water from the surrounding area into Lake Urmia (Figure 3.13). Water quality has also been adversely affected with salinity fluctuating over time, but in recent years reaching a maximum of 340 g/L. This has rendered the underground water unfit for drinking and agricultural purposes and a risk to human health and livelihoods.

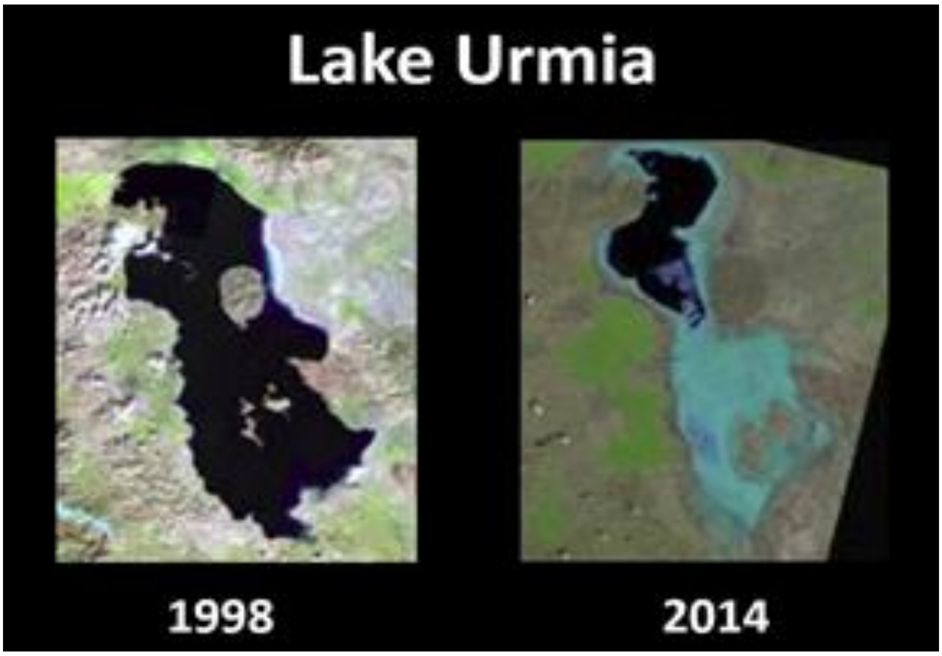

Figure 3.13 Urmia Lake condition in years 1998 and 2014

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

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

In other countries, intrusion of seawater is exacerbated by destruction of mangrove forests. Mangroves are important coastal ecosystems that provide spawning bed for fish, timber for building, livelihood to dependent communities, act as barriers against coastal erosion, storm surges, tropical cyclones and tsunamis (Kalhoro et al., 2017) and are among the most carbon-rich stocks on Earth. They nevertheless face a variety of threats: climatic (storm surges, tidal activities, high temperatures) and human (coastal developments, pollution, deforestation, conversion to aquaculture, rice culture, oil palm plantation), leading to declines in their areas. In Pakistan, using remote sensing (RS), the mangrove forest cover in the Indus delta has decreased from 260,000 ha in 1980s to 160,000 ha in 1990 (Chandio et al., 2011). Based on remotely sensed data, a sharp decline in the mangrove area was found in the arid coastal region of Hormozgan province in southern Iran during 1972, 1987 and 1987 (Etemadi et al., 2016). Myanmar has the highest rate (about 1%) of mangrove deforestation in the world (Atwood et al., 2017), In general, more than one-third of mangroves have been destroyed worldwide (Hamilton and Casey, 2016). Regarding global loss of carbon stored in the mangrove soil due to deforestation, four countries exhibited high levels of loss: Indonesia (3,410 Gg CO_2 yr^{-1}), Malaysia (1,288 GgCO_2 yr^{-1}), US (206 Gg CO_2 yr^{-1}) and Brazil (186 GgCO_2 yr^{-1}). Only in Bangladesh and Guinea Bissau there was no decline in the mangrove area from 2000 to 2012 (Atwood et al., 2017).

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

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

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

3.8.3.1. 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 has caused desertification and dust storms to become a serious problem in the region (Xu et al., 2010). About 60 years ago (i.e., 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 Neutral" 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 was implemented in three stages (1978–2000, 2001–2020, and 2021–2050) following eight engineering schedules. 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 to climate change.

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, and the net decrease of degraded land area was 9,902 km^2, resulting in the "double reduction" since the reduction occurred in 2004; (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) A number of distinctive industrial bases were set up in various places for combating desertification. 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. This has become an important pillar for economic development in the sandy areas and a high priority for peasants as a method to eradicate poverty.

The protection and natural restoration are top priorities for implementation of policies to combat desertification across the country where the key ecological projects are continuing to be implemented, such as, Phase II Project of the sandstorms of Beijing and Tianjin, the projects of Returning Farmland to Forestry and Returning Grazing Land to Grasslands and the "Desertification Prevention Law" which is strictly enforced. Desert ecological compensation policies and subsidies policies for combating desertification have been actively evaluated. The result is that 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 surmounted, the popularisation and application of new technologies have been accelerated, and the trainings of technicians for farmers and herdsmen have been strengthened. Scientific and technological supports for combating desertification have been improved, and public awareness and scientific ways to combat desertification have been raised (e.g. protecting local vegetation in desertification-prone lands and planting suitable vegetation according to local conditions in desertified lands or natural restoration of vegetation, immigration based on ecological carrying capacity for decertified areas); developing the technique of water-saving irrigation, protecting the surface soil and groundwater, optimising the future land utilisation pattern, and changing the agricultural and animal husbandry structure to ensure sustainable development goal. 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). The provincial government responsibilities for desertification prevention and controlling objectives have been strictly implemented.

3.8.3.2. The Green Dam in Algeria

After independence, the Algerian government had the reconstruction of forests destroyed by the war among its top priorities (Belaaz, 2003). The high steppe plains were also seriously desertified by overgrazing. As a means to combat desertification the government invested in revegetation effort “the Green Dam” beginning in 1970 (“Barrage Vert”). This was the first significant experiment to combat desertification, influence the local climate and decrease the aridity by restoring the ecological balance through this 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 3.14). It is over 1,200 km long (from the Algerian-Moroccan border through to the Algerian-Tunisian border) and has an average width of about 20 km, between the Saharan Atlas and the steppe lands. The soils in this area are shallow, low in organic matter and susceptible to erosion. The vegetation is steppic, consisting of Alfa (*stipa tenacissima)* and at higher elevations of forests of pine (*pinus halepensis)* and green oak (*Quercus ilex*). The population in the area of the Green Dam is nomadic, semi-nomadic (agro-pastoralists) and sedentary. The main objectives of the project were to conserve natural resources, facilitate the living conditions of local populations 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, the choice of species (Aleppo pine was not adapted to all areas of the project) and vulnerability of Aleppo pine to insect damage (Processionary Moth) (Bensaid, 1995). The experience of previous years led to re-evaluation and improvements in the concept of the Green Dam with the launch of integrated management studies, the improvement of rangelands through tree and fodder shrub plantations, and the development of appropriate techniques for water conservation. Reforestation during this period was carried out using multiple species, such as *Cypressus, Acacia, Atriplex* and rustic fruit trees to increase and diversify the sources of income of the populations. These actions reduced the rate of monoculture of *Pinus halepensis* from 100% to 65%. The reforestation effort was determined to be successful (greater than 50%). The National Reforestation Plan was adopted in 1999 by the Algerian government with a 20-year lifespan. Its goal is to establish 1,245,900 ha of plantations, over a 20-year span of the plan and contribute to the mitigation of climate change effects.

Figure 3.14 Localisation of green Dam in Algeria (Saifi et al., 2015).

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; 75,000 ha have been planted for the protection of life centres by stabilising dunes; 42,000 ha for pastoral plantations; 21,000 ha for fruit plantations; nine nurseries on 95 ha of productive agricultural lands, which has resulted in the production of nearly 40 million forest seedlings; 1,500 units of water resource mobilisation; and one million m^3 of torrential correction. The 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. Thus, Algeria has presented to the Green Climate Fund an integrated project to restore land in arid and semi-arid areas of the Green Dam. The Green Dam as a pioneering effort to combat desertification and 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).

3.8.3.2. The Afforestation and Erosion Controlling Action and Green Belt in Turkey

Turkey is among the countries facing a high level of land degradation and erosion due to topographical structure, climate and improper agricultural practices including destruction of range and forest lands, and the sensitivity of most of the lands to erosion. Turkey is also considered among those that would be affected adversely by climate change and variability (IPCC, 2013). The impact of climate change and climate variability in the Mediterranean region has consequences on productivity of forest and natural ecosystems, fire risks, and alters the effects of human-induced disturbances.

In order to reduce greenhouse gas emissions and increase carbon sinks (IPCC, 2013), a cooperative project "The National Afforestation and Erosion Control Mobilization Action Plan (NAECMAP)" was implemented by Turkey's Ministry of Environment and Forestry (now the Ministry of Forestry and Water Affairs). Public institutions and organisations, municipalities, non-governmental organisations and the community assigned by National Afforestation and Erosion Control Mobilization Law no. 4122 were organised to combat desertification. NAECMAP (2008 to 2012) prescribed the need for coordinated work among public bodies and institutions as well as all the parties of the community NAECMAP detailed the undertaking of afforestation, rehabilitation and erosion control and rangeland rehabilitation works on an area of 2.3 million ha. The MOEF aimed to accomplish the work on 2.16 million ha of this area and other bodies and institutions on 136,000 ha. The total cost of these works was estimated at more than 2.7 billion Turkish Liras.

Figure 3.15 Each year of the mobilisation, 300,000 people were employed for seed and seedling production, afforestation, rehabilitation and erosion control work in 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 (total 21.7 million ha) through this program. During the five years in the scope of the NAECMAP, Turkey achieved afforestation of 210,169 ha; soil protection afforestation in 315,889 ha; and private afforestation in 49,385 ha (Figure 3.15). Some 1.75 million ha of degraded forest were rehabilitated, and rangeland rehabilitation was achieved in 37,880 ha. In addition to afforestation, rehabilitation, erosion control and rangeland rehabilitation work revegetation was done on 8,135 km of highways and 2,262 km of village road, in 27,000 school yards, 1,095 health centres and hospital orchards 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 3.16). As a result of migration from rural to urban areas, the population's need for recreational space in cities has increased. To address this need, green belt afforestation around urban areas was undertaken and urban forests were established. Over the five years of the NAECMAP, 109 million seedlings were distributed to the population free of charge. NAECMAP also provided employment opportunities to the rural population. Every year during the plan's implementation, 300,000 people were employed for six months in the production of seeds and seedlings, afforestation, rehabilitation and erosion control works.

Figure 3.16 Green belt afforestation was important to address a growing need for recreational space in urban areas of Turkey

3.8.3.3. The Great Green Wall of the Sahara and the Sahel Initiative

The Great Green Wall is an initiative of the Heads of State and Government of the Sahelo-Saharan countries who decided to provide a relevant response from Africa in order to contribute to the mitigation and adaptation to climate change, and to improve food security of the Sahel and Saharan peoples. Launched in 2007, this continental project aims to restore Africa's degraded arid landscapes and support the efforts of local communities in the management and sustainable use of forests and rangelands and promote the fight against biodiversity degradation in the poorest regions of the Sahel. The Great Green Wall involves plantations and proximity projects covering 7,775 km from Senegal on the Atlantic coast to Eritrea on the Red Sea coast (Figure 3.17), with a width of 15 km. The wall will pass through the cooperating countries including Eritrea, Ethiopia, Sudan, Chad, Niger, Nigeria, Mali, Burkina Faso and Mauritania. In North Africa, Algeria and Tunisia are stakeholders in this initiative.

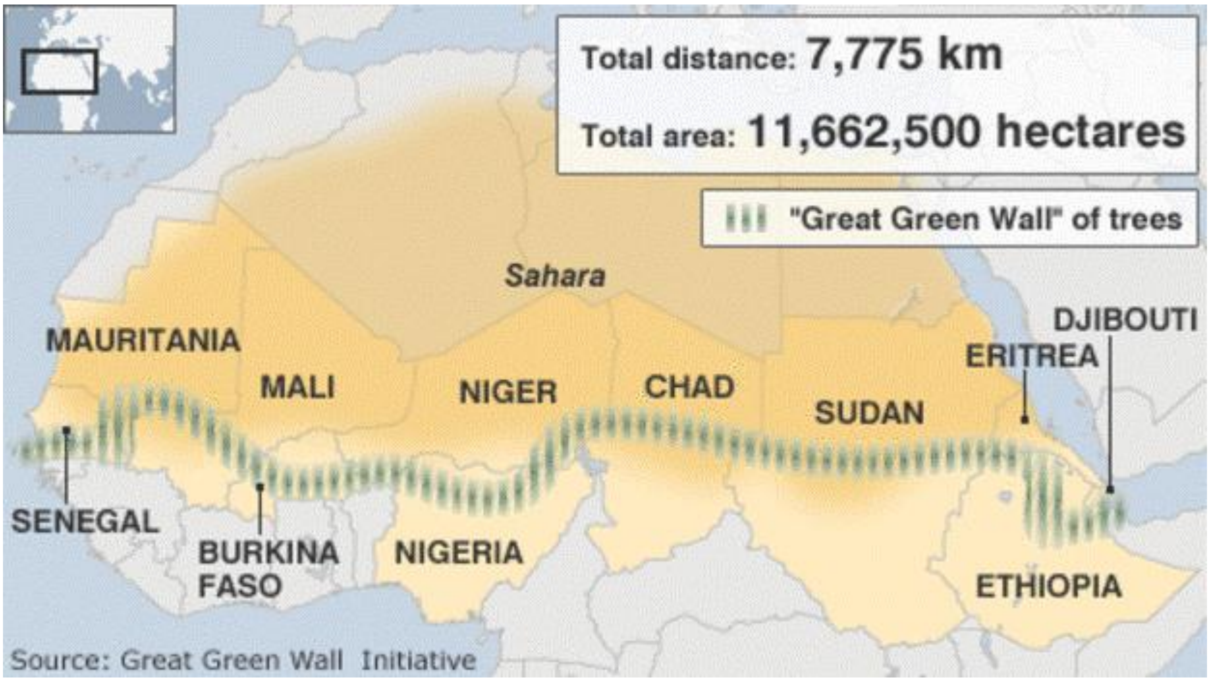

Figure 3.17 The Great Green Wall

The choice of woody and herbaceous species that will be used to recolonise degraded ecosystems will be based on biophysical, ecological and socio-economic criteria. The criteria will be: species with socio-economic value (food, pastoral, commercial, energetic, medicinal, cultural etc.); species of ecological importance (improvement of the environment, sequestration of carbon, soil protection and improvement, and infiltration of rainwater) and species that are resilient to climatic change and variability. Pan African Agency of the Great Green Wall (PAGGW) is an international agency 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 with the mission to fully realise the Great Green Wall. Although officially launched more than 10 years ago, the activities in several member countries have not started at required scales. A monitoring/evaluation system has been defined, allowing all Nations to measure the impact of the efforts made and to propose the necessary adjustments.

3.8.4. Case Study on Invasive Plant Species

3.8.4.1. Introduction

Invasion of plant species is an important consequence of climate change (Bradley et al., 2010; Davis et al., 2000). It is considered a major risk to native biodiversity and can disturb the nutrient dynamics and water balances in affected ecosystems (Ehrenfeld, 2003). Compared to more mesic regions, the number of species that succeed in invading drylands is low. Yet, drylands can be heavily impacted by invasive species in terms of their effect on biodiversity as well as the socio-economic wellbeing of resident communities. Moreover, increasing human populations in dryland areas are responsible for creating new invasion opportunities.

Climate change is expected to further affect the introduction, establishment and spread of alien species in drylands. For instance, Davis et al. (2000) proposed that sites with high rainfall variability promote 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). Although the effects of climate change on invasive species distributions have been explored over the last decade, the associated consequences from communities to ecosystems remain under studied. As a result, the effect of invasive plant species on ecosystem functioning and land degradation is not fully understood (Bradley et al., 2010; Eldridge et al., 2011b). For example, Eldridge et al. (2011b) reported that the invasion of shrubs species into grasslands, increased soil C and N contents in 33 sites, but other studies exhibit unchanged results (20 sites).

Due to the time lag between the first release of invasive species and their impacts, the consequences of invasions are not immediately detected and may only be noticed centuries after introduction. Climate change and invaders such as exotic grasses may act in concert (Ravi et al., 2009). For example, the invasion of dryland ecosystems often changes fuel loads, which can lead to an increase in the frequency and intensity of fire. In areas where the climate is becoming warmer, an increase in the probability 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 already known invasive species (n=99 from Bellard et al. (2013)) predicted to find suitable climate conditions in dryland areas is anticipated to decrease slightly by 2050. The average number of invasive alien species (IUCN list of the 100 worst invasive alien species worldwide, Lowe et al. (2000)) is predicted to reach 5.1 species by 2050 (under an A1B emission scenario) compared to 5.3 species under the current scenario. At a regional scale, Bellard et al. (2013) predicted increasing risk in Africa and Asia, with declining risk in Australia (Figure 3.18). This projection does not represent an exhaustive list of invasive alien species occurring in drylands. Many other invasive alien species occur in these regions and with climate change, new species that are not invasive as yet, are likely to emerge.

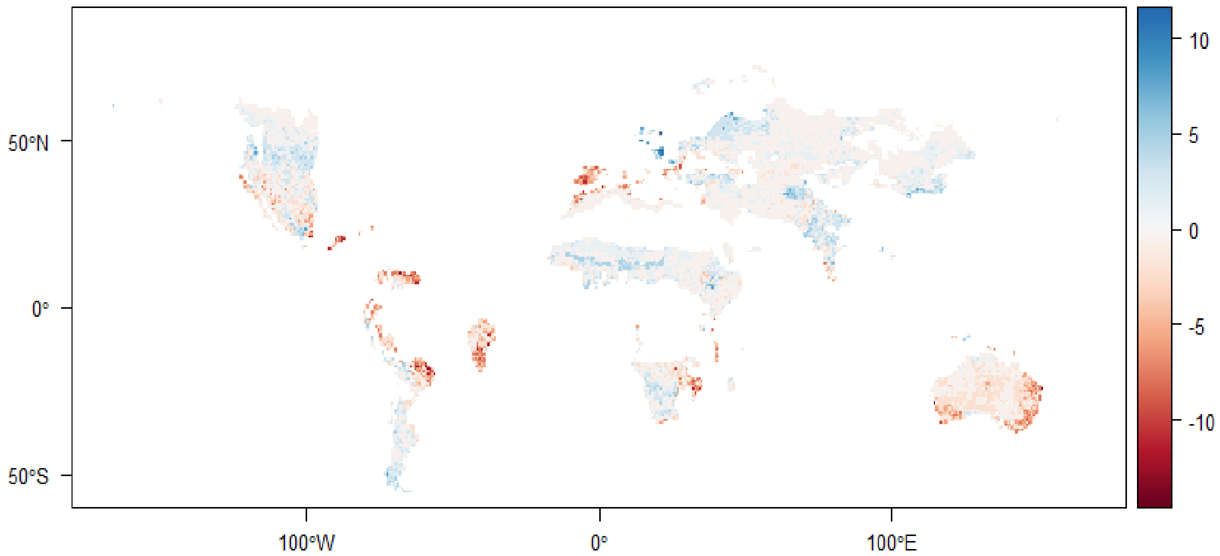

Figure 3.18 Difference between the number of invasive alien species (n=99, from (Bellard et al. 2013)**) predicted to occur by 2050 (under A1B scenario) and current period "2000"**

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

3.8.4.2. Description of the Problem

Ethiopia. A large number of invasive plant species occur in the country. However, the two 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.

Mexico. The 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 one million ha in Sonora State (Castellanos 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 notorious invasive plant in the United States. 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 within five years following the introduction of cheatgrass (Pilliod et al., 2017; Balch et al., 2013).

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% to 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 savanna cleared about 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). This invasive plant/wildfire cycle has severe negative effects on both urban and rural agricultural communities, which in turn contributes to increasing conflicts over land and water.

3.8.4.4. 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 the invader, they are clearly inadequate (G/Selase and Getu, 2009). 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.

United States. Attempts to reduce cheatgrass impacts through reseeding have occurred for more than 60 years (Hull and Stewart, 1948) 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 200mm yr^{-1}) (Davies et al., 2012; Taylor et al., 2014). Current rehabilitation efforts emphasise the use of native and non-native perennial grasses, forbs, and shrubs (Bureau of Land Management, 2005). Recent literature suggests that these treatments are not consistently effective at displacing cheatgrass populations or re-establishing sage-grouse habitat with success varying with elevation and precipitation (Arkle et al., 2014; Knutson et al., 2014). Proper post-fire grazing rest, season-of-use, stocking rates, and subsequent management are essential to restore resilient sagebrush ecosystems before they cross a threshold and become an annual grassland (Chambers et al., 2014; Miller et al., 2011; Pellant et al., 2004). Projections of increasing temperature and (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.

3.8.5. Cross Chapter Case Study on Policy Responses to Drought

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 (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., 2014; 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, while 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). Coupled General Circulation models are still not able to produce an accurate representation East African climate; topography and regionality of seasons, which require high resolution regional-level simulations.

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 to 8 billion USD were incurred in the USA (FEMA, 1995). Somewhat more recent 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. (2014) 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, it 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, 2006), and about 600,000 people were in need of food aid to the value of 19 million US(World Bank, 2006).

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 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. For example, 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 policy approaches for responding to droughts. Firstly, responding to droughts when they occur by providing drought relief to affected communities and households, which is known as crisis management. This has been the most frequently used default policy approach for dealing with droughts. 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 the ex-ante preparation of drought preparedness plans which coordinate the policies for providing relief measures when droughts occur. Drought preparedness emphasises improved coordination of responses to droughts. Clarke and Hill (2013) found that combining resources to respond to droughts at regional level in Sub-Saharan Africa was more effective and cheaper that 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. For example, policies aimed at improving water use efficiency in different sectors of the economy, 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.

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. Famine Early Warning System Network (FEWSNet) in East 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. 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, Zhang, Xu, and Vo, 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 diversity of water property instruments and 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 is high uncertainty 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 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 climate system, however, the information on the future interactions between desertification and climate systems, as modified by future land use and land cover changes, is not available.

High uncertainty persists also on 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. 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 over the past years, 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 would affect people, ecosystems and biodiversity in the future. Filling these gaps requires considerable investments in research and data collection.

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