Chapter 6: Interlinkages between Desertification, Land Degradation, Food Security and GHG fluxes: synergies, trade-offs and Integrated Response Options

Coordinating Lead Authors: Johnson Nkem (Cameroon), Pete Smith (United Kingdom)

Lead Authors: Katherine Calvin (United States of America), Donovan Campbell (Jamaica), Francesco Cherubini (Norway/Italy), Giacomo Grassi (Italy), Vladimir Korotkov (Russian Federation), Anh Le Hoang (Vietnam), Shuaib Lwasa (Uganda), Pamela McElwee (United States of America), Ephraim Nkonya (Tanzania), Nobuko Saigusa (Japan), Jean-Francois Soussana (France), Miguel Angel Taboada (Argentina)

Contributing Authors: Cristina Arias-Navarro (Spain), Otavio Cavalett (Brazil), Alexander Popp (Germany), Matteo Vizzarri (Italy)

Review Editors: Ian Noble (Australia), Yoshiki Yamagata (Japan), Taha Zatari (Saudi Arabia)

Chapter Scientists: Frances Manning (United Kingdom), Dorothy Nampanzira (Uganda)

Date of Draft: 08/06/2018

Table of Contents

6.1 Executive summary

Response options have interlinked implications across the challenges in the land sector; options to address one land challenge may exacerbate other problems (*robust evidence; high agreement*) (Sections 6.4-6.9). Among the response options available to address the land challenges, many have impacts across more than one challenge either by delivering co-benefits across a range of challenges, for example many sustainable land management practices co-deliver to climate change mitigation and adaptation, to preventing or addressing desertification and land degradation, and to food security (*robust evidence; high agreement*) (Section 6.9). Other response options create adverse-side effects for one or more challenges, for example response options that demand land for climate mitigation, could cause adverse side effects if implemented at scale for food production, and thereby food security, *via* increasing competition for land (*robust evidence; high agreement*) (Sections 6.4, 6.9).

Land resources are limited. Competition for land may restrict the scale at which response options can be used (*robust evidence; high agreement*). The land is a finite resource and expansion of the current area of managed land into natural ecosystems would lead to the loss of biodiversity and a range of ecosystem services (*robust evidence; high agreement*) (Section 6.3). For this reason, the scale at which some response options can be applied is limited, with response options that compete for land, e.g. afforestation, BECCS, most affected (*robust evidence; high agreement*) (Section 6.9). Other options that can be applied without changing the use of the land, for example measures to increase the soil organic matter (carbon) content of soils, are not limited by land competition constraints (*robust evidence; high agreement*) (Section 6.9).

The impacts of many response options are scale and context dependent, and are uneven across different regions and communities (*robust evidence; high agreement*) (Sections 6.4-6.9). The efficacy, and impacts, of response options to address each land challenge is location specific, with for example, the mitigation effectiveness or adaptation effectiveness differing by bioclimatic region, land management system or local food system context (*robust evidence; high agreement*) (Sections 6.4-6.9). Further, for some scalable response options, large global impacts are seen only when implemented at large scale (Sections 6.4-6.9). In addition, impacts are context dependent, with certain options producing adverse side-effects only in certain locations, for example response options that use freshwater might have no adverse side effects in regions where water is plentiful, but large adverse side effects in regions where water is scarce (*robust evidence; high agreement*) (Section 6.9).

All land challenges need to be considered when addressing potential solutions, in order to identify response options that co-deliver across the range of challenges (*robust evidence; high agreement*) (Section 6.4-6.9). Because the different land challenges are often the concern of different policy and research communities, response options are often proposed to address a specific land challenge. However, since all land challenges share the same land resource, response options can affect, positively or negatively, a number of land challenges. For this reason, considering the impact of response options on all land challenges simultaneously will allow co-benefits to be maximised and adverse side-effects to be minimised (*medium evidence; high agreement*) (Section 6.9).

Many response options (over 40 in number) have multiple co-benefits across land-related goals, but some are not currently widely implemented (*robust evidence; high agreement*) (Section 6.9). The majority of response options considered have potential to deliver co-benefits across the range of land challenges, with co-benefits ranging from large to small across options and challenges (*robust evidence; high agreement*) (Section 6.9). Other options deliver large co-benefits for one or two challenges, with negligible impact on others, but do no harm. Many of the same options also have either no, or small, context-specific adverse side-effects (*robust evidence; high agreement*) (Section 6.9). There are, therefore, a range of “no regrets” options that are suitable to wider implementation to address multiple land challenges. While some of these response options are implemented widely in some regions, other are not, and even for those with wide regional adoption, there is considerable scope for wider deployment globally (*robust evidence; high agreement*) (Section 6.9).

Some response options, such as large-scale BECCS, have the potential to deliver very well for one land challenge only, with potential detrimental effects on other land challenges (*robust evidence; high agreement*) (Section 6.9). A small number response options have very large potential to address one land challenge, but could lead to large adverse side-effects if implemented at scale. Options that require land use change (e.g. BECCS, afforestation), and thereby contribute to land competition, are most prevalent in this category, with food security the land challenge most often adversely affected (*robust evidence; high agreement*) (Section 6.9). Options that improve land management or improve efficiency of production of food and fibre (sustainable land management options) do not fall into this category and they either do not affect competition for land, or have the potential to decrease it (*robust evidence; high agreement*) (Section 6.9).

There are currently barriers to implementation for many response options; identifying and removing barriers is necessary to make progress toward sustainable solutions (*robust evidence; high agreement*) (Section 6.9). Since there is good evidence that many response options will deliver multiple co-benefits across the range of land challenges, yet these are not applied universally, is evidence that multiple barriers to implementation exist (*robust evidence; high agreement*) (Section 6.9). A combination of economic, biophysical, technological, institutional, education, cultural and behavioural barriers exist for each response option in various regions, and these barriers need to be overcome if response options are to be more widely applied (*robust evidence; high agreement*) (Section 6.9). Options aiming to preserve ecosystem services and biodiversity depend largely on land governance and financial aid, since markets where such services can be traded are not well developed. Improved institutional frameworks would strengthen land governance and facilitate efforts to preserve ecosystem services and biodiversity (*medium evidence; high agreement*) (Section 6.9).

Coordinated action is required across a range of actors, including consumers, land managers, indigenous and local communities and policymakers (*robust evidence; high agreement*) (Section 6.9). Since barriers to implementation are economic, biophysical, technological, institutional, education, cultural and behavioural, action is required across a multiple actors (*robust evidence; high agreement*) (Section 6.9). Because of the wide range of actors, and the wide range of impacts to be considered across the land challenges, action to address barriers to implementation would be most effective if action were coordinated across the range of actors, including consumers, land managers, indigenous and local communities and policymakers (*robust evidence; high agreement*) (Section 6.9).

The need to act is urgent. Delayed action will result in an increased need for response and a decreased potential of response options due to climate change and other pressures (*robust evidence; high agreement*) (Section 6.9). Delayed action to address any of the land challenges of climate change, desertification, land degradation and food security make the challenges more difficult to address in future, and often make the response options less effective. For example, failure to mitigate climate change with increase requirements for adaptation, and may reduce the efficacy of future mitigation options, e.g. by reducing the sink capacity for soil and vegetation carbon sequestration (*robust evidence; high agreement*) (Section 6.9). For this reason, and the extent of the land challenges currently, the need to act is urgent (*robust evidence; high agreement*) (Section 6.9).

Though there are gaps in knowledge and more R&D is required for many response options, enough is known to take action now (*robust evidence; high agreement*) (Section 6.9). There are knowledge gaps for some response options, both in their efficacy and in their broader impacts, particularly among the more recently emerging options (e.g. enhanced weathering of minerals, BECCS; (*robust evidence; high agreement*) (Section 6.9). Nevertheless, many response options have been practiced in some regions for many years and have a broad evidence base, so could be applied more widely immediately, with little risk of adverse side-effects if the best available knowledge is used to design implementation plans for these "no regrets" options (*robust evidence; high agreement*) (Section 6.9).

Cost-effective no regrets options are available for immediate local application, providing that compliance with sustainable development is considered (*robust evidence; high agreement*) (Section 6.9). Many "no regrets" response options which delver across the range of land challenges and beyond (e.g. improved dietary health through improved diets) are also cost-effective, with many being low cost, and some even cost negative (*robust evidence; high agreement*) (Section 6.9). Where not already applied due to local barriers to implementation, these response options are available for immediate application, if barriers can be removed. Assessing impacts against the Sustainable Development Goals (and indicators thereof), or other components of sustainable development, would provide a safeguard against inappropriate local implementation (*medium evidence; high agreement*) (Section 6.9).

Creating an enabling environment, including local engagement, to facilitate the adoption of no-regrets options is required (*robust evidence; high agreement*) (Section 6.9). In addition to the need to engage multiple actors, and to assess the impact of implementation across the range of land challenges and against compliance with sustainable development, implementation of response options would be facilitated by local engagement, and the creation of an enabling environment under which the barriers to implementation could be overcome (*medium evidence; high agreement*) (Section 6.9). Policy will require to address all of these issues (*medium evidence; high agreement*) (Section 6.9; Chapter 7).

6.2 Introduction

6.2.1 Context of this chapter

This chapter focuses on the interlinkages between sustainable land management options[1] to deliver climate mitigation and adaptation, to prevent desertification and degradation, and to enhance food security, and also assess reported impacts on ecosystem services and contributions to the UN Sustainable Development Goals (SDGs). By identifying options that provide many co-benefits with few adverse side-effects, the chapter aims to provide *integrative response options* that could co-deliver across the range of challenges. This chapter *does not consider*, in isolation, sustainable land management or other response options to deliver climate mitigation and adaptation, to prevent desertification and degradation, or to enhance food security, since these are the subjects of chapters 2-5; this chapter *considers only* interlinkages between two or more of these challenges in the land sector.

Since we aim to assess and provide guidance on integrative response options, we examine, in turn, land based interventions that are designed to deliver solutions to each of the challenges of climate mitigation / adaptation, prevention of desertification, prevention of land degradation, and food security – and for each set of interventions, we examine the co-benefits and adverse side-effects[2] on each of the other challenges. In the final section, we assess which response options provide the greatest co-benefits and synergies, and the fewest adverse side-effects for the other challenges. The aim of the final section is to identify which options have the greatest potential to co-deliver across the challenges, and the contexts and circumstances in which they do so.

In providing this evidence-based assessment, drawing on the relevant literature, we do not assess the merits of policies to deliver these interventions - chapter 7 assesses the various policy options currently available to deliver these interventions - rather we provide a list of interventions that are best able to co-deliver across the multiple challenges addressed in this SR. We use case studies in this chapter where these improve clarity.

6.2.2 Definitions of co-benefits and adverse side-effects

In this chapter we examine co-benefits and adverse side-effects. The definitions provided below help to clarify why interlinkages are largely assessed in terms of co-benefits and adverse side-effects.

Co-benefits: We use the IPCC AR5 WGIII definition of co-benefits: “The positive effects that a policy or measure aimed at one objective might have on other objectives, without yet evaluating the net effect on overall social welfare. Co-benefits are often subject to uncertainty and depend on, among others, local circumstances and implementation practices. Co-benefits are often referred to as ancillary benefits.” The AR5 WGII and SYR definitions of co-benefits differ only slightly.

Adverse side-effect is defined by AR5 WGIII as: “The negative effects that a policy or measure aimed at one objective might have on other objectives, without yet evaluating the net effect on overall social welfare. Adverse side-effects are often subject to uncertainty and depend on, among others, local circumstances and implementation practices.”

We use these definitions to characterise inter-linkages between the response options proposed for dealing with each land-based challenge. In section 6.3 we outline the approach taken.

[1] Footnote: The majority of interventions considered are sustainable land management options, but a few response options are not based on land management, for example demand-side measures to address food security, community based adaptation interventions and crop insurance programmes for adaptation.

[2] Footnote: Though many of the co-benefits and adverse side-effects are biophysical, some are socio-economic in nature, and these were also assessed.

6.3 Framing the discussion of combined and interactive effects

In this section we outline the approach used in assessing the evidence for interactions between response options to deliver climate mitigation and adaptation, to prevent desertification and land degradation, and to enhance food security.

6.3.1 Examining multi-dimensional interactions between response options

The procedure for assessing the evidence on interlinkages between response options to deliver climate mitigation and adaptation, to prevent desertification and land degradation, and to enhance food security was as follows:

1) Through reference to chapters 2 to 5, identify the response options (the list of sustainable land management options and other interventions) proposed to address each challenge, in turn, addressed in this SR
2) For each response option, assess its impact on each of the other land challenges in this SR and categorise these interlinkages as either co-benefits or adverse side-effects with reference to chapter 2 to 5 where these were noted, and by assessing the literature.
3) Where evidence exists, note in each section the impact of each response option on ecosystem services (using the list of ecosystem services provided in (Díaz et al. 2018))
4) Where evidence exists, note in each section the impact of each response option on the UN SDGs

Each main section below (6.4 to 6.8) lists the response options (sustainable land management options and other interventions) proposed to address each challenge (climate mitigation and adaptation, desertification, land degradation and food security, drawing on chapter 2-5). Where noted in each chapter, potential co-benefits, and adverse side-effects with each of the other challenges were used – supplemented with an assessment of the literature (since most chapters do not consider co-benefits and adverse side-effects beyond their relationship with climate change mitigation and adaptation). Where evidence exists, the impacts on ecosystem services and the UN SDGs are also noted.

For example, when considering land based mitigation response options, soil carbon sequestration is a possible response option, and its potential co-benefits or adverse side-effects, with each of climate adaptation, prevention of desertification and land degradation and delivery of food security, was assessed through reference to chapter 2 to 5 and the literature. In this way, each response option proposed for addressing each land-based challenge, was assessed using evidence from earlier chapters and the literature.

We assess the literature examining current and historical interlinkages (i.e. those that have been observed and reported in the literature), but also interlinkages envisaged in scenarios of future interventions, as implemented in integrated assessment models, and other model-based approaches examining potential futures. The issues associated with historical and scenario-based response options are discussed in more detail in 6.3.2 and 0.

6.3.2 Response options examined in this chapter in current and historical interventions

Multiple interlinkages between land degradation, desertification, food security, biodiversity and climate change have been reported in previous chapters, with focus on biophysical land-climate interactions (Chapter 2) and on impacts and responses of desertification (Chapter 3), land degradation (Chapter 4) and food security (Chapter 5), respectively. Here, we provide historical and current examples of such interlinkages between challenges and of land-based response options in human-dominated and 'wild' ecosystems.

There is an extensive and globally-relevant scientific literature on the historical and current role of specific land-based mitigation options (see Chapter 2; Smith et al. 2014), including forest management and restoration (Canadell and Raupach 2008; Stanturf et al. 2014) agriculture soils and livestock management (FAO (Food and Agriculture Organization) 2010; Paustian et al. 2016), agro-forestry systems (Ramachandran Nair et al. 2010) and the restoration of wetlands and peatlands (Leifeld and Menichetti 2018).

By contrast, until recently relatively fewer studies assessed - mostly at regional level - the interlinkages between options, e.g. on the role of agriculture intensification for reducing deforestation (Lapola et al. 2014), or between challenges, e.g. between mitigation and adaptation (Locatelli 2011). The reason is that analysing the co-benefits, adverse-effects and trade-offs of land-based response options is challenging for a number of reasons (Bustamante et al. 2014a). First, the effects of each option depend on the context and the scale of the intervention, i.e. the effects are site-specific, and generalisations are difficult. Second, effects do not necessarily overlap geographically, socially or temporally. Third, there is no agreement on how to attribute co-benefits and adverse-effects to specific mitigation measures; and fourth there are no standardised metrics for quantifying many of these effects. However, an increasing numbers of tools are available allowing integrated assessment of multiple outcomes for different challenges (e.g., Vogt et al. 2011; Townsend et al. 2012; Smith et al. 2013; Turner et al. 2016). This is reflected also in the rapidly increasing interest in global-level integrated approaches, taking into account the sustainable development (e.g. Dooley and Kartha 2018), planetary boundaries (Heck et al. 2018) and with a focus on nature-based solutions (Griscom et al. 2017a; Nesshöver et al. 2017).

The human domination of ecosystems has resulted in the development of anthropogenic biomes (or anthromes). Ellis and Ramankutty (2008) identified six major anthrome types through empirical analysis of global population, land use, and land cover: dense settlements, villages, croplands, rangelands, forested and wildlands (without evidence of human occupation or land use, about 22% of Earth's ice-free land) (Figure 6.1). Agricultural land-based response options (see 6.4.2 and 6.5.2) tend to dominate in the croplands and rangelands anthromes, forestry responses (see 6.4.1 and 6.5.1) in the forested anthromes and ecosystem based adaptation (see 6.5.4) responses in the wild anthromes. Specific village and dense settlements land-based response options were also documented in the literature (e.g. Ahrends et al. 2010; Huber-Sannwald et al. 2012; Hassan and Nazem 2016).

Anthromes are exposed to multiple challenges, including land degradation, climate change, food insecurity, water stress and threatened biodiversity. The spatial distribution of individual land challenges is shown in Figure 6.2, based on recent studies:

- Recent land degradation is estimated from long-term (1982-2006) NDVI decline by correcting for rainfall and afforestation and by masking areas with saturated NDVI (Le Quéré et al. 2016);

- Climate change is estimated from the magnitudes of change in local climates between 2000 and 2070 following the dissimilarity index calculated by Neftel et al. (2017), contrasting slow (dissimilarity index below 0.7) and rapid (index equal to 0.7 or above) climate change;

- Food insecurity is shown as the percentage prevalence of chronic undernourishment (higher or equal to 5%) by country in 2015 (FAO 2017);

- Threatened terrestrial biodiversity hotspots correspond to areas where exceptional concentrations of endemic species are undergoing exceptional loss of habitat (Myers et al., 2008; revisited by Conservation International, 2011).

- Groundwater stress is estimated for ratios of groundwater abstraction over recharge above one and is mapped for the Cropland and Village anthromes, which abstract water for irrigation (WRI, 2017).

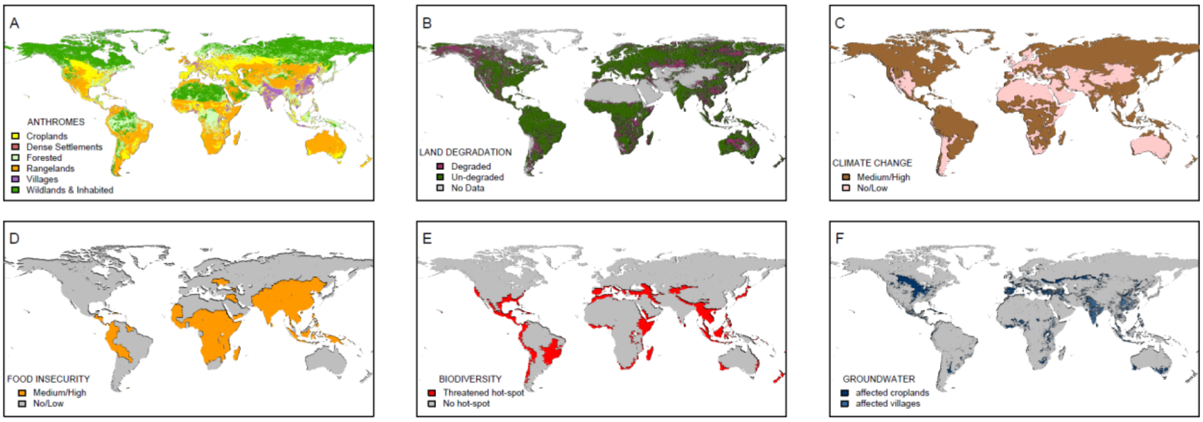

Figure 6.1Global maps of (A) anthropogenic biomes (or anthromes, after Ellis and Ramankutty 2008): dense settlements, villages, croplands, rangelands, forested (semi-natural forests) and wild and inhabited lands (including primary forests and barren); B land degradation (Le et al., 2016); C climate change (Netzel et al., 2017); D, food insecurity (FAO 2017); E, threatened biodiversity hotspots (Conservation International, 2011); F, groundwater stress affecting cropland and village anthromes (World Resources Institute, 2018). For definitions, see text.

Table 6.1 Anthrome area (% ice-free land) and anthrome percentage exposure to individual challenges (see text for definitions)

Anthrome[1]	**Anthrome area**	**Rapid climate change**[2]	**Land degradation**[3]	**Food insecurity**[4]	**Threatened biodiversity hotspot**[5]	**Ground water overuse**[6] **(croplands & villages)**
	% of ice-free land area (1)		*% anthrome area exposed to an individual challenge*			
Dense settlement	1.2	75.0	17.5	30.0	31.9	-
Village	5.4	69.9	24.0	76.5	28.2	66.1
Cropland	14.7	71.1	21.8	27.2	27.1	61.6
Rangeland	26.8	45.0	23.8	42.6	20.4	-
Forested (semi-natural)	14.0	91.1	17.5	36.6	20.6	-
Wild & Inhabited	38.0	77.3	17.2	12.5	2.6	-
All anthromes	100.0	69.3	20.0	29.7	15.2	61.7

(1) Ellis and Ramankutty (2018); (2) Netzel et al. (2017); (3) Le et al., 2016; (4) FAO (2017) (% prevalence of undernourishment by country in 2015); (5) Conservation International (2011) after Myers et al. (2008); (6) World Resources Institute, Aqueduct database (2018).

Anthromes occupy contrasted shares of the ice-free land area, with dense settlements and villages concentrating the majority of the global population in less than 7% of the area, while semi-natural forests, wild and inhabited anthromes occupy more than half of the ice-free land area on a global scale (Table 6.1). Rapid climate change affects close to 70% of the ice-free land area, while the land degradation and food insecurity challenges are concentrated in about 20% and 30% of the global land. All anthromes host threatened biodiversity hotspots. Irrigation potential is constrained by groundwater overuse in more than 60% of the Cropland and Village anthromes and the latter are strongly exposed to food insecurity (Table 6.1).

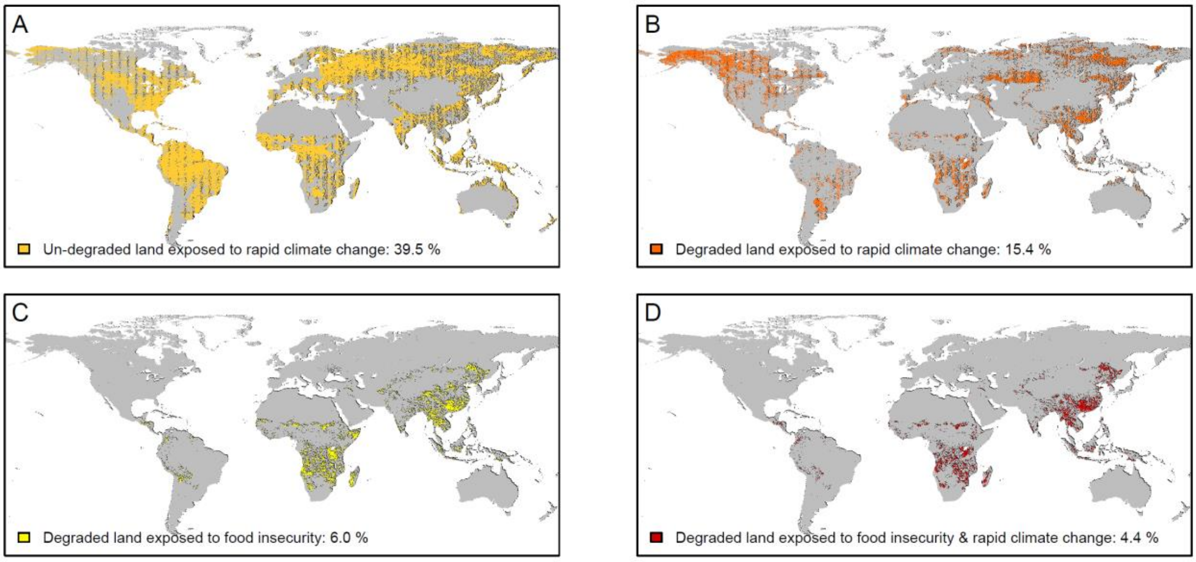

Figure 6.2 Spatial distribution of exposure to selected multiple land challenges. A. Un-degraded land exposed to rapid climate change; B, Degraded land exposed to rapid climate change; C, Degraded land exposed to food insecurity; D, Degraded land exposed to rapid climate change and food insecurity (for definitions, see text; references as in Figure 6.1)

Approximately 15% of the global ice-free land area is exposed to a combination of land degradation and rapid climate change, while the combination of the land degradation and food insecurity challenges is predominantly observed in sub-Saharan Africa and in South Asia. 4.4% of the global ice-free land area is exposed to a combination of land degradation, rapid climate change and food insecurity with largest areas also in sub-Saharan Africa and South Asia (Figure 6.3).

Anthromes used for agriculture show contrasted regional distributions of the combined land degradation, rapid climate change and food insecurity challenges (Figure 6.5), with largest affected areas located in South and East Asia for the Village anthrome, in Sub-Saharan Africa, East Asia and Latin America for the Rangeland anthrome and for the Cropland anthrome areas affected in multiple regions including Asia, sub-Saharan Africa and central America. Anthromes not used for agriculture also show contrasted distributions of areas exposed to combinations of rapid climate change, land degradation and threatened biodiversity hotspots with the largest area in South-East Asia for the semi-natural forests anthrome, while scattered and relatively small areas are exposed to a combination of these challenges in the Wild and Inhabited anthrome (Figure 6.5).

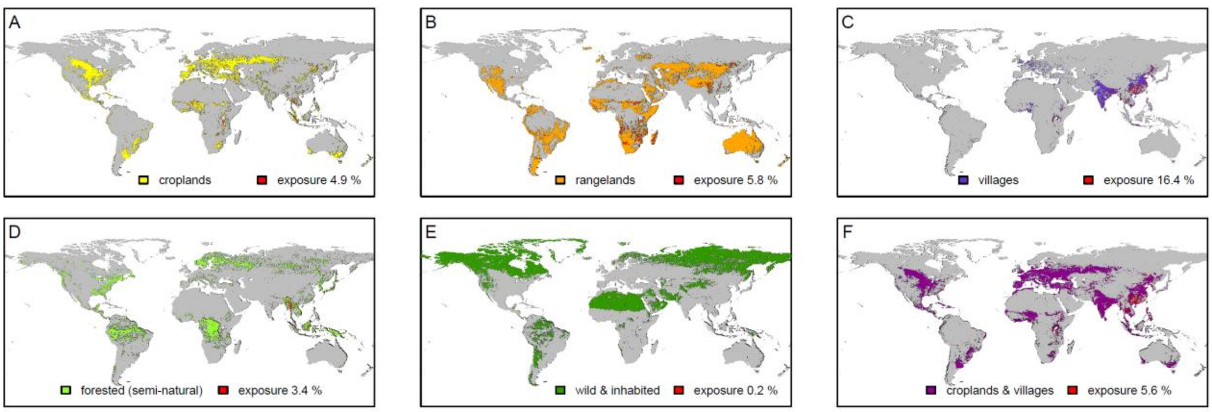

Figure 6.3 Spatial distribution of exposure to selected multiple challenges by anthrome. A, B, C. Cropland, Village and Rangeland anthromes and their exposure to land degradation, rapid climate change and food insecurity. In F, exposure to groundwater stress, rapid climate change, land degradation and food insecurity is mapped for both Cropland and Village anthromes. Semi-natural Forests (D) and Wild and

Inhabited (E) anthromes and their exposure to land degradation and rapid climate change in areas with threatened biodiversity hotspots. In red, anthrome area exposed to the selected multiple challenges. In grey, areas not covered by the anthrome.

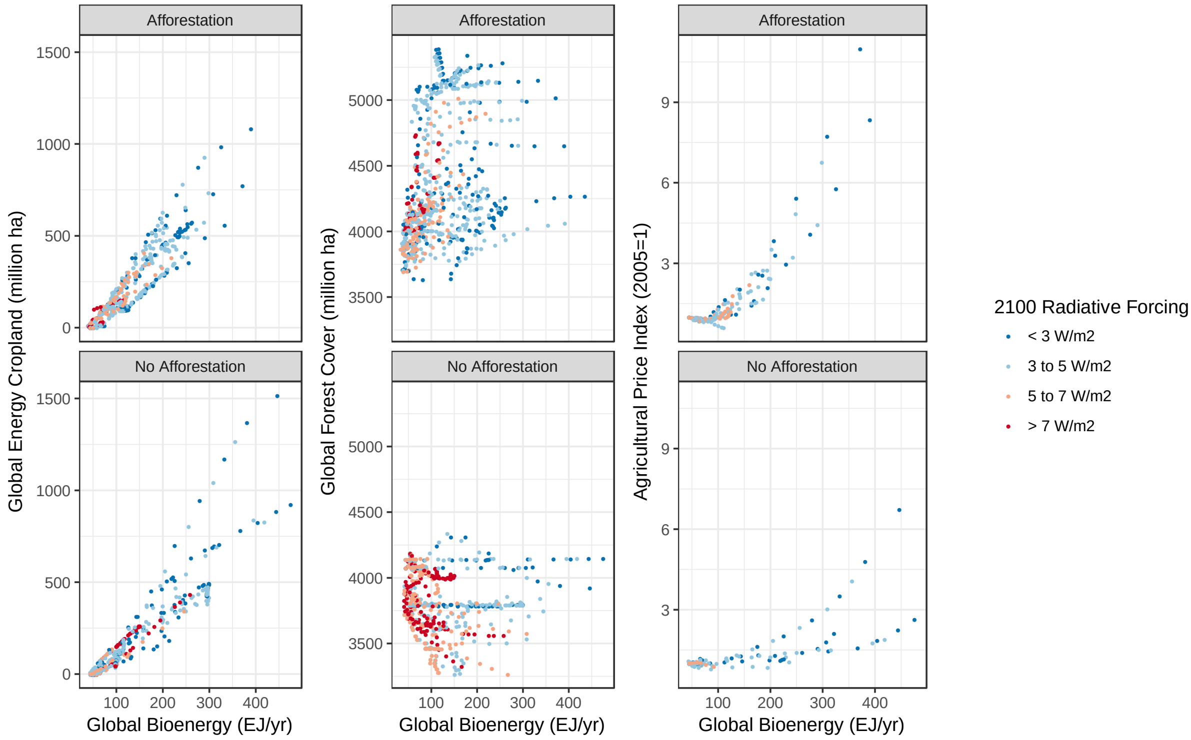

Figure 6.4 Relationship in scenarios from Integrated Assessment Models between future global production of energy from bioenergy, with areas of energy crops (left panels), areas of forest cover (centre panels) and agricultural price index (right panels) under conditions of either afforestation (top panels) or no afforestation (bottom panels)

The global land distribution by anthrome and by number of local land-based challenges (

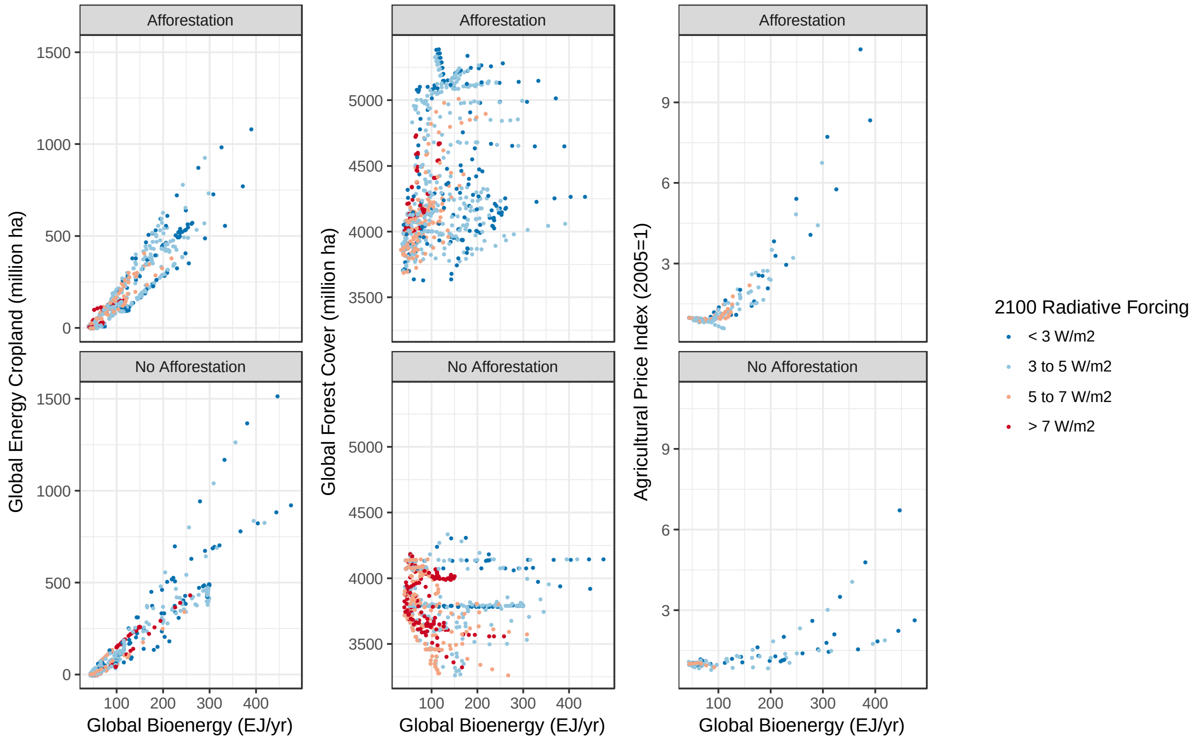

Figure 6.6) shows less frequent exposure to multiple challenges in the Wild & Inhabited anthrome compared to Semi-Natural forests and Rangelands anthromes (often exposed to one challenge or more), to Croplands and Dense Settlements anthromes (often exposed to two challenges or more) and to the Villages anthrome (often exposed to three challenges or more). Therefore, there is a general trend of increased exposure to multiple land challenges in anthromes which are used more intensively.

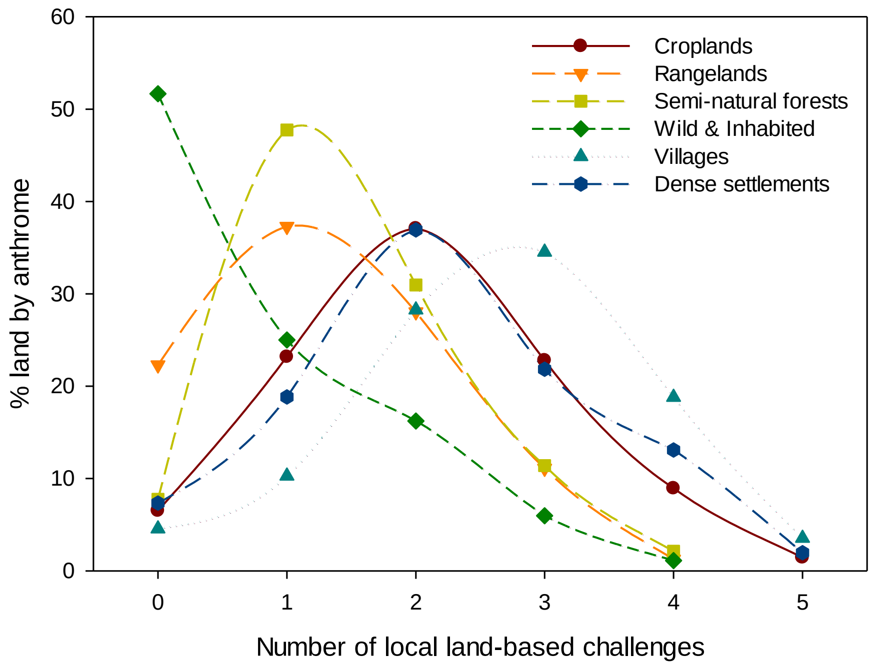

Figure 6.5 Percentage distribution of land area by anthrome and by number of local land-based challenges including: i) land degradation (and desertification in drylands), ii) rapid climate change, iii) food insecurity, iv) threatened biodiversity hot-spot, v) depleted groundwater resources (in anthromes abstracting water: croplands, villages and dense settlements).

Case studies located in different world regions are presented for each anthrome, in order to provide historical context on the interlinkages between multiple challenges and responses (Box 6.1: A to F). Taken together, these case studies illustrate the large contrast across anthromes in land-based interventions and the way these interventions respond to combinations of challenges.

Box 6.1 Case studies by anthrome type showing historical interlinkages between land-based challenges and the development of local responses.

A. Croplands. Land degradation, groundwater stress and food insecurity: soil and water conservation measures in the Tigray region of Ethiopia

In northern Ethiopia, the Tigray region is a drought-prone area that has been subjected to severe land degradation (Frankl et al. 2013) and to recurrent drought and famine during 1888-1892, 1973–1974 and 1984–1985 (Gebremeskel et al. 2018). The prevalence of stunting and underweight among children under five years is still high (Busse et al. 2017) and the region was again exposed to a severe drought during the strong El Niño event of 2015-2016. Croplands are the dominant land-use type in these highlands, with approximately 90% of the households depending on small-scale plough-based cultivation. Gullies affect nearly all slopes and frequently exceed 2m in depth and 5m in top width. Landsat imagery shows that cropland area peaked in 1984-1986 and increased erosion rates in the 1980s and 1990s caused the drainage density and volume to peak in 1994 (Frankl et al. 2013). Since ca. 2000, the large-scale implementation of soil and water conservation (SWC) measures, integrated catchment management, conservation agriculture and tree regeneration started to yield positive effects on the vegetation cover and led to the stabilisation of about 25% of the gullies by 2010 (Frankl et al. 2013).

Since 1991, farmers provide labour for SWC during January as free service for 20 consecutive working days, followed by food for work for the remaining days of the dry season. Most of the degraded landscapes are restored, with positive impacts over the last two decades on soil fertility, water availability and crop productivity. However, mismanagement of fertilisers, low survival of tree seedlings and lack of income from exclosures may affect the sustainability of the land restoration measures (Gebremeskel et al. 2018).

B. Rangelands. Biodiversity hotspot, land degradation and rapid climate change: pasture intensification in the Cerrados of Brazil.

Cerrados are a tropical savannah ecoregion in Brazil corresponding to a biodiversity hot spot with less than 2% of its region protected in national parks and conservation areas (Cava et al. 2018). It has undergone extensive cattle ranching (limited mechanisation, low use of fertiliser and seed inputs) through pasture expansion, including clearing forests to secure properties rights, occurring mainly over 1950-1975 (Martha et al. 2012). Despite observed productivity gains made over the last three decades (Martha et al. 2012), more than half of pasture area is degraded to some extent and challenges remain to reverse grassland degradation, while accommodating growing demand and simultaneously avoiding the conversion of natural habitats (De Oliveira Silva et al. 2018). The largest share of production is on unfertilised pastures often sown with perennial forage grasses of African origin, mainly *Brachiaria* spp. (Cardoso et al. 2016). This initial intensification era was partly at the expense of significant uncontrolled deforestation and average animal stocking rates remained well below the potential carrying capacity (Strassburg et al. 2014). Changes in land use are difficult to reverse since pasture abandonment does not lead to the spontaneous restoration of old-growth savannah (Cava et al. 2018), moreover pasture to crop conversion is frequent, supporting close to half of cropland expansion in Mato Grosso state over 2000-2013 (Cohn et al. 2016). Pasture intensification through liming, fertilisation and controlled grazing increases soil organic carbon and reduces net GHG emission intensity per unit meat product, but only at increased investment cost per unit of area (de Oliveira Silva et al. 2017). Scenarios projecting a decoupling between deforestation, which has already been significantly reduced (−82% emissions from deforestation over 2004- 2014 in the national inventory), and an increased pasture intensification, provides the basis for an Nationally Determined Contribution (NDC) of Brazil that is potentially consistent with accommodating an upward trend in livestock production to meet increasing demand (De Oliveira Silva et al. 2018).

C. Semi-natural forests. Biodiversity hotspot, land degradation, rapid climate change and food insecurity: restoration and resilience of tropical forests in Indonesia

During the last two decades, forest cover in Indonesia reduced by 11.5 Mha in the period 1990-2000 (Stibig et al. 2014), and of approximately 15.8 Mha in the period 2000-2012 (Hansen et al. 2013), mainly due to the conversion of tropical forests into agricultural lands (e.g. oil palm, pulpwood plantations). According to the most recent estimates, deforestation in Indonesia mainly concerns primary, intact, and degraded forests, thus strongly contributing to biodiversity loss, and to the reduction of carbon sequestration potentials (e.g., Margono et al. 2014). For example, Graham et al. (2017) estimated that the following REDD+ strategies may cost-effectively increase carbon sequestration and reduce carbon emissions in 30 years: reforestation (965 MtC), limiting the expansion of oil palm and timber plantations into forest (836 MtC and 831 MtC, respectively), reducing illegal logging (638 MtC), and halting illegal forest loss in Protected Areas (414 MtC); at a total cost of USD 15.7 tC^{-1}. The important role of forest mitigation in Indonesia is confirmed by the Nationally Determined Contribution, where between half and two-thirds of the 2030 emission target relative to business-as-usual scenario is expected to derive from reducing deforestation, forest degradation, peatland drainage and fires (Grassi et al. 2017). In particular, avoiding deforestation and reforestation have multiple co-benefits with adaptation by improving biodiversity conservation, and employment opportunities, while reducing illegal logging in protected areas and afforestation have adverse side-effects since they may deprive local communities' access to natural resources, and reduce highly-diverse (but low-carbon) grasslands (cf. Graham et al. 2017). On the adaptation side, the adoption of the Roundtable on Sustainable Palm Oil (RSPO) certification in oil palm plantations reduced deforestation rates of approximately 33% in the period 2001-2015 (co-benefits with mitigation), and fire rates much more than for non-certified plantations (Carlson et al. 2018). However, considering that oil palm plantations are one of the most impacting driving forces of deforestation in Indonesia (e.g., exacerbated in Borneo), it is argued that RSPO still lacks of information about land-clearing trajectories and of comprehensive assessments (Gaveau et al. 2016). About adaptation options, the community forestry scheme "Hutan Desa" (Village Forest) in Sumatra and Kalimantan was estimated to contribute to avoid deforestation (co-benefits with mitigation: between 0.6 and 0.9 ha km^{-2} in Sumatra, and between 0.6 and 0.8 ha km^{-2} in Kalimantan in the period 2012-2016; Santika et al. 2017), improve local livelihood options, and restore degraded ecosystems (positive side-effects for ecosystem services provision) (e.g., Pohnan et al. 2015). Finally, the establishment of Ecosystem Restoration Concessions in Indonesia (more than 0.55 Mha of forests now covered, and 1.6 Mha allocated to for the future) facilitates the planting of commercial timber species (co-benefits with mitigation), while assisting natural regeneration, preserving important habitats and species, and improving local well-being and incomes (positive side-effects for ecosystem services provision), at relatively lower costs if compared with timber concessions (Silalahi et al. 2017).

D. Wild anthrome. Biodiversity hotspot, land degradation and rapid climate change: rewilding and managing abandoned agricultural land in Mediterranean Europe

Since the 1950s, farmland abandonment has been occurring primarily in developed countries, in Europe, North America and Oceania, but also in some developing and transition economy countries such as China, mainly as a result of a decline in the agricultural labor and of changing socio-economic factors causing small scale farmers to move to cities (Li and Li 2017). Much of the abandoned agricultural land is likely to display altered soil quality, a depleted native biota with established alien species and poor ecosystem connectivity. Land-use policies in abandoned agricultural landscapes can differ across regions with a dominant focus on pre- or post-abandonment conservation (Queiroz et al. 2014). Since the 1990's Europe has experienced a drastic reduction in agricultural land area and most studies have focused on the conservation of the pre-abandonment status, reporting dominantly negative impacts of agricultural abandonment for biodiversity and ecosystem services (Queiroz et al. 2014). Under wet Mediterranean climate conditions, a catchment in the west of Slovenia rewilded for about 70% of the area over a period of 30-50 years, leading to soil (soil organic matter content, bulk density and aggregate stability, van Hall et al. 2017) as well as landscape benefits (reduction of flood risks and runoff discharge). However, the increase in forest cover reduced the stream flow in summer showing that ecosystem services may decline under extensive rewilding (Keesstra et al. 2018).When not the whole catchment area would be forested, but areas with low erosion risks are transformed in extensively managed grasslands, or in mixed systems combining trees and grasses as in agroforestry, multiple benefits for water resource management, biodiversity (especially endangered bird species from grassland areas) and tourism could be obtained (Keesstra et al. 2018). Another example is provided by the traditional agroforestry system in southwestern Iberian peninsula combining extensive pastures and evergreen oak trees. Land abandonment combined with increased droughts and fires have induced loss of productivity and of tree health (Godinho et al. 2016). Nature based solutions adopted in this context include the use of biodiverse pastures rich in nitrogen-fixing legume species which provide soil cover during the year and tend to increase soil organic matter through enhanced plant productivity (Keesstra et al. 2018). Scenarios show that increasing the nature-based use of farmland, forests, and urban areas could create additional jobs in Europe and increase total socioeconomic benefits of ecosystem services (Maes and Jacobs 2017).

E. Villages. Land degradation, groundwater overuse, rapid climate change and food insecurity: climate smart villages in southern India

Indian agriculture, with 80% of farmers being smallholders (less than 0.5 ha), which combines monsoon-dependent rainfed (58%) and irrigated agriculture, is exposed to climatic variability and climate change. Over the past years, the frequency of droughts, cyclone, and hailstorms increased, with 2002, 2004, 2009, 2012, and 2014 being severe droughts (Rao et al. 2016), as well as 2016-2017, with large negative yield impacts for major crops like wheat (Zhang et al. 2017). The development of submersible pump technology in the 1990s resulted in a dramatic increase of the irrigated agricultural area, which has been supported by public policies that provide farmers free electricity for groundwater irrigation (Shah et al. 2012). This shift caused agricultural practices to depend heavily on irrigation from groundwater and induced a groundwater crisis with has large impacts on socio-ecosystems. An increasing number of farmers report borewell failures for two main reasons: borewells have run dry after excessive pumping, or no water was found in newly drilled borewells. The decrease in groundwater table level suppressed the recharge of river beds, turning main permanent rivers into ephemeral streams (Srinivasan et al. 2015). Wells have recently been drilled in upland areas, where groundwater irrigation is also increasing (Robert et al. 2017). Additional challenges are declining soil organic matter and fertility under monocultures and rice/wheat systems. Land is scarce, meaning that the potential for expanding the farmed area is very limited (Aggarwal et al. 2018). In rural areas, diets were deficient in protein, dietary fiber and iron and revolved around the cereals and pulses grown and/or procured through the welfare programs (Vatsala et al. 2017). Cultivators are often indebted and suicide rates among them are much higher than the national average, especially for those strongly indebted (Merriott 2016). Widespread use of diesel pumps for irrigation, especially for paddies, high use of inorganic fertilisers and crop residue burning lead to high GHG emissions (Aggarwal et al. 2018). The climate-smart village (CSV) approach aims at increasing farm yield, income, input use efficiency (water, nutrients, and energy) and reducing GHG emissions (Aggarwal et al. 2018). Climate-smart agriculture interventions are considered in a broad sense by including practices, technologies, climate information services, insurance, institutions, policies, and finance. Options differ based on the CSV site, its agroecological characteristics, level of development, and capacity and interest of the farmers and of the local government (Aggarwal et al. 2018). The selected interventions included crop diversification, conservation agriculture (minimum tillage, residue retention, laser leveling), improved varieties, weather-based insurance, and agro-advisory services, precision agriculture and agroforestry. Farmers' cooperatives were set up for custom hiring farm machinery, securing government credit for inputs, and sharing of experiences and knowledge. Tillage practices and residue incorporation increased rice–wheat yields by 5%–37% and income by 28%–40% and reduced GHG emissions by 16%–25%. Water-use efficiency also increased by 30% (Jat et al. 2014). The resultant portfolio of options proposed by the CSV approach has been integrated with the agricultural development strategy of some states like Haryana.

F. Dense settlements. Rapid climate change, land degradation and groundwater stress: urban farming and green infrastructuctures in USA

Extreme heat events have led to particularly high rates of mortality and morbidity in cities as urban populations are pushed beyond their adaptive capacities, leading to mortality rates increasing by 30%-130% in major cities from developed countries (Norton et al. 2015). There is evidence that increased mortality and morbidity from extreme heat events are exacerbated in urban populations by the urban heat island effect (Gabriel and Endlicher 2011; Schatz and Kucharik 2015), which can be limited by developing green infrastructures in cities. Urban green infrastructure can be defined as public and private green spaces, including remnant native vegetation, parks, private gardens, golf courses, street trees, urban farming and more engineered options such as green roofs, green walls, biofilters and raingardens (Norton et al. 2015). Increasing the amount of vegetation, or green infrastructure, in a city is one way to help reduce urban air and surface temperature maxima and variation and avoid urban heat. During an extreme heat event in Melbourne, Australia, a 10% increase in vegetation cover was estimated to reduce daytime urban surface temperatures by approximately 1°C (Coutts and Harris 2013). Urban farming, is one component of urban green infrastructures which is largely driven by the desire to reconnect food production and consumption (Thomaier et al. 2015). Even though urban farming can only meet a very small share of the overall urban food demand, it can add to the supply of fresh and local food—especially perishable fruits and crops that usually travel a long way into cities and are sold at high prices. Ground-based urban farming dominates urban food production, but faces growing land availability and soil quality constraints. Food-producing urban gardens and farms are often started by grassroots initiatives that occupy vacant urban spaces, creatively transforming them often. In recent years, a growing number of urban farming projects (termed Zero-Acreage farming, or Z-farming, Thomaier et al. 2015) were established in and on existing buildings, using rooftop spaces or abandoned buildings through contracts between food businesses and building owners. Almost all Z-farms are located in cities with more than 150,000 inhabitants, with a majority in N. America in cities such as New York City, Chicago and Toronto (Thomaier et al. 2015), where they depend on the availability of vacant buildings and roof tops thereby competing with other types of use, such as roof-based solar systems. One critical aspect of urban farming is the potentially high level of soil pollution and of air pollutants in urban settings, which may lead to crop contamination and health risks that could be reduced in controlled environments. Comprehensive assessments of the potential of urban green infrastructures and urban farming for improving diets and health in cities exposed to climate change and rising food demand are however still lacking.

6.3.3 Response options represented in future scenarios

Reference Scenarios: Most studies on mitigation and adaptation include a no response counterfactual scenario, referred to as a "reference scenario" in this chapter. For example, Clarke et al. (2014) examine a large suite of reference scenarios, using different underlying socioeconomic and technological assumptions as well as different IAMs. Collins et al. (2013) quantify the effect of one of these reference scenarios, the RCP8.5, on climate change. Other studies examine the effect of these scenarios on desertification (Cook et al. 2014b; Lin et al. 2015; Scheff and Frierson 2015; Fu et al. 2016), land degradation, and food security (Fujimori and ...; Popp et al. 2017a; Calvin et al. 2014).

Mitigation Options: There is a large body of literature developing and quantifying scenarios of climate change mitigation (Clarke et al. 2014; Riahi et al. 2017a). These scenarios include a variety of energy-related climate mitigation interventions (e.g., changes in energy demand, low carbon energy deployment etc.). However, the inclusion of land-based mitigation options is less comprehensive (see Table 6.2). Most of these scenarios include bioenergy and BECCS; many scenarios include non-CO_2 mitigation in the agricultural sector; some scenarios include afforestation or other forestry efforts. For many of the scenarios in the literature, land-based mitigation options are included as part of a suite of mitigation options (Popp et al. 2017a; van Vuuren et al. 2015). As a result, it is difficult to isolate the effect of an individual option on various goals (e.g., desertification, land degradation, food security). A few studies focus on specific land mitigation policies (Calvin et al. 2014; Popp et al. 2014; Kreidenweis et al. 2016; Humpenöder et al. 2018; EMF33 forthcoming); these studies quantify the effect of including an individual mitigation response option on a variety of goals.

The mitigation scenarios typically quantify the effect of the response options included on climate change and various aspects of mitigation (e.g., mitigation cost, land use, energy use). Some scenarios quantify the effect on food security, as measured by food prices (Popp et al. 2017a; Calvin et al. 2014; van Vuuren et al. 2015), agricultural welfare (Stevanovic et al. 2016) or the number of people at risk of hunger (Fujimori et al.). Iyer et al. (2018) and (Krey and ...) quantify the effect of mitigation on a variety of sustainable development goals, including food security. Pereira et al. (2010a) examine the effect of mitigation, including BECCS and afforestation, on biodiversity, using forest cover as a metric. Humpenöder et al. (2018) carry out a multi-criteria sustainability assessment of large-scale bioenergy crop production throughout the 21st century.

Adaptation Options: Increasingly, the scenario literature has expanded to include adaptation options, mostly focusing on energy, agriculture, and water (Nelson et al. 2014; Kim et al. 2016; Reilly et al. 2007; Calvin et al. 2013; Kyle et al. 2014). For example, the Agricultural Model Intercomparison and Improvement Project (AgMIP) explored the effect of changes in agricultural yield on cropland extent in a suite of economic models and Integrated Assessment Models (Nelson et al. 2014).

The adaptation scenarios often quantify the implications of impacts and adaptation for food security, as measured by food price or food production (Nelson et al. 2014; Kim et al. 2016). However, these scenarios often compound impacts and adaptation; thus, it is difficult to disentangle the effect of adaptation from the residual effect of climate change.

Other Scenarios: There are some scenarios focused on achieving the Sustainable Development Goals (SDGs). For example, Obersteiner et al. (2016) develops scenarios targeted at several different SDGs and assesses the implications of these interventions on food prices. van Vuuren et al. (2015) develop scenarios targeting multiple SDGs and evaluate the implications on other SDGs.

Table 6.2 Scenario Exercises Categorised by the Response Options they Include, as well as whether they quantify implications for various goals (Further references to be added)

		Type		**Quantifies Implications for:**						
	Response Option	Individual Options Isolated	Multiple Options Considered Together	Climate Change	Mitigation	Adaptation	Desertification	Land Degradation	Food Security	Other
Reference	None	N/A	N/A	(Clarke et al. 2014; Collins et al. 2013)			Cook et al. 2014b; Lin et al. 2015; Scheff and Frierson 2015; Fu et al. 2016		(Calvin et al. 2014; Popp et al. 2017b; Fujimori and ...; van Vuuren et al. 2015; Tai et al. 2014)	(Pereira et al. 2010a)
Mitigation	Forestry	(Jones et al. 2015; Calvin et al. 2014)	(Popp et al. 2017a)	(Jones et al. 2015)	(Calvin et al. 2014)				(Calvin et al. 2014)	(Pereira et al. 2010a)
	Agriculture Other Land Management									
	BECCS	(Calvin et al. 2014)	(Popp et al. 2017a; van Vuuren et al. 2015; Fujimori and ...)		(van Vuuren et al. 2015; Popp et al. 2017a; Calvin et al. 2014)				(Calvin et al. 2014; Popp et al. 2017a; van Vuuren et al. 2015; Fujimori and ...)	(van Vuuren et al. 2015; Pereira et al. 2010a)
	Other		(Iyer et al. 2018)		(Iyer et al. 2018)				(Iyer et al. 2018)	(Iyer et al. 2018)
Adaptation	Forestry									
	Agriculture	(Nelson et al. 2014; Calvin et al. 2013; Reilly et al. 2007; Kyle et al. 2014)			(Calvin et al. 2013; Kyle et al. 2014)	(Nelson et al. 2014; Reilly et al. 2007; Calvin et al. 2013; Kyle et al. 2014)			(Nelson et al. 2014)	
	Ecosystem Based Adaptation									
	Other	(Kim et al. 2016)				(Kim et al. 2016)			(Kim et al. 2016)	
Desertification	Soil									
	Vegetation									
	Water Management									
	Integrated Options									
Land Degradati	Soil									
	Vegetation									
	Water Management									

Food Security	Integrated Options							
	Increase food production		(van Vuuren et al. 2015)	(van Vuuren et al. 2015)			(van Vuuren et al. 2015)	(van Vuuren et al. 2015)
	Storage and Distribution Supply							
	Food access and nutrition							
	Demand management options		(van Vuuren et al. 2015)	(van Vuuren et al. 2015)			(van Vuuren et al. 2015)	(van Vuuren et al. 2015)
	Integrated Options							
Other	SDGs	(Obersteiner et al. 2016)	(van Vuuren et al. 2015)	(van Vuuren et al. 2015)			(Obersteiner et al. 2016; van Vuuren et al. 2015)	(van Vuuren et al. 2015)
	Biodiversity							

6.4 Land-based mitigation response options (including negative emissions)

6.4.1 Land-based mitigation response options considered

Land-based mitigation options considered in this section include traditional mitigation options and those more recently called negative emission technologies (NETs), carbon dioxide removal (CDR) or greenhouse gas removal (GGR) options. Depending on future CO_2 emission trends, the total amount of negative emissions needed to achieve the 2°C target is between 100 and 230 GtC (Rogelj et al. 2015; Smith et al. 2016a). These response options can be classified into those relating to forestry, agriculture, other land management, bioenergy with carbon capture and storage (BECCS), and other options. *Forestry* options include sustainable forest management, afforestation/reforestation, reducing deforestation and forest degradation, use of forest wood for energy and material substitution. *Agriculture* options include climate-smart agriculture, cropland management, trees in croplands/agroforestry, nutrient management, improved grazing, improved rice management, livestock management, grazing land management, avoided grassland conversion, soil carbon sequestration (in crop- and grazing-land), fire management, biochar and improved water management. *Other land management* options include coastal restoration of wetlands, peat restoration, avoided peat impacts, avoided coastal impacts, soil carbon sequestration (in natural lands) and fire management. *BECCS* includes bioenergy, and bioenergy with carbon capture and storage, and *other options* include enhanced weathering on minerals, material substitution, harvested wood products and energy substitution. The co-benefits and adverse side-effects associated with these response options for climate mitigation, with interventions to tackle climate adaptation, desertification, land degradation and food security are assessed in the sections below.

6.4.2 Forestry-based mitigation response options

Forest-related land use or land cover changes play a key role for climate change mitigation. The main strategies for forestry-based climate change mitigation include: (a) conserving and/or enhancing the existing forest-related carbon sinks and stocks; (b) reducing forest-related emissions, and (c) using wood-based products to reduce emissions in other sectors. These strategies may be implemented through the following response options: **i) sustainable forest management** of existing forests including practices used for the purpose of **forest landscape restoration**; **ii) afforestation/reforestation**; **iii) reducing deforestation and forest degradation**; **iv) use of forest wood for energy and material substitution**.

In this section, we deal with options i), ii) and iii), whereas mitigation activities based on option iv) are discussed in sections 6.4.5 (energy uses) and 6.4.6 (material uses). Sustainable forest management includes a wide variety of practices, including the regeneration modality (natural or artificial), the species planted, the schedule and intensity of operations (thinnings, selective logging, final cut, etc.), aimed at maintaining and enhancing the economic, social and environmental value of forests, for the benefit of present and future generations (UN 2008). Afforestation and reforestation both refer to establishment of trees on non-treed land. Reforestation refers to establishment of forest on land that had recent tree cover, whereas afforestation refers to land that has been without forest for much longer. Forest landscape restoration specifically aims to regain ecological integrity and enhance human well-being in deforested or degraded forest landscape (Maginnis and Jackson 2007; Stanturf et al. 2014). It includes wildfire treatment (Carter et al. 2015), weed control, planting or encouraging regeneration on degraded (e.g. Chazdon et al. 2016; Chazdon 2017), herbivore control through fencing, establishment of exclosures, and selective hunting, soil amelioration, and improvement through biochar additions (Thomas and Gale 2015). There is evidence that forest ecosystem restoration potentially leads to a cumulative carbon sequestration of 220-330 $GtCO_2$ in the 21st century (Dooley and Kartha 2018).

In general, in the different Shared Socio-economic Pathways (SSPs), the global forest area can change from about – 500 Mha up to + 1000 Mha in 2100, and the contributions of biomass used for energy and expansion of global forests increase under stringent targets for climate change mitigation (Popp et al. 2017a). This large-scale deployment potential gives rise to many possible inter-linkages of forestry-based mitigation options with **climate change, adaptation, desertification, land degradation, food security, and ecosystem services**.

Climate change: Forests interact with the climate system in many different ways (see Chapter 2). Sustainable forest management, forest restoration, afforestation and reforestation mitigate climate change by conserving and enhancing the carbon stock in biomass, dead organic matter and soil – while providing wood-based products to reduce emissions in other sectors through material and energy substitution – and reducing deforestation and forest degradation prevent emissions of carbon (Ciais et al. 2013; Grassi et al. 2017; Pan et al. 2011; Smith et al. 2014b). Trade-offs exist between increasing forest carbon stocks and the yield of wood products, e.g. more harvest decreases the carbon in the forest in the short term but increases the carbon in wood products and the potential for substitution effects (Kurz et al. 2016). The most effective forest carbon mitigation strategy is the one that optimises the carbon stocks (in forests and in long-lived products) as well as the wood substitution effects in a given time frame (Smyth et al. 2014).

Forest-based mitigation options also influence the climate system by modifying biophysical properties such as surface albedo and evapotranspiration (Bonan 2008; Pielke et al. 2011), with potentially large implications for the local and global climate (Findell et al. 2017; Mahmood et al. 2014; Perugini et al. 2017). There is a clear latitudinal pattern in the biophysical climate response to forest cover (Cherubini et al. 2017; Davin and de Noblet-Ducoudre 2010; Li et al. 2016, 2015; Mykleby et al. 2017; Zhang et al. 2014). In the tropics, biophysical effects reinforce the climate benefits of CO_2 sequestration in trees (*co-benefit*). On the other hand, at higher latitude the contribution from a decrease in surface albedo dominates, causing a warming effect that counteracts the carbon benefits of afforestation and reforestation measures *(adverse side-effect*). Avoided deforestation, afforestation, and reforestation in the tropics thus have the major *co-benefits* to mitigate climate change, whereas there are larger *adverse side-effects* at high latitude. In general, the net climate benefits that can be achieved through afforestation are not sufficient alone to compensate for avoidance in reducing fossil CO_2 emissions to achieve temperature stabilisation at low levels, especially when constraints related to biophysical effects and land competition with agriculture and biodiversity are taken into account (Arora and Montenegro 2011; Boysen et al. 2017b; Kreidenweis et al. 2016).

Adaptation: Forest mitigation activities, including the management, preservation and/or expansion of forests, have a number of potential *co-benefits* with regard to adaptation. They may facilitate the adaptation and resilience of forests to climate change by reducing anthropogenic pressures on forests, enhancing connectivity between forest areas and conserving biodiversity hotspots (Ellison et al. 2017; Locatelli et al. 2015a, 2011a; Thuy et al. 2014). There are *co-benefits* for adaptation (e.g. through promoting biological diversity) also in the framework of forest landscape restoration. For example, facilitating tree species mixture means storing at least as much carbon as monocultures while enhancing biodiversity (Hulvey et al. 2013). Similarly, other mitigation activities relevant for forest restoration, such as biochar additions, induce greater aboveground biomass accumulation in native species (Gale et al. 2017). On the contrary, prescribed burning and understory thinning maintain positive biomass increment, despite do not sufficiently promote regeneration (*adverse side-effect*) (e.g., oak stands in Illinois, USA; Carter et al. 2015).

Conserving carbon also protects other ecosystem functions and services, which facilitate the adaptation of society: microclimatic regulation for protecting people from heat stresses and crops from climatic variations, wood and fodder as safety nets, soil erosion protection and soil fertility enhancement for agricultural resilience, coastal area protection, water and flood regulation (Locatelli et al. 2015a). Forest resources have been shown to play a role in enabling local livelihood adaptation during extreme events in some African countries (Dasgupta et al. 2014; Robledo et al. 2012). Well-managed reforestation can contribute to adaptation to climate change by reducing the vulnerability of people and ecosystems to current climate hazards and future climate change. For example, restoring forest cover to coastal areas and hillslopes can stabilise land against catastrophic movements associated with wave action and intense run-off during storms and flood events (Locatelli et al. 2015a).

However, there might be also *adverse side-effects* potentially associated to forest mitigation activities, e.g. with the establishment of non-native species, especially with the risks related to the spread of exotic fast growing tree species (Brundu and Richardson 2016; Ellison et al. 2017). Forest management strategies aiming at increasing the biomass stocking levels may decrease stand-level structural complexity and the adaptation potential (D'Amato et al. 2011), and may make forest ecosystems more susceptive to natural disasters like wind throws, fires, and diseases (Seidl et al. 2014). Maximising carbon (e.g., fast-growing tree monoculture) may lead to management that reduces options for ecological adaptation (Locatelli et al. 2015a). REDD+ projects can restrict the rights and access of local people to land and forest resources, or increase the dependence of local people to insecure external funding (Caplow et al. 2011). Experiments with an earth system model show that reforestation/afforestation, besides mitigation and alleviation of the need for adaptation under extreme heat events, has a limited potential to enhance adaptation in some regions of the world (Sonntag et al. 2016). A study in the Congo Basin based on people's perspectives (Few et al. 2017), identified four main adverse side-effects of forest-based mitigation activities: i) the proximity of forest to cropland constitutes a threat to livelihoods in terms of crop raiding by wild animals; ii) constraints in availability of land for farming; iii) conservation restrictions to preserve ecosystem integrity can restrict the access to resources (e.g. firewood); iv) if strengthened forest management implies greater external intervention, then it may lead to a loss of local power and control over forest resources.

Forestry-based mitigation measures need adaptation, and vice versa. For instance, a REDD+ mitigation project is more likely to be sustainable and long-lasting if it integrates adaptation measures for communities and ecosystems. An adaptation project contributing also to mitigation may benefit from carbon funding and capacity building from international instruments such as REDD+. It is therefore important that national and international policies, donors, standards and capacity building initiatives encourage a policy integration of mitigation and adaptation, e.g. through landscape management

(Locatelli et al. 2015a). Additional discussion on the mitigation-adaptation link is included in section 6.5.2.

Prevention of desertification: Afforestation and reforestation measures have significant *co-benefits* and synergies with preventing desertification, as forests tend to maintain water quality by reducing runoff and trapping sediments and nutrients (Medugu et al. 2010; Salvati et al. 2014). Successful implementation needs to address issues about the sufficient scale of activity to impact watershed balances, the hydrological response without displacing crop production and affecting rural communities, and gaining payment for non-forest benefits (Harper et al. 2017). Due to evapotranspiration, trees recharge atmospheric moisture, contributing to rainfall locally and in distant location, and trees' microbial flora and biogenic volatile organic compounds can directly promote rainfall (Arneth et al. 2010). Trees enhance soil infiltration and, under suitable conditions, improve groundwater recharge (Calder 2005; Ellison et al. 2017; Neary et al. 2009). Soil and water conservation management practices associated to forest restoration options, such as traditional planting pits, stone bounds, and permeable rock dams, also help to prevent desertification (*co-benefits* for adaptation) (e.g., African case studies; Spencer et al. 2017).

If afforestation programs do not sufficiently take into account the local environmental constraints and tree species, the likelihood of *adverse side-effects* increases. For instance, afforestation using some exotic species can upset the balance of evapotranspiration regimes, with negative impacts on water availability particularly in arid regions (Ellison et al. 2017; Locatelli et al. 2015a; Trabucco et al. 2008). Afforestation in arid and semiarid regions using species that have evapotranspiration rates exceeding the regional precipitation may aggravate the groundwater decline (Locatelli et al. 2015a; Lu et al. 2016).

Prevention of land degradation: Afforestation and reforestation options are frequently used to counteract land degradation problems (Yirdaw et al. 2017). Complete reforestation of watersheds has been shown to reduce stream salinity (Harper et al. 2017). Planted forests are seen as a degradation driver when they replace natural forests (*adverse side-effects*) (Brockerhoff et al. 2008), whereas when they are established on degraded lands they are instrumental to preserve natural forests (*co-benefit*) (Buongiorno and Zhu 2014). Logging and fires are the major causes of forest degradation in the tropics. Selective logging techniques are "middle way" between deforestation and total protection, allowing to retain substantial levels of biodiversity, carbon, and timber stocks (Putz et al. 2012), and can therefore offer potential co-benefits in terms prevention of land degradation. Other examples of co-benefits include the effects that forests exert to counteract wind-driven degradation of soils, and the positive effects of mangroves to protect coastal zones from extreme events (hurricanes) or sea level rise. Rehabilitation and restoration of drylands, dry forests, or denuded/degraded forest landscapes, through the combination of sustainable livestock husbandry, small scale farming, and agroforestry systems, may prevent land degradation and subsequent negative consequences (loss of ecosystem services and resilience, food insecurity, and social and political instability; i.e., *synergies* with other challenges; *co-benefits* for adaptation), especially among competing land users (e.g. Yirdaw et al. 2017).

Delivering food security: The competition for land between afforestation/reforestation and agricultural production is a potentially large *adverse side-effect* (Boysen et al. 2017a,b; Kreidenweis et al. 2016; Smith et al. 2013). An increase in global forest area can lead to increases in food prices through increasing land competition (Calvin et al. 2014; Kreidenweis et al. 2016; Reilly et al. 2012; Smith et al. 2013; Wise et al. 2009), and it can also have adverse side-effects for reduction of water yield and water availability for human consumption (Bryan and Crossman 2013). Forest and biofuel plantations for climate change mitigation may impede the adaptation of communities because of decreased food security, competition for land, short-term benefits for few stakeholders, although there are few co-benefits from enhancing food supply from forests (Locatelli et al. 2015a). Future needs for food production are a constraint for large-scale afforestation plans. For example, in a future world with 8.0-9.5 billion people in 2050, afforestation of about 1300 Mha of abandoned cropland and pastures theoretically sequesters about 100 GtC by 2100, a factor that decreases of 80% if pasture land is excluded or of 50% if unfavorable albedo changes are considered to restrict land suitability (Boysen et al. 2017a). An additional challenge is that global food crop demand is expected by 50%–97% between 2005 and 2050 (Valin et al. 2014). Future carbon prices will facilitate deployment of afforestation projects at expenses of food availability (*adverse side-effect*), but more liberalised trade in agricultural commodities could buffer food price increases following afforestation in tropical regions (Kreidenweis et al. 2016).

Ecosystem services: Afforestation/reforestation have important *co-benefits* with ecosystem services, as carbon stocks are a key asset for provisioning, regulating, and life supporting ecosystem services (Knoke et al. 2014). Relative to a baseline, large-scale afforestation programs would increase vegetation and soil carbon stocks (49-76 Gt C), decreases annual runoff (up to 2%, corresponding to around 16%–32% of human runoff withdrawal) and nitrogen losses (up to 13.6%) (*co-benefits*) (Krause et al. 2017). Changes in runoff affect water supply but can also contribute to changes in flood risks, and irrigation of forest plantations can increase water consumption. As forests generally reduce run-off compared to grassland and croplands, historic land-cover changes were responsible for a 7% increase in total runoff (Sterling et al. 2013).

Afforestation/reforestation and avoided deforestation benefit biodiversity and species richness, and generally improve the cultural and recreational value of ecosystems (*co-benefits*) (Knoke et al. 2014). The most recent spatial patterns of forest expansions and contractions do not correlate with the geography of climate trends nor with dry versus moist areas, but trends of forest resources of nations are found to positively correlate with UNDP Human Development Index (Kauppi et al. 2018). This shows that forest resources of nations improve along with progresses in human well-being. Avoided deforestation is key for pristine ecosystems, and has important *co-benefits* with biodiversity and other ecosystem services in the tropics, where deforestation rates are usually high (Hansen et al. 2013) and ecosystems are rich in species (Lewis et al. 2015). Avoided deforestation preserves biodiversity more efficiently and at lower costs than afforestation/reforestation (Rey Benayas et al. 2009). Viewing tropical reforestation primarily as a means of mitigating climate change through carbon sequestration overlooks a suite of other roles such as regulation of land–atmosphere interactions, ecosystem services mediated by biota (e.g. pollination), and societal adaptation to climate variability and change (*synergies*).

In the case of afforestation, simply changing the use of land to planted forests is not sufficient to increase abundance of indigenous species, as they depend on type of vegetation, scale of the land transition, and time required for a population to establish (Barry et al. 2014). There are many studies on the relationship between biodiversity and carbon sequestration, with examples of many *co-benefits* and a few *adverse side-effects*. Precipitation filtered through forested catchments delivers purified ground and surface water (*co-benefits*) (Calder 2005; Ellison et al. 2017; Neary et al. 2009). The relationship between plant species richness and afforestation is context and scale dependent. Selection of priorities for landscape restoration and incentives for planting of diverse species in selected locations is a cost-efficient option to restore natural systems and endangered species habitats in the presence of a carbon market (Crossman et al. 2011). Species richness has a non-linear relationship with forest cover share and it is sensitive to land use types, climatic conditions and land use configuration (Lautenbach et al. 2017). At a global scale and in many local context the relationship is mostly *synergistic* (Bai et al. 2011; Izquierdo and Clark 2012; Paoli et al. 2010; Ren et al. 2017), especially in tropical rainforests where high levels of carbon sequestration can be achieved in major biodiversity hotspots (Larsen et al. 2011; Lewis et al. 2015; Siikamäki and Newbold 2012). However, *adverse side-effects* are identified at country or regional scale where afforestation leads to a decrease in plant species richness (Greve et al. 2013; Lautenbach et al. 2017; Thomas et al. 2013).

Particular activities associated with forest landscape restoration, such as mixed planting, assisted natural regeneration, and reducing impact of disturbances (e.g. prescribed burning) have positive implications for the conservation of native species, tree species richness, availability of wood for energy and industry, and fresh water supply, thus sustaining local livelihood (*co-benefits*) (Ciccarese et al. 2012; Suding et al. 2015). Reducing harvesting rates and prolonging rotation periods may induce, on the one hand, an increased vulnerability of stands to external disturbances and catastrophic events (*adverse side-effect*) (Yousefpour et al. 2018), and on the other hand, the accumulation of carbon in biomass and deadwood has positive effects on biodiversity conservation and nutrient retention (*co-benefit*) (Stutz and Lang 2017). Dense tree planting and fire suppression may induce a reduction of biodiversity and ecosystem functions of other biomes, such as e.g. grasslands (*adverse side-effect*) (Veldman et al. 2015).

6.4.3 Agriculture-based mitigation response options

According to IPCC WGIIIAR5, Chapter 11 (Smith et al. 2014b), the Agriculture, Forestry and Other Land Use (AFOLU) sector is responsible for just under a quarter (about 10–12 $GtCO_2eqyr^{-1}$) of anthropogenic greenhouse gases (GHG) emissions mainly from deforestation and agricultural emissions from livestock, soil and nutrient management. The mitigation potential in agriculture-based mitigation (AgbM) response options is derived from both an enhancement of removals of GHG, as well as reduction of emissions through management of land and livestock. Mitigation strategies that go through the reduction of emission sources refer, mainly, to the reduction of: a) CO_2 emissions into the atmosphere due to deforestation and conversion of pastures and grasslands (Houghton and Nassikas, 2017); b) N_2O emissions from agricultural soils, associated with the addition of N as mineral and organic fertiliser (Snyder et al. 2014)Snyder et al., 2009; Snyder et al. 2014) the return of agricultural residues to soils (Shan and Yan, 2013), and the excreta of grazing animals (Gerber et al., 2013); and c) CH_4 emissions from waterlogged soils as rice fields (Wang et al. 2017) and from ruminants by enteric fermentation (Gerber et al., 2013). The strategies associated with the improvement of removals of atmospheric CO_2 are related to the increase in photosynthesis by plants and the storage of fixed carbon, whether as woody tissue in afforestation, fresh organic C in root systems of grasses, and organic C stabilised in the soil (Smith et al. 2014; Paustian et al. 2016; Conant et al, 2017; Merante et al., 2017).

Mitigation: A variety of agricultural management practices and technologies are known to reduce soil GHG emissions and promote C sequestration, most of which also provide environmental co-benefits although many of them at an early stage of implementation and inaccurate quantification of emissions and reductions (Paustian et al. 2016). However, responses may be site specific and variable according to the situation. For example, the impact of no till farming and conservation agriculture on soil carbon stocks may be positive (de Moraes Sá et al. 2017), null or even negative ((Palm et al. 2014); (Powlson et al. 2014); (Cheesman et al. 2016); (Powlson et al. 2016); (VandenBygaart 2016)), as depending on the amount of crop residues returned to the soil. Except under determined circumstances, such as the case of some low organic carbon soils or when no till farming is combined with intensive rotations and complete fertilisation scheme, it is not so easy to increase soil organic carbon in the profile of agricultural soils, at less if only harvest crops are produced. This is so because only a minor proportion of photo-assimilates is derived to the root system in modern crops. The potential of cover crops to sequester carbon in the soil varies between 0.10 and 1 $Mgha^{-1}\ yr^{-1}$ of SOC relative to no-till without cover crops, as depending on cover crop species, soil type, and precipitation input. Livestock systems exhibit great climate change mitigation potential, through management options that sustainably intensify livestock production, promote carbon sequestration in rangelands and reduce emissions from manures, and through reductions in the demand for livestock products (Lemaire et al., 2014; Herrero et al. 2016a), or with autonomous transitions from extensive to more productive systems (Havlik et al. 2014). Climate smart agriculture (CSA) (see 5.4.6.6 Section in Chapter 5) aims -among other objectives- at enhancing the productivity and resilience of natural and agricultural ecosystem functions, showing clear co-benefits with mitigation of climate change (Lipper et al. 2014; Steenwerth et al. 2014). Its implementation requires increasing the adaptive capacity of farmers as well as the resilience and resource use efficiency in agricultural production systems (Lipper et al. 2014).

Adaptation: mitigation options aiming at increasing soil carbon stocks and diversifying crops and products, such as the case of CSA have clear co-benefits with adaptation, as they also promote resilience to climate change (Shirsath et al. 2017). Except for the case of nutrient management and precision agriculture, mitigation based on reducing GHG emissions has neutral impacts on adaptation.

Prevention of desertification: Desertification is a complex process and can result from both abiotic and biotic processes (see 3.1.3 section, Chapter 3), which can be prevented by means of technological, socio-economic and political measures (see 3.6 section, Chapter 3). Technological and SLM options preventing desertification generally based on the increase of land cover and soil carbon stores and the achievement of more resilient climate systems (e.g. better water availability and water productivity) have clear co-benefits with mitigation of climate change. However, there are possible adverse side-effects with regard to erosion control, since this generates conflicting effects on soil carbon: it is lost in eroded sites and gained in deposition sites (Kirkels et al. 2014). The control of invasive species can generate losses, when these species are woody rich in carbon (i.e. shrub encroachment) (D'Odorico et al. 2013; Chiti et al. 2017). Finally, the possibility of agriculture in recovered sites may increase CO_2 and N_2O emissions from agricultural soils (Smith et al. 2014b). Many of these effects are site specific and highly dependent of species and varieties used (crops, livestock), and management (e.g. pastoral farming, grazing systems).

Prevention of land degradation: Actions to address land degradation involve planned interventions based on land restoration and sustainable land management (SLM) (see section 4.9.1, Chapter 4). A study of the status of the world's soils shows that globally erosion is the main process of degradation, followed by nutrient imbalance (deficits and excesses), loss of carbon stocks and salinisation (FAO and ITPS 2015). There are clear co-benefits between the actions of AgbM and the prevention of degradation that occurs mainly due to improvements in soil quality, biodiversity and increases in carbon stocks. Climate extremes such as droughts or storms can lead to a decrease in regional ecosystem carbon stocks and therefore have the potential to negate an expected increase in terrestrial carbon uptake (Reichstein et al. 2013). So, there are clear co-benefits with management options promoting photosynthetic activity by plants and soil C sequestration. However, some adverse side effects may also appear. For example, higher crop productivity in recovered areas could lead to higher CO_2 and N_2O soil emissions. Reconstruction of eroded soils could affect sites in which carbon has been transported and deposited by erosion (Kirkels et al. 2014). As with desertification prevention, effects are site specific and highly dependent of species and varieties used, and management. Mineral fertilisers are used to avoid plant nutrient depletion in agricultural soils, increasing both direct and indirect N_2O emissions from N fertilisers. They may be reduced by a better timing and formulation of fertilisers (Snyder et al. 2009; Snyder et al. 2014; Shcherbak et al. 2014), the use of enhanced-efficiency N fertilisers (Halvorson et al. 2014), and precision agriculture (Yost et al. 2017).

Delivery of food security: Many interventions targeted either to increase food production, climate change adaptation or mitigation are intricately linked with each other leading to complex interaction (Vermeulen et al. 2012; section 5.6, Chapter 5). Smith et al. (2013) noted a number of co-benefits between demand-based mitigation measures and food security. For example, the increase of carbon stores in soils and resilience through CSA, management of rotations, the balance of nutrients and of grazing systems, among others also improve productivity (Wheeler and von Braun 2013; Lipper et al. 2014; (Campbell and Veteto 2015a)(Merante et al. 2017)). Possible adverse side-effects are caused when delivery of food security is achieved by increasing cultivated areas and crop yields, which would result in higher GHG emissions. Increases in food production achieved at the expense of agricultural expansion have clear adverse side effects with mitigation of climate change (i.e. C stock losses), biodiversity and other ecosystem services, with possible negative climate feedbacks (Smith et al. 2014b; Newbold et al. 2015; Salazar et al. 2015). There are several options aiming at increasing crop and livestock productivity, combining genetic improvement, sustainable intensification and closing yields gaps by a better soil, water and crop management. By the hand, a higher crop and livestock production inevitably results in higher GHG emissions from agricultural soils and livestock, although emissions by produced unit (e.g. grain, meat, milk, etc.) may be lower under sustainable intensification. Livestock systems may be a source rather than a fate of atmospheric carbon, if not prioritising mitigation over productivity, as when not targeting emissions from land use changes (Havlik et al. 2014).

Biodiversity and ecosystem services: Intensive agriculture has led to several drawbacks such as biodiversity loss, climate change, erosion, and pollution of air and water. Some ecosystem services are enhanced under conservation agriculture and SLM (e.g. erosion control and runoff reduction), but others (i.e. soil carbon sequestration) are not necessarily (Palm et al. 2014). Compared to the efficiency/substitution paradigm of intensive agriculture, biodiversity-based agriculture, such as the design of agricultural landscapes, exhibit clear co-benefits with biodiversity and ecosystem services. However, they are more knowledge intensive and require implementing a more systemic and holistic view of agricultural systems (Duru et al. 2015; Landis 2017). A recent study by Smith et al. (2018) shows that moving the global warming goal from 2°C to 1.5°C through agriculture-based carbon sequestration has a neutral to positive impact on biodiversity (number of species). Climate mitigation by bioenergy crops can generate conflicts with biodiversity, as long as the design of expanded biofuel production avoids areas of special biodiversity concerns or embeds new production areas within a sustainable matrix of natural and transformed ecosystems (Joly et al. 2015).

UN SDGs: Agriculture based mitigation options will likely contribute positively to SDGs 2, 6, 12, 13, 15, as described in Howden et al. (2007), Challinor et al. (2014) and Altieri and Nicholls (2017).

6.4.4 Other land management-based mitigation response options

A recent study (Griscom et al. 2017a) has highlighted the role of coastal, wetland and peatland restoration/management for providing carbon sequestration for climate mitigation action, and a section in the Carbon Dioxide Removal chapter of the UNEP Emissions Gap Report (UNEP 2017) also featured coastal, wetland and peatland restoration as an option. These ecosystems are defined as high carbon density, anaerobic ecosystems, including "inland organic soils and wetlands on mineral soils, coastal wetlands including mangrove forests, tidal marshes and seagrass meadows, and constructed wetlands for wastewater treatment" (IPCC 2014a). Peatlands and coastal wetlands store 44%-71% of the global terrestrial biological carbon pool (Zedler and Kercher 2005). Like other biological carbon pools, the carbon stocks in coastal, wetland and peat ecosystems are vulnerable to reversal (Parish et al. 2008), but these ecosystems also have significant carbon sequestration potential (Page and Hooijer 2016). Griscom et al. (2017) estimated that, by 2030, avoided coastal wetland impacts, avoided peat impacts and peat restoration could deliver 0.3, 0.7 and 0.8 $GtCO_2$ yr^{-1} of carbon sequestration, respectively.

Adaptation: The restoration of coastal, wetland and peat ecosystems will also likely deliver co-benefits for adaptation, for example *via* restored peatlands and wetlands providing increased resilience to climate change through improved water holding capacity, thereby helping to prevent downstream flooding under extreme rainfall events (Munang et al. 2013b). The storage and regulation of water flow in wetlands may also buffer against the effects of drought (Munang et al. 2014), thereby also conferring adaptation co-benefits. There are no obvious adaptation adverse side-effects from wetland management, except *via* the opportunity costs associated with taking cultivated peats / wetlands out of production, which could affect the ability of smallholders using this land to adapt to future climate change if their livelihoods are adversely impacted (Hadi et al., 2005).

Prevention of desertification: In terms of desertification, coastal, wetland and peat land restoration may have downstream impacts on water flow, thereby regulating water flow to arid areas downstream (Munang et al. 2014), but are unlikely to provide either direct co-benefits nor adverse side effects for desertification prevention.

Prevention of land degradation: Coastal, wetland and peat land restoration provide direct co-benefits to the prevention of land degradation. Large areas of global coastal wetlands and peatlands are degraded (Limpens et al. 2008), so restoration is an effective method to halt and reverse this form of land degradation, providing clear co-benefits. Identical management interventions would be used for both climate mitigation and for prevention / reversal of land degradation (Griscom et al. 2017a). There are no obvious adverse side-effects.

Delivery of food security: There are few synergies between coastal, wetland and peatland restoration and delivery of food security, since these lands are not used for food production (with the exception of fish and edible plants sources from wetlands, which is a minor contributor to food supply globally). Adverse side-effects may arise where the coastal wetlands or peatlands are used for food production. Large areas of tropical peatlands have been drained and cleared for food production (e.g. (Miettinen et al. 2012)) and their restoration could displace food production and damage local food supply in these areas. The same is true for cultivated northern peatlands (Grønlund et al., 2006). The restoration of coastal habitats (e.g. mangroves) could also complete with local fisheries and aquaculture, having local impacts on food supply and livelihoods (Bush et al. 2010).

Biodiversity and ecosystem services: Coastal, wetland and peat land restoration are likely to provide co-benefits for biodiversity, since these ecosystems often are comprised of specialised flora and fauna (Bonn et al. 2014; (Yu et al. 2017)), and are declining in global extent (Limpens et al., 2008). Protection of natural / semi-natural coastal wetlands and peatlands from future degradation would help to prevent biodiversity loss (Yu Minayeva et al., 2017). In terms of other ecosystem services (Yu Minayeva et al., 2017), coastal, wetland and peat land and peatland restoration for climate mitigation would likely provide co-benefits for water provision (Kivaisi, 2001), climate regulation (Joosten et al., 2016), hazard regulation (flood management; Erwin, 2009), soil and water quality (purification; Kivaisi, 2001), supporting services (primary production, soil formation, nutrient cycling, water cycling and ecological interactions; Kivaise, 2001), and in some places, cultural services (e.g. tourism to natural / wild areas, landscape aesthetics; (Daniel 2012)Daniel et al., 2012). Adverse side-effects might occur with food, fibre and energy production if cultivated peatlands are taken out of production.

UN SDGs: The restoration of coastal, wetland and peatland ecosystems will likely contribute positively to SDGs 1, 3, 6, 11, 12, 13, 14, and 15 (from benefits listed in Munang et al. 2013b and Yu Minayeva et al., 2017), and is unlikely to adversely affect any SDG, except for SDG 2 if land used for producing food is taken out of production without compensatory action elsewhere in the food system (Stehfest et al. 2009a; Bajželj et al. 2014a).

6.4.5 Bioenergy with CO_2 Capture and Storage (BECCS)

Bioenergy production mitigates climate change by delivering an energy service, therefore avoiding combustion of fossil energy. It is the most common renewable energy source used today in the world and the one with the largest future potential deployment (Chum et al. 2011; Creutzig et al. 2015; Slade et al. 2014). Bioenergy is produced from dedicated forest or agricultural systems and residues or municipal solid waste. It is widely-deployed as a climate change mitigation option in the energy and transport sector in many future scenarios (Chum et al. 2011; Clarke et al. 2014; Creutzig et al. 2015; Edelenbosch et al. 2016; Popp et al. 2014; Riahi et al. 2017b; Sims et al. 2014), especially those aiming at a stabilisation of global climate at 2°C or less (Edelenbosch et al. 2016; Popp et al. 2014, 2017b; van Vuuren et al. 2016; Van Vuuren et al. 2010; van Vuuren et al. 2011). Bioenergy with CO_2 Capture and Storage (BECCS) entails the use of bioenergy technologies (e.g., bioelectricity or bioliquids) in combination with CO_2 capture and storage such that most or all of the CO_2 produced during conversion or combustion is captured and stored, rather than emitted to the atmosphere. Additionally, since growing bioenergy removes CO_2 from the atmosphere, BECCS is considered a Carbon Dioxide Removal (CDR) or Negative Emissions Technology (NET). BECCS simultaneously provides energy and reduces atmospheric CO_2 concentrations. Note that while a BECCS demonstration project exists, it has yet to be deployed at scale (Kemper 2015a); see Section 6.9 and Chapter 7 for a further discussion of barriers to BECCS deployment.

BECCS is widely deployed in scenarios (see Cross-Chapter Box) to achieve the large-scale negative emissions needed to stabilise global temperature rise (Anderson and Peters 2016; Benson 2014; Riahi et al. 2017b; Rogelj et al. 2011; van Vuuren et al. 2016). The IPCC AR5 found that excluding CDR techniques, like BECCS, could result in infeasibility of low stabilisation targets, like 450ppm CO_2 (Clarke et al. 2014). BECCS can reduce mitigation costs (Muratori et al. 2016; Kemper 2015a), the cost of producing energy, and affect the timing of mitigation. Scenarios without BECCS typically require earlier mitigation action (Van Vuuren et al. 2017).

The use of bioenergy and BECCS in scenarios varies significantly across scenarios, depending on the degree of climate mitigation, the inclusion of other mitigation options, the demand for land for other uses, etc. (see Section 6.3, Chapter 2; Cross-Chapter Box). Scenarios using BECCS show its sequestration as ranging from 2 and 10 $GtCO_2yr^{-1}$ in 2050 (Fuss et al. 2014) and as much as 18 $GtCO_2$ yr^{-1} in 2100 (IPCC SR1.5; Chapter 2; Cross-Chapter Box). Achieving such sequestration requires significant production and use of bioenergy, with estimated energy use of up to 400 EJyr-1 (IPCC SR1.5; Chapter 2; Cross-Chapter Box). Production of bioenergy on that scale has the potential for increased competition for land and interlinkages with food security and ecosystem services (Slade et al. 2014). The estimated cropland area for bioenergy and BECCS in the IAM-based scenarios ranges from 0 to 1500 MHa (Popp et al. 2017b), Section 6.3).

Many aspects directly linked to bioenergy and BECCS can either increase or dampen their climate change mitigation benefits. Examples include a weakening of the global land and ocean sinks as a response to negative emissions, timing of carbon dynamics, direct and indirect land use changes to grow bioenergy crops, including the associated biophysical effects, and life-cycle emissions of GHGs to harvest, transport, and process biomass resources. These issues have case-specific implications and spatial variability, and are discussed in more detail in Chapter 2.

Adaptation: Bioenergy and BECCS compete for land and water with other uses, including adaptation. Increased use of bioenergy and BECCS can result in large expansion in cropland area (Popp et al. 2017b; Calvin et al. 2014; Smith et al. 2016b) and in increased irrigation water use and increased water scarcity (Chaturvedi et al. 2013a; Hejazi et al. 2014, 2015a; Popp et al. 2011; Smith et al. 2016b; Fuss et al. 2018) (*adverse side-effect*). Intensive land use practices may trade short-term increases in forest and agricultural production systems for long-term losses in ecosystem services, including soil health (*adverse side-effect*) (Foley et al. 2011). Under certain circumstances (low inputs of fossil fuels and chemicals, limited irrigation, heat/drought tolerant species, using marginal land), biofuel programs can be beneficial to future adaptation of ecosystems (*co-benefit*) (Dasgupta et al. 2014; Noble et al. 2014). In summary, bioenergy and BECCS can result in ***small co-benefits*** for adaptation under certain circumstances (limited evidence, high agreement), but in general results in ***large adverse side-effects*** for adaptation (medium confidence).

Prevention of desertification: Desertification is classified as a risk in many regions that are key for today and future biofuel production (Albanito et al. 2016; Staples et al. 2017). Intensive agricultural practices aiming to achieve high crop yields, as it is the case for some bioenergy systems, may have significant effects on soil health, including depletion of soil organic matter (SOM) and soil carbon stocks (*adverse side-effect*). The decrease in SOM reduces the resistance of soils to erosion agents (e.g. wind, water), lowers the water holding capacity of soils and affects nutrient content, ultimately contributing to a desertification process (*adverse side-effect*) (FAO 2011; Lal 2014). On the other hand, bioenergy crops like perennial grasses increase soil carbon (Don et al. 2012; Robertson et al. 2017), thereby helping to preserve soil quality and prevent desertification processes without competing with food production (*co-benefit*). For example, growing a perennial grass tolerant to salinity on saline and saline-prone agricultural land can provide soil restoration and simultaneously achieve yields of 22 t dry matter ha^{-1} yr^{-1} (*co-benefit*) (Sánchez et al. 2017). In summary, bioenergy and BECCS has the potential to provide ***small co-benefits*** or ***small adverse side-effects*** for desertification.

Prevention of land degradation: Large-scale production of bioenergy can require significant amounts of land (Smith et al. 2016b; Popp et al. 2017b), increasing pressures for land conversion and land degradation (*adverse side-effect*). However, some studies suggests that BECCS could also help restore degraded lands (Kemper 2015b; Fuss et al. 2018). If bioenergy crops are cultivated on degraded land or land that was previously occupied with a culture with low carbon stocks, the increment of above-ground and soil carbon content due to bioenergy production can be significant (*co-benefit)* (Robertson et al. 2017). Planting bioenergy species on marginal lands or pasture that are prone to degradation and are not used for row crops or livestock represent possible *synergies* of bioenergy and land degradation (Mello et al. 2014). Bioenergy potential on degraded lands ranges from 25 to 190 EJ yr^{-1}, depending on the type of crop and whether certain lands are excluded from use (Nijsen et al. 2012; Schueler et al. 2016). For comparison, the primary energy consumption by 2100 of biomass and BECCS under the different SSPs ranges from 50 EJ yr^{-1} (baseline) to approximately 500 EJ yr^{-1} (SSP5, RCP2.6) (Bauer et al. 2017).

Estimates of marginal lands available for cellulosic feedstock production at global and regional levels vary depending on methodologies and available data, with estimates ranging from 385-1100 Mha (Campbell et al. 2008; Cai et al. 2011). Factors like potential yields, economic costs and life-cycle GHG emissions of feedstock cultivation and transport, and landowner acceptance, will determine how much of this land can be practically available in the future for cellulosic feedstock production (Robertson et al. 2017). For comparison, the land required to meet the demand for cellulosic bioenergy crops in the different SSPs ranges from 200–1500 Mha of land (Popp et al. 2017a). In summary, bioenergy and BECCS has the potential to provide ***small co-benefits*** or ***small adverse side-effects*** or ***large adverse side-effects*** for land degradation.

Delivery of food security: Bioenergy and BECCS competes for land with food and natural systems. Bioenergy demands can result in increased food prices (Muratori et al. 2016; Favero and Mendelsohn 2017; Calvin et al. 2014; Obersteiner et al. 2016; Popp et al. 2017a; van Vuuren et al. 2015; Popp et al. 2011; Fuss et al. 2018), as shown in

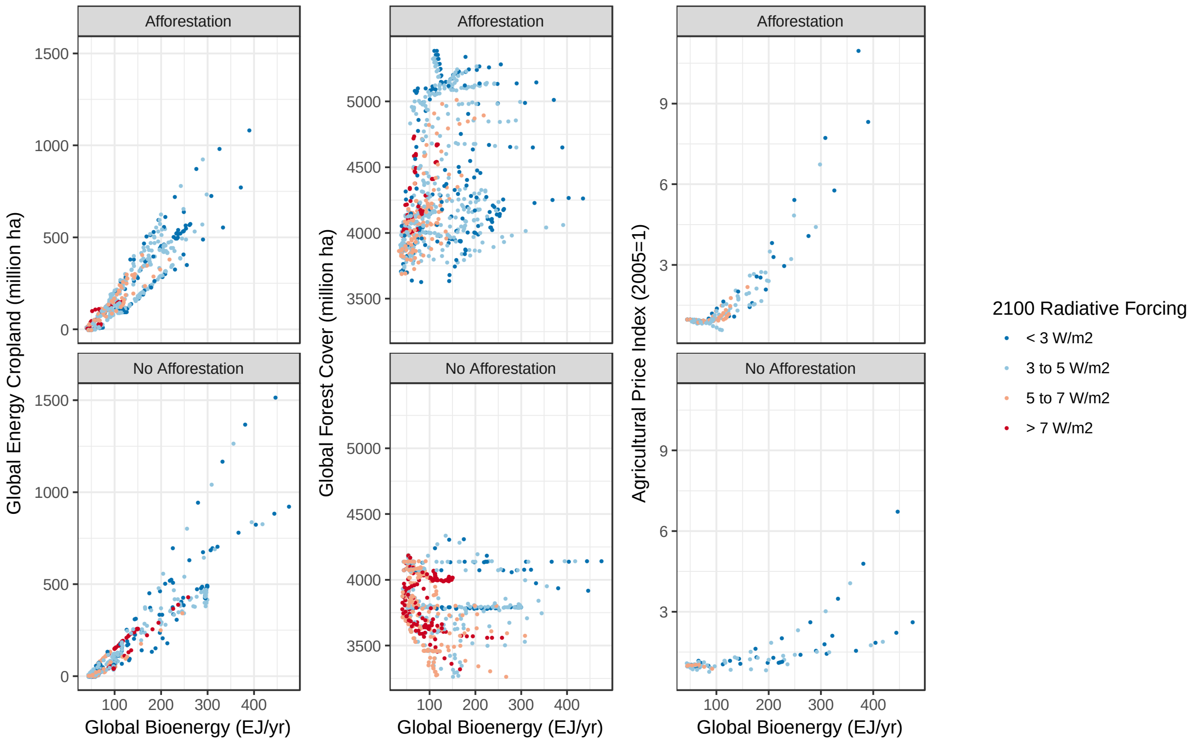

Figure 6.6 Correlation between bioenergy and BECCS consumption and cropland area, forest area, and food price. Data is derived from the IPCC AR5 database (Clarke et al. 2014) and the SSP database (Riahi et al. 2017b). Only scenarios that provide land cover information are included. Scenarios are classified by their 2100 radiative forcing (colors) and whether afforestation is included (top vs. bottom row). (Additional scenarios will be added for SOD) (right panel). Popp et al. (2011) finds that combining bioenergy deployment with forest conservation programs can result in even further increases in food prices. (Frank et al. 2017a) Frank et al. (2017a) find that the increase in price depends strongly on the amount of land-based mitigation, including bioenergy; in extreme scenarios, they find price increases large enough to result in a decline in global food caloric intake by 100–300 kcal per person daily in 2050, resulting in undernourishment of 80–300 million people. Less stringent temperature stabilisation targets and carbon removal from other economic sectors reduce these effects.

Some research suggests that the type of bioenergy feedstock influences the competition for land. Fuss et al. (2016) suggest that using woody or other cellulosic crops may not compete for land with food. Favero and Mendelsohn (2017), however, finds that much of the new forest area needed for woody bioenergy would come from farmland. Other studies show that using "additional" biomass, e.g., residues, can limit competition for land with food (Kemper 2015a); however, even with these resources included, it still may not be able to meet all domestic energy needs without impacts on food systems (Welfle et al. 2014).

In summary, bioenergy and BECCS has the potential to provide ***large adverse side-effects*** for food security (high confidence); however, the magnitude of the side-effects depends on the scale of bioenergy deployment (medium confidence) and the type of feedstock (*limited evidence, low agreement*).

Biodiversity and other ecosystem services: Increased use of BECCS can lead to deforestation or conversion of other natural lands (Calvin et al. 2014; van Vuuren et al. 2015; Favero and Mendelsohn 2017); however, the effect of this conversions on biodiversity is uncertain. Declines in forest area can lead to declines in biodiversity (*adverse side*-effect) (Pereira et al. 2010b). Some studies suggest that inappropriate management will result in negative effects on biodiversity (Kemper 2015a). Other studies indicate that increasing wood outtakes from forests can threaten biodiversity and other ecosystem services (*adverse side-effect*) (Bouget et al. 2012; Kraxner et al. 2013; Michelsen 2008). Others studies suggest that positive impacts are possible if bioenergy crops are planted on degraded areas (Kemper 2015a), as planting perennial grasses can increase biodiversity and ecosystem functions (*co-benefit*) (Meehan et al. 2012; Parish et al. 2012; Robertson et al. 2017; Werling et al. 2014). Additionally, perennial species like switchgrass and other prairie plantings harboured significantly greater plant, methanotrophic bacteria, arthropod, pest suppression, pollination, and bird diversity than maize (*co-benefit*) (Dauber et al. 2010; Meehan et al. 2010; Robertson et al. 2017; Werling et al. 2011). These benefits can also extend to nearby croplands, where cereal yields, pollination, and pest suppression are found to increase following increasing abundance of perennial habitats in the surrounding landscape (*co-benefit*) (Bennett and Isaacs 2014; Liere et al. 2015; Werling et al. 2011). Finally, bioenergy policy that supports coordinated land use can diversify agricultural landscapes with *co-benefits* for ecosystem services (Werling et al. 2014). In summary, while planting perennial grasses for bioenergy can have ***co-benefits*** in terms of increased biodiversity and ecosystem function, the conversion of natural land to bioenergy can also have ***adverse side-effects*** in terms of reduced biodiversity.

Large-scale biofuel production can also impact the water cycle. Bioenergy production can result in increased irrigation water use and increased water scarcity (Chaturvedi et al. 2013a; Hejazi et al. 2014, 2015a; Popp et al. 2011; Smith et al. 2016b) (*adverse side-effect*). Additionally, converting large portions of landscapes to bioenergy production could influence evapotranspiration, precipitation, groundwater recharge and surface water levels (Robertson et al. 2017; McIsaac et al. 2010; Zhuang et al. 2013; Harding et al. 2016; Wang et al. 2017), although the direction depends on what bioenergy replaces (*co-benefit* or *adverse side-effect*) (see Chapter 2).

In terms of nutrient cycling, perennial grasses require lower nitrogen-fertiliser inputs than annual crops, and their nitrogen demands are at a rate that is similar to the harvest nitrogen removal (Davis et al. 2015; Ruan et al. 2016). Nitrate leaching and soil N_2O emissions from established perennial vegetation are also substantially lower than from annual crops (*co-benefit*) (Robertson et al. 2017). The more efficient N use by perennial crops stems from plant growth with associated N uptake in early spring and late fall, when inorganic N is otherwise subject to leaching, and without annual tillage, perennial crops accumulate N in soil organic matter. Wintertime plants can also trap snow in northern regions, protecting soil from freeze-thaw cycles that liberate inorganic N and stimulate N_2O production (Qin et al. 2012; Ruan and Robertson 2017). However, the propensity to retain N and avoid polluting the atmosphere, groundwater, and surface waters with nutrient leaching can be undermined by (even moderately) excessive fertiliser use (Robertson et al. 2017).

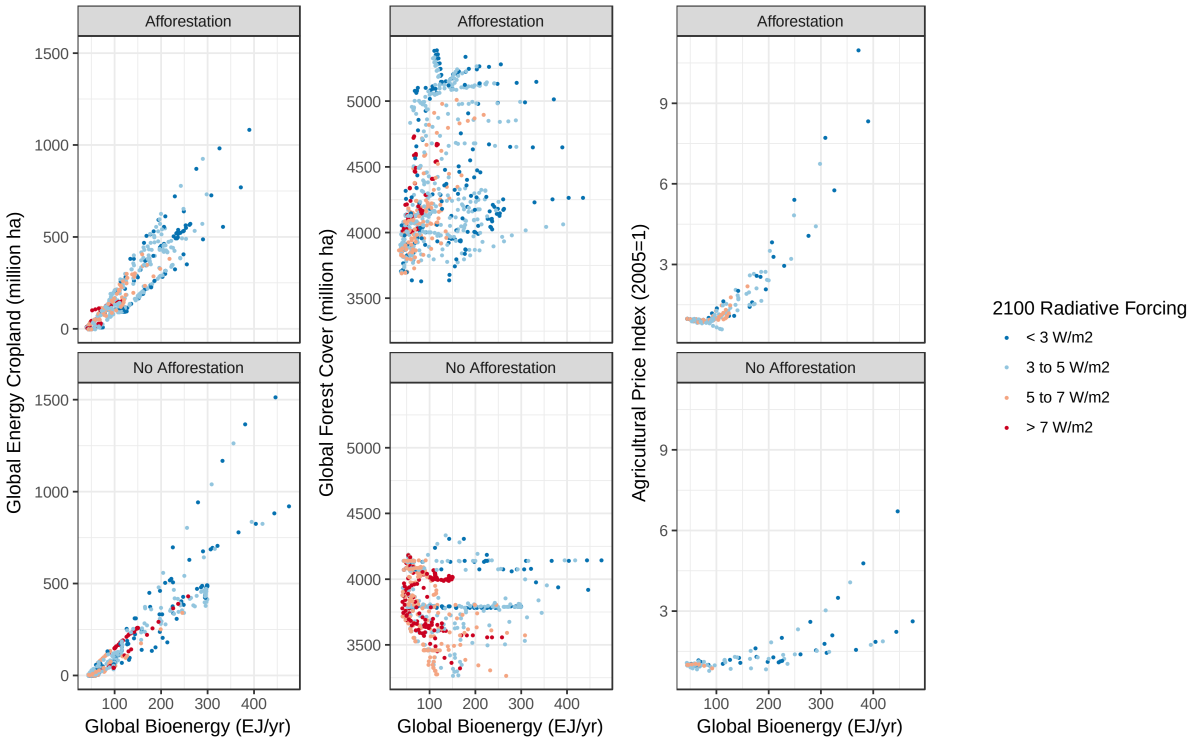

Figure 6.6 Correlation between bioenergy and BECCS consumption and cropland area, forest area, and food price. Data is derived from the IPCC AR5 database (Clarke et al. 2014) and the SSP database (Riahi et al. 2017b). Only scenarios that provide land cover information are included. Scenarios are classified by their 2100 radiative forcing (colors) and whether afforestation is included (top vs. bottom row). (Additional scenarios will be added for SOD)

6.4.6 Other mitigation response options

Enhanced mineral weathering (Schuiling and Krijgsman 2006; Kelemen and Matter 2008; Smith et al. 2016a; Taylor et al. 2016) has been proposed as a negative emission technology whereby minerals that naturally absorb CO_2 are ground to increase the surface area, and then absorb CO_2 either in a controlled environment, or through spreading in the surface of the land (Schuiling and Krijgsman 2006; Kelemen and Matter 2008; Taylor et al. 2016). Depending on the mineral used, spreading on the land may increase the soil pH, which in acidified soils, can also increase production, thereby providing a potential co-benefit for food security (Taylor et al. 2016) and possibly adaptation, though optimal rates of application differ for CO_2 removal and soil improvement (Smith et al. 2016a,c).There are unlikely to be significant impacts of enhanced weathering on prevention of desertification, land degradation. Improving soil fertility is an ecosystem service co-benefit and would help to contribute to SDG 2.

Emission reductions from energy substitution (e.g. bioenergy) are not accounted for in the AFOLU sector. Impacts of widespread production of bioenergy is discussed in detail in section 6.4.5 on BECCS.

Longer term carbon storage through material substitution (e.g. wood replacing concrete and steel in building construction, or use of cereal straw bales for building construction), and carbon storage in harvested wood products provide some mitigation co-benefits associated with removing CO_2 from the atmosphere, at least temporarily, though emission reduction is not accounted for in the AFOLU sector (Smith et al. 2014b). These measures are unlikely to have any appreciable impact upon adaptation, prevention of desertification or land degradation, or delivery of food security. Neither will they impact appreciably on most ecosystem services. They are likely to contribute positively to SDGs 9 (industry, innovation and infrastructure), 11 (sustainable cities and communities), and 12 (responsible consumption and production).

6.5 Land-based adaptation response options and their implications

6.5.1 Land-based adaptation response options considered

Land-based adaptation options considered in this section can be classified into response options relating to forestry, agriculture, ecosystem based adaptation and other options. *Forestry* options mainly include forest landscape restoration (including afforestation/reforestation and "green walls"). *Agriculture* options include cropland management, livestock management, climate-smart agriculture, grazing land management, diversification (of crops, livestock and aquaculture systems) and improved irrigation. *Ecosystem based adaptation* includes ecological restoration, ecological engineering (e.g. biological waste water treatment, landscape corridors, ecosystem based disaster risk reduction, planting mangroves and restoration of marshlands), restoring and enhancing natural and green infrastructure, integrated coastal zone management, integrated water resources management and area-based conservation approaches. *Other* options (not based on sustainable land management) include knowledge transfer, disaster risk reduction, indigenous knowledge, community-based management, early warning systems and insurance schemes. The co-benefits and adverse side-effects associated with these response options for climate adaptation, with interventions to tackle climate mitigation, desertification, land degradation and food security are assessed in the sections below.

6.5.2 Forestry-based adaptation response options

Many forest areas of the world are at risk because of a lack of adaptation to climate change. The linkages between forests and adaptation are twofold (Locatelli 2011). On the one hand, forests may contribute to adaptation by providing local ecosystem services that reduce the vulnerability of local communities to climate change (forestry-based adaptation responses). On the other hand, as climate change will affect forests, management practices need to be designed to decrease the vulnerability of forests to a changing climate (Jandl et al. 2015). These two topics are to be pursued together: for ensuring that forests provide relevant ecosystem services for the society (including adaptation to climate change), healthy and climate-resilient forests must first be achieved through sustainable management (Kongsager et al. 2016). Forest landscape restoration is one of the key options for adaptation (Stanturf et al. 2015). Adaptation options based on forest restoration include (Ramón Vallejo et al. 2012): (i) plant species selection, (ii) improved nursery techniques to improve seedlings quality, and (iii) improved planting techniques.

Mitigation: Forest-based adaptation projects can directly benefit climate change mitigation, the prevention of degradation and desertification, and other ecosystem services. For example, mangrove afforestation is an adaptation measure that can significantly contribute to reduce the loss of life and property against tropical cyclones in coastal areas, improve land stabilisation and soil quality, prevent saline intrusion, protect against erosion and at the same time store carbon (Ahammad et al. 2013).

Climate-suitable planting increases species and functional diversity (*co-benefits* for ecosystem services provision), especially under high-emissions scenarios conditions (e.g. Duveneck and Scheller 2015). Nitrogen fertilisation in the nursery may accelerate the growth (*co-benefits* for mitigation), increase the root/shoot ratio, and reduce tree mortality (e.g. Villar-Salvador et al. 2004). The selection of plants less vulnerable to climatic stresses for planting facilitates survival and growth (e.g. Andivia et al. 2018). On the other hand, if implementing prescribed burning, there is no significant reduction or influence on growth resistance, resilience and recovery (i.e. vulnerability to drought) (e.g. Bottero et al. 2017).

Tree planting in urban areas has been regarded as a beneficial adaptation measure with potential *co-benefits*, e.g. by balancing water flows and providing thermal comfort (Demuzere et al. 2014), and even reducing crime and raising property values (Troy et al. 2012), while at the same time providing mitigation benefits through plant CO_2 sequestration.

Forest-based adaptation options may also produce *adverse side-effects*. For example, adaptation measures in forestry can decrease carbon stocks (e.g. shortening plantation rotation) and the short-term sink (e.g. introduction of tree species that are more tolerant to future climate conditions), or increase the vulnerability of carbon stocks in the long term (e.g. suppressing fire) (Locatelli et al. 2015b; Jandl et al. 2015).

Some REDD+ projects have shown potential to increase local access rights to forests leading to better protection from outside deforestation pressures, and simultaneously increasing participating communities' resilience to climate change (Cerbu et al. 2011; Kashwan 2015). However, review of the literature reveals that very few of the decisions taken at by UNFCCC COPs have explicitly linked forestry and adaptation sectors, for example, in REDD+ projects (McElwee et al. 2017). There are considerable problems in integrating adaptation and mitigation together in forestry work at both national and local levels. For example, time scales may not sync, because mitigation projects often focus on more immediate carbon gains while adaptation benefits may be more longer term. Biophysical impacts on trees like fire risk also requires different responses than impacts on households like loss of income from weather events (Guariguata 2009). While many policymakers have advocated combining mitigation and adaptation in forestry projects, in reality there are few examples of where this has been done successfully; for example, few REDD+ have done both goals well (McElwee et al. 2017; Somorin et al. 2012, 2016; Thuy et al. 2014).

Other interlinkages: Many of the interactions between forestry-based adaptation measures with desertification, land degradation, food security, and ecosystem services are similar to those of forestry-based mitigation measured described in the previous section. Only the complementary relevant aspects mainly pertaining to adaptation measures are reiterated here.

Longer term adaptation could reduce the severity of losses in agricultural and forestry but could include displacement production from southern Europe to the North (Dasgupta et al. 2014). Choosing provenances that are well adapted to current climates but pre-adapted to future climates is difficult because of uncertainties in climate projections at the time scale of a plantation forest rotation (Broadmeadow et al. 2005; Settele et al. 2014). According to (Settele et al. 2014), risk spreading by promoting mixed stands, containing multiple species or provenances, combined with natural regeneration (Kramer et al. 2010), has been advocated as an adaptation strategy for temperate forests (Bolte et al. 2010) and tropical forests (Erskine et al. 2006; Petit and Montagnini 2006). However, incomplete knowledge of the ecology of tropical tree species and little experience in managing mixed tropical tree plantations remains a problem (Hall et al. 2011). Especially at the equator-ward limits of cold adapted species, such as Norway spruce (*Picea abies* Karst.) in Europe, climate change will very likely lead to a shift in the main tree species used for forest plantations (Bolte et al. 2010; Iverson et al. 2008). In addition, indirect impacts can result if an adaptation project prevents activity displacement and induced deforestation, for example if an agricultural adaptation project sustain crop productivity and reduce clearing forest through agricultural expansion (Locatelli 2011).

6.5.3 Agriculture-based adaptation response options

Adaptation to climate change based on agriculture (AgbA) is any practice or action taken at different levels of the agricultural sector (farmers, agribusiness and decision makers), with the aim of managing or minimising the potential risks of the climate for the coming decades. From IPCC WGII AR5, Chapter 14 (Noble et al. 2014), actions for AgbA may be classified as: (i) structural physical, (ii) technological, (iii) social, and (iv) institutional. Structural physical actions require the use of engineering and changes in the physical environment, in response to extreme weather events, such as droughts, floods, and heavy storms. Technological responses are related to changes in management unit decisions in crop and livestock systems, aiming at a better management of crops, livestock and grazing, new varieties and types of crops and animals, genetic techniques, and the use of traditional knowledge (Howden et al., 2007; Challinor et al. 2014). Social and institutional actions can be related to changes in the decision environment (Howden et al., 2007), and are based on information (risk and vulnerability maps, early warning and response systems, systematic monitoring and use of remote sensors) and human behavior and attitudes (adoption of soil and water conservation, changes in practices of livestock, crops and their cultivation practices, patterns and sowing dates).

Mitigation: encompassing the objectives pursued by mitigation and adaptation to climate change require a joint analysis of science and policy (Howden et al. 2007), because there are not always obvious co-benefits such as those raised –for instance- by climate smart agriculture (CSA). This promotes coordinated actions towards climate-resilient pathways, prioritising interventions that can enhance productivity and incomes, help farmers adapt to current risk, and decrease greenhouse gas emissions in current and future (Lipper et al. 2014; Shirsath et al. 2017). On the contrary, embedded emissions may appear when building improved irrigation systems or increased GHG emissions resulting from cropland and grazing land management. Policies to promote the use of biofuels by governments usually pursue the objective of reducing the use of fossil fuels. However, they have significant adverse effects when they promote changes in land use and GHG emissions in other sectors and threats food security (Howden et al. 2007; Miyake et al. 2014).

Prevention of desertification: Adaptive actions promoting famers resilience to climate change, such as cropland and grazing land management or climate smart agriculture have clear co-benefits with the prevention of desertification (Lal 2014; Lipper et al. 2014). For example, technological adaptation options such as the adoption of new varieties, or changing sowing dates, or structural/physical options such as implementing wind or sand dune erosion control or wind storms. Conversion to restorative land use and adoption of conservation-effective practices can reduce the risks of soil erosion, improve soil and water quality, increase soil and ecosystem carbon budget, and adapt to and mitigate the abrupt climate change (Lal 2014). Other practices are highly dependent on technology and management decisions. For instance, some adverse effects may appear when lands vulnerable to erosion are put into cultivation by public policy decisions, or the overuse of water for irrigation, or even worse, soil salinisation due to the use of poor water quality for irrigation (Elliott et al. 2014; Zhou and Turvey, 2014).

Prevention of land degradation: Land degradation adversely affects the provision of ecosystem services and food security (Stringer, 2017). A meta-analysis of 1237 observations in China revealed that terracing –a structural/physical adaptation option- significantly and positively affected erosion control, mainly when combined with land use changes to forests and trees (Chen et al. 2017). As with the prevention of desertification, those actions of AgbA related to changes in cultivation lands or displacement of productions have the inherent risk of invading vulnerable lands, either because of their aridity or of their less fertile soils. The replacement of forests and pastures by annual crops was an autonomous adaptation option to climate change (rainfall increases) by farmers in the South American Chaco during the last 15 years (Viglizzo et al., 2009; Fehlenberg et al. 2017). However, this also caused adverse impacts such as generalised groundwater rises, floods and soil salinisation (Marchesini et al. 2017). The expansion of irrigation with low cost of irrigation water led to water exhaustion in China (Zhou and Turvey, 2014), or reversion to rainfed crop management in Australia (Elliott et al. 2014).

Delivery of food security: Actions of AgbA, such the combination of agriculture conservation practices with integrated or more diverse productions based on agroecology, that seek to increase the resilience of agricultural systems show clear co-benefits with delivery of food security (Bullock et al. 2017). However, all this would not be enough because the whole food system needs to be adjusted to climate change, with strong attention also to trade, stocks, and to nutrition and social policy options (Wheeler and von Braun 2013; Lipper et al. 2014; Campbell et al., 2016). Also effective are crop-level adaptations that increase simulated yields by an average of 7%-15%, with adaptations more effective for wheat and rice than maize (Challinor et al. 2014). There may be adverse effects between AgbA and the delivery of food security. For example, when adaptation involves the adoption of crop varieties or livestock types more resilient to climate change and rustic, but at the same time less productive. A given problem may be solved, but another unwanted problem can be generated, as happens in unfertile soils of Africa where fertilisation makes possible the inclusion of other crops with greater water demand and more prone to suffer water stress (Lobell 2014). This author warns about not falling into the error of "adaptation illusions", for example when the differential impact of adaptation measures at increasing levels of climatic stresses (thermic and hydric) is not taken into account, or the use of cultivation models created for other climatic conditions than those under studied climate change, or not considering the expected changes in crop management and the interactions between genetic progress and management systems.

Biodiversity and ecosystem services: The conservation or improvement of biodiversity and ecosystem services (ES) must be part of all AgbA actions, assuming the implicit existence of several co-benefits. For example: (i) when erosion control practices are implemented and carbon stocks and climate resilience increase in degraded areas (Munang et al. 2013a); (ii) when conservation agriculture (e.g. cover crops, intercrops, reduced tillage) improves the climatic resilience of the system (Palm et al. 2014; Poeplau and Don 2015; Merante et al. 2017); (iii) when rescuing traditional management systems combined with agroecologically-based strategies and plant and crop diversification in peasant agriculture (Altieri and Nicholls 2017); or (iv) when including trees in agricultural systems (i.e. agroforestry) (Mbow et al. 2014; Sain et al. 2017). Despite, its positive impact on diversification and biodiversity, agroforestry can also have some adverse side-effects, such environmental impacts (excessive N_2O emissions, and disruption of regional GHG balances) with leguminous agroforestry (Rosenstock et al. 2014). If AgbA actions like the displacement of cropping areas imply a higher fragmentation in the landscape, this results in adverse side-effects like the loss of biological corridors (Rybicki and Hanski, 2013; Hamilton et al. 2015; Tapia-Armijos et al. 2015).

UN SDGs: Agriculture based adaptation options will likely contribute positively to SDGs 1, 2, 12, 13, 15, (as described in Howden et al., 2007; Challinor et al. 2014; Altieri and Nicholls 2017) and is likely to cause adverse effects on SDG6, when there are competitive uses of freshwater and irrigation (Elliott et al. 2014), and SDG15, when new areas are put into cultivation and threat other ecosystem services (e.g., Viglizzo et al. 2009).

6.5.4 Ecosystem-based adaptation

Ecosystem-based adaptation (EbA) entails the use of biodiversity and ecosystem services (BES) as part of an overall adaptation strategy to help people to adapt to the adverse effects of climate change (CBD, 2009). EbA is intended to provide resilience to climate change and simultaneously to reduce poverty, protect or restore biodiversity and ecosystem services, and remove atmospheric greenhouse gases (mitigation), so is designed around its co-benefits (Scarano 2017). EBA is often promoted as a win-win-win for mitigation, adaptation, and development of livelihoods and poverty alleviation (Munang et al. 2013a, 2014). Some examples of EbA are summarised by (Scarano 2017) and include “sustainable management, conservation and restoration of ecosystems, as part of an overall adaptation strategy that takes into account the multiple social, economic and cultural co-benefits for local communities” (CBD, 2009), “ecosystem restoration to enhance critical ecosystem services (e.g. water flow or food and fisheries provision), and protecting or restoring natural infrastructure (e.g., barrier beaches, mangroves, coral reefs, and forests) to buffer human communities from natural hazards, erosion and flooding” (Munang et al. 2013a; 2013b), “sustainable water management to provide water resilience, disaster-risk reduction through the restoration of coastal habitats, and sustainable agricultural management to enhance livelihoods and increase resilience” (Shaw et al., 2014), “ecological restoration, payment for ecosystem services, creation and effective management of protected areas, and community management” (Magrin et al., 2014), “forest products from agro-biodiversity to provide safety nets to local communities when climate variability causes crop failures”, and “coral reefs to provide protection against erosion and wave damage” (Ojea 2015). EbA’s strengths are that it is often more flexible and cost-effective than other approaches to adaptation like infrastructure development, and more reversible (Jones et al. 2012; Ojea 2015). However, there have been few assessments of the degree to which EBA approaches are integrated into either national or subnational projects and policies, and the limited country experience to date shows that many challenges remain in operationalising EBA (Vignola et al. 2009; Chong 2014), namely due to policy barriers and scalability issues (Ojea 2015; Scarano 2017).

Mitigation: Given that many of the interventions proposed to deliver EbA overlap considerably with those proposed for climate mitigation, particularly those that involve planting trees (see section 6.4.2), sustainable agriculture (see section 6.4.3) and coastal, wetland and peatland restoration/management (see section 6.4.4), the co-benefits between adaptation and mitigation from such interventions are clear. In addition to providing adaptation benefits, many also create carbon sinks (Griscom et al. 2017b), and some might also reduce emissions of non-CO_2 GHGs (e.g. sustainable agricultural management; Lipper et al. 2014), while others (particularly wetland restoration) might temporarily increase methane emissions (Mitsch et al., 2012). In terms of adverse side-effects, large scale afforestation for EbA might increase albedo, particularly at high latitudes (Betts, 2007), and could also affect hydrological cycles with potential climate impacts (Zhao and Jackson 2014).

Prevention of desertification: EbA might also provide co-benefits for desertification control. The prevention of soil erosion is part of EbA (Munang et al. 2013a; 2013b), and decreased erosion rates make soils and vulnerable arid ecosystems less vulnerable to desertification (D’Odorico et al., 2013). Ecological restoration and use of natural / green infrastructure, might also provide vegetation at the margins of areas under threat of desertification, which might slow or halt the process (D’Odorico et al., 2013). Improved cropland management (reduced tillage intensity) and grazing land management (reduction of overgrazing), and improved irrigation provision of water to arid areas) might also help to prevent desertification (D’Odorico et al., 2013). There are no obvious adverse side-effects between EbA and prevention of desertification.

Prevention of land degradation: All of the above mentioned co-benefits between EbA and prevention of desertification are also true for land degradation, with ecological restoration of forest, grasslands, coastal ecosystems, wetland and peatlands for EbA clearly also providing actions to halt and reverse land degradation (Griscom et al. 2017b).

Delivery of food security: There are few synergies between the restoration of many natural ecosystems and delivery of food security, since these lands are not used predominantly for food production (with the exception of fish and edible plant sources from wetlands, wild foods from forests, and fish and seafood products from coastal ecosystems). However, EbA can also be also take the form of sustainable agricultural management (Shaw et al., 2014), and when applied in this way can also improve food supply, thereby contributing to improved food security (Shaw et al., 2014). Diverse farm systems based on mimicking or restoring natural ecosystems (i.e. as in some agroforestry systems) have clear benefits for increases in food security (Vignola et al. 2015). Adverse side-effects may arise when ecosystems are restored on land currently used for food production, and/or some reduced yields in EbA-based agricultural systems given trade-offs (Vignola et al. 2015). Large areas of tropical peatlands have been drained and cleared for food production (Page et al. 2011) and their restoration could displace food production and damage local food supply in these areas. The same is true for cultivated northern peatlands (Grønlund et al., 2006). The restoration of coastal habitats (e.g. mangroves) could also complete with local fisheries and aquaculture, having local impacts on food supply and livelihoods (Bush et al., 2010).

6.5.5 Other interventions

A number of adaptation interventions rely on knowledge, socio-economic or cultural strategies and approaches, and can be used alone or combined with land and ecosystem-based adaptation actions outlined in above sections (Smit and Wandel 2006; Stults and Woodruff 2017). Such actions are often identified as "soft" adaptation measures, as they involve information, capacity or policy, in contrast to "hard" options, which require technology and higher levels of investment, such as infrastructure (Sovacool 2011). Adaptation options discussed in this section include key categories of:

- early warning and forecasting systems and disaster risk reduction programs;
- community-based adaptation and programs to increase social capital and adaptive capacity;
- implementation of indigenous knowledge, social learning, and knowledge transfer for adaptation;
- institutional capacity building at multiple scales

(*Potential figure: summary of 'soft' adaptation options related to land use compiled from Nationally Determined Contributions (NDC) to the Paris Agreement or National Adaptation Programs of Action (NAPAs)*)

Early warning and forecasting systems and disaster risk reduction programs: Providing warnings of impending climate risks (both short and long term) has been a priority for many national governments, expanding from being backward-looking or past-centred to being forward-looking by preparing for expected future climate hazards. Early warning systems (EWS) to enable disaster risk reduction (DRR) can involve mapping and monitoring, forecasting, education and communications systems (Bouwer et al. 2014; Cools et al. 2016). The literature on EWS has stressed that they must be 'end-to-end,' both reaching communities at risk and supporting and empowering vulnerable communities to take appropriate action (Ajibade and McBean 2014); effective EWSs cannot simply be technical systems of information dissemination from top to bottom, but should utilise and develop community capacities, create local ownership of the system, and be based on a shared understanding of needs and purpose (Vogel and O'Brien 2006). Tapping into existing traditional knowledge has also been recommended to be incorporated into EWSs (Alessa et al. 2016). Combined with EWS, DRR approaches have long been considered to reduce the risk of household asset damage during one-off climate events (e.g., often used for response to floods and typhoons), and DRR approaches are increasingly being combined with climate adaptation policies (Thomalla et al. 2006; Mercer 2010). The Hyogo Plan of Action is a UN framework for nations to build resilience to disasters through effective integration of disaster risk considerations into sustainable development policies (Djalante et al. 2012; Sternberg and Batbuyan 2013). For example, in Vietnam a national strategy on disasters based on Hyogo has introduced the concept of a "four-on-the-spot" approach for DRR of proactive prevention; timely response; quick and effective recovery; and sustainable development (Garschagen 2016).

Community-based adaptation and programs to increase social capital and adaptive capacity: Community-based adaptation (CBA) approaches aim to identify, assist and implement activities "that strengthen the capacity of local people to adapt to living in a riskier and less predictable climate" (Ayers and Forsyth 2009), generally through generating adaptation strategies through participatory processes (Elf et al 2014). Such processes usually involve multiple methods to assess how current livelihoods are impacted by climate change and what potential adaptation options may exist: "These actions are intended to ensure that adaptation is more attuned to local needs, and consequently better able to reduce vulnerability to climate change. Consequently, CBA tends to focus on the local social, economic, and political contexts of poverty and vulnerability as well as physical bases of climate risks such as floods and droughts." (Forsyth 2013). Social capital plays a large role in the success of CBA approaches as it helps structure social ties, for example through reciprocity, and strengthens bonds among participants (Aldrich 2017; Pelling and High 2005; Triyanti and Marfai 2017). Examples of culturally-shaped adaptation responses through CBA might include informal non-monetary arrangements and social networks to cope with climate hazards; community organisations sharing communal responsibility; or food-sharing networks (Adger 2003; Adger et al 2007; Agrawal and Perrin 2008). Evaluations of CBA projects have been generally positive; Karim and Thiel (2017) evaluated CBA in Bangladesh where 80% of participants felt it increased their participation and contributed to reduced risks. Yet wider adoption of CBA is potentially hampered by several factors: the fact that most CBA projects are small scale, raising questions about up scaled effectiveness (Forsyth 2013; Ensor et al. 2014) and how to assess criteria of success for CBA (Forsyth 2013). (Nagoda and Nightingale 2017) also caution that CBA often is not able to adequately address the drivers of vulnerability such as inequality and power relations.

Implementation of indigenous knowledge, social learning, and knowledge transfer for adaptation: Cognitive and behavioural factors are being more acknowledged as the constraint to successful adaptation along with other factors (Adger et al. 2007). Traditional knowledge can contribute to accurate predictions of impending natural disasters (Green and Raygorodetsky 2010) as well as longer-term climate changes (Orlove et al. 2010). This often reliable source of information is unfortunately generally overlooked by the formal system, despite the fact that much of the literature encourages an integration of both scientific and indigenous knowledge where possible (Green and Raygorodetsky 2010; Speranza et al. 2010). Local knowledge often plays an important role in facilitating climate adaptation, as responses are often a delicate balance between relying on past experience and needing to introduce new ways to cope, which may require some additional formal learning (Leon et al. 2015). Education can also affect people's ability to make proactive adaptation decisions as well, because lack of education can reflect the inability to read and receive climate warnings, as well as information in post-climate disaster situations about recovery policies (Hallegatte et al. 2016). Knowledge exchange, social learning and other concepts are increasingly being incorporated into adaptation planning. Social learning has been defined as "a change in understanding and skills that becomes situated in groups of actors/communities of practice through social interactions," and such skills in the context of climate adaptation include "capacities to employ the best available knowledge, to incorporate emerging information, and to adapt planning and implementation strategies" (Albert et al. 2012). Social learning is often linked with attempts to increase levels of participation in adaptation decision making, from consultation to more serious community control (Collins and Ison 2009). Social learning has been documented in participatory landscape planning processes that have been piloted in several European contexts (Albert et al. 2012; McCrum et al. 2009) but is not yet widespread elsewhere. Other literatures refer to adaptive risk management (ARM) in which "one takes action based on available information, monitors what happens, learns from the experience and adjusts future actions based on what has been learnt" (Bidwell et al. 2013). Suggestions to facilitate social learning, ARM, and decision-making based on these include extending science-policy networks and using local bridging organisations, such as extension services (Bidwell et al. 2013; Böcher and Krott 2014). Knowledge coproduction, or "approaches that recognise the evolving nature of challenges the research community faces in remaining relevant to real world solutions and filling known evidence gaps" (Howarth and Monasterolo 2017) is another strategy highlighted in the literature.

Institutional capacity building at multiple scales: Natural resources policy has a great effect on vulnerability and adaptation because it influences such issues as land rights, agricultural production, water allocation, and other related topics. In understanding the role of institutions and governance in adaptation, authors have asserted "it is less a question of the type of government, but more of its ability to ensure that its people are able to anticipate, resist, and/or cope with the effects of changes brought about by climate." (Vincent 2007). Institutions have several important roles in this area: "a) they structure impacts and vulnerability, b) they mediate between individual and collective responses to climate impacts and thereby shape outcomes of adaptation, and c) they act as the means of delivery of external resources to facilitate adaptation, and thus govern access to such resources." (Agrawal and Perrin 2009). Key traits associated in the literature with more resilient policy planning for adaptation include decentralisation and autonomy; transparency and accountability; responsiveness and flexibility (Dodman and Satterthwaite 2008). Many of Elinor Ostrom's design principles for common pool resources management likely also apply to adaptation planning: (1) clearly defined boundaries; (2) proportional equivalence between benefits and costs; (3) collective choice arrangements; (4) monitoring; (5) graduated sanctions; (6) conflict-resolution mechanisms; (7) minimal recognition of rights to organise; and (8) nested enterprises (Ostrom 1990; Huntjens et al. 2012). Unfortunately, studies of many climate adaptation policy systems in developing countries in particular show a lack of flexibility, strong hierarchical tendencies, and a lack of local participation (Ampaire et al. 2017). Best practices to reform governance systems to facilitate climate adaption include supporting polycentric governance and recognising that adaptation planning is taking place in both formal and information institutions and among networks at different scales (Juhola and Westerhoff 2011), as well as public, private and civic institutions in different forms (Mubaya and Mafongoya 2017).

Mitigation: Some of the above adaptation approaches can be potentially combined with explicit mitigation strategies; examples include CBA with low carbon agriculture, or building codes encouraging more resilient housing combined with low energy use.

Prevention of desertification and *land degradation:* Adaptation approaches used in situations of impending drought or desertification can be useful, insofar as they make explicit linkages to prioritising prevention of desertification/degradation.

F*ood security:* Many soft adaptation options, like CBA or early warning systems have explicit interest in ensuring food security for households at risk. The literature is unclear if these approaches are more successful in delivering enhanced food security than harder options (e.g. irrigation expansion).

Biodiversity and ES: CBA can be combined with other adaptation approaches with an explicit biodiversity focus, such as EbA. Examples might include a community-based mangrove plantation project. The literature is unclear if soft adaptation approaches provide more biodiversity and ES than other types of adaptation, however.

UN SDGs: Community based adaptation options will likely contribute positively to many of the SDGs. Authors have suggested classifying options as "stand-alone" adaptation (which may not have an explicit development component); 'adaptation plus development', where development actions are made more resilient to climate events; and 'adaptation as development', where development is the basis for adaptation, as is often seen in CBA measures (Huq et al. 2002; McGray et al. 2007; Ayers and Dodman 2010). Synergies can easily be seen between some "climate change policies and the sustainable development agenda in developing countries, such as energy efficiency, renewable energy, transport and sustainable land-use policies" (Beg et al. 2002).

6.6 Desertification response options and their implications

6.6.1 Desertification response options considered

Desertification is a significant global challenge, covering about 5.2 Billion ha (excluding hyper-arid regions; UNEP, 1991), and is discussed in detail in Chapter 3. Response options to tackle desertification can be grouped under soil management options, vegetation management options, water management options, and integrated options. Most of the response options can be classified as forms of sustainable land management. *Soil management options* include management of erosion, soil carbon loss, salinisation, acidification and compaction. *Vegetation management options* include management to prevent overgrazing, to improve cropland management, to reduce forest cover loss, to maintain tree stocking density, to maintain tree species diversity, to prevent fragmentation and to manage fire. *Water management options* include management to reduce aquifer and surface water depletion, and to prevent over extraction, and the management of landslides and natural hazards (such as flooding). *Integrated options* include the management of biodiversity loss, dust storms, sand dune mobilisation, invasive species spread, pollution, urbanisation, wetlands and wildlife corridors. The co-benefits and adverse side-effects associated with these response options to tackle desertification, with interventions to tackle climate mitigation, climate adaptation, land degradation and food security are assessed in the sections below.

6.6.2 Soil management options

Desertification affects dryland regions around the world through vegetation cover, plant community composition, hydrologic conditions, and soil properties changes that cause losses in ecosystem services and threats to sustainable livelihoods (D'Odorico et al. 2013). Soil based options of desertification concern management of erosion, soil carbon loss, salinisation, acidification and compaction. Such options contribute to sustainable land management practices which are regionally differentiated according to climate, land use and soil types (Liniger et al. 2017). Below we address co-benefits and adverse side-effects between these soil-based management options, climate change mitigation, adaptation, prevention of land degradation, delivery of food security and UN SDGs.

Mitigation: Erosion is a carbon loss at site scale, for instance global wind erosion organic carbon losses from croplands range from 0.3 to 1.0 $GtC.y^{-1}$ (Chappell et al. 2016). Nevertheless, despite high uncertainty, the net C balance of erosion removal, compensatory soil sink, transport and re-deposition processes is currently considered to be a net C sink at global scale (Wang et al. 2017). Therefore, erosion control options may not have a direct positive effect on the global biosphere C balance. Nevertheless, by increasing primary productivity and, hence, organic carbon supply to soils (Lugato et al., 2016) erosion control, as well as the prevention of soil compaction, salinisation and acidification, may increase soil organic carbon pools in soils and vegetation. Since desertified lands are highly depleted in their soil organic C pool, their restoration can create a large C-sink capacity (Lal, 2010).

Adaptation: The restoration of degraded soils has potential for climate change adaptation of ecosystems and agriculture. Restoring degraded soils improves water availability to plants through soil organic matter mediated improvements in soil structure, water infiltration and water holding capacity, thereby reducing risks of soil-related droughts as well as flooding since infiltration reduces peak flows (Herrick et al., 2013). Increased soil organic matter also leads to improved soil biological properties (Guimarães et al., 2013). Restoring desertified soils also has large potential for climate change adaptation of agriculture and for the development of resilient production systems, especially under tropical conditions (Branca et al., 2013; Lipper et al. 2014). The use of biochar the solid by-product of the pyrolysis process for the production of bioenergy as a soil amendment increases the water-holding capacity of soil (Laird et al., 2010) and may therefore provide better access to water and nutrients for crops and other vegetation types.

Prevention of land degradation: Interventions to prevent desertification through improved soil management have large co-benefits with the prevention of land degradation, since they consist of options concerning the management of erosion, soil carbon loss, salinisation, acidification and compaction (Liniger et al. 2017).

Delivery of food security: Food insecurity in drylands can be attributed to a range of causes, which include political, economic, social and environmental factors. Addressing desertification, including land and soil degradation, can facilitate or ease the food security dilemma, but may not completely solve it in the presence of other underlying causes (Stringer et al., 2011). Nevertheless, co-benefits for yields could be obtained each year under tropical conditions by improving soil organic matter (Lal, 2006) and improving soil organic carbon stock by one ton C per hectare would have the potential to increase grain production in developing countries by 24-40 million $t.yr^{-1}$ (Lal, 2006).

Biodiversity and ecosystem services: The restoration of soils offers a cost-effective long term solution for hydrological risks and land degradation. Soil solutions aim to enhance the soil health and soil functions through which local eco-system services will be maintained or restored. Co-benefits on soil biodiversity (Guimarães et al., 2013) and ecosystem services are documented from case studies (Keestra et al., 2018).

UN SDGs: Soil management options to prevent desertification contribute positively to SDGs 1, 3, 6, 11, 14, and 15 (see FAO, 2016) and are unlikely to adversely affect any SDG.

6.6.3 Vegetation management options

Vegetation impacts landscape morphology, soil erosivity and desertification processes (Manuel et al., 2017). Prevention of desertification changes in albedo, the hydrological cycle and surface roughness, some of which affect the local and global climate, and it can provide negative side effect for prevention of global warming. Desertification particularly in the mid-latitudes (15-30 and 45-60 degrees North) was found to have a strong effect on global climate, resulting in temperature decreases of more than 2˚C in the boreal summer from 1700 to 2000 AD mainly caused by surface albedo change (Wang et al. 2016). Among the vegetation management options to prevent desertification include the prevention of overgrazing; improvement in cropland management practices; reduction in forest cover loss; maintenance of tree stock density and species diversity; fire management and the prevention of fragmentation.

In semi-arid regions, poor cropland management can lead to desertification (Ferreira et al., 2018) and loss of productive capacity. Improved cropland management and prevention of overgrazing provides direct co-benefits to desertification prevention. Heubes et al. (2011) estimate a 2 million km^2 expansion of grassland into desert by 2050. Reducing forest cover loss decreases erosional efficiency (Manuel et al. 2017) thereby providing positive co-benefits to desertification. In tropical regions, prevention of desertification by improving cropland management and reducing forest cover loss provides multiple benefits as prevention of land degradation, mitigation, adaptation, delivery of food security, biodiversity and ecosystem services. "Shifting agriculture" is a clearing of forests for growing crops until the soil is exhausted of nutrients, and it was greatest in Asia (about 30%) and about 15% over the whole tropical regions (Chakravarty et al. 2012). One of the options to reduce human pressure on natural forests is to assist plantations, but in the tropical forests, plantations often provide serious negative side effects for mitigation, biodiversity, and ecosystem services. Rapidly expanding tree crops and rubber plantation plays a more important role in deforestation in Indonesia than subsistence-oriented shifting agriculture. Plantations also increase deforestation by constructing roads that improve access, which can increase the frequency of large-scale wildfires.

Adaptation: Traditional pastoralists are particularly vulnerable to highly variable climatic regimes (Middleton, 2016). Improved cropland management can provide increased earnings and higher yields to poor farmers (Branca et al., 2016) which can provide direct co-benefits to adaptation. Cropland management strategies related to adaptation can enhance food security and reduce gender inequalities (Jethi et al., 2016). Drought is a driver of vegetation change and projected increases in severity and spatial extent (IPCC 2014b) threatens the livelihood security of the two billion people living in drylands (Middleton, 2016) where some of the most extreme effects of climate change are expected (Villamor and Badmos, 2015).

Mitigation: Maintaining tree stock and species diversity impacts biogeophysical and biogeochemical feedback processes related to carbon storage (Euskirchen et al., 2016) thereby creating co-benefits for climate change mitigation. Beyond the impacts on the carbon cycle, forest cover loss impacts land-atmosphere water and energy fluxes (Alkama and Cescatti 2016). Therefore, mitigation co-benefits can be derived from reduced forest cover loss. Prevention of overgrazing and forest cover loss provides positive co-benefits to mitigation by reducing radiative forcing from anthropogenic emissions (Calabro and Magazu, 2016) and diurnal temperature variations (Alkama and Cescatti 2016) Branca et al (2016) posited that the negative marginal abatement costs for improved cropland management practices are synergistic with climate change mitigation. Land degradation, especially in arid areas, impacts ground albedo and thereby the radiative balance (Calabro and Magazu, 2016).

Prevention of land degradation: Land degradation can reduce the effectiveness of adaptation options (Web et al. 2017) by inducing losses in vegetation and soil productivity (Calabro and Magazu, 2016) thereby increasing the susceptibility of soils to climate change impacts (Reynolds et al. 2007) and desertification. Prevention of overgrazing provides several co-benefits for prevention of land degradation, mitigation, adaptation, delivery of food security, biodiversity and ecosystem services. As an example, more than 90% of grasslands in China and 70% of Mongolia’s land surface are considered degraded by overgrazing, deforestation, and climate extremes. Options are to improve soil quality through natural regeneration and re-seeding with nitrogen fixers, and to maintain vegetation through appropriate rotation plans. To ensure sustainable pasture management, water management are crucial including piping water from rivers, building reservoirs and dams along slopes to slow down the runoff of rainwater, but such water management can provide competition with food production in surrounding regions.

Delivering food security: Improved cropland management creates direct co-benefits for food security by reducing the degradation of arable lands (Siegel, 2016). Prevention of overgrazing, improvement in cropland management and reduced forest cover loss, negates soil degradation threats to food security and prevents income depletion among farming households (Gomiero, 2016). Cropland management strategies such as fallowing can lead to improvements in soil quality (Ferreira et al., 2018) and reduce the expansion of deserts. The length of the fallow period is sensitive to population increase (Behera et al., 2016) and can create adverse side-effects for food security.

Biodiversity and ecosystem services: Reduced forest cover loss and maintenance of tree stocking density and, species diversity creates direct co-benefits for biodiversity and ecosystem services (Bremer et al., 2016). Crop management strategies related to ecological (vegetation) engineering provides multiple benefits for ecosystem services and livelihoods (Horgan et al. 2016) by creating habitat for wildlife, and enhancing regulatory ecosystem services.

UN SDGs: Vegetation management options to prevent desertification contribute positively to SDGs 1, 3, 6, 11, 14, and 15 (see FAO, 2016) and are unlikely to adversely affect any SDG.

6.6.4 Water management options

Climate changes are linked to and directly affected water managements on prevention of desertification. Identifying problems, possible risks on water availability and promote water management to response desertification are important stage for smart investment (Alavian et al. 2009). Change in water availability through improving co-managing floods and groundwater depletion at the river basin such as managed aquifer recharge (MAR), underground taming of floods for irrigation (UTFI), restore over-allocated or brackish aquifers, groundwater dependent ecosystems protection, reducing evaporation losses are significantly contributed to response climate change and reduced impacts of extreme weather event in desertification areas (Dillon and Arshad 2016a). Balance in the use of natural resources enforcement (based water resources, water conservation measures, water allocations) are also good options to response climate change and desertification prevention (Perreira 2005).

Mitigation: Enforcement of integrated land and water planning, restoring soil quality and water conservation and minimising wastewater in use produced co-benefits for mitigating desertification, carbon erosion prevention and reducing GHG emission (Perreira 2005). Increased water conservation to minimise water losses in arid regions (Qaiser et al. 2011), groundwater management, rainwater harvest, conservation and utilisation (Amalraj and Pius 2017) and improving fresh water storage in dryland cropping (Rengasamy 2006) are slightly provides co-benefits for climate mitigation. Such interventions reduce carbon losses, soil erosions and positively contribute to improving agricultural capacity and delivery of food security (Huang et al. 2014).

Adaptation to climate change and response desertification can be implemented through improvement of irrigation system and integrated water resources management. Over exploitation of groundwater and less water prevention are threated to agriculture and push on desertification, e.g. in India, Pakistan, Saudi Arabia, USA, China, Iran and Mexico (Gleeson et al. 2012), in Belgium, Denmark, Saudi Arabia and Austria (Zektser and Everett 2004). Reducing runoff and strengthening on-site water recharge together with groundwater extraction control are feasible options to contribute increasing water storage capacity and significantly reduce desertification, much *co-benefits* from crop production and water rehabilitation in drought areas (Thoa et al. 2008). Reducing water extraction through smart water use based-ecological water system, risk aversion from water extraction helps to improve adaption to response desertification (Alavian et al. 2009). Managed aquifer recharge (MAR) or underground taming of floods for irrigation (UTFI) are significant options to reduce evaporation losses and improve water supply security has *large co-benefits* of adaptation to climate change from water extraction (reducing costs of water supplies and polluted water treatment (Dillon and Arshad 2016a). Barriers includes technology, investment costs, community's awareness and balances of water consumption demands.

Prevention of land degradation through a response to desertification control can be promoted through improving aquifer and surface water depletion, controlling the over-extraction of water, improvement of water management and management of landslides and natural hazards. Watering shifting sand dunes (sprinkler), water resources conservation (Pereira 2002a; Nejad 2013), enhancing rainwater management, reducing recharge and increasing water use in discharge areas (DERM 2011) are potential options to prevent land degradation. Water security for multi-purposes (agriculture, residents, industrial, aquaculture and salinisation) produces *large co-benefits* for improving economic development, construction, acceptable water quantity and quality for health, livelihoods, ecosystems and production, acceptable level of water-related risks to human, environments and economic development (Grey and Sadoff 2007). Prevention of both groundwater and surface water depletion helps to prevent soil losses (by 9.5% on average in Europe, and by 20% for arable lands (Panagos et al. 2015). Roof top rainwater and preserving urban storm water helps to recharged and reduce water depletion in Australia, Germany, India, Jordan, USA that helps mitigating against land subsidence (Dillon and Arshad 2016b). Harmonising water use for energy and food security (Surveys et al. 2013), groundwater capture (Yang et al. 2015), enhancement of local water resource in use (Pereira 2002b), groundwater recharge (Zektser and Everett 2004), reducing runoff and strengthening on-site ground water recharge (Thoa et al. 2008), smart water use based-ecological water system (Alavian et al. 2009), managed aquifer recharge (MAR) or underground taming of floods for irrigation (UTFI) (Dillon and Arshad 2016a) are significantly contributed into enhancing soil production capacity, hence, provides *a large co-benefits* for prevention of land degradation. Barriers of implementation may relate to water security and water sharing among communities inside the borders and among countries.

Delivery of food security: water management options are not only prevent desertification but also significantly contribute into increasing productivity (crop yields or livestock). Integrated, efficient, equitable and sustainable water resource management (as water for agroecosystem) plays importance for food production and benefits to people (Lloyd et al. 2013) while integrated livestock, rangeland and crop production system, mixed and flexible agriculture, water harvesting for small scale irrigation and livestock (Nejad 2013). Improving water irrigation (Rengasamy 2006), improving rainfed agriculture (integrating soil and water management, rainfall infiltration and water harvesting, (UNCTAD 2011) provides *a large co-benefit* to delivery of food security through promoting crop establishment, flowering crop yield and soil texture in drying land areas. The barriers are concerning to information availability, high cost for soil capacity recovery and varieties.

Biodiversity and ecosystem services can be strengthened by provisioning water services, regulating services on water quality; supporting services on water recycling and ecological integration and cultural services that help to enhancing prevention of desertification. Ecosystem health and services can be enhanced by improving water management (Boelee E and E 2011). Securing ecosystem (Lloyd et al. 2013), integrated ecosystem-based management into water resources planning and management, linking ecosystem services and water security (Nicole Bernex 2016), improving correlation between amount of water resources and supply ecosystem services, combining water resources management and supply of ecosystem services (Liu et al. 2016), improving regulations for water sharing, trading and water pricing (ADB 2016), water smart appliance, water smart landscapes (Dawadi and Ahmad 2013), common and unconventional water sources in use (Rengasamy 2006) are reasonable options that provides *large co-benefits* for biodiversity and improving ecosystem services to response desertification.

UN SDGs: Improved water management is likely contributing positively to SDGs: 1, 3, 4, 5, 10, 11 12 and 15 (UN-WWAP 2015; Pradhan et al. 2017). Changes in availability of freshwater resources triggered by climate change can either reinforce the trade-offs with SDGs 3, 15, and 9, or influence negatively the historical synergies with SDGs 1, 3, 4, and 17 (Pradhan et al. 2017).

6.6.5 Integrated options

Among the integrated options for tackling desertification, many are sustainable land management options implemented at the landscape scale. These include management of: biodiversity loss, dust storms, sand dune mobilisation, invasive species spread, pollution, urbanisation, wetlands and wildlife corridors.

Mitigation: Sand dune stabilisation, which will also help to prevent dust storms, is often effected by increasing the vegetation density (Levin and Ben-Dor, 2004; see 6.6.3). Increasing vegetation would be expected to provide a small carbon sink. Any integrated option that also improves soil management, can also reduce soil carbon losses, reduce erosion losses and sequester carbon (Lal, 2001), thereby providing a climate mitigation co-benefit. Prevention of (vegetation) biodiversity loss, and maintaining vegetation to provide wildlife corridors might also protect or enhance carbon stocks (Jantz et al., 2014). Better management of urban spread could help prevent desertification, which in turn can protect carbon stocks lost to soil sealing (Barbero-Sierra et al., 2013). Prevention of the spread of invasive species spread could be either positive, neutral or negative for climate mitigation. Air pollution has been shown to supress precipitation (Rosenfeld, 2000), which will exacerbate desertification in arid regions, so the control of urban and industrial air pollution could aid in tackling desertification. Air pollutants have differing impacts on climate, with some air pollutants (e.g. aerosols at the top of the atmosphere) increasing the reflection of solar radiation to space leading to net cooling (Ramanathan et al., 2001), while others (e.g. nitrogen oxides; Seinfeld and Pandis, 2016) have a net warming effect, so control of these different pollutants will have differing (co-benefits and adverse side-effects) effects on climate mitigation. Wetland restoration to prevent desertification (particularly for steppe peatlands; Erwin, 2009) has co-benefits for carbon sequestration (Griscom et al. 2017b), but might temporarily increase methane emissions (Mitsch et al., 2012).

Adaptation: Sand dune stabilisation and prevention of dust storms, if implemented by increasing the vegetation density would likely provide adaptation co-benefits, through stabilised soils. Similarly, any integrated option that increases soil carbon stocks or reduces erosion (Lal, 2001), will provide an adaptation co-benefit. Prevention of (vegetation) biodiversity loss, and maintaining vegetation to provide wildlife corridors might also protect or enhance carbon stocks (Jantz et al., 2014), thereby providing an adaptation co-benefit. Prevention of the spread of invasive species spread could be either positive, neutral or negative for climate adaptation. The prevention of uncontrolled urban spread (Barbero-Sierra et al., 2013) might provide adaptation co-benefits, but adverse side effects for adaptation might arise due to restricted ability of people to move in response to climate change. Improved control of urban and industrial air pollution to help tackle desertification is likely to provide adaptation co-benefits *via* improved human health (Anderson 2017). Wetland restoration to prevent desertification (particularly for steppe peatlands; Erwin, 2009) has co-benefits for carbon sequestration (Griscom et al. 2017b) so will also have adaptation co-benefits.

Prevention of land degradation: Since desertification is a particular form of land degradation, any intervention aimed at tackling desertification will provide co-benefits to efforts to tackle land degradation.

Delivery of food security: Desertification control will prevent agricultural land from being lost (Lal, 2001; Barbero-Sierra et al., 2013), so will provide a co-benefit for food security. Sand/dust storms are a threat to resilient agriculture (Bal and Minhas, 2017), so sand dune stabilisation and prevention of dust storms to tackle desertification will also provide co-benefits for food security. Since biodiversity is essential for food security and agriculture (FAO 2010), efforts to tackle desertification by improving biodiversity or through use of wildlife corridors will provide co-benefits for food security. Adverse side-effects on food security would only be expected where productive land were taken out of production to create such wildlife corridors. Prevention of the spread of invasive species could benefit food production, particularly in the case of crop and livestock pests and diseases (Lowe et al., 2010), thereby providing co-benefits for food security. The prevention of uncontrolled urban spread might provide co-benefits for food security, since it is often agricultural land that is sealed by the urban expansion (Barbero-Sierra et al., 2013). The control of some air pollutants (such as tropospheric ozone) would have beneficial impacts on food production, since ozone decreases crop production (Carter et al., 2014). Some forms of air pollution that lead to global dimming (enhanced diffuse compared to direct light) are known to increase photosynthesis and might enhance crop productivity (Wild et al., 2012), though the evidence is still weak. Under these circumstances, reduced pollution could lead to adverse side-effects on food production, and thereby on food security. Wetland restoration would have little impact on food security unless it took place on land used for food production, which would lead to an adverse side-effect for food security.

Biodiversity and ecosystem services: Two proposed integrated options for desertification control (improving biodiversity or creation of wildlife corridors) are designed to provide co-benefits for biodiversity. These are also likely to create carbon sinks (Jantz et al., 2014), thereby contributing to climate regulation. Prevention of the spread invasive species would benefit native biodiversity

(McGeoch et al., 2010) and could benefit food, fibre and energy provision (Lowe et al., 2010). Sand dune stabilisation and prevention of dust storms will improve air quality, and pollution control would improve soil, water and air quality. Prevention of uncontrolled urban spread will also deliver across a range of ecosystem services (Kroll et al., 2012), and wetland creation can improve water provision, quality and flood management (Munang et al. 2013a). In some places, cultural services could be protected by desertification control if landscape and heritage value is preserved (Tengberg et al., 2012). Adverse side-effects might occur with food, fibre and energy production if cultivated lands are taken out of production, e.g. for wildlife corridors.

UN SDGs: Integrated options for tackling desertification will contribute positively to SDGs 1, 3, 6, 11, 14, and 15 (see Reynolds et al. 2007) and is unlikely to adversely affect any SDG, except for SDG 2 if land used for producing food were taken out of production without compensatory action elsewhere in the food system (Stehfest et al. 2009a; Bajželj et al. 2014b), though this would likely affect only small areas.

6.7 Land degradation response options and their implications

6.7.1 Land degradation response options considered

Global estimates of total degraded area vary from less than 1 billion ha to over 6 billion ha (Gibbs and Salmon, 2015), and is discussed in detail in Chapter 3. Response options to tackle land degradation can be grouped under soil management options, vegetation management options, water management options, and integrated options. Most of the response options can be classified as forms of sustainable land management. *Soil management options* include management of erosion, soil carbon loss, salinisation, acidification and compaction. *Vegetation management options* include management to prevent overgrazing, to improve cropland management, to reduce forest cover loss, to maintain tree stocking density, to maintain tree species diversity, to prevent fragmentation and to manage fire. *Water management options* include management to reduce aquifer and surface water depletion, and to prevent over extraction, and the management of landslides and natural hazards (such as flooding). *Integrated options* include the management of biodiversity loss, dust storms, sand dune mobilisation, invasive species spread, pollution, urbanisation, wetlands and wildlife corridors. The co-benefits and adverse side-effects associated with these response options to tackle desertification, with interventions to tackle climate mitigation, climate adaptation, desertification and food security are assessed in the sections below.

6.7.2 Soil management options

A recent major global assessment by FAO and ITPS (2015) revealed that the majority of the world's soil resources are in only fair, poor or very poor condition, as a consequence of population growth and economic growth as primary global drivers. From 10 identified main threats, soil erosion, loss of soil organic carbon and nutrient imbalance were the most significant ones at the global scale. More recently FAO (2017) published the Voluntary Guidelines for Sustainable Soil Management, which were associated with: a) minimum rates of soil erosion by water and wind; b) non degraded soil structure; c) sufficient plant cover to protect the soil; d) stable or increasing storage of soil organic matter; e) adequate availability and flow of plant nutrients; f) minimal soil salinisation, sodization and alkalization; g) efficient rainwater infiltration and soil water storage; h) soil pollutants below toxic levels; i) full range of biological functions and soil biodiversity; j) optimal and safe use of agricultural inputs; and j) minimum soil sealing of the. In another comprehensive study, (Smith et al. 2016a) recommend increasing soil resilience, recovering or sustaining their fertility and function, enhancing ecosystem services provided by soils, and considering the soils as a component of the ecosystems.

Mitigation: The prevention of soil degradation can mitigate climate change, either by maximising the capture of CO_2 from the atmosphere and increasing soil carbon stores (e.g. Snyder et al. 2009; Burney et al. 2010; Smith et al. 2011; Bustamante et al. 2014b; Paustian et al. 2016), or by minimising N_2O and CH_4 emissions that are driven by the presence of anaerobic sites and waterlogging (Snyder et al., 2009). Soil erosion control yields benefits on the maintenance of soil organic carbon contents in topsoil (Kirkels et al., 2014), while in eroded lands organic carbon may be also buried in deposition sites at depth (Stockmann et al., 2013), its control or prevention contributes to avoid further soil organic carbon losses. Possible adverse side-effects occur, for example, when nitrogen is added to unfertile soils, which generates higher N_2O emissions from the soil (Snyder et al., 2009).

Prevention of desertification: As described in Chapter 3 (Section 3.2), desertification feedbacks with climate change are due to sand dusts and aerosols, changes in surface albedo, and in vegetation and GHG fluxes. However, Lamchin et al. (2016) attributed the desertification process in Mongolia not only to surface albedo but mostly to TGSI (Topsoil Grain Size Index, an indicator of topsoil texture). Mitigation of sand dust storms require a combination of sustainable land management, integrated landscape and water management (Middleton and Kang 2017), which often results in climatic feedbacks by revegetation and changes in surface albedo (Zhang and Huisingh 2018). An expected direct impact is the increase of the energy available at the surface and the temperature of the atmosphere in layers with less amount of sand and dust present (Kaufman et al., 2002), as well as by land surface albedo distribution. Impact of prevention of desertification on GHG fluxes are largely dependent on vegetation type (i.e. annual or perennial, pasture, forest) and management.

Delivery of food security: Increasing soil degradation threats food security and the challenge of feeding 9 to 10 Billion people by 2050, particularly in tropical regions with less fertile soils and in poorer populations (e.g. McBratney et al., 2014; Lal, 2015; Smith, 2015). Any action tending to reverse the degradation of soils or recover their quality has evident co-benefits with food security. However, a debate that emerges is whether the increased production of food should be done at the expense of incorporating more areas of cultivation, or what margin exists to increase the yields of the main crops. In this sense, there is a consensus that we have reached a ceiling on the availability of fertile land suitable for cultivation, and that further expansion would be the expense of losses of biodiversity reserves (Viglizzo and Frank, 2006; FAO and ITPS 2015; De Clerk, 2016). On the contrary, there is an important margin to further intensification of agriculture, either with combined or integrated productions, implementing climate smart agriculture, or by increasing efficiencies in the use of water and the nutrients applied (Lipper et al. 2014; Smith 2015; Shirsath et al. 2017). There is a long way to go in increasing water productivity in the relationship between yield and evapotranspiration, as shown for different US agroecosystems and Australia (Sadras and Angus 2006), or by crop level adaptations (Challinor et al. 2014). More crop and livestock production increases GHG fluxes, although its impact can be mitigated by appropriate management decisions (e.g. Snyder et al. 2009).

Biodiversity and ecosystem services: Soils have one of the largest reserves of biodiversity on the planet, both because of the plant communities they support and because of the different life forms that inhabit their pores (Mc Bratney et al., 2014; Brevik et al., 2015; Wall et al. 2016). Lost ecosystem services are estimated to represent a significantly larger fraction (~10%) of global Gross Domestic Product-GDP (Sutton et al. 2016), so that any recovery practice that protects soils by limiting changes in cover for vegetation and losses of carbon and soil quality has obvious co-benefits with the maintenance of biodiversity and the provision of ecosystem services.

UN SDGs: Soils and soil science have a significant role for the realisation of SDGs (Keesstra et al., 2016). Management options to combat or prevent soil degradation (as described by FAO (2016)) will likely contribute positively to SDGs 2, 3, and 15, (as described by Montanarella and Lobos Alva, 2015).

6.7.3 Vegetation management options

Reduction in tropical forest cover loss and stocking density of trees provides multiple co-benefits for *prevention of desertification, mitigation, adaptation, biodiversity and ecosystem services, and human health.* South and Southeast Asia occupies one of the largest areas of tropical forests, and these forests account for about 20% of the potential global terrestrial Net Primary Productivity (Cervarich et al. 2016). Tropical land carbon sink exhibits a larger variability than temperate regions, due to the influence of climatic events, such of El Niño and La Niña, and the subsequent ecosystem disturbance such as large-scale forest fires and land use changes (Miettinen et al. 2016). Plantation expansion in Kalimantan is projected to contribute 18%–22% (0.12–0.15 Gt C yr^{-1}) of Indonesia's 2020 CO_2-equivalent emissions. Oil palm development across Kalimantan (538,346 km^2) from 1990 to 2010, and project expansion to 2020 within government-allocated leases represent a critical source of deforestation and carbon emissions (Carlson et al. 2012). Haze pollution over the past four decades in Southeast Asia is mainly a result of forest- and peatland-fire in Indonesia, in particular caused by degraded forests. The economic impacts of haze include adverse health effects and disruption to transport and tourism (Lin et al. 2017).

6.7.4 Water management options

The water management options such as reducing aquifer and surface water depletion and prevention of water over extraction, risk aversion from landslides and natural hazards provide direct *co-benefits* for mitigation and adaptation to climate change, prevention of land degradation, delivery of food security, enhancing biodiversity and ecosystem services. According to the UN Water report (UN-WWAP 2015), half of all countries globally have made significant progress towards implementing the principles of Integrated Water Resources Management (IWRM) to strengthen water conservation and protecting water resources from pollution for domestic use, agriculture, industry, hydropower and navigation, of course for preventing land degradation.

Mitigation: Improved water management options to prevent land degradation are also climate mitigation options. Preventing groundwater and surface water depletion helps to reduce soil losses and significantly contributed into carbon sequestration (Panagos et al. 2015). Reducing water depletion for food production in the upland areas has concerning forest resources and reducing emission from deforestation and forest degradation (REDD+) and produced co-benefits for farmers from enhancing PES (Karsenty et al. 2014). Reducing surface and groundwater depletion through improving irrigated technologies (sprinkle to mini-sprinkle and trickle, sub-surface drip irrigation and regulated deficits irrigation, (Levidow et al. 2014), irrigation without energy consumption such as solar-powered drip irrigation (Burney et al. 2010), optimised subsurface irrigation system (Gunarathna et al. 2017) and farming practices (climate smart agriculture, alternative wet and dry, (CGIAR 2017).

Adaptation: Improving irrigation system and integrated water resources management such as enhancing urban and rural water supplies, reducing water evaporation losses (Dillon and Arshad 2016b) are significant options to enhancing climate adaptation capacity. Improved irrigation for agriculture under water scarcity obviously contributed into improving food production and enhancing adaptation in desertification. Deficit irrigation, defined as water irrigated application below full crop requirements (evapotranspiration) is important practices to enhancing adaptability and co-benefits for desertification prevention and land degradation (Fereres and Soriano 2007). Micro-irrigation technologies (Baumhardt et al. 2015; Datta et al. 2000) enhancing effectives of dams and reservoirs (UNCTAD 2011) improving canal irrigation and improve maintenance for surface flooding (Prathapar 1988); water lifting (human powered pumps, electric and fossil fuel pumps, renewable energy powered pumps); intercept water in the transmission areas (single pumps, surface off-site drainage construction) and increase water use in discharge areas by salted tolerance vegetation and control water quality, (DERM 2011) are important solutions to produce co-benefits and enhance adaptation capacity. Waterlogging in coastal saline soil

(Dagar et al. 2016a), planning water management and water resources strategy for semi-arid for resilience to climate change in the long-term (Herrera-Pantoja and Hiscock 2015) are beneficiary options of adaptation to climate change and increase capacity to respond to land degradation.

Prevention of Desertification: Reducing water over extraction and misuse, preventing water pollutions (Blum 2013; Xia et al. 2017), reducing surface water depletion through roof top rainwater (Dillon and Arshad 2016b; Özdemir et al. 2011), management of aquifer recharge and storage, water storage (Dagar et al. 2016a), purposeful recharge of aquifers (Dillon et al. 2009), rainfed use for tillage, minimising tillage and terracing cultivation (Kassam et al. 2014), intensive rainfall episodes (Nguru and Rono 2013), Preventing over water extraction (Yang et al. 2015) water, fog-capture, water harvesting, cloud seeding and water transfers (Pereira 2002a), diversifying groundwater in use (Zektser and Everett 2004; UN-WWAP 2015; Gleeson et al. 2012), improving aquifer water depletion (Dillon et al. 2009) operating flood insurance, flood risk management (Jenkins et al. 2017), co-managing floods and managed aquifer recharge (MAR) (Baumhardt et al. 2015; Datta et al. 2000), improving soil-water conservation (Nejad 2013), improving early warning system (Winsemius et al. 2016), constructing dams for distributing floods (Yang et al. 2017; Jenkins et al. 2017), harmonising water uses allocating waters for different demands in both long-term and short-term (Dawadi and Ahmad 2013) are large co-benefits and minor adverse side effects and significantly contributed into preventing land degradation.

Delivery food security: water management options helps enhancing mitigation and adaptation capacity and improving prevention of land desertification also can promote increased productivity (crop yields or livestock). Improving water practices in watershed cultivation (Brindha, K. and Pavelic 2016), rainfed harvested for tillage, minimising tillage and terraces (Kassam et al. 2014), improving water productivity (Fereres and Soriano 2007) , accounting water, food and energy (Jiang 2015; Biggs et al. 2015) water quality management for agriculture (Jenkins et al. 2017) are optimal options and produce *large co-benefits* of improving crop yields, livestock and aquaculture for delivery food security to prevent land degradation.

Biodiversity and ecosystem services Low energy riverbank filtration (Dillon and Arshad 2016) mapping hotspot of ecosystem services (Kang et al. 2012), preserving agroecosystem, regenerating natural forest (Palacios-Agundez et al. 2015), improving wastewater agriculture, wastewater generation (Dagar et al. 2016b), developing regulations of wastewater (UN-WWAP 2015), accessing water institution and management (Jiang 2015), effective water management policies (Dawadi and Ahmad 2012), water pricing (Pereira 2005; UN-WWAP 2015) and information exchange produce *moderate co-benefits* and minor *adverse side-effects* for developing biodiversity and ecosystem services to prevent desertification and land degradation (Scherr and Yadav 1996). The barriers may concern governance capacity (engaging the societal actors, decision making across various levels of entities and water management, country capacity and empowerment), environmental pollution control and cost for waste water treatment.

UN SDGs: Improved water management is likely contributing positively to SDGs: 1, 3, 4, 5, 10, 11 12 and 15 (UN-WWAP 2015; Pradhan et al. 2017). However, in the context of climate change, global increase of population and consumption demand, there are negative synergy of water management to other SDGs. Changes in availability of freshwater resources triggered by climate change can either reinforce the trade-offs with SDGs 3, 15, and 9, or influence negatively the historical synergies with SDGs 1, 3, 4, and 17 (Pradhan et al. 2017) governance urban water to reduce influence anthropogenic and natural ecosystem (Milan 2017).

6.7.5 Integrated options

Since desertification is a specific form of land degradation (Chapter 2), the integrated options to tackle land degradation fall into the same categories as those for desertification, though they may be applied in any climatic zone and in any ecosystem threatened by land degradation. Among the integrated options for tackling land degradation, many are sustainable land management options implemented at the landscape scale. These include management of: biodiversity loss, dust storms, sand dune mobilisation, invasive species spread, pollution, urbanisation, wetlands and wildlife corridors. Land tenure has clear implications for how lands are managed for mitigation and adaptation ends, with equivalent impacts on desertification/degradation, BES, and human livelihoods to meet SDGs.

Mitigation: Increasing vegetation density as part of sand dune stabilisation (Levin and Ben-Dor, 2004) would be expected to provide a small carbon sink. Integrated options to tackle land management that also improve soil management, can also reduce soil carbon losses, reduce erosion losses and sequester carbon (Lal, 2001), thereby providing a climate mitigation co-benefit. Prevention of (vegetation) biodiversity loss, and maintaining vegetation to provide wildlife corridors might also protect or enhance carbon stocks (Jantz et al., 2014). Better management of urban spread could help prevent land degradation (Montanarella, 2007), which in turn can protect carbon stocks lost to soil sealing (Barbero-Sierra et al., 2013). Prevention of the spread of invasive species spread could be either positive, neutral or negative for climate mitigation. As for desertification, the control of (air) pollution could have either net cooling of warming impacts, depending on the pollutant controlled (Ramanathan et al., 2001; Seinfeld and Pandis, 2016; see section 6.6.5). Wetland restoration to address land degradation has co-benefits for carbon sequestration (Griscom et al. 2017b), but might temporarily increase methane emissions (Mitsch et al., 2012). Land tenure insecurity has been pointed to as a key driver of deforestation and land degradation at the local level (Clover and Eriksen 2009; Damnyag et al. 2012; Finley-Brook 2007; Robinson et al. 2014; Stickler et al. 2017). Currently, global estimates of the types of land tenure over forests show that approximately 25% are held by indigenous and local communities, 65% by the state (often in reality as open access or unclear tenure rules), and 10% as private lands (double-check and update) (White and Martin 2002). Poor management of state and open-access lands, leading to tragedy of the commons situations, has been combatted in recent years by a move towards forest decentralisation, which has shown considerable success in slowing forest loss and contributing to carbon mitigation (Agrawal et al. 2008; Chhatre and Agrawal 2009; Larson and Pulhin 2012; Pagdee et al. 2006; Holland et al. 2017; Gabay and Alam 2017). Forest titling programs have also improved management of forests, including for carbon, primarily by providing legally secure mechanisms for exclusion of others (Nelson et al. 2001; Blackman et al. 2017), while evidence on land titling for other lands (such as agriculture or grazing) is less definitive (Jacoby and Minten 2007; Kerekes and Williamson 2010).

Adaptation: Sand dune stabilisation and prevention of dust storms, if implemented by increasing the vegetation density, would likely provide adaptation co-benefits, through stabilised soils. Similarly, any integrated option that increases soil carbon stocks or reduces erosion (Lal, 2001), will provide an adaptation co-benefit. Prevention of (vegetation) biodiversity loss, and maintaining vegetation to provide wildlife corridors might also protect or enhance carbon stocks (Jantz et al., 2014), thereby providing an adaptation co-benefit. Prevention of the spread of invasive species spread could be either positive, neutral or negative for climate adaptation. The prevention of uncontrolled urban spread (Barbero-Sierra et al., 2013) might provide adaptation co-benefits, but adverse side effects for adaptation might arise due to restricted ability of people to move in response to climate change. Improved control of urban and industrial air pollution to help tackle land degradation is likely to provide adaptation co-benefits *via* improved human health (Anderson et al. 2017). Wetland restoration to prevent desertification has co-benefits for carbon sequestration (Griscom et al. 2017b) so will also have adaptation co-benefits. The role of secure tenure for indigenous communities in particular has been shown to be effective in reducing deforestation and degradation as compared with neighbouring lands, and is likely to help improve these communities' ability to adapt to climate changes (Suzuki 2012; Balooni et al. 2008; Ceddia et al. 2015; Pacheco et al. 2012; Holland et al. 2017), especially if adaptation is explicitly linked to REDD+ policy development (McElwee et al. 2017).

Prevention of desertification: Since desertification is a particular for of land degradation, any intervention aimed at tackling land degradation in arid regions will provide co-benefits to efforts to tackle desertification. Secure land tenure has been shown to be a useful tool against the spread of desertification/degradation (Gebremedhin and Swinton 2003; Enki et al. 2001; Hajjar et al. 2012), although several authors point out that tenure is usually necessary but not sufficient in many settings, which may be driven more by environmental variability or lack of financial resources (Pender et al. 2004; Bugri 2008; Lanckriet et al. 2015). Further, decentralised community-based activities, such as forestry incorporating enrichment plantings or alien species eradication activities, have been proving to be successful in preventing further degradation (Suyanto et al. 2005; Binns et al. 2001).

Delivery of food security: Land degradation control, when applied to agricultural land, will help to protect food production, so will provide a co-benefit for food security. Sand/dust storms are a threat to resilient agriculture (Bal and Minhas, 2017), so sand dune stabilisation and prevention of dust storms to tackle land degradation, will also provide co-benefits for food security. Since biodiversity is essential for food security and agriculture (FAO 2010), efforts to tackle land degradation by improving biodiversity, or through use of wildlife corridors, will provide co-benefits for food security. Adverse side-effects on food security would, however, be expected where productive land were taken out of production to create such wildlife corridors. Prevention of the spread invasive species could benefit food production, particularly in the case of crop and livestock pests and diseases (Lowe et al., 2010), thereby providing co-benefits for food security. The prevention of uncontrolled urban spread might provide co-benefits for food security, since it is often agricultural land that is sealed by the urban expansion (Barbero-Sierra et al., 2013). As for desertification control, the control of some air pollutants (such as tropospheric ozone) would have beneficial impacts on food production, since ozone decreases crop production (Carter et al., 2014), though some forms of air pollution could enhance crop productivity by increasing diffuse, compared to direct, sunlight (Wild et al., 2012), though the evidence is still weak. Wetland restoration could lead to an adverse side-effect for food security if it occurred on land used for food production. Secure land tenure has been shown to be related to access to markets which promotes stronger food security for smallholders (Maxwell and Wiebe 1998; Holden and Ghebru 2016; Corsi et al. 2017), as well as to the ability to make investments in increasing productivity of land (e.g. through hedgerows, soil improvements, etc.) (Holden and Otsuka 2014; Lawry et al. 2017).

Biodiversity and ecosystem services: Two proposed integrated options for land degradation control (improving biodiversity or creation of wildlife corridors) are designed to provide co-benefits for biodiversity. These are also likely to create carbon sinks (Jantz et al., 2014), thereby contributing to climate regulation. Prevention of the spread invasive species would benefit native biodiversity (McGeoch et al., 2010) and could benefit food, fibre and energy provision (Lowe et al., 2010). Sand dune stabilisation and prevention of dust storms will improve air quality, and pollution control would improve soil, water and air quality. Prevention of uncontrolled urban spread will also deliver across a range of ecosystem services (Kroll et al., 2012), and wetland creation can improve water provision, quality and flood management (Munang et al. 2013a). In some places, cultural services could be protected by land degradation control (Daniel et al., 2012; Tengberg et al., 2012). Adverse side-effects might occur with food, fibre and energy production if cultivated lands are taken out of production, e.g. for wildlife corridors. Lands with secure title or decentralised management are shown to produce more ES and preserve biodiversity better than those with insecure title or centralised management (Somanathan et al. 2009; Robinson et al. 2014; Stickler et al. 2017; Paudyal et al. 2017). REDD+ projects to provide carbon sequestration have found that ensuring secure land title before beginning activities has been a more successful approach than those projects that have not paid attention to tenure (Larson et al. 2013; Awono et al. 2014; Duchelle et al. 2014).

UN SDGs: Integrated options for tackling land degradation will contribute positively to SDGs 1, 3, 6, 11, 14, and 15 (see Reynolds et al. 2007), and is unlikely to adversely affect any SDG, except for SDG

2 if land used for producing food were taken out of production without compensatory action elsewhere in the food system (Stehfest et al. 2009a; Bajželj et al. 2014b), though this would likely affect only small areas. Secure land tenure is likely to provide more access to benefits from land management for local livelihoods, and thereby contribute to SDGs 1,2,3 (Salam et al. 2006; Pokharel et al. 2007; Sunderlin et al. 2008; Chhatre and Agrawal 2009; Riggs et al. 2016). Actions taken to secure land tenure for forest carbon mitigation projects contribute to SDG 13 and SDG 15.

6.8 Food security response options and their implications

6.8.1 Food security response options considered

Response options to improve food security include supply side measures, demand side measures and changes in the food system / value change (see Chapter 5). The response options to enhance food security can be categorised as options that increase food production, options that improve food storage and distribution, options that improve food supply, options that improve food access and nutrition, demand management options, and integrated options that work across the food sector. Options to *increase food production* include increased productivity (crop yields and livestock), stability of supply, closing yield gaps (sustainable intensification), post-harvest options, improved nutrition, technology transfer, fertiliser formulation, decision support tools, and seed sovereignty. Options to improve *storage and distribution* include transport and storage (including refrigeration). Options to improve *supply* include urban food systems and measures to cut down on unplanned urban sprawl that decreases agricultural lands, trade, retail, farm system movements, policy environment, value chains, processing, value added products and waste to biogas (through valuation of waste). Options to improve *food access and nutrition* include distribution, economic access (affordability), governance, nutrition and stability of supply. *Demand management options* include dietary change and waste reduction. *Integrated options* include livelihood diversification, financial risk pooling, insurance and credit options, securing land tenure and ownership, and prevention of land grabbing. The co-benefits and adverse side-effects associated with these response options to improve security, with interventions to tackle climate mitigation, climate adaptation, desertification and land degradation are assessed in the sections below.

6.8.2 Increase food production

Given that further expansion of cropland at the expense of deforestation or grassland losses is no longer a valid option, either due to lack of availability or due to the risk of bearing greater losses of biodiversity, increases in food production must be based on higher land productivity. For this it is necessary to increase the yields of the main crops. This can be done based on increasing the area under irrigation, which is competitive with human consumption (Elliott et al. 2014), or closing the existing yield gaps between what is actually obtained in the fields (Ya) and the potential yield under rainfed (Yw) or under irrigation or potential (Yp) conditions (van Ittersum et al., 2013; Aramburu Merlos et al., 2015). A significant part of closing yield gaps is to increase water use efficiencies, both by higher rainwater caption, soil storage and better utilisation by plants. This can be achieved by implementing the recommended Voluntary Guidelines for Sustainable Soil Management (FAO, 2016), climate smart agriculture (Lipper et al. 2014; Shirsath et al. 2017), or a better integration of production systems (Peyraud et al., 2014; Guilpart et al., 2017), and grazing management (Taboada et al., 2010). Without adaptation, crops such as wheat, corn and rice will be negatively affected by climate change, much more in tropical than temperate regions (Challinor et al. 2014). Although there is a wide field to improve both Yw and Yp, current and expected future rates of progress in yields are a matter of real concern and are insufficient to meet projected demand for cereals by 2050 (Hall and Richards, 2013). In Australia, case studies in wheat and soybean highlight biological links between improved nitrogen nutrition and drought adaptation (Sadras and Richards, 2014) A key issue is that crops such as maize have been selected for ideal conditions and higher yields, but are increasingly affected by non-ideal conditions of drought in the US Midwest (Lobell 2014). Seed sovereignty can potentially help address some of these issues of yield, particularly in the many parts of the developing world that do not rely on commercial seed inputs, through general promotion of local seed saving initiatives. Such actions can include seed networks, banks and exchanges and non-commercial open source plant breeding (Kloppenberg 2010; Luby et al. 2015; Bowman 2015; Campbell and Veteto 2015b; Reisman 2017; Patnaik et al. 2017). These locally developed seeds can both help protect local agrobiodiversity and can often be more climate resilient than generic commercial varieties (Coomes et al. 2015; van Niekerk and Wynberg 2017; Vasconcelos et al. 2013).

Mitigation: There are some co-benefits between the provision of food security and actions to mitigate climate change, for example, the construction of more resilient agricultural systems that decrease the variability of yields to climate (Lipper et al. 2014; Merante et al., 2017; Shirsath et al. 2017), or the improvements in fertility that entails increase of carbon stores in soils (e.g., Baldock et al., 2013; Lal et al., 2015). In any case, it is likely that the adverse side-effects will prevail, since any increase in food production implies increasing total GHG emissions from agricultural systems. The maximum that can be aspired is to achieve greater efficiency in the use of inputs such as water and nutrients, so that emissions per unit of product are lower. For this, actions that tend to limit further deforestation and conversion of pastures, and/or promote the use of best management practices in fertilisation and manuring (Snyder et al., 2009; Snyder et al. 2014; Oenema et al., 2014), and in livestock systems (Luo et al., 2010; Herrero et al. 2016a) contribute to more sustainable food production systems.

Prevention of desertification: The actions to prevent desertification are related in large part to the protection of soils with greater vegetation cover, and the promotion of conservation practices such as avoiding overgrazing and aggressive soil tillage, or diversified and enhanced production and better management of water and soil degradation, water harvesting, improving soil moisture, or reducing runoff. (D'Odorico et al., 2013; Schwilch et al., 2014). All these actions increase the resilience of food production systems, so there are clear co-benefits. However, there may also be some adverse effects, when production is limited or restricted in vulnerable environments in which control measures and strategies such as the establishment of halophytic and xerophytic plants, drainage networks, resilient facilities and infrastructure were proposed. (D'Odorico et al., 2013; Akbari et al., 2016).

Prevention of land degradation: Everything that is done to prevent land degradation has clear co-benefits with food production, especially when it comes to actions that increase the resilience of the system to climate change, such as the Voluntary Guidelines for Sustainable Soil Management (FAO 2017), or climate smart agriculture (Lipper et al. 2014; Shirsath et al. 2017). As with the prevention of desertification, there may be some adverse effects when the productive use of land vulnerable to degradation is restricted or limited, as happened with the Grain to Green Program in China (Deng et al., 2014; Liu et al.; 2014).

Delivery of food security: co-benefits are evident.

Biodiversity and ecosystem services: Land use changes between 1997 and 2011 have resulted in a loss of ecosystem services of between USD 4.3 and USD 20.2 trillion per year, and we believe that these estimates are conservative (Costanza et al., 2014), which can be attributed to a negative impact of food production. Although food production is an ecosystem service by itself, when it is produced on a large scale, other ecosystem services such as biodiversity and water regulation may be compromised, especially in high biodiversity ecosystems. This line of thinking leads to limit changes in the use and conservation of those systems with a high provision of ecosystem services and biodiversity, such as forests and wetlands, because in them the provision of ecosystem services has higher value than the provision of goods (Viglizzo and Frank, 2006). In those destined for large-scale food production, integrated approaches such as those contemplating a greater fragmentation of the landscape and respecting biological corridors are intended to meet both objectives, food production and provision of

SE, often with clear co-benefits. Climate smart agriculture (Lipper et al. 2014; Shirsath et al. 2017), integrated crop – livestock (Peyraud et al., 2014) and agroforestry (Lorenz and Lal, 2014) are clear examples of this. In Africa and the Middle East, Latin America, and East Asia, a shift to mixed systems can reduce pressures on tropical forest from agriculture, increased market-orientated production and improving rural livelihoods (Weindl et al. 2015a). However, there are always adverse effects, since modern agriculture always implies a simplification of the management and simplification of the landscape, with negative impacts on birds, pollinators and the emergence of resistant weeds (Nicholls and Altieri, 2013; Bustamante et al. 2014b Laurance et al., 2014; Nicolia et al., 2014; Lamichchane et al., 2017).

UN SDGs: Increase of food production will likely contribute positively to SDGs 2, 3, 12, and 15, (as described in (Bustamante et al. 2014b) and is likely to cause adverse effects on SDGs 6 (as described in Elliott et al., 2016), when there are competitive uses of freshwater and irrigation (Elliott et al. 2014), and SDGs 12 and 15, when new areas are put into cultivation and threat other ecosystem services and climate (e.g. Viglizzo et al. 2009).

6.8.3 Storage and distribution

Approximately one-third of the food produced for human consumption is wasted in postproduction operations (Bradford et al. 2018; Gustavsson et al. 2011). Technical and organisational innovations can improve food storage and distribution in low income African, Asian and Latin American countries where the problems of food losses are more acute (Ingram et al. 2016). Kumar and Kalita (2017) estimates that scientific storage methods can reduce losses to between 1%-2%. In a recent study, (Accorsi et al. 2017) highlighted the energy-intensive processes involved with food storage and distribution and the need to improve energy-efficiency along the 'cold chain' (Ingram et al. 2016). Another estimate suggests that the cold chain accounts for 1% of CO_2 production globally and improving its efficiency could reduce food waste and health risks associated with poor storage management practices (James and James 2010).

Mitigation: There are co-benefits and adverse side-effects between mitigation and improving food storage and distribution. For example, the use and maintenance of energy-efficient refrigeration equipment can significantly reduce emissions from the food system (Vermeulen et al. 2012; James and James 2010). Regarding adverse side-effects, the cost of refrigeration has implications for food safety and overall energy efficiency gains can be offset by increasing reliance on cold chain dependent foods (Garnett 2011). Utilising energy-saving strategies can support the balancing of benefits from reduced food waste and energy cost (Ingram et al. 2016). Therefore, mitigation gains from improved storage and distribution efficiency are sensitive to behavioural trends such as the increased consumption of refrigeration dependent food (Garnett 2011). While food transportation makes a relatively minor contribution to food chain emissions (Garnett 2011; Edwards-Jones et al. 2008) refrigeration is estimated to account for 15% of global energy consumption and is the most energy-intensive component of the food system (Vermeulen et al. 2012).

Adaptation: Implementing good management practices related to food storage and distribution will likely deliver co-benefits for adaptation (Ingram et al. 2016; IPCC 2014b; Stathers et al. 2013).For example, weatherproofing transport systems and improving the efficiency of food trade (Ingram et al. 2016; Stathers et al. 2013) especially in countries with inadequate infrastructure and weak food distribution systems (Vermeulen et al. 2012), can strengthen climate resilience against future climate-related shocks (Ingram et al. 2016; Stathers et al. 2013). Improving the performance of food storage systems increase resilience to climate change through (Ingram et al. 2016) improvements in the viability of seeds, reduction in the survival and reproduction of storage insect pests, increased shelf-life of products and the performance of storage protectant (Williams et al. 2017; Stathers et al. 2013). There are no obvious adaptation adverse side-effects from improving food storage and distribution.

Prevention of desertification: Climate change and overexploitation of soil are two of the primary drivers of desertification (Huang et al. 2016). Improved food storage and distribution will likely reduce GHG emissions thereby creating direct co-benefits for desertification prevention (Reynolds et al. 2007). The impacts of desertification can be exacerbated by poor postproduction management practices (Bradford et al. 2018; Temba et al. 2016; Stathers et al. 2013; Tirado et al. 2010). Improved storage and distribution can reduce food waste and the need for compensatory extensification of agricultural areas thereby reducing the risk of overexploitation. There are no apparent adverse side-effects from improved food storage and distribution.

Prevention of land degradation: Land degradation can reduce the effectiveness of adaptation options (Webb et al. 2017) and increase the susceptibility of agro-ecological systems to climate change (Reynolds et al. 2007). Improved storage and distribution reduces food waste and the need for compensatory extensification of agricultural areas thereby creating co-benefits for reduced land degradation (Stathers et al. 2013). There are no obvious adverse side-effects from improved postproduction management practices to the prevention of land degradation.

Delivery of food security: Improved storage and distribution systems provide clear co-benefits for food and nutrition security. Approximately one-third of the food produced for human consumption is wasted in postproduction operations (Bradford et al. 2018; Gustavsson et al. 2011). Kumar and Kalita (2017) estimates that most of these losses are due to poor storage management. Improved storage enhances food quality and can reduce mycotoxin intake (Bradford et al. 2018; Temba et al. 2016; Stathers et al. 2013; Tirado et al. 2010) especially in humid climates (Bradford et al. 2018). Improved energy efficiency in food storage and distribution has the potential to reduce emissions which could simultaneously reduce food costs and increase availability (Ingram et al. 2016). The perishability and safety of fresh foods are highly susceptible to temperature increase (Bisbis et al. 2018). Ingram et al. (2016) estimated a 50 percent reduction in bacterial growth for every 1°C drop in temperature below 10°C. Higher temperatures could increase the presence of pathogens and the challenge of managing food safety (Ingram et al. 2016). Mould growth in food storage facilities is primarily the result of elevated moisture content (Bradford et al. 2018). Food storage is directly linked to household level food security and the well-being of producers Stathers et al. (2013). Improving and expanding the 'dry chain' can significantly reduce food losses at the household level (Bradford et al. 2018). Dry chains are analogous to the cold chain and refers to the 'initial dehydration of durable commodities to levels preventing fungal growth followed by storage in moisture-proof containers' (Bradford et al. 2018). Adverse side-effects might occur if improved storage and distribution systems conflict with traditional practices.

Biodiversity and ecosystem services: In terms of biodiversity, improved storage and distribution are likely to provide co-benefits by impacting biomass of paper/card and aluminium and iron-ore mining used for food packaging (Ingram et al. 2016). Food processing and packaging activities such as washing, heating, cooling are heavily dependent on freshwater so improved postharvest storage and distribution could reduce water demand via more efficiently performing systems. Adverse side-effects might occur with energy production as some energy systems (e.g. solar) requires large areas of land which could have adverse effects on ecosystem services. Reducing effluent from food processing and packaging factories and nitrous oxides from transportation of food provides co-benefits for nitrogen and phosphorous cycles (Ingram et al. 2016).

UN SDGS: Improved storage and distribution will contribute positively to SDGs 1, 3, 5, 10 and 13 (Bradford et al. 2018; Temba et al. 2016) and is unlikely to adversely affect any SDG. Reducing food losses from storage and distribution operation can increase economic well-being without additional investment in production activities thereby creating multiple benefits for sustainable development.

6.8.4 Food Supply

The relative increasing welfare at global level is expected to lead to higher consumption and a greater demand for more processed food, meat, dairy, and fish, all of which add pressure to the food supply system (Godfray et al. 2010; van Vuuren et al. 2017). In this context, measures to improve food supply, including processing, retail, trade, value added products, farm system movements, policy environment, urban food systems and waste to biogas may have *co-benefits* and *adverse side-effects* with climate change mitigation, adaptation, desertification, land degradation and ecosystem services (Smith et al. 2013; Vermeulen et al. 2012).

Mitigation: The stages of processing and retail in the food value chain are fundamental to guarantee food security, mostly in terms of more accessible, affordable, reliable, and safe food for all sectors of society (Vermeulen et al. 2012). However, increasing processing and retailing activities in the food value chain may lead to adverse impacts on climate (Godfray et al. 2010). For example, as already identified for storage and distribution, refrigeration is found to be a major source of emissions also at the manufacturing and retailing stages (*adverse side-effect*), mostly due energy demand and refrigerant fluid leakage (e.g. hydrofluorocarbons) (Garnett 2011). On the other hand, reducing food waste and recycling at processing and retail stages are are identified as an important measure for improving food security, and it has direct *co-benefits* with climate mitigation due to increased efficiency in the food supply chain (Garnett 2011; Godfray et al. 2010; Hertel 2015; Smith et al. 2013). However there might be also other potential *adverse side-effects*, for example, if individual actors minimise food waste at the expense of creating increased packaging or transport needs (Vermeulen et al. 2012).

Deployment of urban food systems may also contribute to climate mitigation in food supply due to enhanced ability to feed urban populations through urban and peri-urban agriculture (Lee-Smith 2010; Revi et al. 2014). Climate *co-benefits* of implementing such a food supply chain are mainly associated to a more local food production and consumption chain and lower demand of resources for production, transport and infrastructure (connecting producers, retailers and consumers). In addition, this proximity may allow enhanced waste recycling and simpler food waste management systems (*co-benefit*) (Lee-Smith 2010).

Energetic use of waste and residues is seen as important measure to improve food supply systems (Smith et al. 2013). For example, coproducts, and waste streams in the food supply systems can be used to produce biogas. This may have an important contribution to food security by both increasing the overall system efficiency and making biofuel production locally available (*co-benefit)*. Avoiding emission of GHG in the food supply systems is another important *co-benefit* of this mitigation strategy. For example, better manure management and biogas production form manure in livestock systems may have an important contribution to reduce climate impacts from these food systems (Havlik et al. 2014; Iordan et al. 2016; Lipper et al. 2014). However, there may be *adverse side-effects* as well. For example, using crop residues for biogas production may leave less carbon in cropland ecosystems, and may adversely impact soil quality and the carbon balance of agricultural systems (Blanco-Canqui and Lal 2009; Smith et al. 2013).

Future food security for all will ultimately depend on management of the interacting trajectories of socio-economic and environmental changes (Vermeulen et al. 2012), which include consideration of the police environment in its formulation. For example, the socio-economic pathways have shown that future demand of crops and livestock for food may vary considerably, depending on the policy and regulation environment that will be in place to guarantee a certain global temperature stabilisation target. These different policy scenarios have a mixed effect on food prices, affecting food security. Generally, as a result of land needed for large scale bioenergy production and afforestation programs in future stringent climate mitigation scenarios, the use of land for food and feed production and pasture is reduced, following considerable agricultural intensification and dietary changes (Popp et al. 2017a;

Valin et al. 2014; van Vuuren et al. 2017). Therefore, the policy environment affecting food supply and demand in the different countries will have strong contribution to attend global climate mitigation goals (Golub et al. 2013).

Adaptation: There international trade has been identified with important potential role in facilitating global food security, since well-planned trade systems may act as a buffer to supply food to vulnerable regions where more extreme wheatear events are expected in face of climate change (*co-benefit*) (Baldos and Hertel 2015; Frank et al. 2017a; Porter et al. 2014; Wheeler and von Braun 2013). However, long distance transport has made possible scales of production that are themselves, from a GHG perspective, problematic (Garnett 2011) Climate impacts from long distance transport systems may also have a high contribution to the carbon footprint of some food supply systems, and these may become unfeasible in a future of resource scarcity and stringent climate regulation (*adverse side-effect*). In addition, if increased trade as an adaptation measure drives an expansion of agricultural areas (especially to marginal land and to forests), it would also lead to negative environmental consequences in the form of loss of biodiversity, deforestation, and additional carbon emissions (*adverse side-effects*) (Dasgupta et al. 2014; Lotze-Campen et al. 2010; Schmitz et al. 2012; Verburg et al. 2009).

Farm system movements are anticipated as adaptation strategy to improve food security since climate change impacts may require a reallocation of some food supply systems (Dasgupta et al. 2014; Havlik et al. 2014). However, this reallocation of farming systems may have adverse climate implications, mostly when the land use change harness the production potential of the land for either carbon sequestration, maintenance of carbon stocks or yields. A common illustration of carbon stock losses is when forests or other natural vegetation systems are converted into croplands (*adverse side-effect*) (Smith et al. 2013). For example, climate change may affect coffee production systems in several regions of the world with probable forced migration of coffee plantations toward higher altitude areas, and this can increase the pressure to open new agricultural areas at expense of native vegetation (*adverse side-effect)* (Dasgupta et al. 2014).

Cities with a heavy reliance on food imports would be more significantly affected by eventual food shortages resultant from climate change. Adaptation options may include support for urban and peri urban agriculture, green infrastructure (e.g. green roofs), local markets, enhanced social (food) safety nets and development of alternative food sources (Revi et al. 2014). These new food systems may have diverse unexpected *adverse side-effects* with climates systems, such as lower efficiencies in food supply and higher costs than modern large-scale agriculture.

Diversifying markets considering value added products in the food supply system may help to improve food security by increasing its economic performance and revenues to local farmers (*co-benefit*) (Reidsma et al. 2010). Adding value to residues and side-streams may help some food supply chains to adapt to future markets with more stringent climate regulation and improve income of smallholder farmers. For example, coffee industry by-products can be further processed to yield value added products such as natural antioxidants, vitamins, enzymes, cellulose, starch, lipids, proteins and pigments of high significance to the food, pharmaceutical and cosmetic industries (Murthy and Madhava Naidu 2012). Production of value added products may also have positive impact when the overall efficiency of the food supply chain is increased (*co-benefit*). Negative impacts are expected when further processing of residues and coproducts lead to higher emissions or demand of resources in the food system (*adverse side-effect*).

Prevention of desertification: Well-developed food trade systems may help to provide food in areas at risk of desertification, avoiding the over-exploitation of soil resources due to obsolete practices for agriculture and livestock production, which ultimately can lead to desertification (*co-benefit*). In the same way, some residues and sub-products of the food supply systems, such as manure, digestate, organic wastes and ashes, can be applied to soil for improving its quality attributes (Fernández-Bayo et al. 2017; Nkoa 2014), therefore contributing to prevent desertification and land degradation processes (*co-benefit*).

Prevention of land degradation: Food security is influenced by many factors, including land degradation and increasing competition for land and water from non-food uses. Adverse side-effects will be minimised if, instead of farming in new areas, there will be efforts to rehabilitate degraded, abandoned or underperforming lands. This means not just halting erosion and degradation but reversing it. For example, most farmland in Africa has been degraded over the past century by obsolete practices that were developed when population densities were lower. Some African countries have shown that rehabilitation using efforts to halt soil erosion, using a combination of trees, grasses and crops to build up soil organic matter, can represent a solution against land degradation (Clay 2011). As for preventing desertification, application of digestate to degraded soils is found to have beneficial effects as soil amendment, as it increases plant-available water, total soil carbon, and nutrients (Fernández-Bayo et al. 2017; Nkoa 2014). In general, carbon from digestates is more stable than other organic wastes and therefore has great potential to increase carbon sequestration in the soil (Smith et al. 2014a). Digestates have also shown a high fertilising potential for its high N content (Alburquerque et al. 2012; Möller et al. 2008). On the other hand, some studies argue that direct application of digestates as soil amendments is suspected to enhance carbon mineralisation in soils (priming effect), due to the elevated N concentration of digestates (Insam et al. 2015), and potentially increase salinity (Plaza et al. 2002). In the long term, land application of digestates may be restricted by the risk of accumulation of metals, increased salinity, phytotoxicity, and human and environmental toxicity potentially associated with some of the materials (Alburquerque et al. 2012).

Biodiversity and ecosystem services: Rising demands in food supply systems may increase pressure to further intensify crop production, thereby potentially affecting provision of ecosystem services (Bommarco et al. 2013). As already identified in this section, significant negative impacts on the environment and biodiversity have become evident due to agricultural intensification to guarantee food supply (*adverse side-effect*) (Moss 2008; Potts et al. 2010). In addition to GHG emissions, food supply systems have significant contribution to environmental threats such as pollution from fertilisers and pesticides, and the loss of biodiversity and ecosystem services due to historical conversion of vast amounts of natural ecosystems into croplands and pastures (Bommarco et al. 2013; Hoekstra et al. 2005; Tilman and Clark 2015; Tscharntke et al. 2005).

The hotspot areas where high biodiversity coexists with high food insecurity or a high risk of agricultural expansion were found to mainly occur in the tropics (Molotoks et al. 2017). Addressing food insecurity through methods such as agricultural expansion or intensification could lead to biodiversity loss through destruction of habitats important for conservation. Therefore, biodiversity and food security should not viewed independently, since there is an increasing need for recognising the strong interlinkages of these two issues (Molotoks et al. 2017).

6.8.5 Food access and nutrition

Recent food policies and international trade have increased market-based food access but for Global South smallholder farmers' access remains largely production based. Local food supplies by farmer movements in the global north is trending as a response to perceived reduced impacts and carbon footprint of food supply chains (Bajželj et al. 2014a; McMichael 2015). Trade driven food supply chains are becoming increasingly complex and contributing to emissions (Chapter 5; Wilhelm et al. 2016a). Additionally, globalised food systems are vulnerable to food price volatility, as was seen in the 2007-8 food price shocks that negatively affected food security for millions, most severely in Sub-Saharan Africa (Wodon and Zaman 2010; Haggblade et al. 2017). Food systems can be defined as "a set of dynamic interactions between and within the biogeophysical and human environments that result in the production, processing, distribution, preparation and consumption of food" (Chapter 5). This influences food availability, food access and food utilisation. There is emerging literature on responses targeting food access and nutrition to reduce emissions associated with food access and consumption (Chappell 2018). These include food processing to reduce waste, food transportation measures through energy mix and or renewables, but also regional-based food production to reduce the carbon footprint of accessing food (Göbel et al. 2015). Regional food systems attempt to link the production, distribution and access with social considerations through market-based measures to promote affordability of food (Rocha 2016; Benis and Production 2017). Food value chain governance is used the mediate stability of supply, given that given that climate change may increase production losses (Wheeler and von Braun 2013), leading to even high volatility in global agricultural commodity prices (Lewis and Witham 2012). There is limited evidence of how the regional food system contributes to emissions reduction. Food supply chains and flows are reported to have adverse effects on mitigation due to increased reliance on non-renewable energy (Kurian 2017; Scott 2017). For example regional food systems are under implementation to manage flows of food into, within, and out of the cities (Smit 2016; Benis and Production 2017). The mitigation target is to optimise urban-rural relations, to stimulate regional production, improve the quality of food and its diversity, the multiple functions of food production, distribution and access to foster regional food self-reliance (Aldababseh et al.; Bustamante et al. 2014a). Apart from initiatives in North America and Europe, a limited number of food policy models in the global South (e.g. Belo Horizonte, Brazil) exemplify such approaches. Regional food systems present opportunities for interconnectedness of the food system's component resilient food supply systems and city-regions have an important role (Brinkley et al. 2016; Rocha 2016). This depends on governance systems to enable local and regional policy makers, private sector actors, civil society and farmers to effectively address the availability, access to, nutritional value, and sustainability of food, while enhancing climate change mitigation. Responses for food and nutrition security, greening of the local urban economy and localising the food system as a driver for economic and community development has potential for mitigation.

The concentration of food demand is in urban areas and with rapid urbanisation in developing countries increasing food access challenges, fuelled by food price hikes that also contribute to low calorie intake and inadequate nutrition (Federici et al. 2016). The competition for land between urbanisation and agriculture in the regions around urban areas negatively affects food production around cities. This rapid urban expansion is especially high in emerging towns and cities (Lee et al. 2015), but these trends are being tackled through strategic use of policies including better integrated land use planning, (including use of mixed agricultural zoning or), arable land reclamation through urban redevelopment, and transfer of development rights or easements (Tan et al. 2009; Qian et al. 2015). Additionally, urban areas are becoming the principal territories for intervention in improving food access through innovative strategies that aim to eradicate urban hunger and improve livelihoods. Urban and Peri-urban Agriculture and Forestry (UPAF) has been implemented as a strategy that contributes to enhanced food security and nutrition in urban areas, contributes to local economic development, and addresses gender-based differences in accessing food since women play an important role in the productive reuse of urban food (Tao et al. 2015). The multiple nature of functions that UPAF can provide gives it a prominent role in the development of resilient urban systems (Lwasa et al. 2014).

At global level, local food policy and planning has recently occurred, with many of the policies implemented in North America and Europe. In the US alone there are over 200 Food Policy Councils, and cities like New York, Chicago, Vancouver, Toronto, London and Amsterdam have analysed their food systems and initiated comprehensive regional food strategies (Chappell et al. 2016; Brinkley et al. 2016). These seek to promote inter-linkages of the city and its citizens, and with the rural hinterland in order to create sustainable, and more nutritious food supplies for the city (Ravetz 2016; Lee et al. 2015). They aim at improving the health status of urban dwellers, reduce pollution levels, adapt to and mitigate climate change, and stimulate economic development. In addition, they seek to rebuild local food systems, by promoting marketing channels for locally produced foods, mobilising public demand (school canteens, etc.), and generating logistical infrastructures at appropriate scales as an interface between local supply and demand. The downside of food access and nutrition responses is that there is limited evidence for comprehensive policy frameworks for the development of resilient food systems. There is also limited evidence regarding mandates for food system governance at the local scale, which can help mobilise a required set of interlinked interventions at city-region scale involving various relevant stakeholders from civil society, public administration and market parties. Building a resilient regional food system should ensure sustainable food security through reinforcing rural-urban linkages, incorporating aspects of production, plan and provide for appropriate infrastructure, distribution systems within, and in, the surroundings of the city, matching supply and market demands, and adjusting these to the social and cultural environment and locally-specific natural resource base (Akhtar et al. 2016).

The range of responses for enhancing food access and nutrition include: distribution systems, storage systems for the supply chain, cold infrastructure, preservation, food waste reduction and land-based responses of local and regional food production systems to close yield gaps and enhance access at multiple spatial scales. The co-benefits of the responses are associated with regional food systems in which enhanced urban agriculture and forestry, regional rural food systems have potential for co-benefits of improving nutrition, health but also enhancing ecological services including carbon stocking (Dobbs et al. 2014; Lee et al. 2015; Specht et al. 2014). Evidence suggests that these options are being promoted but with *limited evidence* and *low agreement* on the potential for mitigation. The responses counter challenges of mitigation potential related to food distribution, supply chains and food waste reduction. Improvement of supply chains and distribution is likely to have side effects of increasing emissions due to overreliance on non-renewable energy that increases emissions (Akhtar et al. 2016). Food governance systems are likely to promote known technology for storage, distribution, and preservation with a rebound effect on emissions. Regional food systems, if not managed carefully, can have rebound effects on land-driven emissions as indicated by increasing pressure on agricultural land by urbanisation. Food production based responses may also drive dietary changes with long-term effects on climate change if this is not combined with energy efficient distribution systems. Regional food systems, whether urban focused or rural targeted, may also amplify land grabbing in some regions particularly Africa (Brinkley and Birch 2016).

6.8.6 Demand management options

Demand management options for increasing food security include dietary change (transition to sustainable healthy diets, often involving lower meat consumption in countries / regions of over-consumption) and reduction of food waste (Smith 2013; Bajželj et al. 2014a), since around 30% of all food produced is wasted (Kummu et al. 2012a; Gustavsson et al. 2011; Vermeulen et al. 2012).

Mitigation: A dietary shift away from meat can reduce greenhouse gas emissions (Stehfest et al. 2009b; Bajželj et al. 2014b), reduce cropland and pasture requirements (Stehfest et al. 2009b; Bajželj et al. 2014b), and reduce mitigation costs (Stehfest et al. 2009b). In particular, (Stehfest et al. 2009b) found that a transition to lower meat diets led to increased terrestrial carbon stock as cropland and pasture were abandoned, replaced by higher carbon density natural vegetation types. Combined with additional reductions in methane and nitrous oxide emissions, the authors find that dietary change can reduce the cost of mitigation by as much as 50% in a 450 CO_2-eq stabilisation case. Reductions in cropland and pastureland from dietary change can also lead to higher bioenergy production. Muller et al. (2017) also found that a reduction in meat consumption could reduce GHG emissions and environmental impact.

Bajželj et al. (2014b) found that reducing waste by 50% led to a reduction in cropland area of 14% and a 22%-25% reduction in GHG emissions. The inclusion of Healthy Diets further reduced cropland by 5%, pasture by 25%, and GHG emissions by 45%. Other studies have also found that future diets can have a significant impact on GHG emissions from food production. Havlìk et al. (2011) showed GHG mitigation potentials could be close to 2 $GtCO_2$-eqyr^{-1} under different future scenarios of crop and livestock production, and Popp et al. (2010) showed that GHG emissions of a decreased livestock product scenario were far lower than either a business-as-usual scenario or one in which only supply side mitigation measures were applied. Smith et al. (2013) found similar reductions in GHG emissions through dietary change.

Adaptation: By decreasing pressure on land (Smith 2013), demand reduction through dietary change and waste reduction could allow for decreased production intensity (Muller et al. 2017), which could reduce soil erosion and provide co-benefits to a range of other environmental indicators such as deforestation, and decreases in use of fertiliser (nitrogen and phosphorus), pesticides, water and energy (Muller et al. 2017), leading to potential adaptation co-benefits. No adaptation adverse side-effects have been reported.

Prevention of desertification: By decreasing pressure on land (Smith 2013), demand reduction through dietary change and waste reduction could allow for decreased production intensity (Muller et al. 2017), which could provide co-benefits for prevention of desertification where production intensity is contributing to desertification pressure.

Prevention of land degradation: By decreasing pressure on land (Smith 2013), demand reduction through dietary change and waste reduction could allow for decreased production intensity (Muller et al. 2017), which could provide co-benefits for prevention of land degradation where production intensity is contributing to land degradation.

Biodiversity and other ecosystem services: (Stehfest et al. 2009b) found that a transition to lower meat diets led to an abandonment of pasture and cropland, and subsequent regrowth of natural vegetation. Lamb et al. (2016) found a similar effect of dietary change in the UK. Demand-side management, including reduction in animal product consumption and food losses can enable the production of both food and biodiversity with less need for increased yields (van Vuuren et al. 2015). (Kummu et al. 2012b) reported that 24% of global freshwater and 23% of global fertiliser is used in the production of food losses, so reduction in food waste could provide significant co-benefits for freshwater provision and on nutrient cycling (Kummu et al. 2012b). Muller et al. (2017) found that lower impact agriculture could be practiced if dietary change and waste reduction were implemented, leading to lower GHG emissions, lower rates of deforestation, and decreases in use of fertiliser (nitrogen and phosphorus), pesticides, water and energy.

UN SDGs: Demand management can help to deliver SDG 2, and will also likely benefit SDGs 1, 3, 6, 7, 12, 13 and 15 (Bajželj et al. 2014a; Kummu et al. 2012b; Lamb et al. 2016; Smith 2018).

6.8.7 Other agri-food system options

There are other approaches to food security that combine interest in multiple combinations of increasing food production, improving food distribution, improving food supply, access and nutrition, and demand management options. What these options share in common is a 'social' approach to the issue that focuses both on farmer decision-making and access to resources as well as on the policy-level landscape that is needed to facilitate better decision-making. Some of these options often include explicit linkages to improved climate adaptation as well. Several of these approaches are discussed below, and there are undoubtedly more than could be discussed.

Livelihood diversification: When households' livelihoods depend on a small number of sources of income without much diversification, and when those income sources are in fields that are highly climate dependent, like agriculture and fishing, households have climate sensitive resource dependence (Adger 1999). This dependence can put food security at risk if agricultural systems do not prove resilient to forces of climate change, globalisation and trade, and other drivers acting together. Diversification has been identified as one option to help reduce climate vulnerability and increase food security, as "diverse patterns of resource use and heterogeneity of income sources increases robustness and adaptive capacity in social-ecological systems by helping spread risk in the case of severe disturbances"

(DiGiano and Racelis 2012). Surveys of farmers in climate variable areas find that livelihood diversification is increasingly favoured as an adaptation and food security improvement option (Van Aelst and Holvoet 2016). There is unclear agreement in the literature as to how much diversification can be encouraged through policy, and identification of a number of barriers, particularly for poorer households and female headed households, such as lack of assets to invest in new income streams, lack of education which inhibits proactive searches for new income sources, or discrimination (Berman et al. 2012; Van Aelst and Holvoet 2016; Ngigi et al. 2017).

Commercial crop insurance: Insurance is a risk-hedging strategy to guard against yield declines in agriculture by providing reimbursements to farmers from actual or estimated losses (Havemenn and Muccione 2011; Meze-Hausken et al. 2009), and has been used as a tool for food security as it may lead to expansions in agricultural production areas and increased food supply (Claassen et al. 2011; Goodwin et al. 2004). Commercial crop insurance is in fact somewhat of a misnomer as it tends to be highly subsidised by governments in the Global North (particularly the US) (Smith and Glauber 2012). While it can usefully provide ways for farmers to weather different risks in agricultural production, it may also 'mask' truly risky agriculture and prevent farmers from seeking less risky production strategies, such as diversification (Skees and Collier 2012; Jaworski 2016; Sanderson et al. 2013; Annan and Schlenker 2015). Insurance has also been pointed to as increasing crop production in marginal lands, leading to degradation (Claassen et al. 2011; Goodwin and Smith 2003).

Index insurance: One particularly new risk pooling model is weather-indexed insurance, which allows for pay-outs when a weather parameter is surpassed (e.g. seasonal rainfall falls below threshold, or a storm ranks above a severity index): "The chosen threshold must be objective, reliably measured, and strongly positively correlated with the insured's losses" for such index insurance schemes to work (Akter et al. 2016). Such insurance allows smallholders to reduce farming risks through fairly low-cost payments which are often highly subsidised by governments, as such programs have often failed to attract sufficient buyers or have remained financially unfeasible for commercial insurance sellers (Giné et al. 2008; Meze-Hausken et al. 2009). Peterson (2012) cautions that index insurance that relies too much on technical expertise has the potential to neglect local context in design and implementation. Gender differences have also been noted, with female farmers (who often comprise more than 50% of the rural workforce in many countries) exhibiting stronger loss aversion behavior and less likely to purchase weather insurance (Akter et al. 2016). The overall impact of index insurance on food production supply and access also has not been assessed.

Other forms of financial credit and risk pooling: For poor farmers in rural areas who may be subjected to increased climate risk, financial savings are almost always held in the form of assets (like livestock), which are more vulnerable to climate hazards than wealthier households' bank account (Hallegatte et al. 2016). Thus, credit and financing has been promoted to deal with food security and climate adaptation challenges; financial assistance policies that allow rapid access to subsidised credit for those facing extreme climate variability has been one approached (Linnerooth-Bayer and Hochrainer-Stigler 2014). Such financing can be used, for example, in assistance for households seeking to diversify sources of ecosystem-based income. Credit services have been shown to be important for both food security and adaptation actions, as lack of access to credit is considered a barrier to adaptation (Bryan et al 2009). Hammill et al. (2008) note that "credit acts as an ex-post source of capital when environmental and climate hazards occur" which can help reduce exposure or sensitivity, although there is limited evidence of mostly incremental adaptation actions (Fenton et al. 2017). However, there is an incomplete understanding of the role of credit in facilitating different types of adaptation or food security actions (Fenton et al. 2015).

There are also many types of informal risk pooling schemes used by rural farmers who are vulnerable to climate and price variability. Intra-household risk pooling is commonly encountered, such as through extended family financial transfers; one study found 65% of poor households in Jamaica report receiving transfers, and such transfers can account for up to 75% of household income or more after crisis events (Morduch and Sharma 2002). Community rotating credit associations (ROSCAs) have long been used for general risk pooling in rural areas (Bhattamishra and Barrett 2010), although the impact of these informal mechanisms on food security and access in general is unquantified in the literature.

Securing land tenure: Ensuring that farmer, particularly poorer ones in developing countries, have secure and defendable land tenure rights has been a policy supported by both national governments and international donors for many years. Tenure formalisation through land titling and registration, and through modes of recognition of common property, community co-management, and customary rights, have both been promoted (Deininger and Feder 2009). Land tenure security has been pointed to as a key policy for reduced deforestation and degradation of agricultural and forest lands (Suzuki 2012; Balooni et al. 2008; Ceddia et al. 2015; Pacheco et al. 2012). There are also impacts on increased food security, as tenure increases the probability that farmers will make improvements in their lands (Rao et al. 2016; Holden and Otsuka 2014; Lawry et al. 2017; Gebremedhin and Swinton 2003; Enki et al. 2001; Hajjar et al. 2012) and secure tenure correlates with increases in food production (Maxwell and Wiebe 1999; Holden and Ghebru 2016; Corsi et al. 2017).

Combatting land grabbing: Concerns about land grabbing have increased over the past decade, driven by a series of large-scale land acquisitions, and there are strong warnings that food security (especially at local levels) may be threatened by these large agribusiness deals (Daniel 2011; Lavers 2012; Golay and Biglino 2013). The scope of land acquisitions attributed to land grabbing over the past years is indicated in databases to be around 45 million ha of land by 2010 (Borras et al. 2011), and around 200 million by 2018 (Land Matrix 2018). In Africa alone, nearly 500 land investment projects in 18 countries over nearly 40 million ha had been documented by 2015 (Balehegn 2015). Because much of the land grabs are driven by northern consumer demand, there is concern that southern smallholders (the site of most interventions) are being unduly impacted, contributing to concerns about equity (Grant and Das 2015; Coscieme et al. 2016). There is some evidence that land grabbing has already led to the impoverishment of some communities and as many as 12 million people (Adnan 2013; Davis et al. 2014). There is inconclusive evidence that the food price crisis of 2008 was linked to expanding biofuel land deals (Kugelman and Levenstein 2013; Bush and Martiniello 2017). In at least some cases, the causal process is that land grabs contribute to increased tenure insecurity in surrounding lands, leading farmers to shift to cultivating smaller farms with less investments, potentially leading to food shortages (Aha and Ayitey 2017). A recent meta-analysis has shown that undernourished areas tend to export more “embodied agricultural lands” in foodstuffs for trade than they import (Marselis et al. 2017). Some scholars are concerned with displacement of smallholders from these grabs, potentially leading to impoverishment and increased (unsustainable) production elsewhere once pushed off lands (Borras et al. 2011; Adnan 2013); these have happened with frequency in many countries in Africa, where communal land tenure authorities have allowed expropriation of locally used lands without other farmers’ knowledge or compensation (Osinubi et al. 2016). Others see land grabs as investments that can contribute to more efficient food production at larger scales (World Bank 2011; Deininger and Byerlee 2012). The primary mechanisms for combatting large scale land grabs have included restrictions on the size of land sales (Fairbairn 2015); pressure on agribusiness companies to agree to voluntary guidelines and principles for responsible investment (Collins 2014; Goetz 2013); attempts to repeal biofuels standards (Palmer 2014); and direct protests against the land acquisitions (Hall et al. 2015; Fameree 2016).

Mitigation: These integrated options for food security overall have fairly small benefits for climate mitigation. Although secure land tenure tends to lead to improved management of forests with mitigation benefits (Nelson et al. 2002; Holland et al. 2017; Blackman et al. 2017), less is known about mitigation benefits from secure land titling in agriculture. Land grabbing can threaten not only agricultural lands of farmers, but also protected ecosystems, like forests and wetlands, therefore preventing land grabbing in general is likely to have some mitigation co-benefits, particularly in countries with good land availability and poor accessibility (Hunsberger et al. 2017; Carter et al. 2017).

Adaptation: Most of these integrated options provide good co-benefits for adaptation. Diversification and different forms of credit and insurance can help households ride out short-term shocks and crises and allow them to have a broader range of options for the future (Thornton and Herrero 2014). Land tenure security and prevention of land grabbing can help strengthen local systems of common property management, which can make some communities more able to adapt to climate changes in the future (Gabay and Alam 2017; Dell'Angelo et al. 2017).

Prevention of desertification and land degradation: Livelihood diversification offers potentials for prevention and reversal of desertification and land degradation, particularly through non-traditional crops or trees in agroforestry systems which improve soil (Antwi-Agyei et al. 2014). However, some insurance programs or other forms of financial assistance might provide perverse incentives for farmers to bring additional lands into crop production, particularly marginal or risky lands that could be at greater risk of degradation (Claassen et al. 2011). Preventing land grabbling and securing land tenure likely will help prevent some forms of degradation, as shifts from polyculture (often practiced by smallholders) to large scale monocrops in large scale land deals have negative consequences for soil degradation (Balehegn 2015). Many of the large land investments intensify unsustainable lands uses, and rarely practice more sustainable forms of agriculture such as organic or low-till (Friis and Nielsen 2016).

Biodiversity and other ecosystem services: The benefits to biodiversity from integrated options can be mixed. Livelihood diversification may increase on-farm biodiversity due to investments in more ecosystem-mimicking production systems, like agroforestry and polycultures. However, financial products like insurance and credit may increase the likelihood that farmers will invest in high-yield short-return farming instead, which may reduce biodiversity. Preventing land grabbing will likely have positive impacts for biodiversity, because many large-scale acquisitions, particularly in African countries, exceed the documented cultivable land area for the country; thus many of these investments are likely expanding cultivation into forest, wetlands and grasslands, which will have negative impacts on biodiversity and other ES (D'Odorico et al. 2017; Balehegn 2015). Water demands for intensification of large scale investments are also likely to increase with impacts on other users of water (Lazarus 2014).

SDGs: Livelihood diversification and financial mechanisms can contribute to SDG 1 and 2. Policies to restrict inappropriate land grabbing and secure land tenure are likely to contribute to SDGs 1, 2 and 3. Women may be at particular risk of negative impacts from land grabbing and thus combatting this may contribute to SDG 5 (Chu 2011; Collins 2014).

6.9 Managing interactions and interlinkages

6.9.1 Assessing multiple interactions and interlinkages of integrative response options

In the sections above, the impacts of each suggested response option to deal with a given challenge (climate change mitigation, climate change adaptation, desertification, land degradation and food security) has been assessed, on each of the other challenges. These sections (6.4 to 6.8) assess the co-benefits and adverse side-effects of each potential intervention in the context of the primary challenge it seeks to address. In this final section (6.9), we use the synthesis presented in sections 6.4 to 6.8 to assess the interlinkages across all challenges.

A number of issues related to the response options, co-benefits and adverse side effects should be noted:

1) The response options often overlap, so are not additive. For example, sustainable intensification to increase food productivity will involve changes to cropland, grazing land and livestock management, which in turn my include increasing soil carbon stocks. The response options should not therefore be summed, nor regarded as entirely mutually exclusive interventions.
2) The efficacy of a response option for addressing the primary challenge for which it is implemented needs to be weighed against any co-benefits and adverse side-effects for the other challenges, e.g. if a response option has a major impact in addressing one challenge but results in relatively minor and manageable adverse-side effects for another challenge, it may remain a powerful response option despite the adverse side-effects, particularly if they can be minimised or managed.
3) There is no equivalence implied in terms co-benefits or adverse side-effects, either in number or in magnitude of the impact, i.e. one co-benefit *does not equal* one adverse side-effect. As a consequence:
    a. Large co-benefits for one challenge might outweigh relatively minor adverse side-effects in addressing another challenge.
    b. Some response options may deliver mostly co-benefits with few adverse-side effects, but the co-benefits might be small in magnitude, i.e. the response options do no harm, but present only minor co-benefits.
4) A number of co-benefits and adverse side-effects are context specific; the context specificity is discussed in each of the following sub-sections where relevant.
5) Issues relating to saturation and reversibility of the response options are considered where applicable
6) Current barriers to implementation and any knowledge gaps are noted.

In this section we deal only with integrative response options, not the policies that are currently / could be implemented to enable their application; that is the subject of Chapter 7.

6.9.2 Integrative response options

The response options considered below can largely be treated as distinct options, though there are some overlaps. For example, sustainable intensification to deliver increased food productivity (6.9.2.5), will include the improved cropland management, grazing land management and livestock management 6.9.2.2-6.9.2.4). Similarly, cropland management and grazing land management partly act through increasing soil carbon sequestration 6.9.2.1), and one component of sustainable forest management (6.9.2.8), involves managing fire (6.9.2.19). Cross reference to the relevant sections are made where overlaps occur.

Other suggested methods to address land challenges are better described as goals than as response options. For example, the conservation of biodiversity is the very broad goal of other response options, including e.g. reduced deforestation (6.9.2.22), peatland restoration (6.9.2.32), coastal wetland restoration (6.9.2.29), ecosystem-based adaptation (6.9.2.21), management of pollution (6.9.2.26), management of invasive species (6.9.2.27) and various forms of sustainable land management that include sustainable intensification (6.9.2.5) and sustainable forest management (6.9.2.8), some of which might also reduce habitat fragmentation. Other suggested methods to address land challenges are better describes as over arching frameworks than as response options. For example, climate smart agriculture is a collection of response options aimed at delivering mitigation and adaptation in agriculture, including improved cropland management, grazing land management and livestock management (6.9.2.2-6.9.2.4). For this reason, broad goals such as "conservation of biodiversity", and overarching frameworks, such as "climate smart agriculture" do not appear as response options in the folloing sections.

Each of the response options is dealt with in turn below, roughly in order of the options with the largest co-benefits and fewest adverse-side effect first, and summarised in Table 6.3 and Figure 6.7 at the end of the section.

6.9.2.1 Increased soil organic matter content (and reduced losses)

Increased soil organic matter content (and reduced losses) can be achieved across a range of different land uses, including cropland, grazing land and forestry – and can be promoted by improved cropland, grazing land and forest management, as well as through afforestation / reforestation in most circumstances (Smith, 2012 – see also sections on these response options in this section; 6.9). Practices that increase soil organic matter content include a) *land use change* to an ecosystem with higher equilibrium soil carbon levels (e.g. from cropland to forest), b) *management of the vegetation*: including high input carbon practices, e. g., improved varieties, rotation and cover crops, perennial cropping systems, biotechnology, c) *nutrient management to increase plant carbon returns to the soil*: including optimised fertiliser application rate, fertiliser type, timing and precision application, d) *reduced tillage intensity and residue retention*, e) *improved water management*: including irrigation in arid / semi-arid conditions, f) *biochar application* (Smith et al. 2014; Smith, 2016). Increased soil organic matter content provides ***large co-benefits*** for climate mitigation by creating soil carbon sinks with a technical potential in the 3-5 $GtCO_2yr^{-1}$ range (Chapter 2; section 6.4; Smith et al. 2014; Smith, 2016), ***large co-benefits*** for climate adaptation by improving the resilience of food crop production systems to future climate change (Chapter 2; section 6.5; Porter et al. 2014), ***large co-benefits*** for prevention or reversal of desertification by improving soil health and sustainable use of land in arid areas (Chapter 3; section 6.6; D'Odorico et al., 2013), ***large co-benefits*** for prevention or reversal of land degradation by forming a major component of sustainable land management (Chapter 4; section 6.7; Altieri and Nicholls 2017) and ***large co-benefits*** for food security by increasing yield and yield stability to enhance food production (Chapter 5; section 6.8; Pan et al. 2009). There are ***few adverse side-effects*** across the challenges (Bustamante et al. 2014; Smith, 2016) as long as soil organic matter sinks are not increased by methods that increase the emissions of other greenhouse gases (Liao et al., 2015). The soil carbon sink, however, both saturates and is reversible (Smith, 2012). Increasing soil organic matter content is a low cost option, which can be cost negative (Smith et al. 2008; McKinsey and Company 2009). Barriers to implementation include biophysical (e.g. soil type; Baveye et al. 2018), technological (e.g. difficult to measure and verify; Smith, 2004), can be institutional in some regions (e.g. lack of institutional capacity; Bustamante et al. 2014), educational (e.g. poor knowledge of best practices among farmers; Reichardt, 2010), though cultural / behavioural barriers are likely to be small compared to other barriers (Smith et al. 2007; Wollenberg et al. 2016).

6.9.2.2 Improved cropland management

Improved cropland management is a collection of practices consisting of a) *management of the crop*: including high input carbon practices, e. g., improved crop varieties, crop rotation, use of cover crops, perennial cropping systems, agricultural biotechnology, b) *nutrient management*: including optimised fertiliser application rate, fertiliser type, timing, precision application, inhibitors, c) *reduced tillage* *intensity and residue retention*, d) *improved water management*: including drainage of waterlogged mineral soils and irrigation of crops in arid / semi-arid conditions, e) *improved rice management*: including water management such as mid-season drainage and improved fertilisation and residue management in paddy rice systems, and f) *biochar application* (Smith et al. 2014b). Improved cropland management provides ***large co-benefits*** for climate mitigation by reducing greenhouse gas emissions and creating soil carbon sinks in the range of 1.4 $GtCO_2yr^{-1}$ (Chapter 2; section 6.4; Smith et al. 2008, 2014), ***large co-benefits*** for climate adaptation by improving the resilience of food crop production systems to future climate change (Chapter 2; section 6.5; Porter et al. 2014), ***large co-benefits*** for prevention or reversal of desertification by improving sustainable use of land in arid areas (Chapter 3; section 6.6; Bryan et al., 2009; Chen et al., 2010), ***large co-benefits*** for prevention or reversal of land degradation by forming a major component of sustainable land management (Chapter 4; section 6.7; Labrière et al., 2015) and ***large co-benefits*** for food security by improving agricultural productivity for food production (Chapter 5; section 6.8; Porter et al. 2014). There are ***few adverse side-effects*** across the challenges (Bustamante et al. 2014a). While the soil carbon sink component of improved cropland management both saturates and is reversible (see section on increasing soil organic matter content; Smith, 2012), other components (such as reduced methane and nitrous oxide emissions) do not. However, if practices under improved cropland management are discontinued, the beneficial impacts will also cease. Improved cropland management is a low cost option, which can be cost negative (Smith et al. 2008, 2014b). Barriers to implementation include biophysical (e.g. land access; Bryan et al., 2009; Bustamante et al. 2014), technological (e.g. need for further development of nitrification inhibitors; Singh and Verma, 2017), can be institutional in some regions (e.g. poor sustainability frameworks; Madlener et al., 2006), educational (e.g. lack of knowledge; Reichardt, 2010), and cultural / behavioural (e.g. promotion of cover crops needs to account for farmers' needs; Roesch-McNally et al. 2017).

6.9.2.3 Improved livestock management

Improved livestock management is a collection of practices consisting of a) *improved feed and dietary additives:* to increase productivity and reduce emissions from enteric fermentation; including improved forage, dietary additives (bioactive compounds, fats), ionophores / antibiotics, propionate enhancers, archaea inhibitors, nitrate and sulphate supplements, b) *breeding and other long-term management*: including improved breeds with higher productivity or with reduced emissions from enteric fermentation; microbial technology such as archaeal vaccines, methanotrophs, acetogens, defaunation of the rumen, bacteriophages and probiotics; improved fertility, and c) *improved manure management:* including manipulation of bedding and storage conditions, anaerobic digesters; biofilters, dietary change and additives, soil applied and animal fed nitrification inhibitors, urease inhibitors, fertiliser type, rate and timing, manipulation of manure application practices, grazing management (Smith et al. 2014b). Improved livestock management provides ***moderate co-benefits*** for climate mitigation by reducing greenhouse gas emissions, particularly from enteric methane and manure management in the range of 0.5-0.7 $GtCO_2yr^{-1}$ (Chapter 2; section 6.4; Smith et al. 2008, 2014), ***large co-benefits*** for climate adaptation by improving the resilience of livestock production systems to future climate change (Chapter 2; section 6.5; Porter et al. 2014), ***large co-benefits*** for prevention or reversal of desertification by tackling overgrazing in arid areas (Chapter 3; section 6.6; Archer et al. 2011), ***large co-benefits*** for prevention or reversal of land degradation by allowing for reduced stocking density (Chapter 4; section 6.7; Tighe et al., 2012) and ***large co-benefits*** for food security by improving livestock sector productivity for food (Chapter 5; section 6.8; Herrero et al., 2015). There are ***few adverse side-effects*** across the challenges (Bustamante et al. 2014a). There are no saturation or reversibility issues associated with improved livestock management. The different practices contributing to improved livestock managed vary greatly in cost, with some cost negative (such as improved productivity; Smith et al. 2008; Herrero et al., 2015) and others expensive (such as some of the dietary additives; McKinsey and Co., 2011). Barriers to implementation include biophysical (e.g. climate suitability of different cattle breeds in a changing climate; Thornton et al., 2009; Rojas-Downing et al., 2017), technological (e.g. many dietary additives are still at low technology readiness level; Beauchemin et al., 2008), can be institutional in some regions (e.g. need for extension services; Ndoro et al., 2014), educational (e.g. poor knowledge of best animal husbandry practices among farmers; Ndoro et al., 2014), and cultural / behavioural (e.g. strong cultural importance of livestock in some communities (Herrero et al., 2015).

6.9.2.4 Improved grazing land management

Improved grazing land management is a collection of practices consisting of a) *management of the vegetation*: including improved grass varieties / sward composition, deep rooting grasses, increased productivity, and nutrient management, b) *animal management*: including appropriate stocking densities to fit carrying capacity, fodder banks, and improved grazing management fodder production, and fodder diversification, and c) *fire management*: improved use of fire for sustainable grassland management, including fire prevention and improved prescribed burning (Smith et al. 2014b). Improved grazing land management provides ***large co-benefits*** for climate mitigation by increasing soil carbon sinks and reducing greenhouse gas emissions in the range of 1.3 $GtCO_2yr^{-1}$ (Chapter 2; section 6.4; Herrero et al. 2016), ***large co-benefits*** for climate adaptation by improving the resilience of grazing lands to future climate change (Chapter 2; section 6.5; Porter et al. 2014), ***large co-benefits*** for prevention or reversal of desertification by tackling overgrazing in arid areas (Chapter 3; section 6.6; Archer et al. 2011), ***large co-benefits*** for prevention or reversal of land degradation by optimising stocking density (Chapter 4; section 6.7; Tighe et al., 2012), and ***large co-benefits*** for food security by improving livestock sector productivity for food (Chapter 5; section 6.8; Herrero et al., 2014). There are ***few adverse side-effects*** across the challenges (Bustamante et al. 2014a). While the soil carbon sink component of improved grazing land management both saturates and is reversible (see section on increasing soil organic matter content; Smith, 2012), other components (such as reduced methane and nitrous oxide emissions) do not. However, if practices under improved grazing land management are discontinued, the beneficial impacts will also cease. Improved grazing land management is a low cost option, which can be cost negative (Smith et al. 2008; McKinsey and Co., 2011). Barriers to implementation include biophysical (e.g. unless degraded, grazing lands are already closer to saturation than croplands; Smith, 2015), technological (e.g. e.g. need for further development of nitrification inhibitors; Singh and Verma, 2017), can be institutional in some regions (e.g. need for extension services; Ndoro et al., 2014), educational (e.g. poor knowledge of best animal husbandry practices among farmers; Ndoro et al., 2014), and cultural / behavioural (e.g. strong cultural importance of livestock and traditional practices in some communities (Herrero et al., 2015).

6.9.2.5 Sustainable intensification to deliver increased food productivity

Increased productivity of food (which could arise from many other interventions such as improved cropland, grazing land and livestock management) could help in addressing a number of the land challenges, but only if it is achieved in a sustainable way. Many interventions to increase food production, particularly those predicated on very large inputs of agro-chemicals, have resulted in a wide range of negative externalities (e.g. Godfray et al. 2010; Foley et al. 2011), leading to the proposal of sustainable intensification as a mechanism to deliver future sustainable increases in productivity (Burney et al. 2010; Tilman et al. 2011; Smith 2013; Garnett et al., 2013). Sustainable intensification to deliver increased food productivity could provide ***large co-benefits*** with yield improvement estimated to have contributed to emissions savings of >13 $GtCO_2yr^{-1}$ since 1961 (Burney et al. 2010) It also reduces the greenhouse gas intensity of products (Bennetzen et al. 2016a,b) which means a smaller environmental footprint of production, since demand can be met using less land and/or with fewer animals (Chapter 2; section 6.4). By reducing pressure on land and food production, there are also ***large co-benefits*** for adaptation (Chapter 2; section 6.4; Campbell et al. 2014). In some circumstances, there are ***large co-benefits*** for prevention of desertification (Chapter 3; section 6.6; Dai, 2010) and ***large co-benefits*** for prevention or reversal of land degradation (Chapter 4; section 6.7; Clay et al., 1995). There are ***large co-benefits*** for food security, through increased production of food (Chapter 5; section 6.8; Godfray et al. 2010; Tilman et al. 2011; Godfray and Garnett 2014). Intensification has led to a wide range of negative impacts on water quality, air quality and biodiversity (Tilman et al. 2011), but sustainable intensification (by definition) aims to increase food productivity ***without adverse side-effects*** (Garnett et al., 2013), since it would not be considered sustainable intensification if there were negative impacts. Barriers to implementation include technological barriers, for example limited ability to define and measure indicators of sustainable intensification (Barnes and Thomson 2014), biophysical, since increasing food productivity can be limited by climatic and environmental factors (Olesen and Bindi, 2002), institutional (e.g. better access to credit, services, inputs and markets, Schut et al., 2016), educational (e.g. educational needs of women; Pretty and Bharucha 2014), and cultural / behavioural (Martin et al. 2015).

6.9.2.6 Agro-forestry

Agro-forestry includes planting trees in croplands and silvo-pastural systems. Agro-forestry provides ***large co-benefits*** for climate mitigation by increasing carbon sinks in vegetation and soils (Chapter 2; section 6.4; Delgado et al., 2011; Mbow et al., 2013) in the range of 1.4 $GtCO_2yr^{-1}$ (Griscom et al. 2017a), ***large co-benefits*** for climate adaptation by improving the resilience of agricultural lands to future climate change (Chapter 2; section 6.5; Mbow et al., 2013), ***large co-benefits*** for prevention or reversal of desertification by providing perennial vegetation in arid areas (Chapter 3; section 6.6; Nair, 2007; Lal, 2001), and ***large co-benefits*** for prevention or reversal of land degradation by stabilising soils through perennial vegetation (Narain et al., 1998; Lal, 2001). Depending on how implemented, adding trees to the landscape could reduce the land area available for food production, though well planned agro-forestry can enhance productivity (Bustamante et al. 2014a), so could have ***large co-benefits*** for food security (Chapter 5; section 6.8; Baliton, et al., 2017; Paudela et al., 2017). There are ***few adverse side-effects*** across the challenges (Bustamante et al. 2014a), though removal of land for food production could occur. The carbon sink provided by agro-forestry both saturates and is reversible (see also section on increasing soil organic matter content; Smith, 2012). Agro-forestry is a low cost option (Smith et al. 2014b). Barriers to implementation include biophysical (susceptibility to pests; Sileshi et al. 2008), institutional in some regions (e.g. seed availability; Lillesø et al., 2011), educational (e.g. poor knowledge of how best to integrate trees into agro-ecosystems; Meijer et al., 2015), and cultural / behavioural (e.g. farmers perceptions; Meijer et al., 2015). There are likely to be relatively few technological barriers (Smith et al. 2007).

6.9.2.7 Sustainable healthy diets

Sustainable healthy diets represent a range of dietary changes to improve human diets, to make them healthy in terms of the nutrition delivered, and also (economically, environmentally and socially) sustainable. A "contract and converge" model of transition to sustainable healthy diets would involve a reduction in overconsumption (particularly of livestock products) in over-consuming populations, with increased consumption of some food groups in populations where minimum nutritional needs are not met. A transition to sustainable healthy diets would provide ***large co-benefits*** for human health, but also ***large co-benefits*** for climate mitigation, by reducing greenhouse gas emissions from the global food system significantly (Chapter 2; section 6.4; Bajželj et al. 2014a; Tilman and Clark 2014) in the region of 4.4 $GtCO_2yr^{-1}$ (Stehfest et al. 2009a), and ***moderate co-benefits*** for climate adaptation (Chapter 2; section 6.5; Soret et al., 2014; Song et al. 2017), potentially ***small co-benefits*** (due to relatively limited global area) for prevention or reversal of desertification by freeing cropland to be replaced by perennial vegetation in arid areas (Chapter 3; section 6.6), and ***large co-benefits*** for prevention or reversal of land degradation by reduced overgrazing or freeing cropland to be replaced by perennial vegetation (Chapter 4; section 6.7; Stehfest et al. 2009b), all by reducing pressure on land (Bajželj et al. 2014a; Clark and Tilman 2017). By definition, this would also deliver ***large co-benefits*** for food security (Chapter 5; section 6.8; Tilman and Clark 2014). There are likely to be ***few adverse side-effects*** across the challenges (Bajželj et al. 2014a; Tilman and Clark 2014; Clark and Tilman 2017). The main barriers to implementation are cultural / behavioural (e.g. diets are deeply culturally embedded and behaviour change is extremely difficult to effect, even when health benefits are well known; Macdiarmid et al., 2016). Biophysical barriers include poor accessibility of healthy foods such and fruit and vegetables (e.g. Hearn et al. 1998; Lock et al., 2005) and technological barriers include inadequate storage options for e.g. fresh fruit and vegetables (Kitijona et al., 2011). Barriers might also be institutional in some regions (e.g. poorly developed dietary health advice; Penny et al., 2005) and educational (e.g. poor knowledge of what constitutes a healthy diet; Wardle et al., 2000).

6.9.2.8 Sustainable forest management

Sustainable forest management includes a wide variety of practices, including the regeneration modality (natural or artificial), the species planted, the schedule and intensity of operation (thinnings, selective logging, final cut, etc.), aimed at maintaining and enhancing the economic, social and environmental value of existing forests, for the benefit of present and future generations (UN 2008). Sustainable forest management provides ***large co-benefits*** for climate mitigation by conserving and enhancing carbon stocks in forests and long-lived products and by providing wood used to reduce GHG emissions through material and energy substitution (section 6.4, Smith et al. 2014). Given the trade-offs between forest management options (i.e., increasing forest carbon stocks vs. increasing the substitution effects), it is the overall climate impact of both options that should be maximised in a given time frame. Sustainable forest management provides ***large co-benefits*** for climate adaptation, e.g., by conserving biodiversity and improving the resilience of both forests and local communities to future climate change (section 6.5, e.g. Locatelli et al. 2011), and ***moderate co-benefits*** for prevention or reversal of both desertification and land degradation, e.g., by affecting land stabilisation, water and microclimatic regulation (section 6.5, e.g. (Locatelli et al. 2015c; Alkama and Cescatti 2016). For example, sustainable forest management such as selective logging allows to retain substantial levels of carbon stocks, biodiversity, and timber volumes (Putz et al. 2012), and can therefore offer co-benefits in terms and mitigation, adaptation and prevention of land degradation. The sustainable management of already existing forests typically has ***small or negligible impact*** on food security. There are few possible ***adverse side-effects*** across the challenges. The carbon sink provided by forest management saturates and is reversible (Smith et al. 2014b). To this regard, integrating adaptation and mitigation allows reducing the impacts of climate change on forests, as such impacts may jeopardise the permanence of carbon storage (Locatelli et al. 2011a). Forest management affects the climate also through biophysical effects and the emissions of biogenic volatile organic compounds (BVOCs), which are both influenced by species composition. Forest management strategies aiming at maximise the carbon sink (e.g., fast-growing tree monoculture) may reduce biodiversity and options for ecological adaptation (Locatelli et al. 2015b). Barriers to implementation of sustainable forest management practices are mainly educational (limited knowledge of the most appropriate techniques, ref.) and institutional (e.g. better access to credit and markets, etc.). Forest certification may be an effective instrument to promote sustainable forest management (Bustamante et al. 2015).

6.9.2.9 Agricultural diversification

Agricultural diversification includes a set of agricultural practices and products obtained in the field that aim to improve the resilience of farmers to climate variability and climate change, and to the economic risks posed by fluctuating market forces. In general, the agricultural system is moved from one based on low-value agricultural commodities to another more diverse one composed of a basket of higher value-added products (e.g. Lipper et al. 2014; Waha et al., 2018). This diversification has ***small co-benefits*** for mitigation, but has the potential to deliver ***large co-benefits*** for adaptation to climate change. Diversification could also deliver ***moderate co-benefits*** for the prevention of desertification and land degradation since it can reduce the pressure on land (Lambin and Meyfroidt 2011), and ***large co-benefits*** for the achievement of food security (e.g. Birthal et al. 2015; Massawe et al. 2016; Waha et al. 2018) and household income (Pellegrini and Tasciotti, 2011). There are likely ***few adverse side effects*** (Massawe et al. 2016; Waha et al. 2018). However, diversification is not always economically viable (Barnes et al. 2015), and technological, biophysical, educational, and cultural barriers may emerge that limit the adoption of more diverse farming systems by farmers (Barnett and Palutikof 2015; Ahmed and Stepp 2016; Roesch-McNally et al., 2016). More support from extension services, access to inputs and markets, economic incentives for producing a certain crop or livestock product, research and investments focused on adapted varieties and climatic resilient systems, a combination of agricultural and non-agricultural activities (e.g. off farm jobs) are all important interventions aimed at overcoming barriers to agricultural diversification (Martin and Lorenzen 2016; Waha et al. 2018).

6.9.2.10 Management of erosion

Soil erosion is the removal of soil from the land surface by water, wind or tillage, which occurs worldwide but it is particularly severe in Asia, Latin America and the Caribbean, and the Near East and North Africa (FAO and ITPS 2015). Soil erosion management includes conservation practices such as the use of minimum tillage or zero tillage (Derpsch et al. 2010; de Moraes Sá et al. 2017), crop rotations and cover crops, rational grazing systems, among others (Poeplau and Don 2015) and also engineering-like practices such as construction of terraces and contour cropping for controlling water erosion, or forest barriers and strip cultivation for controlling wind erosion (Chen 2017). In eroded soils, the advance of erosion gullies and sand dunes can be limited by revegetation, among other practices. The management and control of erosion has ***small co-benefits for mitigation***, mostly by avoiding greater losses of organic carbon in water- or wind- transported sediments, but since the final fate of eroded material is still debated, the overall impact of erosion control on mitigation is context specific and at the global level, uncertain (Hoffmann et al., 2013). Soil erosion control measures can deliver ***large co-benefits*** for adaptation, since soil erosion control is a physical / structural adaptation option (IPCC AR5 WG2, Chapter 14; Noble et al. 2014). It can also deliver ***large co-benefits*** for prevention and reversal of land degradation and ***large co-benefits*** for prevention and reversal of desertification, since soil erosion is the most important soil degradation process (FAO and ITPS 2015). Erosion control measures deliver ***moderate co-benefits*** for food security, mainly through the preservation of crop productivity (Cook, 2016). There are likely ***no adverse side-effects*** from erosion control measures. The most prominent barriers to implementation include institutional factors (e.g. laws, regulations of use), technological (e.g. limited technology choices), educational (lack of farmer knowledge of erosion control measures), social, and economic factors (e.g. credit support, tax discounts). For instance, in Ethiopia farmers have shown an increased understanding of the soil erosion problem, but soil conservation programs face a host of barriers related to limited access to capital, limited benefits, land tenure insecurity, limited technology choices and technical support, and poor community participation (Haregeweyn et al. 2015).

6.9.2.11 Land tenure / ownership

Establishing secure land tenure includes options such as: 1) formalisation through land titling and registration; 2) community co-management; and 3) changing legal and policy frameworks to recognise customary rights (Deininger and Feder 2009). Land tenure security options are likely to have ***small co-benefits for mitigation,*** namely due to the fact that forest titling programs tend to lead to improved management of forests (Nelson et al. 2001; Holland et al. 2017; Blackman et al. 2017). Land tenure security is likely to lead to ***large co-benefits*** for adaptation, as it leads to reduced deforestation and degradation, increasing communities' ability to use forest resources to adapt (Suzuki 2012; Balooni et al. 2008; Ceddia et al. 2015; Pacheco et al. 2012). Tenure security can also deliver ***moderate co-benefits*** for land degradation and ***moderate co-benefits*** for desertification, as secure land tenure increases households' ability to make investments like agroforestry, hedgerows or soil improvements (Rao et al. 2016; Holden and Otsuka 2014; Lawry et al. 2017; Gebremedhin and Swinton 2003; Enki et al. 2001; Hajjar et al. 2012). There are likely to be ***large co-benefits*** for food security, as strong land tenure is positively correlated with food production increases (Maxwell and Wiebe 1999; Holden and Ghebru 2016; Corsi et al. 2017), although in some cases land formalisation has led to reduced food security for smallholders when they have been pushed to commercialise farming (Pritchard 2013). There are likely ***few adverse side-effects*** from land tenure security measures, although in some cases, formalisation can increase exclusion and has been associated with more confusion over land rights, not less, in some areas where poorly implemented (Broegaard et al. 2017). Barriers to stronger land security include lack of political will and the costs of adopting land formalisation programs (Deininger and Feder 2009).

6.9.2.12 Prevent / reverse soil salinisation

Soil salinisation is a major process of land degradation that decreases soil fertility, is a significant component of the desertification process for the world's drylands, and affects agricultural production, aquaculture and forestry. Prevention of soil salinisation can be achieved through improvement of water management including water-use efficiency and irrigation in arid/semi-arid area, improvement of soil health through increase in soil organic matter content and improving cropland management, agroforestry and conservation agriculture (Dagar et al. 2016a; Evans and Sadler 2008; He et al. 2015; UNCTAD 2011; DERM 2011; Rengasamy 2006; Baumhardt et al. 2015; Datta et al. 2000; Prathapar 1988). Techniques to prevent and reverse soil salinisation have ***small co-benefits*** for mitigation since they may benefit soil carbon sinks, ***moderate co-benefits*** for adaptation, since they allow existing crop systems to be maintained and crop shifting to be reduced (UNCTAD 2011; Dagar et al. 2016a), and can deliver a ***large co-benefits*** for prevention and reversal of desertification (Chapter 3, section 3.6, section 6.4-6.8 and section 6.9 this Chapter, (Rengasamy 2006; Dagar et al. 2016a) and ***large co-benefits*** for prevention and reversal of land degradation, since soil salinisation is a main driver of both desertification and land degradation in the world's drylands (Chapter 4, section 4.8, section 6.4-6.8 and section 6.9 this Chapter, (Rengasamy 2006; Dagar et al. 2016a). Prevention of soil salinisation delivers ***moderate co-benefits*** for food security by maintaining existing crop systems, and helping to close yield gaps in rainfed crops (Chapter 5, section 5.8, section 6.4-6.8 and section 6.9 this Chapter). There are likely to be ***few adverse side-effects***, apart from potential additional fossil fuel use for irrigation or increasing the efficiency of water use usually means reduced yields at some level. Barriers depend on how salinisation and sodification are tackled, but can include biophysical (e.g. lack of alternative water sources; (Bhattacharyya et al. 2015; Dagar et al. 2016a)), technological (e.g. lack of appropriate irrigation technology; (Machado and Serralheiro 2017; CGIAR 2016; Bhattacharyya et al.

2015), institutional (lack of alternative irrigation infrastructure; (Evans and Sadler 2008; CGIAR 2016), educational (poor knowledge of the causes and salinisation and how to address it; (Greene et al. 2016; Dagar et al. 2016a), and cultural / behavioural (persistence of traditional practices; (Greene et al. 2016; Dagar et al. 2016a).

6.9.2.13 Prevention of land grabbing

The primary mechanisms for combatting large scale land acquisitions ('land grabs') have included restrictions on the size of land sales (Fairbairn 2015); pressure on agribusiness companies to agree to voluntary guidelines and principles for responsible investment (Collins 2014; Goetz 2013); and repeals of biofuels standards that have contributed to expansion of lands (Palmer 2014). Preventing land grabbing has ***small co-benefits for mitigation,*** namely due to avoiding conversion of forests to agriculture, although there is no quantification available of how much land has been converted due to land grabs in recent years; however, because many land investments in African countries in particular exceed the documented cultivable land area for the country, forest, wetlands and grasslands have likely been converted, leading to increased GHG emissions (D'Odorico et al. 2017; Balehegn 2015). Preventing land grabbing leads to ***large co-benefits*** for adaptation as land grabbing threatens local systems of land management (Vermeulen and Cotula 2010), which can make some communities less able to adapt over time (Gabay and Alam 2017; Dell'Angelo et al. 2017; Kenney-Lazar 2012). Such land grabs have occurred on nearly 45 Mha of land, likely impacting tens of millions of people. Preventing land grabbing can also deliver ***moderate co-benefits*** for prevention and reversal of land degradation and ***small co-benefits*** for prevention and reversal of desertification, as many large-scale investments intensify unsustainable lands uses, leading to soil degradation (Friis and Nielsen 2016; Balehegn 2015). ***Large co-benefits*** for food security are likely to be realised; for example, the food price crisis of 2008 was linked to expanding biofuel land deals (Kugelman and Levenstein 2013; Bush and Martiniello 2017). There is some evidence that land grabbing has already led to the impoverishment of up to 12 million people (Adnan 2013; Davis et al. 2014). In at least some cases, the causal process is that land grabs contribute to increased tenure insecurity in surrounding lands, leading farmers to shift to cultivating smaller farms with less investments, potentially leading to food shortages (Aha and Ayitey 2017). There are likely ***no adverse side-effects*** from preventing land grabbing measures. Barriers to policies against land grabbing include agribusiness opposition, as they are the main funders of such investments, and lack of political will to enact policies to reduce agricultural investments among poor country governments (The World Bank 2011).

6.9.2.14 Prevention of compaction

Soil compaction is related to degradation of soil structure through the deformation of the soil surface due to external stresses applied to the soil (e.g. agricultural traffic, tillage, livestock trampling, etc.). As a result of soil compaction, higher bulk density and soil strength values, and lower water infiltration rates and hydraulic conductivity values are usually found in compacted soils compared to non-compacted soils. Excessive soil compaction may reach up to 50 cm soil depth, hindering plant root growth in this layer and below and increasing N losses in anaerobic zones (Soane and van Ouwerkerk 1994; Hamza and Anderson 2005; FAO and ITPS 2015). Although soil compaction alleviation may be natural, it can be promoted and accelerated if the following practices are implemented (FAO 2017): a) the avoidance of excessive and inappropriate tillage, b) the minimisation of vehicular traffic, particularly in bare soils, c) the adjustment of machines and vehicles used in the field to soil strength, d) the selection of cropping systems with strong tap roots able to penetrate and break up compacted soils, e) the maintenance of adequate amount of soil organic matter to improve and stabilise soil structure , f) the promotion of macrofauna and microbial (especially fungal) activity, and g) sufficient cover by growing plants in grazing systems. These prevention and reversion techniques have ***small co-benefits*** for mitigation due to variable impacts on GHG emissions, and ***small co-benefits*** for adaptation by improving soil climatic resilience (Tim Chamen et al. 2015; Epron et al. 2016; Tullberg et al. 2018).

Prevention of soil compaction can deliver ***large co-benefits*** for prevention and reversal of land degradation and ***large co-benefits*** for prevention and reversal of desertification, since soil compaction is a main driver of both desertification and land degradation (FAO and ITPS 2015). Prevention of compaction delivers ***moderate co-benefits*** for food security by helping to close yield gaps in rainfed crops (Anderson and Peters 2016). Implementation costs are not high since compaction avoidance technologies require less fuel and provide a win–win strategy for farmers and the environment (Tim Chamen et al. 2015). There are likely to be ***few adverse side-effects***. Although both compaction process and remediation technologies are well-known, barriers include biophysical (some soils are prone to compaction), technological (e.g. few decision support systems for implementation of precision management of traffic compaction) and educational (knowledge gaps; Antille et al. 2016).

6.9.2.15 Reduce post-harvest losses

Underlying post-harvest food losses, is a food system failure to equitably enable accessible and affordable food in all countries (Wilhelm et al. 2016b). Food loss in developed countries mostly occurs at the retail/consumer stage (Bajželj et al. 2014a; Göbel et al. 2015)– this is dealt with in section 6.9.2.15. In developing countries, food loss occurs mainly at the post-harvest stage, and less at consumption stage (Ritzema et al. 2017). The key drivers for post-harvest waste in developing countries are structural and infrastructure deficiencies (Chaboud and Daviron 2017; Sheahan and Barrett 2017). Thus reducing food waste at the post-harvest stage requires responses that process, preserve and, where appropriate, redistribute food to where it can be consumed immediately (Bajželj et al. 2014a; Ritzema et al. 2017). Differences exist between farm food waste reduction technologies between small-scale agricultural systems and large scale agricultural systems (Ansah et al. 2017; Hengsdijk and de Boer 2017). A suite of options includes farm level storage facilities, trade or exchange processing technologies including food drying, onsite farm processing for value addition, and seed systems which take from harvests. For large scale agri-food systems, options include cold chains for preservation, processing for value addition and linkages to value chains that absorb the harvests almost instantly into the supply chain. In addition to the specific options to reduce food loss and waste, there are more systemic possibilities related to food systems. Regional and local food systems are now being promoted to enable production, distribution, access and affordability of food (Billen et al. 2018; Kissinger et al. 2018). Emissions attributable to food waste and loss amount to 3.3 $GtCO_{2e}yr^{-1}$ with a large share of this loss in developing countries (Chapter 5), so reducing post-harvest losses has ***large co-benefits*** for mitigation, though increased use of refrigeration could increase emissions (6.9.2.39 from energy use. Since reduced food losses reduces pressure on the land, there are also ***large co-benefits*** for adaptation. There are likely to be ***small co-benefits*** for prevention and reversal of land degaradation, and ***small co-benefits*** for prevention and reversal of desertification, both through reduced pressure on land. There are ***large co-benefits*** for food security, since most of the 30% of all food wasted globally arises from post harvest losses in developing countries (Ritzema et al. 2017). There are likely to be ***no adverse side-effects***. Barriers are largely institutional, since solutions may require dismantling and redesigning current food value chains, and technological barriers are lack of low cost storage and preservation technologies. There are few biophysical, educational or cultural barriers, since preventing food loss is a priority in many developing countries.

6.9.2.16 Reduce food waste (consumer or retailer)

Approximately 30% of all food is wasted (Kummu et al. 2012a; Gustavsson et al. 2011; Vermeulen et al. 2012). A reduction in food waste would provide ***large co-benefits*** for mitigation, by reducing greenhouse gas emissions (section 6.8.6), on the order of 5 $GtCO_2yr^{-1}$ (Bajželj et al. 2014a). Reducing food waste would have ***small co-benefits*** for desertification and ***large co-benefits*** for prevention or reversal of land degradation by reducing agricultural area on the order of 7 million km^2 (Bajželj et al. 2014a) and reducing production intensity (Muller et al. 2017). These decreases in pressure on land and decreases in production intensity could also have ***small co-benefits*** for adaptation (section 6.8.6; Muller et al., 2017). By definition, this would also deliver ***large co-benefits*** for food security (Chapter 5; section 6.8; Bajželj et al. 2014a). Reductions in food waste would also deliver ***moderate co-benefits*** for other sustainable development goals, including reducing water scarcity through reductions in irrigation water, reducing pollution through reductions in fertiliser use, and reducing biodiversity loss through reductions in agricultural area and fertiliser use (section 6.8.6; Bajželj et al. 2014a). There are likely to be ***few adverse side-effects*** across the challenges (Bajželj et al. 2014a; Muller et al. 2017). The main barriers to implementation are cultural / behavioural (e.g. potential for rebound effects; Smith et al. 2013), economic, and institutional (REF).

6.9.2.17 Promote value-added products

Value-added food production is a collection of practices that enable producers to increase the economic value or reduce risks of commodities through production processes (e.g. packaging, processing, cooling, drying, extracting). Adding value to products requires improved innovation, coordination and efficiency in the food supply chain (Chapter 5; section 6.8; Garnett 2011; Godfray et al. 2010; Hertel 2015). Promoting value-added products can provide ***large co-benefits*** for climate mitigation by making better use of food already grown and reducing the need for compensatory extensification of agricultural areas (Chapter 2; section 6.4; Bajželj et al. 2014a); energy use in pre and post-harvesting processes (Chapter 5; section 6.8; Accorsi et al. 2017); and emissions from food loss and waste (e.g. methane from landfills) (Ingram et al. 2016; James and James 2010). This response option also provides ***large co-benefits*** for climate adaptation by diversifying and increasing flexibility in the food system to climate stressors and shocks while simultaneously creating economic alternatives for the poor (thereby strengthening adaptive capacity) and lowering expenditures of food processors and retailers by reducing losses (Chapter 5; section 6.5; Muller et al. 2017). Adding value to products can extend a producer's marketing season and provide unique opportunities to capture niche markets thereby increasing their adaptive capacity to climate change (Evans, 2009; Anderson and Hanselka, 2009; Brees, Parcell, and Giddens, 2010). Promoting value-added product would deliver ***large co-benefits*** for food security (Chapter 5; section 6.8; Tilman and Clark 2014). Diversifying markets and developing value-added products in the food supply system improves food security by increasing its economic performance and revenues to local farmers (Reidsma et al. 2010) and strengthens the capacity of the food production chains to adapt to future markets with more stringent climate regulation and improve income of smallholder farmers, increasing their food security (Murthy and Madhava Naidu 2012). Value-added products may also have positive impact when the overall efficiency of the food supply chain (Niles et al., 2018) and can create closer and more direct links between producers and consumers. There are ***small co-benefits*** for land degradation and desertification and there are likely to be ***few adverse side-effects*** across the challenges (Chapter 3: section 6.6; Clark and Tilman 2017) except in cases where processing of value-added products lead to higher emissions or demand of resources in the food system. Reversibility could be an issue and while there are low cost options, the implementations can be expensive. Developing technical knowledge and building capacity in value-added processing and logistics systems, targeting smallholder farmers and small and medium enterprises can help to overcome inherent technological and educational barriers. While there are no obvious biophysical or cultural barriers, there are institutional barriers in some contexts (e.g. in low income African, Asian and Latin American countries where challenges associated with food insecurity and climate change vulnerability are more acute) (Ingram et al. 2016). Strengthening Institutional and organisational innovation capacities and value-chain collaboration (Mirjam et al.) can enhance the effectiveness of strategies to promote value-added products (Capone et al. 2014).

6.9.2.18 Management of urban sprawl

Unplanned urbanisation leading to sprawl and extensification of cities along the rural-urban fringe has been pointed to as a driver of agricultural land loss. China has lost 3%-5% of productive farmlands to industrial and urban development in recent years (Chen 2007; Cai et al. 2013), and the US is on a similar trajectory (Francis et al. 2012), while in India, more urban land is reclaimed from woodlands and grassland than from cropland (Gibson et al. 2015). Policies to prevent such urbanisation have included land use planning, agricultural zoning ordinances and agricultural districts, urban redevelopment, arable land reclamation, and transfer/purchase of development rights (Tan et al. 2009; Qian et al. 2015); China in particular has a strict national Requisition–Compensation Balance of Farmland policy requiring balance in expropriations of farmland (Shen et al. 2017). Such policies promoting densification and less haphazard development are estimated to have the potential to save 62,000 km^2 of arable land by 2030 in India alone (Gibson et al. 2015). Preventing urban sprawl has ***moderate co-benefits*** for mitigation, namely due to the fact that extensive and less dense urban developments tend to have higher energy usage (Liu et al. 2015), such that a 10% reduction of very low density urban fabrics is correlated with 9% fewer emissions per capita in Europe (Baur et al 2015). However, the exact contribution to mitigation from the prevention of land conversion in particular has not been well quantified; suggestions from select studies in the US are that biomass decreases by half in cases of conversion from forest to urban land uses (Briber et al. 2015), and a study in Bangkok found a decline by half in carbon sinks in the urban area in the past 30 years (Ali et al. 2018). Managing urban expansion offers ***moderate co-benefits*** for adaptation. In particular, reducing urban sprawl is likely to provide adaptation co-benefits *via* improved human health, as noted earlier (Anderson, 2017), although other adaption co-benefits are less well understood. It can also deliver ***small co-benefits*** for prevention and reversal of desertification and ***large co-benefits*** for prevention and reversal of land degradation. Urban expansion has been pointed to as the culprit in major soil degradation in China, for example, affecting 20 million ha, almost one-sixth of the cultivated land total, and causing an annual grain yield loss of 10 million tons. Pollution from urban development has included water and soil pollution from industry and wastes and sewage as well as acid deposition from increasing energy use in cities (Chen 2007), all resulting in major losses to ecosystem services from urban conversion (Song and Deng 2015). Managing urban sprawl offers ***large co-benefits*** for food security as concerns about urban expansion threatening productive agricultural lands are longstanding. Evidence in the US indicates ambiguous trends; on one hand most urban expansion in the US has primarily been on lands of low and moderate soil productivity with only 6% of total urban land on highly productive soil. On the other hand, highly productive soils were experiencing the highest rate of conversion of any soil type (Nizeyimana et al. 2001). Specific types of agriculture are often practiced in urban-influenced fringes, such as fruits, vegetables, and poultry and eggs in the US, the loss of which can have an impact on the types of nutritious foods available in urban areas (Francis et al 2012). ***Adverse side-effects*** from managing urbanisation may include increased prices for housing if more expensive densification is pursued versus often cheaper extensification. Barriers to policies against urban sprawl include institutional barriers to integrated land use planning and the costs to national governments of restricting or buying back development rights (Tan et al. 2009).

6.9.2.19 Fire management

Fire management is a land management option aiming at the safeguarding of life, property and resources through the prevention, detection, control, restriction and suppression of fire in forest and other vegetation in rural areas (FAO 2006). It includes improved use of fire for sustainable forestry management, including wildfire prevention and improved prescribed burning (Smith et al. 2014b). Prescribed burning is used to reduce the risk of large, uncontrollable fires breaking out in the forest areas, and controlled burning is among the most effective and economical methods of reducing fire danger and stimulating a natural reforestation process under the forest canopy and after clear felling (Valendik et al., 2010). The frequency and severity of large wildfires has increased around the globe in past decades, and it strongly impacts forest carbon budgets (Seidl et al. 2014; Westerling et al. 2006). For example, the disturbance-related reduction (including wildfires, pests and wind) of the carbon storage potential in Europe's forests is estimated to be 503.4 Tg C in 2021–2030 (Seidl et al. 2014).

Fire management provides ***large co-benefits*** for climate mitigation by reduced size, severity, and frequency of wildfires (Chapter 2; section 6.4; Whitehead et al. 2008), ***large co-benefits*** for climate adaptation by improving the resilience of forestry ecosystems to future severe disturbances (Chapter 2; section 6.4; (Locatelli et al. 2015b; Jandl et al. 2015), ***large co-benefits*** for prevention or reversal of desertification by increased control of wildfires and long-term maintenance of tree stock density to protect against soil erosion (Chapter 3; section 6.6; Freeman et al. 2017) and ***large co-benefits*** for prevention or reversal of land degradation by stabilising forest ecosystems (Chapter 4; section 6.7; Freeman et al. 2017). Forest fire management can guarantee forest products availability and prevention of fire expansion to agricultural land, so it has ***moderate co-benefits*** also for food security (Chapter 5; section 6.8). ***Few adverse side-effects*** are also expected, for example prescribed burning can cause emissions of GHGs (methane) and short-lived climate pollutants (Chapter 2), including its adverse health effects and economic impacts to transport and tourism (Lin et al. 2017). The carbon sink in forest both saturates and is reversible due intrinsic nature of forest ecosystems. If fire management practices are discontinued, the beneficial impacts may also cease. Technologies for fire management exist, but the cost of its implementation is relatively moderate, since it requires constant maintenance (North et al. 2015), and can be excessive for some local communities. Barriers to implementation include biophysical (e.g. susceptibility to climate and other unpredicted events; Hurteau et al. 2014 or steep or remote areas to its application; North et al. 2015), technological, institutional (e.g. lacks of social or political acceptance; Freeman et al. 2017) and educational (e.g. poor knowledge of best practices, liability issues, casualty risks and little tolerance for management errors; North et al. 2015).

6.9.2.20 Management of landslides and natural hazards

Landslides and natural hazards (e.g. floods, storm surges, droughts) are due to intentional, non-malicious human activities, such as subsurface material extraction, subsurface material addition, land use change, surface material extraction, surface material addition, hydrological change, explosion, and combustion (fire) (McGill and Malamud, 2017). Decision analysis for management typically involves estimating the probability of extreme events, assessing the potential impact of those events from a variety of perspectives, and evaluating options to plan for, mitigate, or react to events (Simpson et al. 2016), but nevertheless multi-risk approaches do not often consider the effects of climate change and mostly rely on the analysis of static vulnerability (i.e. no time-dependent vulnerabilities, no changes among exposed elements; (Gallina et al. 2016). The prevention and management of landslides and natural hazards has ***small co-benefits*** for mitigation, because of little impact of GHG emissions and eventual preservation of topsoil carbon stores, but ***large co-benefits*** for adaptation, since they are structural/physical options for climate change adaptation (IPCC AR5 WG2, Chapter 14; Noble et al. 2014). In the same way, they can deliver ***small co-benefits*** for desertification, but ***large co-benefits*** for land degradation, since landslides and natural hazards are among the most severe degradation processes (FAO and ITPS 2015). In countries in which mountain slopes are cropped for food crops, such as the case of Pacific Islands (Campbell 2015), the management and prevention of landslides can deliver ***moderate co-benefits*** for food security. There are few ***adverse side effects*** from measures to reduce the risk oflandslides and natural hazards. Most of the deaths caused due to different disasters have occurred in developing countries, in which poverty, poor education and health facilities and other aspects of human population increase exposure and high levels of vulnerability and risk (Mal et al. 2018). In the tropics, the most cited barriers for implementing landslide risk reduction measures are scientific and political in nature, and the ratio of implemented versus recommended landslide risk reduction measures is low for most landslide risk reduction components (Maes et al. 2017). The implementation of practices for management of landslides and natural hazards is based on engineering works and more resilient cropping systems (Noble et al. 2014; McGill and Malamud, 2017), which are is often limited by their high costs, as well as biophysical, technological and educational barriers.

6.9.2.21 Ecosystem-based adaptation

Ecosystem-based adaptation (EbA) involves the use of natural ecosystems and capital to assist in adaptation to climate change, based on the principle that intact and healthy ecosystems are more resilient to climate stressors (Scarano 2017). EbA is promoted as a win-win for mitigation, adaptation and other benefits with low adverse side effects. Response options within EbA include targeted management, conservation, and restoration activities, such as protecting or increasing the extent of an ecosystem, enhancing connectivity, or reducing stressors on ecosystems. Examples of benefits from such actions include mangrove protection leading to buffers against storm surges or protecting floodplains to recharge groundwater supplies and provide low-cost wastewater treatment (Ojea 2015; Munang et al. 2013a). EbA has ***moderate co-benefits for mitigation,*** namely due to protection of existing ecosystems which may serve as carbon sinks (mangroves, grasslands, wetlands, etc) (Jones et al. 2012). Ecosystem-based adaptation offers ***large co-benefits*** for adaptation, as by its very name EbA involves the use of ecosystems to increase adaptive capacity. EbA is often based on local, available and renewable inputs, and is asserted to be more cost-effective than other approaches to adaptation (such as hard infrastructure) (Jones et al. 2012; Ojea 2015), as well as being more flexible, less path dependent and potentially reversible (van Wesenbeeck et al. 2014; Jones et al. 2012). EbA can also deliver ***large co-benefits*** for prevention and reversal of desertification and ***large co-benefits*** for prevention and reversal of land degradation, as EbA involves ecologically-based management practices known to reduce degradation, such as management of trees in agroforestry or silvopastoral systems; use of integrated pest-management strategies to reduce chemical pesticide use; use of mulching or cover crops to improve soil moisture; planting of windbreaks to help reduce impacts on soil structure; or conservation of riparian vegetation to ensure water provision, among others (Vignola et al. 2015). EbA also promises ***large co-benefits*** for food security; diverse farm systems based on agroforestry systems can increase food security of smallholders and improve income generation (Vignola et al. 2015). ***Adverse side-effects*** from EbA include potentially lower yields in agricultural systems adopting EbA and trade-offs between long term improvements and short-term labour and other costs (Vignola et al. 2015). Barriers to EbA include lack of information about what roles ecosystems play, trade-offs for farmers in terms of labor and benefits, and challenges in scalability and governance as EbA is challenging for siloed policy systems (Scarano 2017; Vignola et al. 2013; Ojea 2015; Burch et al. 2014).

6.9.2.22 Reduced deforestation

Reduced deforestation includes conservation of existing carbon pools in forest vegetation and soil by controlling deforestation, protecting forest in reserves, controlling other anthropogenic disturbances such as fire and pest outbreaks, reducing slash and burn agriculture, protection of peatland forest, and reduction of wildfires (Smith et al. 2014b). Currently emissions from AFOLU Sections accounts for 25 % (about 10-12 $GtCO_2yr^{-1}$) of net anthropogenic emissions, and deforestation alone contributes approximately 10% of all human-induced GHG emissions (Chapter 1, Box 1.1). Reduced deforestation provides ***large co-benefits*** for climate mitigation by maintaining the carbon sequestration in forest ecosystems (Chapter 2; section 6.4; Pan et al., 2011), ***large co-benefits*** for climate adaptation from continued provision of local ecosystem services, including climate regulation (Locatelli et al. 2015b) (Chapter 2; section 6.5; Kongsager et al., 2016; Reyer et al., 2009), ***large co-benefits*** for prevention or reversal of desertification by maintaining forest ecosystems in sensible areas (Chapter 3; section 6.6; Nasiru Idris et al., 2010; Salvati et al., 2014), ***large co-benefits*** for prevention of land degradation through maintenance of perennial vegetation, (Chapter 4; section 6.7; Ellison et al. 2017). However, considerable ***large adverse side-effects*** are expected in food security due to potential land competition with food production (Chapter 5; section 6.8; Frank et al. 2017). The carbon stock in the forest is prone to both reversibility and saturation. The reduced deforestation practices have relatively moderate costs, but it requires transaction and administration costs (Overmars et al. 2014; Kindermann et al. 2008). Barriers to its implementation include biophysical (e.g. susceptibility to climate and other unpredicted events; Ellison et al. 2017), institutional (e.g. land tenure, economic disincentives and transaction costs; Kindermann et al. 2008), educational (e.g. little information available in some regions) and cultural (different realities, e.g., small holder versus industrial production).

6.9.2.23 Livelihood diversification

Livelihood diversification can help reduce climate vulnerability given that "diverse patterns of resource use and heterogeneity of income sources increases robustness and adaptive capacity in social-ecological systems by helping spread risk in the case of severe disturbances" (DiGiano and Racelis 2012). Diversification has ***small co-benefits*** for mitigation, namely due to a lack of linkages between diversification and specific GHG reduction measures (Wise et al. 2016), although diversification into agroforestry is likely to have ***small co-benefits*** for mitigation (Altieri et al. 2015; Descheemaeker et al. 2016). Diversification offers ***large co-benefits*** for adaptation as it can help households smooth out income fluctuations and provide a broader range of options for the future (Thornton and Herrero 2014). Surveys of farmers in climate variable areas find that livelihood diversification is increasingly favoured as an adaptation option (Van Aelst and Holvoet 2016). Diversification is likely to deliver ***small co-benefits*** for prevention and reversal of desertification ***and moderate co-benefits*** for prevention and reversal of land degradation, as diversification may involves adding non-traditional crops or trees that may reduce the need for tillage (Antwi-Agyei et al. 2014). Diversification offers ***large co-benefits for food security***, as diversification has been linked to increasing incomes and decreasing the probability of poverty in several country studies (Arslan et al. 2018; Asfaw et al. 2018). A***dverse side-effects*** from diversification are minimal. Barriers to diversification include the fact that poorer households and female headed households may lack assets to invest in new income streams or have a lack of education about new income sources (Berman et al. 2012; Van Aelst and Holvoet 2016; Ngigi et al. 2017).

6.9.2.24 Promotion of seed sovereignty

Seed sovereignty refers to movements to retain control over "people's right to save, replant, breed and share seeds, and their right to participate in decision-making processes regarding rules and laws that regulate their access and use" (Wattnem 2016). Options for seed sovereignty include farmer seed networks and community seed banks; open source plant breeding and use of open pollinated seeds; declaration of GM-free zones; and educational programs (Kloppenberg 2010; Luby et al. 2015; Bowman 2015; Campbell and Veteto 2015b; Reisman 2017; Patnaik et al. 2017). Seed sovereignty has ***small co-benefits*** for mitigation, namely due to the lack of explicit linkages between local seeds and GHG emission reductions, although use of local seeds likely reduces emissions associated with transport for commercial seeds (however minimal). Seed sovereignty offers ***large co-benefits*** for adaptation, given that from 60 to 100% of seeds used in various countries of the global South are likely local farmer-bred (noncommercial) seed (Louwaars 2002; Santilit 2012), and moving to use of commercial seed would increase costs considerably for these farmers (Howard 2015). Seed networks and banks protect local agrobiodiversity and landraces, which are important to facilitate adaptation, and can provide crucial lifelines when crop harvests fail (Coomes et al. 2015; van Niekerk and Wynberg 2017; Vasconcelos et al. 2013); for example, problems of seed scarcity and dependence on outside supplies can be overcome by local control over seeds (Reisman 2017). Seed sovereignty is likely to deliver ***small co-benefits*** for prevention and reversal of desertification and ***small co-benefits*** for prevention and reversal of land degradation namely due to the likelihood of local seeds as being less dependent on inputs like chemical fertilisers or mechanical tillage; for example, in India, local legumes are retained in seed networks while commercial crops like sorghum and rice dominate food markets (Reisman 2017a). Seed sovereignty provides ***large co-benefits*** for food security because of the increased ability of farmers to revive and strengthen local food systems; several studies have reported more diverse and healthy food in areas with strong food sovereignty networks (Coomes et al. 2015; Bisht et al. 2018). Women in particular may benefit from seed banks for low value but nutritious crops (Patnaik et al. 2017). ***Adverse side-effects*** from seed sovereignty are minimal. Barriers to seed sovereignty include concerns about equitability in access to seed networks and the difficulty of sustaining such projects when development donors leave (Reisman 2017), and disputes over the intellectual property rights associated with seeds (Timmermann and Robaey 2016).

6.9.2.25 Management of pollution including acidification

Total emissions from the fires have been in the order of 1.75 $GtCO_2$ (Tacconi, 2016) and there are important synergies between air pollution and climate change control policies. Acid deposition can destroy trees and other vegetation (ref) so managing acid deposition would be expected to provide ***small co-benefits*** for mitigation with respect to preserving or enhancing carbon sinks (ref), and since controlling acid deposition improves ecosystem health, it would provide ***moderate co-benefits*** for adaptation. Acid deposition is a significant driver of land degradation (Smith et al. 2015), so its management would be expected to deliver ***large co-benefits*** for prevention and reversal of land degradation (Smith et al. 2015), and may provide ***small co-benefits*** for prevention and reversal of desertification where acid deposition is a stressor. Since acid deposition is harmful to crops (ref), its management would be expected to increase production, thereby producing ***moderate co-benefits*** for food security. Management of air pollution through a combination of stringent policies on air pollution control and climate change mitigation could provide ***large co-benefits*** to human health with projected improvements to those exposed to PM levels below the WHO air quality guideline; with the largest improvements estimated for India, China, and Middle East (Rao et al. 2016). West et al. (2013) estimated the global GHG mitigation would avoid 0.5±0.2, 1.3±0.5, and 2.2±0.8 million premature deaths in 2030, 2050 and 2100, and Anenberg et al. (2012) estimated GHG mitigation can reduce global mean PM2.5 by 23–34%, avoiding 0.6-4.4 million premature deaths per year by 2030, with more than 80% of the health benefits occurring in Asia. The costs of reducing greenhouse gas emissions could be compensated with the health co-benefits alone for China and India, whereas the proportion of health co-benefits varied but could be substantial in the European Union (7–84%) and USA (10–41%), respectively (Markandya et al. 2018). There are likely to be ***few adverse side-effects***, though atmospheric nitrogen deposition can be an important source of nitrogen in low input agriculture and forestry, so reducing emissions could have small negative impacts on crop and tree growth (ref). Barriers to implementation are mainly biophysical (since acid deposition is transboundary, so sources are often far distant from the site of impact; ref) technological (e.g. lack of technology to inject liquid fertilisers below ground to prevent ammonia emissions; (Sutton et al. 2007), institutional (e.g. poor regulation and enforcement of environmental regulations; ref), educational (e.g. poor knowledge among farmers about how to reduce ammonia emissions; ref) and behavioural / cultural (e.g. desire to maintain traditional practices, ref).

6.9.2.26 Management of invasive species

Agricultural and forests can be very high in diversity but much of it is often non-native. Invasive species in different biomes have been introduced through intended and unintended processes of exportation of ornamental plants or animals, and many times through the promotion of modern agriculture and forestry. Non-native species tend to be more numerous in larger than smaller human-modified landscapes (e.g. over 50% of species in an urbanised area or extensive agricultural field can be non-native). Management of invasive species can be done through manual clearance of invasive species, which has been done in many landscapes, while in some areas natural enemies of the invasive species are introduced to control them (Dresner et al. 2015). There are potentially ***small co-benefits*** for mitigation if removal of invasive species allows higher productivity, thereby increasing carbon stocks or sparing land, and ***small co-benefits*** for adaptation if removal of invasive species allows for production of crops and livestock, improving resilience to climate change. The resilience of rural livelihoods can also be improved since manual clearance can create job opportunities for the local population, local populations can breed indigenous plants and secure investments in nurseries. In some places, the replanting of the original species is tied with education and public campaigns to ensure ownership and future protection from introduction of non-native species. In this respect, the co-benefits of manual clearance of invasive species are higher than introduction of natural enemies of the invasive species. Since invasive species are a primary cause of land degradation, management of invasive species will deliver ***large co-benefits*** to preventon and reversal of land degradation, and in arid areas could provide ***large co-benefits*** for preventon and reversal of desertification though similar mechanisms. There may also be ***moderate co-benefits*** for food security in cases where the invasive species are surpressing food production. There are ***no adverse side-effects*** though natural enemies need to be well targeted so that they do not present similar problems to the invasive sprecies (ref). Barriers are partly biophysical, since restoration programmes can take a long time, and in the case of natural enemies can be technological (Dresner et al. 2015). Education can be a barrier, where populations are unaware of the damamge caused by the invasive species (ref), and institution barriers occur where agricultural extension and advice services are poorly developed (ref). Cultural / behavioural barriers are likely to be small (ref).

6.9.2.27 Reforestation

Reforestation is conversion of recently non-forested land to forest, often with a conservation or landscape protection background, generally focussing on restoration of ‘‘nature-like’’ ecosystems (Reyer et al. 2009). Reforestation also includes improved biomass stocks by planting trees on non-forested agricultural lands that were previously forested (see also afforestation for non-forest land) and can include either monocultures or mixed species plantings (Smith et al. 2014b). Reforestation provides ***large co-benefits*** for climate mitigation by rebuilding the carbon sequestration stocks in forest ecosystems, although decreases in surface albedo can reduce the net climate benefits in areas affected by seasonal snow cover (Chapter 2; section 6.4; Sonntag et al. 2016; Mahmood et al. 2014). There are ***large co-benefits*** for climate adaptation by increasing provision of forest ecosystem services (section 6.5; Locatelli et al. 2011b; Reyer et al. 2009), ***large co-benefits*** for prevention or reversal of desertification by restoring forest ecosystems in sensible areas (Chapter 3; section 6.6; Nasiru Idris et al., 2010; Salvati et al. 2014), and ***large co-benefits*** for reversal of land degradation through reestablishment of perennial vegetation (Chapter 4; section 6.7; Ellison et al. 2017). However, considerable ***adverse side-effects*** are expected in food security due to potential land competition with food production (Chapter 5; section 6.8; Frank et al. 2017b). The carbon sink provided by forest both saturates and is reversible. The reforestation practices have relatively moderate costs (Strengers et al. 2008). Barriers to its implementation include biophysical (e.g. availability of native species seedlings for planting), institutional, educational (e.g. low genetic diversity of planted forests) and cultural (e.g. care of forest cultures).

6.9.2.28 Restoration and avoid conversion of coastal wetlands

Coastal wetland restoration involves restoring degraded / damaged coastal wetlands including mangroves, salt marshes and seagrass ecosystems, which has the capacity to increase carbon sinks (Griscom et al. 2017a). Coastal wetland restoration provides ***large co-benefits*** for climate mitigation, with avoided coastal wetland impacts and coastal wetland restoration estimated to deliver 0.3 and 0.8 (total 1.1) $GtCO_2yr^{-1}$, respectively, by 2030 (Section 6.4.4; Griscom et al. 2017). Coastal wetland restoration may also provide ***large co-benefits*** for climate adaptation by regulating water flow and preventing downstream flooding (Section 6.4.4; Munang et al. 2014). There are likely ***no co-benefits*** (nor adverse side effects) of of coastal wetland restoration for prevention of desertification (Section 6.3), since these do not occur in in arid areas. Since large areas of global coastal wetlands are degraded (Lotze et al., 2006; Griscom et al. 2017), restoration provides ***large co-benefits*** for preventing and reversing land degradation (Section 6.4.4). Since large areas of coastal wetlands are used for food production (e.g. mangroves converted for aquaculture; Naylor et al. 2000), restoration could displace food production and damage local food supply (Section 6.4.4), potentially leading to a ***small adverse side-effect*** for food security globally, though the impact may be more significant in the affected areas.

This could be offset by more careful management, such as the careful siting of ponds within mangroves (Naylor et al. 2000). The carbon sink associated with coastal wetland restoration is reversible, though saturation is likely to take many decades (Griscom et al. 2017a). Costs for coastal wetland restoration projects vary, but they can be cost-effective at scale (Erwin, 2009). Barriers to implementation include biophysical (e.g. loss of large predators, herbivores, spawning and nursery habitat; Lotze et al., 2006), can be institutional in some regions (e.g. poor governance of wetland use in some regions; Lotze et al., 2006), and educational (e.g. lack of knowledge of impact of wetland conversion), though technological and cultural / behavioural barriers are likely to be small compared to other barriers.

6.9.2.29 Stability of food supply

Increasing the stability of food supplies is a key goal to increase food security, given that climate change threatens to lead to more production shocks in the future (Wheeler and von Braun 2013). Such shocks have in the past led to high volatility in global commodity markets (Lewis and Witham 2012). Measures to improve stability of food supply in traded markets can include a range of options, such as: 1) financial and trade policies, such as reductions on food taxes and import tariffs; 2) shortening food supply chains (SFSCs); 3) increasing food production; 4) designing alternative distribution networks; 5) increasing food market transparency and reducing speculation in futures markets; 6) increasing storage options; and 7) increasing subsidies and food-based safety nets (Mundler and Rumpus 2012; Barthel and Isendahl 2013; Wodon and Zaman 2010; Michelini et al. 2018; Minot 2014; Tadesse et al 2014). Stability of food supply has ***low to no co-benefits*** for mitigation, as food distribution is unlikely to be combined with GHG emissions reductions (and may actually increase them if food is being distributed more widely to meet stability demands) (Vermeulen et al. 2012). Food supply stability offers ***large co-benefits*** for adaptation, because when households are faced with negative shocks to food supplies, as may happen with price increases or volatility of production, they may sell other productive assets leading to declines in long term livelihoods (Fafchamps et al. 1998). Further, coping with higher food prices associated with food instability reduces income available for other adaptation options, especially for the poor (Haggblade et al. 2017). Food supply stability is likely to deliver ***small co-benefits*** for prevention and reversal of desertification and ***small co-benefits*** for prevention and reversal of land degradation, as reducing volatility in prices and supply may decrease pressure to expand production into marginal lands to make up shortfalls. There are ***large co-benefits*** for food security because there are clear links between higher food prices as a result of volatility, leading to lower caloric intake and lower quality diet, eventually leading to increases in child malnutrition in particular, which have affected millions in recent decades (Vellakkal et al. 2015; Arndt et al. 2016). Shifts in food availability and stability of food supply, caused in part by export bans and competition with land for biofuels, likely led to the 2007-2008 food price shocks that negatively affected food security for millions around the globe, and particularly in Sub-Saharan Africa (Wodon and Zaman 2010; Haggblade et al. 2017). ***Adverse side-effects*** from food stability policies are likely to be minimal and depend on the particular policies put in place to regulate issues like speculation in futures markets. Barriers to tackling food supply stability include political will within trade regimes, economic laissez-faire policies that discourage interventions in markets, and the difficulties of coordination across economic sectors (Poulton et al. 2006; Cohen et al. 2009; Gilbert 2012).

6.9.2.30 Biochar

Biochar is carbonised organic material produced by burning biomass in a low oxygen environment (pyrolysis). Biochar is added to soils as an amendment to improve fertility and to increase carbon stocks (Woolf et al. 2010). Biochar can be added to soils across a range different land uses, including cropland, grazing land and forestry – and can form part of cropland, grazing land and forest management (Smith, 2016). Use of biochar as a soil amendment provides ***large co-benefits*** for climate mitigation by creating soil carbon sinks in the 3-5 $GtCO_2yr^{-1}$ range (Chapter 2; section 6.4; Smith, 2016), ***large co-benefits*** for climate adaptation by improving the resilience of food crop production systems to future climate change by increasing yield in some regions and improving water holding capacity (Chapter 2; Section 6.5; Woolf et al. 2010; Sohi, 2012), potentially ***small co-benefits*** for prevention or reversal of desertification due to relatively small land areas (Chapter 3; section 6.6), and ***moderate co-benefits*** for prevention or reversal of land degradation, in both cases by improving water holding capacity, improving nutrient use efficiency, managing heavy metal pollution and other co-benefits (section 4.9.5.2; section 6.7; Sohi, 2012). There are, on balance, ***moderate co-benefits*** for food security from improved yield in the tropics but less so in temperate regions (Jeffery et al., 2017), through improved water holding capacity and nutrient use efficiency (Chapter 5; section 6.8; Sohi, 2012), though these co-benefits could be tempered by additional pressure on land if large quantities of biomass are required as feedstock for biochar production, causing potential conflicts with food security (Smith, 2016). There are ***few adverse side-effects*** across the challenges, other than the land requirement for biomass feedstock (Smith, 2016). The biochar carbon sink is thought to be less reversible than soil organic matter sinks (Smith, 2016), though there is mixed evidence about its residence time (Sohi, 2012). The biochar sink would also be expected to be less susceptible to saturation (Sohi, 2012). Use of biochar can be a low cost option, and can be cost negative depending on markets for the biochar as a soil amendment (Shackley et al. 2011; Meyer et al., 2011; Dickinson et al., 2014). Barriers to implementation include biophysical (e.g. land available for biomass production; (Woolf et al. 2010), technological (e.g. feedstock and pyrolysis temperature have large impacts on biochar properties; ref), can be institutional in some regions (e.g. lack of quality standards; Guo et al., 2016), educational (e.g. low awareness among end users; Guo et al., 2016), and cultural / behavioural (Guo et al., 2016).

6.9.2.31 Peatland restoration

Peatland restoration involves restoring degraded / damaged peatlands which both increases carbon sinks, but also avoids the ongoing CO_2 emissions from degraded peatlands, so it both prevents future emissions and creates a sink (Griscom et al. 2017a). Peatland restoration provides ***large co-benefits*** for climate mitigation, with avoided peat impacts and peat restoration estimated to deliver 0.7 and 0.8 (total 1.5) $GtCO_2yr^{-1}$, respectively, by 2030 (Section 6.4.4; Griscom et al. 2017), though there could be a temporary increase in methane emissions after restoration (Jauhiainen et al. 2008). It may also provide ***large co-benefits*** for climate adaptation by regulating water flow and preventing downstream flooding (Section 6.4.4; Munang et al. 2014). There are likely ***no co-benefits*** (nor adverse side effects) of peatland restoration for prevention of desertification (Section 6.3), as peatlands occur in wet areas and deserts in arid areas so there are not connected. Considering that large areas of global peatlands are degraded (Limpens et al., 2008), peatland restoration provides ***large co-benefits*** for preventing and reversing land degradation (Section 6.4.4). Since large areas of tropical peatlands and some northern peatlands have been drained and cleared for food production their restoration could displace food production and damage local food supply (Section 6.4.4), potentially leading to a ***small adverse side-effect*** for food security globally, though the impact may be more significant in the affected areas. Avoided emissions from peatlands are permanent upon restoration, but the carbon sink is reversible (Griscom et al. 2017a). Since peatlands continue to accumulate carbon over hundreds or thousands of years under suitable conditions, unlike mineral soils, the carbon sink does not saturate (Dommain et al., 2014). Direct CO_2 removal costs for wetland restoration range from USD 10-100/tCO_2 (Worrall et al., 2009), suggesting potential low-cost options for projects. Barriers to implementation include biophysical (e.g. site inaccessibility; Bonn et al. 2014), can be institutional in some regions (e.g. lack of inputs; Bonn et al. 2014), and educational (e.g. lack of skilled labour; Bonn et al. 2014), though technological and cultural / behavioural barriers are likely to be small compared to other barriers.

6.9.2.32 Afforestation

Afforestation includes improved biomass stocks by planting trees on non-forested agricultural lands that have not previously been forested (see also reforestation for planting on former forest land) and can include either monocultures or mixed species plantings (Smith et al. 2014b). Afforestation provides

large co-benefits to climate change mitigation, especially if occurring in the tropics and in areas that are not significantly affected by seasonal snow cover. There are ***large co-benefits*** for climate adaptation (Chapter 2; section 6.4; Kongsager et al., 2016; Reyer et al. 2009), ***large co-benefits*** for prevention or reversal of desertification by providing perennial vegetation in arid areas (Chapter 3; section 6.6; (Medugu et al. 2010; Salvati et al. 2014), ***large co-benefits*** for prevention or reversal of land degradation by stabilising soils through perennial vegetation (Chapter 4; section 6.7; Lal, 2001). Afforestation also has ***large co-benefits*** with a number of ecosystem services, as it increases carbon storage in biomass and soil organic matter, reduced erosion and improved regulation of flooding, improved water quality and increasing habitat provision to enhance biodiversity (Whitehead 2011). Afforestation also has the potential to filter out sediment and excess nutrients before entering streams (Newbold et al. 2010). The competition for land between afforestation/reforestation and agricultural production is a potential ***large adverse side-effect*** (Boysen et al. 2017a,b; Kreidenweis et al. 2016; Smith et al. 2013). Planting monocultures of non-native or native improved-growth species will likely yield greater carbon accumulation rates but ***adverse side-effect*** in terms of biodiversity loss. Under poor management, afforestation can result in a reduction of biodiversity in the local ecosystem, with introduction of potentially invasive and non-native species, reduced stream flow and loss of agricultural revenue (Cunningham et al. 2015). The carbon sink provided by afforestation both saturates and is reversible. The reduced deforestation practices have relatively low cost (Kreidenweis et al. 2016). Barriers to its implementation include biophysical, technological (e.g. achieve necessary rates of yields; Kreidenweis et al. 2016), institutional (e.g. policy makers commitment; (Medugu et al. 2010), educational and cultural.

6.9.2.33 Early warning systems for disaster risk reduction

Early warning systems (EWS) to enable disaster risk reduction (DRR) can include options such as 1) education systems; 2) hazard and risk maps; 3) hydrological and meteorological monitoring (such as flood forecasting or extreme weather warnings); 4) and communications systems to pass on information to enable action (Bouwer et al. 2014; Cools et al. 2016). EWS have ***no co-benefits*** for mitigation, namely due to such systems primarily being focused on adaptation. EWS offers ***large co-benefits*** for adaptation; for example, the Famine Early Warning System funded by the USAID has operated across 3 continents since the 1980s, and is praised for the timeliness, quantity, and quality of the warnings provided to countries, focusing on assessing agricultural changes due to climate/weather events, staple food prices, and health (Hillbruner and Moloney 2012). Such information can assist communities and households in adapting to onset conditions. However, concerns have been raised as to how many people are actually reached by such systems; for example, less than 50% of respondents in Bangladesh had heard a cyclone warning before it hit, even though an EWS existed (Mahmud and Prowse 2012). Further, there are concerns that current EWS systems "tend to focus on response and recovery rather than on addressing livelihood issues as part of the process of reducing underlying risk factors," (Birkmann et al 2013), leading to less adaptation potential realised. EWS are likely to deliver ***small co-benefits*** for prevention and reversal of desertification and ***small co-benefits*** for prevention and reversal of land degradation, as most warning systems are not focused on land use. Only examples like the Global Drought Early Warning System (GDEWS) (currently in development), which will monitor precipitation, soil moisture, evapotranspiration, river flows, groundwater, agricultural productivity and natural ecosystem health, may have some potential co-benefits to reduce degradation/desertification (Pozzi et al. 2013). There are ***moderate co-benefits*** for food security from EWS when such systems may be focused on warnings to help farmers harvest crops in advance of impending weather events or otherwise make agricultural decisions to prepare for adverse events (Fakhruddin et al. 2015). Surveys with farmers reporting food insecurity from climate impacts have indicated their strong interest in having such EWS (Shisanya and Mafongoya 2016). Additionally, famine early warning systems have been successful in Sahelian Africa to alert authorities to impending food shortages so that food acquisition and transportation from outside the region can begin, potentially helping millions of people

(Genesio et al 2011; Hillbruner and Moloney 2012). ***Adverse side-effects*** from EWS are minimal. Barriers to EWS include cost; an early warning system for the 80 most climate vulnerable countries in the world is estimated to cost USD 2 billion over five years to develop (Hallegatte 2012). Institutional and governance barriers such as coordination and synchronisation among levels also effect some EWS (Birkmann et al 2013).

6.9.2.34 Improved food transport and distribution

Improved food transportation and distribution are a collection of practices geared towards a) *improving energy-efficiency* (to reduce GHG emissions and simultaneously improve availability and affordability of food), (b) *reducing food loss and waste* and (c) *minimising risk to human health*. While efficient use of energy and resources in food transport and distribution contribute to a reduction in GHG emissions (Chapter 5; section 6.8; James and James 2010; Vermeulen et al. 2012) thereby providing ***small co-benefits*** for climate mitigation. Strategies such as weatherproofing transport systems, distribution infrastructure and improving the efficiency of food trade (Chapter 2; section 6.4; Ingram et al. 2016; Stathers et al. 2013) especially in countries with inadequate infrastructure and weak food distribution systems (Puma et al. 2015; Wellesley et al. 2017; Vermeulen et al. 2012) Vermeulen et al., 2012), can strengthen climate resilience against future climate-related shocks (Ingram et al. 2016; Stathers et al. 2013) and deliver ***large co-benefits*** for adaptation. Land degradation and desertification can be exacerbated by poor postproduction management practices related to food transport and distribution (Bradford et al. 2018; Temba et al. 2016; Stathers et al. 2013; Tirado et al. 2010). Improved food transportation and distribution can reduce food waste and the need for compensatory extensification of agricultural areas thereby reducing the risk of overexploitation and providing ***small co-benefits*** for land degradation and desertification (Stathers et al. 2013; see also sections on these response options in section 6.9.2.16). Improved storage and distribution systems provide ***large co-benefits*** for food and nutrition security. Improved energy efficiency in food storage and distribution has the potential to reduce food costs and increase availability (Ingram et al. 2016). The perishability and safety of fresh foods are highly susceptible to temperature increase (Bisbis et al. 2018) and are directly linked to household level food security and the well-being of producers (Stathers et al. 2013). Improving and expanding the ‘dry chain’ can significantly reduce food losses at the household level (Bradford et al. 2018). Higher temperatures could increase the presence of pathogens and the challenge of managing food safety (Ingram et al. 2016). The implementation of innovations related to food transport and distribution can be expensive and while there are no obvious biophysical and cultural/behavioural barriers, there are technological (technological barriers include inadequate storage options (Kitijona et al., 2011)), educational and context-specific institutional barriers (e.g. in low income African, Asian and Latin American countries where problems are associated with food loss and institutional and organisational innovation capacities) (Ingram et al. 2016). Reversibility can be an issue and there are likely to be ***few adverse side-effects*** across the challenges (Bajželj et al. 2014a; Tilman and Clark 2014; Clark and Tilman 2017). Technical, organisational and climate communication innovations can improve food storage and distribution in poorer countries and reduce losses to between 1%-2% in some cases (Kumar and Kalita 2017).

6.9.2.35 Avoidance of conversion of grassland to cropland

Since croplands have a lower soil carbon content than grasslands and are also more prone to erosion that grasslands, avoidance of conversion of grassland to croplands will prevent soil carbon losses by oxidation and soil loss through erosion. Avoidance of conversion of grassland to cropland could provide ***large co-benefits*** for climate mitigation by retaining soil carbon stocks that might otherwise be lost. Historical losses of soil carbon have been on the order of 500 $GtCO_2$ (Sanderman et al., 2017). Mean annual global cropland conversion rates (1961-2003) have been 0.36% per year (Krause et al., 2009), i.e. around 4.7 $Mhayr^{-1}$ – so preventing conversion could potentially save significant emissions of CO_2. There could be ***small co-benefits*** for adapatation in terms of stabilising soils to improve resilience (Lal, 2003), or ***moderate adverse side-effects*** by limiting the capacity of farmers to adapt to future challenges (i.e. no possibility to convert grassland to grow crops). In areas where shifting to arable use can provide high-yielding grain crops, there are ***large co-benefits*** for prevention or reversal of desertification by stabilising soils in arid areas (Chapter 3; section 6.6), and ***large co-benefits*** for prevention or reversal of land degradation through the same mechanism (Chapter 4; section 6.7). There are likely to be ***moderate to large adverse side-effects*** for food security, since conversion of grassland to cropland usually occurs to remedy food security, and much more land is required to produce human food from livestock products on grassland than from crops on cropland (Chapter 5; section 6.8; de Ruiter et al. 2017; Clark and Tilman 2017). The soil carbon sink both saturates and is reversible (see section on increasing soil organic matter content; Smith, 2012), though this response option is about protecting existsing stocks rather than increasing them. Avoiding conversion is low cost, but there may be significant opportunity costs associated with foregone production of crops (ref). Since the response option involves not cultivating a current grassland, there are likely to be few biophysical or technological barriers, but there could be institutional barriers in some regions (e.g. poor governance to prevent conversion), and educational (e.g. poor knowledge of the impacts of ploughing grasslands, and cultural / behavioural (e.g. strong cultural importance of crop production in some communities. Avoidance of grassland conversion to cropland also avoids risks for the livelihoods of pastoralists in extensively managed rangelands.

6.9.2.36 Commercial crop insurance

Crop insurance is one of the most widely used financial vehicles to guard against yield losses in agriculture. Crop insurance can involve both traditional indemnity-based insurance that reimburses clients for estimated financial losses from shortfalls, or index insurance that pays out the value of an index rather than actual losses; the former is more common for large farms in the developed world and the latter for smaller noncommercial farms in developing countries (Havemenn and Muccione 2011; Meze-Hausken et al. 2009). Crop insurance is highly subsidised in much of the developed world, with little private sector involvement (Smith and Glauber 2012). From the literature, it appears that cropping insurance offers ***no co-benefits for mitigation,*** and may in fact lead to carbon losses as there is evidence that subsidised crop insurance programmes can induce producers to bring additional land into crop production, particularly marginal or risky lands that may be more environmentally sensitive (Claassen et al. 2011). Crop insurance offers ***moderate co-benefits*** for adaptation, as it provides a means of buffering and transferring weather risk, saving farmers the cost of crop losses (Meze-Hausken et al. 2009). However, overly subsidised insurance can undermine the market’s role in pricing risks and thus depress more rapid adaptation strategies (Skees and Collier 2012; Jaworski 2016). For example, availability of crop insurance was observed to reduce farm-level diversification in the US, a factor cited as increasing adaptive capacity (Sanderson et al. 2013), and crop insurance-holding soybean farmers in the US have been less likely to adapt to extreme weather events than those not holding insurance (Annan and Schlenker 2015). Crop insurance is likely to deliver ***no co-benefits*** for prevention and reversal of desertification and ***no co-benefits*** for prevention and reversal of land degradation, as evidence suggests that subsidised insurance in particular can increase crop production in marginal lands. Crop insurance could have been responsible for shifting up to 0.9 percent of rangelands to cropland in the Upper US Midwest (Claassen et al. 2011), and another study found a 1% increase in farm receipts generated from subsidised farm programs (including crop insurance and others) increased soil erosion by 0.135 tons per acre (Goodwin and Smith 2003). There are ***moderate co-benefits*** for food security from crop insurance, as crop insurance has generally lead to (modest) expansions in cultivated land area and increased food production (Claassen et al. 2011; Goodwin et al. 2004). ***Adverse side-effects*** from crop insurance include water quality problems associated with marginal lands brought into production with greater chemical use due to soil fertility issues (Goodwin and Smith 2003). Barriers to crop insurance include the high costs; few farmers are willing to pay the full commercial cost, which is why most governments subsidise insurance; in the US, this is equivalent to billions of USD every year (Goodwin and Smith 2013).

6.9.2.37 Urban food systems

Urban areas are becoming the principal territories for intervention in improving food access through innovative strategies that aim to eradicate urban hunger and improve livelihoods (section 6.8.5). Interventions include Urban and Peri-urban Agriculture and Forestry (UPAF; Tao et al. 2015; Lwasa et al. 2014) and local food policy and planning initiatives such as Food Policy Councils and city-region-wide regional food strategies (Brinkley et al. 2016; Chappell et al. 2016). Such systems aim to promote inter-linkages of the city and its citizens with surrounding rural areas to create sustainable, and more nutritious food supplies for the city, while improving the health status of urban dwellers, reducing pollution levels, adapting to and mitigating climate change, and stimulating economic development (Akhtar et al. 2016). There are ***small co-benefits*** for climate change mitigation, ***moderate co-benefits*** for adaptation, ***negligible co-benefits*** and ***negligible adverse side-effects*** for prevention or reversal of desertification and ***negligible co-benefits*** and ***negligible adverse side-effects*** for prevention or reversal of land degrdadation. Well developed urban food systems should provide ***large co-benefits*** for food security (section 6.8.5; Chappell et al. 2016). There are likely to be few biophysical, technological or cultural / behavioural barriers to implementing improved urban food systems, though institutional and education barriers could play a role (ref).

6.9.2.38 Improved efficiency and sustainability of food processing, retail and agri-food industries

Improved efficiency and sustainability of retail and agri-food industries involve several practices related to a) *greening supply chains* (e.g. utilising products and services with a reduced impact on the environment and human health), b) *adoption of specific sustainability instruments among agri-food companies* (e.g. eco-innovation practices), c) *adopting emission accounting tools* (e.g. carbon and water footprinting), d) *implementing "demand forecasting" strategies* (e.g. changes in consumer preference for 'green' products) and, e) *supporting polycentric supply-chain governance processes*. Improved efficiency and sustainability of retail and agri-food industries provides ***small co-benefits*** for climate mitigation (Chapter 2; section 6.4; Song et al. 2017) as GHG-friendly foods can create significant savings in agri-food GHG (Song et al. 2017) by reducing greenhouse gas emissions from transportation (Avetisyan et al. 2014), waste (Porter et al. 2016), and energy use (Mohammadi et al. 2014). There are ***small co-benefits*** for climate adaptation (Chapter 2; Macfadyena et al. 2015) especially in cases where climate extremes and natural disasters disrupt supply chain networks (Godfray et al. 2010) And buffer the impacts of changing temperature and rainfall patterns on upstream agricultural production—yields and quality (Ridoutt et al. 2016). There are limited direct co-benefits to land degradation or desertification. This response option also provides ***large co-benefits*** for food security by supporting healthier diets and reducing food loss and waste (Chapter 5; section 6.8; Garnett 2011). As result of decreasing costs in information technology, biotechnology, and renewable energy systems, implementation of this response option is relatively inexpensive (Ridoutt et al. 2016). Reversibility can be an issue and there are likely to be ***few adverse side-effects*** across the challenges (Clark and Tilman 2017). The implementation of strategies to improve the efficiency and sustainability of retail and agri-food industries can be expensive and while there are no obvious biophysical and cultural/behavioural barriers, there are technological (adoption of specific sustainability instruments and eco-innovation practices, educational and context-specific institutional barriers. Successful implementation is dependent on organisational capacity, the agility and flexibility of business strategies, the strengthening of public-private policies and effectiveness of supply-chain governance.

6.9.2.39 Increased energy efficiency in agriculture

Energy is an input into agricultural production that can determine the profitability of farming, which, in turn, impacts heavily upon the farmers' investment in improved farming systems (Baptista et al.

2013). Energy efficiency of agriculture can be improved to reduce the dependency on non-renewable energy sources. This can be realised either by deceased energy inputs, or through increased outputs per unit of input. Although energy efficiency expressed per area unit generally increases with reducing tillage intensity (de Moraes Sá et al. 2017), or replacing chemical inputs by manures and manual weed control, when efficiencies are expressed per unit of product, it does not always increase, since crop yields can be lower in organic and agroecology production systems (Alluvione et al. 2011; Reganold and Wachter 2016). Increased energy efficiency in agriculture delivers ***small co-benefits*** for mitigation, when reducing CO_2 emissions by decreasing the use of fossil fuels or energy-intensive products, though the emission reduction is not accounted for in the AFOLU sector (Smith et al. 2014; IPCC AR5 WG3 Chapter 11). It also delivers ***small co-benefits*** for adaptation, since it often forms part of ecosystem approaches that combine both food and energy production, such as agroforestry or integrated crop–livestock–biogas systems, though not necessarily in agricultural systems more focussed on productivity (Bogdanski 2012; Lipper et al. 2014). For the same reason, it also delivers ***small co-benefits*** for food security. There are ***no co-benefits*** (or adverse side-effects) for prevention and reversal of desertification and ***no co-benefits*** (or adverse side-effects) for prevention and reversal of land degradation. There are ***no adverse side-effects*** from improving energy efficiency in agriculture. Energy efficiency improvement is very cost effective as it decreases energy costs. There are no biophysical barriers to implementation of energy efficiency measures. The main barriers are technological (e.g. low levels of farm mechanisation), institutional (e.g. energy efficiency in agriculture depends strongly on the technology level; Vlontzos et al. 2014), educational (e.g. poor knowledge of alternative energy sources), and behavioural / cultural (e.g. high levels of repetitive labour, making farming unattractive to the youth, and disproportionally affecting women; Baudron et al. 2015).

6.9.2.40 Enhanced weathering of minerals

The enhanced weathering of minerals that naturally absorb CO_2 from the atmosphere has been proposed as a greenhouse gas removal technology (Smith et al. 2016a). The rocks are ground to increase the surface area and the ground material is then applied to the land where it absorbs atmospheric CO_2 (Schuiling and Krijgsman 2006), with potential co-benefits as a soil amendment to raise the pH of acidic soils (Taylor et al. 2016). Enhanced mineral weathering provides ***large co-benefits*** for climate mitigation, with a global mitigation potential in the region of ~0.7-3.7 $GtCO_2yr^{-1}$ (Lenton, 2014; (Smith et al. 2016a); Taylor et al. 2016). Enhanced mineral weathering would not be expected to impact adaptation or desertification so there are ***no co-benefits*** for adaptation and ***no co-benefits*** for the prevention or reversal of desertification. Since ground minerals can increase pH (Taylor et al. 2016), there could be ***small co-benefits*** for prevention and reversal of land degradation, where acidification is the driver of degradation (Taylor et al. 2016). Since increasing soil pH in acidified soils increases productivity, the same effect could provide ***small co-benefits*** for food security (Taylor et al. 2016). Minerals used for enhanced weathering need to be mined, and mining has ***large impacts locally***, though the total area mined is likely to be small on the global scale, so there are likely to be ***small adverse-side effects globally***. Permanence is not an issue of the timescales of relevance since the CO_2 absorbed from the atmosphere is mineralised. The main costs (and large energy input) is in the mining and comminution of the minerals (Renforth, 2012), with higher total costs compared to low cost. Land management options (Smith et al. 2016a). Barriers to implementation include biophysical (e.g. limited and inaccessible mineral formations; Renforth, 2012), institutional in some regions (e.g. lack of infrastructure for this new technology; Taylor et al. 2016), technological (high energy costs of comminution; (Smith et al. 2016a) and educational (e.g. lack of knowledge of how to use these new materials in agriculture). Cultural barriers could occur in some regions, e.g. due to minerals lying under undisturbed natural areas where mining might generate public acceptance issues (e.g. Renforth, 2012).

6.9.2.41 Material substitution

Material substitution involves the use of wood products instead of traditional building materials (e.g., iron, steel, aluminium, etc.). Such a substitution reduces carbon emissions both because the wood sequesters carbon during the growth phase and because it reduces the demand for fossil fuel-based materials. Material substitution has the potential for ***large co-benefits*** for mitigation, with one study estimating a 14% to 31% reduction in global CO_2 emissions (Oliver et al. 2014). These measures are unlikely to have any appreciable impact upon adaptation, prevention of desertification or land degradation, or delivery of food security. Neither will they impact appreciably on most ecosystem services.

6.9.2.42 Bioenergy and BECCS

Bioenergy and BECCS are often used in mitigation scenarios as a low carbon energy option. Bioenergy and BECCS has the potential to reduce energy system emissions. However, bioenergy production could result in increase in land use change CO_2 emissions or in increased N_2O emissions from fertiliser use. Despite these effects, the use of bioenergy and BECCS provides ***large co-benefits*** for climate mitigation (Chapter 2; section 6.4.5; IPCC SR1.5), with a cumulative mitigation potential as high as 20 $GtCO_2eyr^{-1}$ (Kriegler et al. 2017). Intensive management of bioenergy crops can result in declines in soil organic matter (FAO (Food and Agriculture Organization) 2011; Lal 2014), but the production of bioenergy on marginal lands can increase soil carbon (Robertson et al. 2017; Mello et al. 2014). Large-scale production of bioenergy can require significant amounts of land, e.g., 380 to 700 Mha in 2°C scenarios (Smith et al. 2016a), increasing pressures for land conversion and land degradation. As a result, bioenergy and BECCS has the potential to provide ***small co-benefits*** or ***small adverse side-effects*** or ***large adverse side-effects*** for desertification and land degradation. Competition for land between bioenergy and food crops can result in increased food prices, creating ***large adverse side-effects*** for food security (section 6.4.5; Muratori et al. 2016; Favero and Mendelsohn 2017; Calvin et al. 2014; Obersteiner et al. 2016; Popp et al. 2017a; van Vuuren et al. 2015; Popp et al. 2011). While planting perennial grasses for bioenergy can have ***co-benefits*** in terms of increased biodiversity and ecosystem function, the conversion of natural land to bioenergy can also have ***adverse side-effects*** in terms of reduced biodiversity (Santangeli et al. 2016; Smith 2018). The sign and magnitude of the effects of bioenergy and BECCS depends on the scale of deployment, the type of bioenergy feedstock, and where bioenergy is grown. For example, limiting bioenergy production to marginal lands or abandoned cropland would have negligible effects on biodiversity, food security, and potentially small co-benefits for land degradation; however, the benefits for mitigation would also be smaller (section 6.4.5). The main barriers are biophysical, technological, institutional and cultural (IPCC SR1.5; chapter 7). In terms of technological barriers, while there is a small-scale BECCS demonstration facility, BECCS has not been implemented at scale (Kemper 2015b). Cultural barriers include social acceptance (Sanchez and Kammen, 2016), with CCS facing concerns of safety and environmental issues and bioenergy facing additional scrutiny because of competition for land and water.

Table 6.3 Summary of co-benefits, adverse side effects, permanence/saturation issues, costs and barriers of response options to address land challenges

Response option	Sector	Impacts on land challenges – co-benefits and adverse side effects*					Saturation or reversibility issues	Cost	Barriers
		M	A	D	L	F			
Increased soil organic matter (and reduced losses)									
Improved cropland management									
Improved livestock management									
Improved grazing land management									
Sustainable intensification to deliver increased food productivity									
Agro-forestry									
Sustainable healthy diets									
Sustainable forest management									
Agricultural diversification									
Management of erosion									
Land tenure / ownership									
Prevent / reverse soil salinization									
Prevention of land grabbing									
Prevention of compaction									
Reduce post-harvest losses									
Reduce food waste (consumer or retailer)									
Promote value added products									
Management of urban sprawl									
Fire management									
Management of landslides and natural hazards									
Ecosystem-based adaptation									
Reduced deforestation									
Livelihood diversification									
Promotion of seed sovereignty									
Management of pollution including acidification									
Management of invasive species									
Reforestation									
Restoration and avoid conversion of coastal wetlands									
Stability of food supply									

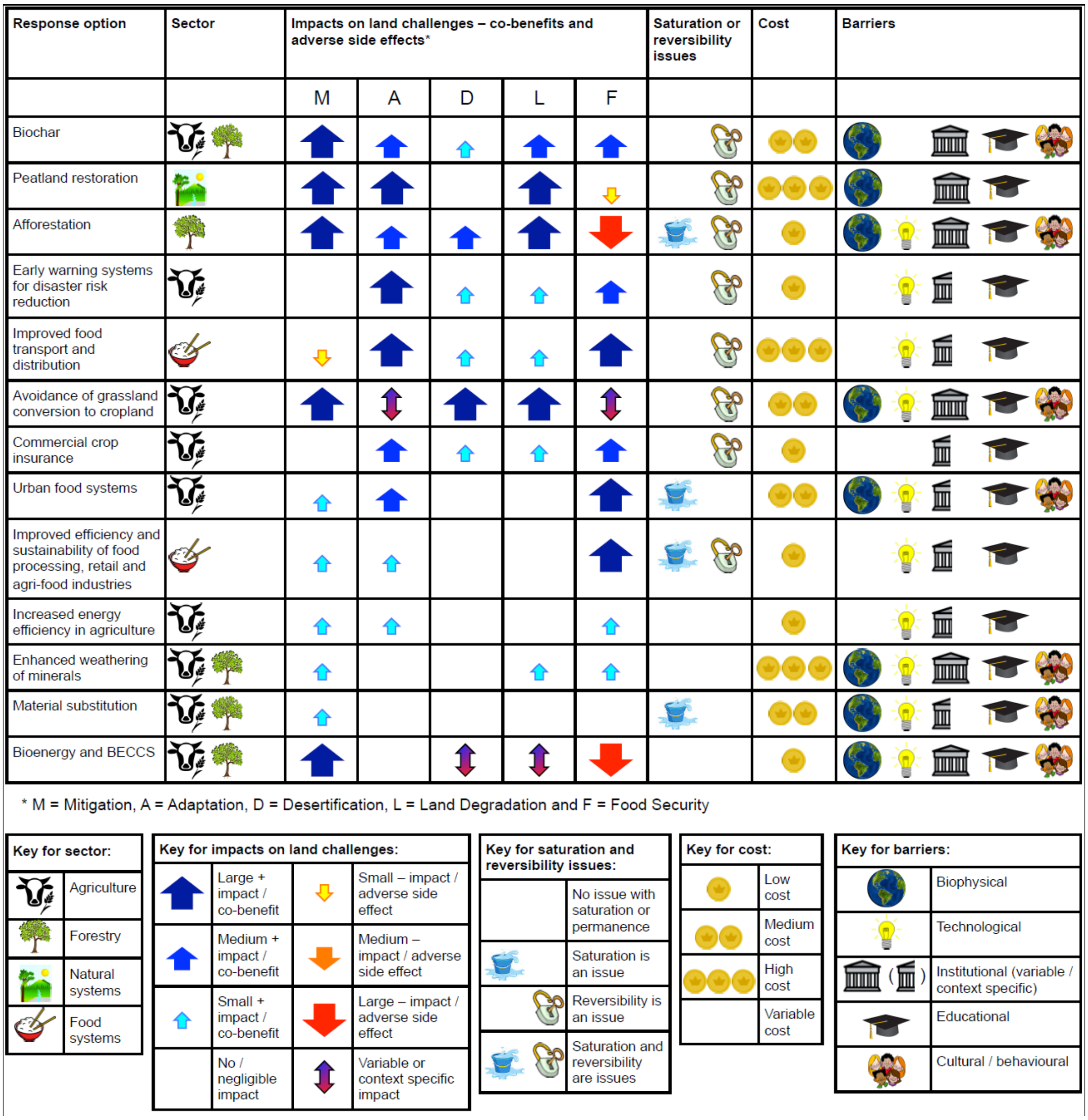

Table 6.4. Key for criteria used to define size of impact in Table 6.3

	Mitigation	Adaptation	Land Deg	Desertification	Food
3	Around more than 1 GtCe per year	Positively impacts greater than 25 million people	Positively impacts 500 million hectares	Positively impacts 300 million hectares	Positively impacts greater than 100 million people
2	0.1 – 1 GtCe	1 million – 25 million	100 – 500 million hectares	50 – 300 million hectares	1 million – 100 million
1	>0	Under 1 million	>0	>0	Under 1 million
0	0	No effect	No effect	No effect	No effect
-1	<0	Under 1 million	<0	<0	Under 1 million

-2	-0.1 – 1 GtCe	1 million – 25 million	100 – 500 million hectares	50 – 300 million hectares	1 million – 100 million
-3	More than -1 GtCe	Negatively impacts greater than 25 million people	Negatively impacts 500 million hectares	Negatively impacts 300 million hectares	Negatively impacts greater than 100 million people

*All numbers above are guidelines, not strict cut-off points
*All above are global numbers. Consider impacts relative to the population / region
*Consider linking to global targets (e.g. meet 10% of target). Consider space-specificity
*Mitigation: 100 $GtCO_2$ in 2100 to go from baseline to 2˚C
*Food security: 800 million people undernourished
*Land deg: 1-6 billion hectares degraded land
*Desertification: 0.6-1.20 billion hectares
*Revisit adaptation numbers. Total number of people globally vulnerable to climate change. 5 million lives lost per year. 100 million lives between now and next decade
*All technical potential. In notes, elaborate on economic potential, cost, barriers etc.

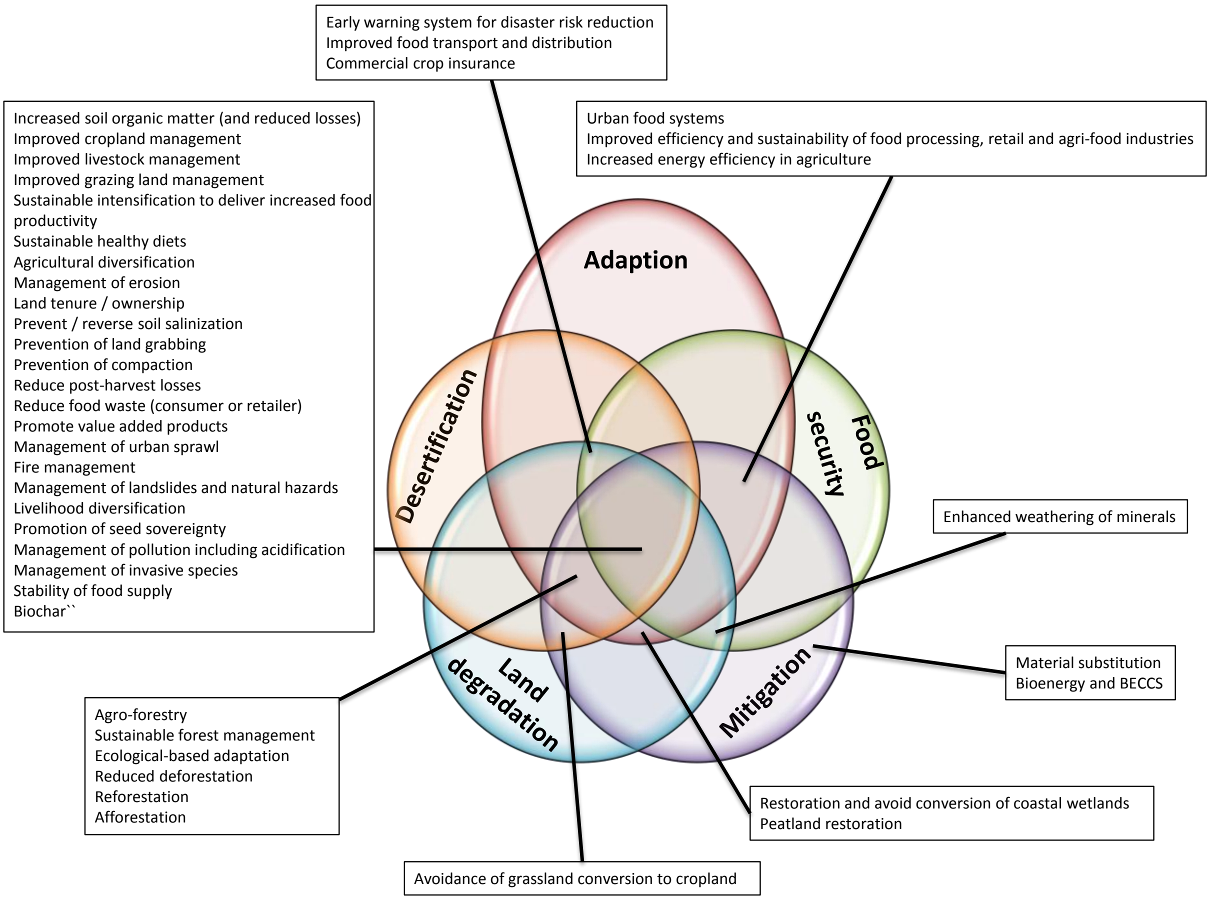

Figure 6.7 Summary of co-benefits delivered by response options across the five land challenges

As shown in section 6.3.2, a large part of the land area is exposed to multiple land challenges, especially in the Villages, Croplands and Rangelands anthromes. Appropriate responses will vary in each anthrome with the local exposure to one or more land challenges. For instance, in croplands exposed to a combination of rapid climate change, land degradation and food insecurity (see Figure 6.3A) responses could be considered as appropriate when delivering co-benefits for climate change adaptation, for land conservation and for food security. As the challenge of climate change mitigation is not local but global, co-benefits for this challenge would also be required for the design of appropriate responses.

Since there are only few responses that deliver co-benefits for all of mitigation, adaptation, food security, land degradation and desertification and to additional local constraints like the conservation of biodiversity hot-spots and of depleted groundwater resources (see 6.3.2), the number of appropriate responses declines with the number of local land based challenges. For instance with croplands, there are 12 appropriate options available (with large co-benefits for mitigation) when there is no local land challenge, but this number declines to eight, on average, with two local challenges (in 37% of global cropland area) and to four with four local challenges (in 9% of global cropland area) (Figure 6.8) .

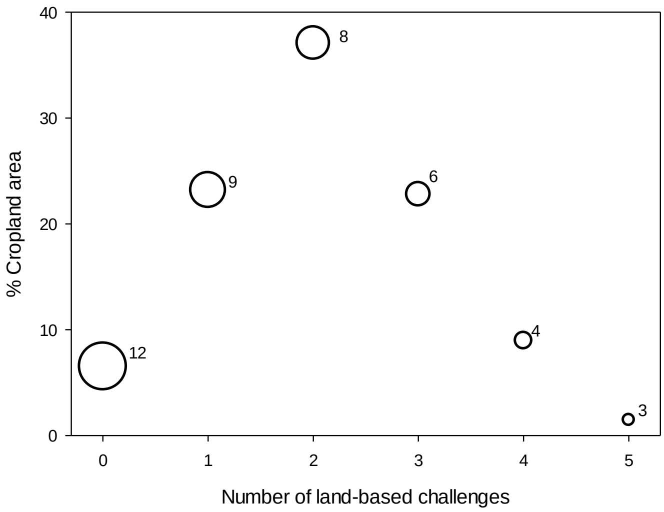

Figure 6.8 Distribution of global cropland area by number of local land based challenges and mean number of appropriate responses providing co-benefits for each of the local challenges and for the (universal) challenge of climate change mitigation. The number of appropriate response is plotted and is indicated by the size of the open circle symbol.

6.9.3 Opportunities for Integrative Response Options

6.9.3.1 Interlinkages in future scenarios

Reference Scenarios: Absent any efforts to mitigate, global mean temperatures are expected to continue to rise. For example, the SSP baseline scenarios cover a range between about 5.0–8.7 W/m^2 by 2100 (Riahi et al. 2017a) and Collins et al. (2013) report temperature rise ranging from 2.6°C to 4.8°C in the RCP8.5 in 2100 (see Chapter 2). Climate change is expected to result in increases in aridity, and associated dryland expansion (Cook et al. 2014b; Lin et al. 2015; Scheff and Frierson 2015; Fu et al. 2016) (Need to get these citations from Chapter 3), leading to increased desertification (see Chapter 3). Increases in income, as well as agricultural productivity, result in declines in the number of malnourished people (Fujimori and ...), ameliorating food security; however, climate change has implications for food production, with some studies showing a decrease in food produced in due to climate change impacts (Tai et al. 2014) as well as increasing food prices (Popp et al. 2017b; Wiebe et al. 2015), see Chapter 5) and agricultural welfare (Stevanovic et al. 2016). The majority of these reference scenarios are characterised by proceeding and even increasing negative environmental impacts such as loss of natural land (especially forests in tropical areas) (Popp et al. 2014, 2017a), ongoing emissions from agricultural production (Frank et al. 2017a; Popp et al. 2017b), enhanced nitrogen pollution (mainly due to fertilisation of croplands) (Bodirsky et al. 2014; Humpenöder et al. 2018; Obersteiner et al. 2016; Mogolí On et al. 2018) and water withdrawal for irrigation (Bonsch et al. 2015; Hejazi et al. 2014), with severe consequences for ecosystems and human well-being.

Mitigation Options: The scenario literature has largely focused on emission reductions from agricultural production and land-use change, as well as the CDR options bioenergy and BECCS and afforestation and/or reforestation as land-based mitigation options (see Section 6.3.3, IPCC SR1.5 Chapter 2). The inclusion of bioenergy/BECCS can result in increased nitrogen pollution, land devoted to energy crops, declines in forest and other natural land, additional water withdrawal for irrigation (Chaturvedi et al. 2013b; Hejazi et al. 2015b, 2014; Mouratiadou et al. 2016; Bonsch et al. 2015) and increases in food prices (Calvin et al. 2014; Popp et al. 2014; Muratori et al. 2016; Wise et al. 2009) (Figure 6.8). Muratori et al. (2016) shows that these effects are exacerbated in scenarios with bioenergy but no BECCS, as bioenergy deployment is higher to reach the same climate mitigation target. In scenarios with both bioenergy/BECCS and afforestation/reforestation, bioenergy production occurs on other natural and pasture land, enabling an expansion of forest (Humpenöder et al. 2014); however, this can result in higher food prices than scenarios with bioenergy/BECCS alone (Calvin et al. 2014; Kreidenweis et al. 2016) (Figure 6.8). In general, the effect of mitigation on desertification, land degradation, food security, biodiversity, and other sustainable development goals depends strongly on which mitigation options are included and the extent to which they are deployed (Figure 6.9). (*This paragraph will investigate sustainability interactions of scenario archetypes introduced and assessed for climate and land-consequences in chapter2 as soon as scenario data base for SRCCL is available*)

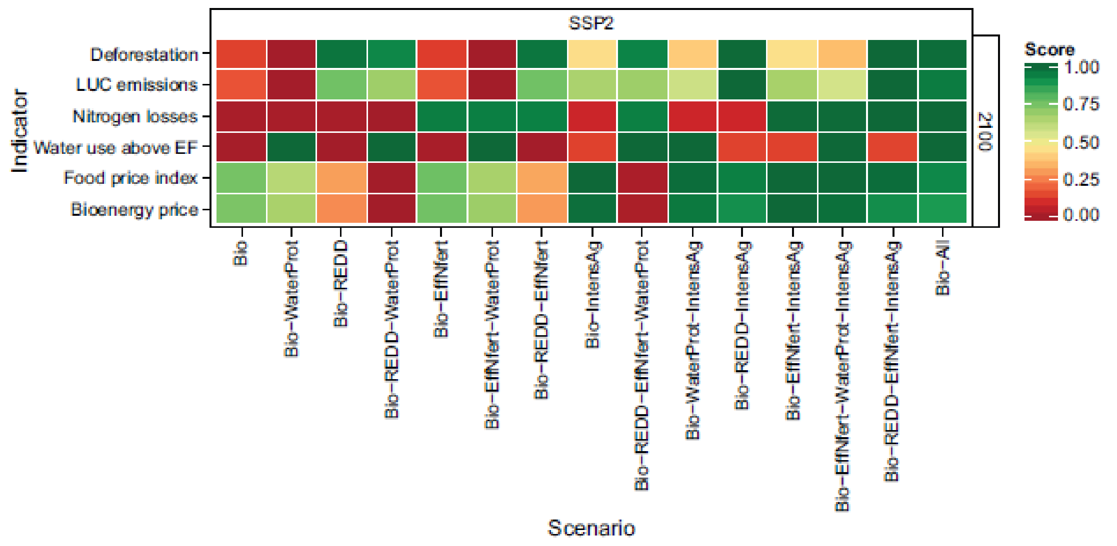

Figure 4. Additional scenarios with all possible combinations of measures considered in our study (see table 2). Results are shown for year 2100 at the global scale. To facilitate straightforward comparison of scenario outcomes across all indicators, we calculate for each indicator normalized scores ranging between 0 and 1. First we determine the worst (0) and the best outcome (1) for each indicator across all scenarios and then align all outcomes in-between accordingly. Finally, we sort the scenarios according to the sum of the six indicator scores (best scenario is on the right-hand side).

Figure 6.9 Implications of different mitigation scenarios for land-related goals (*This figure will use the scenario archetypes introduced in Chapter 2 and quantify them across different metrics related to land degradation, desertification, food security, biodiversity, other sustainability indicators depending on availability in SRCCL database*. *Current figure is a place holder from* (Humpenöder et al. 2018)*; figure will be updated for SOD.*)

Adaptation Options: Scenarios including the implications of changes in climate on agricultural yield quantify the implications of these changes for cropland area, in general finding increases in cropland area when yields decline and decreases when yields rise (Nelson et al. 2014; Calvin et al. 2013; Kyle et al. 2014; Reilly et al. 2007). Some scenarios examine the implications of climate for livestock production, including effects on the productivity of rangeland and feed crops. These studies show that shifts in livestock production systems represent a resource- and cost-efficient adaptation options, simultaneously abating deforestation and hence climate change (Weindl et al. 2015b). Some other studies examine the effect of adaptation on mitigation in croplands, quantifying differences in bioenergy production (Calvin et al. 2013; Kyle et al. 2014) or carbon price (Calvin et al. 2013) as a result of climate impacts. Kyle et al. (2014) finds increase in bioenergy production due to increases in bioenergy yields, while Calvin et al. (2013) finds declines in bioenergy production and increases in carbon price due to the negative effects of climate on crop yield.

6.9.3.2 Dealing with interlinkages when formulating integrated response options

Response options have interlinked implications across the challenges in the land sector; options to address one land challenge may exacerbate other problems. Among the response options available to address the land challenges, many have impacts across more than one challenge either by delivering co-benefits across a range of challenges, for example many sustainable land management practices co-deliver to climate change mitigation and adaptation, to preventing or addressing desertification and land degradation, and to food security (Table 6.3; Figure 6.7). Other response options create adverse-side effects for one or more challenges, for example response options that demand land for climate mitigation, could cause adverse side effects if implemented at scale for food production, and thereby food security, *via* increasing competition for land.

Land resources are limited. Competition for land may restrict the scale at which response options can be used. The land is a finite resource and expansion of the current area of managed land into natural ecosystems would lead to the loss of biodiversity and a range of ecosystem services (Sections 6.4-6.8). For this reason, the scale at which some response options can be applied is limited, with response options that compete for land, e.g. afforestation, BECCS, most affected (Table 6.3; Figure 6.7). Other ptions that can be applied without changing the use of the land, for example measures to increase the soil organic matter (carbon) content of soils, are not limited by land competition constraints.

The impacts of many response options are scale and context dependent, and are uneven across different regions and communities. The efficacy, and impacts, of response options to address each land challenge is location specific, with for example, the mitigation effectiveness or adaptation effectiveness differing by bioclimatic region, land management system or local food system context (Sections 6.4-6.8). Further, for some scalable response options, large global impacts are seen only when implemented at large scale (Sections 6.4-6.9). In addition, impacts are context dependent, with certain options producing adverse side-effects only in certain locations, for example response options that use freshwater might have no adverse side effects in regions where water is plentiful, but large adverse side effects in regions where water is scarce (Section 6.9.2).

All land challenges need to be considered when addressing potential solutions, in order to identify response options that co-deliver across the range of challenges (Section 6.4-6.9). Because the different land challenges are often the concern of different policy and research communities, response options are often proposed to address a specific land challenge. However, since all land challenges share the same land resource, response options can affect, positively or negatively, a number of land challenges. For this reason, considering the impact of response options on all land challenges simultaneously will allow co-benefits to be maximised and adverse side-effects to be minimised.

Many response options (over 40 in number – section 6.9.2.1-6.9.2.42) have multiple co-benefits across land-related goals, but some are not currently widely implemented (Table 6.3; Figure 6.7). The majority of response options considered have potential to deliver co-benefits across the range of land challenges, with co-benefits ranging from large to small across options and challenges. Other options deliver large co-benefits for one or two challenges, with negligible impact on others, but do no harm. Many of the same options also have either no, or small, context-specific adverse side-effects. There are, therefore, a range of “no regrets” options that are suitable to wider implementation to address multiple land challenges. While some of these response options are implemented widely in some regions, other are not, and even for those with wide regional adoption, there is considerable scope for wider deployment globally.

Some response options, such as large-scale BECCS, have the potential to deliver very well for one land challenge only, with potential detrimental effects on other land challenges (Section 6.9.2.42). A small number response options have very large potential to address one land challenge, but could lead to large adverse side-effects if implemented at scale. Options that require land use change (e.g. BECCS, afforestation; 6.9.2.45; 6.9.2.35), and thereby contribute to land competition, are most prevalent in this category, with food security the land challenge most often adversely affected (Section 6.8). Options that improve land management or improve efficiency of production of food and fibre (sustainable land management options) do not fall into this category and they either do not affect competition for land, or have the potential to decrease it.

There are currently barriers to implementation for many response options; identifying and removing barriers is necessary to make progress toward sustainable solutions (Table 6.3; Figure 6.7). Since there is good evidence that many response options will deliver multiple co-benefits across the range of land challenges, yet these are not applied universally, is evidence that multiple barriers to implementation exist. A combination of economic, biophysical, technological, institutional, education, cultural and behavioural barriers exist for each response option in various regions, and these barriers need to be overcome if response options are to be more widely applied. Options aiming to preserve ecosystem services and biodiversity depend largely on land governance and financial aid, since markets where such services can be traded are not well developed. Improved institutional frameworks would strengthen land governance and facilitate efforts to preserve ecosystem services and biodiversity.

Coordinated action is required across a range of actors, including consumers, land managers, indigenous and local communities and policymakers. Since barriers to implementation are economic, biophysical, technological, institutional, education, cultural and behavioural, action is required across a multiple actors. Because of the wide range of actors, and the wide range of impacts to be considered across the land challenges, action to address barriers to implementation would be most effective if action were coordinated across the range of actors, including consumers, land managers, indigenous and local communities and policymakers.

The need to act is urgent. Delayed action will result in an increased need for response and a decreased potential of response options due to climate change and other pressures. Delayed action to address any of the land challenges of climate change, desertification, land degradation and food security make the challenges more difficult to address in future, and often make the response options less effective. For example, failure to mitigate climate change with increase requirements for adaptation, and may reduce the efficacy of future mitigation options, e.g. by reducing the sink capacity for soil and vegetation carbon sequestration (e.g. section 6.9.2.1). For this reason, and the extent of the land challenges currently, the need to act is urgent.

Though there are gaps in knowledge and more R&D is required for some response options, enough is known to take action now. There are knowledge gaps for some response options, both in their efficacy and in their broader impacts, particularly among the more recently emerging options (e.g. enhanced weathering of minerals, BECCS; section 6.9.2.43, 6.9.2.45). Nevertheless, many response options have been practiced in some regions for many years and have a broad evidence base, so could be applied more widely immediately, with little risk of adverse side-effects if the best available knowledge is used to design implementation plans for these "no regrets" options.

Cost-effective no regrets options are available for immediate local application, providing that compliance with sustainable development is considered. Many "no regrets" response options which delver across the range of land challenges and beyond (e.g. improved dietary health through improved diets) are also cost-effective, with many being low cost, and some even cost negative. Where not already applied due to local barriers to implementation, these response options are available for immediate application, if barriers can be removed. Assessing impacts against the SDGs (and indicators thereof), or other components of sustainable development, would provide a safeguard against inappropriate local implementation.

Creating an enabling environment, including local engagement, to facilitate the adoption of no-regrets options is required. In addition to the need to engage multiple actors, and to assess the impact of implementation across the range of land challenges and against compliance with sustainable development, implementation of response options would be facilitated by local engagement, and the creation of an enabling environment under which the barriers to implementation could be overcome. Policy will be required to address all of these issues.

6.9.3.3 Moving from response options to policies

Understanding the integrative response options available in a given context requires an understanding of the specificities of social vulnerability, adaptive capacity, and institutional support. Vulnerability often reflects how access to resources are distributed within and among communities, shaped by such factors as "poverty and inequality, marginalisation, food entitlements, access to insurance, and housing quality" (Adger et al. 2004), which are not easily overcome with technical solutions. Adaptive capacity relates to the ability of institutions or people to modify or change characteristics or behaviour so as to cope better with existing or anticipated external stresses (Moss et al. 2001; Brenkert and Malone 2005; Brooks et al. 2005). Adaptive capacity reflects institutional and policy support networks, and has often been associated at the national level with strong developments in the fields of economics, education, health, and governance and political rights (Smit et al. 2001). Conjoining response options to maximise social, climatic and environmental benefits will require framings of such actions as strong pathways to sustainable development (Ayers and Dodman 2010). Chapter 7 discusses in further depth the risks and challenges involved in formulating policy responses that meet these demands for sustainable land management and development outcomes, such as food security, community adaptation and poverty alleviation.

References

Accorsi, R., A. Gallo, and R. Manzini, 2017: A climate driven decision-support model for the distribution of perishable products. *J. Clean. Prod.*, doi:10.1016/j.jclepro.2017.07.170.

ADB, 2016: *Asian Water Development Outlook 2016; Strengthening Water Security in Asia and the Pacific*. 66 pp.

Adger, W. N., 1999: Social vulnerability to climate change and extremes in coastal Vietnam. *World Dev.*, **27**, 249–269, doi:10.1016/S0305-750X(98)00136-3.

Adger, W. N., N. Brooks, G. Bentham, and M. Agnew, 2004: New indicators of vulnerability and adaptive capacity. *Change*,.

Adnan, S., 2013: Land grabs and primitive accumulation in deltaic Bangladesh: Interactions between neoliberal globalization, state interventions, power relations and peasant resistance. *J. Peasant Stud.*, **40**, 87–128, doi:10.1080/03066150.2012.753058.

Van Aelst, K., and N. Holvoet, 2016: Intersections of Gender and Marital Status in Accessing Climate Change Adaptation: Evidence from Rural Tanzania. *World Dev.*, doi:10.1016/j.worlddev.2015.11.003.

Aggarwal, P. K., and Coauthors, 2018: The climate-smart village approach: framework of an integrative strategy for scaling up adaptation options in agriculture. *Ecol. Soc.*, **23**, art14, doi:10.5751/ES-09844-230114. https://www.ecologyandsociety.org/vol23/iss1/art14/ (Accessed June 2, 2018).

Agrawal, A., and N. Perrin, 2009: Climate Adaptation, Local Institutions and Rural Livelihoods. *Adapting to Climate Change: Thresholds, Values, Governance*.

Agrawal, A., A. Chhatre, and R. Hardin, 2008: Forests in Flux. *Science (80-.).*, **320**, 1460–1462, doi:10.1126/science.320.5882.1435.

Aha, B., and J. Z. Ayitey, 2017: Biofuels and the hazards of land grabbing: Tenure (in)security and indigenous farmers' investment decisions in Ghana. *Land use policy*, **60**, 48–59, doi:10.1016/j.landusepol.2016.10.012.

Ahammad, R., P. Nandy, and P. Husnain, 2013: Unlocking ecosystem based adaptation opportunities in coastal Bangladesh. *J. Coast. Conserv.*, **17**, 833–840, doi:10.1007/s11852-013-0284-x.

Ahmed, S., and J. R. Stepp, 2016: Beyond yields: Climate change effects on specialty crop quality and agroecological management. *Elem. Sci. Anthr.*, **4**, 92, doi:10.12952/journal.elementa.000092. http://www.elementascience.org/articles/10.12952/journal.elementa.000092 (Accessed May 31, 2018).

Ahrends, A., and Coauthors, 2010: Predictable waves of sequential forest degradation and biodiversity loss spreading from an African city. *Proc. Natl. Acad. Sci. U. S. A.*, **107**, 14556–14561, doi:10.1073/pnas.0914471107. http://www.ncbi.nlm.nih.gov/pubmed/20679200 (Accessed June 2, 2018).

Ajibade, I., and G. McBean, 2014: Climate extremes and housing rights: A political ecology of impacts, early warning and adaptation constraints in Lagos slum communities. *Geoforum*, doi:10.1016/j.geoforum.2014.05.005.

Akhtar, P., Y. Tse, Z. Khan, and R Rao-Nicholson, 2016: Data-driven and adaptive leadership contributing to sustainability: Global agri-food supply chains connected with emerging markets. https://www.sciencedirect.com/science/article/pii/S0925527315005125 (Accessed May 1, 2018).

Akter, S., T. J. Krupnik, F. Rossi, and F. Khanam, 2016: The influence of gender and product design on farmers' preferences for weather-indexed crop insurance. *Glob. Environ. Chang.*, doi:10.1016/j.gloenvcha.2016.03.010.

Alavian, V., and Coauthors, 2009: Water and Climate Change: Understanding the Risks and Making Climate Smart Investment Decisions.

Albanito, F., T. Beringer, R. Corstanje, B. Poulter, A. Stephenson, J. Zawadzka, and P. Smith, 2016: Carbon implications of converting cropland to bioenergy crops or forest for climate mitigation: A global assessment. *GCB Bioenergy*, doi:10.1111/gcbb.12242.

Albert, C., T. Zimmermann, J. Knieling, and C. von Haaren, 2012: Social learning can benefit decision-making in landscape planning: Gartow case study on climate change adaptation, Elbe valley biosphere reserve. *Landsc. Urban Plan.*, doi:10.1016/j.landurbplan.2011.12.024.

Alburquerque, J. A., C. de la Fuente, A. Ferrer-Costa, L. Carrasco, J. Cegarra, M. Abad, and M. P. Bernal, 2012: Assessment of the fertiliser potential of digestates from farm and agroindustrial residues. *Biomass and Bioenergy*, doi:10.1016/j.biombioe.2012.02.018.

Aldababseh, A., M. Temimi, and P. Maghelal, Multi-Criteria Evaluation of Irrigated Agriculture Suitability to Achieve Food Security in an Arid Environment. doi:10.3390/su10030803.

Aldrich, D. P., 2017: The Importance of Social Capital in Building Community Resilience. *Rethinking Resilience, Adaptation and Transformation in a Time of Change*, Springer International Publishing, Cham, 357–364 http://link.springer.com/10.1007/978-3-319-50171-0_23 (Accessed June 2, 2018).

Alessa, L., A. Kliskey, J. Gamble, M. Fidel, G. Beaujean, and J. Gosz, 2016: The role of Indigenous science and local knowledge in integrated observing systems: moving toward adaptive capacity indices and early warning systems. *Sustain. Sci.*, **11**, 91–102, doi:10.1007/s11625-015-0295-7. http://link.springer.com/10.1007/s11625-015-0295-7 (Accessed December 26, 2017).

Ali, G., N. Pumijumnong, and S. Cui, 2018: Valuation and validation of carbon sources and sinks through land cover/use change analysis: The case of Bangkok metropolitan area. *Land use policy*, **70**, 471–478, doi:10.1016/J.LANDUSEPOL.2017.11.003. https://www.sciencedirect.com/science/article/pii/S0264837717311316 (Accessed May 31,

2018).

Alkama, R., and A. Cescatti, 2016: Biophysical climate impacts of recent changes in global forest cover. *Science (80-.).*, **351**, 600–604, doi:10.1126/science.aac8083.

Alluvione, F., B. Moretti, D. Sacco, and C. Grignani, 2011: EUE (energy use efficiency) of cropping systems for a sustainable agriculture. *Energy*, **36**, 4468–4481, doi:10.1016/J.ENERGY.2011.03.075. https://www.sciencedirect.com/science/article/pii/S036054421100243X (Accessed June 1, 2018).

Altieri, M. A., and C. I. Nicholls, 2017: The adaptation and mitigation potential of traditional agriculture in a changing climate. *Clim. Change*, doi:10.1007/s10584-013-0909-y.

——, ——, A. Henao, and M. A. Lana, 2015: Agroecology and the design of climate change-resilient farming systems. *Agron. Sustain. Dev.*, **35**, 869–890, doi:10.1007/s13593-015-0285-2. http://link.springer.com/10.1007/s13593-015-0285-2 (Accessed June 1, 2018).

Amalraj, A., and A. Pius, 2017: Assessment of groundwater quality for drinking and agricultural purposes of a few selected areas in Tamil Nadu South India: a GIS-based study. *Sustain. Water Resour. Manag.*, **4**, 1–21, doi:10.1007/s40899-017-0113-8. http://link.springer.com/10.1007/s40899-017-0113-8.

Ampaire, E. L., L. Jassogne, H. Providence, M. Acosta, J. Twyman, L. Winowiecki, and P. van Asten, 2017: Institutional challenges to climate change adaptation: A case study on policy action gaps in Uganda. *Environ. Sci. Policy*, doi:10.1016/j.envsci.2017.05.013.

Anderson, C. M., C. B. Field, and K. J. Mach, 2017: Forest offsets partner climate-change mitigation with conservation. *Front. Ecol. Environ.*, **15**, 359–365, doi:10.1002/fee.1515. http://doi.wiley.com/10.1002/fee.1515 (Accessed May 1, 2018).

Anderson, H. R., 2017: Implications for the science of air pollution and health. *Lancet Respir. Med.*, doi:10.1016/S2213-2600(17)30396-X.

Anderson, K., and G. Peters, 2016: The trouble with negative emissions. *Science (80-.).*, doi:10.1126/science.aah4567.

Andivia, E., F. Natalini, M. Fernández, R. Alejano, and J. Vázquez-Piqué, 2018: Contrasting holm oak provenances show different field performance but similar resilience to drought events eight years after planting in a Mediterranean environment. *http://www.sisef.it/iforest*, **11**, 259, doi:10.3832/IFOR2573-011.

Annan, F., and W. Schlenker, 2015: Federal Crop Insurance and the Disincentive to Adapt to Extreme Heat. *Am. Econ. Rev.*, **105**, 262–266, doi:10.1257/aer.p20151031. http://pubs.aeaweb.org/doi/10.1257/aer.p20151031 (Accessed June 1, 2018).

Ansah, I. G. K., B. K. D. Tetteh, and S. A. Donkoh, 2017: Determinants and income effect of yam postharvest loss management: evidence from the Zabzugu District of Northern Ghana. *Food Secur.*, **9**, 611–620, doi:10.1007/s12571-017-0675-1. http://link.springer.com/10.1007/s12571-017-0675-1 (Accessed May 10, 2018).

Antille, D. L., J. M. Bennett, and T. A. Jensen, 2016: Soil compaction and controlled traffic considerations in Australian cotton-farming systems. *Crop Pasture Sci.*, **67**, 1, doi:10.1071/CP15097. http://www.publish.csiro.au/?paper=CP15097 (Accessed May 31, 2018).

Antwi-Agyei, P., L. C. Stringer, and A. J. Dougill, 2014: Livelihood adaptations to climate variability: insights from farming households in Ghana. *Reg. Environ. Chang.*, **14**, 1615–1626, doi:10.1007/s10113-014-0597-9. http://link.springer.com/10.1007/s10113-014-0597-9 (Accessed June 1, 2018).

Arndt, C., M. A. Hussain, V. Salvucci, and L. P. Østerdal, 2016: Effects of food price shocks on child malnutrition: The Mozambican experience 2008/2009. *Econ. Hum. Biol.*, **22**, 1–13, doi:10.1016/J.EHB.2016.03.003. https://www.sciencedirect.com/science/article/pii/S1570677X16300119 (Accessed June 1, 2018).

Arneth, A., and Coauthors, 2010: From biota to chemistry and climate: towards a comprehensive description of trace gas exchange between the biosphere and atmosphere. *Biogeosciences*, **7**, 121–149, doi:10.5194/bg-7-121-2010. http://www.biogeosciences.net/7/121/2010/ (Accessed June 2, 2018).

Arora, V. K., and A. Montenegro, 2011: Small temperature benefits provided by realistic afforestation efforts. *Nat. Geosci.*, doi:10.1038/ngeo1182.

Arslan, A., R. Cavatassi, F. Alfani, N. Mccarthy, L. Lipper, and M. Kokwe, 2018: Diversification Under Climate Variability as Part of a CSA Strategy in Rural Zambia. *J. Dev. Stud.*, **54**, 457–480, doi:10.1080/00220388.2017.1293813. https://www.tandfonline.com/doi/full/10.1080/00220388.2017.1293813 (Accessed June 1, 2018).

Asfaw, S., G. Pallante, and A. Palma, 2018: Diversification Strategies and Adaptation Deficit: Evidence from Rural Communities in Niger. *World Dev.*, **101**, 219–234, doi:10.1016/J.WORLDDEV.2017.09.004. https://www.sciencedirect.com/science/article/pii/S0305750X16301863 (Accessed June 1, 2018).

Avetisyan, M., T. Hertel, and G. Sampson, 2014: Is Local Food More Environmentally Friendly? The GHG Emissions Impacts of Consuming Imported versus Domestically Produced Food. *Environ. Resour. Econ.*, **58**, 415–462, doi:10.1007/s10640-013-9706-3.

Awono, A., O. A. Somorin, R. Eba'a Atyi, and P. Levang, 2014: Tenure and participation in local REDD+ projects: Insights from southern Cameroon. *Environ. Sci. Policy*, **35**, 76–86, doi:10.1016/j.envsci.2013.01.017. http://dx.doi.org/10.1016/j.envsci.2013.01.017.

Ayers, J., and T. Forsyth, 2009: Community-Based Adaptation to Climate Change. *Environ. Sci. Policy Sustain. Dev.*, doi:10.3200/ENV.51.4.22-31.

——, and D. Dodman, 2010: Climate change adaptation and development I: The state of the debate. *Prog. Dev. Stud.*, doi:10.1177/146499340901000205.

Bai, Y., C. Zhuang, Z. Ouyang, H. Zheng, and B. Jiang, 2011: Spatial characteristics between biodiversity and ecosystem services in a human-dominated watershed. *Ecol. Complex.*, doi:10.1016/j.ecocom.2011.01.007.

Bajželj, B., K. Richards, J. Allwood, and … P. S., 2014a: Importance of food-demand management for climate mitigation. *Nat. Clim.*,. https://www.nature.com/articles/doi:10.1038%2Fnclimate2353 (Accessed May 1, 2018).

Bajželj, B., K. S. Richards, J. M. Allwood, P. Smith, J. S. Dennis, E. Curmi, and C. A. Gilligan, 2014b: Importance of food-demand management for climate mitigation. *Nat. Clim. Chang.*, doi:10.1038/nclimate2353.

Baldos, U. L. C., and T. W. Hertel, 2015: The role of international trade in managing food security risks from climate change. *Food Secur.*, doi:10.1007/s12571-015-0435-z.

Balehegn, M., 2015: Unintended consequences: The ecological repercussions of land grabbing in sub-Saharan Africa. *Environment*, **57**, 4–21, doi:10.1080/00139157.2015.1001687.

Balooni, K., J. M. Pulhin, and M. Inoue, 2008: The effectiveness of decentralisation reforms in the Philippines's forestry sector. *Geoforum*, **39**, 2122–2131, doi:10.1016/j.geoforum.2008.07.003. http://dx.doi.org/10.1016/j.geoforum.2008.07.003.

Baptista, F., L. L. Silva, C. De Visser, J. Gołaszewski, A. Meyer-Aurich, D. Briassoulis, H. Mikkola, and D. Murcho, 2013: Energy Efficiency in Agriculture. https://dspace.uevora.pt/rdpc/bitstream/10174/8648/1/Energy_efficiency_in_agriculture_Lisbon.pdf (Accessed June 1, 2018).

Barnes, A. P., and S. G. Thomson, 2014: Measuring progress towards sustainable intensification: How far can secondary data go? *Ecol. Indic.*, doi:10.1016/j.ecolind.2013.07.001.

Barnes, A. P., H. Hansson, G. Manevska-Tasevska, S. Shrestha, and S. G. Thomson, 2015: The influence of diversification on long-term viability of the agriculture sector. *Land use policy*, **49**, 404–412, doi:10.1016/j.landusepol.2015.08.023. http://creativecommons.org/licenses/by-nc-nd/4.0/ (Accessed May 31, 2018).

Barnett, J., and J. Palutikof, 2015: The limits to adaptation. A comparative analysis. *Applied Studies in Climate Adaptation 1*, John Wiley & Sons, Ltd.

Barry, L. E., R. T. Yao, D. R. Harrison, U. H. Paragahawewa, and D. J. Pannell, 2014: Enhancing ecosystem services through afforestation: How policy can help. *Land use policy*, doi:10.1016/j.landusepol.2014.03.012.

Barthel, S., and C. Isendahl, 2013: Urban gardens, agriculture, and water management: Sources of resilience for long-term food security in cities. *Ecol. Econ.*, **86**, 224–234, doi:10.1016/J.ECOLECON.2012.06.018. https://www.sciencedirect.com/science/article/pii/S0921800912002431 (Accessed June 1, 2018).

Baudron, F., and Coauthors, 2015: Re-examining appropriate mechanization in Eastern and Southern Africa: two-wheel tractors, conservation agriculture, and private sector involvement. *Food Secur.*, **7**, 889–904, doi:10.1007/s12571-015-0476-3. http://link.springer.com/10.1007/s12571-015-0476-3 (Accessed June 1, 2018).

Bauer, N., and Coauthors, 2017: Shared Socio-Economic Pathways of the Energy Sector – Quantifying the Narratives. *Glob. Environ. Chang.*, doi:10.1016/j.gloenvcha.2016.07.006.

Baumhardt, R. L., B. A. Stewart, and U. M. Sainju, 2015: North American soil degradation: Processes, practices, and mitigating strategies. *Sustain.*, **7**, 2936–2960, doi:10.3390/su7032936.

Baveye, P. C., J. Berthelin, D. Tessier, and G. Lemaire, 2018: The "4 per 1000" initiative: A credibility issue for the soil science community? *Geoderma*, **309**, 118–123, doi:10.1016/j.geoderma.2017.05.005. http://linkinghub.elsevier.com/retrieve/pii/S0016706117306171 (Accessed May 30, 2018).

Beg, N., and Coauthors, 2002: Linkages between climate change and sustainable development. *Clim. Policy*, doi:10.1016/S1469-3062(02)00028-1.

Benis, K., and P. F.- Production, 2017: Potential mitigation of the environmental impacts of food systems through urban and peri-urban agriculture (UPA)–a life cycle assessment approach. *J. Clean. Prod.*,. https://www.sciencedirect.com/science/article/pii/S0959652616306552 (Accessed May 1, 2018).

Bennett, A. B., and R. Isaacs, 2014: Landscape composition influences pollinators and pollination services in perennial biofuel plantings. *Agric. Ecosyst. Environ.*, doi:10.1016/j.agee.2014.04.016.

Bennetzen, E. H., P. Smith, and J. R. Porter, 2016a: Agricultural production and greenhouse gas emissions from world regions—The major trends over 40 years. *Glob. Environ. Chang.*, **37**, 43–55, doi:10.1016/J.GLOENVCHA.2015.12.004. https://www.sciencedirect.com/science/article/pii/S0959378015300753 (Accessed May 30, 2018).

——, ——, and ——, 2016b: Decoupling of greenhouse gas emissions from global agricultural production: 1970-2050. *Glob. Chang. Biol.*, **22**, 763–781, doi:10.1111/gcb.13120. http://doi.wiley.com/10.1111/gcb.13120 (Accessed May 30, 2018).

Benson, S. M., 2014: Negative-emissions insurance. *Science*, **344**, 1431, doi:10.1126/science.1257423. http://www.ncbi.nlm.nih.gov/pubmed/24970054 (Accessed December 24, 2017).

Berman, R., C. Quinn, and J. Paavola, 2012: The role of institutions in the transformation of coping capacity to sustainable adaptive capacity. *Environ. Dev.*, doi:10.1016/j.envdev.2012.03.017.

Bhattacharyya, R., and Coauthors, 2015: Soil Degradation in India: Challenges and Potential Solutions. *Sustainability*, **7**, 3528–3570, doi:10.3390/su7043528. http://www.mdpi.com/2071-1050/7/4/3528/.

Bhattamishra, R., and C. B. Barrett, 2010: Community-based risk management arrangements: A Review. *World Dev.*, doi:10.1016/j.worlddev.2009.12.017.

Bidwell, D., T. Dietz, and D. Scavia, 2013: Fostering knowledge networks for climate adaptation. *Nat. Clim. Chang.*, doi:10.1038/nclimate1931.

Biggs, E. M., and Coauthors, 2015: Sustainable development and the water-energy-food nexus: A perspective on livelihoods. *Environ. Sci. Policy*, **54**, 389–397, doi:10.1016/j.envsci.2015.08.002. http://dx.doi.org/10.1016/j.envsci.2015.08.002.

Billen, G., L. Lassaletta, J. Garnier, and … J. L. N., 2018: Opening to distant markets or local reconnection of agro-food systems? Environmental consequences at regional and global scales. *Farming Syst.*,.

Binns, J. A., P. M. Illgner, and E. L. Nel, 2001: Water shortage, deforestation and development: South Africa's working for water programme. *L. Degrad. Dev.*, **12**, 341–355, doi:10.1002/ldr.455.

Birthal, P. S., D. Roy, and D. S. Negi, 2015: Assessing the Impact of Crop Diversification on Farm Poverty in India. *World Dev.*, **72**, 70–92, doi:10.1016/J.WORLDDEV.2015.02.015. https://www.sciencedirect.com/science/article/pii/S0305750X15000480 (Accessed May 31, 2018).

Bisbis, M. B., N. Gruda, and M. Blanke, 2018: Potential impacts of climate change on vegetable production and product quality – A review. *J. Clean. Prod.*, doi:10.1016/j.jclepro.2017.09.224.

Bisht, I. ., P. . Mehta, K. . Negi, S. . Verma, R. . Tyagi, and S. . Garkoti, 2018: Farmers' rights, local food systems, and sustainable household dietary diversification: A case of Uttarakhand Himalaya in north-western India. *Agroecol. Sustain. Food Syst.*, **42**, 77–113, doi:10.1080/21683565.2017.1363118. https://www.tandfonline.com/doi/full/10.1080/21683565.2017.1363118 (Accessed June 1, 2018).

Blackman, A., L. Corral, E. S. Lima, and G. P. Asner, 2017: Titling indigenous communities protects forests in the Peruvian Amazon. *Proc. Natl. Acad. Sci.*, **114**, 4123–4128, doi:10.1073/pnas.1603290114. http://www.pnas.org/lookup/doi/10.1073/pnas.1603290114.

Blanco-Canqui, H., and R. Lal, 2009: Crop Residue Removal Impacts on Soil Productivity and Environmental Quality. *Crit. Rev. Plant Sci.*, doi:10.1080/07352680902776507.

Blum, W. E. H., 2013: Soil and Land Resources for Agricultural Production: General Trends and Future Scenarios-A Worldwide Perspective. *Int. Soil Water Conserv. Res.*, doi:10.1016/S2095-6339(15)30026-5.

Böcher, M., and M. Krott, 2014: The RIU model as an analytical framework for scientific knowledge transfer: the case of the "decision support system forest and climate change." *Biodivers. Conserv.*, doi:10.1007/s10531-014-0820-5.

Bodirsky, B. L., and Coauthors, 2014: Reactive nitrogen requirements to feed the world in 2050 and potential to mitigate nitrogen pollution. *Nat. Commun.*, **5**, doi:10.1038/ncomms4858.

Boelee E, C. T., and K. E, 2011: *An Ecosystem Services Approach to Water and Food Security*. 35 pp.

Bogdanski, A., 2012: Integrated food–energy systems for climate-smart agriculture. *Agric. Food Secur.*, **1**, 9, doi:10.1186/2048-7010-1-9. http://agricultureandfoodsecurity.biomedcentral.com/articles/10.1186/2048-7010-1-9 (Accessed June 1, 2018).

Bolte, A., L. Hilbrig, B. Grundmann, F. Kampf, J. Brunet, and A. Roloff, 2010: Climate change impacts on stand structure and competitive interactions in a southern Swedish spruce-beech forest. *Eur. J. For. Res.*, doi:10.1007/s10342-009-0323-1.

Bommarco, R., D. Kleijn, and S. G. Potts, 2013: Ecological intensification: Harnessing ecosystem services for food security. *Trends Ecol. Evol.*, doi:10.1016/j.tree.2012.10.012.

Bonan, G. B., 2008: Forests and climate change: Forcings, feedbacks, and the climate benefits of forests.

Science (80-.)., doi:10.1126/science.1155121.

Bonn, A., M. Reed, C. Evans, H. Joosten, and C. B. Services, 2014: Investing in nature: developing ecosystem service markets for peatland restoration. *Ecosystem*,. http://www.sciencedirect.com/science/article/pii/S2212041614000692 (Accessed December 20, 2017).

Bonsch, M., and Coauthors, 2015: Environmental flow provision: Implications for agricultural water and land-use at the global scale. *Glob. Environ. Chang.*, **30**, 113–132, doi:10.1016/j.gloenvcha.2014.10.015.

Borras, S. M., R. Hall, I. Scoones, B. White, and W. Wolford, 2011: Towards a better understanding of global land grabbing: An editorial introduction. *J. Peasant Stud.*, **38**, 209–216, doi:10.1080/03066150.2011.559005.

Bottero, A., A. W. D'Amato, B. J. Palik, C. C. Kern, J. B. Bradford, and S. S. Scherer, 2017: Influence of Repeated Prescribed Fire on Tree Growth and Mortality in Pinus resinosa Forests, Northern Minnesota. *For. Sci.*, **63**, 94–100, doi:10.5849/forsci.16-035.

Bouget, C., A. Lassauce, and M. Jonsell, 2012: Effects of fuelwood harvesting on biodiversity — a review focused on the situation in Europe. *Can. J. For. Res.*, doi:10.1139/x2012-078.

Bouwer, L. M., E. Papyrakis, J. Poussin, C. Pfurtscheller, and A. H. Thieken, 2014: The Costing of Measures for Natural Hazard Mitigation in Europe. *Nat. Hazards Rev.*, **15**, 4014010, doi:10.1061/(ASCE)NH.1527-6996.0000133. http://ascelibrary.org/doi/10.1061/%28ASCE%29NH.1527-6996.0000133 (Accessed June 1, 2018).

Bowman, A., 2015: Sovereignty, Risk and Biotechnology: Zambia's 2002 GM Controversy in Retrospect. *Dev. Change*, **46**, 1369–1391, doi:10.1111/dech.12196. http://doi.wiley.com/10.1111/dech.12196 (Accessed June 1, 2018).

Boysen, L. R., W. Lucht, and D. Gerten, 2017a: Trade-offs for food production, nature conservation and climate limit the terrestrial carbon dioxide removal potential. *Glob. Chang. Biol.*, doi:10.1111/gcb.13745.

——, ——, ——, V. Heck, T. M. Lenton, and H. J. Schellnhuber, 2017b: The limits to global-warming mitigation by terrestrial carbon removal. *Earth's Futur.*, doi:10.1002/2016EF000469.

Bradford, K. J., P. Dahal, J. Van Asbrouck, K. Kunusoth, P. Bello, J. Thompson, and F. Wu, 2018: The dry chain: Reducing postharvest losses and improving food safety in humid climates. *Trends Food Sci. Technol.*, doi:10.1016/j.tifs.2017.11.002.

Brenkert, A. L., and E. L. Malone, 2005: Modeling vulnerability and resilience to climate change: A case study of India and Indian states. *Clim. Change*, doi:10.1007/s10584-005-5930-3.

Briber, B. M., L. R. Hutyra, A. B. Reinmann, S. M. Raciti, V. K. Dearborn, C. E. Holden, and A. L. Dunn, 2015: Tree Productivity Enhanced with Conversion from Forest to Urban Land Covers. *PLoS One*, **10**, e0136237, doi:10.1371/journal.pone.0136237. http://dx.plos.org/10.1371/journal.pone.0136237 (Accessed May 31, 2018).

Brindha, K. and Pavelic, P., 2016: Identifying priority watersheds to mitigate flood and drought impacts by novel conjunctive water use management. *Mitig. Adapt. Strateg. Glob. Chang.*, 2–17, doi:10.1007/s12665-015-4989-z.

Brinkley, C., and E. Birch, 2016: Feeding cities: Charting a research and practice agenda toward food security. *Food Syst.*,. http://www.foodsystemsjournal.org/index.php/fsj/article/view/190 (Accessed May 1, 2018).

——, ——, and A. Keating, 2016: Feeding cities: Charting a research and practice agenda toward food security. *Agric. Food Secur.*,. http://www.foodsystemsjournal.org/index.php/fsj/article/view/190 (Accessed May 1, 2018).

Broadmeadow, M. S. J., D. Ray, and C. J. A. Samuel, 2005: Climate change and the future for broadleaved tree species in Britain. *Forestry*, doi:10.1093/forestry/cpi014.

Brockerhoff, E. G., H. Jactel, J. A. Parrotta, C. P. Quine, and J. Sayer, 2008: Plantation forests and biodiversity: Oxymoron or opportunity? *Biodivers. Conserv.*, doi:10.1007/s10531-008-9380-x.

Broegaard, R. B., T. Vongvisouk, and O. Mertz, 2017: Contradictory Land Use Plans and Policies in Laos: Tenure Security and the Threat of Exclusion. *World Dev.*, **89**, 170–183, doi:10.1016/J.WORLDDEV.2016.08.008. https://www.sciencedirect.com/science/article/pii/S0305750X15304848 (Accessed May 31, 2018).

Brooks, N., W. N. Adger, and P. M. Kelly, 2005: The determinants of vulnerability and adaptive capacity at the national level and the implications for adaptation. *Glob. Environ. Chang.*, doi:10.1016/j.gloenvcha.2004.12.006.

Brundu, G., and D. M. Richardson, 2016: Planted forests and invasive alien trees in Europe: A Code for managing existing and future plantings to mitigate the risk of negative impacts from invasions. *NeoBiota*, doi:10.3897/neobiota.30.7015.

Bryan, B. A., and N. D. Crossman, 2013: Impact of multiple interacting financial incentives on land use change and the supply of ecosystem services. *Ecosyst. Serv.*, **4**, 60–72, doi:10.1016/J.ECOSER.2013.03.004. https://www.sciencedirect.com/science/article/pii/S221204161300020X (Accessed June 2, 2018).

Bugri, J. T., 2008: The dynamics of tenure security, agricultural production and environmental degradation in Africa: Evidence from stakeholders in north-east Ghana. *Land use policy*, **25**, 271–285, doi:10.1016/j.landusepol.2007.08.002.

Bullock, J. M., K. L. Dhanjal-Adams, A. Milne, T. H. Oliver, L. C. Todman, A. P. Whitmore, and R. F. Pywell, 2017: Resilience and food security: rethinking an ecological concept. *J. Ecol.*, **105**, 880–884, doi:10.1111/1365-2745.12791. http://doi.wiley.com/10.1111/1365-2745.12791 (Accessed June 3, 2018).

Buongiorno, J., and S. Zhu, 2014: Assessing the impact of planted forests on the global forest economy. *New Zeal. J. For. Sci.*, **44**, S2, doi:10.1186/1179-5395-44-S1-S2. http://www.nzjforestryscience.com/content/44/S1/S2 (Accessed June 2, 2018).

Burch, S., P. Berry, and M. Sanders, 2014: Embedding climate change adaptation in biodiversity conservation: A case study of England. *Environ. Sci. Policy*, **37**, 79–90, doi:10.1016/J.ENVSCI.2013.08.014. https://www.sciencedirect.com/science/article/pii/S1462901113001779 (Accessed June 1, 2018).

Burney, J., L. Woltering, M. Burke, R. Naylor, and D. Pasternak, 2010: Solar-powered drip irrigation enhances food security in the Sudano-Sahel. *Proc. Natl. Acad. Sci.*, doi:10.1073/pnas.0909678107.

Bush, R., and G. Martiniello, 2017: Food Riots and Protest: Agrarian Modernizations and Structural Crises. *WORLD Dev.*, **91**, 193–207, doi:10.1016/j.worlddev.2016.10.017.

Bush, S., N. Khiem, and N. Chau, 2010: Is there a business case for small-holders in Vietnamese Pangasius? *Aquac. Asia Mag.*, **15**, 18–23, doi:20113123446. https://www.cabdirect.org/cabdirect/abstract/20113123446 (Accessed June 4, 2018).

Busse, H., W. Jogo, G. Leverson, F. Asfaw, and H. Tesfay, 2017: Prevalence and predictors of stunting and underweight among children under 5 years in Tigray, Ethiopia: Implications for nutrition-sensitive agricultural interventions. *J. Hunger Environ. Nutr.*, 1–20, doi:10.1080/19320248.2017.1393364. https://www.tandfonline.com/doi/full/10.1080/19320248.2017.1393364 (Accessed June 2, 2018).

Bustamante, M., and Coauthors, 2014a: Co-benefits, trade-offs, barriers and policies for greenhouse gas mitigation in the agriculture, forestry and other land use (AFOLU) sector. *Glob. Chang. Biol.*,

20, 3270–3290, doi:10.1111/gcb.12591. http://doi.wiley.com/10.1111/gcb.12591 (Accessed May 30, 2018).

——, and Coauthors, 2014b: Co-benefits, trade-offs, barriers and policies for greenhouse gas mitigation in the agriculture, forestry and other land use (AFOLU) sector. *Glob. Chang. Biol.*, **20**, 3270–3290, doi:10.1111/gcb.12591. http://doi.wiley.com/10.1111/gcb.12591 (Accessed December 18, 2017).

Cai, H., X. Yang, and X. Xu, 2013: Spatiotemporal Patterns of Urban Encroachment on Cropland and Its Impacts on Potential Agricultural Productivity in China. *Remote Sens.*, **5**, 6443–6460, doi:10.3390/rs5126443. http://www.mdpi.com/2072-4292/5/12/6443 (Accessed May 31, 2018).

Cai, X., X. Zhang, and D. Wang, 2011: Land availability for biofuel production. *Environ. Sci. Technol.*, doi:10.1021/es103338e.

Calder, I. R., 2005: *Blue Revolution: Integrated Land and Water Resource Management - Ian R. Calder - Google Books*. Second. Earthscan, London, 347 pp. https://books.google.com.jm/books?hl=en&lr=&id=fnUjZxtHjJcC&oi=fnd&pg=PR2&dq=Calder,+I.R.+(2005)+Blue+revolution:+integrated+land+and+water+resource+management.+Routledge.&ots=IU3nyCbdAu&sig=qAuizNG8xANuYdSUgmEZPbbDapg&redir_esc=y#v=onepage&q=Calder%2C (Accessed December 22, 2017).

Calvin, K., M. Wise, L. Clarke, J. Edmonds, P. Kyle, P. Luckow, and A. Thomson, 2013: Implications of simultaneously mitigating and adapting to climate change: Initial experiments using GCAM. *Clim. Change*, **117**, 545–560, doi:10.1007/s10584-012-0650-y.

——, ——, P. Kyle, P. Patel, L. Clarke, and J. Edmonds, 2014: Trade-offs of different land and bioenergy policies on the path to achieving climate targets. *Clim. Change*, **123**, 691–704, doi:10.1007/s10584-013-0897-y.

Campbell, B. C., and J. R. Veteto, 2015a: Free seeds and food sovereignty: anthropology and grassroots agrobiodiversity conservation strategies in the US South. *J. Polit. Ecol.*, **22**. http://jpe.library.arizona.edu/volume_22/CampbellVeteto.pdf (Accessed June 1, 2018).

——, and ——, 2015b: Free seeds and food sovereignty: anthropology and grassroots agrobiodiversity conservation strategies in the US South. *J. Polit. Ecol.*, **22**. http://jpe.library.arizona.edu/volume_22/CampbellVeteto.pdf (Accessed May 31, 2018).

Campbell, J. E., D. B. Lobell, R. C. Genova, and C. B. Field, 2008: The global potential of bioenergy on abandoned agriculture lands. *Environ. Sci. Technol.*, doi:10.1021/es800052w.

Campbell, J. R., 2015: Development, global change and traditional food security in Pacific Island countries. *Reg. Environ. Chang.*, **15**, 1313–1324, doi:10.1007/s10113-014-0697-6. http://link.springer.com/10.1007/s10113-014-0697-6 (Accessed May 31, 2018).

Canadell, J. G., and M. R. Raupach, 2008: Managing forests for climate change mitigation. *Science*, **320**, 1456–1457, doi:10.1126/science.1155458. http://www.ncbi.nlm.nih.gov/pubmed/18556550 (Accessed June 2, 2018).

Caplow, S., P. Jagger, K. Lawlor, and E. Sills, 2011: Evaluating land use and livelihood impacts of early forest carbon projects: Lessons for learning about REDD+. *Environ. Sci. Policy*, **14**, 152–167, doi:10.1016/j.envsci.2010.10.003.

Capone, R., H. El Bilali, P. Debs, G. Cardone, and N. Driouech, 2014: Food System Sustainability and Food Security: Connecting the Dots. *J. Food Secur.*, **2**, 13–22, doi:10.12691/JFS-2-1-2.

Cardoso, A. S., A. Berndt, A. Leytem, B. J. R. Alves, I. das N. O. de Carvalho, L. H. de Barros Soares, S. Urquiaga, and R. M. Boddey, 2016: Impact of the intensification of beef production in Brazil on greenhouse gas emissions and land use. *Agric. Syst.*, **143**, 86–96, doi:10.1016/J.AGSY.2015.12.007. https://www.sciencedirect.com/science/article/pii/S0308521X15300652 (Accessed June 2, 2018).

Carlson, K. M., L. M. Curran, G. P. Asner, A. M. Pittman, S. N. Trigg, and J. Marion Adeney, 2012:

Carbon emissions from forest conversion by Kalimantan oil palm plantations. *Nat. Clim. Chang.*, doi:10.1038/nclimate1702.

Carlson, K. M., and Coauthors, 2018: Effect of oil palm sustainability certification on deforestation and fire in Indonesia. *Proc. Natl. Acad. Sci. U. S. A.*, **115**, 121–126, doi:10.1073/pnas.1704728114. http://www.ncbi.nlm.nih.gov/pubmed/29229857 (Accessed June 2, 2018).

Carter, D. R., R. T. Fahey, K. Dreisilker, M. B. Bialecki, and M. L. Bowles, 2015: Assessing patterns of oak regeneration and C storage in relation to restoration-focused management, historical land use, and potential trade-offs. *For. Ecol. Manage.*, **343**, 53–62, doi:https://doi.org/10.1016/j.foreco.2015.01.027.

Carter, S., A. M. Manceur, R. Seppelt, K. Hermans-Neumann, M. Herold, and L. Verchot, 2017: Large scale land acquisitions and REDD plus : a synthesis of conflictsand opportunities. *Environ. Res. Lett.*, **12**, doi:10.1088/1748-9326/aa6056.

Cava, M. G. B., N. A. L. Pilon, M. C. Ribeiro, and G. Durigan, 2018: Abandoned pastures cannot spontaneously recover the attributes of old-growth savannas. *J. Appl. Ecol.*, **55**, 1164–1172, doi:10.1111/1365-2664.13046. http://doi.wiley.com/10.1111/1365-2664.13046 (Accessed June 2, 2018).

Ceddia, M. ., U. Gunter, and A. Corriveau-Bourque, 2015: Land tenure and agricultural expansion in Latin America: The role of Indigenous Peoples' and local communities' forest rights. *Glob. Environ. Chang.*, **35**, 316–322, doi:10.1016/j.gloenvcha.2015.09.010.

Cerbu, G. A., B. M. Swallow, and D. Y. Thompson, 2011: Locating REDD: A global survey and analysis of REDD readiness and demonstration activities. *Environ. Sci. Policy*, **14**, 168–180, doi:10.1016/J.ENVSCI.2010.09.007. https://www.sciencedirect.com/science/article/pii/S1462901110001176 (Accessed June 3, 2018).

Cervarich, M., and Coauthors, 2016: The terrestrial carbon budget of South and Southeast Asia. *Environ. Res. Lett.*, doi:10.1088/1748-9326/11/10/105006.

CGIAR, 2016: The drought crisis in the Central Highlands of Vietnam. 1–36.

——, 2017: options for future agriculture production in the South Central Coast region of Vietnam Assessment of potential CSA.

Chaboud, G., and B. Daviron, 2017: Food losses and waste: navigating the inconsistencies. *Glob. Food Sec.*,. https://www.sciencedirect.com/science/article/pii/S2211912416300499 (Accessed May 10, 2018).

Chakravarty, S., S. Ghosh, C. Suresh, A. Dey, and S. Gopal, 2012: Deforestation: Causes, Effects and Control Strategies, Global Perspectives on Sustainable Forest Management. *Global Perspectives on Sustainable Forest Management*, O. Clement, Ed., InTech.

Challinor, A. J., J. Watson, D. B. Lobell, S. M. Howden, D. R. Smith, and N. Chhetri, 2014: A meta-analysis of crop yield under climate change and adaptation Feeding a growing global population in a changing climate presents a significant challenge to society. doi:10.1038/NCLIMATE2153.

Chappell, M., 2018: *Beginning to end hunger: Food and the environment in Belo Horizonte, Brazil, and beyond*. https://books.google.com/books?hl=en&lr=&id=YqtCDwAAQBAJ&oi=fnd&pg=PR9&dq=Food+systems+in+Belo+Horizonte&ots=E8IYMnwq7S&sig=qizWb7JuantT8dDi6X1P4y-kSEk (Accessed May 1, 2018).

Chappell, M. J., J. R. Moore, and A. A. Heckelman, 2016: Participation in a city food security program may be linked to higher ant alpha- and beta-diversity: an exploratory case from Belo Horizonte, Brazil. *Agroecol. Sustain. Food Syst.*, **40**, 804–829, doi:10.1080/21683565.2016.1160020. https://www.tandfonline.com/doi/full/10.1080/21683565.2016.1160020 (Accessed May 1, 2018).

Chaturvedi, V., M. Hejazi, J. Edmonds, L. Clarke, P. Kyle, E. Davies, and M. Wise, 2013a: Climate mitigation policy implications for global irrigation water demand. *Mitig. Adapt. Strateg. Glob.*

Chang., doi:10.1007/s11027-013-9497-4.

——, ——, ——, ——, ——, ——, and ——, 2013b: Climate mitigation policy implications for global irrigation water demand. *Mitig. Adapt. Strateg. Glob. Chang.*, **20**, 389–407, doi:10.1007/s11027-013-9497-4.

Chazdon, R. L., 2017: Landscape Restoration, Natural Regeneration, and the Forests of the Future. *Ann. Missouri Bot. Gard.*, **102**, 251–257, doi:10.3417/2016035.

——, and Coauthors, 2016: Carbon sequestration potential of second-growth forest regeneration in the Latin American tropics. *Sci. Adv.*, **2**.

Cheesman, S., C. Thierfelder, N. S. Eash, G. T. Kassie, and E. Frossard, 2016: Soil carbon stocks in conservation agriculture systems of Southern Africa. *Soil Tillage Res.*, **156**, 99–109, doi:10.1016/J.STILL.2015.09.018. https://www.sciencedirect.com/science/article/pii/S0167198715300350 (Accessed June 5, 2018).

Chen, D., W. Wei, and L. Chen, 2017: Effects of terracing practices on water erosion control in China: A meta-analysis. *Earth-Science Rev.*, **173**, 109–121, doi:10.1016/J.EARSCIREV.2017.08.007. https://www.sciencedirect.com/science/article/pii/S0012825217300090 (Accessed June 3, 2018).

Chen, J., 2007: Rapid urbanization in China: A real challenge to soil protection and food security. *CATENA*, **69**, 1–15, doi:10.1016/J.CATENA.2006.04.019. https://www.sciencedirect.com/science/article/pii/S0341816206000920 (Accessed May 31, 2018).

Chen, W., 2017: Environmental externalities of urban river pollution and restoration: A hedonic analysis in Guangzhou (China). *Landsc. Urban Plan.*, **157**, 170–179. https://www.sciencedirect.com/science/article/pii/S0169204616301177 (Accessed May 1, 2018).

Cherubini, F., S. Vezhapparambu, W. Bogren, R. Astrup, and A. H. Strømman, 2017: Spatial, seasonal, and topographical patterns of surface albedo in Norwegian forests and cropland. *Int. J. Remote Sens.*, doi:10.1080/01431161.2017.1320442.

Chhatre, A., and A. Agrawal, 2009: Trade-offs and synergies between carbon storage and livelihood benefits from forest commons. *Proc. Natl. Acad. Sci.*, **106**, 17667–17670, doi:10.1073/pnas.0905308106. http://www.pnas.org/content/106/42/17667%7B%25%7D5Cnhttp://www.ncbi.nlm.nih.gov/pubmed/19815522%7B%25%7D5Cnhttp://www.pnas.org/content/106/42/17667.full.pdf%7B%25%7D5Cnhttp://www.pnas.org/content/106/42/17667.short.

Chiti, T., V. Mihindou, K. J. Jeffery, Y. Malhi, F. L. De Oliveira, L. J. T. White, and R. Valentini, 2017: Impact of woody encroachment on soil organic carbon storage in the Lopé National Park, Gabon. *Biotropica*, **49**, 9–12, doi:10.1111/btp.12369. http://doi.wiley.com/10.1111/btp.12369 (Accessed June 5, 2018).

Chong, J., 2014: Ecosystem-based approaches to climate change adaptation: progress and challenges. *Int. Environ. Agreements Polit. Law Econ.*, **14**, 391–405, doi:10.1007/s10784-014-9242-9. http://link.springer.com/10.1007/s10784-014-9242-9 (Accessed June 3, 2018).

Chu, J., 2011: Gender and `Land Grabbing' in Sub-Saharan Africa: Women's land rights and customary land tenure. *Development*, **54**, 35–39.

Chum, H., and Coauthors, 2011: Bioenergy. *IPCC Special Report on Renewable Energy Sources and Climate Change Mitigation*, O. Edenhofer et al., Eds., Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Ciais, P., and Coauthors, 2013: Carbon and Other Biogeochemical Cycles. *Clim. Chang. 2013 Phys. Sci. Basis. Contrib. Work. Gr. I to Fifth Assess. Rep. Intergov. Panel Clim. Chang.*, doi:10.1017/CBO9781107415324.015.

Ciccarese, L., A. Mattsson, and D. Pettenella, 2012: Ecosystem services from forest restoration: thinking ahead. *New For.*, **43**, 543–560, doi:10.1007/s11056-012-9350-8.

Claassen, R., F. Carriazo, J. Cooper, D. Hellerstein, and K. Ueda, 2011: *Grassland to Cropland Conversion in the Northern Plains. The Role of Crop Insurance, Commodity, and Disaster Programs*.

Clark, M., and D. Tilman, 2017: Comparative analysis of environmental impacts of agricultural production systems, agricultural input efficiency, and food choice. *Environ. Res. Lett.*, **12**, 64016, doi:10.1088/1748-9326/aa6cd5. http://stacks.iop.org/1748-9326/12/i=6/a=064016?key=crossref.f80d1b1b72259fb25e1f060a8362e9ca (Accessed May 30, 2018).

Clarke, L. E., and Coauthors, 2014: Assessing transformation pathways. *Climate Change 2014: Mitigation of Climate Change. Contribution of Working Group III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*, 413–510 http://scholar.google.com/scholar?hl=en&btnG=Search&q=intitle:Assessing+Transformation+Pathways#4.

Clay, J., 2011: Freeze the footprint of food. *Nature*, doi:10.1038/475287a.

Clover, J., and S. Eriksen, 2009: The effects of land tenure change on sustainability: human security and environmental change in southern African savannas. *Environ. Sci. Policy*, **12**, 53–70, doi:10.1016/j.envsci.2008.10.012.

Cohen, M. J., J. Clapp, and Centre for International Governance Innovation., 2009: *The global food crisis : governance challenges and opportunities*. Wilfrid Laurier University Press, 267 pp. https://books.google.co.nz/books?hl=en&lr=&id=Wa_wKWgbTSgC&oi=fnd&pg=PR5&dq=The+Global+Food+Crisis:+Governance+Challenges+and+Opportunities.+&ots=yWgJADp8R_&sig=w6fauWwuIuQyUQgrtVGWp_0aNAU#v=onepage&q=The Global Food Crisis%3A Governance Challenges and Opportunities.&f=false (Accessed June 1, 2018).

Cohn, A. S., J. Gil, T. Berger, H. Pellegrina, and C. Toledo, 2016: Patterns and processes of pasture to crop conversion in Brazil: Evidence from Mato Grosso State. *Land use policy*, **55**, 108–120, doi:10.1016/J.LANDUSEPOL.2016.03.005. https://www.sciencedirect.com/science/article/pii/S0264837716301909 (Accessed June 2, 2018).

Collins, A. M., 2014: Governing the Global Land Grab: What role for Gender in the VoluntaryGuidelines and the Principles for Responsible Investment? *GLOBALIZATIONS*, **11**, 189–203, doi:10.1080/14747731.2014.887388.

Collins, K., and R. Ison, 2009: Jumping off Arnstein's ladder: Social learning as a new policy paradigm for climate change adaptation. *Environ. Policy Gov.*, doi:10.1002/eet.523.

Collins, M., and Coauthors, 2013: Long-term Climate Change: Projections, Commitments and Irreversibility. *Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*, G.-K.P. T. F. Stocker, D. Qin, Ed., Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Cools, J., D. Innocenti, and S. O'Brien, 2016: Lessons from flood early warning systems. *Environ. Sci. Policy*, **58**, 117–122, doi:10.1016/J.ENVSCI.2016.01.006. https://www.sciencedirect.com/science/article/pii/S1462901116300065 (Accessed June 1, 2018).

Coomes, O. T., and Coauthors, 2015: Farmer seed networks make a limited contribution to agriculture? Four common misconceptions. *Food Policy*, **56**, 41–50, doi:10.1016/J.FOODPOL.2015.07.008. https://www.sciencedirect.com/science/article/pii/S030691921500086X (Accessed June 1, 2018).

Corsi, S., L. V. Marchisio, and L. Orsi, 2017: Connecting smallholder farmers to local markets: Drivers of collective action, land tenure and food security in East Chad. *Land use policy*, **68**, 39–47, doi:10.1016/j.landusepol.2017.07.025.

Coscieme, L., F. M. Pulselli, V. Niccolucci, N. Patrizi, and P. C. Sutton, 2016: Accounting for ``land-grabbing'' from a biocapacity viewpoint. *Sci. Total Environ.*, **539**, 551–559, doi:10.1016/j.scitotenv.2015.09.021.

Coutts, A., and R. Harris, 2013: *A multi-scale assessment of urban heating inMelbourne during an extreme heat event and policy approaches for adaptation*. 64 pp.

Creutzig, F., and Coauthors, 2015: Bioenergy and climate change mitigation: An assessment. *GCB Bioenergy*, doi:10.1111/gcbb.12205.

Crossman, N. D., B. A. Bryan, and D. M. Summers, 2011: Carbon Payments and Low-Cost Conservation. *Conserv. Biol.*, **25**, 835–845, doi:10.1111/j.1523-1739.2011.01649.x. http://doi.wiley.com/10.1111/j.1523-1739.2011.01649.x (Accessed June 2, 2018).

Cunningham, S. C., R. Mac Nally, P. J. Baker, T. R. Cavagnaro, J. Beringer, J. R. Thomson, and R. M. Thompson, 2015: Balancing the environmental benefits of reforestation in agricultural regions. *Perspect. Plant Ecol. Evol. Syst.*, **17**, 301–317, doi:10.1016/J.PPEES.2015.06.001. https://www.sciencedirect.com/science/article/pii/S1433831915000463 (Accessed June 1, 2018).

D'Amato, A. W., J. B. Bradford, S. Fraver, and B. J. Palik, 2011: Forest management for mitigation and adaptation to climate change: Insights from long-term silviculture experiments. *For. Ecol. Manage.*, **262**, 803–816, doi:10.1016/j.foreco.2011.05.014.

D'Odorico, P., Y. He, S. Collins, S. F. J. De Wekker, V. Engel, and J. D. Fuentes, 2013: Vegetation-microclimate feedbacks in woodland-grassland ecotones. *Glob. Ecol. Biogeogr.*, **22**, 364–379, doi:10.1111/geb.12000. http://doi.wiley.com/10.1111/geb.12000 (Accessed June 5, 2018).

——, M. C. Rulli, J. Dell'Angelo, and K. F. Davis, 2017: NEW FRONTIERS OF LAND AND WATER COMMODIFICATION: SOCIO-ENVIRONMENTALCONTROVERSIES OF LARGE-SCALE LAND ACQUISITIONS. *L. Degrad. Dev.*, **28**, 2234–2244, doi:10.1002/ldr.2750.

Dagar, J. C., D. K. Sharma, P. C. Sharma, and A. K. Singh, 2016a: *Innovative saline agriculture*. 1-519 pp.

——, ——, ——, and ——, 2016b: Innovative saline agriculture. *Innov. Saline Agric.*, 1–519, doi:10.1007/978-81-322-2770-0.

Damnyag, L., O. Saastamoinen, M. Appiah, and A. Pappinen, 2012: Role of tenure insecurity in deforestation in Ghana's high forest zone. *For. Policy Econ.*, **14**, 90–98, doi:10.1016/j.forpol.2011.08.006. http://dx.doi.org/10.1016/j.forpol.2011.08.006.

Daniel, S., 2011: Land Grabbing and Potential Implications for World Food Security. *Sustainable Agricultural Development*, 25–42 http://www.springerlink.com/index/10.1007/978-94-007-0519-7_2.

——, 2012: Situating private equity capital in the land grab debate. *J. Peasant Stud.*, **39**, 703–729, doi:10.1080/03066150.2012.674941.

Dasgupta, P., J. F. Morton, D. Dodman, B. Karapinar, F. Meza, M. G. Rivera-Ferre, A. Toure Sarr, and K. E. Vincent, 2014: Rural Areas. *Climate Change 2014: Impacts, Adaptation, and Vulnerability. Part A: Global and Sectoral Aspects. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*.

Datta, K. K., C. De Jong, and O. P. Singh, 2000: Reclaiming salt-affected land through drainage in Haryana, India: A financial analysis. *Agric. Water Manag.*, **46**, 55–71, doi:10.1016/S0378-3774(00)00077-9.

Dauber, J., M. B. Jones, and J. C. Stout, 2010: The impact of biomass crop cultivation on temperate biodiversity. *GCB Bioenergy*, doi:10.1111/j.1757-1707.2010.01058.x.

Davin, E. L., and N. de Noblet-Ducoudre, 2010: Climatic impact of global-scale Deforestation: Radiative versus nonradiative processes. *J. Clim.*, doi:10.1175/2009JCLI3102.1.

Davis, K. F., P. D'Odorico, and M. C. Rulli, 2014: Land grabbing: a preliminary quantification of economic impacts on rurallivelihoods. *Popul. Environ.*, **36**, 180–192, doi:10.1007/s11111-014-0215-2.

Davis, M. P., M. B. David, T. B. Voigt, and C. A. Mitchell, 2015: Effect of nitrogen addition on

Miscanthus ×giganteus yield, nitrogen losses, and soil organic matter across five sites. *GCB Bioenergy*, doi:10.1111/gcbb.12217.

Dawadi, S., and S. Ahmad, 2012: Changing climatic conditions in the Colorado River Basin: Implications for water resources management. *J. Hydrol.*, doi:10.1016/j.jhydrol.2012.02.010.

——, and ——, 2013: Evaluating the impact of demand-side management on water resources under changing climatic conditions and increasing population. *J. Environ. Manage.*, doi:10.1016/j.jenvman.2012.10.015.

Deininger, K., and O. Feder, 2009: Land registration, governance, and development: Evidence and implications for policy. *World Bank Res. Obs.*,. https://academic.oup.com/wbro/article-abstract/24/2/233/1684787 (Accessed May 31, 2018).

Deininger, K., and D. Byerlee, 2012: The rise of large farms in land abundant countries: Do they have a future? *World Dev.*, doi:10.1016/j.worlddev.2011.04.030.

Dell'Angelo, J., P. D'Odorico, M. C. Rulli, and P. Marchand, 2017: The Tragedy of the Grabbed Commons: Coercion and Dispossession in the Global Land Rush. *World Dev.*, **92**, 1–12, doi:10.1016/j.worlddev.2016.11.005.

Demuzere, M., and Coauthors, 2014: Mitigating and adapting to climate change: Multi-functional and multi-scale assessment of green urban infrastructure. *J. Environ. Manage.*, **146**, 107–115, doi:10.1016/j.jenvman.2014.07.025.

DERM, 2011: Salinity management handbook Second edition. 188.

Derpsch, R., T. Friedrich, A. Kassam, and H. Li, 2010: Current Status of Adoption of No-till Farming in the World and Some of its Main Benefits. *Int. J. Agric. Biol. Eng.*, **3**, 1–25, doi:10.25165/IJABE.V3I1.223. https://ijabe.org/index.php/ijabe/article/view/223 (Accessed May 31, 2018).

Descheemaeker, K., S. J. Oosting, S. Homann-Kee Tui, P. Masikati, G. N. Falconnier, and K. E. Giller, 2016: Climate change adaptation and mitigation in smallholder crop–livestock systems in sub-Saharan Africa: a call for integrated impact assessments. *Reg. Environ. Chang.*, **16**, 2331–2343, doi:10.1007/s10113-016-0957-8. http://link.springer.com/10.1007/s10113-016-0957-8 (Accessed June 1, 2018).

Díaz, S., and Coauthors, 2018: Assessing nature's contributions to people. *Science*, **359**, 270–272, doi:10.1126/science.aap8826. http://www.ncbi.nlm.nih.gov/pubmed/29348221 (Accessed June 4, 2018).

DiGiano, M. L., and A. E. Racelis, 2012: Robustness, adaptation and innovation: Forest communities in the wake of Hurricane Dean. *Appl. Geogr.*, doi:10.1016/j.apgeog.2011.10.004.

Dillon, P., and M. Arshad, 2016a: Managed aquifer recharge in integrated water resource management. *Integrated Groundwater Management: Concepts, Approaches and Challenges*.

——, and ——, 2016b: Integrated Groundwater Management. 435–452, doi:10.1007/978-3-319-23576-9.

——, I. Gale, S. Contreras, P. Pavelic, R. Evans, and J. Ward, 2009: Managing aquifer recharge and discharge to sustain irrigation livelihoods under water scarcity and climate change. *IAHS-AISH Publ.*, **330**, 1–12, doi:Cited By (since 1996) 2\nExport Date 9 July 2012\nSource Scopus.

Djalante, R., F. Thomalla, M. S. Sinapoy, and M. Carnegie, 2012: Building resilience to natural hazards in Indonesia: Progress and challenges in implementing the Hyogo Framework for Action. *Nat. Hazards*, doi:10.1007/s11069-012-0106-8.

Dobbs, C., D. Kendal, and C. Nitschke, 2014: Multiple ecosystem services and disservices of the urban forest establishing their connections with landscape structure and sociodemographics. *Ecol. Indic.*,. http://www.sciencedirect.com/science/article/pii/S1470160X14000594 (Accessed December 20, 2017).

Dodman, D., and D. Satterthwaite, 2008: Institutional capacity, climate change adaptation and the urban poor. *Ids Bull.*, doi:10.1111/j.1759-5436.2008.tb00478.x.

Don, A., and Coauthors, 2012: Land-use change to bioenergy production in Europe: implications for the greenhouse gas balance and soil carbon. *Glob. Chang. Biol. Bioenergy*, doi:10.1111/j.1757-1707.2011.01116.x.

Dooley, K., and S. Kartha, 2018: Land-based negative emissions: risks for climate mitigation and impacts on sustainable development. *Int. Environ. Agreements Polit. Law Econ.*, **18**, 79–98, doi:10.1007/s10784-017-9382-9.

Dresner, M., C. Handelman, S. Braun, and G. Rollwagen-Bollens, 2015: Environmental identity, pro-environmental behaviors, and civic engagement of volunteer stewards in Portland area parks. *Environ. Educ. Res.*, **21**, 991–1010, doi:10.1080/13504622.2014.964188.

Duchelle, A. E., and Coauthors, 2014: Linking forest tenure reform, environmental compliance, and incentives: Lessons from redd+ initiatives in the brazilian amazon. *World Dev.*, **55**, 53–67, doi:10.1016/j.worlddev.2013.01.014.

Duru, M., and Coauthors, 2015: How to implement biodiversity-based agriculture to enhance ecosystem services: a review. *Agron. Sustain. Dev.*, **35**, 1259–1281, doi:10.1007/s13593-015-0306-1. http://link.springer.com/10.1007/s13593-015-0306-1 (Accessed June 3, 2018).

Duveneck, M. J., and R. M. Scheller, 2015: Climate-suitable planting as a strategy for maintaining forest productivity and functional diversity. *Ecol. Appl.*, **25**, 1653–1668, doi:10.1890/14-0738.1.

Edelenbosch, O. Y., and Coauthors, 2016: Decomposing passenger transport futures: Comparing results of global integrated assessment models. *Transp. Res. Part D Transp. Environ.*, doi:10.1016/j.trd.2016.07.003.

Edwards-Jones, G., and Coauthors, 2008: Testing the assertion that "local food is best": the challenges of an evidence-based approach. *Trends Food Sci. Technol.*, doi:10.1016/j.tifs.2008.01.008.

Elliott, J., and Coauthors, 2014: Constraints and potentials of future irrigation water availability on agricultural production under climate change. *Proc. Natl. Acad. Sci.*, **111**, 3239–3244, doi:10.1073/pnas.1222474110. http://www.pnas.org/lookup/doi/10.1073/pnas.1222474110.

Ellis, E. C., and N. Ramankutty, 2008: Putting people in the map: anthropogenic biomes of the world. *Front. Ecol. Environ.*, **6**, 439–447, doi:10.1890/070062. http://doi.wiley.com/10.1890/070062 (Accessed June 2, 2018).

Ellison, D., and Coauthors, 2017: Trees, forests and water: Cool insights for a hot world. *Glob. Environ. Chang.*, doi:10.1016/j.gloenvcha.2017.01.002.

Enki, M., K. Belay, and L. Dadi, 2001: Determinants of adoption of physical soil conservation measures in central highlands of ethiopia the case of three districts of North Shewa. *Agrekon*, **40**, 293–315, doi:10.1080/03031853.2001.9524955.

Ensor, J., R. Berger, and S. Huq, 2014: *Community-based adaptation to climate change: Emerging lessons*. Practical Action Publishing, Rugby, 183--197 pp.

Epron, D., C. Plain, F.-K. Ndiaye, P. Bonnaud, C. Pasquier, and J. Ranger, 2016: Effects of compaction by heavy machine traffic on soil fluxes of methane and carbon dioxide in a temperate broadleaved forest. *For. Ecol. Manage.*, **382**, 1–9, doi:10.1016/J.FORECO.2016.09.037. https://www.sciencedirect.com/science/article/pii/S037811271630648X (Accessed May 31, 2018).

Erskine, P. D., D. Lamb, and M. Bristow, 2006: Tree species diversity and ecosystem function: Can tropical multi-species plantations generate greater productivity? *For. Ecol. Manage.*, doi:10.1016/j.foreco.2006.05.013.

Evans, R. G., and E. J. Sadler, 2008: Methods and technologies to improve efficiency of water use. *Water Resour. Res.*, **44**, 1–15, doi:10.1029/2007WR006200.

Fafchamps, M., C. Udry, and K. Czukas, 1998: Drought and saving in West Africa: are livestock a buffer stock? *J. Dev. Econ.*, **55**, 273–305, doi:10.1016/S0304-3878(98)00037-6. https://www.sciencedirect.com/science/article/pii/S0304387898000376 (Accessed June 1, 2018).

Fairbairn, M., 2015: Foreignization, Financialization and Land Grab Regulation. *J. Agrar. Chang.*, **15**, 581–591, doi:10.1111/joac.12112.

Fakhruddin, S. H. M., A. Kawasaki, and M. S. Babel, 2015: Community responses to flood early warning system: Case study in Kaijuri Union, Bangladesh. *Int. J. Disaster Risk Reduct.*, **14**, 323–331, doi:10.1016/J.IJDRR.2015.08.004. https://www.sciencedirect.com/science/article/pii/S2212420915300509 (Accessed June 1, 2018).

Fameree, C., 2016: Political contestations around land deals: insights from Peru. *Can. J. Dev. Stud. Can. D ETUDES DUDEVELOPPEMENT*, **37**, 541–559, doi:10.1080/02255189.2016.1175340.

FAO, 2006: *Fire management: voluntary guidelines. Principles and strategic actions.* Rome,.

——, 2011: *The State of the World's land and water resources for Food and Agriculture. Managing systems at risk*.

——, 2017: *Voluntary Guidelines for Sustainable Soil Management*.

FAO (Food and Agriculture Organization), 2010: *Climate-Smart agriculture : policies, practice and financing for food security, adaptation and migration.*

——, 2011: *The State of the World's Land and Water Resources for Food and Agriculture*.

FAO and ITPS, 2015: *Status of the World's Soil Resources (SWSR) - Main Report*.

Favero, A., and R. Mendelsohn, 2017: The Land-Use Consequences of Woody Biomass with More Stringent Climate Mitigation Scenarios. *J. Environ. Prot. (Irvine,. Calif).*, doi:10.4236/jep.2017.81006.

Federici, C., G. Fattore, and G. Rudolfsson, 2016: The Effects Of Food Prices On Weight Outcomes. A Systematic Review Of The Moderating Effects Of Income And Initial BMI. *Value Heal.*,. http://www.valueinhealthjournal.com/article/S1098-3015(16)31993-3/abstract (Accessed May 1, 2018).

Fehlenberg, V., M. Baumann, N. I. Gasparri, M. Piquer-Rodriguez, G. Gavier-Pizarro, and T. Kuemmerle, 2017: The role of soybean production as an underlying driver of deforestation in the South American Chaco. *Glob. Environ. Chang.*, **45**, 24–34, doi:10.1016/J.GLOENVCHA.2017.05.001. https://www.sciencedirect.com/science/article/pii/S0959378017305964 (Accessed June 3, 2018).

Fenton, A., J. Paavola, and A. Tallontire, 2015: Microfinance and climate change adaptation: an overview of the current literature. *Enterp. Dev. Microfinance*, doi:10.3362/1755-1986.2015.023.

Fenton, A., J. Paavola, and A. Tallontire, 2017: The Role of Microfinance in Household Livelihood Adaptation in Satkhira District, Southwest Bangladesh. *World Dev.*, doi:10.1016/j.worlddev.2016.12.004.

Fereres, E., and M. A. Soriano, 2007: Deficit irrigation for reducing agricultural water use. *Journal of Experimental Botany*.

Fernández-Bayo, J. D., and Coauthors, 2017: Assessment of Two Solid Anaerobic Digestate Soil Amendments for Effects on Soil Quality and Biosolarization Efficacy. *J. Agric. Food Chem.*, doi:10.1021/acs.jafc.6b04816.

Few, R., A. Martin, and N. Gross-Camp, 2017: Trade-offs in linking adaptation and mitigation in the forests of the Congo Basin. *Reg. Environ. Chang.*, **17**, 851–863, doi:10.1007/s10113-016-1080-6.

Findell, K. L., A. Berg, P. Gentine, J. P. Krasting, B. R. Lintner, S. Malyshev, J. A. Santanello, and E. Shevliakova, 2017: The impact of anthropogenic land use and land cover change on regional climate extremes. *Nat. Commun.*, doi:10.1038/s41467-017-01038-w.

Finley-Brook, M., 2007: Indigenous land tenure insecurity fosters illegal logging in Nicaragua. *Int. For. Rev.*, **9**, 850–864, doi:10.1505/ifor.9.4.850.

Foley, J. A., and Coauthors, 2011: Solutions for a cultivated planet. *Nature*, doi:10.1038/nature10452.

Forsyth, T., 2013: Community-based adaptation: A review of past and future challenges. *Wiley Interdiscip. Rev. Clim. Chang.*, doi:10.1002/wcc.231.

Frank, S., and Coauthors, 2017a: Reducing greenhouse gas emissions in agriculture without compromising food security? *Environ. Res. Lett.*, doi:10.1088/1748-9326/aa8c83.

——, and Coauthors, 2017b: Reducing greenhouse gas emissions in agriculture without compromising food security? *Environ. Res. Lett.*, **12**, 105004, doi:10.1088/1748-9326/aa8c83. http://stacks.iop.org/1748-9326/12/i=10/a=105004?key=crossref.000b6ab1748d1b0af07f66e0f496f0a3 (Accessed June 1, 2018).

Frankl, A., J. Poesen, M. Haile, J. Deckers, and J. Nyssen, 2013: Quantifying long-term changes in gully networks and volumes in dryland environments: The case of Northern Ethiopia. *Geomorphology*, **201**, 254–263, doi:10.1016/J.GEOMORPH.2013.06.025. https://www.sciencedirect.com/science/article/pii/S0169555X13003474 (Accessed June 2, 2018).

Freeman, J., L. Kobziar, E. W. Rose, and W. Cropper, 2017: A critique of the historical-fire-regime concept in conservation. *Conserv. Biol.*, **31**, 976–985, doi:10.1111/cobi.12942. http://doi.wiley.com/10.1111/cobi.12942 (Accessed May 31, 2018).

Friis, C., and J. Ø. Nielsen, 2016: Small-scale land acquisitions, large-scale implications: Exploring the case of Chinese banana investments in Northern Laos. *Land use policy*, **57**, 117–129, doi:10.1016/j.landusepol.2016.05.028.

Fujimori, S., and ..., A multi-model assessment of food security implications of well below 2°C scenarios.

Fuss, S., and Coauthors, 2014: Betting on negative emissions. *Nat. Clim. Chang.*, doi:10.1038/nclimate2392.

Fuss, S., and Coauthors, 2016: Research priorities for negative emissions. *Environ. Res. Lett.*, doi:10.1088/1748-9326/11/11/115007.

Fuss, S., and Coauthors, 2018: Negative emissions - Part 2: Costs, potentials and side effects. *Environ. Res. Lett.*, **13**, 2–4, doi:https://doi.org/10.1088/1748-9326/aabf9f.

Gabay, M., and M. Alam, 2017: Community forestry and its mitigation potential in the Anthropocene: The importance of land tenure governance and the threat of privatization. *For. Policy Econ.*, **79**, 26–35, doi:10.1016/j.forpol.2017.01.011. http://dx.doi.org/10.1016/j.forpol.2017.01.011.

Gabriel, K. M. A., and W. R. Endlicher, 2011: Urban and rural mortality rates during heat waves in Berlin and Brandenburg, Germany. *Environ. Pollut.*, **159**, 2044–2050, doi:10.1016/J.ENVPOL.2011.01.016. https://www.sciencedirect.com/science/article/pii/S0269749111000388 (Accessed June 2, 2018).

Gale, N., A. Halim, M. Horsburgh, and S. C. Thomas, 2017: Comparative responses of early-successional plants to charcoal soil amendments. *Ecosphere*, **8**, e01933, doi:10.1002/ecs2.1933.

Gallina, V., S. Torresan, A. Critto, A. Sperotto, T. Glade, and A. Marcomini, 2016: A review of multi-risk methodologies for natural hazards: Consequences and challenges for a climate change impact assessment. *J. Environ. Manage.*, **168**, 123–132, doi:10.1016/J.JENVMAN.2015.11.011. https://www.sciencedirect.com/science/article/pii/S0301479715303650 (Accessed May 31, 2018).

Garnett, T., 2011: Where are the Best Opportunities for Reducing Greenhouse Gas Emissions in the Food System? *Food Policy*, doi:10.1016/j.foodpol.2010.10.010.

Garschagen, M., 2016: Decentralizing urban disaster risk management in a centralized system?

Agendas, actors and contentions in Vietnam. *Habitat Int.*, doi:10.1016/j.habitatint.2015.08.030.

Gaveau, D. L. A., D. Sheil, Husnayaen, M. A. Salim, S. Arjasakusuma, M. Ancrenaz, P. Pacheco, and E. Meijaard, 2016: Rapid conversions and avoided deforestation: examining four decades of industrial plantation expansion in Borneo. *Sci. Rep.*, **6**, 32017, doi:10.1038/srep32017. http://www.nature.com/articles/srep32017 (Accessed June 2, 2018).

Gebremedhin, B., and S. M. Swinton, 2003: Investment in soil conservation in northern Ethiopia: The role of land tenure security and public programs. *Agric. Econ.*, **29**, 69–84, doi:10.1016/S0169-5150(03)00022-7.

Gebremeskel, G., T. G. Gebremicael, and A. Girmay, 2018: Economic and environmental rehabilitation through soil and water conservation, the case of Tigray in northern Ethiopia. *J. Arid Environ.*, **151**, 113–124, doi:10.1016/J.JARIDENV.2017.12.002. https://www.sciencedirect.com/science/article/pii/S014019631730232X (Accessed June 2, 2018).

Gibson, J., G. Boe-Gibson, and G. Stichbury, 2015: Urban land expansion in India 1992–2012. *Food Policy*, **56**, 100–113, doi:10.1016/J.FOODPOL.2015.08.002. https://www.sciencedirect.com/science/article/pii/S0306919215001013 (Accessed May 31, 2018).

Gilbert, C. L., 2012: International agreements to manage food price volatility. *Glob. Food Sec.*, **1**, 134–142, doi:10.1016/J.GFS.2012.10.001. https://www.sciencedirect.com/science/article/pii/S2211912412000168 (Accessed June 1, 2018).

Giné, X., R. Townsend, and J. Vickery, 2008: Patterns of rainfall insurance participation in rural India. *World Bank Econ. Rev.*, doi:10.1093/wber/lhn015.

Gleeson, T., Y. Wada, M. F. P. Bierkens, and L. P. H. Van Beek, 2012: Water balance of global aquifers revealed by groundwater footprint. *Nature*, **488**, 197–200, doi:10.1038/nature11295.

Göbel, C., N. Langen, and A. Blumenthal, 2015: Cutting food waste through cooperation along the food supply chain. *Sustainability*,. http://www.mdpi.com/2071-1050/7/2/1429/htm (Accessed May 1, 2018).

Godfray, H. C. J., and T. Garnett, 2014: Food security and sustainable intensification. *Philos. Trans. R. Soc. Lond. B. Biol. Sci.*, **369**, 20120273, doi:10.1098/rstb.2012.0273. http://www.ncbi.nlm.nih.gov/pubmed/24535385 (Accessed May 30, 2018).

Godfray, H. C. J., and Coauthors, 2010: Food security: The challenge of feeding 9 billion people. *Science (80-.).*, doi:10.1126/science.1185383.

Godinho, S., N. Guiomar, R. Machado, P. Santos, P. Sá-Sousa, J. P. Fernandes, N. Neves, and T. Pinto-Correia, 2016: Assessment of environment, land management, and spatial variables on recent changes in montado land cover in southern Portugal. *Agrofor. Syst.*, **90**, 177–192, doi:10.1007/s10457-014-9757-7. http://link.springer.com/10.1007/s10457-014-9757-7 (Accessed June 2, 2018).

Goetz, A., 2013: Private Governance and Land Grabbing: The Equator Principles and theRoundtable on Sustainable Biofuels. *GLOBALIZATIONS*, **10**, 199–204, doi:10.1080/14747731.2013.760949.

Golay, C., and I. Biglino, 2013: Human Rights Responses to Land Grabbing: a right to food perspective. *Third World Q.*, **34**, 1630–1650, doi:10.1080/01436597.2013.843853.

Golub, A. A., B. B. Henderson, T. W. Hertel, P. J. Gerber, S. K. Rose, and B. Sohngen, 2013: Global climate policy impacts on livestock, land use, livelihoods, and food security. *Proc. Natl. Acad. Sci.*, doi:10.1073/pnas.1108772109.

Goodwin, B. K., and V. H. Smith, 2003: An Ex Post Evaluation of the Conservation Reserve, Federal Crop Insurance, and Other Government Programs: Program Participation and Soil Erosion. *J. Agric. Resour. Econ.*, **28**, 201–216, doi:10.2307/40987182. http://www.jstor.org/stable/40987182 (Accessed June 1, 2018).

Goodwin, B. K., and V. H. Smith, 2013: What Harm Is Done By Subsidizing Crop Insurance? *Am. J. Agric. Econ.*, **95**, 489–497, doi:10.1093/ajae/aas092. https://academic.oup.com/ajae/article-lookup/doi/10.1093/ajae/aas092 (Accessed June 1, 2018).

——, M. L. Vandeveer, and J. L. Deal, 2004: An Empirical Analysis of Acreage Effects of Participation in the Federal Crop Insurance Program. *Am. J. Agric. Econ.*, **86**, 1058–1077, doi:10.1111/j.0002-9092.2004.00653.x. https://academic.oup.com/ajae/article-lookup/doi/10.1111/j.0002-9092.2004.00653.x (Accessed June 1, 2018).

Graham, V., S. G. Laurance, A. Grech, and O. Venter, 2017: Spatially explicit estimates of forest carbon emissions, mitigation costs and REDD+ opportunities in Indonesia. *Environ. Res. Lett.*, **12**, 44017, doi:10.1088/1748-9326/aa6656. http://stacks.iop.org/1748-9326/12/i=4/a=044017?key=crossref.969b6ca7c50dd7aa01dc8119235e7322 (Accessed June 2, 2018).

Grant, E., and O. Das, 2015: Land Grabbing, Sustainable Development and Human Rights. *Transnatl. Environ. LAW*, **4**, 289–317, doi:10.1017/S2047102515000023.

Grassi, G., J. House, F. Dentener, S. Federici, M. Den Elzen, and J. Penman, 2017: The key role of forests in meeting climate targets requires science for credible mitigation. *Nat. Clim. Chang.*, doi:10.1038/nclimate3227.

Green, D., and G. Raygorodetsky, 2010: Indigenous knowledge of a changing climate. *Clim. Change*, doi:10.1007/s10584-010-9804-y.

Greene, R., W. Timms, P. Rengasamy, M. Arshad, and R. Cresswell, 2016: *Integrated Groundwater Management*. 377-412 pp. http://link.springer.com/10.1007/978-3-319-23576-9.

Greve, M., B. Reyers, A. Mette Lykke, and J. C. Svenning, 2013: Spatial optimization of carbon-stocking projects across Africa integrating stocking potential with co-benefits and feasibility. *Nat. Commun.*, doi:10.1038/ncomms3975.

Grey, D., and C. W. Sadoff, 2007: Sink or Swim? Water security for growth and development. *Water Policy*, **9**, 545–571, doi:10.2166/wp.2007.021.

Griscom, B. W., and Coauthors, 2017a: Natural climate solutions. *Proc. Natl. Acad. Sci. U. S. A.*, **114**, 11645–11650, doi:10.1073/pnas.1710465114. http://www.ncbi.nlm.nih.gov/pubmed/29078344 (Accessed May 30, 2018).

——, and Coauthors, 2017b: Natural climate solutions. *Proc. Natl. Acad. Sci. U. S. A.*, **114**, 11645–11650, doi:10.1073/pnas.1710465114. http://www.ncbi.nlm.nih.gov/pubmed/29078344 (Accessed June 2, 2018).

Guariguata, M. R., 2009: Tropical Forest Management and Climate Change Adaptation. *Rev. Estud. Soc.*, **32**, 98–112. http://www.scielo.org.co/scielo.php?pid=S0123-885X2009000100008&script=sci_arttext&tlng=es (Accessed June 3, 2018).

Gunarathna, M. H. J. P., K. Sakai, T. Nakandakari, M. Kazuro, T. Onodera, H. Kaneshiro, H. Uehara, and K. Wakasugi, 2017: Optimized subsurface irrigation system (OPSIS): Beyond traditional subsurface irrigation. *Water (Switzerland)*, doi:10.3390/w9080599.

Gustavsson, J., C. Cederberg, U. Sonesson, R. van Otterdijk, and A. Meybeck, 2011: *Global food losses and food waste - Extent, causes and prevention.*

Haggblade, S., N. M. Me-Nsope, and J. M. Staatz, 2017: Food security implications of staple food substitution in Sahelian West Africa. *Food Policy*, **71**, 27–38, doi:10.1016/J.FOODPOL.2017.06.003. https://www.sciencedirect.com/science/article/pii/S0306919216302366 (Accessed June 1, 2018).

Hajjar, R. F., R. A. Kozak, and J. L. Innes, 2012: Is decentralization leading to “real” decision-making power for forest-dependent communities? Case studies from mexico and Brazil. *Ecol. Soc.*, doi:10.5751/ES-04570-170112.

Hall, J. S., M. S. Ashton, E. J. Garen, and S. Jose, 2011: The ecology and ecosystem services of native trees: Implications for reforestation and land restoration in Mesoamerica. *For. Ecol. Manage.*, doi:10.1016/j.foreco.2010.12.011.

Hall, R., M. Edelman, S. M. Borras Jr., I. Scoones, B. White, and W. Wolford, 2015: Resistance, acquiescence or incorporation? An introduction to landgrabbing and political reactions `from below'. *J. Peasant Stud.*, **42**, 467–488, doi:10.1080/03066150.2015.1036746.

van Hall, R. L., L. H. Cammeraat, S. D. Keesstra, and M. Zorn, 2017: Impact of secondary vegetation succession on soil quality in a humid Mediterranean landscape. *CATENA*, **149**, 836–843, doi:10.1016/J.CATENA.2016.05.021. https://www.sciencedirect.com/science/article/pii/S0341816216301904 (Accessed June 2, 2018).

Hallegatte, S., 2012: A Cost Effective Solution to Reduce Disaster Losses in Developing Countries: Hydro-Meteorological Services, Early Warning, and Evacuation. https://papers.ssrn.com/sol3/papers.cfm?abstract_id=2051341 (Accessed June 1, 2018).

Hallegatte, S., and Coauthors, 2016: *Shock Waves: Managing the Impacts of Climate Change on Poverty*.

Halvorson, A. D., C. S. Snyder, A. D. Blaylock, and S. J. Del Grosso, 2014: Enhanced-Efficiency Nitrogen Fertilizers: Potential Role in Nitrous Oxide Emission Mitigation. *Agron. J.*, **106**, 715, doi:10.2134/agronj2013.0081. https://www.agronomy.org/publications/aj/abstracts/106/2/715 (Accessed June 3, 2018).

Hamilton, S. K., M. Z. Hussain, A. K. Bhardwaj, B. Basso, and G. P. Robertson, 2015: Comparative water use by maize, perennial crops, restored prairie, and poplar trees in the US Midwest. *Environ. Res. Lett.*, doi:10.1088/1748-9326/10/6/064015.

Hammill, a, R. Matthew, and E. McCarter, 2008: Microfinance and climate change adaptation. *IDS Bull.*,.

Hamza, M. A., and W. K. Anderson, 2005: Soil compaction in cropping systems: A review of the nature, causes and possible solutions. *Soil Tillage Res.*, **82**, 121–145, doi:10.1016/J.STILL.2004.08.009. https://www.sciencedirect.com/science/article/pii/S0167198704001849 (Accessed May 31, 2018).

Hansen, M. C., and Coauthors, 2013: High-resolution global maps of 21st-century forest cover change. *Science (80-.).*, doi:10.1126/science.1244693.

Harding, K. J., T. E. Twine, A. VanLoocke, J. E. Bagley, and J. Hill, 2016: Impacts of second-generation biofuel feedstock production in the central U.S. on the hydrologic cycle and global warming mitigation potential. *Geophys. Res. Lett.*, doi:10.1002/2016GL069981.

Haregeweyn, N., and Coauthors, 2015: Soil erosion and conservation in Ethiopia. *Prog. Phys. Geogr.*, **39**, 750–774, doi:10.1177/0309133315598725. http://journals.sagepub.com/doi/10.1177/0309133315598725 (Accessed May 31, 2018).

Harper, R. J., S. J. Sochacki, and J. F. McGrath, 2017: The development of reforestation options for dryland farmland in south-western Australia: a review§. *South. For.*, doi:10.2989/20702620.2016.1255417.

Hassan, M. M., and M. N. I. Nazem, 2016: Examination of land use/land cover changes, urban growth dynamics, and environmental sustainability in Chittagong city, Bangladesh. *Environ. Dev. Sustain.*, **18**, 697–716, doi:10.1007/s10668-015-9672-8. http://link.springer.com/10.1007/s10668-015-9672-8 (Accessed June 2, 2018).

Havemenn, T., and V. Muccione, 2011: *Mechanisms for agricultural climate change mitigation incentives for smallholders. CCAFS Report no. 6*. Copenhagen, Denmark,.

Havlik, P., and Coauthors, 2014: Climate change mitigation through livestock system transitions. *Proc. Natl. Acad. Sci.*, doi:10.1073/pnas.1308044111.

He, B., Y. Cai, W. Ran, X. Zhao, and H. Jiang, 2015: Spatiotemporal heterogeneity of soil salinity after the establishment of vegetation on a coastal saline field. *Catena*, **127**, 129–134, doi:10.1016/j.catena.2014.12.028. http://dx.doi.org/10.1016/j.catena.2014.12.028.

Hearn, M. D., T. Baranowski, J. Baranowski, C. Doyle, M. Smith, L. S. Lin, and K. Resnicow, 1998: Environmental Influences on Dietary Behavior among Children: Availability and Accessibility of Fruits and Vegetables Enable Consumption. *J. Heal. Educ.*, **29**, 26–32, doi:10.1080/10556699.1998.10603294. https://www.tandfonline.com/doi/full/10.1080/10556699.1998.10603294 (Accessed May 30, 2018).

Heck, V., D. Gerten, W. Lucht, and A. Popp, 2018: Biomass-based negative emissions difficult to reconcile with planetary boundaries. *Nat. Clim. Chang.*, doi:10.1038/s41558-017-0064-y. http://www.nature.com/articles/s41558-017-0064-y.

Hejazi, M. I., and Coauthors, 2014: Integrated assessment of global water scarcity over the 21st century under multiple climate change mitigation policies. *Hydrol. Earth Syst. Sci.*, **18**, 2859–2883, doi:10.5194/hess-18-2859-2014.

Hejazi, M. I., and Coauthors, 2015a: 21st century United States emissions mitigation could increase water stress more than the climate change it is mitigating. *Proc. Natl. Acad. Sci.*, doi:10.1073/pnas.1421675112.

——, and Coauthors, 2015b: 21st century United States emissions mitigation could increase water stress more than the climate change it is mitigating. *Proc. Natl. Acad. Sci.*, **112**, 10635–10640, doi:10.1073/pnas.1421675112. http://www.pnas.org/lookup/doi/10.1073/pnas.1421675112.

Hengsdijk, H., and W. J. de Boer, 2017: Post-harvest management and post-harvest losses of cereals in Ethiopia. *Food Secur.*, **9**, 945–958, doi:10.1007/s12571-017-0714-y. http://link.springer.com/10.1007/s12571-017-0714-y (Accessed May 10, 2018).

Herrera-Pantoja, M., and K. M. Hiscock, 2015: Projected impacts of climate change on water availability indicators in a semi-arid region of central Mexico. *Environ. Sci. Policy*, **54**, 81–89, doi:10.1016/j.envsci.2015.06.020. http://dx.doi.org/10.1016/j.envsci.2015.06.020.

Herrero, M., and Coauthors, 2016a: Greenhouse gas mitigation potentials in the livestock sector. *Nat. Clim. Chang.*, **IN PRESS**, doi:10.1038/nclimate2925. https://www.nature.com/nclimate/journal/v6/n5/abs/nclimate2925.html (Accessed December 20, 2017).

Herrero, M., and Coauthors, 2016b: Greenhouse gas mitigation potentials in the livestock sector. *Nat. Clim. Chang.*, **6**, 452–461, doi:10.1038/nclimate2925. http://www.nature.com/articles/nclimate2925 (Accessed May 30, 2018).

Hertel, T. W., 2015: The challenges of sustainably feeding a growing planet. *Food Secur.*, doi:10.1007/s12571-015-0440-2.

Hillbruner, C., and G. Moloney, 2012: When early warning is not enough—Lessons learned from the 2011 Somalia Famine. *Glob. Food Sec.*, **1**, 20–28, doi:10.1016/J.GFS.2012.08.001. https://www.sciencedirect.com/science/article/pii/S2211912412000107 (Accessed June 1, 2018).

Hoekstra, J. M., T. M. Boucher, T. H. Ricketts, and C. Roberts, 2005: Confronting a biome crisis: Global disparities of habitat loss and protection. *Ecol. Lett.*, doi:10.1111/j.1461-0248.2004.00686.x.

Holden, S. T., and K. Otsuka, 2014: The roles of land tenure reforms and land markets in the context of population growth and land use intensification in Africa. *Food Policy*, **48**, 88–97, doi:10.1016/j.foodpol.2014.03.005. http://dx.doi.org/10.1016/j.foodpol.2014.03.005.

——, and H. Ghebru, 2016: Land tenure reforms, tenure security and food security in poor agrarian economies: Causal linkages and research gaps. *Glob. Food Sec.*, **10**, 21–28, doi:10.1016/j.gfs.2016.07.002.

Holland, M. B., K. W. Jones, L. Naughton-Treves, J. L. Freire, M. Morales, and L. Suárez, 2017: Titling land to conserve forests: The case of Cuyabeno Reserve in Ecuador. *Glob. Environ. Chang.*, **44**, 27–38, doi:10.1016/j.gloenvcha.2017.02.004. http://dx.doi.org/10.1016/j.gloenvcha.2017.02.004.

Howard, P. H., 2015: Intellectual Property and Consolidation in the Seed Industry. *Crop Sci.*, **55**, 2489, doi:10.2135/cropsci2014.09.0669. https://dl.sciencesocieties.org/publications/cs/abstracts/55/6/2489 (Accessed June 1, 2018).

Howarth, C., and I. Monasterolo, 2017: Opportunities for knowledge co-production across the energy-food-water nexus: Making interdisciplinary approaches work for better climate decision making. *Environ. Sci. Policy*, doi:10.1016/j.envsci.2017.05.019.

Howden, S. M., J.-F. Soussana, F. N. Tubiello, N. Chhetri, M. Dunlop, and H. Meinke, 2007: Adapting agriculture to climate change. *Proc. Natl. Acad. Sci. U. S. A.*, **104**, 19691–19696, doi:10.1073/pnas.0701890104. http://www.ncbi.nlm.nih.gov/pubmed/18077402 (Accessed June 5, 2018).

Huang, J., T. Wang, W. Wang, Z. Li, and H. Yan, 2014: Climate effects of dust aerosols over east asian arid and semiarid regions. *J. Geophys. Res.*, doi:10.1002/2014JD021796.

——, H. Yu, X. Guan, G. Wang, and R. Guo, 2016: Accelerated dryland expansion under climate change. *Nat. Clim. Chang.*, doi:10.1038/nclimate2837.

Huber-Sannwald, E., M. R. Palacios, J. T. A. Moreno, M. Braasch, R. M. M. Peña, J. G. de A. Verduzco, and K. M. Santos, 2012: Navigating challenges and opportunities of land degradation and sustainable livelihood development in dryland social-ecological systems: a case study from Mexico. *Philos. Trans. R. Soc. Lond. B. Biol. Sci.*, **367**, 3158–3177, doi:10.1098/rstb.2011.0349. http://www.ncbi.nlm.nih.gov/pubmed/23045713 (Accessed June 2, 2018).

Hulvey, K. B., R. J. Hobbs, R. J. Standish, D. B. Lindenmayer, L. Lach, and M. P. Perring, 2013: Benefits of tree mixes in carbon plantings. *Nat. Clim. Chang.*, **3**, 869–874, doi:10.1038/nclimate1862.

Humpenöder, F., and Coauthors, 2014: Investigating afforestation and bioenergy CCS as climate change mitigation strategies. *Environ. Res. Lett.*, **9**, 64029, doi:10.1088/1748-9326/9/6/064029. http://stacks.iop.org/1748-9326/9/i=6/a=064029?key=crossref.5fa44a1462d2acebebaa002315d8e4a6 (Accessed May 30, 2018).

——, and Coauthors, 2018: Large-scale bioenergy production: how to resolve sustainability trade-offs? *Environ. Res. Lett.*, **13**, 24011, doi:10.1088/1748-9326/aa9e3b.

Hunsberger, C., and Coauthors, 2017: Climate change mitigation, land grabbing and conflict: towards a landscape-based and collaborative action research agenda. *Can. J. Dev. Stud.*, **38**, 305–324, doi:10.1080/02255189.2016.1250617.

Huntjens, P., L. Lebel, C. Pahl-Wostl, J. Camkin, R. Schulze, and N. Kranz, 2012: Institutional design propositions for the governance of adaptation to climate change in the water sector. *Glob. Environ. Chang.*, **22**, 67–81, doi:10.1016/j.gloenvcha.2011.09.015.

Huq, S., Y. Sokona, and A. Najam, 2002: *Climate change and sustainable development beyond Kyoto*. UNEP, http://wedocs.unep.org/handle/20.500.11822/8289 (Accessed December 26, 2017).

Hurteau, M. D., J. B. Bradford, P. Z. Fulé, A. H. Taylor, and K. L. Martin, 2014: Climate change, fire management, and ecological services in the southwestern US. *For. Ecol. Manage.*, **327**, 280–289, doi:10.1016/J.FORECO.2013.08.007. https://www.sciencedirect.com/science/article/pii/S0378112713005343 (Accessed May 31, 2018).

Ingram, J., R. Dyball, M. Howden, S. Vermeulen, T. Ganett, B. Redlingshöfer, S. Guilbert, and J. Porter, 2016: Food security, food systems, and environmental change. *Solut. J.* , **7**, 2154–0926.

Insam, H., M. Gómez-Brandón, and J. Ascher, 2015: Manure-based biogas fermentation residues -

Friend or foe of soil fertility? *Soil Biol. Biochem.*, doi:10.1016/j.soilbio.2015.02.006.

Iordan, C., C. Lausselet, and F. Cherubini, 2016: Life-cycle assessment of a biogas power plant with application of different climate metrics and inclusion of near-term climate forcers. *J. Environ. Manage.*, doi:10.1016/j.jenvman.2016.10.030.

IPCC, 2014a: Summary for Policymakers. *Climate Change 2014: Mitigation of Climate Change. Contribution of Working Group III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*, O. Edenhofer et al., Eds., Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

——, 2014b: *Climate Change 2014: Mitigation of Climate Change. Contribution of Working Group III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*. O. Edenhofer et al., Eds. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA,.

IPCC SR1.5, *Global Warming of 1.5oC: A Special Report of Working Groups I, II and III of the Ingergovernmental Panel on Climate Change.*

Iverson, L., A. Prasad, and S. Matthews, 2008: Modeling potential climate change impacts on the trees of the northeastern United States. *Mitig. Adapt. Strateg. Glob. Chang.*, doi:10.1007/s11027-007-9129-y.

Iyer, G., and Coauthors, 2018: Implications of sustainable development considerations for comparability across nationally determined contributions. *Nat. Clim. Chang.*, **8**, doi:10.1038/s41558-017-0039-z.

Izquierdo, A. E., and M. L. Clark, 2012: Spatial analysis of conservation priorities based on ecosystem services in the Atlantic forest region of Misiones, Argentina. *Forests*, doi:10.3390/f3030764.

Jacoby, H. G., and B. Minten, 2007: Is land titling in Sub-Saharan Africa cost-effective? Evidence from Madagascar. *World Bank Econ. Rev.*, **21**, 461–485, doi:10.1093/wber/lhm011.

James, S. J., and C. James, 2010: The food cold-chain and climate change. *Food Res. Int.*, doi:10.1016/j.foodres.2010.02.001.

Jandl, R., J. Bauhus, A. Bolte, A. Schindlbacher, and S. Schüler, 2015: Effect of Climate-Adapted Forest Management on Carbon Pools and Greenhouse Gas Emissions. *Curr. For. Reports*, doi:10.1007/s40725-015-0006-8.

Jauhiainen, J., S. Limin, H. Silvennoinen, and H. Vasander, 2008: Carbon dioxide and methane fluxes in drained tropical peat before and after hydrological restoration. *Ecology*, **89**, 3503–3514, doi:10.1890/07-2038.1. http://doi.wiley.com/10.1890/07-2038.1 (Accessed June 1, 2018).

Jaworski, A., 2016: Encouraging Climate Adaptation through Reform of Federal Crop Insurance Subsidies. *New York Univ. Law Rev.*, **91**. https://heinonline.org/HOL/Page?handle=hein.journals/nylr91&id=1721&div=&collection= (Accessed June 1, 2018).

Jenkins, K., S. Surminski, J. Hall, and F. Crick, 2017: Assessing surface water flood risk and management strategies under future climate change: Insights from an Agent-Based Model. *Sci. Total Environ.*, **595**, 159–168, doi:10.1016/j.scitotenv.2017.03.242. http://dx.doi.org/10.1016/j.scitotenv.2017.03.242.

Jiang, Y., 2015: China's water security: Current status, emerging challenges and future prospects. *Environ. Sci. Policy*, **54**, 106–125, doi:10.1016/j.envsci.2015.06.006. http://dx.doi.org/10.1016/j.envsci.2015.06.006.

Joly, C. A., L. M. Verdade, B. J. Huntley, V. H. Dale, G. Mace, B. Muok, and N. H. Ravindranath, 2015: Biofuel Impacts on Biodiversity and Ecosystem Services. http://bioenfapesp.org/scopebioenergy/images/chapters/bioen-scope_chapter16.pdf (Accessed June 3, 2018).

Jones, A. D., K. V. Calvin, W. D. Collins, and J. Edmonds, 2015: Accounting for radiative forcing from albedo change in future global land-use scenarios. *Clim. Change*, **131**, 691–703, doi:10.1007/s10584-015-1411-5.

Jones, H. P., D. G. Hole, and E. S. Zavaleta, 2012: Harnessing nature to help people adapt to climate change. *Nat. Clim. Chang.*, **2**, 504–509, doi:10.1038/nclimate1463. http://www.nature.com/doifinder/10.1038/nclimate1463 (Accessed June 1, 2018).

Juhola, S., and L. Westerhoff, 2011: Challenges of adaptation to climate change across multiple scales: A case study of network governance in two European countries. *Environ. Sci. Policy*, doi:http://dx.doi.org/10.1016/j.envsci.2010.12.006.

Kang, H., H. Chang, and M. G. Chung, 2012: Rapid land use change impacts on coastal ecosystem services: a South Korean case study. *Water Ecosyst. Serv.*, 119–126, doi:10.1017/CBO9781316178904.015.

Karim, M. R., and A. Thiel, 2017: Role of community based local institution for climate change adaptation in the Teesta riverine area of Bangladesh. *Clim. Risk Manag.*, doi:10.1016/j.crm.2017.06.002.

Karsenty, A., A. Vogel, and F. Castell, 2014: "Carbon rights", REDD+ and payments for environmental services. *Environ. Sci. Policy*, doi:10.1016/j.envsci.2012.08.013.

Kashwan, P., 2015: Forest Policy, Institutions, and REDD+ in India, Tanzania, and Mexico. *Glob. Environ. Polit.*, **15**, 95–117, doi:10.1162/GLEP_a_00313. http://www.mitpressjournals.org/doi/10.1162/GLEP_a_00313 (Accessed June 3, 2018).

Kassam, A., R. Derpsch, and T. Friedrich, 2014: Global achievements in soil and water conservation: The case of Conservation Agriculture. *Int. Soil Water Conserv. Res.*, doi:10.1016/S2095-6339(15)30009-5.

Kauppi, P. E., V. Sandström, and A. Lipponen, 2018: Forest resources of nations in relation to human well-being. *PLoS One*, **13**, e0196248, doi:10.1371/journal.pone.0196248. http://dx.plos.org/10.1371/journal.pone.0196248 (Accessed June 2, 2018).

Keesstra, S., J. Nunes, A. Novara, D. Finger, D. Avelar, Z. Kalantari, and A. Cerdà, 2018: The superior effect of nature based solutions in land management for enhancing ecosystem services. *Sci. Total Environ.*, doi:10.1016/j.scitotenv.2017.08.077.

Kelemen, P. B., and J. Matter, 2008: In situ carbonation of peridotite for CO2 storage. *Proc. Natl. Acad. Sci.*, **105**, 17295–17300, doi:10.1073/PNAS.0805794105. http://www.pnas.org/content/105/45/17295.short (Accessed June 2, 2018).

Kemper, J., 2015a: Biomass and carbon dioxide capture and storage: A review. *Int. J. Greenh. Gas Control*, **40**, 401–430, doi:10.1016/j.ijggc.2015.06.012. http://linkinghub.elsevier.com/retrieve/pii/S1750583615002650.

——, 2015b: Biomass and carbon dioxide capture and storage: A review. *Int. J. Greenh. Gas Control*, doi:10.1016/j.ijggc.2015.06.012.

Kenney-Lazar, M., 2012: Plantation rubber, land grabbing and social-property transformation in southern Laos. *J. Peasant Stud.*, **39**, 1017–1037, doi:10.1080/03066150.2012.674942.

Kerekes, C. B., and C. R. Williamson, 2010: Propertyless in Peru, Even with a Government Land Title. *Am. J. Econ. Sociol.*, **69**, 1011–1033, doi:10.1111/j.1536-7150.2010.00734.x.

Kim, S. H., and Coauthors, 2016: Balancing global water availability and use at basin scale in an integrated assessment model. *Clim. Change*, doi:10.1007/s10584-016-1604-6.

Kindermann, G., and Coauthors, 2008: Global cost estimates of reducing carbon emissions through avoided deforestation. *Proc. Natl. Acad. Sci. U. S. A.*, **105**, 10302–10307, doi:10.1073/pnas.0710616105. http://www.ncbi.nlm.nih.gov/pubmed/18650377 (Accessed June 1, 2018).

Kirkels, F. M. S. A., L. H. Cammeraat, and N. J. Kuhn, 2014: The fate of soil organic carbon upon erosion, transport and deposition in agricultural landscapes — A review of different concepts. *Geomorphology*, **226**, 94–105, doi:10.1016/J.GEOMORPH.2014.07.023. https://www.sciencedirect.com/science/article/pii/S0169555X14003870 (Accessed June 5, 2018).

Kissinger, M., C. Sussmann, and C. Dorward, 2018: Local or global: A biophysical analysis of a regional food system. *Food Syst.*,. https://www.cambridge.org/core/journals/renewable-agriculture-and-food-systems/article/local-or-global-a-biophysical-analysis-of-a-regional-food-system/BEB1826C608FC06CE6CD221D25C1EA6D (Accessed May 10, 2018).

Kloppenberg, J., 2010: Impeding Dispossession, Enabling Repossession: Biological Open Source and the Recovery of Seed Sovereignty. *J. Agrar. Chang.*, **10**, 367–388, doi:10.1111/j.1471-0366.2010.00275.x. http://doi.wiley.com/10.1111/j.1471-0366.2010.00275.x (Accessed June 1, 2018).

Knoke, T., and Coauthors, 2014: Afforestation or intense pasturing improve the ecological and economic value of abandoned tropical farmlands. *Nat. Commun.*, doi:10.1038/ncomms6612.

Kongsager, R., B. Locatelli, and F. Chazarin, 2016: Addressing Climate Change Mitigation and Adaptation Together: A Global Assessment of Agriculture and Forestry Projects. *Environ. Manage.*, **57**, 271–282, doi:10.1007/s00267-015-0605-y. http://link.springer.com/10.1007/s00267-015-0605-y (Accessed June 3, 2018).

Kramer, K., B. Degen, J. Buschbom, T. Hickler, W. Thuiller, M. T. Sykes, and W. de Winter, 2010: Modelling exploration of the future of European beech (Fagus sylvatica L.) under climate change-Range, abundance, genetic diversity and adaptive response. *For. Ecol. Manage.*, doi:10.1016/j.foreco.2009.12.023.

Krause, A., and Coauthors, 2017: Global consequences of afforestation and bioenergy cultivation on ecosystem service indicators. *Biogeosciences*, doi:10.5194/bg-14-4829-2017.

Kraxner, F., and Coauthors, 2013: Global bioenergy scenarios - Future forest development, land-use implications, and trade-offs. *Biomass and Bioenergy*, doi:10.1016/j.biombioe.2013.02.003.

Kreidenweis, U., F. Humpenöder, M. Stevanović, B. L. Bodirsky, E. Kriegler, H. Lotze-Campen, and A. Popp, 2016: Afforestation to mitigate climate change: Impacts on food prices under consideration of albedo effects. *Environ. Res. Lett.*, doi:10.1088/1748-9326/11/8/085001.

Krey, V., and ..., Implications of the Paris agreement for achieving the Sustainable Develoment Goals. *Nat. Clim. Chang.*,.

Kriegler, E., and Coauthors, 2017: Fossil-fueled development (SSP5): An energy and resource intensive scenario for the 21st century. *Glob. Environ. Chang.*, **42**, doi:10.1016/j.gloenvcha.2016.05.015.

Kugelman, M., and S. Levenstein, 2013: *The global farms race: land grabs, agricultural investment, and the scramble for food security*. Island Press, Washington DC,.

Kumar, D., and P. Kalita, 2017: Reducing Postharvest Losses during Storage of Grain Crops to Strengthen Food Security in Developing Countries. *Foods*, doi:10.3390/foods6010008.

Kummu, M., H. de Moel, M. Porkka, S. Siebert, O. Varis, and P. J. Ward, 2012a: Lost food, wasted resources: Global food supply chain losses and their impacts on freshwater, cropland, and fertiliser use. *Sci. Total Environ.*, **438**, 477–489, doi:10.1016/j.scitotenv.2012.08.092. http://dx.doi.org/10.1016/j.scitotenv.2012.08.092.

——, ——, ——, ——, ——, and ——, 2012b: Lost food, wasted resources: Global food supply chain losses and their impacts on freshwater, cropland, and fertiliser use. *Sci. Total Environ.*, doi:10.1016/j.scitotenv.2012.08.092.

Kurian, M., 2017: The water-energy-food nexus: trade-offs, thresholds and transdisciplinary approaches to sustainable development. *Environ. Sci. Policy*,. https://www.sciencedirect.com/science/article/pii/S1462901116305184 (Accessed May 6, 2018).

Kurz, W., C. Smyth, and T. Lemprière, 2016: Climate change mitigation through forest sector activities:

Principles, potential and priorities 1. *Unasylva*, **67**, 61–67.

Kyle, P., C. Müller, K. Calvin, and A. Thomson, 2014: Meeting the radiative forcing targets of the representative concentration pathways in a world with agricultural climate impacts. *Earth's Futur.*, **2**, 83–98, doi:10.1002/2013EF000199. http://doi.wiley.com/10.1002/2013EF000199.

Lal, R., 2014: Soil Carbon Management and Climate Change. *Soil Carbon*.

Lambin, E. F., and P. Meyfroidt, 2011: Global land use change, economic globalization, and the looming land scarcity. *Proc. Natl. Acad. Sci. U. S. A.*, **108**, 3465–3472, doi:10.1073/pnas.1100480108. http://www.ncbi.nlm.nih.gov/pubmed/21321211 (Accessed May 30, 2018).

Lamchin, M., J.-Y. Lee, W.-K. Lee, E. J. Lee, M. Kim, C.-H. Lim, H.-A. Choi, and S.-R. Kim, 2016: Assessment of land cover change and desertification using remote sensing technology in a local region of Mongolia. *Adv. Sp. Res.*, **57**, 64–77, doi:10.1016/J.ASR.2015.10.006. https://www.sciencedirect.com/science/article/pii/S027311771500705X (Accessed June 3, 2018).

Lanckriet, S., B. Derudder, J. Naudts, H. Bauer, J. Deckers, M. Haile, and J. Nyssen, 2015: A Political Ecology Perspective of Land Degradation in the North Ethiopian Highlands. *L. Degrad. Dev.*, **26**, 521–530, doi:10.1002/ldr.2278.

Landis, D. A., 2017: Designing agricultural landscapes for biodiversity-based ecosystem services. *Basic Appl. Ecol.*, **18**, 1–12, doi:10.1016/J.BAAE.2016.07.005. https://www.sciencedirect.com/science/article/pii/S1439179116300950 (Accessed June 3, 2018).

Lapola, D. M., and Coauthors, 2014: Pervasive transition of the Brazilian land-use system. *Nat. Clim. Chang.*, **4**, 27–35, doi:10.1038/nclimate2056. http://www.nature.com/articles/nclimate2056 (Accessed June 2, 2018).

Larsen, F. W., M. C. Londo??o-Murcia, and W. R. Turner, 2011: Global priorities for conservation of threatened species, carbon storage, and freshwater services: Scope for synergy? *Conserv. Lett.*, doi:10.1111/j.1755-263X.2011.00183.x.

Larson, A., and J. Pulhin, 2012: Enhancing forest tenure reforms through more responsive regulations. *Conserv. Soc.*, **10**, 103, doi:10.4103/0972-4923.97482. http://www.conservationandsociety.org/text.asp?2012/10/2/103/97482.

Larson, A. M., and Coauthors, 2013: Land tenure and REDD+: The good, the bad and the ugly. *Glob. Environ. Chang.*, **3**, 678–689, doi:10.1016/j.gloenvcha.2013.02.014. http://dx.doi.org/10.1016/j.gloenvcha.2013.02.014.

Lautenbach, S., A. Jungandreas, J. Blanke, V. Lehsten, S. Mühlner, I. Kühn, and M. Volk, 2017: Trade-offs between plant species richness and carbon storage in the context of afforestation – Examples from afforestation scenarios in the Mulde Basin, Germany. *Ecol. Indic.*, doi:10.1016/j.ecolind.2016.09.035.

Lawry, S., C. Samii, R. Hall, A. Leopold, D. Hornby, and F. Mtero, 2017: The impact of land property rights interventions on investment and agricultural productivity in developing countries: a systematic review. *J. Dev. Eff.*, **9**, 61–81, doi:10.1080/19439342.2016.1160947.

Lazarus, E. D., 2014: Land grabbing as a driver of environmental change. *Area*, **46**, 74–82, doi:10.1111/area.12072.

Lee-Smith, D., 2010: Cities feeding people: An update on urban agriculture in equatorial Africa. *Environ. Urban.*, doi:10.1177/0956247810377383.

Lee, Y., J. Ahern, and C. Yeh, 2015: Ecosystem services in peri-urban landscapes: The effects of agricultural landscape change on ecosystem services in Taiwan's western coastal plain. *Landsc. Urban Plan.*,. https://www.sciencedirect.com/science/article/pii/S0169204615000596 (Accessed May 1, 2018).

Leifeld, J., and L. Menichetti, 2018: The underappreciated potential of peatlands in global climate change mitigation strategies. *Nat. Commun.*, **9**, 1071, doi:10.1038/s41467-018-03406-6. http://www.nature.com/articles/s41467-018-03406-6 (Accessed June 2, 2018).

Leon, J. X., J. Hardcastle, R. James, S. Albert, J. Kereseka, and C. D. Woodroffe, 2015: Supporting Local and Traditional Knowledge with Science for Adaptation to Climate Change: Lessons Learned from Participatory Three-Dimensional Modeling in BoeBoe, Solomon Islands. *Coast. Manag.*, doi:10.1080/08920753.2015.1046808.

Levidow, L., D. Zaccaria, R. Maia, E. Vivas, M. Todorovic, and A. Scardigno, 2014: Improving water-efficient irrigation: Prospects and difficulties of innovative practices. *Agric. Water Manag.*, doi:10.1016/j.agwat.2014.07.012.

Lewis, K., and C. Witham, 2012: Agricultural commodities and climate change. *Clim. Policy*, **12**, S53–S61, doi:10.1080/14693062.2012.728790. http://www.tandfonline.com/doi/abs/10.1080/14693062.2012.728790 (Accessed June 1, 2018).

Lewis, S. L., D. P. Edwards, and D. Galbraith, 2015: Increasing human dominance of tropical forests. *Science (80-.).*, doi:10.1126/science.aaa9932.

Li, S., and X. Li, 2017: Global understanding of farmland abandonment: A review and prospects. *J. Geogr. Sci.*, **27**, 1123–1150, doi:10.1007/s11442-017-1426-0. http://link.springer.com/10.1007/s11442-017-1426-0 (Accessed June 2, 2018).

Li, Y., M. Zhao, S. Motesharrei, Q. Mu, E. Kalnay, and S. Li, 2015: Local cooling and warming effects of forests based on satellite observations. *Nat. Commun.*, doi:10.1038/ncomms7603.

——, and Coauthors, 2016: Potential and actual impacts of deforestation and afforestation on land surface temperature. *J. Geophys. Res.*, doi:10.1002/2016JD024969.

Liere, H., T. N. Kim, B. P. Werling, T. D. Meehan, D. A. Landis, and C. Gratton, 2015: Trophic cascades in agricultural landscapes: indirect effects of landscape composition on crop yield. *Ecol. Appl.*, **25**, 652–661, doi:10.1890/14-0570.1.

Limpens, J., and Coauthors, 2008: Peatlands and the carbon cycle: from local processes to global implications – a synthesis. *Biogeosciences*, **5**, 1475–1491. www.biogeosciences.net/5/1475/2008/ (Accessed June 7, 2018).

Lin, Y., L. S. Wijedasa, and R. A. Chisholm, 2017: Singapore's willingness to pay for mitigation of transboundary forest-fire haze from Indonesia. *Environ. Res. Lett.*, doi:10.1088/1748-9326/aa5cf6.

Liniger, H., R. Mekdaschi Studer, P. Moll, and U. Zander, 2017: *Making sense of research for sustainable land management*. Leipzig,.

Linnerooth-Bayer, J., and S. Hochrainer-Stigler, 2014: Financial instruments for disaster risk management and climate change adaptation. *Clim. Change*, doi:10.1007/s10584-013-1035-6.

Lipper, L., and Coauthors, 2014: Climate-smart agriculture for food security. *Nat. Clim. Chang.*, doi:10.1038/nclimate2437.

Liu, J., J. Li, Z. Gao, M. Yang, K. Qin, and X. Yang, 2016: Ecosystem services insights into water resources management in China: A case of Xi'an City. *Int. J. Environ. Res. Public Health*, **13**, doi:10.3390/ijerph13121169.

Liu, Y., Y. Zhou, and W. Wu, 2015: Assessing the impact of population, income and technology on energy consumption and industrial pollutant emissions in China. *Appl. Energy*, **155**, 904–917, doi:10.1016/J.APENERGY.2015.06.051. https://www.sciencedirect.com/science/article/pii/S0306261915008090 (Accessed May 31, 2018).

Lloyd, G. J., L. Korsgaard, R. Tharme, E. Boelee, F. Clement, J. Barron, and N. Eriyagama, 2013: Water management for ecosystem health and food production. *Manag. water agroecosystems food Secur.*, 197.

Lobell, D. B., 2014: Climate change adaptation in crop production: Beware of illusions. *Glob. Food Sec.*, doi:10.1016/j.gfs.2014.05.002.

Locatelli, B., 2011: Synergies between adaptation and mitigation in a nutshell. *Cobam*, **8**.

——, V. Evans, A. Wardell, A. Andrade, and R. Vignola, 2011a: Forests and climate change in latin America: Linking adaptation and mitigation. *Forests*, **2**, 431–450, doi:10.3390/f2010431.

Locatelli, B., V. Evans, A. Wardell, A. Andrade, and R. Vignola, 2011b: Forests and climate change in Latin America: linking adaptation and mitigation. *Forests*, doi:10.3390/f2010431.

Locatelli, B., and Coauthors, 2015a: Tropical reforestation and climate change: Beyond carbon. *Restor. Ecol.*, doi:10.1111/rec.12209.

——, C. Pavageau, E. Pramova, and M. Di Gregorio, 2015b: Integrating climate change mitigation and adaptation in agriculture and forestry: Opportunities and trade-offs. *Wiley Interdiscip. Rev. Clim. Chang.*, doi:10.1002/wcc.357.

——, ——, ——, and ——, 2015c: Integrating climate change mitigation and adaptation in agriculture and forestry: Opportunities and trade-offs. *Wiley Interdiscip. Rev. Clim. Chang.*, **6**, 585–598, doi:10.1002/wcc.357.

Lotze-Campen, H., A. Popp, T. Beringer, C. Müller, A. Bondeau, S. Rost, and W. Lucht, 2010: Scenarios of global bioenergy production: The trade-offs between agricultural expansion, intensification and trade. *Ecol. Modell.*, doi:10.1016/j.ecolmodel.2009.10.002.

Louwaars, N. P., 2002: Seed Policy, Legislation and Law. *J. New Seeds*, **4**, 1–14, doi:10.1300/J153v04n01_01. http://www.tandfonline.com/doi/abs/10.1300/J153v04n01_01 (Accessed June 1, 2018).

Lu, C., T. Zhao, X. Shi, and S. Cao, 2016: Ecological restoration by afforestation may increase groundwater depth and create potentially large ecological and water opportunity costs in arid and semiarid China. *J. Clean. Prod.*, doi:https://doi.org/10.1016/j.jclepro.2016.03.046.

Luby, C. H., J. Kloppenburg, T. E. Michaels, and I. L. Goldman, 2015: Enhancing Freedom to Operate for Plant Breeders and Farmers through Open Source Plant Breeding. *Crop Sci.*, **55**, 2481, doi:10.2135/cropsci2014.10.0708. https://dl.sciencesocieties.org/publications/cs/abstracts/55/6/2481 (Accessed June 1, 2018).

Lwasa, S., F. Mugagga, B. Wahab, D. Simon, and J. C. Climate, 2014: Urban and peri-urban agriculture and forestry: transcending poverty alleviation to climate change mitigation and adaptation. *Urban Clim.*,. https://www.sciencedirect.com/science/article/pii/S2212095513000552 (Accessed May 1, 2018).

Machado, R., and R. Serralheiro, 2017: Soil Salinity: Effect on Vegetable Crop Growth. Management Practices to Prevent and Mitigate Soil Salinization. *Horticulturae*, **3**, 30, doi:10.3390/horticulturae3020030. http://www.mdpi.com/2311-7524/3/2/30.

Maes, J., and S. Jacobs, 2017: Nature-Based Solutions for Europe's Sustainable Development. *Conserv. Lett.*, **10**, 121–124, doi:10.1111/conl.12216. http://doi.wiley.com/10.1111/conl.12216 (Accessed June 2, 2018).

Maes, J., and Coauthors, 2017: Landslide risk reduction measures: A review of practices and challenges for the tropics. *Prog. Phys. Geogr.*, **41**, 191–221, doi:10.1177/0309133316689344. http://journals.sagepub.com/doi/10.1177/0309133316689344 (Accessed May 31, 2018).

Maginnis, S., and W. Jackson, 2007: What is FLR and how does it differ from current approaches? *The forest landscape restoration handbook*, Earthscan, London, UK.

Mahmood, R., and Coauthors, 2014: Land cover changes and their biogeophysical effects on climate. *Int. J. Climatol.*, doi:10.1002/joc.3736.

Mahmud, T., and M. Prowse, 2012: Corruption in cyclone preparedness and relief efforts in coastal Bangladesh: Lessons for climate adaptation? *Glob. Environ. Chang.*, **22**, 933–943, doi:10.1016/J.GLOENVCHA.2012.07.003. https://www.sciencedirect.com/science/article/pii/S0959378012000799 (Accessed June 1, 2018).

Mal, S., R. B. Singh, C. Huggel, and A. Grover, 2018: Introducing Linkages Between Climate Change, Extreme Events, and Disaster Risk Reduction. Springer, Cham, 1–14 http://link.springer.com/10.1007/978-3-319-56469-2_1 (Accessed May 31, 2018).

Marchesini, V. A., R. Giménez, M. D. Nosetto, and E. G. Jobbágy, 2017: Ecohydrological transformation in the Dry Chaco and the risk of dryland salinity: Following Australia's footsteps? *Ecohydrology*, **10**, e1822, doi:10.1002/eco.1822. http://doi.wiley.com/10.1002/eco.1822 (Accessed June 3, 2018).

Margono, B. A., P. V. Potapov, S. Turubanova, F. Stolle, and M. C. Hansen, 2014: Primary forest cover loss in Indonesia over 2000–2012. *Nat. Clim. Chang.*, **4**, 730–735, doi:10.1038/nclimate2277. http://www.nature.com/doifinder/10.1038/nclimate2277 (Accessed June 2, 2018).

Markandya, A., J. Sampedro, S. J. Smith, R. Van Dingenen, C. Pizarro-Irizar, I. Arto, and M. González-Eguino, 2018: Health co-benefits from air pollution and mitigation costs of the Paris Agreement: a modelling study. *Lancet Planet. Heal.*, **2**, e126–e133, doi:10.1016/S2542-5196(18)30029-9. https://www.sciencedirect.com/science/article/pii/S2542519618300299 (Accessed June 1, 2018).

Marselis, S. M., K. Feng, Y. Liu, J. D. Teodoro, and K. Hubacek, 2017: Agricultural land displacement and undernourishment. *J. Clean. Prod.*, **161**, 619–628, doi:10.1016/j.jclepro.2017.05.125.

Martha, G. B., E. Alves, and E. Contini, 2012: Land-saving approaches and beef production growth in Brazil. *Agric. Syst.*, **110**, 173–177, doi:10.1016/J.AGSY.2012.03.001. https://www.sciencedirect.com/science/article/pii/S0308521X12000340 (Accessed June 7, 2018).

Martin, S., A. Rieple, J. Chang, B. Boniface, and A. Ahmed, 2015: Small farmers and sustainability: Institutional barriers to investment and innovation in the Malaysian palm oil industry in Sabah. *J. Rural Stud.*, **40**, 46–58, doi:10.1016/J.JRURSTUD.2015.06.002. https://www.sciencedirect.com/science/article/pii/S0743016715000662 (Accessed May 30, 2018).

Martin, S. M., and K. Lorenzen, 2016: Livelihood Diversification in Rural Laos. *World Dev.*, **83**, 231–243, doi:10.1016/J.WORLDDEV.2016.01.018. https://www.sciencedirect.com/science/article/pii/S0305750X16000164 (Accessed May 31, 2018).

Massawe, F., S. Mayes, and A. Cheng, 2016: Crop Diversity: An Unexploited Treasure Trove for Food Security. *Trends Plant Sci.*, **21**, 365–368, doi:10.1016/J.TPLANTS.2016.02.006. https://www.sciencedirect.com/science/article/pii/S1360138516000601 (Accessed May 31, 2018).

Maxwell, D., and K. Wiebe, 1998: Land Tenure and Food Security : A Review of Concepts , Evidence , and Methods. *Access*, 37. http://scholar.google.com/scholar?hl=en&btnG=Search&q=intitle:Land+Tenure+and+Food+Security+:+A+Review+of+Concepts+,+Evidence+,+and+Methods#0.

——, and ——, 1999: Land tenure and food security: Exploring dynamic linkages. *Dev. Change*, **30**, 825–849, doi:10.1111/1467-7660.00139. http://doi.wiley.com/10.1111/1467-7660.00139.

Mbow, C., P. Smith, D. Skole, L. Duguma, and M. Bustamante, 2014: Achieving mitigation and adaptation to climate change through sustainable agroforestry practices in africa. *Curr. Opin. Environ. Sustain.*, doi:10.1016/j.cosust.2013.09.002.

McCrum, G., K. Blackstock, K. Matthews, M. Rivington, D. Miller, and K. Buchan, 2009: Adapting to climate change in land management: the role of deliberative workshops in enhancing social learning. *Environ. Policy Gov.*, **19**, 413–426, doi:10.1002/eet.525. http://doi.wiley.com/10.1002/eet.525 (Accessed December 26, 2017).

McElwee, P., V. H. T. Nguyen, D. V. Nguyen, N. H. Tran, H. V. T. Le, T. P. Nghiem, and H. D. T. Vu, 2017: Using REDD+ policy to facilitate climate adaptation at the local level: Synergies and challenges in Vietnam. *Forests*, **8**, 1–25, doi:10.3390/f8010011.

McGray, H., A. Hammill, R. Bradley, E. L. F. Schipper, and J.-E. Parry, 2007: *Weathering the Storm. Options for Framing Adaptation and Development*.

McIsaac, G. F., M. B. David, and C. A. M. U. of Illinois, 2010: Miscanthus and Switchgrass Production in Central Illinois: Impacts on Hydrology and Inorganic Nitrogen Leaching. *J. Environ. Qual.*, **39**, 1790, doi:10.2134/jeq2009.0497. https://www.agronomy.org/publications/jeq/abstracts/39/5/1790 (Accessed December 24, 2017).

McKinsey and Company, 2009: *Pathways to a low-carbon economy: Version 2 of the global greenhouse gas abatement cost curve | McKinsey & Company*. https://www.mckinsey.com/business-functions/sustainability-and-resource-productivity/our-insights/pathways-to-a-low-carbon-economy (Accessed May 30, 2018).

McMichael, P., 2015: Food security governance. Empowering communities, regulating corporations. *J. Peasant Stud.*, **42**, 1051–1052, doi:10.1080/03066150.2015.1072974. http://www.tandfonline.com/doi/full/10.1080/03066150.2015.1072974 (Accessed May 1, 2018).

Medugu, N. I., M. R. Majid, F. Johar, and I. D. Choji, 2010: The role of afforestation programme in combating desertification in Nigeria. *Int. J. Clim. Chang. Strateg. Manag.*, doi:10.1108/17568691011020247.

Meehan, T. D., A. H. Hurlbert, and C. Gratton, 2010: Bird communities in future bioenergy landscapes of the Upper Midwest. *Proc. Natl. Acad. Sci.*, doi:10.1073/pnas.1008475107.

Meehan, T. D., B. P. Werling, D. A. Landis, and C. Gratton, 2012: Pest-suppression potential of midwestern landscapes under contrasting bioenergy scenarios. *PLoS One*, doi:10.1371/journal.pone.0041728.

Mello, F. F. C., and Coauthors, 2014: Payback time for soil carbon and sugar-cane ethanol. *Nat. Clim. Chang.*, doi:10.1038/nclimate2239.

Merante, P., and Coauthors, 2017: Adopting soil organic carbon management practices in soils of varying quality: Implications and perspectives in Europe. *Soil Tillage Res.*, **165**, 95–106, doi:10.1016/J.STILL.2016.08.001. https://www.sciencedirect.com/science/article/pii/S0167198716301507 (Accessed June 5, 2018).

Mercer, J., 2010: Policy Arena Disaster Risk Reduction or Climate Change Adaptation: Are We Reinventing the Wheel? *J. Int. Dev.*, doi:10.1002/jid.

Merriott, D., 2016: Factors associated with the farmer suicide crisis in India. *J. Epidemiol. Glob. Health*, **6**, 217–227, doi:10.1016/J.JEGH.2016.03.003. https://www.sciencedirect.com/science/article/pii/S2210600615300277 (Accessed June 2, 2018).

Meze-Hausken, E., A. Patt, and S. Fritz, 2009: Reducing climate risk for micro-insurance providers in Africa: A case study of Ethiopia. *Glob. Environ. Chang.*, doi:10.1016/j.gloenvcha.2008.09.001.

Michelini, L., L. Principato, and G. Iasevoli, 2018: Understanding Food Sharing Models to Tackle Sustainability Challenges. *Ecol. Econ.*, **145**, 205–217, doi:10.1016/J.ECOLECON.2017.09.009. https://www.sciencedirect.com/science/article/pii/S0921800916308370 (Accessed June 1, 2018).

Michelsen, O., 2008: Assessment of land use impact on biodiversity. *Int. J. Life Cycle Assess.*, doi:10.1065/lca2007.04.316.

Middleton, N., and U. Kang, 2017: Sand and Dust Storms: Impact Mitigation. *Sustainability*, **9**, 1053, doi:10.3390/su9061053. http://www.mdpi.com/2071-1050/9/6/1053 (Accessed June 3, 2018).

Miettinen, J., A. Hooijer, C. Shi, D. Tollenaar, R. Vernimmen, S. C. Liew, C. Malins, and S. E. Page, 2012: Extent of industrial plantations on Southeast Asian peatlands in 2010 with analysis of historical expansion and future projections. *GCB Bioenergy*, **4**, 908–918, doi:10.1111/j.1757-1707.2012.01172.x. http://doi.wiley.com/10.1111/j.1757-1707.2012.01172.x (Accessed June 7,

2018).

——, C. Shi, and S. C. Liew, 2016: Land cover distribution in the peatlands of Peninsular Malaysia, Sumatra and Borneo in 2015 with changes since 1990. *Glob. Ecol. Conserv.*, doi:10.1016/j.gecco.2016.02.004.

Milan, B. F., 2017: Clean water and sanitation for all: interactions with other sustainable development goals. *Sustain. Water Resour. Manag.*, **3**, 479–489, doi:10.1007/s40899-017-0117-4. http://link.springer.com/10.1007/s40899-017-0117-4.

Minot, N., 2014: Food price volatility in sub-Saharan Africa: Has it really increased? *Food Policy*, **45**, 45–56, doi:10.1016/J.FOODPOL.2013.12.008. https://www.sciencedirect.com/science/article/pii/S0306919213001863 (Accessed June 1, 2018).

Mogolí On, J. M., L. Lassaletta, A. H. W. Beusen, J. M. Van Grinsven, H. Westhoek, and A. F. Bouwman, 2018: Assessing future reactive nitrogen inputs into global croplands based on the shared socioeconomic pathways. *Environ. Res. Lett*, **13**, doi:10.1088/1748-9326/aab212.

Mohammadi, A., S. Rafiee, A. Jafari, A. Keyhani, S. H. Mousavi-Avval, and S. Nonhebel, 2014: Energy use efficiency and greenhouse gas emissions of farming systems in north Iran. *Renew. Sustain. Energy Rev.*, **30**, 724–733, doi:10.1016/J.RSER.2013.11.012.

Möller, K., W. Stinner, A. Deuker, and G. Leithold, 2008: Effects of different manuring systems with and without biogas digestion on nitrogen cycle and crop yield in mixed organic dairy farming systems. *Nutr. Cycl. Agroecosystems*, doi:10.1007/s10705-008-9196-9.

Molotoks, A., M. Kuhnert, T. Dawson, and P. Smith, 2017: Global Hotspots of Conflict Risk between Food Security and Biodiversity Conservation. *Land*, doi:10.3390/land6040067.

de Moraes Sá, J. C., R. Lal, C. C. Cerri, K. Lorenz, M. Hungria, and P. C. de Faccio Carvalho, 2017: Low-carbon agriculture in South America to mitigate global climate change and advance food security. *Environ. Int.*, **98**, 102–112, doi:10.1016/J.ENVINT.2016.10.020. https://www.sciencedirect.com/science/article/pii/S0160412016306341 (Accessed May 31, 2018).

Morduch, J., and M. Sharma, 2002: Strengthening Public Safety Nets from the Bottom Up. *Dev. Policy Rev.*, doi:10.1111/1467-7679.00190.

Moss, B., 2008: Water pollution by agriculture. *Philos. Trans. R. Soc. Lond. B. Biol. Sci.*, **363**, 659–666, doi:10.1098/rstb.2007.2176. http://www.ncbi.nlm.nih.gov/pubmed/17666391 (Accessed December 24, 2017).

Moss, R. H., A. L. Brenkert, and E. L. Malone, 2001: Vulnerability to Climate Change. A Quantitative Approach. *Prep. US Dep. Energy Available online httpwww Glob. umd educgibinDetails pl*,.

Mouratiadou, I., and Coauthors, 2016: The impact of climate change mitigation on water demand for energy and food: An integrated analysis based on the Shared Socioeconomic Pathways. *Environ. Sci. Policy*, **64**, 48–58, doi:10.1016/j.envsci.2016.06.007.

Mubaya, C. P., and P. Mafongoya, 2017: The role of institutions in managing local level climate change adaptation in semi-arid Zimbabwe. *Clim. Risk Manag.*, doi:10.1016/j.crm.2017.03.003.

Muller, A., and Coauthors, 2017: Strategies for feeding the world more sustainably with organic agriculture. *Nat. Commun.*, doi:10.1038/s41467-017-01410-w.

Munang, R., and Coauthors, 2013a: Climate change and Ecosystem-based Adaptation: a new pragmatic approach to buffering climate change impacts This review comes from a themed issue on Terrestrial systems. *Curr. Opin. Environ. Sustain.*, **5**, 67–71, doi:10.1016/j.cosust.2012.12.001. http://dx.doi.org/10.1016/j.cosust.2012.12.001.

——, I. Thiaw, K. Alverson, J. Liu, and Z. Han, 2013b: The role of ecosystem services in climate change adaptation and disaster risk reduction. *Curr. Opin. Environ. Sustain.*, doi:10.1016/j.cosust.2013.02.002.

——, J. Andrews, K. Alverson, and D. Mebratu, 2014: Harnessing Ecosystem-based Adaptation To Address the Social Dimensions of Climate Change. *Environ. Sci. Policy Sustain. Dev.*, **56**, 18–24, doi:10.1080/00139157.2014.861676. http://www.tandfonline.com/doi/abs/10.1080/00139157.2014.861676 (Accessed May 30, 2018).

Mundler, P., and L. Rumpus, 2012: The energy efficiency of local food systems: A comparison between different modes of distribution. *Food Policy*, **37**, 609–615, doi:10.1016/J.FOODPOL.2012.07.006. https://www.sciencedirect.com/science/article/pii/S0306919212000802 (Accessed June 1, 2018).

Muratori, M., K. Calvin, M. Wise, P. Kyle, and J. Edmonds, 2016: Global economic consequences of deploying bioenergy with carbon capture and storage (BECCS). *Environ. Res. Lett.*, **11**, 95004, doi:10.1088/1748-9326/11/9/095004. http://stacks.iop.org/1748-9326/11/i=9/a=095004?key=crossref.962cb328d4e5c48882928ffcb3cdc7fe.

Murthy, P. S., and M. Madhava Naidu, 2012: Sustainable management of coffee industry by-products and value addition - A review. *Resour. Conserv. Recycl.*, doi:10.1016/j.resconrec.2012.06.005.

Mykleby, P. M., P. K. Snyder, and T. E. Twine, 2017: Quantifying the trade-off between carbon sequestration and albedo in midlatitude and high-latitude North American forests. *Geophys. Res. Lett.*, doi:10.1002/2016GL071459.

Nagoda, S., and A. J. Nightingale, 2017: Participation and Power in Climate Change Adaptation Policies: Vulnerability in Food Security Programs in Nepal. *World Dev.*, doi:10.1016/j.worlddev.2017.07.022.

Naylor, R., R. Goldburg, J. Primavera, and N. Kautsky, 2000: Effect of aquaculture on world fish supplies. *Nature*,. https://www.nature.com/articles/35016500 (Accessed May 30, 2018).

Neary, D. G., G. G. Ice, and C. R. Jackson, 2009: Linkages between forest soils and water quality and quantity. *For. Ecol. Manage.*, doi:10.1016/j.foreco.2009.05.027.

Nejad, A. N., 2013: *Soil and water conservation for desertification control in Iran*. 377-400 pp.

Nelson, G. C., and Coauthors, 2014: Climate change effects on agriculture: Economic responses to biophysical shocks. *Proc. Natl. Acad. Sci.*, **111**, 3274–3279, doi:10.1073/pnas.1222465110. http://www.pnas.org/lookup/doi/10.1073/pnas.1222465110.

Nelson, V., K. Meadows, T. Cannon, J. Morton, and A. Martin, 2002: Uncertain predictions, invisible impacts, and the need to mainstream gender in climate change adaptations. *Gend. Dev.*, doi:10.1080/13552070215911.

Nesshöver, C., and Coauthors, 2017: The science, policy and practice of nature-based solutions: An interdisciplinary perspective. *Sci. Total Environ.*, **579**, 1215–1227, doi:10.1016/J.SCITOTENV.2016.11.106. https://www.sciencedirect.com/science/article/pii/S0048969716325578 (Accessed June 2, 2018).

Newbold, J. D., S. Herbert, B. W. Sweeney, P. Kiry, and S. J. Alberts, 2010: Water Quality Functions of a 15-Year-Old Riparian Forest Buffer System [1]. *JAWRA J. Am. Water Resour. Assoc.*, **46**, 299–310, doi:10.1111/j.1752-1688.2010.00421.x. http://doi.wiley.com/10.1111/j.1752-1688.2010.00421.x (Accessed June 1, 2018).

Newbold, T., and Coauthors, 2015: Global effects of land use on local terrestrial biodiversity. *Nature*, **520**, 45–50, doi:10.1038/nature14324. http://www.nature.com/articles/nature14324 (Accessed June 3, 2018).

Ngigi, M. W., U. Mueller, and R. Birner, 2017: Gender Differences in Climate Change Adaptation Strategies and Participation in Group-based Approaches: An Intra-household Analysis From Rural Kenya. *Ecol. Econ.*, doi:10.1016/j.ecolecon.2017.03.019.

Nguru, P. M., and D. K. Rono, 2013: Combating desertification in Kenya. *Combat. Desertif. Asia, Africa Middle East Proven Pract.*, 139–151, doi:10.1007/978-94-007-6652-5_7.

Nicole Bernex, 2016: Linking ecosystem services and water security – SDGs offer a new opportunity for integration. www.gwp.org%0Awww.gwp.org.

Nijsen, M., E. Smeets, E. Stehfest, and D. P. van Vuuren, 2012: An evaluation of the global potential of bioenergy production on degraded lands. *GCB Bioenergy*, doi:10.1111/j.1757-1707.2011.01121.x.

Nizeyimana, E. L., G. W. Petersen, M. L. Imhoff, H. R. Sinclair, S. W. Waltman, D. S. Reed-Margetan, E. R. Levine, and J. M. Russo, 2001: Assessing the Impact of Land Conversion to Urban Use on Soils with Different Productivity Levels in the USA. *Soil Sci. Soc. Am. J.*, **65**, 391, doi:10.2136/sssaj2001.652391x. https://www.soils.org/publications/sssaj/abstracts/65/2/391 (Accessed May 31, 2018).

Nkoa, R., 2014: Agricultural benefits and environmental risks of soil fertilization with anaerobic digestates: A review. *Agron. Sustain. Dev.*, doi:10.1007/s13593-013-0196-z.

Noble, I., S. Huq, Y. Anokhin, J. Carmin, D. Goudou, F. Lansigan, B. Osman-Elasha, and A. Villamizar, 2014: Adaptation needs and options. *Climate Change 2014: Impacts, Adaptation, and Vulnerability. Part A: Global and Sectoral Aspects. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*, C. Field et al., Eds., Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, 833–868.

North, M. P., S. L. Stephens, B. M. Collins, J. K. Agee, G. Aplet, J. F. Franklin, and P. Z. Fulé, 2015: Reform forest fire management. *Science*, **349**, 1280–1281, doi:10.1126/science.aab2356. http://www.ncbi.nlm.nih.gov/pubmed/26383934 (Accessed May 31, 2018).

Norton, B. A., A. M. Coutts, S. J. Livesley, R. J. Harris, A. M. Hunter, and N. S. G. Williams, 2015: Planning for cooler cities: A framework to prioritise green infrastructure to mitigate high temperatures in urban landscapes. *Landsc. Urban Plan.*, **134**, 127–138, doi:10.1016/J.LANDURBPLAN.2014.10.018. https://www.sciencedirect.com/science/article/pii/S0169204614002503 (Accessed June 2, 2018).

Obersteiner, M., and Coauthors, 2016: Assessing the land resource-food price nexus of the Sustainable Development Goals. *Sci. Adv.*, doi:10.1126/sciadv.1501499.

Ojea, E., 2015: Challenges for mainstreaming Ecosystem-based Adaptation into the international climate agenda. *Curr. Opin. Environ. Sustain.*, doi:10.1016/j.cosust.2015.03.006.

de Oliveira Silva, R., L. G. Barioni, J. A. J. Hall, A. C. Moretti, R. Fonseca Veloso, P. Alexander, M. Crespolini, and D. Moran, 2017: Sustainable intensification of Brazilian livestock production through optimized pasture restoration. *Agric. Syst.*, **153**, 201–211, doi:10.1016/J.AGSY.2017.02.001. https://www.sciencedirect.com/science/article/pii/S0308521X16303845 (Accessed June 2, 2018).

De Oliveira Silva, R., L. G. Barioni, G. Queiroz Pellegrino, and D. Moran, 2018: The role of agricultural intensification in Brazil's Nationally Determined Contribution on emissions mitigation. *Agric. Syst.*, **161**, 102–112, doi:10.1016/J.AGSY.2018.01.003. https://www.sciencedirect.com/science/article/pii/S0308521X17307655 (Accessed June 2, 2018).

Oliver, C. D., N. T. Nassar, B. R. Lippke, and J. B. McCarter, 2014: Carbon, Fossil Fuel, and Biodiversity Mitigation With Wood and Forests. *J. Sustain. For.*, **33**, 248–275, doi:10.1080/10549811.2013.839386.

Orlove, B., C. Roncoli, M. Kabugo, and A. Majugu, 2010: Indigenous climate knowledge in southern Uganda: The multiple components of a dynamic regional system. *Clim. Change*, doi:10.1007/s10584-009-9586-2.

Osinubi, S. T., K. Hand, D. C. C. Van Oijen, B. A. Walther, and P. Barnard, 2016: Linking science and policy to address conservation concerns about African land use, land conversion and land grabs in the era of globalization. *Afr. J. Ecol.*, **54**, 265–267, doi:10.1111/aje.12355.

Ostrom, E., 1990: An Institutional Approach to the Study of Self-organization and Self-governance in

CPR Situations. *Gov. Commons Evol. institutions Collect. action*, doi:10.1017/CBO9781316423936.003.

Overmars, K. P., E. Stehfest, A. Tabeau, H. van Meijl, A. M. Beltrán, and T. Kram, 2014: Estimating the opportunity costs of reducing carbon dioxide emissions via avoided deforestation, using integrated assessment modelling. *Land use policy*, **41**, 45–60, doi:10.1016/J.LANDUSEPOL.2014.04.015. https://www.sciencedirect.com/science/article/pii/S0264837714000799 (Accessed June 1, 2018).

Özdemir, S., M. Elliott, J. Brown, P. K. Nam, V. Thi Hien, and M. D. Sobsey, 2011: Rainwater harvesting practices and attitudes in the Mekong Delta of Vietnam. *J. Water, Sanit. Hyg. Dev.*, doi:10.2166/washdev.2011.024.

Pacheco, P., D. Barry, P. Cronkleton, and A. Larson, 2012: The Recognition of Forest Rights in Latin America: Progress and Shortcomings of Forest Tenure Reforms. *Soc. Nat. Resour.*, **25**, 556–571, doi:10.1080/08941920.2011.574314.

Pagdee, A., Y. S. Kim, and P. J. Daugherty, 2006: What makes community forest management successful: A meta-study from community forests throughout the world. *Soc. Nat. Resour.*, **19**, 33–52, doi:10.1080/08941920500323260.

Page, S. E., and A. Hooijer, 2016: In the line of fire: the peatlands of Southeast Asia. *Philos. Trans. R. Soc. Lond. B. Biol. Sci.*, **371**, 20150176, doi:10.1098/rstb.2015.0176. http://www.ncbi.nlm.nih.gov/pubmed/27216508 (Accessed June 3, 2018).

Page, S. E., J. O. Rieley, and C. J. Banks, 2011: Global and regional importance of the tropical peatland carbon pool. *Glob. Chang. Biol.*, **17**, 798–818, doi:10.1111/j.1365-2486.2010.02279.x. http://doi.wiley.com/10.1111/j.1365-2486.2010.02279.x (Accessed June 3, 2018).

Palacios-Agundez, I., M. Onaindia, M. Potschin, J. A. Tratalos, I. Madariaga, and R. Haines-Young, 2015: Relevance for decision making of spatially explicit, participatory scenarios for ecosystem services in an area of a high current demand. *Environ. Sci. Policy*, **54**, 199–209, doi:10.1016/j.envsci.2015.07.002. http://dx.doi.org/10.1016/j.envsci.2015.07.002.

Palm, C., H. Blanco-Canqui, F. DeClerck, L. Gatere, and P. Grace, 2014: Conservation agriculture and ecosystem services: An overview. *Agric. Ecosyst. Environ.*, **187**, 87–105, doi:10.1016/J.AGEE.2013.10.010. https://www.sciencedirect.com/science/article/pii/S0167880913003502 (Accessed June 5, 2018).

Palmer, J. R., 2014: Biofuels and the politics of land-use change: Tracing the interactions of discourse and place in European policy making. *Environ. Plan. A*, doi:10.1068/a4684.

Pan, G., P. Smith, and W. Pan, 2009: The role of soil organic matter in maintaining the productivity and yield stability of cereals in China. *Agric. Ecosyst. Environ.*, **129**, 344–348, doi:10.1016/J.AGEE.2008.10.008. https://www.sciencedirect.com/science/article/pii/S0167880908002703 (Accessed May 30, 2018).

Pan, Y., and Coauthors, 2011: A large and persistent carbon sink in the world's forests. *Science*, **333**, 988–993, doi:10.1126/science.1201609. http://www.ncbi.nlm.nih.gov/pubmed/21764754 (Accessed June 2, 2018).

Panagos, P., P. Borrelli, J. Poesen, C. Ballabio, E. Lugato, K. Meusburger, L. Montanarella, and C. Alewell, 2015: The new assessment of soil loss by water erosion in Europe. *Environ. Sci. Policy*, doi:10.1016/j.envsci.2015.08.012.

Paoli, G. D., and Coauthors, 2010: Biodiversity Conservation in the REDD. *Carbon Balance Manag.*, doi:10.1186/1750-0680-5-7.

Parish, E. S., and Coauthors, 2012: Multimetric spatial optimization of switchgrass plantings across a watershed. *Biofuels, Bioprod. Biorefining*, doi:10.1002/bbb.342.

Parish, F., A. Sirin, D. Charman, H. Joosten, T. Minayeva, M. Silvius, and L. Stringer, 2008: *Assessment* *on peatlands, biodiversity and climate change: main report.* Wetlands International , Wageningen, 179 pp. http://journals.sagepub.com/doi/10.1177/0309133310365595 (Accessed June 7, 2018).

Patnaik, A., J. Jongerden, and G. Ruivenkamp, 2017: Repossession through sharing of and access to seeds: different cases and practices. *Int. Rev. Sociol.*, **27**, 179–201, doi:10.1080/03906701.2016.1235213. https://www.tandfonline.com/doi/full/10.1080/03906701.2016.1235213 (Accessed June 1, 2018).

Paudyal, K., H. Baral, L. Putzel, S. Bhandari, and R. J. Keenan, 2017: Change in land use and ecosystem services delivery from community-based forest landscape restoration in the Phewa Lake watershed , Nepal. *Int. For. Rev.*, **19**, 1–14.

Paustian, K., J. Lehmann, S. Ogle, D. Reay, G. P. Robertson, and P. Smith, 2016: Climate-smart soils. *Nature*, **532**, 49–57, doi:10.1038/nature17174. http://www.nature.com/doifinder/10.1038/nature17174 (Accessed June 2, 2018).

Pelling, M., and C. High, 2005: Understanding adaptation: What can social capital offer assessments of adaptive capacity? *Glob. Environ. Chang.*, **15**, 308–319, doi:10.1016/J.GLOENVCHA.2005.02.001. https://www.sciencedirect.com/science/article/pii/S0959378005000154 (Accessed June 2, 2018).

Pender, J., E. Nkonya, P. Jagger, D. Sserunkuuma, and H. Ssali, 2004: Strategies to increase agricultural productivity and reduce land degradation: Evidence from Uganda. *Agricultural Economics*, Vol. 31 of, 181–195.

Pereira, H. M., and Coauthors, 2010a: Scenarios for Global Biodiversity in the 21st Century. *Science (80-.).*, **330**, 1496–1502. http://api.iucnredlist.org/go/panthera-leo.

——, and Coauthors, 2010b: Scenarios for Global Biodiversity in the 21st Century: Supporting information. *Science (80-.).*, **330**, 1496–1502. http://api.iucnredlist.org/go/panthera-leo.

Pereira, L. S., 2002a: by.

——, 2002b: *Coping with water scarcity*. Technical Documents in Hydrotogy No 58, 49-200 pp.

——, 2005: Combating Desertification: Water Conservation and Water Saving Issues in Agriculture.

Perreira, L. S., 2005: Combating Desertification : Water Conservation and Water Saving Issues in Agriculture. *New Medit N.1/2005*,.

Perugini, L., L. Caporaso, S. Marconi, A. Cescatti, B. Quesada, N. De Noblet-Ducoudré, J. I. House, and A. Arneth, 2017: Biophysical effects on temperature and precipitation due to land cover change. *Environ. Res. Lett.*, doi:10.1088/1748-9326/aa6b3f.

Peterson, N. D., 2012: Developing Climate Adaptation: The Intersection of Climate Research and Development Programmes in Index Insurance. *Dev. Change*, doi:10.1111/j.1467-7660.2012.01767.x.

Petit, B., and F. Montagnini, 2006: Growth in pure and mixed plantations of tree species used in reforesting rural areas of the humid region of Costa Rica, Central America. *For. Ecol. Manage.*, doi:10.1016/j.foreco.2006.05.030.

Pielke, R. A., and Coauthors, 2011: Land use/land cover changes and climate: Modeling analysis and observational evidence. *Wiley Interdiscip. Rev. Clim. Chang.*, doi:10.1002/wcc.144.

Plaza, C., N. Senesi, J. C. García-Gil, G. Brunetti, V. D'Orazio, and A. Polo, 2002: Effects of pig slurry application on soils and soil humic acids. *J. Agric. Food Chem.*, doi:10.1021/jf020195p.

Poeplau, C., and A. Don, 2015: Carbon sequestration in agricultural soils via cultivation of cover crops – A meta-analysis. *Agric. Ecosyst. Environ.*, **200**, 33–41, doi:10.1016/J.AGEE.2014.10.024. https://www.sciencedirect.com/science/article/pii/S0167880914004873 (Accessed May 31, 2018).

Pohnan, E., H. Ompusunggu, and C. Webb, 2015: Does Tree Planting Change Minds? Assessing the Use of Community Participation in Reforestation to Address Illegal Logging in West Kalimantan. *Trop. Conserv. Sci.*, **8**, 45–57, doi:10.1177/194008291500800107. http://journals.sagepub.com/doi/10.1177/194008291500800107 (Accessed June 2, 2018).

Pokharel, B. K., P. Branney, M. Nurse, and Y. B. Malla, 2007: Community Forestry: Conserving Forests, Sustaining Livelihoods and Strengthening Democracy. *J. For. Livelihood*, **6**, 8–19, doi:10.3126/jfl.v6i2.2321.

Popp, A., and Coauthors, 2011: The economic potential of bioenergy for climate change mitigation with special attention given to implications for the land system. *Environ. Res. Lett.*, **6**, doi:10.1088/1748-9326/6/3/034017.

——, and Coauthors, 2014: Land-use transition for bioenergy and climate stabilization: Model comparison of drivers, impacts and interactions with other land use based mitigation options. *Clim. Change*, doi:10.1007/s10584-013-0926-x.

——, and Coauthors, 2017a: Land-use futures in the shared socio-economic pathways. *Glob. Environ. Chang.*, doi:10.1016/j.gloenvcha.2016.10.002.

——, and Coauthors, 2017b: Land-use futures in the shared socio-economic pathways. *Glob. Environ. Chang.*, **42**, 331–345, doi:10.1016/j.gloenvcha.2016.10.002. http://dx.doi.org/10.1016/j.gloenvcha.2016.10.002.

Porter, J. R., L. Xie, A. J. Challinor, K. Cochrane, S. M. Howden, M. M. Iqbal, D. B. Lobell, and M. I. Travasso, 2014: Food security and food production systems. *Climate Change 2014: Impacts, Adaptation, and Vulnerability. Part A: Global and Sectoral Aspects. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*.

Porter, S. D., D. S. Reay, P. Higgins, and E. Bomberg, 2016: A half-century of production-phase greenhouse gas emissions from food loss & waste in the global food supply chain. *Sci. Total Environ.*, **571**, 721–729, doi:10.1016/j.scitotenv.2016.07.041.

Potts, S. G., J. C. Biesmeijer, C. Kremen, P. Neumann, O. Schweiger, and W. E. Kunin, 2010: Global pollinator declines: Trends, impacts and drivers. *Trends Ecol. Evol.*, doi:10.1016/j.tree.2010.01.007.

Poulton, C., J. Kydd, S. Wiggins, and A. Dorward, 2006: State intervention for food price stabilisation in Africa: Can it work? *Food Policy*, **31**, 342–356, doi:10.1016/J.FOODPOL.2006.02.004. https://www.sciencedirect.com/science/article/pii/S0306919206000261 (Accessed June 1, 2018).

Powlson, D. S., C. M. Stirling, M. L. Jat, B. G. Gerard, C. A. Palm, P. A. Sanchez, and K. G. Cassman, 2014: Limited potential of no-till agriculture for climate change mitigation. *Nat. Clim. Chang.*, **4**, 678–683, doi:10.1038/nclimate2292. http://www.nature.com/doifinder/10.1038/nclimate2292 (Accessed June 5, 2018).

——, ——, C. Thierfelder, R. P. White, and M. L. Jat, 2016: Does conservation agriculture deliver climate change mitigation through soil carbon sequestration in tropical agro-ecosystems? *Agric. Ecosyst. Environ.*, **220**, 164–174, doi:10.1016/J.AGEE.2016.01.005. https://www.sciencedirect.com/science/article/pii/S0167880916300056 (Accessed June 5, 2018).

Pozzi, W., and Coauthors, 2013: Toward Global Drought Early Warning Capability: Expanding International Cooperation for the Development of a Framework for Monitoring and Forecasting. *Bull. Am. Meteorol. Soc.*, **94**, 776–785, doi:10.1175/BAMS-D-11-00176.1. http://journals.ametsoc.org/doi/abs/10.1175/BAMS-D-11-00176.1 (Accessed June 1, 2018).

Pradhan, P., L. Costa, D. Rybski, W. Lucht, and J. P. Kropp, 2017: A Systematic Study of Sustainable Development Goal (SDG) Interactions Earth ' s Future. *Earth's Futur.*, 1169–1179, doi:10.1002/eft2.266.

Prathapar, S. A., 1988: *How to Manage Salinity in Irrigated Lands: A Selective Review with Particular Reference to Irrigation in Developing Countries*.

Pretty, J., and Z. Bharucha, 2014: Sustainable intensification in agricultural systems. *Ann. Botony*, **114**, 1571–1596. https://academic.oup.com/aob/article-abstract/114/8/1571/210078 (Accessed May 30, 2018).

Pritchard, M. F., 2013: Land, power and peace: Tenure formalization, agricultural reform, and livelihood insecurity in rural Rwanda. *Land use policy*, **30**, 186–196, doi:10.1016/J.LANDUSEPOL.2012.03.012. https://www.sciencedirect.com/science/article/pii/S0264837712000464 (Accessed May 31, 2018).

Putz, F. E., and Coauthors, 2012: Sustaining conservation values in selectively logged tropical forests: The attained and the attainable. *Conserv. Lett.*, **5**, 296–303, doi:10.1111/j.1755-263X.2012.00242.x.

Qaiser, K., S. Ahmad, W. Johnson, and J. Batista, 2011: Evaluating the impact of water conservation on fate of outdoor water use: A study in an arid region. *J. Environ. Manage.*, **92**, 2061–2068, doi:10.1016/j.jenvman.2011.03.031. http://dx.doi.org/10.1016/j.jenvman.2011.03.031.

Qian, J., Y. Peng, C. Luo, C. Wu, and Q. Du, 2015: Urban Land Expansion and Sustainable Land Use Policy in Shenzhen: A Case Study of China's Rapid Urbanization. *Sustainability*, **8**, 16, doi:10.3390/su8010016. http://www.mdpi.com/2071-1050/8/1/16 (Accessed May 31, 2018).

Qin, Z., Q. Zhuang, and M. Chen, 2012: Impacts of land use change due to biofuel crops on carbon balance, bioenergy production, and agricultural yield, in the conterminous United States. *GCB Bioenergy*, doi:10.1111/j.1757-1707.2011.01129.x.

Queiroz, C., R. Beilin, C. Folke, and R. Lindborg, 2014: Farmland abandonment: threat or opportunity for biodiversity conservation? A global review. *Front. Ecol. Environ.*, **12**, 288–296, doi:10.1890/120348. http://doi.wiley.com/10.1890/120348 (Accessed June 2, 2018).

Le Quéré, C., and Coauthors, 2016: Global Carbon Budget 2016. *Earth Syst. Sci. Data*, **8**, 605–649, doi:10.5194/essd-8-605-2016.

Ramachandran Nair, P. K., V. D. Nair, B. Mohan Kumar, and J. M. Showalter, 2010: Carbon Sequestration in Agroforestry Systems. *Advances in Agronomy*, D.L. Sparks, Ed., Vol. 108 of, p. 237–307.

Ramón Vallejo, V., A. Smanis, E. Chirino, D. Fuentes, A. Valdecantos, and A. Vilagrosa, 2012: Perspectives in dryland restoration: Approaches for climate change adaptation. *New For.*, **43**, 561–579, doi:10.1007/s11056-012-9325-9.

Rao, S., and Coauthors, 2016: A multi-model assessment of the co-benefits of climate mitigation for global air quality. *Environ. Res. Lett.*, **11**, 124013, doi:10.1088/1748-9326/11/12/124013. http://stacks.iop.org/1748-9326/11/i=12/a=124013?key=crossref.b4de2f0ebe431be6d994766c93883b95 (Accessed May 1, 2018).

Ravetz, J., 2016: *City-region 2020: integrated planning for a sustainable environment*. https://www.taylorfrancis.com/books/9781134900923 (Accessed May 1, 2018).

Reganold, J. P., and J. M. Wachter, 2016: Organic agriculture in the twenty-first century. *Nat. Plants*, **2**, 15221, doi:10.1038/nplants.2015.221. http://www.nature.com/articles/nplants2015221 (Accessed June 1, 2018).

Reichstein, M., and Coauthors, 2013: Climate extremes and the carbon cycle. *Nature*, **500**, 287–295, doi:10.1038/nature12350. http://www.nature.com/articles/nature12350 (Accessed June 5, 2018).

Reidsma, P., F. Ewert, A. O. Lansink, and R. Leemans, 2010: Adaptation to climate change and climate variability in European agriculture: The importance of farm level responses. *Eur. J. Agron.*, doi:10.1016/j.eja.2009.06.003.

Reilly, J., and Coauthors, 2007: Global economic effects of changes in crops, pasture, and forests due to changing climate, carbon dioxide, and ozone. *Energy Policy*, **35**, 5370–5383, doi:10.1016/j.enpol.2006.01.040.

Reilly, J., and Coauthors, 2012: Using land to mitigate climate change: Hitting the target, recognizing the trade-offs. *Environ. Sci. Technol.*, doi:10.1021/es2034729.

Reisman, E., 2017: Troubling Tradition, Community, and Self-Reliance: Reframing Expectations for Village Seed Banks. *World Dev.*, **98**, 160–168, doi:10.1016/J.WORLDDEV.2017.04.024. https://www.sciencedirect.com/science/article/pii/S0305750X17301353 (Accessed June 1, 2018).

Ren, Y., Y. Lü, B. Fu, and K. Zhang, 2017: Biodiversity and Ecosystem Functional Enhancement by Forest Restoration: A Meta-analysis in China. *L. Degrad. Dev.*, doi:10.1002/ldr.2728.

Rengasamy, P., 2006: World salinization with emphasis on Australia. *J. Exp. Bot.*, **57**, 1017–1023, doi:10.1093/jxb/erj108.

Revi, A., D. E. Satterthwaite, F. Aragón-Durand, J. Corfee-Morlot, R. B. R. Kiunsi, M. Pelling, Roberts. D.C., and W. Solecki, 2014: Urban Areas. *Climate Change 2014: Impacts, Adaptation, and Vulnerability. Part A: Global and Sectoral Aspects. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*.

Rey Benayas, J. M., A. C. Newton, A. Diaz, and J. M. Bullock, 2009: Enhancement of biodiversity and ecosystem services by ecological restoration: A meta-analysis. *Science (80-.).*, doi:10.1126/science.1172460.

Reyer, C., M. Guericke, and P. L. Ibisch, 2009: Climate change mitigation via afforestation, reforestation and deforestation avoidance: and what about adaptation to environmental change? *New For.*, **38**, 15–34, doi:10.1007/s11056-008-9129-0. http://link.springer.com/10.1007/s11056-008-9129-0 (Accessed June 1, 2018).

Reynolds, J. F., and Coauthors, 2007: Global Desertificaiton: Building a Science for Dryland Development. *Science (80-.).*,.

Riahi, K., and Coauthors, 2017a: The Shared Socioeconomic Pathways and their energy, land use, and greenhouse gas emissions implications: An overview. *Glob. Environ. Chang.*, **42**, 153–168, doi:10.1016/j.gloenvcha.2016.05.009.

——, and Coauthors, 2017b: The Shared Socioeconomic Pathways and their energy, land use, and greenhouse gas emissions implications: An overview. *Glob. Environ. Chang.*, doi:10.1016/j.gloenvcha.2016.05.009.

Ridoutt, B., P. Sanguansri, L. Bonney, S. Crimp, G. Lewis, and L. Lim-Camacho, 2016: Climate Change Adaptation Strategy in the Food Industry—Insights from Product Carbon and Water Footprints. *Climate*, doi:10.3390/cli4020026.

Riggs, R. A., J. Sayer, C. Margules, A. K. Boedhihartono, J. D. Langston, and H. Sutanto, 2016: Forest tenure and conflict in Indonesia: Contested rights in Rempek Village, Lombok. *Land use policy*, **57**, 241–249, doi:10.1016/j.landusepol.2016.06.002. http://linkinghub.elsevier.com/retrieve/pii/S0264837716305403.

Ritzema, R., R. Frelat, S. Douxchamps, and S. Silvestri, 2017: intensification likely to make farm households food-adequate? A simple food availability analysis across smallholder farming systems from East and West Africa. *Food Secur.*,. https://link.springer.com/article/10.1007/s12571-016-0638-y (Accessed May 10, 2018).

Robert, M., A. Thomas, M. Sekhar, S. Badiger, L. Ruiz, M. Willaume, D. Leenhardt, and J.-E. Bergez, 2017: Farm Typology in the Berambadi Watershed (India): Farming Systems Are Determined by Farm Size and Access to Groundwater. *Water*, **9**, 51, doi:10.3390/w9010051. http://www.mdpi.com/2073-4441/9/1/51 (Accessed June 2, 2018).

Robertson, G. P., and Coauthors, 2017: Cellulosic biofuel contributions to a sustainable energy future: Choices and outcomes. *Science (80-.).*, doi:10.1126/science.aal2324.

Robinson, B. E., M. B. Holland, and L. Naughton-Treves, 2014: Does secure land tenure save forests?

A meta-analysis of the relationship between land tenure and tropical deforestation. *Glob. Environ. Chang.*, **29**, 281–293, doi:10.1016/j.gloenvcha.2013.05.012. http://dx.doi.org/10.1016/j.gloenvcha.2013.05.012.

Robledo, C., N. Clot, A. Hammill, and B. Riché, 2012: The role of forest ecosystems in community-based coping strategies to climate hazards: Three examples from rural areas in Africa. *For. Policy Econ.*, doi:10.1016/j.forpol.2011.04.006.

Rocha, C., 2016: Belo Horizonte: the Opportunities and Challenges of Urban Food Security Policy. http://fondazionefeltrinelli.it/app/uploads/2016/05/The-Governance-of-City-Food-Systems_The-Cases-Study-from-Around-The-World.pdf#page=29 (Accessed May 1, 2018).

Roesch-McNally, G. E., A. D. Basche, J. G. Arbuckle, J. C. Tyndall, F. E. Miguez, T. Bowman, and R. Clay, 2017: The trouble with cover crops: Farmers' experiences with overcoming barriers to adoption. *Renew. Agric. Food Syst.*, 1–12, doi:10.1017/S1742170517000096. https://www.cambridge.org/core/product/identifier/S1742170517000096/type/journal_article (Accessed May 31, 2018).

Rogelj, J., and Coauthors, 2011: Emission pathways consistent with a 2°C global temperature limit. *Nat. Clim. Chang.*, doi:10.1038/nclimate1258.

——, G. Luderer, R. C. Pietzcker, E. Kriegler, M. Schaeffer, V. Krey, and K. Riahi, 2015: Energy system transformations for limiting end-of-century warming to below 1.5 °C. *Nat. Clim. Chang.*, doi:10.1038/nclimate2572.

Rosenstock, T. S., K. L. Tully, C. Arias-Navarro, H. Neufeldt, K. Butterbach-Bahl, and L. V. Verchot, 2014: Agroforestry with N2-fixing trees: Sustainable development's friend or foe? *Curr. Opin. Environ. Sustain.*, doi:10.1016/j.cosust.2013.09.001.

Ruan, L., and G. P. Robertson, 2017: Reduced Snow Cover Increases Wintertime Nitrous Oxide (N2O) Emissions from an Agricultural Soil in the Upper U.S. Midwest. *Ecosystems*, doi:10.1007/s10021-016-0077-9.

——, A. K. Bhardwaj, S. K. Hamilton, and G. P. Robertson, 2016: Nitrogen fertilization challenges the climate benefit of cellulosic biofuels. *Environ. Res. Lett.*, doi:10.1088/1748-9326/11/6/064007.

de Ruiter, H., J. I. Macdiarmid, R. B. Matthews, T. Kastner, L. R. Lynd, and P. Smith, 2017: Total global agricultural land footprint associated with UK food supply 1986–2011. *Glob. Environ. Chang.*, **43**, 72–81, doi:10.1016/J.GLOENVCHA.2017.01.007. https://www.sciencedirect.com/science/article/pii/S0959378017301176 (Accessed May 30, 2018).

Salam, M. A., T. Noguchi, and R. Pothitan, 2006: Community forest management in Thailand: Current situation and dynamics in the context of sustainable development. *New For.*, **31**, 273–291, doi:10.1007/s11056-005-7483-8.

Salazar, A., G. Baldi, M. Hirota, J. Syktus, and C. McAlpine, 2015: Land use and land cover change impacts on the regional climate of non-Amazonian South America: A review. *Glob. Planet. Change*, **128**, 103–119, doi:10.1016/J.GLOPLACHA.2015.02.009. https://www.sciencedirect.com/science/article/pii/S0921818115000557 (Accessed June 3, 2018).

Salvati, L., A. Sabbi, D. Smiraglia, and M. Zitti, 2014: Does forest expansion mitigate the risk of desertification? Exploring soil degradation and land-use changes in a Mediterranean country. *Int. For. Rev.*, doi:10.1505/146554814813484149.

Sánchez, J., M. D. Curt, and J. Fernández, 2017: Approach to the potential production of giant reed in surplus saline lands of Spain. *GCB Bioenergy*, doi:10.1111/gcbb.12329.

Sanderson, M. A., and Coauthors, 2013: Diversification and ecosystem services for conservation agriculture: Outcomes from pastures and integrated crop–livestock systems. *Renew. Agric. Food Syst.*, **28**, 129–144, doi:10.1017/S1742170512000312. http://www.journals.cambridge.org/abstract_S1742170512000312 (Accessed May 31, 2018).

Santangeli, A., T. Toivonen, F. M. Pouzols, M. Pogson, A. Hastings, P. Smith, and A. Moilanen, 2016: Global change synergies and trade-offs between renewable energy and biodiversity. *GCB Bioenergy*, **8**, 941–951, doi:10.1111/gcbb.12299. http://doi.wiley.com/10.1111/gcbb.12299 (Accessed May 30, 2018).

Santika, T., and Coauthors, 2017: Community forest management in Indonesia: Avoided deforestation in the context of anthropogenic and climate complexities. *Glob. Environ. Chang.*, **46**, 60–71, doi:10.1016/J.GLOENVCHA.2017.08.002. https://www.sciencedirect.com/science/article/pii/S0959378016305933 (Accessed June 2, 2018).

Scarano, F. R., 2017: Ecosystem-based adaptation to climate change: concept, scalability and a role for conservation science. *Perspect. Ecol. Conserv.*, **15**, 65–73, doi:10.1016/J.PECON.2017.05.003. https://www.sciencedirect.com/science/article/pii/S1679007316301621 (Accessed June 1, 2018).

Schatz, J., and C. J. Kucharik, 2015: Urban climate effects on extreme temperatures in Madison, Wisconsin, USA. *Environ. Res. Lett.*, **10**, 94024, doi:10.1088/1748-9326/10/9/094024. http://stacks.iop.org/1748-9326/10/i=9/a=094024?key=crossref.296eeb5909d7fedd3e8fecf131a2a592 (Accessed June 2, 2018).

Scherr, S. J., and S. Yadav, 1996: Land Degradation in the Developing World: Implications for Food, Agriculture, and the Environment to 2020.

Schmitz, C., A. Biewald, H. Lotze-Campen, A. Popp, J. P. Dietrich, B. Bodirsky, M. Krause, and I. Weindl, 2012: Trading more food: Implications for land use, greenhouse gas emissions, and the food system. *Glob. Environ. Chang.*, doi:10.1016/j.gloenvcha.2011.09.013.

Schueler, V., S. Fuss, J. C. Steckel, U. Weddige, and T. Beringer, 2016: Productivity ranges of sustainable biomass potentials from non-agricultural land. *Environ. Res. Lett.*, doi:10.1088/1748-9326/11/7/074026.

Schuiling, R. D., and P. Krijgsman, 2006: Enhanced Weathering: An Effective and Cheap Tool to Sequester Co2. *Clim. Change*, **74**, 349–354, doi:10.1007/s10584-005-3485-y. http://link.springer.com/10.1007/s10584-005-3485-y (Accessed June 1, 2018).

Scott, A., 2017: Making governance work for water–energy–food nexus approaches.

Seidl, R., M. J. Schelhaas, W. Rammer, and P. J. Verkerk, 2014: Increasing forest disturbances in Europe and their impact on carbon storage. *Nat. Clim. Chang.*, doi:10.1038/nclimate2318.

Settele, J., R. J. Scholes, R. A. Betts, S. Bunn, P. Leadley, D. Nepstad, J. T. Overpeck, and M. A. Toboada, 2014: Terrestrial and inland water systems. *Climate Change 2014: Impacts, Adaptation, and Vulnerability. Part A: Global and Sectoral Aspects. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*.

Shackley, S., J. Hammond, J. Gaunt, and R. Ibarrola, 2011: The feasibility and costs of biochar deployment in the UK. *Carbon Manag.*, **2**, 335–356, doi:10.4155/cmt.11.22. http://www.tandfonline.com/doi/abs/10.4155/cmt.11.22 (Accessed May 30, 2018).

Shah, T., M. Giordano, and A. Mukherji, 2012: Political economy of the energy-groundwater nexus in India: exploring issues and assessing policy options. *Hydrogeol. J.*, **20**, 995–1006, doi:10.1007/s10040-011-0816-0. http://link.springer.com/10.1007/s10040-011-0816-0 (Accessed June 2, 2018).

Shcherbak, I., N. Millar, and G. P. Robertson, 2014: Global metaanalysis of the nonlinear response of soil nitrous oxide (N2O) emissions to fertilizer nitrogen. *Proc. Natl. Acad. Sci. U. S. A.*, **111**, 9199–9204, doi:10.1073/pnas.1322434111. http://www.ncbi.nlm.nih.gov/pubmed/24927583 (Accessed June 3, 2018).

Sheahan, M., and C. B. Barrett, 2017: Review: Food loss and waste in Sub-Saharan Africa. *Food Policy*, **70**, 1–12, doi:10.1016/j.foodpol.2017.03.012. https://www.sciencedirect.com/science/article/pii/S0306919217302440 (Accessed May 10,

2018).

Shen, X., L. Wang, C. Wu, T. Lv, Z. Lu, W. Luo, and G. Li, 2017: Local interests or centralized targets? How China's local government implements the farmland policy of Requisition–Compensation Balance. *Land use policy*, **67**, 716–724, doi:10.1016/J.LANDUSEPOL.2017.06.012. https://www.sciencedirect.com/science/article/pii/S0264837715301952 (Accessed May 31, 2018).

Shirsath, P. B., P. K. Aggarwal, P. K. Thornton, and A. Dunnett, 2017: Prioritizing climate-smart agricultural land use options at a regional scale. *Agric. Syst.*, **151**, 174–183, doi:10.1016/J.AGSY.2016.09.018. https://www.sciencedirect.com/science/article/pii/S0308521X16305881 (Accessed June 2, 2018).

Shisanya, S., and P. Mafongoya, 2016: Adaptation to climate change and the impacts on household food security among rural farmers in uMzinyathi District of Kwazulu-Natal, South Africa. *Food Secur.*, **8**, 597–608, doi:10.1007/s12571-016-0569-7. http://link.springer.com/10.1007/s12571-016-0569-7 (Accessed June 1, 2018).

Siikamäki, J., and S. C. Newbold, 2012: Potential biodiversity benefits from international programs to reduce carbon emissions from deforestation. *Ambio*, doi:10.1007/s13280-011-0243-4.

Silalahi, M., A. Utomo, T. Walsh, A. Ayat, Andriansyah, and S. Bashir, 2017: Indonesia's ecosystem restoration concessions. *Unasylva*, **68**, 63–70.

Sileshi, G., E. Kuntashula, P. Matakala, P. N.-A. Systems, and U. 2008, 2008: Farmers' perceptions of tree mortality, pests and pest management practices in agroforestry in Malawi, Mozambique and Zambia. *Springer*,. https://link.springer.com/article/10.1007/s10457-007-9082-5 (Accessed May 30, 2018).

Simpson, M., and Coauthors, 2016: Decision Analysis for Management of Natural Hazards. *Annu. Rev. Environ. Resour.*, **41**, 489–516, doi:10.1146/annurev-environ-110615-090011. http://www.annualreviews.org/doi/10.1146/annurev-environ-110615-090011 (Accessed May 31, 2018).

Sims, R., and Coauthors, 2014: Transport. *Climate Change 2014: Mitigation of Climate Change. Contribution of Working Group III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*, O. Edenhofer et al., Eds., Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Skees, J. R., and B. Collier, 2012: The Roles of Weather Insurance and the Carbon Market. *Greening the Financial Sector*, Springer Berlin Heidelberg, Berlin, Heidelberg, 111–164 http://link.springer.com/10.1007/978-3-642-05087-9_4 (Accessed June 1, 2018).

Slade, R., A. Bauen, and R. Gross, 2014: Global bioenergy resources. *Nat. Clim. Chang.*, doi:10.1038/nclimate2097.

Smit, B., and J. Wandel, 2006: Adaptation, adaptive capacity and vulnerability. *Glob. Environ. Chang.*, doi:10.1016/j.gloenvcha.2006.03.008.

Smit, W., 2016: Urban governance and urban food systems in Africa: Examining the linkages. *Cities*,. https://www.sciencedirect.com/science/article/pii/S026427511630083X (Accessed May 1, 2018).

Smith, J., A. Abegaz, R. B. Matthews, M. Subedi, E. R. Orskov, V. Tumwesige, and P. Smith, 2014a: What is the potential for biogas digesters to improve soil fertility and crop production in Sub-Saharan Africa? *Biomass and Bioenergy*, doi:10.1016/j.biombioe.2014.02.030.

Smith, P., 2013: Delivering food security without increasing pressure on land. *Glob. Food Sec.*, doi:10.1016/j.gfs.2012.11.008.

Smith, P., 2018: Managing the global land resource. *Proc. R. Soc. B* , **(in review**.

Smith, P., and Coauthors, 2007: *Agriculture. In Climate Change 2007: Mitigation. Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate*

Change. B. Metz, O. Davidson, P. Bosch, R. Dave, and L. Meyer, Eds. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.,.

——, and Coauthors, 2008: Greenhouse gas mitigation in agriculture. *Philos. Trans. R. Soc. Lond. B. Biol. Sci.*, **363**, 789–813, doi:10.1098/rstb.2007.2184. http://www.ncbi.nlm.nih.gov/pubmed/17827109 (Accessed May 30, 2018).

——, and Coauthors, 2013: How much land-based greenhouse gas mitigation can be achieved without compromising food security and environmental goals? *Glob. Chang. Biol.*, doi:10.1111/gcb.12160.

Smith, P., H. Clark, H. Dong, E. Elsiddig, and H. Haberl, 2014b: Agriculture, forestry and other land use (AFOLU). http://pure.iiasa.ac.at/11115/ (Accessed December 20, 2017).

——, and Coauthors, 2015: Biogeochemical cycles and biodiversity as key drivers of ecosystem services provided by soils. **1**, 665–685, doi:10.5194/soil-1-665-2015. www.soil-journal.net/1/665/2015/ (Accessed May 30, 2018).

——, S. Davis, F. Creutzig, S. Fuss, and … J. M.-, 2016a: Biophysical and economic limits to negative CO2 emissions. *Nat. Clim.*,. https://www.nature.com/articles/nclimate2870 (Accessed May 1, 2018).

Smith, P., S. J. Davis, F. Creutzig, S. Fuss, J. Minx, and B. Gabrielle, 2016b: Biophysical and economic limits to negative CO 2 emissions. *Nat. Clim. Chang.*, doi:10.1038/NCLIMATE2870.

——, R. S. Haszeldine, and S. M. Smith, 2016c: Preliminary assessment of the potential for, and limitations to, terrestrial negative emission technologies in the UK. *Environ. Sci. Process. Impacts*, doi:10.1039/C6EM00386A.

——, J. Price, A. Molotoks, R. Warren, and Y. Malhi, 2018: Impacts on terrestrial biodiversity of moving from a 2°C to a 1.5°C target. *Philos. Trans. A. Math. Phys. Eng. Sci.*, **376**, 20160456, doi:10.1098/rsta.2016.0456. http://www.ncbi.nlm.nih.gov/pubmed/29610386 (Accessed June 3, 2018).

Smith, V. H., and J. W. Glauber, 2012: Agricultural Insurance in Developed Countries: Where Have We Been and Where Are We Going? *Appl. Econ. Perspect. Policy*, **34**, 363–390, doi:10.1093/aepp/pps029. https://academic.oup.com/aepp/article-lookup/doi/10.1093/aepp/pps029 (Accessed May 31, 2018).

Smyth, C. E., G. Stinson, E. Neilson, T. C. Lemprière, M. Hafer, G. J. Rampley, and W. A. Kurz, 2014: Quantifying the biophysical climate change mitigation potential of Canada's forest sector. *Biogeosciences*, **11**, 3515–3529, doi:10.5194/bg-11-3515-2014.

Snyder, C., E. Davidson, P. Smith, and R. Venterea, 2014: Agriculture: sustainable crop and animal production to help mitigate nitrous oxide emissions. *Curr. Opin. Environ. Sustain.*, **9**, 46–54. http://www.sciencedirect.com/science/article/pii/S1877343514000384 (Accessed December 20, 2017).

Soane, B. D., and C. van Ouwerkerk, 1994: Soil Compaction Problems in World Agriculture. *Dev. Agric. Eng.*, **11**, 1–21, doi:10.1016/B978-0-444-88286-8.50009-X. https://www.sciencedirect.com/science/article/pii/B978044488286850009X (Accessed May 31, 2018).

Somanathan, E., R. Prabhakar, B. S. Mehta, and B. Singh, 2009: Decentralization for cost-effective conservation. *Proc. Natl. Acad. Sci.*, **106**, 4143–4147, doi:10.1073/pnas.0810049106. http://www.pubmedcentral.nih.gov/articlerender.fcgi?artid=2657449&tool=pmcentrez&rendertype=abstract.

Somorin, O. A., H. C. P. Brown, I. J. Visseren-Hamakers, D. J. Sonwa, B. Arts, and J. Nkem, 2012: The Congo Basin forests in a changing climate: Policy discourses on adaptation and mitigation (REDD+). *Glob. Environ. Chang.*, **22**, 288–298, doi:10.1016/J.GLOENVCHA.2011.08.001. https://www.sciencedirect.com/science/article/pii/S095937801100118X (Accessed June 3, 2018).

Somorin, O. A., I. J. Visseren-Hamakers, B. Arts, A.-M. Tiani, and D. J. Sonwa, 2016: Integration through interaction? Synergy between adaptation and mitigation (REDD+) in Cameroon. *Environ. Plan. C Gov. Policy*, **34**, 415–432, doi:10.1177/0263774X16645341. http://journals.sagepub.com/doi/10.1177/0263774X16645341 (Accessed June 3, 2018).

Song, G., M. Li, P. Fullana-i-Palmer, D. Williamson, and Y. Wang, 2017: Dietary changes to mitigate climate change and benefit public health in China. *Sci. Total Environ.*, **577**, 289–298, doi:10.1016/j.scitotenv.2016.10.184.

Song, W., and X. Deng, 2015: Effects of Urbanization-Induced Cultivated Land Loss on Ecosystem Services in the North China Plain. *Energies*, **8**, 5678–5693, doi:10.3390/en8065678. http://www.mdpi.com/1996-1073/8/6/5678 (Accessed May 31, 2018).

Sonntag, S., J. Pongratz, C. H. Reick, and H. Schmidt, 2016: Reforestation in a high-CO2 world—Higher mitigation potential than expected, lower adaptation potential than hoped for. *Geophys. Res. Lett.*, doi:10.1002/2016GL068824.

Sovacool, B. K., 2011: Hard and soft paths for climate change adaptation. *Clim. Policy*, doi:10.1080/14693062.2011.579315.

Specht, K., R. Siebert, I. Hartmann, and … U. F., 2014: Urban agriculture of the future: an overview of sustainability aspects of food production in and on buildings. *Agric. Human Values*,. https://link.springer.com/article/10.1007/s10460-013-9448-4 (Accessed May 1, 2018).

Spencer, B., and Coauthors, 2017: Case studies in co-benefits approaches to climate change mitigation and adaptation. *J. Environ. Plan. Manag.*, **60**, 647–667, doi:10.1080/09640568.2016.1168287.

Speranza, C. I., B. Kiteme, P. Ambenje, U. Wiesmann, and S. Makali, 2010: Indigenous knowledge related to climate variability and change: Insights from droughts in semi-arid areas of former Makueni District, Kenya. *Clim. Change*, doi:10.1007/s10584-009-9713-0.

Srinivasan, V., S. Thompson, K. Madhyastha, G. Penny, K. Jeremiah, and S. Lele, 2015: Why is the Arkavathy River drying? A multiple-hypothesis approach in a data-scarce region. *Hydrol. Earth Syst. Sci*, **19**, 1950–2015, doi:10.5194/hess-19-1905-2015. www.hydrol-earth-syst-sci.net/19/1905/2015/ (Accessed June 2, 2018).

Stanturf, J. A., B. J. Palik, M. I. Williams, R. K. Dumroese, and P. Madsen, 2014: Forest Restoration Paradigms. *J. Sustain. For.*, **33**, S161–S194, doi:10.1080/10549811.2014.884004.

——, P. Kant, J.-P. B. Lillesø, S. Mansourian, M. Kleine, L. Graudal, and P. Madsen, 2015: *Forest Landscape Restoration as a Key Component of Climate Change Mitigation and Adaptation*. Vienna, Austria,.

Staples, M. D., R. Malina, and S. R. H. Barrett, 2017: The limits of bioenergy for mitigating global life-cycle greenhouse gas emissions from fossil fuels. *Nat. Energy*, doi:10.1038/nenergy.2016.202.

Stathers, T., R. Lamboll, and B. M. Mvumi, 2013: Postharvest agriculture in changing climates: Its importance to African smallholder farmers. *Food Secur.*, doi:10.1007/s12571-013-0262-z.

Steenwerth, K. L., and Coauthors, 2014: Climate-smart agriculture global research agenda: scientific basis for action. *Agric. Food Secur.*, **3**, 11, doi:10.1186/2048-7010-3-11. http://agricultureandfoodsecurity.biomedcentral.com/articles/10.1186/2048-7010-3-11 (Accessed June 3, 2018).

Stehfest, E., L. Bouwman, D. P. Van Vuuren, M. G. J. Den Elzen, B. Eickhout, and P. Kabat, 2009a: Climate benefits of changing diet. *Clim. Change*, **95**, 83–102, doi:10.1007/s10584-008-9534-6.

——, ——, ——, ——, ——, and ——, 2009b: Climate benefits of changing diet. *Clim. Change*, doi:10.1007/s10584-008-9534-6.

Sterling, S. M., A. Ducharne, and J. Polcher, 2013: The impact of global land-cover change on the terrestrial water cycle. *Nat. Clim. Chang.*, doi:Doi 10.1038/Nclimate1690.

Sternberg, T., and B. Batbuyan, 2013: Integrating the Hyogo Framework into Mongolia's disaster risk reduction (DRR) policy and management. *Int. J. Disaster Risk Reduct.*, doi:10.1016/j.ijdrr.2013.05.003.

Stevanovic, M., and Coauthors, 2016: Climate Change Impacts on Agricultural Welfare. *Sci. Adv.*, 1–10.

Stibig, H.-J., F. Achard, S. Carboni, R. Raši, and J. Miettinen, 2014: Change in tropical forest cover of Southeast Asia from 1990 to 2010. *Biogeosciences*, **11**, 247–258, doi:10.5194/bg-11-247-2014. www.biogeosciences.net/11/247/2014/ (Accessed June 2, 2018).

Stickler, M. M., H. Huntington, A. Haflett, S. Petrova, and I. Bouvier, 2017: Does de facto forest tenure affect forest condition? Community perceptions from Zambia. *For. Policy Econ.*, **85**, 32–45, doi:10.1016/j.forpol.2017.08.014.

Strassburg, B. B. N., A. E. Latawiec, L. G. Barioni, C. A. Nobre, V. P. da Silva, J. F. Valentim, M. Vianna, and E. D. Assad, 2014: When enough should be enough: Improving the use of current agricultural lands could meet production demands and spare natural habitats in Brazil. *Glob. Environ. Chang.*, **28**, 84–97, doi:10.1016/J.GLOENVCHA.2014.06.001. https://www.sciencedirect.com/science/article/pii/S0959378014001046 (Accessed June 2, 2018).

Strengers, B. J., J. G. Van Minnen, and B. Eickhout, 2008: The role of carbon plantations in mitigating climate change: potentials and costs. *Clim. Change*, **88**, 343–366, doi:10.1007/s10584-007-9334-4. http://link.springer.com/10.1007/s10584-007-9334-4 (Accessed June 1, 2018).

Stults, M., and S. C. Woodruff, 2017: Looking under the hood of local adaptation plans: shedding light on the actions prioritized to build local resilience to climate change. *Mitig. Adapt. Strateg. Glob. Chang.*, **22**, 1249–1279, doi:10.1007/s11027-016-9725-9.

Stutz, K. P., and F. Lang, 2017: Potentials and unknowns in managing coarse Woody debris for soil functioning. *Forests*, **8**, 37, doi:10.3390/f8020037.

Suding, K., and Coauthors, 2015: Committing to ecological restoration. *Science (80-.).*, **348**, 638 LP-640.

Sunderlin, W. D., S. Dewi, A. Puntodewo, D. Müller, A. Angelsen, and M. Epprecht, 2008: Why forests are important for global poverty alleviation: A spatial explanation. *Ecol. Soc.*, **13**, doi:24.

Surveys, S., F. Effersø, and R. J. Zafiryadis, 2013: *An Intergrated Management Approach for Water Security in All Sectors*. 14-16 pp.

Sutton, M. A., and Coauthors, 2007: Challenges in quantifying biosphere–atmosphere exchange of nitrogen species. *Environ. Pollut.*, **150**, 125–139, doi:10.1016/J.ENVPOL.2007.04.014. https://www.sciencedirect.com/science/article/pii/S0269749107001960 (Accessed May 30, 2018).

Sutton, P. C., S. J. Anderson, R. Costanza, and I. Kubiszewski, 2016: The ecological economics of land degradation: Impacts on ecosystem service values. *Ecol. Econ.*, **129**, 182–192, doi:10.1016/J.ECOLECON.2016.06.016. https://www.sciencedirect.com/science/article/pii/S0921800915301725 (Accessed June 3, 2018).

Suyanto, S., R. Pandu Permana, N. Khususiyah, and L. Joshi, 2005: Land tenure, agroforestry adoption, and reduction of fire hazard in a forest zone: A case study from Lampung, Sumatra, Indonesia. *Agrofor. Syst.*, **65**, 1–11, doi:10.1007/s10457-004-1413-1.

Suzuki, R., 2012: *Linking Adaptation and Mitigation through Community Forestry: Case Studies from Asia. RECOFTC – The Center for People and Forests*. 80 pp.

Tai, A. P. K., M. V. Martin, and C. L. Heald, 2014: Threat to future global food security from climate change and ozone air pollution. *Nat. Clim. Chang.*, **4**, 817–821, doi:10.1038/nclimate2317.

Tan, R., V. Beckmann, L. van den Berg, and F. Qu, 2009: Governing farmland conversion: Comparing China with the Netherlands and Germany. *Land use policy*, **26**, 961–974, doi:10.1016/J.LANDUSEPOL.2008.11.009.

https://www.sciencedirect.com/science/article/pii/S0264837708001543 (Accessed May 31, 2018).

Tao, Y., F. Li, X. Liu, D. Zhao, X. Sun, and L. X. Modelling, 2015: Variation in ecosystem services across an urbanization gradient: A study of terrestrial carbon stocks from Changzhou, China. *Ecol. Appl.*,. https://www.sciencedirect.com/science/article/pii/S0304380015001994 (Accessed May 1, 2018).

Taylor, L. L., and Coauthors, 2016: Enhanced weathering strategies for stabilizing climate and averting ocean acidification. *Nat. Clim. Chang.*, **6**, 402–406, doi:10.1038/nclimate2882. http://www.nature.com/articles/nclimate2882 (Accessed May 30, 2018).

Temba, B. A., Y. Sultanbawa, D. J. Kriticos, G. P. Fox, J. J. W. Harvey, and M. T. Fletcher, 2016: Tools for Defusing a Major Global Food and Feed Safety Risk: Nonbiological Postharvest Procedures to Decontaminate Mycotoxins in Foods and Feeds. *J. Agric. Food Chem.*, doi:10.1021/acs.jafc.6b03777.

The World Bank, 2011: *Rising Global Interest in Farmland: Can It Yield Sustainable and Equitable Benefits?* World Bank, Washington DC, http://elibrary.worldbank.org/doi/book/10.1596/978-0-8213-8591-3.

Thoa, N. T. K., G. Arduino, P. Bono, N. Van Giang, P. T. K. Van, and B. T. Vuong, 2008: Special Feature : Water Harvesting Technologies Special Feature : Water Harvesting Technologies Groundwater storage in the sand dunes of Viet Nam. 31–37.

Thomaier, S., K. Specht, D. Henckel, A. Dierich, R. Siebert, U. B. Freisinger, and M. Sawicka, 2015: Farming in and on urban buildings: Present practice and specific novelties of Zero-Acreage Farming (ZFarming). *Renew. Agric. Food Syst.*, **30**, 43–54, doi:10.1017/S1742170514000143. http://www.journals.cambridge.org/abstract_S1742170514000143 (Accessed June 2, 2018).

Thomalla, F., T. Downing, E. Spanger-Siegfried, G. Han, and J. Rockström, 2006: Reducing hazard vulnerability: Towards a common approach between disaster risk reduction and climate adaptation. *Disasters*, doi:10.1111/j.1467-9523.2006.00305.x.

Thomas, C. D., and Coauthors, 2013: Reconciling biodiversity and carbon conservation. *Ecol. Lett.*, doi:10.1111/ele.12054.

Thomas, S. C., and N. Gale, 2015: Biochar and forest restoration: a review and meta-analysis of tree growth responses. *New For.*, **46**, 931–946, doi:10.1007/s11056-015-9491-7.

Thornton, P. K., and M. Herrero, 2014: Climate change adaptation in mixed crop–livestock systems in developing countries. *Glob. Food Sec.*, **3**, 99–107, doi:10.1016/J.GFS.2014.02.002. https://www.sciencedirect.com/science/article/pii/S2211912414000108 (Accessed June 1, 2018).

Thuy, P. T., M. Moeliono, B. Locatelli, M. Brockhaus, M. Di Gregorio, and S. Mardiah, 2014: Integration of adaptation and mitigation in climate change and forest policies in Indonesia and Vietnam. *Forests*, doi:10.3390/f5082016.

Tilman, D., and M. Clark, 2014: Global diets link environmental sustainability and human health. *Nature*, **515**, 518–522, doi:10.1038/nature13959. http://www.nature.com/articles/nature13959 (Accessed May 30, 2018).

——, and ——, 2015: Food, Agriculture & the Environment: Can We Feed the World & Save the Earth? *Daedalus*, doi:10.1162/DAED_a_00350.

——, C. Balzer, J. Hill, and B. L. Befort, 2011: Global food demand and the sustainable intensification of agriculture. *Proc. Natl. Acad. Sci. U. S. A.*, **108**, 20260–20264, doi:10.1073/pnas.1116437108. http://www.ncbi.nlm.nih.gov/pubmed/22106295 (Accessed May 30, 2018).

Tim Chamen, W. C., A. P. Moxey, W. Towers, B. Balana, and P. D. Hallett, 2015: Mitigating arable soil compaction: A review and analysis of available cost and benefit data. *Soil Tillage Res.*, **146**, 10–25, doi:10.1016/J.STILL.2014.09.011. https://www.sciencedirect.com/science/article/pii/S0167198714001901 (Accessed May 31,

2018).

Timmermann, C., and Z. Robaey, 2016: Agrobiodiversity Under Different Property Regimes. *J. Agric. Environ. Ethics*, **29**, 285–303, doi:10.1007/s10806-016-9602-2. http://link.springer.com/10.1007/s10806-016-9602-2 (Accessed June 1, 2018).

Tirado, M. C., R. Clarke, L. A. Jaykus, A. McQuatters-Gollop, and J. M. Frank, 2010: Climate change and food safety: A review. *Clim. Chang. Food Sci.*, doi:10.1016/j.foodres.2010.07.003.

Townsend, P. V., R. J. Harper, P. D. Brennan, C. Dean, S. Wu, K. R. J. Smettem, and S. E. Cook, 2012: Multiple environmental services as an opportunity for watershed restoration. *For. Policy Econ.*, **17**, 45–58, doi:10.1016/J.FORPOL.2011.06.008. https://www.sciencedirect.com/science/article/pii/S1389934111000888 (Accessed June 2, 2018).

Trabucco, A., R. J. Zomer, D. A. Bossio, O. van Straaten, and L. V. Verchot, 2008: Climate change mitigation through afforestation/reforestation: A global analysis of hydrologic impacts with four case studies. *Agric. Ecosyst. Environ.*, doi:10.1016/j.agee.2008.01.015.

Triyanti, A., and M. A. Marfai, 2017: Social capital, interactive governance and coastal protection: The effectiveness of mangrove ecosystem-based strategies in promoting inclusive development in Demak, Indonesia. *Ocean Coast. Manag.*, **150**, 3–11, doi:10.1016/J.OCECOAMAN.2017.10.017. https://www.sciencedirect.com/science/article/pii/S0964569117308542 (Accessed June 2, 2018).

Troy, A., J. Morgan Grove, and J. O'Neil-Dunne, 2012: The relationship between tree canopy and crime rates across an urban–rural gradient in the greater Baltimore region. *Landsc. Urban Plan.*, **106**, 262–270, doi:10.1016/J.LANDURBPLAN.2012.03.010. https://www.sciencedirect.com/science/article/pii/S0169204612000977 (Accessed June 3, 2018).

Tscharntke, T., A. M. Klein, A. Kruess, I. Steffan-Dewenter, and C. Thies, 2005: Landscape perspectives on agricultural intensification and biodiversity - Ecosystem service management. *Ecol. Lett.*, doi:10.1111/j.1461-0248.2005.00782.x.

Tullberg, J., D. L. Antille, C. Bluett, J. Eberhard, and C. Scheer, 2018: Controlled traffic farming effects on soil emissions of nitrous oxide and methane. *Soil Tillage Res.*, **176**, 18–25, doi:10.1016/J.STILL.2017.09.014. https://www.sciencedirect.com/science/article/pii/S0167198717301812 (Accessed May 31, 2018).

Turner, K. G., and Coauthors, 2016: A review of methods, data, and models to assess changes in the value of ecosystem services from land degradation and restoration. *Ecol. Modell.*, **319**, 190–207, doi:10.1016/J.ECOLMODEL.2015.07.017. https://www.sciencedirect.com/science/article/pii/S0304380015003233?via%3Dihub (Accessed June 2, 2018).

UN-WWAP, 2015: *The United Nations World Water Development Report 2015: Water for a Sustainable World*. 1-139 pp.

UNCTAD, 2011: Water for Food – Innovative Water Management Technologies for Food Security and Poverty Alleviation. 39. http://unctad.org/en/docs/dtlstict2011d2_en.pdf.

UNEP, 2017: *The Emissions Gap Report 2017*. United Nations Environment Programme (UNEP), Nairobi,.

Valin, H., and Coauthors, 2014: The future of food demand: understanding differences in global economic models. *Agric. Econ.*, doi:10.1111/agec.12089.

VandenBygaart, A. J., 2016: The myth that no-till can mitigate global climate change. *Agric. Ecosyst. Environ.*, **216**, 98–99, doi:10.1016/J.AGEE.2015.09.013. https://www.sciencedirect.com/science/article/pii/S0167880915300797 (Accessed June 5, 2018).

Vasconcelos, A. C. F., M. Bonatti, S. L. Schlindwein, L. R. D'Agostini, L. R. Homem, and R. Nelson, 2013: Landraces as an adaptation strategy to climate change for smallholders in Santa Catarina, Southern Brazil. *Land use policy*, **34**, 250–254, doi:10.1016/J.LANDUSEPOL.2013.03.017.

https://www.sciencedirect.com/science/article/pii/S026483771300063X (Accessed June 1, 2018).

Vatsala, L., J. Prakash, and S. Prabhavathi, 2017: Food security and nutritional status of women selected from a rural area in South India. *J. Food, Nutr. Popul. Heal.*, **1**, 10.

Veldman, J. W., and Coauthors, 2015: Where Tree Planting and Forest Expansion are Bad for Biodiversity and Ecosystem Services. *Bioscience*, **65**, 1011–1018, doi:10.1093/biosci/biv118.

Vellakkal, S., J. Fledderjohann, S. Basu, S. Agrawal, S. Ebrahim, O. Campbell, P. Doyle, and D. Stuckler, 2015: Food Price Spikes Are Associated with Increased Malnutrition among Children in Andhra Pradesh, India. *J. Nutr.*, **145**, 1942–1949, doi:10.3945/jn.115.211250. https://academic.oup.com/jn/article/145/8/1942/4585797 (Accessed June 1, 2018).

Verburg, R., E. Stehfest, G. Woltjer, and B. Eickhout, 2009: The effect of agricultural trade liberalisation on land-use related greenhouse gas emissions. *Glob. Environ. Chang. Policy Dimens.*, doi:DOI 10.1016/j.gloenvcha.2009.06.004.

Vermeulen, S., and L. Cotula, 2010: Over the heads of local people: consultation, consent, and recompense in large-scale land deals for biofuels projects in Africa. *J. Peasant Stud.*, **37**, 899–916, doi:10.1080/03066150.2010.512463. https://www.tandfonline.com/doi/full/10.1080/03066150.2010.512463 (Accessed May 31, 2018).

Vermeulen, S. J., B. M. Campbell, and J. S. I. Ingram, 2012: Climate Change and Food Systems. *Annu. Rev. Environ. Resour.*, doi:10.1146/annurev-environ-020411-130608.

Vignola, R., B. Locatelli, C. Martinez, and P. Imbach, 2009: Ecosystem-based adaptation to climate change: what role for policy-makers, society and scientists? *Mitig. Adapt. Strateg. Glob. Chang.*, **14**, 691–696, doi:10.1007/s11027-009-9193-6. http://link.springer.com/10.1007/s11027-009-9193-6 (Accessed June 3, 2018).

——, T. L. McDaniels, and R. W. Scholz, 2013: Governance structures for ecosystem-based adaptation: Using policy-network analysis to identify key organizations for bridging information across scales and policy areas. *Environ. Sci. Policy*, doi:10.1016/j.envsci.2013.03.004.

——, C. A. Harvey, P. Bautista-Solis, J. Avelino, B. Rapidel, C. Donatti, and R. Martinez, 2015: Ecosystem-based adaptation for smallholder farmers: Definitions, opportunities and constraints. *Agric. Ecosyst. Environ.*, **211**, 126–132, doi:10.1016/J.AGEE.2015.05.013. https://www.sciencedirect.com/science/article/pii/S0167880915002157 (Accessed June 1, 2018).

Villar-Salvador, P., R. Planelles, E. Enríquez, and J. P. Rubira, 2004: Nursery cultivation regimes, plant functional attributes, and field performance relationships in the Mediterranean oak Quercus ilex L. *For. Ecol. Manage.*, **196**, 257–266, doi:10.1016/j.foreco.2004.02.061.

Vincent, K., 2007: Uncertainty in adaptive capacity and the importance of scale. *Glob. Environ. Chang.*, doi:10.1016/j.gloenvcha.2006.11.009.

Vlontzos, G., S. Niavis, and B. Manos, 2014: A DEA approach for estimating the agricultural energy and environmental efficiency of EU countries. *Renew. Sustain. Energy Rev.*, **40**, 91–96, doi:10.1016/J.RSER.2014.07.153. https://www.sciencedirect.com/science/article/pii/S1364032114006054 (Accessed June 1, 2018).

Vogel, C., and K. O'Brien, 2006: Who can eat information? Examining the effectiveness of seasonal climate forecasts and regional climate-risk management strategies. *Clim. Res.*, doi:10.3354/cr033111.

Vogt, J. V., U. Safriel, G. Von Maltitz, Y. Sokona, R. Zougmore, G. Bastin, and J. Hill, 2011: Monitoring and assessment of land degradation and desertification: Towards new conceptual and integrated approaches. *L. Degrad. Dev.*, **22**, 150–165, doi:10.1002/ldr.1075. http://doi.wiley.com/10.1002/ldr.1075 (Accessed June 2, 2018).

van Vuuren, D. P., and Coauthors, 2011: RCP2.6: exploring the possibility to keep global mean temperature increase below 2 degrees C. *Clim. Change*, doi:10.1007/s10584-011-0152-3.

van Vuuren, D. P., and Coauthors, 2015: Pathways to achieve a set of ambitious global sustainability objectives by 2050: Explorations using the IMAGE integrated assessment model. *Technol. Forecast. Soc. Change*, doi:10.1016/j.techfore.2015.03.005.

van Vuuren, D. P., H. van Soest, K. Riahi, and E. Al., 2016: Carbon budgets and energy transition pathways. *Environ. Res. Lett.*, doi:10.1088/1748-9326/11/7/075002.

van Vuuren, D. P., and Coauthors, 2017: Energy, land-use and greenhouse gas emissions trajectories under a green growth paradigm. *Glob. Environ. Chang.*, doi:10.1016/j.gloenvcha.2016.05.008.

Van Vuuren, D. P., E. Bellevrat, A. Kitous, and M. Isaac, 2010: Bio-energy use and low stabilization scenarios. *Energy J.*, doi:10.5547/ISSN0195-6574-EJ-Vol31-NoSI-8.

——, A. F. Hof, M. A. E. Van Sluisveld, and K. Riahi, 2017: Open discussion of negative emissions is urgently needed. *Nat. Energy*, **2**, 902–904, doi:10.1038/s41560-017-0055-2. http://dx.doi.org/10.1038/s41560-017-0055-2.

Waha, K., M. T. van Wijk, S. Fritz, L. See, P. K. Thornton, J. Wichern, and M. Herrero, 2018: Agricultural diversification as an important strategy for achieving food security in Africa. *Glob. Chang. Biol.*, doi:10.1111/gcb.14158. http://doi.wiley.com/10.1111/gcb.14158 (Accessed May 31, 2018).

Wang, M., and Coauthors, 2017: On the long-term hydroclimatic sustainability of perennial bioenergy crop expansion over the United States. *J. Clim.*, doi:10.1175/JCLI-D-16-0610.1.

Wang, Y., X. Yan, and Z. Wang, 2016: A preliminary study to investigate the biogeophysical impact of desertification on climate based on different latitudinal bands. *Int. J. Climatol.*, **36**, 945–955, doi:10.1002/joc.4396. http://doi.wiley.com/10.1002/joc.4396 (Accessed June 3, 2018).

Webb, N. P., N. A. Marshall, L. C. Stringer, M. S. Reed, A. Chappell, and J. E. Herrick, 2017: Land degradation and climate change: building climate resilience in agriculture. *Front. Ecol. Environ.*, doi:10.1002/fee.1530.

Weindl, I., H. Lotze-Campen, A. Popp, C. Müller, P. Havlík, M. Herrero, C. Schmitz, and S. Rolinski, 2015a: Livestock in a changing climate: production system transitions as an adaptation strategy for agriculture Related content Reducing greenhouse gas emissions in agriculture without compromising food security? Implications of climate mitigation for future agr. *Environ. Res. Lett.*, **10**. http://iopscience.iop.org/article/10.1088/1748-9326/10/9/094021/pdf.

——, ——, ——, ——, ——, ——, ——, and ——, 2015b: Livestock in a changing climate: production system transitions as an adaptation strategy for agriculture Related content Reducing greenhouse gas emissions in agriculture without compromising food security? Implications of climate mitigation for future agr. *Environ. Res. Lett.*, **10**.

Welfle, A., P. Gilbert, and P. Thornley, 2014: Securing a bioenergy future without imports. *Energy Policy*, **68**, 1–14, doi:10.1016/j.enpol.2013.11.079.

Werling, B. P., T. D. Meehan, B. A. Robertson, C. Gratton, and D. A. Landis, 2011: Biocontrol potential varies with changes in biofuel-crop plant communities and landscape perenniality. *GCB Bioenergy*, doi:10.1111/j.1757-1707.2011.01092.x.

Werling, B. P., and Coauthors, 2014: Perennial grasslands enhance biodiversity and multiple ecosystem services in bioenergy landscapes. *Proc. Natl. Acad. Sci.*, doi:10.1073/pnas.1309492111.

van Wesenbeeck, B. K., J. P. M. Mulder, M. Marchand, D. J. Reed, M. B. de Vries, H. J. de Vriend, and P. M. J. Herman, 2014: Damming deltas: A practice of the past? Towards nature-based flood defenses. *Estuar. Coast. Shelf Sci.*, **140**, 1–6, doi:10.1016/J.ECSS.2013.12.031. https://www.sciencedirect.com/science/article/pii/S0272771413005556 (Accessed June 1, 2018).

West, J. J., and Coauthors, 2013: Co-benefits of mitigating global greenhouse gas emissions for future air quality and human health. *Nat. Clim. Chang.*, **3**, 885–889, doi:10.1038/nclimate2009. http://www.nature.com/articles/nclimate2009 (Accessed June 1, 2018).

Westerling, A., H. Hidalgo, D. Cayan, and T. Swetnam, 2006: Warming and Earlier Spring Increase Western U.S. Forest Wildfire Activity. *Science*, **313**, 940–943, doi:10.1126/science.262.5135.885. http://www.ncbi.nlm.nih.gov/pubmed/17757357 (Accessed May 31, 2018).

Wheeler, T., and J. von Braun, 2013: Climate Change Impacts on Global Food Security. *Science (80-.).*, doi:10.1126/science.1239402.

White, A., and A. Martin, 2002: *Who owns the world's forests*. Forest Trends, Washington D.C., 30 pp. pp. http://www.era-mx.org/biblio/politica/ForestTenure.pdf.

Whitehead, D., 2011: Forests as carbon sinks--benefits and consequences. *Tree Physiol.*, **31**, 893–902, doi:10.1093/treephys/tpr063. https://academic.oup.com/treephys/article-lookup/doi/10.1093/treephys/tpr063 (Accessed May 31, 2018).

Whitehead, P. J., P. Purdon, J. Russell-Smith, P. M. Cooke, and S. Sutton, 2008: The management of climate change through prescribed Savanna burning: Emerging contributions of indigenous people in Northern Australia. *Public Adm. Dev.*, **28**, 374–385, doi:10.1002/pad.512. http://doi.wiley.com/10.1002/pad.512 (Accessed May 31, 2018).

Wiebe, K., and Coauthors, 2015: Climate change impacts on agriculture in 2050 under a range of plausible socioeconomic and emissions scenarios. *Environ. Res. Lett.*, **10**, doi:10.1088/1748-9326/10/8/085010.

Wilhelm, M., C. Blome, and V. Bhakoo, 2016a: Sustainability in multi-tier supply chains: Understanding the double agency role of the first-tier supplier. *Elsevier*,. https://www.sciencedirect.com/science/article/pii/S0272696315300115 (Accessed May 1, 2018).

——, ——, ——, and A. Paulraj, 2016b: Sustainability in multi-tier supply chains: Understanding the double agency role of the first-tier supplier. https://www.sciencedirect.com/science/article/pii/S0272696315300115 (Accessed May 1, 2018).

Williams, S. B., L. L. Murdock, and D. Baributsa, 2017: Sorghum seed storage in Purdue Improved Crop Storage (PICS) bags and improvised containers. *J. Stored Prod. Res.*, doi:10.1016/j.jspr.2017.04.004.

Winsemius, H. C., and Coauthors, 2016: Global drivers of future river flood risk. *Nat. Clim. Chang.*, doi:10.1038/nclimate2893.

Wise, M., and Coauthors, 2009: Implications of Limiting CO2 Concentrations for Land Use and Energy. *Science (80-.).*, doi:10.1126/science.1168475.

Wise, R. M., J. R. A. Butler, W. Suadnya, K. Puspadi, I. Suharto, and T. D. Skewes, 2016: How climate compatible are livelihood adaptation strategies and development programs in rural Indonesia? *Clim. Risk Manag.*, **12**, 100–114, doi:10.1016/J.CRM.2015.11.001. https://www.sciencedirect.com/science/article/pii/S2212096315000352 (Accessed June 1, 2018).

Wodon, Q., and H. Zaman, 2010: Higher Food Prices in Sub-Saharan Africa: Poverty Impact and Policy Responses. *World Bank Res. Obs.*, **25**, 157–176, doi:10.1093/wbro/lkp018. https://academic.oup.com/wbro/article-lookup/doi/10.1093/wbro/lkp018 (Accessed June 1, 2018).

Wollenberg, E., and Coauthors, 2016: Reducing emissions from agriculture to meet the 2 °C target. *Glob. Chang. Biol.*, **22**, 3859–3864, doi:10.1111/gcb.13340. http://doi.wiley.com/10.1111/gcb.13340 (Accessed May 30, 2018).

Woolf, D., J. E. Amonette, F. A. Street-Perrott, J. Lehmann, and S. Joseph, 2010: Sustainable biochar to mitigate global climate change. *Nat. Commun.*, **1**, 1–9, doi:10.1038/ncomms1053. http://www.nature.com/doifinder/10.1038/ncomms1053 (Accessed May 30, 2018).

Xia, J., Q.-Y. Duan, Y. Luo, Z.-H. Xie, Z.-Y. Liu, and X.-G. Mo, 2017: Climate change and water resources: Case study of Eastern Monsoon Region of China. *Adv. Clim. Chang. Res.*, doi:10.1016/j.accre.2017.03.007.

Yang, H. F., S. L. Yang, K. H. Xu, H. Wu, B. W. Shi, Q. Zhu, W. X. Zhang, and Z. Yang, 2017: Erosion potential of the Yangtze Delta under sediment starvation and climate change. *Sci. Rep.*, doi:10.1038/s41598-017-10958-y.

Yang, X., and Coauthors, 2015: Groundwater sapping as the cause of irreversible desertification of Hunshandake Sandy Lands, Inner Mongolia, northern China. *Proc. Natl. Acad. Sci.*, doi:10.1073/pnas.1418090112.

Yirdaw, E., M. Tigabu, and A. Monge, 2017: Rehabilitation of degraded dryland ecosystems – review. *Silva Fenn.*, doi:10.14214/sf.1673.

Yost, M. A., N. R. Kitchen, K. A. Sudduth, E. J. Sadler, S. T. Drummond, and M. R. Volkmann, 2017: Long-term impact of a precision agriculture system on grain crop production. *Precis. Agric.*, **18**, 823–842, doi:10.1007/s11119-016-9490-5. http://link.springer.com/10.1007/s11119-016-9490-5 (Accessed June 3, 2018).

Yousefpour, R., A. L. D. Augustynczik, C. P. O. Reyer, P. Lasch-Born, F. Suckow, and M. Hanewinkel, 2018: Realizing Mitigation Efficiency of European Commercial Forests by Climate Smart Forestry. *Sci. Rep.*, **8**, 345, doi:10.1038/s41598-017-18778-w.

Yu, H., B. Li, D. Shen, J. Cao, and B. Mao, 2017: Study on prediction model of grain post-harvest loss. *Procedia Computer Science*, Vol. 122 of, 122–129 https://www.sciencedirect.com/science/article/pii/S1877050917325796 (Accessed May 10, 2018).

Zedler, J. B., and S. Kercher, 2005: WETLAND RESOURCES: Status, Trends, Ecosystem Services, and Restorability. *Annu. Rev. Environ. Resour.*, **30**, 39–74, doi:10.1146/annurev.energy.30.050504.144248. http://www.annualreviews.org/doi/10.1146/annurev.energy.30.050504.144248 (Accessed June 7, 2018).

Zektser, I. S., and L. G. Everett, 2004: *Groundwater resources of the world and their use*. 1-346 pp.

Zhang, M., and Coauthors, 2014: Response of surface air temperature to small-scale land clearing across latitudes. *Environ. Res. Lett.*, doi:10.1088/1748-9326/9/3/034002.

Zhang, X., R. Obringer, C. Wei, N. Chen, and D. Niyogi, 2017: Droughts in India from 1981 to 2013 and Implications to Wheat Production. *Sci. Rep.*, **7**, 44552, doi:10.1038/srep44552. http://www.nature.com/articles/srep44552 (Accessed June 2, 2018).

Zhang, Z., and D. Huisingh, 2018: Combating desertification in China: Monitoring, control, management and revegetation. *J. Clean. Prod.*, **182**, 765–775, doi:10.1016/J.JCLEPRO.2018.01.233. https://www.sciencedirect.com/science/article/pii/S0959652618302646 (Accessed June 3, 2018).

Zhao, K., and R. B. Jackson, 2014: Biophysical forcings of land-use changes from potential forestry activities in North America. *Ecol. Monogr.*, doi:10.1890/12-1705.1.

Zhuang, Q., Z. Qin, and M. Chen, 2013: Biofuel, land and water: Maize, switchgrass or Miscanthus? *Environ. Res. Lett.*, doi:10.1088/1748-9326/8/1/015020.