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Working Group III – Mitigation of Climate Change

Chapter 11

Agriculture, Forestry and Other Land Use (AFOLU)

Chapter:	11	
Title:	Agriculture, Forestry and Other Land Use (AFOLU)	
Author(s):	CLAs:	Pete Smith, Mercedes Bustamante
	LAs:	Helal Ahammad, Harry Clark, Hongmin Dong, Elnour A. Elsiddig, Helmut Haberl, Richard Harper, Joanna House, Mostafa Jafari, Omar Masera, Cheikh Mbow, Nijavalli H. Ravindranath, Charles W. Rice, Carmenza Robledo Abad, Anna Romanovskaya, Frank Sperling, Francesco Tubiello
	CAs:	Göran Berndes, Simon Bolwig, Hannes Böttcher, Ryan Bright, Francesco Cherubini, Helena Chum, Esteve Corbera, Felix Creutzig, Mark Delucchi, Andre Faaij, Joe Fargione, Gesine Hänsel, Garvin Heath, Mario Herrero, Heather Jacobs, Oswaldo Lucon, Daniel Pauly, Richard Plevin, Alexander Popp, John R. Porter, Steven Rose, Alexandre de Siqueira Pinto, Saran Sohi, Anders Stromman, Sangwon Suh
	REs:	Thelma Krug, Gert-Jan Nabuurs
	CSA:	Marina Molodovskaya

Chapter 11: Agriculture, Forestry and Other Land Use (AFOLU)

Contents

Executive Summary

Agriculture, Forestry and Other Land Use (AFOLU) is unique among the sectors considered in this volume, since the mitigation potential is derived from both an enhancement of removals of GHGs, as well as reduction of emissions through management of land and livestock [*11.1, robust evidence; high agreement*]. The land provides food which feeds the Earth's human population of ca. 7 billion, fibre for a variety of purposes, livelihoods for billions of people worldwide, and is a critical resource for sustainable development in many regions. Agriculture is frequently central to the livelihoods of many social groups, especially in developing countries where it often accounts for a significant share of production. In addition to food and fibre, the land provides a multitude of ecosystem services; greenhouse gas mitigation is just one of many that are vital to human wellbeing [*11.1, robust evidence; high agreement*]. Mitigation options in the AFOLU sector, therefore, need to be assessed, as far as possible, for their potential impact on all other services provided by land.

The AFOLU sector is responsible for just under a quarter (~9-12 Gt CO_2eq/yr) of anthropogenic GHG emissions mainly from deforestation and agricultural emissions from livestock, soil and nutrient management [*11.2, medium evidence; high agreement*]. Anthropogenic forest degradation and biomass burning (forest fires and agricultural burning) also represent relevant contributions. Annual GHG emissions from agricultural production in 2000-2010 were estimated at 5.0-5.8 Gt CO_2eq/yr while annual GHG flux from land use and land use change activities accounted for approximately 4.3-5.5 Gt CO_2eq/yr. Leveraging the mitigation potential in the sector is extremely important in meeting emission reduction targets [*11.9, robust evidence; high agreement*]. Since publication of the AR4, emissions from the AFOLU sector have remained similar but the share of anthropogenic emissions has decreased to 24% (in 2010), largely due to increases in emissions in the energy sector [*robust evidence, high agreement*]. In spite of a large range across global FOLU flux estimates, most approaches indicate a decline in FOLU CO_2 emissions over the most recent years, largely due to decreasing deforestation rates. As in AR4, most projections suggest declining annual net CO_2 emissions in the long run. In part, this is driven by technological change, as well as projected declining rates of agriculture area expansion, which, in turn, is related to the expected slowing in population growth. However, unlike AR4, none of the more recent scenarios projects growth in the near-term [11.9].

Opportunities for mitigation include supply-side and demand-side options. On the supply side, emissions from land use change, land management and livestock management can be reduced, terrestrial carbon stocks can be increased by sequestration in soils and biomass, and emissions from energy production can be saved through the substitution of fossil fuels by biomass [*11.3, robust evidence; high agreement*]. On the demand side, GHG emissions could be mitigated by reducing losses and wastes of food, changes in diet and changes in wood consumption [*11.4, robust evidence; high agreement*] though quantitive estimates of the potential are few and highly uncertain. Increasing production without a commensurate increase in emissions also reduces emission intensity, i.e. the GHG emissions per unit of product which could be delivered through sustainable intensification; another mechanism for mitigation explored in more detail here than in AR4. Supply-side options depend on the efficacy of land and livestock management [*11.6, medium evidence; high agreement*]. Considering demand-side options, changes in human diet can have a significant impact on GHG emissions from the food production life cycle [*11.4, medium evidence; medium agreement*]. There are considerably different challenges involved in delivering demand-side and supply-side options, which also have very different synergies and risk-tradeoffs.

The nature of the sector means that there are potentially many barriers to implementation of available mitigation options, including accessibility to AFOLU financing, poverty, institutional, ecological, technological development, diffusion and transfer barriers [*11.7, 11.8, medium evidence; medium agreement*]. Similarly, there are important feedbacks to adaptation, conservation of natural resources, such as water and terrestrial and aquatic biodiversity [11.5, 11.8, *robust evidence; high agreement*]. There can be competition between different land-uses if alternative options to use available land are mutually exclusive, but there are also potential synergies, e.g. integrated systems or multi-functionality at landscape scale [*11.4, medium evidence; high agreement*]. Recent frameworks, such as those for assessing environmental or ecosystem services, provide one mechanism for valuing the multiple synergies and trade-offs that may arise from mitigation actions [*11.1, medium evidence; medium agreement*]. Sustainable management of agriculture, forests, and other land is an underpinning requirement of sustainable development [*11.4, robust evidence; high agreement*].

AFOLU forms a significant component of mitigation in transformation pathways, offering a variety of mitigation options and a large, cost-competitive mitigation potential [*limited evidence; medium agreement*]. Recent multi-model comparisons have found that all land-related mitigation strategies (agriculture, forestry, bioenergy) were projected to contribute 20 to 60% of total cumulative abatement to 2030, and still 15 to 45% to 2100 [11.9]. Large-scale energy generation or carbon sequestration in the AFOLU sector provides flexibility for the development of mitigation technologies in the energy supply and energy end-use sectors, as many technologies already exist and some of them are commercial [11.3, *limited evidence; medium agreement*], but there are potential implications for biodiversity, food security and other services provided by land [11.7 *medium evidence, high agreement*]. Implementation challenges, including institutional barriers and inertia related to governance issues, make the costs and net emission reduction potential of near-term mitigation uncertain. In climate management scenarios with idealized comprehensive climate policies, agriculture, forestry and bioenergy contribute substantially to mitigation of global CO_2, CH_4, and N_2O, and to the energy system, thereby reducing policy costs [*11.9, medium evidence; high agreement*]. More realistic partial and delayed policies for global land mitigation have potentially significant spatial and temporal leakage, and economic implications, but could still be cost-effectively deployed [*11.9, limited evidence; limited agreement*].

Economic mitigation potential of supply-side measures in the AFOLU sector is estimated to be 7.18 to 10.60 (full range: 0.49-13.78) $GtCO_2eq$/yr at carbon prices up to 100 US$/ tCO_2eq, about a third of which can be achieved at <20 US$/ tCO_2eq [*11.6, medium evidence; medium agreement*]. These estimates are based on studies that cover both forestry and agriculture and that include agricultural soil carbon sequestration. Estimates from agricultural sector-only studies range from 0.26 to 4.6 Gt CO_2eq/yr at prices up to 100 USD/t CO_2eq, and estimates from forestry sector-only studies from 0.2 to 13.8 Gt CO_2eq/yr at prices up to 100 USD/t CO_2eq [*11.6, medium evidence; medium agreement*]. The large range in the estimates arises due to widely-different collections of options considered in each study, and because not all GHGs are considered in all of the studies. The composition of the agricultural mitigation portfolio varies with the carbon price, with the restoration of organic soils having the greatest potential at higher (100 USD/t CO_2eq) and cropland and grazing land management at lower (20 USD/t CO_2eq) carbon prices. In forestry there is less difference between measures at different carbon prices, but there are significant differences between regions, with reduced deforestation dominating the forestry mitigation potential LAM and MAF, but very little potential in OECD90 and REF. Forest management, followed by afforestation, dominate in OECD90, REF and Asia [*11.6, medium evidence, strong agreement*]. Among demand-side measures, which are under-researched compared to supply-side measures, changes in diet and reductions of losses in the food supply chain can have a significant impact on GHG emissions from food production (0.76-9.31 Gt CO_2eq/yr by 2050), with the range being determined by assumptions about how the freed land is used [*11.4, limited evidence; medium agreement*]. More research into demand-side mitigation options is merited. There are significant regional differences in terms of mitigation potential, costs and applicability, due to differing local biophysical, socioeconomic and cultural circumstances, for instance between developed and developing regions, and among developing regions [*11.6, medium evidence; high agreement*].

The size and regional distribution of future mitigation potential is difficult to estimate accurately as it depends on a number of inherently uncertain factors. Critical factors include population (growth), economic and technological developments, changes in behaviour over time (depending on cultural and normative backgrounds, market structures and incentives), and how these translate into demand for food, fibre, fodder and fuel, as well as development in the agriculture, aquaculture and forestry sectors. Other factors important to mitigation potential are: potential climate change impacts on carbon stocks in soils and forests including their adaptive capacity [*11.5, medium evidence; high agreement*]; considerations set by biodiversity and nature conservation requirements; and interrelations with land degradation and water scarcity [*11.8, robust evidence; high agreement*].

Land use change associated with bioenergy expansion, afforestation or deforestation can affect GHG balances, albedo and other climate drivers in several ways. Bioenergy can be deployed as solid, liquid and gaseous fuels to provide transport, electricity, and heat for a wide range of uses, including cooking, and depending on how and where implemented, can lead to either beneficial or undesirable consequences for climate change mitigation [*11.13, robust evidence, high agreement*]. With limited availability of productive land, increased competition for land may result from large deployment of dedicated energy crops, which may induce substantial land use change (LUC), causing high GHG emissions and/or agricultural intensification, which could result in more fertilizer use (leading to higher N_2O emissions), and energy use for irrigation [*11.9, medium evidence; limited agreement*]. However, societal preferences and technological changes also shape the LUC and intensification outcomes. AFOLU mitigation options can promote innovation, and many technological supply-side mitigation options also increase agricultural and silvicultural efficiency [*11.3, robust evidence; high agreement*].

Bioenergy could play a critical role in stabilizing climate change, if conversion of high carbon density ecosystems (forests, grasslands and peat-lands) is avoided and best-practice land management is implemented [*robust evidence, medium agreement*] (see 11.13). Integrated assessments suggest a wide range of between 10 and 245 EJ/yr primary energy from biomass by 2050. Bioenergy from fast-growing tree species, sugarcane, and Miscanthus, and residues have significantly lower life-cycle emissions than bioenergy from corn and soybean, for most pathways and site-specific conditions [11.13, *robust evidence, medium agreement*]. Scientific debate about the marginal emissions of most bioenergy pathways, in particular around land-mediated equilibrium effects (such as indirect land use change), remains unresolved [11.13, *robust evidence, high agreement*]. BECCS may be critical to scenarios for stabilization at <2◦C; however, the potential and costs of BECCS are highly uncertain with some integrated assessment models being more optimistic than bottom-up studies. Biomass for energy, in combination with improved cookstoves, biogas and small-scale biopower could reduce marginal GHG emissions and also improve livelihoods and health of 2.7 billion rural inhabitants. But if policy conditions (e.g. price on both fossil and terrestrial carbon; land-use planning, and others) are not met, bioenergy deployment could also lead to increased emissions, and compromise livelihoods (distributional consequences), biodiversity and ecosystem services [11.13, *medium evidence, medium agreement*].

Any large-scale change in land use, for biomass for energy, or for sequestration in vegetation, will likely increase the competition for land, water, and other resources, and conflicts may arise with important sustainability objectives such as food security, soil and water conservation, and the protection of terrestrial and aquatic biodiversity [*11.4, 11.7, 11.13, medium evidence; medium agreement*]. In some cases land-based mitigation projects may provide land, water and biodiversity co-benefits [*11.7, medium evidence; medium* agreement]. Sustainability frameworks to guide development of such mitigation projects need to consider competition for land [*11.8, medium evidence; limited agreement*]. Risks could be reduced by focussing on multifunctional systems that allow the delivery of multiple services from land [*11.7, medium evidence; high agreement*].

Policies governing practices in agriculture and in forest conservation and management need to account for both mitigation and adaptation. One of the most visible current policies in the AFOLU sector is the implementation of REDD+, that can represent a cost-effective option for mitigation [*11.10, medium evidence; high agreement*], with economic, social and other environmental co-benefits (e.g. conservation of biodiversity and water resources).

11.1 Introduction

Agriculture, Forestry and Other Land Use (AFOLU[1]) plays a central role for food security and sustainable development (11.9). Plants take up carbon dioxide (CO_2) from the atmosphere and N from the soil when they grow, re-distributing it among different pools, including above and below-ground living biomass, dead residues, and soil organic matter. CO_2 and other non-CO_2 GHG gases, largely methane (CH_4) and nitrous oxide (N_2O), are in turn released to the atmosphere by plant respiration, by decomposition of dead plant biomass and soil organic matter, and by combustion (11.2). Anthropogenic land use activities (e.g. management of croplands, forests, grasslands, wetlands), and changes in land use/cover (e.g. conversion of forest lands and grasslands to cropland and pasture, afforestation) cause changes superimposed on these natural fluxes. AFOLU activities lead to both sources of CO_2 (e.g. deforestation, peatland drainage) and sinks of CO_2 (e.g. afforestation, management for soil carbon sequestration), and to non-CO_2 emissions primarily from agriculture (e.g. CH_4 from livestock and rice cultivation, N_2O from manure storage and agricultural soils and biomass burning (11.2).

The main mitigation options within AFOLU involve one or more of three strategies: *reduction/prevention* of emissions to the atmosphere by conserving existing carbon pools in soils or vegetation that would otherwise be lost or by reducing emissions of CH_4 and N_2O (11.3); *sequestration* – enhancing the uptake of carbon in terrestrial reservoirs, and thereby removing carbon dioxide from the atmosphere (11.3); and reducing carbon dioxide emissions by *substitution* of biological products for fossil fuels (Appendix 1) or energy-intensive products (11.4). Demand-side options (e.g. by lifestyle changes, reducing losses and wastes of food, changes in human diet, changes in wood consumption), though known to be difficult to implement, may also play a role (11.4).

Land is the critical resource for the AFOLU sector and it provides food and fodder to feed the Earth's population of ~7 billion, and fibre and fuel for a variety of purposes. It provides livelihoods for billions of people worldwide. It is finite and provides a multitude of goods and ecosystem services, fundamental to human well-being (MEA, 2005). Human economies and quality of life are directly dependent on the services and the resources provided by land. Figure 11.1 shows the many provisioning, regulating, cultural and supporting services provided by land, of which climate regulation is just one. Implementing mitigation options in the AFOLU sector may potentially affect other services provided by land in positive or negative ways.

In the IPCC SAR (IPCC, 1996) and in AR4 (IPCC, 2007), agricultural and forestry mitigation were dealt with in separate chapters. In the TAR (IPCC, 2001), there were no separate sectoral chapters on either agriculture or forestry. In AR5, for the first time, the vast majority of the terrestrial land surface, comprising agriculture, forestry and other land use AFOLU (IPCC, 2006), is considered together in a single chapter, though settlements (which are important, with urban areas forecasted to triple in size from 2000 global extent by 2030; 12.2), are dealt with in Chapter 12. This approach ensures that all land based mitigation options can be considered together; it minimises the risk of double counting or inconsistent treatment (e.g. different assumptions about available land) between

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[1] The term AFOLU used here consistent with the (IPCC, 2006) Guidelines is also consistent with LULUCF (IPCC, 2003), and other similar terms used in the scientific literature.

different land categories, and allows the consideration of systemic feedbacks between mitigation options related to the land surface (11.4). Considering AFOLU in a single chapter allows phenomena common across land use types, such as competition for land (Smith et al., 2010; Lambin and Meyfroidt, 2011) and water (e.g., Jackson et al., 2007), co-benefits (Sandor et al., 2002; Venter et al., 2009), risk-tradeoffs, uncertainty and spill-overs (11.7) and interactions between mitigation and adaptation (11.5) to be considered consistently. The complex nature of land presents a unique range of barriers and opportunities (11.8), and policies to promote mitigation in the AFOLU sector (11.10) need to take account of this complexity.

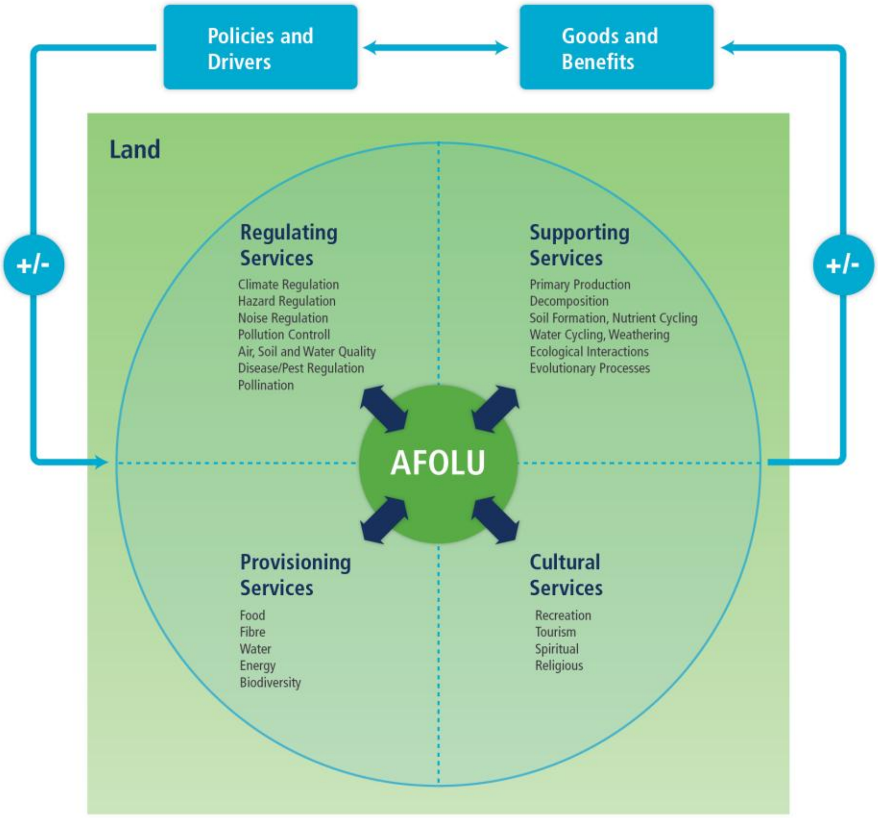

Figure 11.1. Multiple ecosystem services, goods and benefits provided by land (after (MEA, 2005; UNEP-WCMC, 2011). Mitigation actions aim to enhance climate regulation, but this is only one of the many functions fulfilled by land.

In this chapter, we consider the competing uses of land for mitigation and for providing other services (11.7; 11.8). Unlike the chapters on agriculture and forestry in AR4, impacts of sourcing bioenergy from the AFOLU sector are considered explicitly in a dedicated appendix (11.13). Also new to this assessment is the explicit consideration of food / dietary demand-side options for GHG mitigation in the AFOLU sector (11.4), and some consideration of freshwater fisheries and aquaculture, which may compete with the agriculture and forestry sectors, mainly through their requirements for land and/or water, and indirectly, by providing fish and other products to the same markets as animal husbandry.

We deal with AFOLU in an integrated way with respect to the underlying scenario projections of population growth, economic growth, dietary change, land use change and cost of mitigation. We draw evidence from both “bottom-up” studies that estimate mitigation potentials at small scales or for individual options or technologies and then scale up, and multi-sectoral “top-down” studies that consider AFOLU as just one component of a total multi-sector system response (11.9). In this chapter we provide updates on emissions trends and changes in drivers and pressures in the AFOLU sector (11.2), describe the practices available in the AFOLU sector (11.3), and we provide refined estimates of mitigation costs and potentials for the AFOLU sector, by synthesising studies that have become available since IPCC AR4 (11.6). We conclude the chapter by identifying gaps in knowledge and data (11.11), providing a selection of Frequently Asked Questions, and presenting an Appendix on bioenergy to update the IPCC Special Report on Renewables (SRREN; 11.13).

11.2 New developments in emission trends and drivers

Estimating and reporting the anthropogenic component of gross and net AFOLU GHG fluxes to the atmosphere, globally, regionally, and at country level, is difficult compared to other sectors. First, it is not always possible to separate anthropogenic and natural GHG fluxes from land. Second, the input data necessary to estimate GHG emissions globally and regionally, often based on country-level statistics or on remote sensing information, are very uncertain. Third, methods for estimating GHG emissions use a range of approaches, from simple default methodologies such as those specified in the IPCC GHG Guidelines[2] (IPCC, 2006), to more complex estimates based on terrestrial carbon cycle modelling and/or remote sensing information. Global trends in total GHG emissions from AFOLU activities between 1971 and 2010 are shown in figure 11.2; figure 11.3 shows trends of major drivers of emissions.

11.2.1 Supply and consumption trends in agriculture and forestry

In 2010 world agricultural land occupied 4889 Mha, an increase of 7% (311 Mha) since 1970 (FAOSTAT, 2013). Agricultural land area has decreased by 53 Mha since 2000 due to a decline of the cropland area (OECD90, REF) and a decrease in permanent meadows and pastures (OECD90 and Asia). The average amount of cropland and pasture land per-capita in 1970 was 0.4 and 0.8 ha and by 2010 this had decreased to 0.2 and 0.5 ha per capita, respectively (FAOSTAT, 2013).

Changing land-use practices, technological advancement and varietal improvement have enabled world grain harvests to double from 1.2 to 2.5 billion tonnes per year between 1970 and 2010 (FAOSTAT, 2012). Average world cereal yields increased from 1602 kg/ha to 3034 kg/ha over the same period (FAOSTAT, 2012) while there has also been a 233% increase in global fertilizer use from 31.8 to 105.9 Mt/yr, and a 73% increase in the irrigated cropland area (FAOSTAT, 2013).

Globally, since 1970, there has been a 1.4 fold increase in the numbers of cattle and buffalo, sheep and goats (which is closely linked to the trend of CH_4 emissions in the sector; section 11.2.2), and increases of 1.6 and 3.7 fold for pigs and poultry, respectively (FAOSTAT, 2013). Major regional trends between 1970 and 2010 include a decrease in the total number of animals in REF and OECD90 (except poultry), and continuous growth in other regions, particularly MAF and Asia (Figure 11.3b). The soaring demand for fish has led to the intensification of freshwater and marine fisheries worldwide, and an increased freshwater fisheries catch which topped 11 Mt in 2010, although the marine fisheries catch has slowly declined (78 Mt in 2010; FAOSTAT, 2013). The latter is, however, compensated in international markets by tremendous growth of aquaculture production to 60 Mt wet weight in 2010, of which 37 Mt originate from freshwater, overwhelmingly in Asia (FAOSTAT, 2013).

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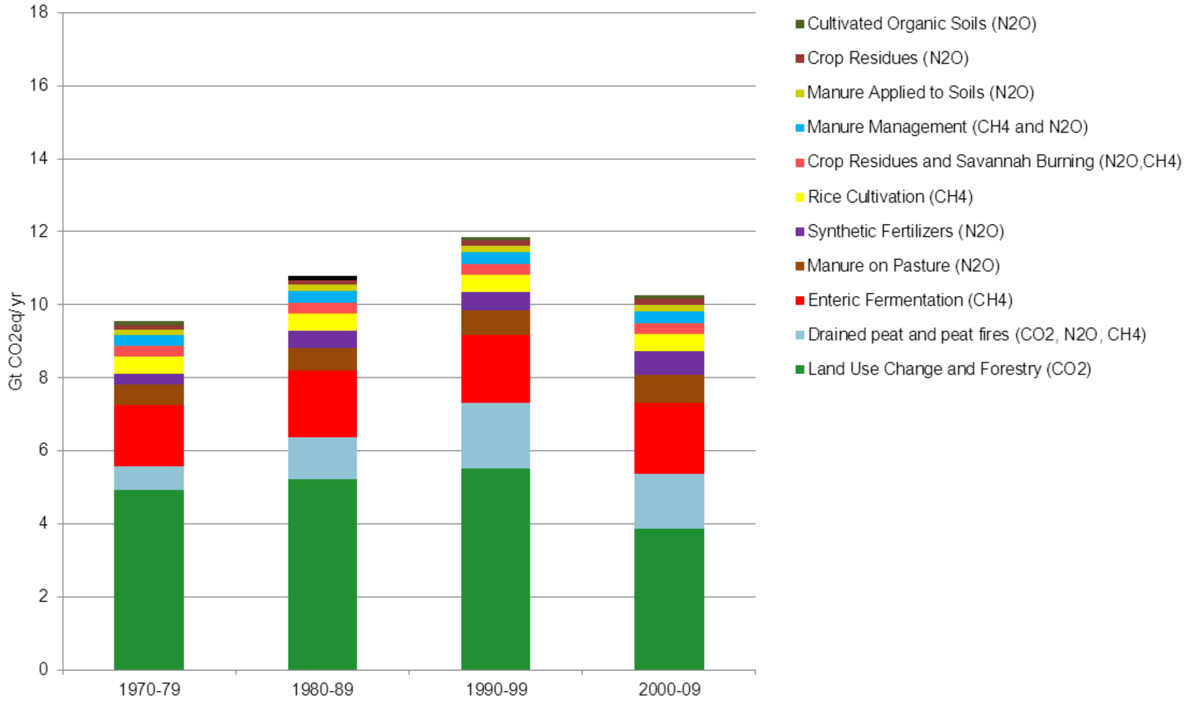
[2] Parties to the UNFCCC report net GHG emissions according to IPCC methodologies (IPCC, 2006). Reporting is based on a range of methods and approaches dependent on available data and national capacities, from default equations and emission factors applicable to global or regional cases and assuming instantaneous emissions of all carbon that will be eventually lost from the system following human action (tier 1) to more complex approaches such as model-based spatial analyses (tier 3).

Between 1970 and 2010, global daily per-capita food availability, expressed in energy units, has risen from 10,008 to 11,850 kJ (2391 to 2831 kcal), an increase of 18.4%; growth in MAF (10,716 kJ in 2010) has been 22%, and in Asia, 32% (11,327 kJ in 2010; FAOSTAT, 2013). The percentage of animal products in daily per-capita total food consumption has increased consistently in Asia since 1970 (7% to 16%), remained constant in MAF (8%) and, since 1985, has decreased in OECD90 countries (32% to 28%), comprising, respectively, 1793, 865 and 3801 kJ in 2010 (FAOSTAT, 2013).

a)



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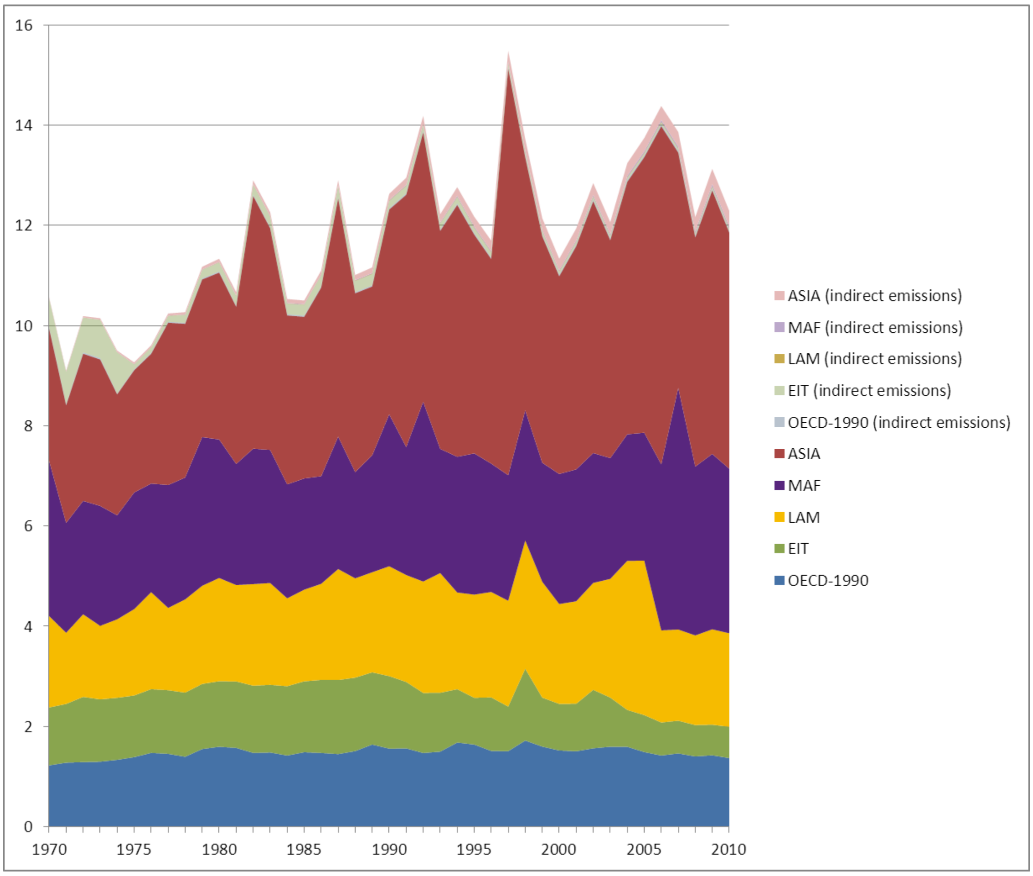

Figure 11.2. a) AFOLU emissions for the last four decades. For the agricultural sub-sectors emissions are shown for separate categories, based on FAOSTAT (2013). Emissions from crop residues, manure applied to soils, manure left on pasture, cultivated organic soils and synthetic fertilizers are typically aggregated to the category “agricultural soils” for IPCC reporting. For the FOLU sub-sector, (land use change and forestry) data are from the Houghton book-keeping model results (Houghton et al., 2012). Emissions from drained peat and peat fires are from JRC/PBL (2012), derived from Hooijer et al. (2010) and van der Werf et al. (2006); b) Emissions from AFOLU for each RC5 region (see Annex II.7) using data from JRC/PBL (2012), with emissions from energy end-use in the AFOLU sector from IEA (2012) included in a single aggregated category, see Annex II.8, used in the AFOLU section of Chapter 5.7.4 for cross-sectoral comparisons. The direct emission data from JRC/PBL (2012; see Annex II.8) represents land-based CO_2 emissions from forest and peat fires and decay that approximate to net CO_2 flux from the FOLU (Forestry and Other Land Use) sub-sector. Differences between FAOSTAT/Houghton data and JRC/PBL (2012) are discussed in the text. See Figures 11.4 and 11.6 for the range of differences among available databases for AFOLU emissions.

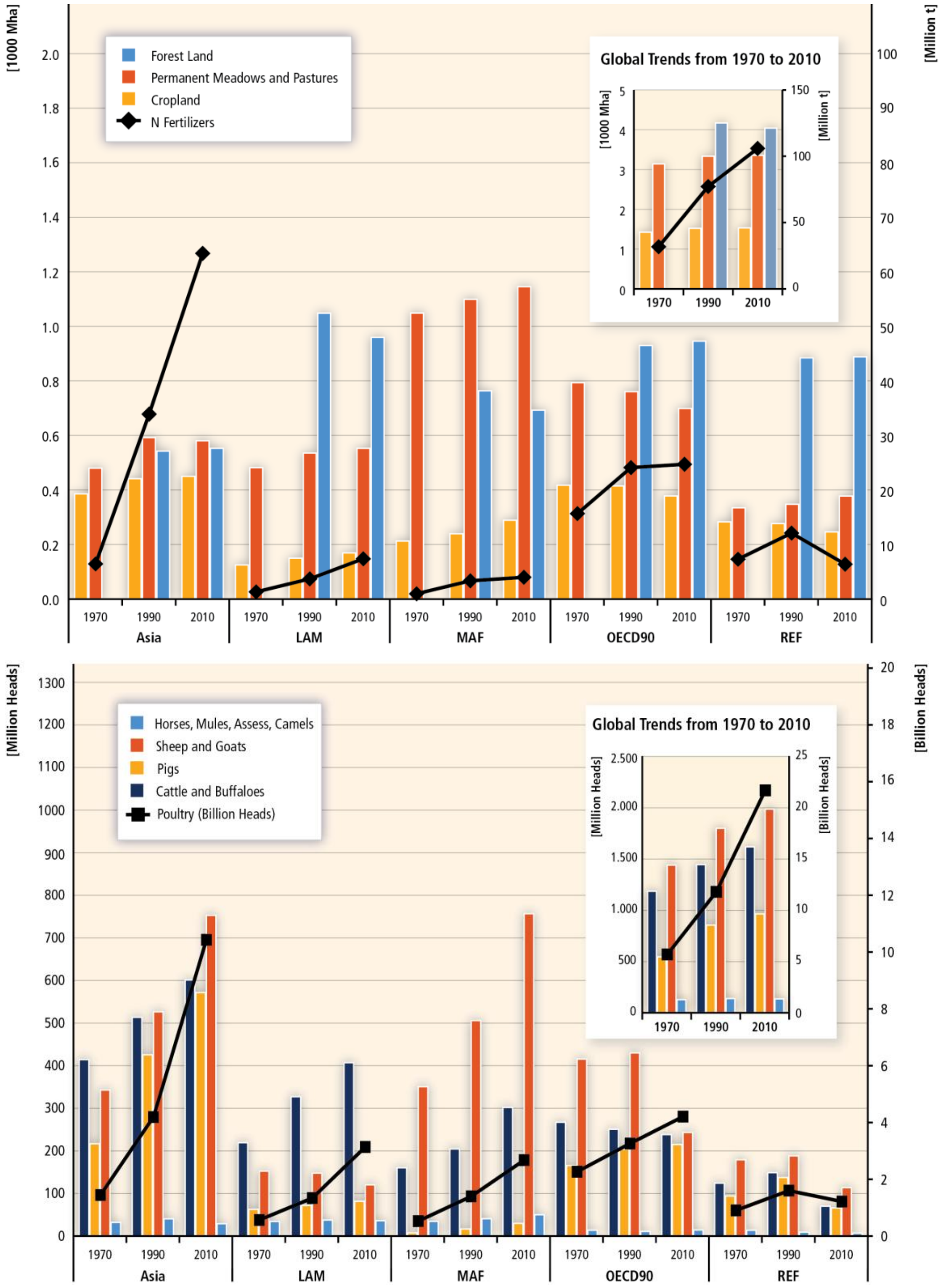

Figure 11.3. Global trends from 1971 to 2010 in (a) area of land use (forest land – available only from 1990; 1000 Mha) and amount of N fertilizer use (million tonnes), and (b) number of livestock (million head) and poultry (billion head). Data presented by regions: 1) Asia, 2) Latin America (LAM), 3) Middle East and Africa (MAF), 4) OECD90 countries; 5) countries with reforming economies (REF). The area extent of AFOLU land use categories, from FAOSTAT, (2013): ''Cropland'' corresponds to the sum of FAOSTAT categories ''arable land'' and ''temporary crops'' and coincides with the IPCC category (IPCC, 2003); ''Forest'' is defined according to FAO (FRA, 2010); countries reporting to UNFCCC may use different definitions. ''Permanent meadows and pasture'', are a subset of IPCC category ''grassland'' (IPCC, 2003), as the latter, by definition, also includes unmanaged natural grassland ecosystems.

11.2.2 Trends of GHG emissions from agriculture

Organic and inorganic material provided as inputs or output in the management of agricultural systems are typically broken down through bacterial processes, releasing significant amounts of CO_2, CH_4 and N_2O to the atmosphere. Only agricultural non-CO_2 sources are reported as anthropogenic GHG emissions however. The CO_2 gas emitted is considered neutral, being associated to annual cycles of carbon fixation and oxidation through photosynthesis. The agricultural sector is the largest contributor to global anthropogenic non-CO_2 GHGs, accounting for 56% of emissions in 2005 (U.S. EPA, 2011). Other important, albeit much smaller non-CO_2 emissions sources from other AFOLU categories, and thus not treated here, include fertilizer applications in forests. Annual total non-CO_2 GHG emissions from agriculture in 2010 are estimated to be 5.2-5.8 Gt CO_2eq/yr (FAOSTAT, 2013; Tubiello et al., 2013) and comprised about 10-12% of global anthropogenic emissions. Fossil fuel CO_2 emissions on croplands added another 0.4-0.6 Gt CO_2eq/yr in 2010 from agricultural use in machinery, such as tractors, irrigation pumps, etc. (Ceschia et al., 2010; FAOSTAT, 2013), but these emissions are accounted for in the energy sector rather than the AFOLU sector. Between 1990 and 2010, agricultural non-CO_2 emissions grew by 0.9%/yr, with a slight increase in growth rates after 2005 (Tubiello et al., 2013).

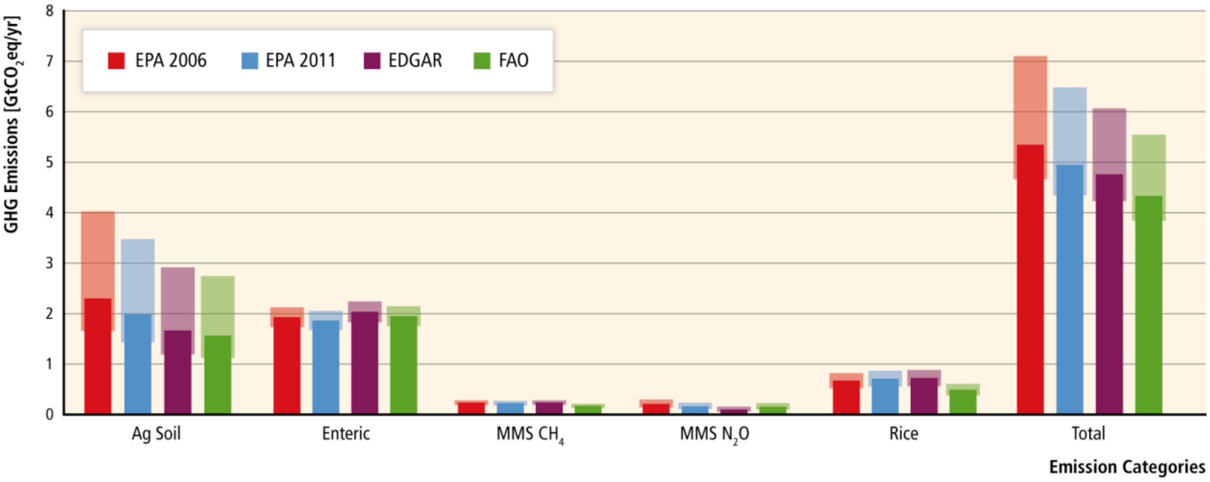

Figure 11.4. Data comparison between FAOSTAT, EPA (2006 and 2013) and EDGAR databases for key agricultural emission categories, grouped as: agricultural soils, enteric fermentation, manure management systems and rice cultivation, for 2005. Transparent ranges represent 95% confidence intervals of global aggregated categories, computed using IPCC guidelines (IPCC, 2006) for uncertainty estimation (from Tubiello et al., 2013).

Three independent sources of disaggregated non-CO_2 GHG emissions estimates from agriculture at global, regional and national levels are available. They are mostly based on FAOSTAT activity data and IPCC Tier 1 approaches (IPCC, 2006; FAOSTAT, 2012; JRC/PBL, 2012; US EPA, 2013). EDGAR and FAOSTAT also provide data at country level. Estimates of global emissions for enteric fermentation, manure management and manure, estimated using IPCC Tier 2 / 3 approaches are also available (e.g. Herrero et al., 2013). FAOSTAT, EDGAR and US EPA estimates are slightly different, although statistically consistent given the large uncertainties in IPCC default methodologies (Tubiello et al., 2013). They cover emissions from enteric fermentation; manure deposited on pasture; synthetic fertilizers; rice cultivation; manure management; crop residues; biomass burning; and manure applied to soils. Enteric fermentation, biomass burning and rice cultivation are reported separately under IPCC inventory guidelines, with the remaining categories aggregated into ''agricultural soils.'' According to EDGAR and FAOSTAT, emissions from enteric fermentation are the largest emission source, while US EPA lists emissions from agricultural soils as the dominant source (Figure 11.4).

The following analyses refer to annual total non-CO_2 emissions by all categories. All three databases agree that that enteric fermentation and agricultural soils represent together about 70% of total emissions, followed by paddy rice cultivation (9-11%), biomass burning (6-12%) and manure management (7-8%). If all emission categories are disaggregated, both EDGAR and FAOSTAT agree that the largest emitting categories after enteric fermentation (32-40% of total agriculture emissions) are manure deposited on pasture (15%) and synthetic fertilizer (12%), both contributing to emissions from agricultural soils. Paddy rice cultivation (11%) is a major source of global CH_4 emissions, which in 2010 were estimated to be 493-723 Mt CO_2eq/yr. The lower end of the range corresponds to estimates by FAO (FAOSTAT, 2013), with EDGAR and US EPA data at the higher end. Independent analyses suggest that emissions from rice may be at the lower end of the estimated range (Yan et al., 2009).

Enteric Fermentation. Global emissions of this important category grew from 1.4 to 2.1 Gt CO_2eq/yr between 1961 and 2010, with average annual growth rates of 0.70% (FAOSTAT, 2013). Emission growth slowed during the 1990s compared to the long-term average, but became faster again after the year 2000. In 2010, 1.0-1.5 Gt CO_2eq/yr (75% of the total emissions), were estimated to come from developing countries (FAOSTAT, 2013). Over the period 2000-2010, Asia and the Americas contributed most, followed by Africa and Europe (FAOSTAT, 2013); see Figure 11.5). Emissions have grown most in Africa, on average 2.4%/yr. In both Asia (2.0%/yr) and the Americas, (1.1%/yr) emissions grew more slowly, and decreased in Europe (-1.7%/yr). From 2000 to 2010, cattle contributed the largest share (75% of the total), followed by buffalo, sheep and goats (FAOSTAT, 2013).

Manure. Global emissions from manure, as either organic fertilizer on cropland or manure deposited on pasture, grew between 1961 and 2010 from 0.57 to 0.99 Gt CO_2eq/yr. Emissions grew by 1.1%/yr on average. Manure deposited on pasture led to far larger emissions than manure applied to soils as organic fertilizer, with 80% of emissions from deposited manures coming from developing countries (FAOSTAT, 2013; Herrero et al., 2013). The highest emitting regions from 2000-2010 were the Americas, Asia and Africa. Growth over the same period was most pronounced in in Africa, with an average of 2.5%/yr, followed by Asia (2.3%/yr) and the Americas (1.2%/yr), while there was a decrease in Europe of -1.2%/yr. Two-thirds of the total came from grazing cattle, with smaller contributions from sheep and goats. In this decade, emissions from manure applied to soils as organic fertilizer were greatest in Asia, then in Europe and the Americas. Though the continent with the highest growth rates of 3.4%/yr, Africa's share in total emissions remained small. In this sub-category, swine and cattle contributed more than three quarters (77%) of the emissions. Emissions from manure management grew from 0.25 to 0.36 Gt CO_2eq/yr, resulting in average annual growth rates of only 0.6%/yr during the period 1961-2010. From 2000-2010 most emissions came from Asia, then Europe and the Americas (Figure 11.5).

Synthetic Fertilizer. Emissions from synthetic fertilizers grew at an average rate of 3.9%/yr from 1961 to 2010, with absolute values increasing more than 9-fold, from 0.07 to 0.68 Gt CO_2eq/yr (Tubiello et al., 2013). Considering current trends, synthetic fertilizers will become a larger source of emissions than manure deposited on pasture in less than ten years and the second largest of all agricultural emission categories after enteric fermentation. Close to three quarters (70%) of these emissions were from developing countries in 2010. In the decade 2000-2010, the largest emitter by far was Asia, then the Americas and then Europe (FAOSTAT, 2012). Emissions grow in Asia by 5.3%/yr, in Africa by 2.0%/yr and in the Americas by 1.5%/yr. Emissions decreased in Europe (-1.8%/yr).

Rice. Emissions from rice are limited to paddy rice cultivation. From 1961 to 2010, global emissions increased with average annual growth rates of 0.4%/yr (FAOSTAT, 2013) from 0.37 to 0.52 Gt CO_2eq/yr., The growth in global emissions has slowed in recent decades, consistent with trends in rice cultivated area. During 200-2010, the largest share of emissions (94%) came from developing countries, with Asia being responsible for almost 90% of the total (Figure 11.5). The largest growth of emissions took place in in Africa (2.7%/yr), followed by Europe (1.4%/yr). Growth rates in Asia and the Americas were much smaller over the same period (0.4-0.7%/yr).

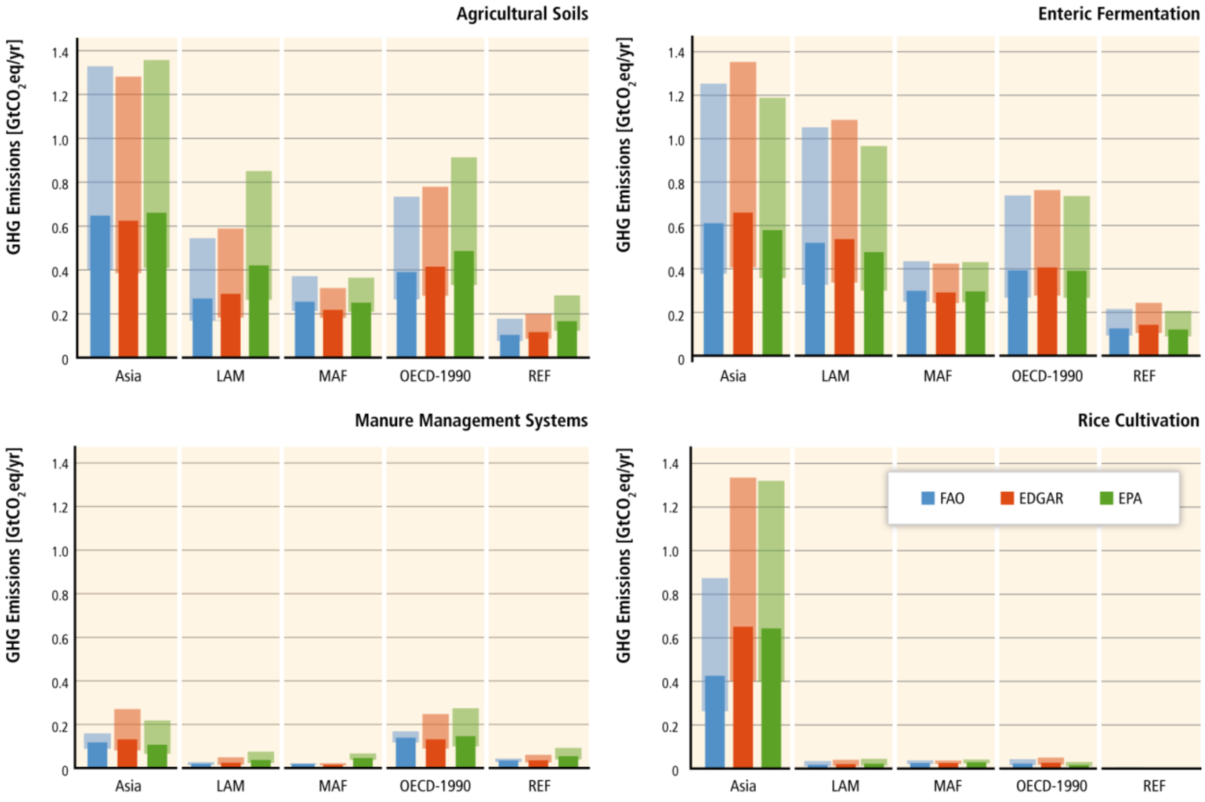

Figure 11.5. Regional data comparisons for key agricultural emission categories in 2010. Transparent ranges represent 95% confidence intervals computed using IPCC guidelines (IPCC, 2006; Tubiello et al., 2013). The data show that most of the differences between regions and databases are of the same magnitude as the underlying emission uncertainties.

11.2.3 Trends of GHG fluxes from forestry and other land use (FOLU)[3]

This section focuses on the most significant non-agricultural GHG fluxes to the atmosphere for which there are global trend data. Fluxes resulting directly from anthropogenic FOLU activity are dominated by CO_2 fluxes, primarily emissions due to deforestation, but also uptake due to reforestation/regrowth. Non-CO_2 greenhouse gas emissions from FOLU are small in comparison, and mainly arise from peat degradation through drainage and biomass fires (Box 11.1; Box 11.2).

FOLU accounted for about a third of anthropogenic CO_2 emissions from 1750 to 2011 and 12% of emissions in 2000 to 2009 (Table 11.1). At the same time, atmospheric measurements indicate the land as a whole was a net sink for CO_2, implying a “residual” terrestrial sink offsetting FOLU emissions (Table 11.1). This sink is confirmed by inventory measurements in both managed and unmanaged forests in temperate and tropical regions (Phillips et al., 1998; Luyssaert et al., 2008; Lewis et al., 2009; Pan et al., 2011). A sink of the right order of magnitude has been accounted for in models as a result of the indirect effects of human activity on ecosystems, i.e. the fertilising effects

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[3] The term FOLU used here, is consistent with AFOLU in the (IPCC, 2006) Guidelines and consistent with LULUCF Land Use, Land Use Change and Forestry (IPCC, 2003).

of increased levels of CO_2 and N in the atmosphere and the effects of climate change (WGI Chapter 6; Le Quéré et al., 2013), although some of it may be due to direct AFOLU activities not accounted for in current estimates (Erb et al., 2013). This sink capacity of forests is relevant to AFOLU mitigation through forest protection.

Global FOLU CO_2 flux estimates (Table 11.1 and Figure 11.6) are based on a wide range of data sources, and include different processes, definitions, and different approaches to calculating emissions (Houghton et al., 2012; Le Quéré et al., 2013; Pongratz et al., 2013). This leads to a large range across global FOLU flux estimates. Nonetheless, most approaches agree that there has been a decline in FOLU CO_2 emissions over the most recent years. This is largely due to a decrease the rate of deforestation (FRA, 2010; FAOSTAT, 2013).

Table 11.1: Net global CO_2 flux from AFOLU

	1750 to 2011 Cumulative Gt CO_2	1980–1989 Gt CO_2/yr	1990–1999 Gt CO_2/yr	2000–2009 Gt CO_2/yr
IPCC WGI Carbon Budget, Table 6.1[a]:				
Net AFOLU CO_2 flux [b]	660 ± 293	5.13 ± 2.93	5.87 ± 2.93	4.03 ± 2.93
Residual terrestrial sink[c]	-550 ± 330	-5.50 ± 4.03	-9.90 ± 4.40	-9.53 ± 4.40
Fossil fuel combustions and cement production[d]	1338 ± 110	20.17 ± 1.47	23.47 ± 1.83	28.60 ± 2.20
Meta-analyses of Net AFOLU CO_2 flux:				
IPCC WGI Table 6.2[e]		4.77 ± 2.57	4.40 ± 2.20	2.93 ± 2.20
Houghton et al, 2012[f]		4.18 ± 1.83	4.14 ± 1.83	4.03 ± 1.83

Notes:
Positive fluxes represent net emissions and negative fluxes represent net sinks.
(a) Selected components of the carbon budget in IPCC, Fifth Assessment Report, Working Group 1, Chapter 6, Table 6.1.
(b) From the bookkeeping model accounting method of Houghton (2003), updated in Houghton et al., (2012), uncertainty based on expert judgement ; 90% confidence uncertainty interval.
(c) Calculated as residual of other terms in the carbon budget.
(d) Fossil fuel flux shown for comparison (Boden et al., 2011).
(e) Average of estimates from 12 process models, only 5 were updated to 2009 and included in the 2000-2009 mean. Uncertainty based on standard deviation across models, 90% confidence uncertainty interval (WGI Chapter 6).
(f) Average of 13 estimates including process models, book-keeping model and satellite/model approaches, only 4 were updated to 2009 and included in the 2000-2009 mean. Uncertainty based on expert judgement.

Regional trends in FOLU CO_2 emissions are shown in Figure 11.7. Model results indicate FOLU emissions peaked in the 1980s in Asia and LAM regions and declined thereafter. This is consistent with a reduced rate of deforestation, most notably in Brazil[4], and some areas of afforestation, the latter most notably in China, Vietnam and India (FAOSTAT, 2013). In MAF the picture is mixed, with the Houghton model (Houghton et al., 2012) showing a continuing increase from the 1970s to the 2000s, while the VISIT model (Kato et al., 2011) indicates a small sink in the 2000s. The results for temperate and boreal areas represented by OECD and REF regions are very mixed ranging from large net sources (ISAM) to small net sinks. The general picture in temperate and boreal regions is of declining emissions and/or increasing sinks. These regions include large areas of managed forests

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[4] For annual deforestation rates in Brazil see http://www.obt.inpe.br/prodes/index.php

subjected to harvest and regrowth, and areas of reforestation (e.g. following cropland abandonment in the USA and Europe). Thus results are sensitive to whether and how the models include forest management and environmental effects on regrowing forests.

The book-keeping model method (Houghton, 2003; Houghton et al., 2012) uses regional biomass, growth and decay rates from the inventory literature that are not varied to account for changes in climate or CO_2. It includes forest management associated with shifting cultivation in tropical forest regions as well as global wood harvest and regrowth cycles. The primary source of data for the most recent decades is FAO forest area and wood harvest (FRA, 2010). FAOSTAT (2013) uses the default IPCC methodologies to compute: stock-difference to estimate emissions and sinks from forest management; carbon loss associated to forest conversion to other land uses as a proxy for emissions from deforestation; GFED4 data on burned area to estimate emissoins from peat fires; and spatial analyses to determine emissions from drained organic soils (IPCC, 2007). The other models in Fig 11.6 & 11.7 are process-based terrestrial ecosystem models that simulate changing plant biomass and carbon fluxes, and include climate and CO_2 effects, with a few now including the nitrogen cycle (Zaehle et al., 2011; Jain et al., 2013). Inclusion of the nitrogen cycle results in much higher modelled net emissions in the ISAM model (Jain et al., 2013) as N limitation due to harvest removals limits forest regrowth rates, particularly in temperate and boreal forests. Change in land cover in the process models is from the HYDE dataset (Goldewijk et al., 2011; Hurtt et al., 2011), based on FAO cropland and pasture area change data. Only some process models include forest management in terms of shifting cultivation (VISIT) or wood harvest and forest degradation (ISAM); none account for emissions from peatlands (see Box 11.1).

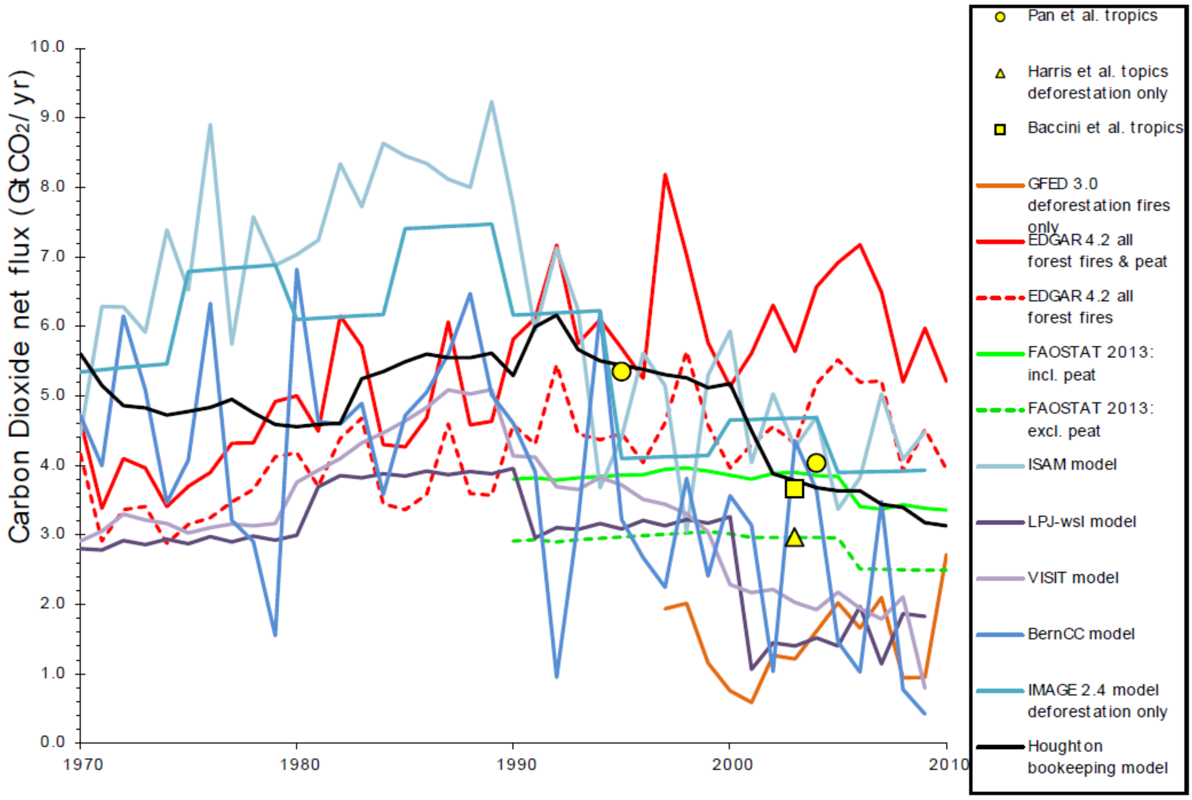

Figure 11.6. Global net CO_2 emission estimates from FOLU (Forestry and Other Land Use including land use change). The symbols represent mean values for the tropics only. Yellow circles: tropical deforestation and forest management for the 1990s and 2000-2007 (Pan et al., 2011) using the Houghton (2003) book-keeping model approach and FAO data. Yellow triangle: tropical deforestation only, mean over 2000 to 2007 based on satellite forest area and biomass data (Baccini et al., 2012; Harris et al., 2012b). Yellow square: tropical deforestation and forest management, mean over 2000 to 2007 based on satellite forest area and biomass data and FAO data using bookkeeping model (Baccini et al., 2012; Harris et al., 2012b). Orange line: deforestation and degradation fires only based on satellite fire data from GFED 3.0 database (van der Werf et al., 2010). Dark Red line: EDGAR

"LULUCF" emissions derived from the GFED 2.0 database (van der Werf et al., 2006) of emissions due to all forest fires (includes both FOLU and non-FOLU fires), with (solid line) and without (dashed line) peat fires and decay. Green line: emissions from land use change and management from FAO agricultural and forest inventory data (FAOSTAT, 2013), shown with (solid line) and without (dashed line) peat fires and peat degradation. Black line: Houghton book-keeping model approach updated to 2010 as in (Houghton et al., 2012), including land use change and forest management but no peatlands. Other coloured lines: a selection of process-based vegetation model results, updated for WGI Chapter 6; (Le Quéré et al., 2013) include land use change, some include forest management, none include peatlands. LPJ-wsl: (Poulter et al., 2010); BernCC: (Stocker et al., 2011); VISIT: (Kato et al., 2011); ISAM: (Jain et al., 2013), IMAGE 2.4 (van Minnen et al., 2009, deforestation only).

Figure 11.7

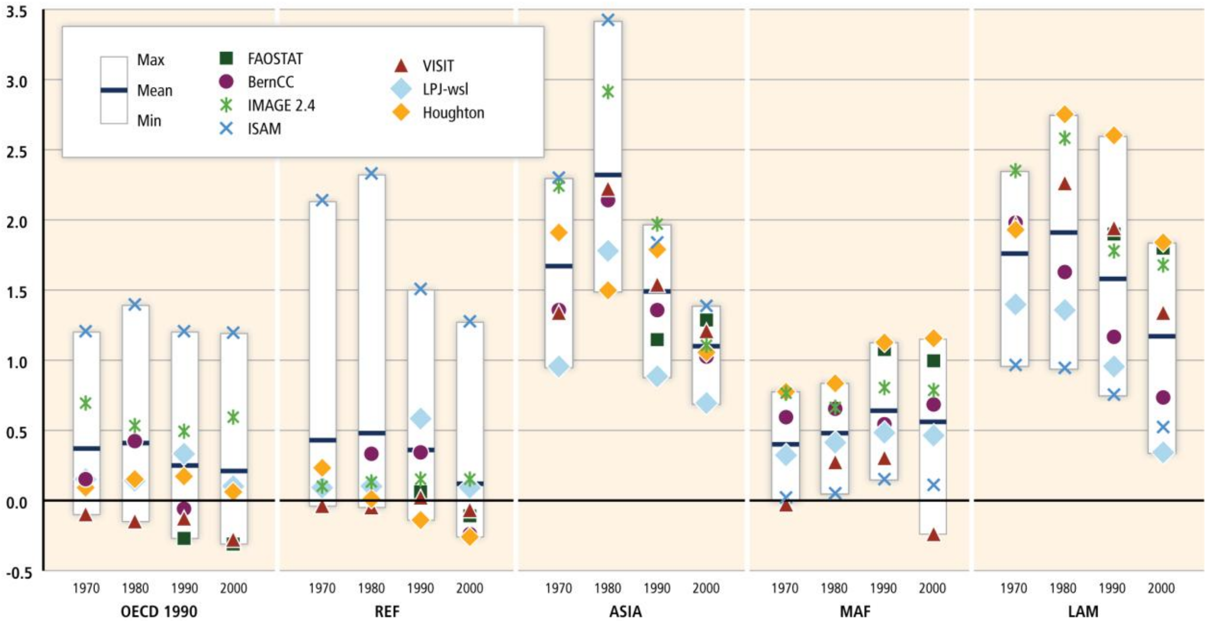

Figure 11.7. Regional trends in net CO$_2$ fluxes from FOLU (Forestry and Other Land Use including land use change). Houghton book-keeping model approach updated to 2010 as in (Houghton et al., 2012) and 5 process-based vegetation models updated to 2010 for WGI Chapter 6; (Le Quéré et al., 2013): LPJ-wsl: (Poulter et al., 2010); BernCC: (Stocker et al., 2011); VISIT: (Kato et al., 2011); ISAM: (Jain et al., 2013), IMAGE 2.4 (van Minnen et al., 2009, deforestation only). Only the FAO estimates (FAOSTAT, 2013) include peatlands.

Satellite estimates of change in land cover have been combined with model approaches to calculate tropical forest emissions (Hansen et al., 2010). The data is high resolution and verifiable, but only covers recent decades, and does not account for fluxes due to land use change that occurred prior to the start of the study period (e.g. decay or regrowth). Satellite data alone cannot distinguish the cause of change in land use (deforestation, natural disturbance, management) , but can be used in conjunction with acitivity data for attribution (Baccini et al., 2012). A recent development is the use of satellite-based forest biomass estimates (Saatchi et al., 2011) together with satellite land cover change in the tropics to estimate "gross deforestation" emissions (Harris et al., 2012a) or further combining it with FAO and other activity data to estimate net fluxes from forest area change and forest management (Baccini et al., 2012).

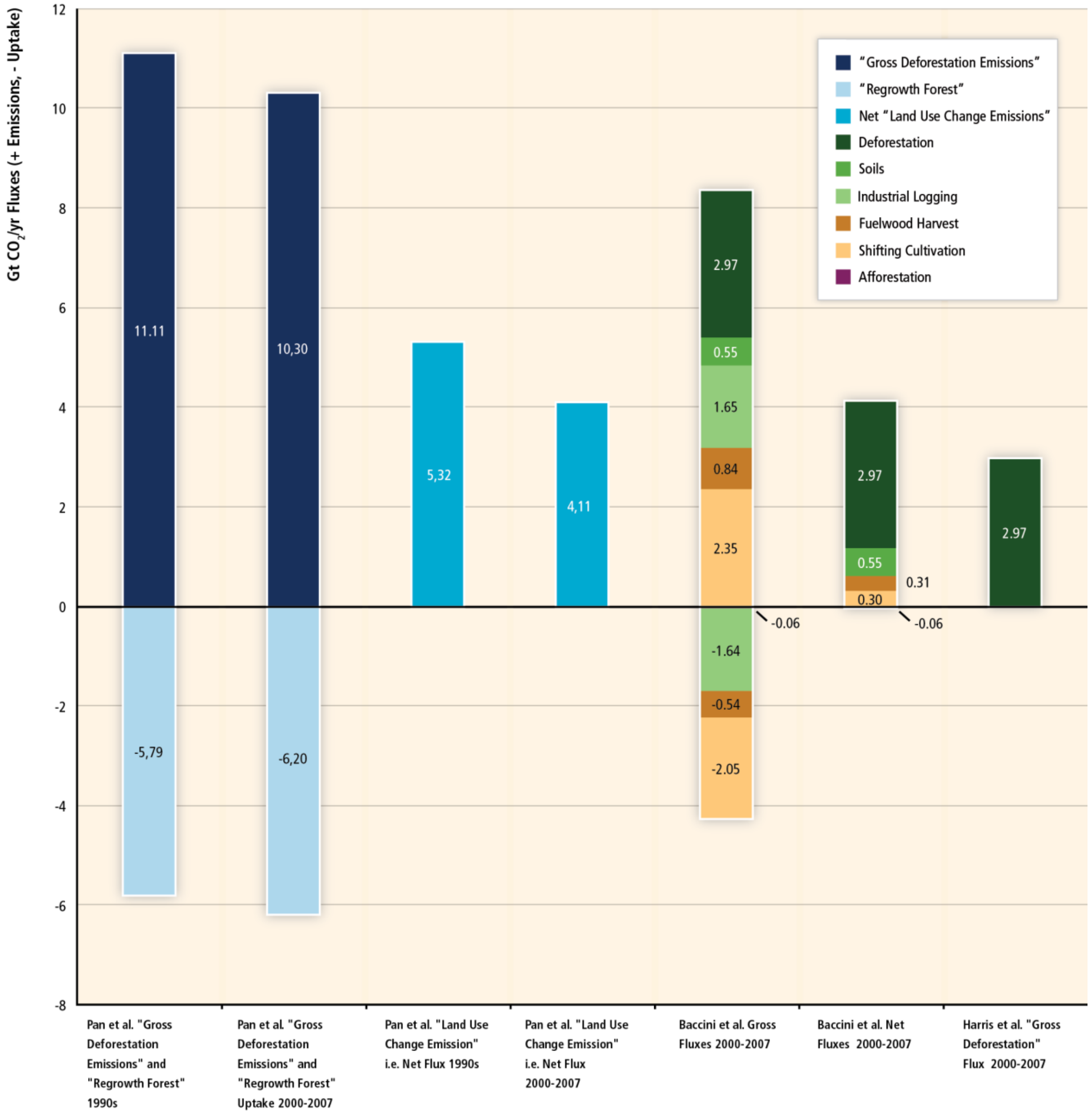

Figure 11.8. Breakdown of mean annual CO_2 fluxes from deforestation and forest management in tropical countries (Gt CO_2/yr). (Pan et al., 2011) estimates are based on FAO data and the Houghton book-keeping model (Houghton, 2003). (Baccini et al., 2012) estimates are based on satellite land cover change and biomass data with FAO data, and the (Houghton, 2003) book-keeping model, with the detailed breakdown of these results shown in Houghton 2013. Harris et al. (2012) estimates are based on satellite land cover change and biomass data.

A detailed breakdown of the component fluxes in (Baccini et al., 2012) is shown in Figure 11.8. Where there is temporary forest loss through management, "gross" forest emissions can be as high as for permanent forest loss (deforestation), but are largely balanced by "gross" uptake in regrowing forest, so net emissions are small. When regrowth does not balance removals, it leads to a degradation of forest carbon stocks. In Baccini et al. (2012) this degradation was responsible for 15% of total net emissions from tropical forests (Houghton, 2013; Figure 11.8). Huang and Asner, (2010), estimated that forest degradation in the Amazon, particularly from selective logging, is responsible for 15-19% higher C emissions than reported from deforestation alone. Pan et al., (2011) separated "gross emissions" from deforestation and forest management on the one hand, from uptake in regrowing vegetation on the other. Deforestation emissions decline from the 1990s to 2000-2007, and uptake in regrowing vegetation increases, both contributing to the decline in net tropical CO_2 emissions.

Satellite fire data has also been used to estimate FOLU emissions (van der Werf et al., 2006; Box 11.2). The EDGAR[5] database "Land Use Change and Forestry" emissions are based on forest and peat fire data from GFED 2.0 (van der Werf et al., 2006), with additional estimates of post-burn decay, and emissions from degraded peatlands based on (Joosten, 2010; Box 11.1). However, GFED 2.0 fire data does not distinguish anthropogenic AFOLU fires from other fires, unlike GFED 3.0 (van der Werf et al., 2010; Box 11.2). Fire data also does not capture significant additional AFOLU fluxes due to land clearing and forest management that is by harvest rather than fire (e.g. deforestation activities outside the humid tropics) or regrowth following clearing. Thus EDGAR data only approximates the FOLU flux.

FAO estimates AFOLU GHG emissions (FAOSTAT, 2013)[6] based on IPCC Tier 1 methodology[7]. With reference to the decade 2001-2010, total GHG FOLU emissions were +3.2 Gt CO_2 eq/yr including: deforestation (+3.8 Gt CO_2 eq/yr), forest degradation and forest management (-1.8 Gt CO_2eq /yr), biomass fires including peatland fires (+0.3 Gt CO_2eq/yr), and drained peatlands (+0.9 Gt CO_2eq/yr). The FAO estimated total mean net GHG FOLU flux to the atmosphere decreased from +3.9 Gt CO_2eq/yr in 1991-2000 to +3.2 Gt CO_2eq/yr in 2001-2010 (FAOSTAT, 2013).

Box 11.1 AFOLU GHG emissions from Peatlands and Mangroves

Undisturbed waterlogged peatlands (organic soils) store a large amount of carbon and act as small net sinks (Hooijer et al., 2010). Drainage of peatlands for agriculture and forestry results in a rapid increase in decomposition rates, leading to increased emissions of CO_2, and N_2O, and vulnerability to further GHG emissions through fire. The FAO emissions database estimates globally 250,000 km^2 of drained organic soils under cropland and grassland, with total GHG emissions of 0.9 Gt CO_2eq/yr in 2010 - with the largest contributions from Asia (0.44 Gt CO_2eq/yr) and Europe (0.18 Gt CO_2eq/yr; (FAOSTAT, 2013). Joosten, (2010), estimated that there are >500,000 km^2 of drained peatlands in the world including under forests, with CO_2 emissions having increased from 1.06 Gt CO_2/yr in 1990 to 1.30 Gt CO_2/yr in 2008, despite a decreasing trend in Annex I countries, from 0.65 to 0.49 Gt CO_2/yr, primarily due to natural and artificial rewetting of peatlands. In Southeast Asia, CO_2 emissions from drained peatlands in 2006 were 0.61 ± 0.25 Gt CO_2/yr (Hooijer et al., 2010). Satellite estimates indicate that peat fires in equatorial Asia emitted on average 0.39 Gt CO_2 eq/yr over the period 1997-2009 (van der Werf et al., 2010), but only 0.2 Gt CO_2 eq/yr over the period 1998-2009. This lower figure is consistent with recent independent FAO estimates over the same period and region. Mangrove ecosystems have declined in area by 20% (36Mha) since 1980, although the rate of loss has been slowing in recent years, reflecting an increased awareness of the value of these ecosystems (FAO, 2007). A recent study estimated that deforestation of mangroves released 0.07 to 0.42 Gt CO_2/yr (Donato et al., 2011).

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[5] http://edgar.jrc.ec.europa.eu/index.php

[6] http://faostat.fao.org/

[7] Parties to the UNFCCC report net GHG emissions according to IPCC methodologies (IPCC, 2003, 2006). Reporting is based on a range of methods and approaches dependent on available data and national capacities, from default equations and emission factors applicable to global or regional cases and assuming instantaneous emissions of all carbon that will be eventually lost from the system following human action (tier 1) to more complex approaches such as model-based spatial analyses (tier 3).

Box 11.2. AFOLU GHG emissions from Fires

Burning vegetation releases CO_2, CH_4, N_2O, ozone-precursors and aerosols (including black carbon) to the atmosphere. When vegetation regrows after a fire, it takes up CO_2 and nitrogen. Anthropogenic land management or land conversion fire activities leading to permanent clearance or increasing levels of disturbance result in net emissions to the atmosphere over time. Satellite-detection of fire occurrence and persistence has been used to estimate fire emissions (e.g. GFED 2.0 database; van der Werf et al., 2006). It is hard to separate the causes of fire as natural or anthropogenic, especially as the drivers are often combined. An update of the GFED methodology now distinguishes FOLU deforestation and degradation fires from other management fires (GFED 3.0 database; van der Werf et al., 2010; Figure 11.6). The estimated tropical deforestation and degradation fires emissions were 1.39 Gt CO_2eq/yr during 1997 to 2009 (total carbon including CO_2, CH_4, CO and black carbon), 20% of all fire emissions. CO_2 FOLU fire emissions are already included as part of the global models results such as those presented in Table 1.1 and figures 11.6 and 11.7. According to (FAOSTAT, 2013)[8], in 2010 the non-CO_2 component of deforestation and forest degradation fires totalled 0.1 Gt CO_2eq/yr, with forest management and peatland fires (Box 11.1) responsible for an additional 0.2 Gt CO_2eq/yr.

11.3 Mitigation technology options and practices, and behavioural aspects

Greenhouse gases can be reduced by supply-side mitigation options (i.e. by reducing GHG emissions per unit of land / animal, or per unit of product), or by demand-side options (e.g. by changing demand for food and fibre products, reducing waste). In IPCC AR4 the forestry chapter (Nabuurs et al., 2007) considered some demand-side options, but the agriculture chapter focused on supply-side options only (Nabuurs et al., 2007; Smith, et al., 2007). In this section we discuss only supply-side options (11.3.1). Demand-side options are discussed in 11.4.

Mitigation activities in the AFOLU sector can reduce climate forcing in different ways:

- Reductions in CH_4 or N_2O emissions from croplands, grazing lands and livestock.
- Conservation of existing carbon stocks, e.g. conservation of forest biomass, peatlands and soil carbon that would otherwise be lost.
- Reductions of carbon losses from biota and soils, e.g. through management changes within the same land-use type (e.g. reducing soil carbon loss by switching from tillage to no-till cropping) or by reducing losses of carbon-rich ecosystems, e.g. reduced deforestation, rewetting of drained peatlands.
- Enhancement of carbon sequestration in soils, biota and long-lived products through increases in the area of carbon-rich ecosystems such as forests (afforestation, reforestation), increased carbon storage per unit area, e.g. increased stocking density in forests, carbon sequestration in soils, and wood use in construction activities.
- Changes in albedo resulting from land-use and land-cover change that increase reflection of visible light.
- Provision of products with low GHG emissions that can replace products with higher GHG emissions for delivering the same service (e.g. replacement of concrete and steel in buildings with wood, some bioenergy options; see 11.13).

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[8] FOLU GHG emissions by fires include, as per IPCC GHG guidelines, all fires on managed land. Most current FOLU estimates are limited however to fires associated to deforestation, forest management and peat fires. Emissions form prescribed burning of savannahs are reported under agriculture. Both CO_2 and non-CO_2 emissions are accounted under these FOLU components, but CO_2 emissions dominate.

- Reductions of direct (e.g. agricultural machinery, pumps, fishing craft) or indirect (e.g. production of fertilizers, emissions resulting from fossil energy use in agriculture, fisheries, aquaculture and forestry or from production of inputs); though indirect emission reductions are accounted for in the energy end-use sectors (buildings, industry, energy generation, transport) so are not discussed further in detail in this chapter.

11.3.1 Supply-side mitigation options

Mitigation potentials for agricultural mitigation options were given on a “per-area” and “per-animal” in AR4 (Nabuurs et al., 2007; Smith, et al., 2007). All options are summarised in Table 11.2 with impacts on each GHG noted, and a categorisation of technical mitigation potential, ease of implementation and availability (supported by recent references). These mitigation options can have additive positive effects, but can also work in opposition, e.g. zero tillage can reduce the effectiveness of residue incorporation. Most mitigation options were described in detail in AR4 so are not described further here; additional practices that were not considered in AR4, i.e. biochar, reduced emissions from aquaculture, and bioenergy are described in Boxes 11.3, 11.4 and 11.5, respectively.

In addition to the per-area and per-animal mitigation options described in AR4, more attention has recently been paid to options that reduce emissions intensity by improving the efficiency of production (i.e. less GHG emissions per unit of agricultural product; Burney et al., 2010; Bennetzen et al., 2012); a reduction in emissions intensity has long been a feature of agricultural emissions reduction and is one component of a process more broadly referred to as sustainable intensification (Tilman et al., 2009; Godfray et al., 2010; Smith, 2013; Garnett et al., 2013);. This process does not rely on reducing inputs *per se*, but relies on the implementation of new practices that result in an increase in product output that is larger than any associated increase in emissions (Smith, 2013). Even though per-area emissions could increase, there is a net benefit since less land is required for production of the same quantity of product. The scope to reduce emissions intensity appears considerable since there are very large differences in emissions intensity between different regions of the world (Herrero et al., 2013). Sustainable intensification is discussed further in section 11.4.2, and trends in changes in emissions intensity are discussed further in section 11.6.

Table 11.2: Summary of supply-side mitigation options in the AFOLU sector. Technical Mitigation Potential: Area = t CO_2eq/ha/y; Animal = % reduction of enteric emissions. Low = < 1; <5% (light colour), Medium =1-10; 5-15% (medium colour), High = >10, >15% (dark colour); Ease of Implementation (acceptance or adoption by land manager): Difficult (light colour), Medium (medium colour), Easy, i.e. universal applicability (dark colour); Timescale for Implementation: Long-term (at research and development stage; light colour), Mid-term (trials in place, within 5-10 years; medium colour), Immediate (technology available now, dark colour).

Categories	**Practices and Impacts**	**Technical Mitigation Potential**	**Ease of Implementation**	**Timescale for implementation**	**References**
Forestry					
Reducing deforestation	**C**: Conservation of existing C pools in forest vegetation and soil by controlling deforestation protecting forest in reserves, and controlling other anthropogenic disturbances such as fire and pest outbreaks. Reducing slash and burn agriculture, reducing forest fires.	▒	▒	█	1
	CH_4, N_2O: Protection of peatland forest, reduction of wildfires	▒	░	▒	2
Afforestation / Reforestation	**C**: Improved biomass stocks by planting trees on non-forested agricultural lands. This can include either monocultures or mixed species plantings. These activities may also provide a range of other social, economic and environmental benefits.	▒	█	█	3, 4, 5
Forest management	**C:** Management of forests for sustainable timber production including	▒	▒	█	6, 7, 8, 9

	extending rotation cycles, reducing damage to remaining trees, reducing logging waste, implementing soil conservation practices, fertilization, and using wood in a more efficient way, sustainable extortion of wood energy	▒	▒	█	
	CH_4, N_2O: wildfire behaviour modification	▒	░	▒	10, 11, 12
Forest restoration	**C**: Protecting secondary forests and other degraded forests whose biomass and soil C densities are less than their maximum value and allowing them to sequester C by natural or artificial regeneration, Rehabilitation of degraded lands, long term fallows	▒	▒	█	13, 14
	CH_4, N_2O : Wildfire behaviour modification	▒	░	▒	
Land-based Agriculture					
Cropland management					
Croplands – plant management	**C**: High input carbon practices, e.g. improved crop varieties, crop rotation, use of cover crops, perennial cropping systems, agricultural biotechnology	▒	█	█	15, 16, 17
	N_2O: Improved N use efficiency	▒	█	█	18
Croplands – nutrient management	**C**: Fertilizer input to increase yields and residue inputs (especially important in low-yielding agriculture)	▒	▒	█	19, 20
	N_2O: Changing N fertilizer application rate, fertiliser type, timing, precision application, inhibitors	▒	▒	█	21, 22, 23, 24, 25, 105, 106
Croplands – tillage/residues management	**C**: Reduced tillage intensity; Residue retention	█	█	█	17, 24, 26, 27
	N_2O:	█	█	█	28, 96, 97
	CH_4:	█	█	█	96
Croplands – water management	**C**: Improved water availability in cropland including water harvesting and application	▒	▒	▒	29
	CH_4: Decomposition of plant residues	▒	▒	▒	
	N_2O: Drainage management to reduce emissions, reduce N runoff leaching	▒	▒	▒	
Croplands – rice management	**C**: Straw retention,	▒	█	█	30
	CH_4: Water management, mid-season paddy drainage	█	▒	█	31, 32, 98
	N_2O: Water management, N fertilizer application rate, fertiliser type, timing, precision application	░	█	█	32, 98, 99
Rewet peatlands drained for agriculture	**C**: Ongoing CO_2 emissions from reduced drainage (but CH_4 emissions may increase)	█	░	█	33
Croplands – set-aside & LUC	**C**: Replanting to native grasses and trees. Increase C sequestration	█	▒	█	34, 35, 36, 37, 38
	N_2O: N inputs decreased resulting in reduced N_2O	█	▒	█	
Biochar application	**C**: Soil amendment to increase biomass productivity, and sequester C (Biochar was not covered in AR4 so is described in Box 11.3).	█	▒	░	39, 40, 41
	N_2O: Reduced N inputs will reduce emissions	░	▒	░	39, 42
Grazing Land Management					
Grazing Lands – Plant management	**C:** Improved grass varieties / sward composition, e.g. deep rooting grasses, increased productivity and nutrient management. Appropriate stocking densities, carrying capacity, fodder banks and improved grazing management	░	█	█	43, 44, 45
	N_2O	░	█	█	46
Grazing Lands –Animal management	**C:** Appropriate stocking densities, carrying capacity management, fodder banks and improved grazing management, fodder production and fodder diversification	░	█	█	43, 47
	CH_4	░	█	█	
	N_2O: Stocking density, animal waste management	░	▒	█	
Grazing Land- Fire management	**C:** Improved use of fire for sustainable grassland management. Fire prevention and improved prescribed burning	░	▒	█	
Revegetation					
Revegetation	**C:** The establishment of vegetation that does not meet the definitions of afforestation and reforestation (e.g. *Atriplex* spp.)	█	█	█	48
	CH_4: Increased grazing by ruminants may increase net emissions	░	█	█	
	N_2O: Reduced N inputs will reduce emissions	█	█	█	
Other					
Organic soils – restoration	**C:** Soil carbon restoration on peatlands; and avoided net soil carbon emissions using improved land management	█	█	█	49
	CH_4: May increase	░	█	█	
Degraded soils – restoration	Land reclamation (afforestation, soil fertility management, water conservation soil nutrients enhancement, improved fallow.)	▓	▓	▓	100, 101, 102, 103, 104
Biosolid applications	**C:** Use of animal manures and other biosolids for improved management of nitrogen; integrated livestock agriculture techniques	▓	▓	▓	26

	N_2O:	░	▓	▒	
Livestock					
Livestock – feeding	**CH_4**: Improved feed and dietary additives to reduce emissions from enteric fermentation; including improved forage, dietary additives (bioactive compounds, fats), ionophores / antibiotics, propionate enhancers, archaea inhibitors, nitrate and sulphate supplements	▒	▒	▒	50, 51, 52, 53, 54, 55, 56, 57, 58, 59
Livestock – breeding and other long term management	**CH_4**: Improved breeds with higher productivity (so lower emissions per unit of product) or with reduced emissions from enteric fermentation; microbial technology such as archaeal vaccines, methanotrophs, acetogens, defaunation of the rumen, bacteriophages and probiotics; improved fertility	▒	░	░	54, 55, 56, 58, 60, 61, 62, 63, 64, 65, 66, 67, 68, 69, 70, 71
Manure management	**CH_4**: Manipulate bedding and storage conditions, anaerobic digesters; biofilters, dietary additives	▓	▓	▓	56, 58, 72, 73
	N_2O: Manipulate livestock diets to reduce N excreta, soil applied and animal fed nitrification inhibitors, urease inhibitors, fertilizer type, rate and timing, manipulate manure application practices, grazing management	▓	▒	▓	56, 58, 72, 74, 75, 76, 77, 78
Integrated Systems					
Agroforestry (including agropastoral and agrosilvopastoral systems)	**C:** Mixed production systems can increase land productivity and efficiency in the use of water and other resources and protect against soil erosion as well as serve carbon sequestration objectives.	▒	▒	█	79, 80, 81, 82, 83, 84, 85, 86, 87, 88
	N_2O: Reduced N inputs will reduce emissions	▒	▒	█	
Other mixed biomass production systems	**C:** Mixed production systems such as double-cropping systems and mixed crop-livestock systems can increase land productivity and efficiency in the use of water and other resources as well as serve carbon sequestration objectives. Perennial grasses (e.g. bamboo) can in the same way as woody plants be cultivated in shelter belts and riparian zones/buffer strips provide environmental services and supports C sequestration and biomass production.	▒	█	█	82, 89, 90
	N_2O: Reduced N inputs will reduce emissions	▒	█	█	
Integration of biomass production with subsequent processing in food and bioenergy sectors	**C:** Integrating feedstock production with conversion, typically producing animal feed that can reduce demand for cultivated feed such as soy and corn and can also reduce grazing requirements. Using agricultural and forestry residues for energy production.	▒	█	█	91, 92, 93, 94, 95
	N_2O: Reduced N inputs will reduce emissions	░	█	█	
Bioenergy (seeBox 11.5 and Section 11.13)					

[1]Van Bodegom et al. (2009); [2]Malmsheimer et al. (2009); [3]Reyer et al. (2009); [4]Sochacki et al. (2012); [5]IPCC (2000); [6]DeFries and Rosenzweig (2010); [7]Takimoto et al. (2008); [8] Masera et al. (2003); [9]Silver et al. (2000); [10] Dezzeo et al. (2005); [11]Ito (2005); [12] Sow et al. (2013); [13]Reyer et al. (2009); [14]Palm et al. (2004); [15]Godfray et al. (2010); [16] Burney et al. (2010); [17]Conant et al. (2007); [18] Huang and Tang (2010); [19]Lemke et al. (2010); [20]Eagle and Olander (2012); [21]Snyder et al. (2007); [22] Akiyama et al. (2010); [23]Barton et al. (2011); [24]Powlson et al. (2011); [25] van Kessel et al. (2013); [26]Farage et al. (2007); [27] Smith (2012); [28] Abdalla et al. (2013); [29]Bayala et al. (2008); [30] Yagi et al. (1997); [31] Tyagi et al. (2010); [32]Feng et al. (2013); [33]Lohila et al. (2004); [34]Seaquist et al. (2008); [35]Mbow (2010); [36]Assogbadjo et al. (2012); [37] Laganiere et al. (2010); [38] Bayala et al. (2011); [39] Singh et al. (2010); [40]Woolf et al. (2010); [41]Lehmann et al. (2003); [42]Taghizadeh-Toosi et al. (2011); [43]Franzluebbers and Studemann (2009); [44]Follett and Reed (2010); [45]McSherry and Ritchie (2013); [46]Saggar et al. (2004); [47]Thornton and Herrero (2010); [48]Harper et al. (2007); [49]Smith and Wollenberg (2012); [50]Beauchemin et al. (2008); [51]Beauchemin et al. (2009); [52]Martin et al. (2010); [53]Grainger and Beauchemin (2011); [54]Clark (2013); [55]Cottle et al. (2011); [56]Eckard et al. (2010); [57]Sauvant and Giger-Reverdin (2007); [58]Hristov et al. (2013); [59]Bryan et al. (2013); [60]Attwood and McSweeney (2008); [61]Attwood et al. (2011); [62]Hegarty et al. (2007); [63]Hook et al. (2010); [64]Janssen and Kirs (2008); [65]Martin et al. (2010); [66]Morgavi et al. (2008); [67]Morgavi et al. (2010); [68]Place and Mitloehner (2010); [69]Waghorn and Hegarty (2011); [70]Wright and Klieve (2011); [71]Yan et al. (2010) [72]Chadwick et al. (2011); [73]Petersen and Sommer (2011); [74]de Klein et al. (2010); [75]de Klein and Eckard (2008); [76]Dijkstra et al. (2011); [77]Schils et al. (2013); [78]VanderZaag et al. (2011); [79]Oke and Odebiyi (2007); [80]Rice (2008); [81]Takimoto et al. (2008); [82]Lott et al. (2009); [83]Sood and Mitchell (2011); [84]Assogbadjo et al. (2012); [85]Wollenberg et al. (2012); [86]Semroc et al. (2012); [87]Souza et al. (2012); [88]Luedeling and Neufeldt (2012); [89]Heggenstaller et al. (2008); [90] Herrero et al. (2010); [91]Dale et al. (2009); [92]Dale et al. (2010); [93]Sparovek et al. (2007; [94]Sood and Mitchell (2011); [95]Vermeulen et al. (2012); [96]Metay et al. (2007); [97]Rochette (2008); [98]Ma et al. (2009); [99]Yao et al. (2010); [100]Arnalds (2004); [101]Batjes (2004); [102]Hardner et al. (2000); [103]May et al. (2004); [104]Zhao et al. (2005); [105]Huang and Tang (2010); [106]Kim et al. (2013).

Box 11.3 BiocharThis box summarises the mitigation potential for biochar technologies, which were not considered in AR4. Biomass C stabilisation could be combined with (or substitute) bioenergy capture as part of a land-based mitigation strategy (Lehmann, 2007). Heating biomass with air excluded (pyrolysis) generates energy-containing volatiles and gases. Hydrogen and O are preferentially eliminated, creating a stable (biologically recalcitrant) C-rich co-product (char). By adding char to soil as ‘biochar’ a system can be established that may have a higher carbon abatement than typical bioenergy alternatives (Woolf et al., 2010). The gain is probably highest where efficient bioenergy is constrained by a remote, seasonal or diffuse biomass resource (Shackley et al., 2012). The benefit of pyrolysis–biochar systems (PBS) is increased considerably if allowance is made for the indirect effects of using biochar *via* the soil. These effects include increased crop and biomass production and decreased N_2O and CH_4 emissions. Realising the mitigation potential for biochar technologies will be constrained by the need for sustainable feedstock acquisition, competing biomass use options are an important influence of the production process on biochar properties. Considering sustainable feedstock production and targeting biochar deployment on less fertile land, Woolf et al. (2010) calculated maximum global abatement of 6.6 Gt CO_2eq/yr from 2.27 Gt biomass C. Allowing for competition for virgin non-waste biomass the value was lower (3.67 Gt CO_2eq/yr from 1.01 Gt biomass C), accruing 240-480 Gt CO_2eq abatement within 100 years. Meta-analysis shows that in experimental situations crop productivity has, on average, been enhanced by ca. 15% near-term, but with a wide range of effects (Jeffery et al., 2011; Biederman and Harpole, 2013). This range is probably explained by the nature and extent of pre-existing soil constraints. The Woolf et al. (2010) analysis accordingly assumed crop yield increases of 0–90% (relative). Relaxing this assumption by one-half decreased projected abatement by 10%. Decreasing an assumed 25% suppression on soil N_2O flux by the same proportion had a smaller impact. Beneficial interactions of biochar and the soil N cycle are beginning to be understood with effects on mineralisation, nitrification, denitrification, immobilisation and adsorption persisting at least for days and months after biochar addition (Nelissen et al., 2012; Clough et al., 2013). Although the often large suppression of soil N_2O flux observed under laboratory conditions can be increasingly explained (Cayuela et al., 2013), this effect is not yet predictable and there has been only limited validation of N_2O suppression by biochar in planted field soils (Liu et al., 2012; Van Zwieten et al., 2013) or over longer timeframes (Spokas, 2013). The potential to gain enhanced mitigation using biochar by tackling gaseous emissions from manures and fertilisers before and after application to soil are less well explored (Steiner et al., 2010; Angst et al., 2013). The abatement potential for PBS remains most sensitive to the absolute stability of the C stored in biochar. Estimates of ‘half-life’ have been inferred from wildfire charcoal (Lehmann, 2007) or extrapolated from direct short-term observation. These give values that range from <50 to >10,000 years, but predominantly between 100–1000 years (Spokas, 2010; Singh et al., 2012). Nonetheless, the assumption made by Woolf et al., (2010) for the proportion of biochar C that is stable long-term (85%) is subject to refinement and field validation. Demonstration of the equipment and infrastructure required for effective use of energy products from biomass pyrolysis is still limited, especially across large and small unit scales. Preliminary analyses shows, however, that the break even cost of biochar production is likely to be location and feedstock specific (Shackley et al., 2012; Field et al., 2013). Until economic incentives are established for the stabilisation of C, biochar adoption will depends on predictable, positive effects on crop production. This requires more research on the use of biochar as a regular low-dose soil input, rather than single applications at rates >10t/ha which have so far been the norm (Sohi, 2012). Product standards are also required, to ensure that biochar is produced in a way that does not create or conserve problematic concentrations of toxic contaminants, and to support regulated deployment strategies (IBI Biochar, 2012; Downie et al., 2012).

Box 11.4 **Aquaculture**

Aquaculture is defined as the farming of fish, shellfish, and aquatic plants (Hu et al., 2013). Although it is an ancient practice in some parts of world, this sector of the food system is growing rapidly. Since the mid-1970s, total aquaculture production has grown at an average rate of 8.3% per year (1970−2008; Hu et al., 2013). The estimated aquaculture production in 2009 was 55.10 Mt, which accounts for approximately 47% of all the fish consumed by humans (Hu et al., 2013). The sector is diverse, being dominated by shellfish and herbivorous and omnivorous pond fish, either entirely or partly utilizing natural productivity, but globalizing trade and favourable economic conditions are driving intensive farming at larger scales (Bostock et al., 2010). Potential impacts of aquaculture, in terms emissions of N_2O, have recently been considered (Williams and Crutzen, 2010; Hu et al., 2012). Global N_2O emissions from aquaculture in 2009 were estimated to be 93 kt N_2O-N (~43 Mt CO_2eq), and will increase to 383 kt N_2O-N (~178 Mt CO_2eq) by 2030, which could account for 5.7% of anthropogenic N_2O−N emissions if aquaculture continues to grow at the present growth rate (~7.1%/yr; Hu et al., 2012).

Some studies have focused on rice-fish farming which is a practice associated with wet rice cultivation in south-east Asia, providing protein, especially for subsistence oriented farmers Bhattacharyya et al., 2013). Cultivation of fish along with rice increases emissions of CH_4 (Frei et al., 2007; Bhattacharyya et al., 2013), but decreases N_2O emissions, irrespective of the fish species used (Datta et al., 2009; Bhattacharyya et al., 2013). Although rice–fish farming systems might be globally important in terms of climate change, they are also relevant for local economy, food security and efficient water use (shared water), which makes it difficult to design appropriate mitigation measures, because of the trade-offs between mitigation measures and rice and fish production (Datta et al., 2009; Bhattacharyya et al., 2013). Feeding rate and dissolved oxygen (DO) concentration could affect N_2O emissions from aquaculture systems significantly, and nitrification and denitrification processes were equally responsible for the emissions of N_2O in these systems. Measures to control N_2O from aquaculture are described by (Hu et al., 2012), and include the maintenance of optimal operating conditions of the system, such as appropriate pH and temperature, sufficient DO and good quality feed. Additionally, two potential ways to minimize N_2O emissions from aquaculture systems include: Aquaponic Aquaculture (polyculture consisting of fish tanks [aquaculture] and plants which are cultivated in the same water cycle [hydroponic]), and Bioflocs Technology (BFT) Aquaculture (which involves the development and control of heterotrophic bacteria in flocs within the fish culture component), where the growth of heterotrophic bacteria is stimulated, leading to nitrogen uptake; (Hu et al., 2012).

Box 11.5 Bioenergy

Bioenergy deployment offers significant potential for climate change mitigation, but also carries considerable risks. The IPCC's Special Report on Renewable Energy Sources and Climate Change Mitigation (SRREN), suggested potential bioenergy deployment levels to be between 100-300EJ. This assessment agrees on a technical bioenergy potential of around 100 EJ, and possibly 300 EJ and higher. Integrated models project between 15-245 EJ/yr deployment in 2050, excluding traditional bioenergy. Achieving high deployment levels would require, amongst others, extensive use of agricultural residues and second-generation biofuels to mitigate adverse impacts on land use and food production, and the co-processing of biomass with coal or natural gas with CCS to produce low net GHG-emitting transportation fuels and/or electricity. Integration of crucial sectoral research (albedo effects, evaporation, counterfactual land carbon sink assumptions) into transformation pathways research, and exploration of risks of imperfect policy settings (for example, in absence of a global CO_2 price on land carbon) is subject of further research (see Sections 11.9, 11.13.2 and 11.13.4). Small-scale bioenergy systems aimed at meeting rural energy needs synergistically provide mitigation and energy access benefits. Decentralized deployment of biomass for energy, in combination with improved cookstoves, biogas, and small-scale biopower, could improve livelihoods and health of around 3 billion people. Both mitigation potential and sustainability hinges crucially on the protection of land carbon (high density carbon ecosystems), careful fertilizer application, interaction with food markets, and good land and water management. Sustainability and livelihood concerns might constrain beneficial deployment of dedicated biomass plantations to lower values. (see Sections 11.13.3, 11.13.5, 11.13.7)

Lifecycle assessments for bioenergy options demonstrate a plethora of pathways, site-specific conditions and technologies produce a wide range of climate-relevant effects. Specifically, land-use change emissions, nitrous oxide emissions from soil and fertilizers, co-products, process design and process fuel use, end-use technology, and reference system can all influence the total attributional lifecycle emissions of bioenergy use. The large variance for specific pathways points to the importance of management decisions in reducing the lifecycle emissions of bioenergy use. The total marginal global warming impact of bioenergy can only be evaluated in a comprehensive setting that also addresses equilibrium effects, e.g. indirect land-use change emissions, actual fossil fuel substitution and other effects. Structural uncertainty in modeling decisions renders such evaluation exercises uncertain. Available data suggest a differentiation between options that offer low lifecycle emissions under good land-use management (e.g. sugarcane, Miscanthus, and fast-growing tree species) and those that are unlikely to contribute to climate change mitigation (e.g. corn and soybean), pending new insights from more comprehensive consequential analyses (see Sections 8.7, 11.13.4)

Coupling bioenergy and CCS (BECCS) has attracted particular attention since AR4 because it offers the prospect of negative emissions. Until 2050 the economic potential is estimated to be between 2-10 $GtCO_2$ per year. Some climate stabilization scenarios see considerable higher deployment towards the end of the century, even in some 580-650ppm scenarios, operating under different time scales, socioeconomic assumptions, technology portfolios, CO_2 prices, and interpreting BECCS as part of an overall mitigation framework. Technological challenges and potential risks of BECCS include those associated with the provision of the biomass feedstock as well as with the capture, transport and long-term underground storage of CO_2. BECCS faces large challenges in financing and currently no such plants have been built and tested at scale (see Sections 7.5.5., 7.9, 11.13.3)

Land-demand and livelihoods are often affected by bioenergy deployment. Land demand for bioenergy depends on (1) the share of bioenergy derived from wastes and residues; (2) the extent to which bioenergy production can be integrated with food and fibre production, and conservation to minimize land-use competition; (3) the extent to which bioenergy can be grown on areas with little current production; and (4) the quantity of dedicated energy crops and their yields. Considerations of trade-offs with water, land and biodiversity are crucial to avoid adverse effects. The total impact on livelihood and distributional consequences depends on global market factors, impacting income and income-related food-security, and site-specific factors such as land tenure and social dimensions. The often site-specific effects of bioenergy deployment on livelihoods have not yet been comprehensively evaluated (see Section 11.13.7).

11.3.2 Mitigation effectiveness (non-permanence: saturation, human and natural impacts, displacement)

Since carbon sequestration in soil and vegetation and the retention of existing carbon stocks forms a significant component of the mitigation potential in the AFOLU sector, this section considers the factors affecting this strategy compared to avoided GHG emissions.

Non-permanence / reversibility. Reversals are the release of previously sequestered carbon, which negates some or all of the benefits from sequestration that has occurred in previous years. This issue is sometimes referred to as “non-permanence” (Smith, 2005). Various types of carbon sinks (e.g. afforestation/reforestation, agricultural soil C) have an inherent risk of future reversals.

Certain types of mitigation activities (e.g. avoided N_2O from fertilizer, emission reductions from changed diet patterns or reduced food-chain losses) are effectively permanent since the emissions, once avoided, cannot be re-emitted. The same applies to the use of bioenergy to displace fossil-fuel emissions (see 11.13) or the use of biomass-based products to displace more emissions-intensive products (e.g. wood in place of concrete or steel) in construction.

Reversals may be caused by natural events that affect yields / growth. In some cases (e.g. frost damage, pest infestation or fire; Reichstein et al., 2013), these effects may be temporary or short-term. Although these events will affect the annual increment of C sequestration, they may not result in a permanent decline in carbon stocks. In other cases, such as stand replacing forest fires, insect or disease outbreaks or drought, the declines may be more profound. Although a substantial loss of aboveground stored carbon could occur following a wildfire, whether this represents a loss depends on what happens following the fire and whether the forest recovers, or changes to a lower carbon storage state (see Box 11.2). Similarly, some systems are naturally adapted to fire and carbon stocks will recover following fire, whereas in other cases the fire results in a change to a system with a lower carbon stock (e.g., Brown and Johnstone, 2011). For a period of time following fire (or other disruptive event) the stock of carbon will be less than that before the fire. Similarly, emissions of non-CO_2 gases also need to be considered.

The permanence of the AFOLU carbon stock relates to the longevity of the stock, i.e. how long the increased carbon stock remains in the soil or vegetation. This is linked to consideration of the reversibility of the increased carbon stock (Smith et al., 2005), as discussed in 11.5.2.

Saturation. Substitution of fossil fuel and material with biomass, and energy intensive building materials with wood can continue in perpetuity. In contrast, it is often considered that carbon sequestration in soils (Guldea et al., 2008) or vegetation cannot continue indefinitely. The carbon stored in soils and vegetation reaches a new equilibrium (as the trees mature or as the soil carbon stock saturates). As the soils / vegetation approach the new equilibrium, the annual removal (sometimes referred to as the sink strength) decreases until it becomes zero at equilibrium. This process is called saturation (Smith et al., 2005; Körner, 2006, 2009; Johnston et al., 2009c), and the uncertainty associated with saturation has been estimated (Kim and McCarl, 2009). An alternative view is that saturation does not occur, with studies from old-growth forests, for example, showing that they can continue to sequester C in soil and dead organic matter even if net living biomass increment is near zero (e.g., Luyssaert et al., 2008). Peatlands are unlikely to saturate in carbon storage, but the rate of C uptake may be very slow (see Box 11.1).

Human and natural impacts. Soil and vegetation carbon sinks can be impacted upon by direct human induced, indirect human induced and natural changes (Smith, 2005). All of the mitigation practices discussed in section 11.3.1 arise from direct human-induced impacts (deliberate management). Both sink processes and carbon stocks can be affected by natural factors such as soil and hydrological conditions. Indirect human-induced changes can impact carbon sinks and are influenced by human activity, but are not directly related to the management of that piece of land; examples include climate change and atmospheric nitrogen deposition. For some tree species, rising concentrations of tropospheric ozone caused by human activities may counteract the effects of increased atmospheric CO_2 or N deposition on tree growth (Sitch et al., 2007; Matyssek et al., 2010). Natural changes that threaten to impact the efficacy of mitigation measures are discussed in 11.5.

Displacement / leakage. Displacement / leakage arises from a change in land-use or land management that causes a positive or negative change in emissions elsewhere. This can occur within or across national boundaries, and the efficacy of mitigation practices must consider the leakage implications. For example, if reducing emissions in one place leads to increased emissions elsewhere, no net reduction occurs; the emissions are simply displaced (Powlson et al., 2011; Kastner et al., 2011b; a), however this assumes a one to one correspondence. (Murray et al., 2004) estimated the leakage from different forest carbon programs and this varied from <10% to >90% depending on the nature of the activity. (West et al., 2010a) examined the impact of displaced activities in different geographic contexts; for example land clearing in the tropics will release twice the carbon, but only produce half the crop yield of temperate areas. Indirect land use change (iLUC) is an important component to consider for displaced emissions and assessments of this are an emerging area. iLUC is discussed further in 11.4 and in relation to bioenergy in 11.13.

The timing of mitigation benefits from actions (e.g. bioenergy, forest management, forest products use/storage) can vary as a result both of the nature of the activity itself (e.g. from the temporal pattern of soil or forest sequestration compared to biomass substitution), and rates of adoption. Timing thus needs to be considered when judging the effectiveness of a mitigation action. Cherubini et al. (2012) modelled the impact of timing of benefits for three different wood applications (fuel, non-structural panels and housing construction materials) and showed that the options provide mitigation over different time-frames, and thus have different impacts on CO_2 concentrations and radiative forcing. The temporal pattern of emissions and removals is especially important in mitigating emissions of short-lived gases through carbon sequestration (Lauder et al., 2013).

Additionality: Another consideration for gauging the effectiveness of mitigation is determining whether the activity would have occurred anyway, with this encompassed in the concept of “additionality” (see Glossary).

Impacts of climate change: An area of emerging activity is predicting the likely impacts of climate change on mitigation potential, both in terms of impacts on existing carbon stocks, but also on the rates of carbon sequestration. This is discussed further in 11.5.

11.4 Infrastructure and systemic perspectives

Only supply-side mitigation options are considered in 11.3. In this section, we consider infrastructure and systemic perspectives, which include potential demand-side mitigation options in the AFOLU sector. Since infrastructure is a minor issue in AFOLU compared to energy end use sectors, this section focusses on systemic perspectives.

11.4.1 Land: a complex, integrated system

Mitigation in the AFOLU sector is embedded in the complex interactions between socioeconomic and natural factors simultaneously affecting land systems (Turner et al., 2007). Land is used for a variety of purposes, including housing and infrastructure (Chapter 12), production of goods and services through agriculture, aquaculture and forestry and absorption or deposition of wastes and emissions (Dunlap and Catton, Jr., 2002). Agriculture and forestry are important for rural livelihoods and employment (Coelho et al., 2012), while aquaculture and fisheries can be regionally important (FAO, 2012). More than half of the planet’s total land area (134 Mkm^2) is used for urban and infrastructure land, agriculture and forestry. Less than one quarter shows relatively minor signs of direct human use (Erb et al., 2007; Ellis et al., 2010; Figure 11.9). Some of the latter areas are inhabited by indigenous populations, which depend on the land for the supply of vitally important resources (Read et al., 2010).

Land use change is a pervasive driver of global environmental change (Foley et al., 2005, 2011). From 1950 to 2005, farmland (cropland plus pasture) increased from 28% to 38% of the global land area excluding ice sheets and inland waters (Hurtt et al., 2011). The growth of farmland area (+33%) was lower than that of population, food production and GDP due to increases in yields and biomass conversion efficiency (Krausmann et al., 2012). In the year 2000, almost one quarter of the global terrestrial net primary production (one third of the aboveground part) was ‘appropriated’ by humans. This means that it was either lost because the net primary productivity (the biomass production of green plants, abbreviated NPP) of agro-ecosystems or urban areas was lower than that of the vegetation they replaced or it was harvested for human purposes, destroyed during harvest or burned in human-induced fires (Imhoff et al., 2004; Haberl et al., 2007). The fraction of terrestrial NPP appropriated by humans doubled in the last century (Krausmann et al., 2013), exemplifying the increasing human domination of terrestrial ecosystems (Ellis et al., 2010). Growth trajectories of the use of food, energy and other land-based resources, as well as patterns of urbanization and infrastructure development are influenced by increasing population and GDP, as well as the on-going agrarian-industrial transition (Haberl et al., 2011b; Kastner et al., 2012).

Growing resource use and land demand for biodiversity conservation and carbon sequestration (Soares-Filho et al., 2010), result in increasing competition for land (Harvey and Pilgrim, 2011); 11.4.2). Influencing ongoing transitions in resource use is a major challenge (Fischer-Kowalski, 2011; WBGU, 2011). Changes in cities, e.g. in terms of infrastructure, governance and demand, can play a major role in this respect (Seto et al., 2012b; Seitzinger et al., 2012; Chapter 12).

Many GHG mitigation activities in the AFOLU sector affect land use or land cover and, therefore, have socioeconomic as well as ecological consequences, e.g. on food security, livelihoods, ecosystem services or emissions (11.1, 11.4.5, 11.7). Feedbacks involved in implementing mitigation in AFOLU may influence different, sometimes conflicting, social, institutional, economic and environmental goals (Madlener et al., 2006). Climate change mitigation in the AFOLU sector faces a complex set of interrelated challenges (11.4.5; 11.7):

- Full GHG impacts, including those from feedbacks (e.g. 'indirect' land use change) or leakage, are often difficult to determine (Searchinger et al., 2008b).
- Feedbacks between GHG reduction and other important objectives such as provision of livelihoods and sufficient food or the maintenance of ecosystem services and biodiversity are not completely understood.
- Maximizing synergies and minimizing negative effects involves multi-dimensional optimization problems involving various social, economic and ecological criteria or conflicts of interest between different social groups (Martinez-Alier, 2002).
- Changes in land use and ecosystems are scale-dependent and may proceed at different speeds, or perhaps even move in different directions, at different scales.

11.4.2 Mitigation in AFOLU – feedbacks with land use competition

Driven by economic and population growth, increased demand for food and bioenergy as well as land demand for conservation and urbanisation (e.g. aboveground biomass carbon losses associated with land-clearing from new urban areas in the pan-tropics are estimated to be 5% of the tropical deforestation and land-use-change emissions, Seto et al., 2012a, 12.2), competition for land is expected to intensify (Smith et al., 2010; Woods et al., 2010). Maximization of one output or service (e.g. crops) often excludes, or at least negatively affects, others (e.g. conservation; Phalan et al., 2011). Mitigation in the AFOLU sector may affect land use competition. Reduced demand for AFOLU products generally decreases inputs (fertilizer, energy, machinery) and land demand. The ecological feedbacks of demand-side options are mostly beneficial since they reduce competition for land and water (Smith et al., 2013b).

Some supply-side options, though not all, may intensify competition for land and other resources. Based on Figure 11.9 one may distinguish three cases:

- **Optimization of biomass-flow cascades;** that is, increased use of residues and by-products, recycling of biogenic materials and energetic use of wastes (WBGU, 2009). Such options increase resource use efficiency and may reduce competition, but there may also be trade-offs. For example, using crop residues for bioenergy or roughage supply may leave less C and nutrients on cropland, reduce soil quality and C storage in soils and increase the risk of losses of carbon through soil erosion. Residues are also often used as forage, particularly in the tropics. Forest residues are currently also used for other purposes, e.g. chipboard manufacture, pulp and paper production (González-Estrada et al., 2008; Blanco-Canqui and Lal, 2009; Muller, 2009; Ceschia et al., 2010).

- **Increases in yields** of cropland (Burney et al., 2010; Foley et al., 2011; Tilman et al., 2011; Mueller et al., 2012; Lobell et al., 2013), grazing land or forestry and improved livestock feeding efficiency (Steinfeld et al., 2010; Thornton and Herrero, 2010) can reduce land competition if yield increases relative to any additional inputs and the emission intensity (i.e. GHG emissions per unit of product) decreases. This may result in trade-offs with other ecological, social and economic costs (IAASTD, 2009) although these can to some extent be mitigated if intensification is sustainable (Tilman et al., 2011). Another caveat is that increases in yields may result in rebound effects that increase consumption (Lambin and Meyfroidt, 2011; Erb, 2012) or provide incentives to farm more land (Matson and Vitousek, 2006), and hence may fail to spare land (11.10).
- **Land-demanding options** reduce GHG emissions by harnessing the potential of the land for either C sequestration or growing energy crops (including food crops used as feedstocks for bioenergy production). These options result in competition for land (and sometimes other resources such as water) that may have substantial social, economic and ecological effects (positive or negative; UNEP, 2009; WBGU, 2009; Chum et al., 2011; Coelho et al., 2012). Such options may increase pressures on ecosystems (e.g. forests) and GHG emissions related to direct and indirect land use change, contribute to price increases of agricultural products, or negatively affect livelihoods of rural populations. These possible impacts need to be balanced against possible positive effects such as GHG reduction, improved water quality (Townsend et al., 2012), restoration of degraded land (Harper et al., 2007), biodiversity protection (Swingland et al., 2002) and job creation (Chum et al., 2011; Coelho et al., 2012).

Therefore, an integrated energy/agriculture/land-use approach for mitigation in AFOLU can help to optimize synergies and mitigate negative effects (Popp et al., 2011b; Creutzig et al., 2012; Smith, 2012; Smith et al., 2013b).

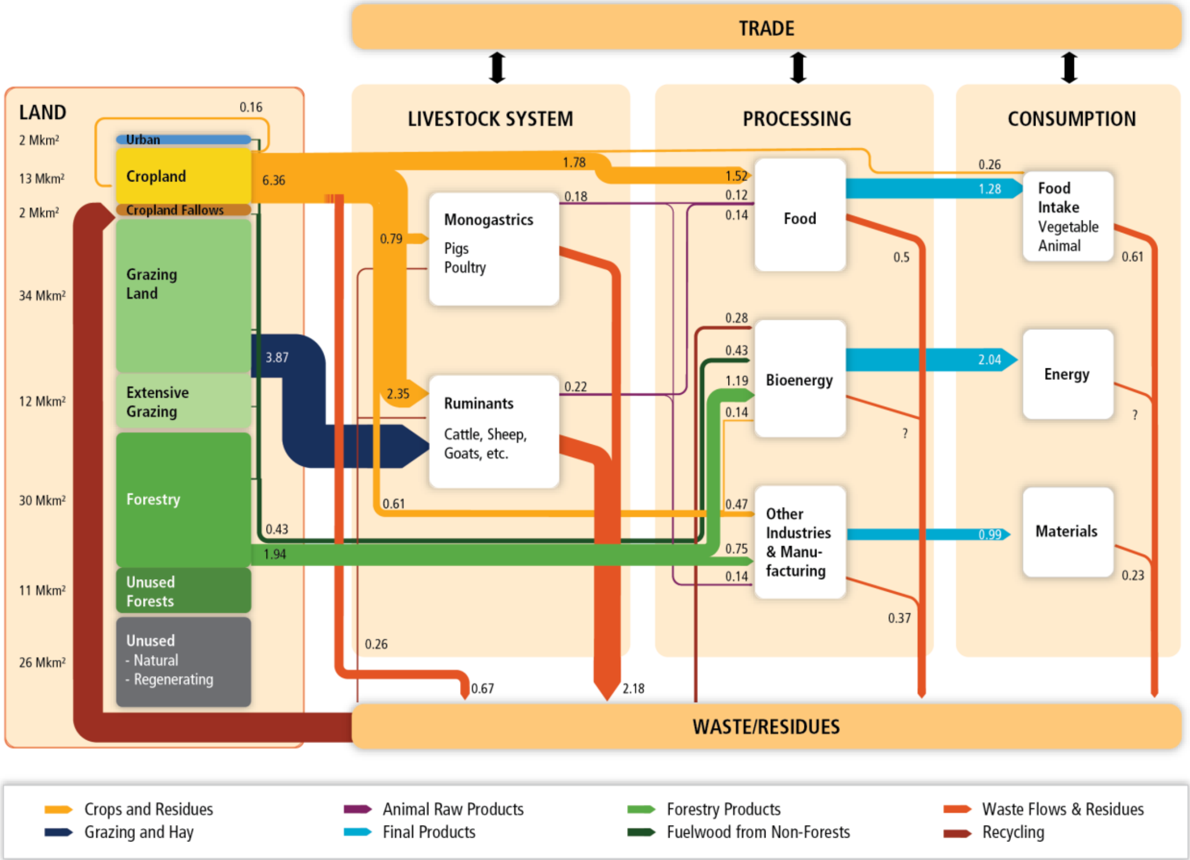

Figure 11.9. Global land use and biomass flows arising from human economic activity in 2000 from the cradle to the grave. Values in Gt dry matter biomass/yr. Figure source: (Smith et al., 2013b). If a source reported biomass flows in energy units, the numbers were converted to dry matter assuming a gross energy value of 18.5 MJ/kg. The difference between inputs and outputs in the consumption compartment is assumed to be released to the atmosphere (respiration, combustion); small differences may result from rounding. Note that data sources a) area: (Erb et al., 2007; Schneider et al., 2009; FAO, 2010a); b) biomass flows: (Wirsenius, 2003; Sims et al., 2006; Krausmann et al., 2008; FAOSTAT, 2012; Kummu et al., 2012) are incomplete; more research is needed to close data gaps between different statistical sources such as agricultural, forestry and energy statistics (11.11). "Unused forests" are pristine forests not harvested or otherwise used.

11.4.3 Demand-side options for reducing GHG emissions from AFOLU

Some changes in demand for food and fibre can reduce GHG emissions in the production chain (Table 11.3) through (i) a switch to the consumption of products with higher GHG emissions in the process chain to products with lower GHG emissions and (ii) by making land available for other GHG reduction activities e.g. afforestation or bioenergy (11.4.4). Food demand change is a sensitive issue due to the prevalence of hunger, malnutrition and the lack of food security in many regions (Godfray et al., 2010). Sufficient production of, and equitable access to, food are both critical for food security (Misselhorn et al., 2012). GHG emissions may be reduced through changes in food demand without jeopardizing health and well-being by (1) reducing losses and wastes of food in the supply chain as well as during final consumption and (2) changing diets towards less GHG intensive food, e.g. substitution of animal products with plant-based food, while quantitatively and qualitatively maintaining adequate protein content, in regions with high animal product consumption, and (3) reduction of overconsumption in regions where this is prevalent. Substituting plant based diets for animal based diets is complex since, in many circumstances, livestock can be fed on plants not suitable for human consumption or growing on land with high soil carbon stocks not suitable for cropping; hence, food production by grazing animals contributes to food security in many regions of the world (Wirsenius, 2003; Gill et al., 2010).

Table 11.3: Overview of demand-side mitigation options in the AFOLU sector

Measure	Description	References
Reduced losses in the food supply chain	Reduced losses in the food supply chain and in final consumption reduces energy use and GHG emissions from agriculture, transport, storage and distribution, and reduce land demand.	(Godfray et al., 2010; Gustavsson et al., 2011), see text.
Changes in human diets towards less emission-intensive products	Where appropriate, reduced consumption of food items with high greenhouse gas emissions per unit of product, to those with low GHG products can reduce GHG emissions. Such demand changes can reduce energy inputs in the supply chain and reduces land demand.	(Stehfest et al., 2009; FAO, 2011), see text
Demand-side options related to wood and forestry	Wood harvest in forests releases GHG and at least temporarily reduces forest C stocks. Conservation of wood (products) through more efficient use or replacement with recycled materials and replacing wood from illegal logging or destructive harvest with wood from certified sustainable forestry (section 11.10) can save GHG emissions. Substitution of wood for non-renewable resources can reduce GHG emissions, e.g. when wood is substituted for emission-intensive materials such as aluminium, steel or concrete in buildings. Integrated optimization of C stocks in forests and in long-lived products, as well as the use of by-products and wastes for energy, can deliver the highest GHG benefits.	(Gustavsson et al., 2006; Werner et al., 2010; Ingerson, 2011), see text.

Reductions of losses in the food supply chain – Globally, rough estimates suggest that ~30-40% of all food produced is lost in the supply chain from harvest to consumption (Godfray et al., 2010). Energy embodied in wasted food is estimated at ~36 EJ/yr (FAO, 2011). In developing countries, up to 40% is lost on farm or during distribution due to poor storage, distribution and conservation technologies and procedures. In developed countries, losses on farm or during distribution are smaller, but the same amount is lost or wasted in service sectors and at the consumer level (Foley et al., 2005; Godfray et al., 2010; Parfitt et al., 2010; Gustavsson et al., 2011; Hodges et al., 2011). However, uncertainties related to losses in the food supply chain are large and more research is needed.

Not all losses are (potentially) avoidable because losses in households also include parts of products normally not deemed edible (e.g. peels of some fruits and vegetables). According to (Parfitt et al., 2010), in the UK, 18% of the food waste is unavoidable, 18% is potentially avoidable and 64% is avoidable. Data for Austria, Netherlands, Turkey, the UK and the USA, derived with a variety of methods, show that food wastes at the household level in industrialized countries are 150-300 kg per household per year (Parfitt et al., 2010). According to a top-down mass-flow modelling study based on FAO commodity balances completely covering the food supply chain, but excluding non-edible fractions, food loss values range from 120-170 kg/cap/yr in Sub-Saharan Africa to 280-300 kg/cap/yr in Europe and North-America. Losses ranging from 20% in Sub-Saharan Africa to more than 30% in the industrialized countries were calculated (Gustavsson et al., 2011).

A range of options exist to reduce wastes and losses in the supply chain: investments into harvesting, processing and storage technologies in the developing countries, awareness raising, taxation and other incentives to reduce retail and consumer-related losses primarily in the developed countries. Different options can help to reduce losses (i.e. increase efficiency) in the supply chain and at the household level. Substantial GHG savings could be realised by saving one quarter of the wasted food according to (Gustavsson et al., 2011); see Table 11.4.

Changes in human diets – Land use and GHG effects of changing diets require widespread behavioural changes to be effective; i.e. a strong deviation from current trajectories (increasing demand for food, in particular for animal products). Cultural, socioeconomic and behavioural aspects of implementation are discussed in 11.4.5 and 11.7.

Studies based on Life-Cycle Assessment (LCA) methods show substantially lower GHG emissions for most plant-based food than for animal products (Carlsson-Kanyama and González, 2009; Pathak et al., 2010; Bellarby et al., 2012; Berners-Lee et al., 2012), although there are exceptions, e.g. vegetables grown in heated greenhouses or transported by airfreight (Carlsson-Kanyama and González, 2009). A comparison of three meals served in Sweden with similar energy and protein content based on (1) soy, wheat, carrots and apples, (2) pork, potatoes, green beans and oranges, and (3) beef, rice, cooked frozen vegetables and tropical fruits revealed GHG emissions of 0.42 kgCO_2eq for the first option, 1.3 kgCO_2eq for the second and 4.7 kgCO_2eq for the third, i.e. a factor of >10 difference (Carlsson-Kanyama and González, 2009). Most LCA studies quoted here use attributional LCA; differences to results from consequential LCA (see Annex II “Methodology and Metrics”) are generally not large enough to reverse the picture (Thomassen et al., 2008). GHG benefits of plant-based food over animal products hold when compared per unit of protein (González et al., 2011). In addition to plant-based foods having lower emissions than animal based ones, GHG emissions of livestock products also vary considerably; emissions per unit of protein are highest for beef and lower for pork, chicken meat, eggs and dairy products (de Vries and de Boer, 2010) due to their feed and land use intensities. Figure 11.10 presents a comparison between milk and beef for different production systems and regions of the world (Herrero et al., 2013). Beef production can use up to five times more biomass for producing 1 kg of animal protein than dairy. Emissions intensities for the same livestock product also vary largely between different regions of the world due to differences in agro-ecology, diet quality and intensity of production (Herrero et al., 2013). In overall terms, Europe and North America have lower emissions intensities per kg of protein than Africa, Asia and Latin America. This shows that the highest potential for improving emissions intensities lies in developing countries, if intensification strategies can be matched to local resources and contexts.

Studies based on integrated modelling show that changes in diets strongly affect future GHG emissions from food production (Stehfest et al., 2009; Popp et al., 2010; Davidson, 2012). Popp et al. (2010) estimated that agricultural non-CO_2 emissions (CH_4 and N_2O) would triple by 2055 to 15.3 Gt CO_2eq/yr if current dietary trends and population growth were to continue. Technical mitigation options on the supply side, such as improved cropland or livestock management, alone could reduce that value to 9.8 Gt CO_2eq/yr, whereas emissions were reduced to 4.3 Gt CO_2eq/yr in a ‘decreased livestock product’ scenario and to 2.5 Gt CO_2eq/yr if both technical mitigation and dietary change were assumed. Hence, the potential to reduce GHG emissions through changes in consumption was found to be substantially higher than that of technical GHG mitigation options. Stehfest et al., (2009) evaluated effects of dietary changes on CO_2 (including C sources/sinks of ecosystems), CH_4 and N_2O emissions. In a ‘business as usual’ scenario largely based on (FAO, 2006a), total GHG emissions were projected to reach 11.9 Gt CO_2eq/yr in 2050. The following changes were evaluated: no ruminant meat, no meat, and a diet without any animal products. Changed diets resulted in GHG emission savings of 34-64% compared to the ‘business as usual’ scenario; a switch to a ‘healthy diet’ recommended by the Harvard Medical School would save 4.3 Gt CO_2eq/yr (-36%). Adoption of the ‘healthy diet’ (which includes a meat, fish and egg consumption of 90 g/cap/day) would reduce global GHG abatement costs to reach a 450 ppm CO_2eq concentration target by ~50% compared to the reference case (Stehfest et al., 2009). The analysis assumed nutritionally sufficient diets; reduced supply of animal protein was compensated by plant products (soy, pulses etc.). Considerable cultural and social barriers against a widespread adoption of dietary changes to low-GHG food may be expected (Davidson, 2012; Smith et al., 2013b; 11.4.5).

A limitation of food-related LCA studies is that they have so far seldom considered the emissions resulting from land-use change induced by changing patterns of food production (Bellarby et al., 2012). A recent study (Schmidinger and Stehfest, 2012) found that cropland and pastures required for the production of beef, lamb, calf, pork, chicken and milk could annually sequester an amount of carbon equivalent to 30%-470% of the GHG emissions usually considered in LCA of food products if the land were to be reforested. Land-related GHG costs differ greatly between products and depend on the time horizon (30-100 yr) assumed (Schmidinger and Stehfest, 2012). If cattle production contributes to tropical deforestation (Zaks et al., 2009; Bustamante et al., 2012; Houghton et al., 2012), land-use related GHG emissions are particularly high (Cederberg et al., 2011). These findings underline the importance of diets for GHG emissions in the food supply chain (Garnett, 2011; Bellarby et al., 2012). A potential co-benefit is a reduction in diet-related health risks in regions where overconsumption of animal products is prevalent (McMichael et al., 2007).

Demand-side options related to wood and forestry – A comprehensive global, long-term dataset on carbon stocks in long-lived wood products in use (excluding landfills) shows an increase from approximately 2.2 GtC in 1900 to 6.9 GtC in 2008 (Lauk et al., 2012). Per-capita, carbon stored in wood products amounted to ~1.4 t C / capita in 1900 and ~1.0 t C / capita in 2008. The net yearly accumulation of long-lived wood products in use varied between 35 and 91 MtC / yr in the period 1960-2008 (Lauk et al., 2012). The yearly accumulation of C in products and landfills was ~200 MtC / yr in the period 1990-2008 (Pan et al., 2011). If more long-lived wood products were used, C sequestration and GHG mitigation could be enhanced.

Increased wood use does not reduce GHG emissions under all circumstances because wood harvest reduces the amount of carbon stored in the forest, at least temporarily, and increases in wood harvest levels may result in reduced long-term carbon storage in forests (Ingerson, 2011; Böttcher et al., 2012; Holtsmark, 2012; Lamers and Junginger, 2013a) . Reducing wood consumption, e.g. through paper recycling, can reduce GHG emissions (Acuff and Kaffine, 2013), as may the use of wood from sustainable forestry in place of emission-intensive materials such as concrete, steel or aluminium. Recent studies suggest that, where technically possible, substitution of wood from sustainably managed forests for non-wood materials in the construction sector (concrete, steel, etc.) in single family homes, apartment houses and industrial buildings, reduces GHG emissions in most cases (Werner et al., 2010; Sathre and O'Connor, 2010; Ximenes and Grant, 2013). Most of the emission reduction results from reduced production emissions, whereas the role of carbon sequestration in products is relatively small (Sathre and O'Connor, 2010). (Werner et al., 2010) show that GHG benefits are highest when wood is primarily used for long-lived products, the lifetime of products is maximized, and energy use of woody biomass is focused on by-products, wood wastes and end of life cycle use of long lived wood products.

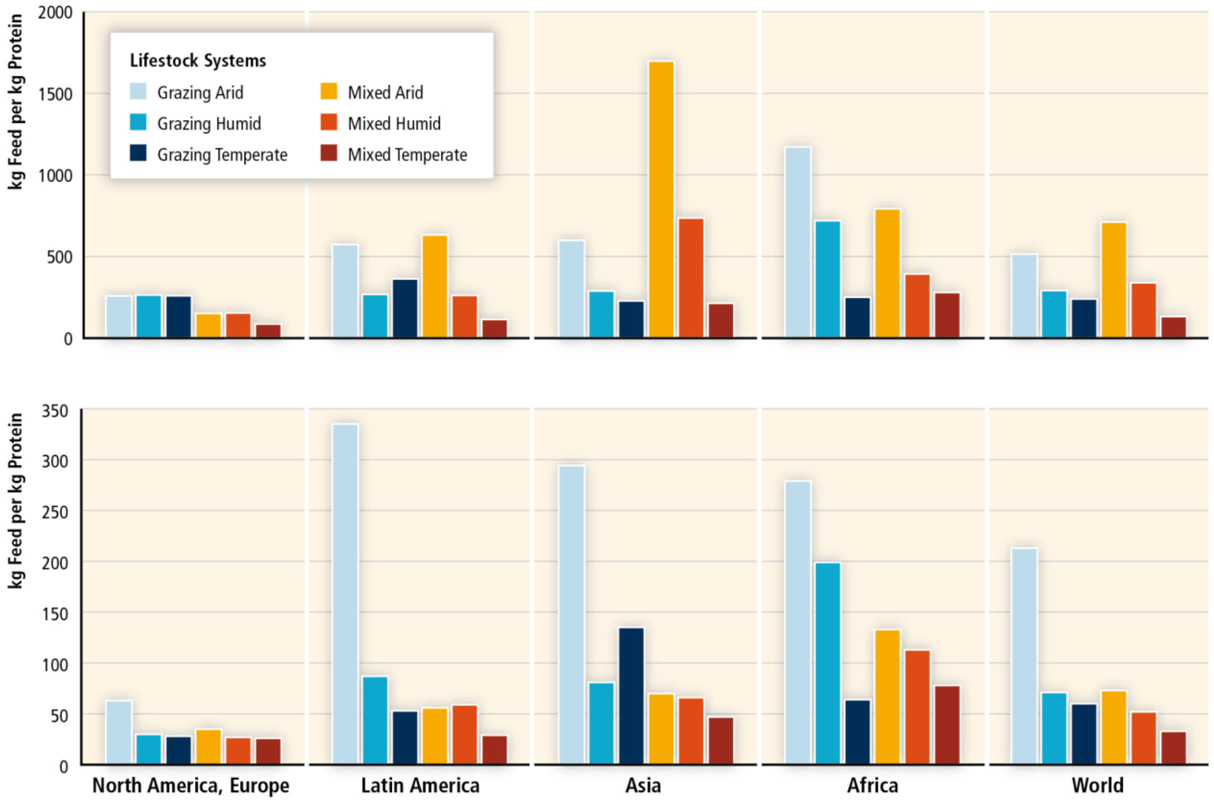

Figure 11.10. Biomass use efficiencies for the production of edible protein from (A) beef and (B) milk for different production systems and regions of the world (Herrero et al., 2013).

11.4.4 Feedbacks of changes in land demand

Mitigation options in the AFOLU sector, including options such as biomass production for energy, are highly interdependent due to their direct and indirect impacts on land demand. Indirect interrelationships, mediated *via* area demand for food production, which in turn affects the area available for other purposes, are difficult to quantify and require systemic approaches. Table 11.4 (Smith et al., 2013b) shows the magnitude of possible feedbacks in the land system in 2050. It first reports the effect of single mitigation options compared to a reference case, and then the combined effect of all options. The reference case is similar to the (FAO, 2006a) projections for 2050 and assumes a continuation of on-going trends towards richer diets, considerably higher cropland yields (+54%) and moderately increased cropland areas (+9%). The diet change case assumes a global contract-and-converge scenario towards a nutritionally sufficient low animal product diet (8% of food calories from animal products). The yield growth case assumes that yields in 2050 are 9% higher than those in the reference case, according to the 'Global Orchestration' scenario in (MEA, 2005). The feeding efficiency case assumes on average 17% higher livestock feeding efficiencies than the reference case. The waste reduction case assumes a reduction of the losses in the food supply chain by 25% (11.4.3). The combination of all options results in a substantial reduction of cropland and grazing areas (Smith et al., 2013b), even though the individual options cannot simply be added up due to the interactions between the individual compartments.

Table 11.4 shows that demand-side options save GHG by freeing up land for bioenergy or afforestation and related C-sequestration. The effect is strong and non-linear, and more than cancels out reduced C sequestration potentials on farmland. Demand-side potentials are substantial compared to supply-side mitigation potentials (11.3), but implementation may be difficult (sections 11.7; 11.8). Estimates of GHG savings from bioenergy are subject to large uncertainties related to the assumptions regarding power plants, utilization pathway, energy crop yields, and effectiveness of sustainability criteria (see 11.4.5, 11.7 and 11.13).

Table 11.4: Changes in global land use and related GHG reduction potentials in 2050 assuming the implementation of options to increase C sequestration on farmland, and use of spared land for either biomass production for energy or afforestation. Afforestation and biomass for bioenergy are both assumed to be implemented only on spare landand are mutually exclusive (Smith et al., 2013b).

Cases	Food crop area	Livestock grazing area	C sink on farm-land*	Afforestation of spare land**,^{1}	Biomass for bioenergy on spare land**,^{2}	Total mitigation potential	Difference in mitigation from Reference case
	[Gha]		Gt CO_2eq/yr				
Reference	1.60	4.07	3.5	6.1	1.2-9.4	4.6-12.9	0
Diet change	1.38	3.87	3.2	11.0	2.1-17.0	5.3-20.2	0.7-7.3
Yield growth	1.49	4.06	3.4	7.3	1.4-11.4	4.8-14.8	0.2-1.9
Feeding efficiency	1.53	4.04	3.4	7.2	1.4-11-1	4.8-14.5	0.2-1.6
Waste reduction	1.50	3.82	3.3	10.1	1.9-15.6	5.2-18.9	0.6-6.0
Combined	1.21	3.58	2.9	16.5	3.2-25.6	6.1-28.5	1.5-15.6

* Potential for C sequestration on cropland for food production and livestock grazing land with improved soil C management. The potential C sequestration rate was derived from (Smith et al., 2008)

** Spare land is cropland or grazing land not required for food production, assuming increased but still sustainable stocking densities of livestock based on (Haberl et al., 2011a; Erb et al., 2012a).

[1] Assuming 11.8 t CO_2eq/ha/yr (Smith et al., 2000).

[2] Assumptions were as follows. High bioenergy value: short-rotation coppice or energy grass directly replaces fossil fuels, energy return on investment 1:30, dry-matter biomass yield 190 GJ/ha/yr (WBGU, 2009). Low bioenergy value: ethanol from maize replaces gasoline and reduces GHG by 45%, energy yield 75 GJ/ha/yr (Chum et al., 2011). Some energy crops may, under certain conditions, sequester C in addition to delivering bioenergy; the effect is context-specific and was not included. Whether bioenergy or afforestation is a better option to use spare land for GHG mitigation needs to be decided on a case-by-case basis.

The systemic effects of land-demanding GHG mitigation options such as bioenergy or afforestation depend not only on their own area demand, but also on land demand for food and fibre supply (Chum et al., 2011; Coelho et al., 2012; Erb et al., 2012b). In 2007, energy crops for transport fuels covered about 26.6 Mha or 1.7% of global cropland (UNEP, 2009). Assumptions on energy crop yields (see 11.13) are the main reason for the large differences in estimates of future area demand of energy crops in the next decades, which vary from <100 Mha to >1000 Mha, i.e. 7%-70% of current cropland (Sims et al., 2006; Smeets et al., 2007a; Pacca and Moreira, 2011; Coelho et al., 2012). Increased pressure on land systems may also emerge when afforestation claims land, or forest conservation restricts farmland expansion (Murtaugh and Schlax, 2009; Popp et al., 2011a).

Land-demanding mitigation options may result in feedbacks such as GHG emissions from land expansion or agricultural intensification, higher yields of food crops, higher prices of agricultural products, reduced food consumption, displacement of food production to other regions and consequent land clearing, as well as impacts on biodiversity and non-provisioning ecosystem services (Plevin et al., 2010b; Popp et al., 2012).

Restrictions to agricultural expansion due to forest conservation, increased energy crop area, afforestation and reforestation may increase costs of agricultural production and food prices. In a modeling study, conserving C-rich natural vegetation such as tropical forests was found to increase food prices by a factor of 1.75 until 2100, due to restrictions of cropland expansion, even if no growth of energy crop area was assumed (Wise et al., 2009b). Food price indices (weighted average of crop and livestock products) are estimated to increase until 2100 by 82% in Africa, 73% in Latin America and 52% in Pacific Asia if large scale bioenergy deployment is combined with strict forest conservation, compared to a reference scenario without forest conservation and bioenergy (Popp et al., 2011a). Further trade liberalisation can lead to lower costs of food, but also increases the pressure on land, especially on tropical forests (Schmitz et al., 2011).

Increased land demand for GHG mitigation can be partially compensated by higher agricultural yield per unit area (Popp et al., 2011a). While yield increases can lead to improvements in output from less land, generate better economic returns for farmers, help to reduce competition for land and alleviate environmental pressures (Burney et al., 2010; Smith et al., 2010), agricultural intensification if poorly implemented incurs economic costs (Lotze-Campen et al., 2010) and may also create social and environmental problems such as nutrient leaching, soil degradation, pesticide pollution, impact on animal welfare and many more (IAASTD, 2009). Maintaining yield growth while reducing negative environmental and social effects of agricultural intensification is, therefore, a central challenge, requiring sustainable management of natural resources as well as the increase of resource efficiency (DeFries and Rosenzweig, 2010), two componets of sustainable intensification (Garnett et al., 2013).

Additional land demand may put pressures on biodiversity, as land-use change is one of the most important drivers of biodiversity loss (Sala et al., 2000). Improperly managed large-scale agriculture (or bioenergy) may negatively affect biodiversity (Groom et al., 2008), which is a key prerequisite for the resilience of ecosystems, i.e. their ability to adapt to changes such as climate change, and to continue to deliver ecosystem services in the future (Díaz et al., 2006; Landis et al., 2008). However, implementing appropriate management, such as establishing bioenergy crops or plantations for carbon sequestration in already degraded ecosystems areas represents an opportunity where bioenergy can be used to achieve positive environmental outcomes (e.g., Hill et al., 2006; Semere and Slater, 2007; Campbell et al., 2008; Nijsen et al., 2012). Because climate change is also an important driver of biodiversity loss (Sala et al., 2000), bioenergy for climate change mitigation may also be beneficial for biodiversity if it is planned with biodiversity conservation in mind (Heller and Zavaleta, 2009; Dawson et al., 2011); see 11.13).

Trade-offs related to land demand may be reduced through multifunctional land use, i.e. the optimization of land to generate more than one product or service such as food, animal feed, energy or materials, soil protection, wastewater treatment, recreation, or nature protection (de Groot, 2006; DeFries and Rosenzweig, 2010; see 11.7). This also applies to the potential use of ponds and other small water bodies for raising fish fed with agricultural waste (Pullin et al., 2007).

11.4.5 Sustainable development and behavioural aspects

The assessment of impacts of AFOLU mitigation options on sustainable development requires an understanding of a complex multilevel system where social actors make land use decisions aimed at various development goals, one of them being GHG mitigation. Depending on the specific objectives, the beneficiaries of a particular land-use choice may differ. Thus trade-offs between global, national and local concerns and various stakeholders need to be considered (see also Section 4.3.7 and WGII, Chapter 20). The development context provides opportunities or barriers for AFOLU (May et al., 2005; Madlener et al., 2006; Smith and Trines, 2006; Smith et al., 2007; Angelsen, 2008; Howden et al., 2008; Corbera and Brown, 2008; Cotula et al., 2009; Cattaneo et al., 2010; Junginger et al., 2011; 11.8 and figure 11.11).

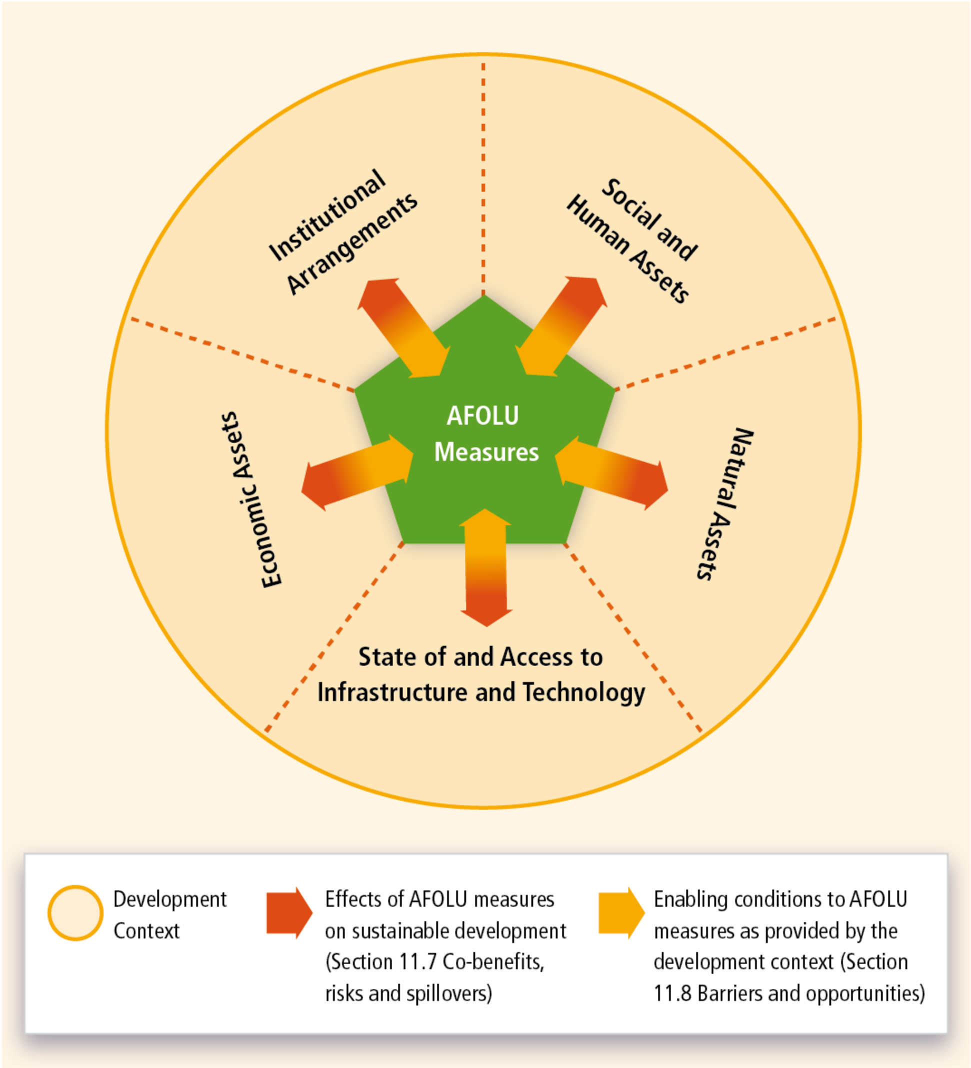

Figure 11.11. Dynamic interactions between the development context and AFOLU

Further, AFOLU measures have additional effects on development, beyond improving the GHG balance (Foley et al., 2005; Alig et al., 2010; Calfapietra et al., 2010; Busch et al., 2011; Smith et al., 2013b; Branca et al., 2013; Albers and Robinson, 2013). These effects can be positive (co-benefits) or negative (adverse side-effects) and do not necessarily overlap geographically, socially or in time (11.7 and figure 11.11). This creates the possibility of trade-offs, because an AFOLU measure can bring co-benefits to one social group in one area (e.g. increasing income), while bringing adverse side-effects to others somewhere else (e.g. reducing food availability).

Table 11.5 summarizes the issues commonly considered when assessing the above-mentioned interactions at various levels between sustainable development and AFOLU.

Table 11.5: Issues related to AFOLU measures and sustainable development

Dimensions	**Issues**
Social and human assets	Population growth and migration, level of education, human capacity, individual skills, indigenous and traditional knowledge, cultural values, equity and health, animal welfare, organizational capacity
Natural assets	Availability of natural resources (land, forest, water, agricultural land, minerals, fauna), GHG balance, ecosystem integrity, biodiversity conservation, ecosystem services, the productive capacity of ecosystems, ecosystem health and resilience
State of infrastructure and technology	Availability of infrastructure and technology and industrial capacity, technology development, appropriateness, acceptance
Economic factors	Credit capacity, employment creation, income, wealth distribution/distribution mechanisms, carbon finance, available capital/investments, market access
Institutional arrangements	Land tenure and land use rights, participation and decision making mechanisms (e.g. through Free, Prior and Informed Consent, FPIC), sectoral and cross-sectoral policies, investment in research, trade agreements and incentives, benefit sharing mechanisms, existence and forms of social organization,

Based on (Madlener et al., 2006; Sneddon et al., 2006; Pretty, 2008; Corbera and Brown, 2008; Macauley and Sedjo, 2011; de Boer et al., 2011).

Social complexity: Social actors in the AFOLU sector include individuals (farmers, forest users), social groups (communities, indigenous groups), private companies (e.g. concessionaires, food-producer multinationals), subnational authorities and national States (see table 11.6).

Table 11.6: Characterization of social actors in AFOLU

Social actors	***Characterization***
Individuals (forest users – legal and illegal-, farmers)	Rather small scale interventions, although some can be medium scale Decisions taken rather at the local level
Social groups (communities, indigenous peoples)	Small to medium interventions Decisions taken at the local or regional levels
Sub-national authorities (provinces, states)	Medium to large interventions Decisions taken at the national or sub-national level, depending on the governance structure
State (national level)	Rather large interventions Decisions taken at the national level, often in line with international agreements
Corporate (at the national or multinational levels)	Rather large interventions. Decisions can be taken within a specific region/country, in another country or at global level (e.g. for multinational companies). National and international markets play a key role in decision-making

Spatial scale refers on the one hand to the size of an intervention (e.g. in number of hectares) and on the other hand to the biophysical characterization of the specific land (e.g. soil type, water availability, slope). Social interactions tend to become more complex the bigger the area of an AFOLU intervention, on a social-biophysical continuum: family/farm – neighborhood – community – village – city - province – country – region – globe. Impacts from AFOLU measures on sustainable development are different along this spatial scale continuum (Table 11.6). The challenge is to provide landscape governance that responds to societal needs as well as biophysical capacity at different spatial scales (Görg, 2007; Moilanen and Arponen, 2011; van der Horst and Vermeylen, 2011).

Temporal scale: As the concept of sustainable development includes current and future generations, the impacts of AFOLU over time need to be considered (see chapter 4). Positive and negative impacts of AFOLU measures can be realized at different times. For instance, while reducing deforestation has an immediate positive impact on reducing GHG emissions, reforestation will have a positive impact on C sequestration over time. Further, in some circumstances, there is the risk of reversing current emission reductions in the future (see 11.3.2 on non-permanence).

Behavioural aspects: Level of education, cultural values and tradition, as well as access to markets and technology, and the decision power of individuals and social groups, all influence the perception of potential impacts and opportunities from AFOLU measures, and consequently have a great impact on local land management decisions (see Chapters 2, 3 and 4; Guthinga, 2008; Durand and Lazos, 2008; Gilg, 2009; Bhuiyan et al., 2010; Primmer and Karppinen, 2010; Durand and Vázquez, 2011). When decisions are taken at a higher administrative level (e.g. international corporations, regional authorities or national States), other factors or values play an important role, including national and international development goals and priorities, policies and commitments, international markets or corporate image (see Chapters 3 and 4). Table 11.7 summarizes the emerging behavioural aspects regarding AFOLU mitigation measures.

Table 11.7: Emerging behavioural aspects relevant for AFOLU mitigation measures

Change in	Emerging behavioural aspects in AFOLU
Consumption patterns	***Dietary change***: Several changes in diet can potentially reduce GHG emissions, including reduction of food waste and reduction of / changes in meat consumption (especially in industrialized countries). On the other hand, increasing income and evolving lifestyles with increasing consumption of animal protein in developing countries are projected to increase food related GHG emissions. The potential of reducing GHG emissions in the food sector needs to be understood in a wider and changing socio-cultural context that determines nutrition. Potential drivers of change: Health awareness and information, income increase, lifestyle References 1, 2,3, 4, 5
Production patterns	***Large-scale land acquisition***: The acquisition of (long-term rights) of large areas of farmland in lower income countries, by transnational companies, agribusiness, investments funds or government agencies. There are various links between these acquisitions and GHG emissions in the AFOLU sector. On one hand because some acquisitions are aimed at producing energy crops (through non-food or "flex-crops"), on the other because these can cause the displacement of peoples and activity, increasing GHG leakage. Impacts on livelihood, local users rights, local employment, economic activity, or on biodiversity conservation are of concern Potential drivers of change: International markets and their mechanisms, national and international policies References 6, 7, 8
Production and consumption patterns	***Switching to low carbon products***: land managers are sensitive to market changes. The promotion of low carbon products as a means for reducing GHG emissions can increase the land area dedicated to these products. Side effects from this changes in land management (positive and negative), and acceptability of products and technologies at the production and consumption sides are context related and cannot be generalized Potential drivers of change: International agreements and markets, accessibility to rural energy, changes in energy demand References 9, 10, 11
Relation between producers and	***Certification***: Labelling, certification or other information-based instruments have been developed for promoting behavioural changes towards more sustainable products (11.10). Recently, the role of certification in reducing GHG while improving sustainability has been explored, especially for bioenergy (see 11.13) Potential drivers of change: Consumer awareness, international agreements, cross-national sector policies and initiatives References 11, 12, 13, 14
Management priorities	***Increasing interest in conservation and sustainable (land) management***: Changing management practices towards more sustainable ones as alternative for gaining both environmental and social co-benefits, including climate change mitigation is gaining recognition. Concerns about specific management practices, accountability methods of co-benefits and sharing mechanisms seem to be elements of concerns when promoting a more sustainable management of natural resources Potential drivers of change: Policies and international agreements and their incentive mechanisms, schemes for Payments for environmental services References 15, 16, 17, 18, 19

[1]Stehfest et al. (2009); [2]Roy et al. (2012); [3]González et al. (2011); [4]Popp et al. (2010); [5]Schneider et al. (2011); [6]Cotula (2012); [7]Messerli et al. (2013); [8]German et al. (2013); [9]Muys et al. (2013); [10]MacMillan Uribe et al. (2012); [11]Chakrabarti (2010); [12]Karipidis et al. (2010); [13]Auld et al. (2008); [14]Diaz-Chavez (2011); [15]Calegari et al. (2008); [16]Deal et al. (2012); [17]DeFries and Rosenzweig (2010); [18]Hein and van der Meer (2012);[19]Lippke et al. (2003).

Land use policies (11.10) have the challenge of balancing impacts considering these parameters: social complexity, spatial scale, temporal scale and behavioural aspects. (Vlek and Keren, 1992; Vlek, 2004) indicate the following dilemmas relevant to land management decisions: who should take the risks, when (this generation or future generations) and where (specific place) co-benefits and potential adverse effects will take place and how to mediate between individual *vs.* social benefits (Vlek and Keren, 1992; Vlek, 2004). Addressing these dilemmas is context specific. Nevertheless, the fact that a wide range of social actors need to face these dilemmas explains, to a certain extent, disagreements about environmental decision-making in general, and land management decisions in particular (Villamor et al., 2011; Le et al., 2012; see Section 11.10).

11.5 Climate change feedback and interaction with adaptation (includes vulnerability)

When reviewing the inter-linkages between climate change mitigation and adaptation within the AFOLU sector the following issues need to be considered: (i) the impact of climate change on the mitigation potential of a particular activity (e.g. forestry and agricultural soils) over time, (ii) potential synergies / tradeoffs within a land-use sector between mitigation and adaptation objectives, and (iii) potential risk-tradeoffs across sectors between mitigation and adaptation objectives.

Mitigation and adaptation in land-based ecosystems are closely interlinked through a web of feedbacks, synergies and risk-tradeoffs (see Section 11.8). The mitigation options themselves may be vulnerable to climatic change (see 11.3.2) or there may be possible synergies or trade-offs between mitigation and adaptation options within or across AFOLU sectors.

IPCC WG I presents feedbacks between climate change and the carbon cycle (WGI Chapter 6; Le Quéré et al., 2013), while WGII assesses the impacts of climate change on terrestiral ecosystems (WGII Chapter 4) and crop production systems (WGII Chapter 7), including vulnerability and adaptation. This section focuses particularly on the impacts of climate change on mitigation potential of land use sectors and interactions that arise with adaptation, linking to the relevant chapters of WGI and WGII reports.

11.5.1 Feedbacks between ALOFU and climate change

AFOLU activities can either reduce or accelerate climate change by affecting biophysical processes (e.g. evapotranspiration, albedo) and change in greenhouse gas fluxes to and from the atmosphere (WGI). Whether a particular ecosystem is functioning as sink or source of greenhouse gas emission may change over time, depending on its vulnerability to climate change and other stressors and disturbances. Hence, mitigation options available today (11.3) in the AFOLU sectors may no longer be available in the future.

There is *robust evidence* that human-induced land-use changes have led to an increased surface albedo (WGI, Chapter 8; Myhre et al. 2013). Changes in evapotranspiration and surface roughness, may counteract the effect of changes in albedo. Land-use changes affect latent heat flux and influence the hydrological cycle. Biophysical climate feedbacks of forest ecosystems differ depending on regional climate regime and forest types. For example, a decrease in tropical forests has a possitive climate forcing through a decrease in evaporative cooling (Bala et al., 2007; Bonan, 2008). An increase in coniferous-boreal forests compared to grass and snow provides a positive climate forcing through lowering albedo (Bala et al., 2007; Bonan, 2008; Swann et al., 2010). There is currently *low agreement* on the net biophysical effect of land-use changes on the global mean temperature (WGI, Chapter 8; Myhre et al. 2013). By contrast the biogeochemical effects of land use change on raditive forcing through emissions of greenhouse gases is is possitive (WGI Chapter 8; see also 11.2.2 and 11.2.3).

11.5.2 Implications of climate change on terrestrial carbon pools and mitigation potential of forests

Projections of the global carbon cycle to 2100 using 'CMIP5 Earth System Models' (WGI Chapter 6; Le Quéré et al., 2013) that represent a wider range of complex interactions between the carbon cycle and the physical climate system consistently estimate a positive feedback between climate and the carbon cycle, i.e. reduced natural sinks or increased natural CO_2 sources in response to future climate change. Implications of climate change on terrestrial carbon pools biomes and mitigation potential of forests.

Rising temperatures, drought and fires may lead to forests becoming a weaker sink or a net carbon source before the end of the century (Sitch et al., 2008). Pervasive droughts, disturbances such as fire and insect outbreaks, exacerbated by climate extremes and climate change put the mitigation benefits of the forests at risk (Canadell and Raupach, 2008; Phillips et al., 2009; Herawati and Santoso, 2011). Forest disturbances and climate extremes have associated carbon balance implications (Millar et al., 2007; Kurz et al., 2008; Zhao and Running, 2010; Potter et al., 2011; Davidson, 2012; Reichstein et al., 2013). Allen et al. (2010) suggest that at least some of the world's forested ecosystems may already be responding to climate change.

Experimental studies and observations suggest that predicted changes in temperature, rainfall regimes and hydrology may promote the die-back of tropical forests (e.g., Nepstad et al., 2007). The prolonged drought conditions in the Amazon region during 2005 contributed to a decline in aboveground biomass and triggered a release of 4.40 to 5.87 Gt CO_2 (Phillips et al., 2009). Earlier model studies suggested Amazon die-back in the future. Earlier model studies suggested Amazon die-back in the future (Cox et al., 2013; Huntingford et al., 2013). However recent model estimates suggest that rainforests may be more resilient to climate change, projecting a moderate risk of tropical forest reduction in South America and even lower risk for African and Asian tropical forests (Gumpenberger et al., 2010; Cox et al., 2013; Huntingford et al., 2013).

(Arcidiacono-Bársony et al., 2011) suggest that the mitigation benefits from deforestation reduction under REDD+ (see 11.10.1) could be reversed due to increased fire events, and climate-induced feedbacks, while (Gumpenberger et al., 2010) conclude that the protection of forests under the forest conservation (including REDD) programmes could increase carbon uptake in many tropical countries, mainly due to CO_2 fertilization effects, even under climate change conditions.

11.5.3 Implications of climate change on peat lands, grasslands and croplands

Peatlands: Wetlands, peatlands and permafrost soils contain higher carbon densities relative to mineral soils, and together they comprise extremely large stocks of carbon globally (Davidson and Janssens, 2006). Peatlands cover approximately 3% of the earth's land area and are estimated to contain 350-550 Gt of carbon, roughly between 20 to 25% of the world's soil organic carbon stock (Gorham, 1991; Fenner et al., 2011). Peatlands can lose CO_2 through plant respiration and aerobic peat decomposition (Clair et al., 2002) and with the onset of climate change, may become a source of CO_2 (Koehler et al., 2010). Large carbon losses are likely from deep burning fires in boreal peatlands under future projections of climate warming and drying (Flannigan et al., 2009). A study by Fenner et al. (2011) suggests that climate change is expected to increase the frequency and severity of drought in many of the world's peatlands which, in turn, will release far more GHG emissions than thought previously. Climate change is projected to have a severe impact on the peatlands in northern regions where most of the perennially frozen peatlands are found (Tarnocai, 2006). According to Schuur et al. (2008), the thawing permafrost and consequent microbial decomposition of previously frozen organic carbon, is one of the most significant potential feedbacks from terrestrial ecosystems to the atmosphere in a changing climate. Large areas of permafrost will experience thawing (WGI Chapter 12; Collins et al. 2013), but uncertainty over the magnitude of frozen carbon losses through CO_2 or CH_4 emissions to the atmosphere are large, ranging between 180 and 920 $GtCO_2$ by the end of the 21st century under the RCP8.5 scenario (WGI Chapter 6; Le Quéré et al., 2013).

Grasslands: Tree cover and biomass in savannah has increased over the past century (Angassa and Oba, 2008; Witt et al., 2009; Lunt et al., 2010; Rohde and Hoffman, 2012) leading to increased carbon storage per hectare (Hughes et al., 2006; Liao et al., 2006; Throop and Archer, 2008; Boutton et al., 2009) which has been attributed to land management, rising CO_2, climate variability and climate change. Climate change and CO_2 may affect grazing systems by altering species composition; for example, warming will favour tropical (C4) species over temperate (C3) species but CO_2 increase would favour C3 grasses (Howden et al., 2008).

Croplands: Climate change impacts on agriculture will affect not only crop yields, but also SOC levels in agricultural soils (Rosenzweig and Tubiello, 2007). Such impacts can be either positive or negative, depending on the particular effect considered, which highlights the uncertainty of the impacts. Elevated CO_2 alone are expected to have positive effects on soil carbon storage, because of increased above- and below-ground biomass production in agro-ecosystems. Similarly, the lengthening of the growing season under future climate will allow for increased carbon inputs into soils. Warmer temperatures could have negative impacts on SOC, by speeding decomposition and by reducing inputs by shortening crop life cycles (Rosenzweig and Tubiello, 2007), but increased productivity could increase SOC stocks (Gottschalk et al., 2012).

11.5.4 Potential adaptation options to minimize the impact of climate change on carbon stocks in forests and agricultural soils

Forests: Forest ecosystems require a longer response time to adapt, the development and implementation of adaptation strategies is also lengthy (Leemans and Eickhout, 2004; Ravindranath, 2007). Some examples of the adaptation practices (Murthy et al., 2011) are as follows: anticipatory planting of species along latitude and altitude, assisted natural regeneration, mixed species forestry, species mix adapted to different temperature tolerance regimes, fire protection and management practices, thinning, sanitation and other silvicultural practices, *in situ* and *ex situ* conservation of genetic diversity, drought and pest resistance in commercial tree species, adoption of sustainable forest management practices, increase in Protected Areas and linking them wherever possible to promote migration of species, forests conservation and reduced forest fragmentation enabling species migration, and energy efficient fuel-wood cooking devices to reduce pressure on forests.

Agricultural soils: On current agricultural land, mitigation and adaptation interaction can be mutually re-enforcing, particularly for improving resilience to increased climate variability under climate change (Rosenzweig and Tubiello, 2007). Many mitigation practices implemented locally for soil carbon sequestration will increase the ability of soils to hold soil moisture and to better withstand erosion and will enrich ecosystem biodiversity by establishing more diversified cropping systems, and may also help cropping systems to better withstand droughts and floods, both of which are projected to increase in frequency and severity under a future warmer climate (Rosenzweig and Tubiello, 2007).

11.5.5 Mitigation and adaptation synergies and risk-tradeoffs

Mitigation choices taken in a particular land-use sector may further enhance or reduce resilience to climate variability and change within or across sectors. In light of the multiple, and often competing, pressures on land (11.4), and shifting demographics and consumption patterns (e.g., O'Brien et al., 2004; Sperling et al., 2008; Hunsberger and Evans, 2012). Land-use choices driven by mitigation concerns (e.g. forest conservation, afforestation) may have consequences for adaptive responses and/or development objectives of other sectors (e.g. expansion of agricultural land). For example, reducing emissions from deforestation and degradation may also yield co-benefits for adaptation by maintaining biodiversity and other ecosystem goods and services, while plantations, if they reduce biological diversity may diminish adaptive capacity to climate change (e.g., Chum et al., 2011). Primary forests tend to be more resilient to climate change and other human induced environmental changes than secondary forests and plantations (Thompson et al., 2009). The impact of plantations on the carbon balance is dependent on the land-use system they replace, while plantation forests are often monospecies stands, they may be more vulnerable to climatic change (see IPCC WG 2, Chapter 4). Smith and Olesen (2010) identified a number of synergies between options that deliver climate mitigation in agriculture while also enhancing resilience to future climate change, the most prominent of which was enhancement of soil carbon stocks.

Adaptation measures in return my help maintain the mitigation potential of land-use systems. For example, projects that prevent fires and restore degraded forest ecosystems also prevent release of GHGs and enhance carbon stocks (CBD and GiZ, 2011). Mitigation and adaptation benefits can also be achieved within broader level objectives of AFOLU measures, which are linked to sustainable development considerations. Given the exposure of many livelihoods and communities to multiple stressors, recommendations from case studies suggest that climate risk management strategies need to appreciate the full hazard risk envelope, as well as the compounding socioeconomic stressors (O’Brien et al., 2004; Sperling et al., 2008). Within this broad context, the potential trade-offs and synergies between mitigation, adaptation and development strategies and measures need to be considered. Forest and biodiversity conservation, protected area formation and mixed species forestry based afforestation are practices that can help to maintain or enhance carbon stocks, while also providing adaptation options to enhance resilience of forest ecosystems to climate change (Ravindranath, 2007). Use of organic soil amendments as a source of fertility could potentially increase soil carbon (Gattinger et al., 2012). Most categories of adaptation options for climate change have positive impacts on mitigation. In the agriculture sector, cropland adaptation options that also contribute to mitigation are: “soil management practices that reduce fertilizer use and increase crop diversification; promotion of legumes in crop rotations; increasing biodiversity, the availability of quality seeds and integrated crop/livestock systems; promotion of low energy production systems; improving the control of wildfires and avoiding burning of crop residues; and promoting efficient energy use by commercial agriculture and agro-industries” (FAO, 2008, 2009a). Agroforestry is an example of mitigation-adaptation synergy in agriculture sector, since trees planted sequester carbon and tree products provide livelihood to communities, especially during drought years (Verchot et al., 2007).

11.6 Costs and potentials

This section deals with economic costs and potentials of greenhouse gas (GHG) mitigation (emission reduction or sequestration of carbon) within the AFOLU sector. Economic mitigation potentials are distinguished from technical or market mitigation potentials (Smith, 2012). Technical mitigation potentials represent the full biophysical potential of a mitigation option, without accounting for economic or other constraints. These estimates account for constraints and factors such as land availability and suitability (Smith, 2012), but not any associated costs (at least explicitly). By comparison, economic potential refers to mitigation that could be realised at a given carbon price over a specific period, but does not take into consideration any socio-cultural (for example, life-style choices) or institutional (for example, political, policy and informational) barriers to practice or technology adoption. Economic potentials are expected to be lower than the corresponding technical potentials. Also, policy incentives (e.g. a carbon price; see also 11.10) and competition for resources across various mitigation options, tend to affect the size of economic mitigation potentials in the AFOLU sector (McCarl and Schneider, 2001). Finally, market potential is the realised mitigation outcome under current or forecast market conditions encompassing biophysical, economic, socio-cultural and institutional barriers to, as well as policy incentives for, technological and/or practice adoption, specific to a sub-national, national or supra-national market for carbon. Figure 11.12 (Smith, 2012) provides a schematic view of the three types of mitigation potentials.

Economic (as well as market) mitigation potentials tend to be context-specific and are likely to vary across spatial and temporal scales. Unless otherwise stated, in the rest of this section, economic potentials are expressed in million tonnes (Mt) of GHG mitigation in carbon dioxide equivalent (CO_2eq) terms, that can arise from an individual mitigation option or from an AFOLU sub-sector at a given cost per tonne of CO_2eq. (USD/t CO_2eq) over a given period to 2030, which is 'additional' to the corresponding baseline or reference case levels.

Various supply-side mitigation options within the AFOLU sector are dedescribed in 11.3, and 11.4 considers a number of potential demand-side options. Estimates for costs and potentials are not always available for the individual options described. Also, aggregate estimates covering both the supply- and demand-side options for GHG mitigation within the AFOLU sector are lacking, so this section mostly focuses on the supply-side options. Key uncertainties and sensitivities around mitigation costs and potentials in the AFOLU sector are (1) carbon price, (2) prevailing biophysical and climatic conditions, (3) existing management heterogeneity (or differences in the baselines), (4) management interdependencies (arising from competition or co-benefits across tradition production, environmental outcomes and mitigation strategies or competition/co-benefits across mitigation options), (5) the extent of leakage, (6) differential impact on different GHGs associated with a particular mitigation option, and (7) timeframe for abatement activities and the discount rate. In this section we, a) provide aggregate mitigation potentials for the AFOLU sector (because these wereprovided separately for agriculture and forestry in AR4), b) provide estimates of global mitigation costs and potentials published since AR4, and c) provide a regional disaggregation of the potentials to show how potential, and the portfolio of available options, varies in different world regions.

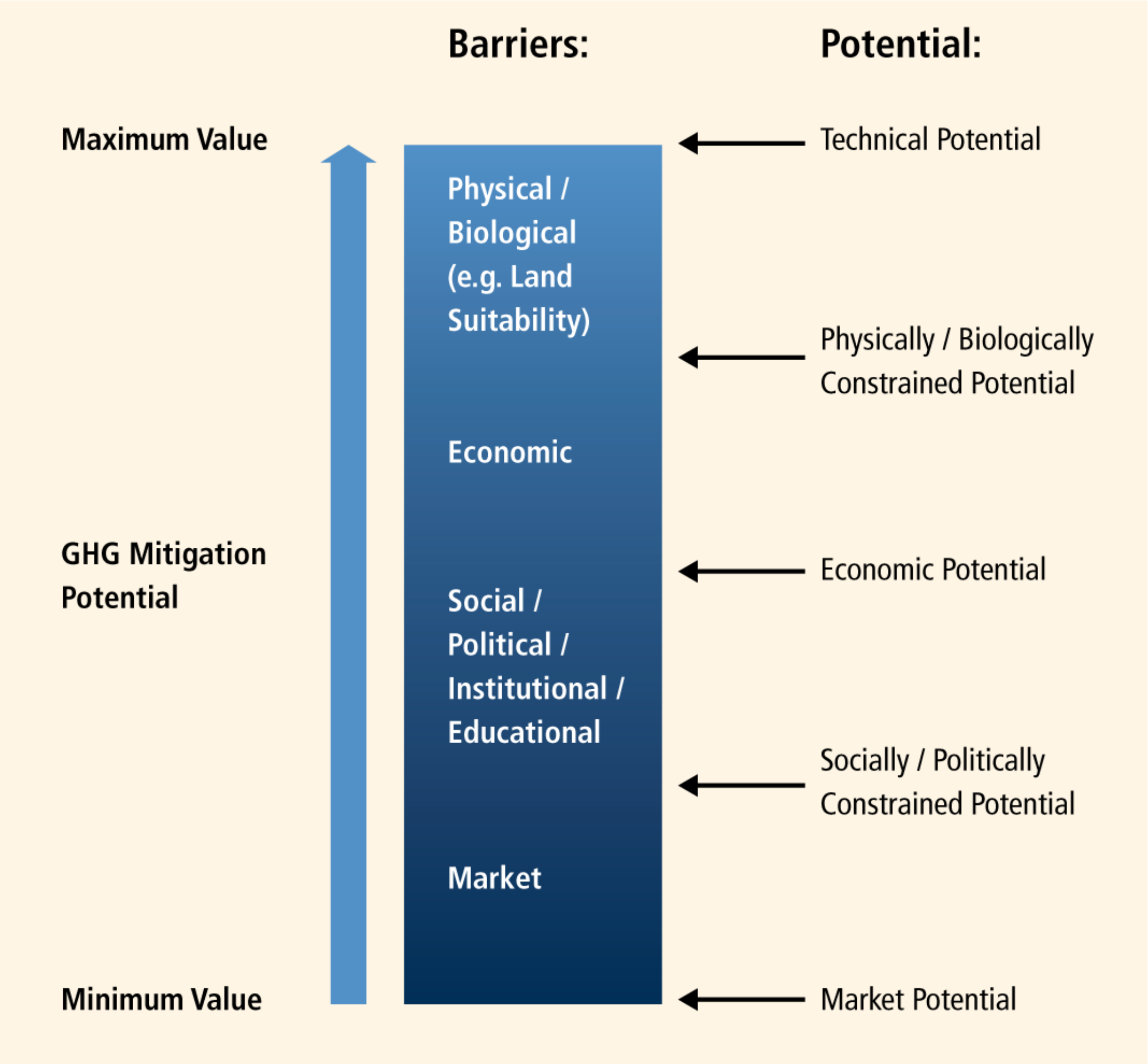

Figure 11.12. Relationship between technical, economic and market potential (after Smith, 2012)

11.6.1 Approaches to estimating economic mitigation potentials

Bottom-up and top-down modelling approaches are used to estimate AFOLU mitigation potentials and costs. While both approaches provide useful estimates for mitigation costs and potentials, comparing bottom-up and top-down estimates is not straightforward.

Bottom-up estimates are typically derived for discrete abatement options in agriculture at a specific location or time, and are often based on detailed technological, engineering and process information and data on individual technologies (DeAngelo et al., 2006). These studies provide estimates of how much technical potential of particular AFOLU mitigation options will become economically viable at certain carbon dioxide-equivalent prices. Bottom-up mitigation responses are typically restricted to input management (for example, changing practices with fertiliser application and livestock feeding) and mitigation costs estimates are considered ‘partial equilibrium’ in that the relevant input-output prices (and, sometimes, quantities such as area or production levels) are held fixed. As such, unless adjusted for potential overlaps and trade-offs across individual mitigation options, adding up various individual estimates to arrive at an aggregate for a particular landscape or at a particular point in time could be misleading.

With a 'systems' approach, top-down models (described in Chapter 6; 11.9) typically take into account possible interactions between individual mitigation options. These models can be sector-specific or economy-wide, and can vary across geographical scales: sub-national, national, regional and global. Mitigation strategies in top-down models may include a broad range of management responses and practice changes (for example, moving from cropping to grazing or grazing to forestry) as well as changes in input-output prices (for example, land and commodity prices). Such models can be used to assess the cost competitiveness of various mitigation options and implications across input-output markets, sectors, and regions over time for large-scale domestic or global adoption of mitigation strategies. In top-down modelling, dynamic cost-effective portfolios of abatement strategies are identified incorporating the lowest cost combination of mitigation strategies over time from across sectors, including agricultural, forestry and other land-based sectors across the world that achieve the climate stabilisation target (see Chapter 6). Top-down estimates for 2030 are included in this section, and are revisited in 11.9 when considering the role of the AFOLU sector in transformation pathways.

Providing consolidated estimates of economic potentials for GHG mitigation within the AFOLU sector as a whole is complicated because of complex interdependencies, largely stemming from competing demands on land for various agricultural and forestry (production and mitigation) activities, as well as for the provision of many ecosystem services (Smith et al., 2013a). These interactions are discussed in more detail in 11.4.

11.6.2 Global estimates of costs and potentials in the AFOLU sector

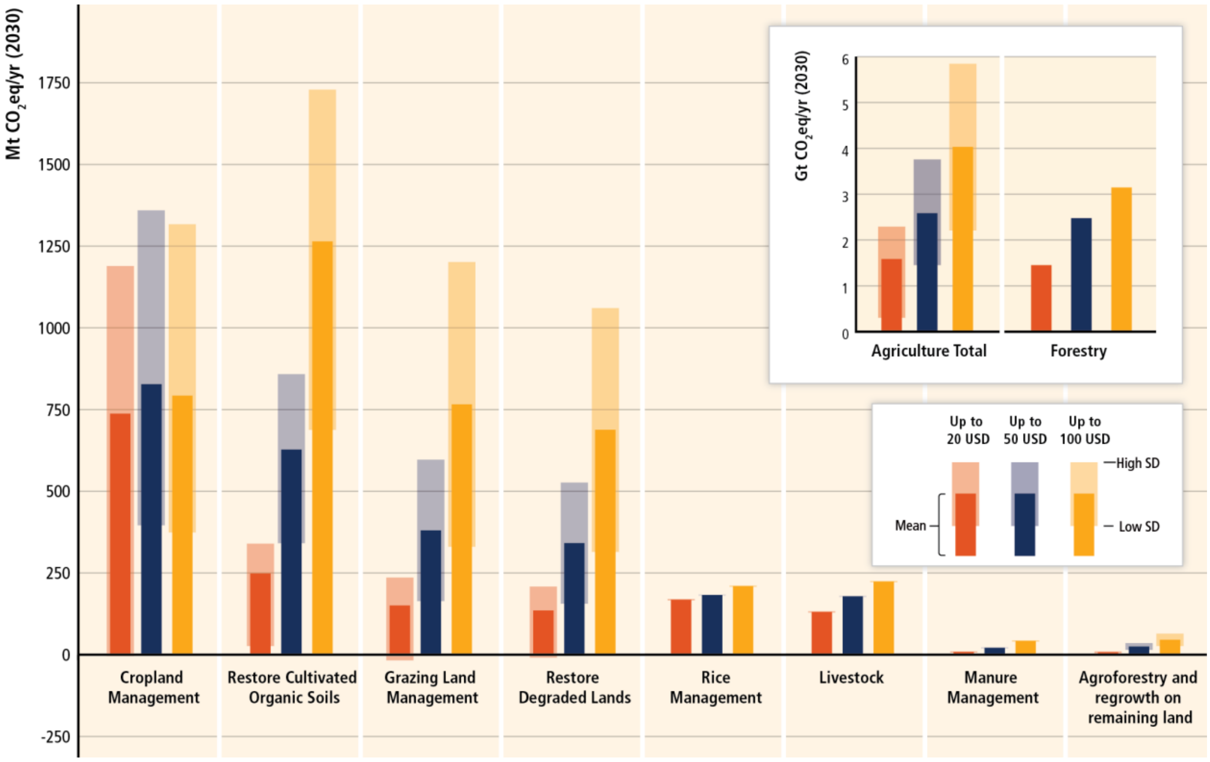

Figure 11.13. Mitigation potential for the AFOLU sector, plotted using data from IPCC AR4 (Nabuurs et al., 2007; Smith, et al., 2007). Transparent ranges show the range of estimates (+/- 1 standard deviation) for agricultural options for which estimates are available.

Through combination of forestry and agriculture potentials from IPCC AR4, total mitigation potentials for the AFOLU sector are estimated to be ~3 to ~7.2 Gt CO_2eq/yr in 2030 at 20 and 100 USD/t CO_2eq, respectively (Figure 11.13), including only supply-side options in agriculture (Smith, et al., 2007) and a combination of supply- and demand-side options for forestry (Nabuurs et al., 2007). Estimates of global economic mitigation potentials in the AFOLU sector published since AR4 are shown in Figure 11.14, with AR4 estimates shown for comparison (IPCC AR4, 2007 in figure 11.14).

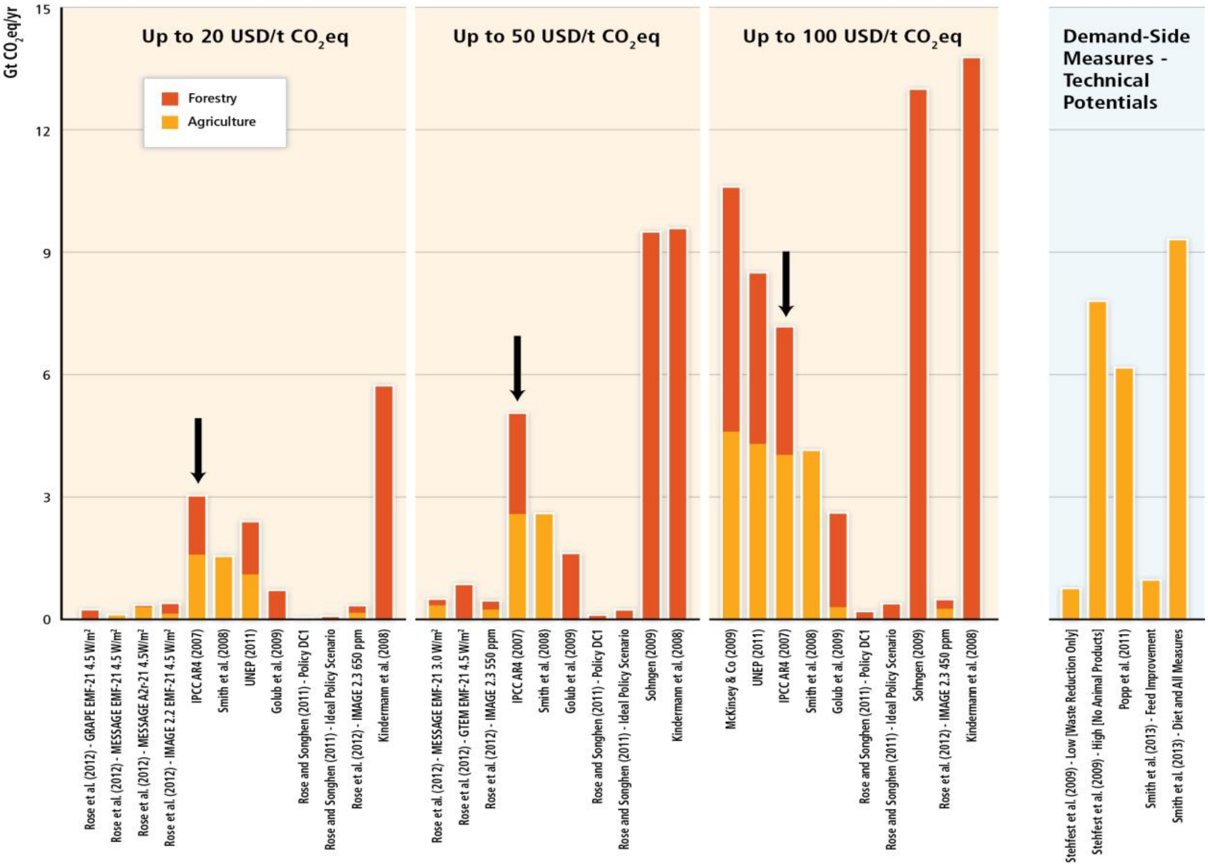

Figure 11.14. Estimates of economic mitigation potentials in the AFOLU sector published since AR4, (AR4 estimates shown for comparison, denoted by red arrows), including bottom-up, sectoral studies, and top-down, multi-sector studies. Some studies estimate potential for agriculture and forestry, others for one or other sector. Supply-side mitigation potentials are estimated for around 2030, but studies range from estimates for 2025 (Rose et al., 2012) to 2035 (Rose and Sohngen, 2011). Studies are collated for those reporting potentials at up to ~20 USD/t CO$_2$eq (actual range 1.64-21.45), up to ~50 USD/t CO$_2$eq (actual range 31.39-50.00), and up to ~100 USD/t CO$_2$eq (actual range 70.0-120.91). Demand-side options (shown on the right hand side of the figure) are for ~2050 and are not assessed at a specific carbon price, and should be regarded as technical potentials. Smith et al. (2013) values are mean of the range. Not all studies consider the same options or the same GHGs; further details are given in the text.

Table 11.8 summarises the ranges of global economic mitigation potentials from AR4 (Nabuurs et al., 2007; Smith et al., 2007), and studies published since AR4 that are shown in full in Figure 11.14, for agriculture, forestry and AFOLU combined.

Table 11.8: Ranges of global mitigation potential (Gt CO$_2$eq/yr) reported since IPCC AR4. All values are for 2030 except demand-side options which are for ~2050 (full data shown in Figure 11.14)

	up to 20 USD/t CO$_2$eq	up to 50 USD/t CO$_2$eq	up to 100 USD/t CO$_2$eq	Technical potential only
Agriculture only[1]	0-1.59	0.03-2.6	0.26-4.6	
Forestry only	0.01-1.45	0.11-9.5	0.2-13.8	
AFOLU total[1,2]	0.12-3.03	0.5-5.06	0.49-10.6	
Demand-side options				0.76-9.31

[1] All lower range values for agriculture are for non-CO$_2$ GHG mitigation only and do not include soil C sequestration

[2] AFOLU total includes only estimates where both agriculture and forestry have been considered together.

As described in 11.3, since AR4, more attention has been paid to options that reduce emissions intensity by improving the efficiency of production (i.e. less GHG emissions per unit of agricultural product; Burney et al., 2010; Bennetzen et al., 2012). As agricultural and silvicultural efficiency have improved over recent decades, emissions intensities have declined (Figure 11.15). Whilst emissions intensity has increased (1960s to 2000s) by 45% for cereals, emissions intensities have decreased by 38% for milk, 50% for rice, 45% for pig meat, 76% for chicken and 57% for eggs.

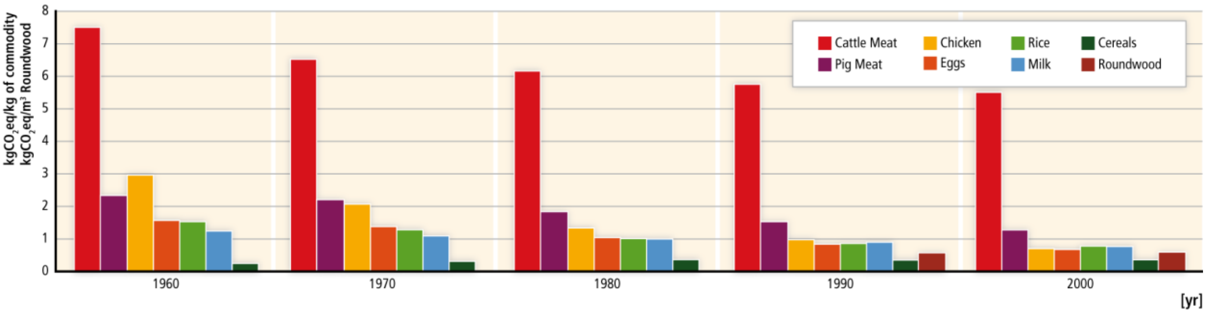

Figure 11.15. GHG emissions intensities of selected major AFOLU commodities for decades 1960s-2000s, based on (Tubiello et al., 2012). i) Cattle meat, defined as GHG (Enteric fermentation+ Manure management of Cattle, Dairy and Non-Dairy)/meat produced; ii) Pig meat, defined as GHG (Enteric fermentation+ Manure management of Swine, market and breeding) /meat produced; iii) Chicken meat, defined as GHG (Manure management of Chickens)/meat produced; iv) Milk, defined as GHG (Enteric fermentation+ Manure management of Cattle, dairy)/milk produced; v) Eggs, defined as GHG (Manure management of Chickens, layers)/egg produced; vi) Rice, defined as GHG (Rice cultivation)/rice produced; vii) Cereals, defined as GHG (Synthetic fertilizers)/cereals produced; viii) Wood, defined as GHG (Carbon loss from harvest)/Roundwood produced. Data Source: (FAOSTAT, 2013).

The implementation of mitigation measures can contribute to further decrease emission intensities of AFOLU commodities (Figure 11.16; which shows changes of emissions intensities when a commodity-specific mix of mitigation measures is applied). For cereal production, mitigation measures considered include improved cropland agronomy, nutrient and fertilizer management, tillage and residue management and the establishment of agro-forestry systems. Improved rice management practices where considered for paddy rice cultivation. Mitigation measures applied in the livestock sector include improved feeding and dietary additives. Countries can improve emission intensities of AFOLU commodities through increasing production at the same level of input, the implementation of mitigation measures, or a combination of both. In some regions, increasing current yields is still an option with a significant potential to improve emission intensities of agricultural production. Foley et al. (2011) analysed current and potential yields that could be achieved for 16 staple crops using available agricultural practices and technologies and identified large “yield gaps”, especially across many parts of Africa, Latin America and Eastern Europe. Better crop management practices can help to close yield gaps and improve emission intensities if measures are selected that also have a mitigation potential.

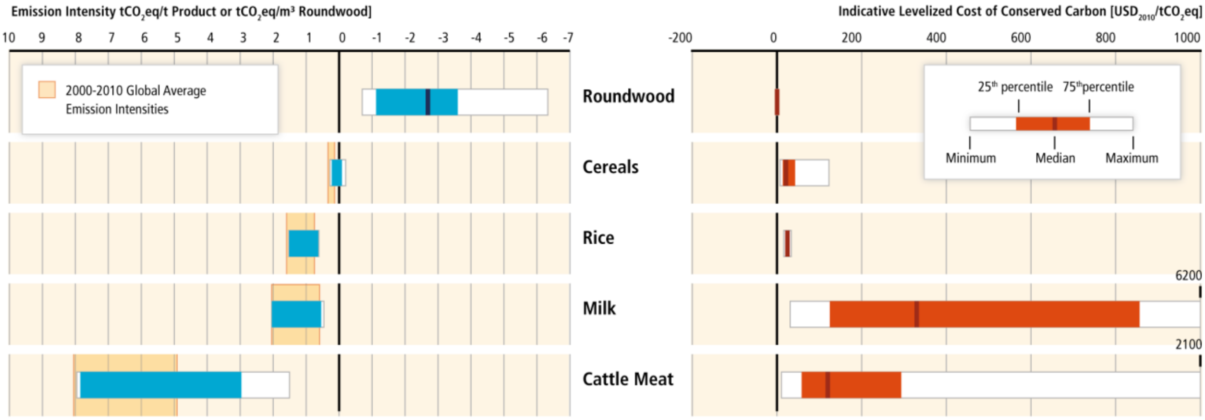

Figure 11.16. Potential changes of emission intensities of major AFOLU commodities through implementation of commodity-specific mitigation measures (left panel) and related mitigation costs (right panel). Commodities and GHG emission sources are defined as in Figure 11.15, except for roundwood, expressed as the amount of carbon sequestered per unit roundwood from reforestation and afforestation within dedicated plantation cycles. Agricultural emission intensities represent regional averages, calculated based on 2000-2010 data (FAOSTAT, 2013) for selected commodities. Data on mitigation potentials and costs of measures are calculated using the mean values reported by (Smith et al., 2008) and the maximum and minimum are defined by the highest and lowest values for four climate zones for cereals and rice, or five geographical regions for milk and cattle meat. Emission intensities and mitigation potentials of roundwood production are calculated using data from (Sathaye et al., 2005, 2006), (FAO, 2006b) and (IPCC, 2006); maximum and minimum values are defined by the highest and lowest values for ten geographical regions. In the left panel, red stacks of bars show current GHG emissions (in tCO_2eq) per unit commodity (in t or m^3), white stacks of bars indicate the GHG mitigation potential of selected measures (in tCO_2eq) per unit commodity (in t or m^3), and striped stacks of bars represent the area where ranges of GHG emissions and mitigation potentials overlap. The right panel shows the mitigation costs (in USD/tCO_2eq) of commodity-specific mitigation measures (1:3 quartile range).

Mitigation potentials and costs differ largely between AFOLU commodities (Figure 11.16). While average abatement costs are low for roundwood production under the assumption of perpetual rotation, costs of mitigation options applied in meat and dairy production systems have a wide range (1:3 quartile range: 58-856 USD/tCO_2eq). Calculations of emission intensities are based on the conservative assumption that production levels stay the same after the application of the mitigation option. However, some mitigation options can increase production. This would not only improve food security but could also increase the cost-effectiveness of mitigation actions in the agricultural sector.

Agriculture and forestry related GHG mitigation could cost-effectively contribute to transformation pathways associated with long-run climate change management (11.9 and 6.3.5). Transformation pathway modeling includes land use change, as well as land management options that reduce emissions intensities and increase sequestration intensities. However, the resulting transformation pathway emissions (sequestration) intensities are not comparable to those discussed here. Transformation pathways are the result of integrated modelling and the resulting intensities are the net result of many effects. The intensities capture mitigation technology adoption, but also changes in levels of production, land cover change, mitigation technology competition, and model specific definitions for sectors/regions/and assigned emissions inventories. Mitigation technology competition, in particular, can lead to intensification (and increases in agricultural emissions intensities) that support cost-effective adoption of other mitigation strategies, such as afforestation or bioenergy (11.9 and 6.3.5).

11.6.3 Regional disaggregation of global costs and potentials in the AFOLU sector

Figure 11.17 shows the economically viable mitigation opportunities in AFOLU in 2030 by region and by main mitigation option at carbon prices of up to USD20, 50 and 100/t CO_2eq. The composition of the agricultural mitigation portfolio varies greatly with the carbon price (Smith, 2012), with low cost options such as cropland management being favoured at low carbon prices, but higher cost options such as restoration of cultivated organic soils being more cost effective at higher prices. Figure 11.17 also reveals some very large differences in mitigation potential, and different ranking of most effective options, between regions. Across all AFOLU options, Asia has the largest mitigation potential, with the largest mitigation in both forestry and agriculture, followed by LAM, OECD90, MAF then REF.

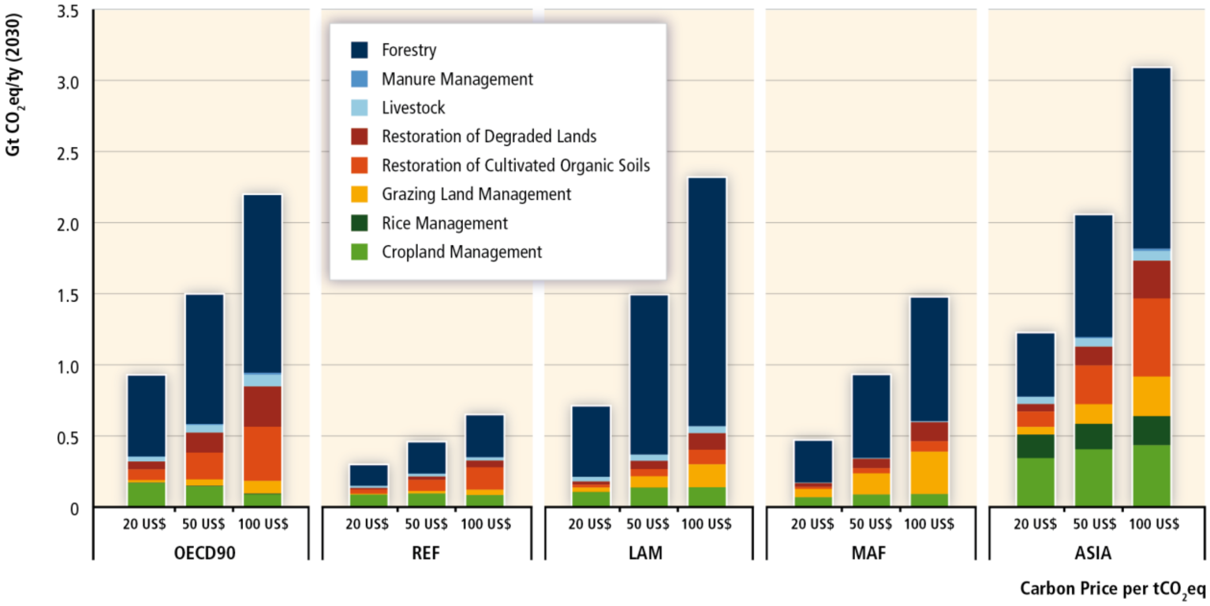

Figure 11.17. Economic mitigation potentials in the AFOLU sector by region. Agriculture values are from (Smith, et al., 2007). Forestry values are from (Nabuurs et al., 2007). For forestry, 20 USD values correspond to “low”, and 100 USD values correspond to “high” values from (Nabuurs et al., 2007). 50 USD values represent the mean of the “high” and “low” values from (Nabuurs et al., 2007).

Differences between the most effective forectry options in each region (Figure 11.18) are particularly striking, with reduced deforestation dominating the forestry mitigation potential LAM and MAF, but very little potential in OECD90 and REF. Forest management, followed by afforestation, dominate in OECD90, REF and Asia (Figure 11.18). Among agricultural options, among the most striking of regional differences are the rice management practices for which almost all of the global potential is in Asia, and the large potential for restoration of organic soils also in Asia (due to cultivated south east Asian peats), and OECD90 (due to cultivated northern peatlands; Figure 11.18).

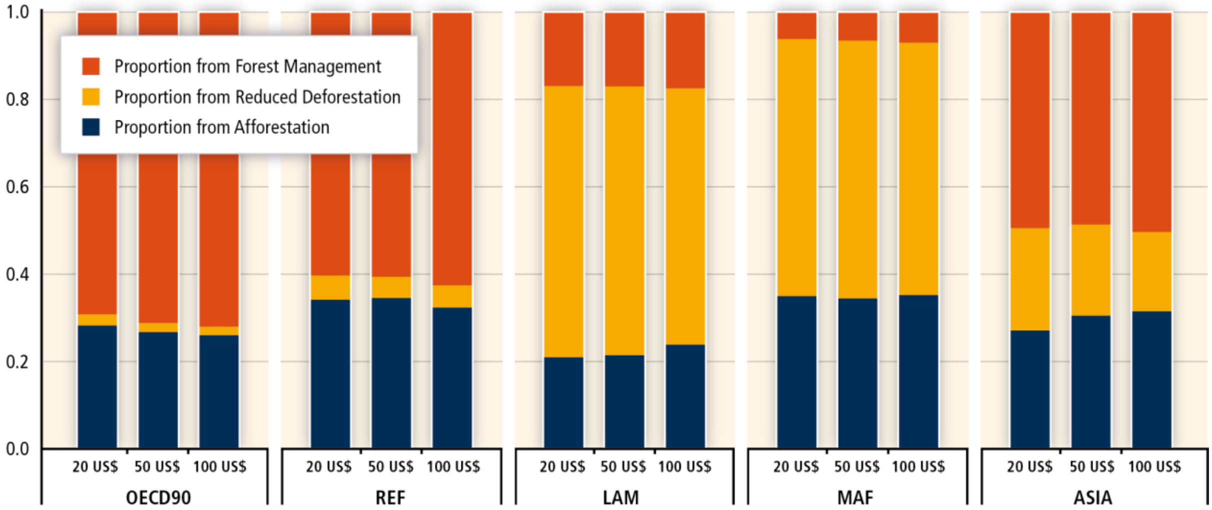

Figure 11.18. Regional differences in forestry options, shown as a proportion of total potential available in forestry in each region. Global forestry activities (annual amount sequestered or emissions avoided above the baseline for forest management, reduced deforestation and afforestation), at carbon prices up to 100 USD/t CO_2 are aggregated to regions from results from three models of global forestry and land use: the Global Timber Model (GTM; (Sohngen and Sedjo, 2006), the Generalized Comprehensive Mitigation Assessment Process (Sathaye et al., 2006), and the Dynamic Integrated Model of Forestry and Alternative Land Use (Benítez et al., 2007).

11.7 Co-benefits, risks and spillovers

Implementation of AFOLU mitigation measures (11.3) will result in a range of outcomes beyond changes in GHG balances with respect to institutional, economic, social and environmental objectives. To the extent these effects are positive, they can be deemed ‘co-benefits’; if adverse and uncertain, they imply risks.[9] A global assessment of the co-benefits and adverse side-effects of AFOLU mitigation measures is challenging for a number of reasons. First, co-benefits and adverse side-effects depend on the development context and the scale of the intervention (size), i.e. implementing the same AFOLU mitigation measure in two different areas (different countries or different regions within a country) can have different socio-economic, institutional or environmental effects (Forner et al., 2006; Koh and Ghazoul, 2008; Trabucco et al., 2008; Zomer et al., 2008; Alves Finco and Doppler, 2010; Alig et al., 2010; Colfer, 2011; Davis et al., 2013a; Muys et al., 2013; Albers and Robinson, 2013). Thus the effects are site specific and generalizations are difficult. Second, these effects do not necessarily overlap geographically, socially or over the same time scales (11.4.5). Third, there is no general agreement on attribution of co-benefits and adverse side-effects to specific AFOLU mitigation measures; and fourth there are no standardized metrics for quantifying many of these effects. Modelling frameworks are being developed which allow an integrated assessment of multiple outcomes at landscape (Bryant et al., 2010), project (Townsend et al., 2012) and smaller (Smith et al., 2013a) scales. Table 11.9 presents an overview of the potential effects from AFOLU mitigation measures, while in the text we present the most relevant co-benefits and potential adverse side-effects from the recent literature.

Maximising co-benefits of AFOLU mitigation measures can increase efficiency in achieving the objectives of other international agreements, including the United Nations Convention to Combat Desertification (UNCCD, 2011) or the Convention on Biological Diversity (CBD), and mitigation

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[9] Co-benefits and adverse side-effects describe effects in physical units without yet evaluating the net effect on overall social welfare. Please refer to the respective sections in the framing chapters as well as to the glossary in Annex I for concepts and definitions – particularly 2.2, 3.6.3, and 4.8.

actions may also contribute to a broader global sustainability agenda (Harvey et al., 2010; Gardner et al., 2012); see chapter 4). In many cases, implementation of these agendas is limited by capital, and mitigation may provide a new source of finance (Tubiello et al., 2009).

Box 11.6 Challenges for mitigation in Developing Countries in the AFOLU sector

Mitigation challenges related to the AFOLU sector

The contribution of Developing Countries to future GHG emissions is expected to be very significant due to projected increases in food production by 2030 driving short-term land conversion in these countries. Mitigation efforts in the AFOLU sector rely mainly on reduction of GHG emissions and an increase in carbon sequestration (Table 11.2). Potential activities include reducing deforestation, increasing forest cover, agroforestry, agriculture and livestock management, and production of sustainable renewable energy (Sathaye et al., 2005; Smith et al., 2013b). Although agriculture and forestry are important sectors for GHG abatement (11.2.3), it is likely that technology alone will not be sufficient to deliver the necessary transitions to a low GHG future (Alig et al., 2010); 11.3.2). Other barriers include access to market and credits, technical capacities to implement mitigation options including accurate reporting of emission levels and emission factors based on activity data, and institutional frameworks and regulations (Corbera and Schroeder, 2011; Mbow et al., 2012); 11.7; 11.8). Additionally, the diversity of circumstances among developing countries makes it difficult to establish the modelled relationships between GDP and CO_2 emissions per capita found by using the Kaya identity. This partly arises from the wide gap between rural and urban communities, and the difference in livelihoods (e.g. the use of fuel wood, farming practices in various agro-ecological conditions, dietary preferences with a rising middle class in developing countries, development of infrastructure, and behavioural change, etc.; Lambin and Meyfroidt, 2011). Also, some mitigation pathways raise the issue of non-permanence and leakage that can lead to the transfer activities to non-protected areas, which may threaten conservation areas in countries with low capacities (Lippke et al., 2003; Jackson and Baker, 2010; 11.3.2).

Critical issues to address are the co-benefits and adverse side-effects associated with changed agricultural production, the necessary link between mitigation and adaptation, and how to manage incentives for a substantial GHG abatement initiative without compromising food security (Smith and Wollenberg, 2012; 11.5; 11.7). The challenge is to strike a balance between emissions reductions/adaptation and development/poverty alleviation priorities, or to find policies that co-deliver. Mitigation pathways in developing countries should address the dual need for mitigation and adaptation through clear guidelines to manage multiple options (11.5.4). Prerequisites for the successful implementation of AFOLU mitigation projects are ensuring that, a) communities are fully engaged in implementing mitigation strategies, b) any new strategy is consistent with ongoing policies or programmes, and c) *a priori* consent of small holders is given. Extra effort is required to address equity issues including gender, challenges and prospects (Mbow et al., 2012).

Mitigation Challenges related to the bioenergy sector

Bioenergy has a significant GHG mitigation potential, provided that the resources are developed sustainably and that bioenergy systems are efficient (Chum et al., 2011; 11.9.1). Bioenergy production can be integrated with food production in developing countries, e.g. through suitable crop rotation schemes, or use of by-products and residues (Berndes et al., 2013). If implemented sustainably this can result in higher food and energy outcomes and hence reduce land use competition. Some bioenergy options in developing countries include perennial cropping systems, use of biomass residues and wastes, and advanced conversion systems (Popp et al., 2011a; Beringer et al., 2011; Box 7.1). Agricultural and forestry residues can provide low carbon and low costs feedstock for bioenergy. Biomass from cellulosic bioenergy crops feature substantially in future energy systems, especially in the framework of global climate policy that aims at stabilizing CO_2 concentration at low levels (Popp et al., 2011a; 11.13). The large-scale use of bioenergy is controversial in the context of developing countries because of the risk of reducing carbon stocks and releasing carbon to the atmosphere (Bailis and McCarthy, 2011), threats to food security in Africa (Mbow, 2010), and threats to biodiversity *via* the conversion of forests to biofuel (e.g. palm oil) plantations. Several studies underline the inconsistency between the need for bioenergy and the requirement for, e.g. Africa, to use its productive lands for sustainable food production (Cotula et al., 2009). Efficient biomass production for bioenergy requires a range of sustainability requirements to safeguard food production, biodiversity and terrestrial carbon storage.

11.7.1 Socio-economic effects

AFOLU mitigation measures can affect institutions and living conditions of the various social groups involved. This section includes potential effects of AFOLU mitigation measures on three dimensions of sustainable development: institutional, social and economic (see 11.4.5).

AFOLU mitigation measures may have impacts on ***land tenure and land use rights*** for several social groups including indigenous peoples, local communities and other social groups, dependant on natural assets. Co-benefits from AFOLU mitigation measures can be clarification of land tenure and harmonization of rights, while adverse side-effects can be lack of recognition of customary rights, loss of tenure or possession rights, and even displacement of social groups (Sunderlin et al., 2005, 2013; Chhatre and Agrawal, 2009; Blom et al., 2010; Sikor et al., 2010; Robinson et al., 2011; Rosemary, 2011; Larson, 2011; Rosendal and Andresen, 2011). Whether an impact on land tenure and use rights is positive or negative depends upon two factors: a) the institutions regulating land tenure and land use rights (e.g. laws, policies) and b) the level of enforcement by such institutions (Corbera and Brown, 2008; Araujo et al., 2009; Rosemary, 2011; Larson et al., 2013; Albers and Robinson, 2013). More research is needed on specific tenure forms (e.g. individual property, state ownership or community rights), and on the specific effects from tenure and rights options, on enabling AFOLU mitigation measures and co-benefits in different regions under specific circumstances (Sunderlin et al., 2005; Katila, 2008; Chhatre and Agrawal, 2009; Blom et al., 2010; Sikor et al., 2010; Robinson et al., 2011; Rosemary, 2011; Larson, 2011; Rosendal and Andresen, 2011).

AFOLU mitigation measures can support ***enforcement of sectoral policies*** (e.g. conservation policies) as well as ***cross-sectoral coordination*** (e.g. facilitating a landscape view for policies in the agriculture, energy and forestry sectors (Brockhaus et al., 2013). However, AFOLU mitigation activities can also introduce or reduce clashes with existing policies in other sectors (e.g. if a conservation policy cover a forest area, where agricultural land is promoted by another policy; 11.10; Madlener et al., 2006; Smith et al., 2007; Halsnæs and Verhagen, 2007; Beach et al., 2009; Alig et al., 2010; Jackson and Baker, 2010; DeFries and Rosenzweig, 2010; Pettenella and Brotto, 2011).

An area of increasing concern since AR4 is the potential impact of AFOLU mitigation measures on ***food security***. Efforts to reduce hunger and malnutrition will increase individual food demand in many developing countries, and population growth will increase the number of individuals requiring a secure and nutritionally sufficient. Thus, a net increase in food production is an essential component for securing sustainable development (Ericksen et al., 2009; FAO, WFP, and IFAD, 2012). AFOLU mitigation measures linked to increases in food production (e.g. agroforestry, intensification of agricultural production or integrated systems) can increase food availability and access especially at the local level, while other measures (e.g. forest or energy crop plantations) can reduce food production at least locally (Foley et al., 2005; McMichael et al., 2007; Pretty, 2008; Godfray et al., 2010; Jackson and Baker, 2010; Jeffery et al., 2011; Graham-Rowe, 2011).

Regarding ***human health*** reduced emissions from agriculture and forestry may also improve air, soil and water quality (Smith et al., 2013a), thereby indirectly providing benefits to human health and well-being. Demand-side measures aimed at reducing the proportion of livestock products in human diets that are high in animal products are also associated with multiple health benefits (McMichael et al., 2007; Stehfest et al., 2009; Marlow et al., 2009). AFOLU mitigation measures, particularly in the livestock sector, can have an impact on ***animal welfare*** (Sundrum, 2001; Lund and Algers, 2003; Keeling et al., 2011; Kehlbacher et al., 2012; Koknaroglu and Akunal, 2013).

A major area of concern is related to the potential impacts of AFOLU mitigation measures on ***equity*** (3.3, 4.2, 4.7 and 4.8). Depending on the actual and perceived distribution of socio-economic benefits, responsibilities (burden-sharing), as well the access to decision-making, financing mechanisms and technology, AFOLU mitigation measures can promote inter- and intra- generational equity (Di Gregorio et al., 2013). Conversely, depending on the policy instruments and the implementation schemes of these mitigation measures, they can increase inequity and land conflicts, or marginalize small scale farm/forest owners or users (Robinson et al., 2011; Kiptot et al., 2012; Huettner, 2012; Mattoo and Subramanian, 2012). Potential impacts on equity and benefit-sharing mechanisms arise for AFOLU activities using forestry measures in developing countries including conservation, restoration, reduced deforestation and degradation, as well as sustainable management and afforestation/reforestation (Combes Motel et al., 2009; Cattaneo et al., 2010; Rosemary, 2011).

Large-scale land acquisition (often referred as “land grabbing”) related to the promotion of AFOLU mitigation measures (especially for production of bioenergy crops) and its links to sustainable development in general, and equity in particular, are emerging issues in the literature (Cotula et al., 2009; Scheidel and Sorman, 2012; Mwakaje, 2012; Messerli et al., 2013; German et al., 2013).

In many cases, the implementation of agricultural and forestry systems with positive impacts mitigating climate change are limited by capital, and ***carbon payments or compensation mechanisms*** may provide a new source of finance (Tubiello et al., 2009). For instance, in some cases, mitigation payments can help to make production of non-timber forest products (NTFP) economically viable, further ***diversifying income*** at the local level (Singh, 2008). However, depending on the accessibility of the financing mechanisms (payments, compensation or other) economic benefits can become concentrated, marginalizing many local stakeholders (Combes Motel et al., 2009; Alig et al., 2010; Asante et al., 2011; Asante and Armstrong, 2012; 11.8). The realisation of economic co-benefits is related to the design of the specific mechanisms and depends upon three main variables a) the amount and coverage of these payments, b) the recipient of the payments and c) timing of payments (*ex-ante* or *ex-post*; Corbera and Brown, 2008; Skutsch et al., 2011)**.** Further considerations on financial mechanisms and carbon payments, both within and outside UNFCCC agreements, are described in 11.10.

Financial flows supporting AFOLU mitigation measures (e.g. those resulting from the REDD+) can have positive effects on conserving biodiversity, but could eventually create conflicts with conservation of biodiversity hotspots, when their respective carbon stocks are low (Gardner et al., 2012; 11.10). Some authors propose that carbon payments can be complemented with biodiversity payments as an option for reducing trade-offs with biodiversity conservation (Phelps et al., 2010a). Bundling of ecosystem service payments, and links to carbon payments, is an emerging area of research (Deal and White, 2012).

11.7.2 Environmental effects

Availability of land and land competition can be affected by AFOLU mitigation measures. Different stakeholders may have different views on what land is available, and when considering several AFOLU mitigation measures for the same area, there can be different views on the importance of the goods and ecosystem services provided by the land, e.g. some AFOLU measures can increase food production but reduce water availability or other environmental services. Thus decision makers need to be aware of potential site-specific trade-offs within the sector. A further potential adverse side-effect is that of increasing land rents and food prices due to a reduction in land availability for agriculture in developing countries (Muller, 2009; Smith et al., 2010, 2013b; Rathmann et al., 2010;

Godfray et al., 2010; de Vries and de Boer, 2010; Harvey and Pilgrim, 2011; Amigun et al., 2011; Janzen, 2011; Cotula, 2012; Scheidel and Sorman, 2012; Haberl et al., 2013a).

AFOLU mitigation options can promote conservation of **biological diversity** (Smith et al., 2013a) both by reducing deforestation (Chhatre et al., 2012; Murdiyarso et al., 2012; Putz and Romero, 2012; Visseren-Hamakers et al., 2012), and by using reforestation/afforestation to restore biodiverse communities on previously developed farmland (Harper et al., 2007). However, promoting land use changes (e.g. through planting monocultures on biodiversity hot spots) can have adverse side-effects, reducing biodiversity (Koh and Wilcove, 2008; Beringer et al., 2011; Pandit and Grumbine, 2012; Ziv et al., 2012; Hertwich, 2012; Gardner et al., 2012).

In addition to potential climate impacts, land-use intensity drives the three main N loss pathways (nitrate leaching, denitrification and ammonia volatilization) and typical ***N balances*** for each land use indicate that total N losses also increase with increasing land-use intensity (Stevenson et al., 2010). Leakages from the N cycle can cause air (e.g. NH_3^+, NO_x) [10], soil (NO_3^-) and water pollution (e.g. eutrophication) and agricultural intensification can lead to a variety of other adverse environmental impacts (Smith et al., 2013a). Combined strategies (e.g. diversified crop rotations and organic N sources) or single-process strategies (e.g. reduced N rates, nitrification inhibitors, and changing chemical forms of fertilizer) can reduce N losses (Bambo et al., 2009; Gardner and Drinkwater, 2009). Integrated systems may be an alternative approach to reduce leaching (see also 11.10).

AFOLU mitigation measures can have either positive or negative ***impacts on water resources***, with responses dependant on the mitigation measure used, site conditions (e.g. soil thickness and slope, hydrological setting, climate; (Yu et al., 2013) and how the particular mitigation measure is managed. There are two main components; water yield and water quality. Water yields can be manipulated with forest management, through afforestation, reforestation, forest thinning or deforestation. In general, reduction in water yields in afforestation / reforestation projects has been reported in both groundwater or surface catchments (Jackson et al., 2005), or where irrigation water is used to produce bioenergy crops. For water supply security it is important to consider the relative yield reduction and this can have have severe consequences in dry regions with inherent water shortages (Wang et al., 2011c). Where there is a water imbalance, however, this additional water use can be beneficial by reducing the efflux of salts (Jackson et al., 2005). Another aspect of water yield is the reduction of flood peaks, and also prolonged periods of water flow, because discharge is stabilised (Jackson et al., 2005), however low flows can be reduced because of increased forest water use. Water quality can be affected by AFOLU in several ways. For example, minimum tillage systems have been reported to reduce water erosion and thus sedimentation of water courses (Lal, 2001). Deforestation is well known to increase erosion and thus efflux of silt; avoiding deforestation will prevent this. In other situations, watershed scale reforestation can result in the restoration of water quality (e.g., Townsend et al., 2012). Furthermore, strategic placement of tree belts in lands affected by dryland salinity can remediate the affected lands by lowering the water table (Robinson et al., 2004). Various types of AFOLU mitigation can result in degradation of water sources through the losses of pesticides and nutrients to water (Smith et al., 2013a).

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[10] Please see section 7.9.2 and WGII chapter 11.9 for a discussion of health effects related to air pollution.

Table 11.9: Summary of potential co-benefits (green arrows) and adverse side-effects (orange arrows) from AFOLU mitigation measures; arrows pointing up/down denote positive/negative effect on the respective issue. These effects depend on the specific context (including bio-physical, institutional and socio-economic aspects) as well as on the scale of implementation. For an assessment of macroeconomic, cross-sectoral effects associated with mitigation policies (e.g., on energy prices, consumption, growth, and trade), see Sections 3.9, 6.3.6, 13.2.2.3 and 14.4.2. Note: Co-benefits/adverse side-effects of bioenergy are discussed in 11.13.

	Issue	Potential co-benefit or adverse-side effect	Scale	AFOLU mitigation measure
Institutional	Land tenure and use rights	Improving (↑) or diminishing (↓) tenure and use rights for local communities and indigenous peoples, including harmonization of land tenure and use regimes (e.g. with customary rights)	Local to national	Forestry (4, 5, 6, 8, 9, 21)
	Sectoral policies	Promoting (↑) or contradicting (↓) the enforcement of sectoral (forest and/or agriculture) policies	National	Forestry (5, 6, 9, 2, 21); land-based agriculture (7, 20, 21)
	Cross-sectoral policies	Cross-sectoral coordination (↑) or clashes (↓) between forestry, agriculture, energy and/or mining policies	Local to national	Forestry (7, 21); agriculture (7, 20, 21)
	Participative mechanisms	Creation/use of participative mechanisms (↑) for decision-making regarding land management (including participation of various social groups e.g. indigenous peoples or local communities)	Local to national	Forestry (4, 5, 6, 8, 9, 21); agriculture (21, 33); integrated systems (21, 35)
	Benefit sharing mechanisms	Creation/use of benefits-sharing mechanisms (↑) from AFOLU mitigation measures	Local to national	Forestry (4, 5, 6, 21,8)
Social	Food security	Increase (↑) or decrease (↓) on food availability and access	Local to national	Forestry (18, 19); agriculture (7, 18, 19,15, 24, 29, 31); livestock (2, 3, 19, 36, 37); integrated systems (18,19); biochar (17, 27)
	Local/traditional knowledge	Recognition (↑) or denial (↓) of indigenous and local knowledge in managing (forest/agricultural) land	Local/sub-national	Forestry (4, 5, 6, 21, 8), agriculture (21,29); integrated systems (2); livestock (2, 3, 36); biochar (2)
	Animal welfare	Changes in perceived or measured animal welfare (perceived due to cultural values or measured e.g. through amount of stress hormones)	Local to national	Livestock (32, 2, 36, 38, 39)
	Cultural values	Respect and value cultural habitat and traditions (↑), reduce (↓) or increase (↑) existing conflicts or social discomfort (4, 5, 6, 21, 8)	Local to trans-boundary	Forestry (4, 5, 6, 9, 21)
	Human health	Impacts on health due to dietary changes specially in societies with a high consumption of animal protein (↓)	Local to global	Changes in demand patterns (32, 37)
	Equity	Promote (↑) or not (↓) equal access to land, decision-making, value chain and markets as well as to knowledge and benefit-sharing mechanisms	Local to global	Forestry (4, 5, 6, 21, 8, 9); agriculture (20, 24, 33)
Economic	Income	Increase (↑) or decrease (↓) in income. There are concerns regarding income distribution (↑)	Local	Forestry (6, 7, 8, 16, 21, 22, 23); agriculture (16, 19, 21, 24, 29); livestock (2, 3); integrated systems (7, 21); biochar (25); changes in demand patterns (2)
	Employment	Employment creation (↑) or reduction of employment (especially for small farmers or local communities) (↓)	Local	Forestry (8, 21), agriculture (21, 24); livestock (2, 3); integrated systems (7, 21)
	Financing mechanisms	Access (↑) or lack of access (↓) to new financing schemes	Local to global	Forestry (6, 8, 16, 21); agriculture (16, 21); livestock (2, 3)

	Economic activity	Diversification and increase in economic activity (↑) while concerns on equity (↑)	Local	Forestry (6, 7, 21, 8); land based agriculture (16, 19, 21, 24, 29); livestock (2, 3)
Environmental	Land availability	Competition between land uses and risk of activity or community displacement (↑)	Local to trans-boundary	Forestry and land based-agriculture (5, 6, 15, 18, 21, 30, 31); livestock (2, 3, 30, 41)
	Biodiversity	Monocultures can reduce biodiversity (↓). Ecological restoration increases biodiversity and ecosystem services (↑) by 44 and 25% respectively (28) Conservation, forest management and integrated systems can keep biodiversity (↑) and/or slow desertification (↓)	Local to trans-boundary	Forestry (1, 21, 19, 28) On conservation and forest management (1, 19, 22, 28, 31); agriculture and integrated systems (15, 19, 21, 29, 31);
	Albedo	Positive impacts (↑) on albedo and evaporation and interactions with ozone	Local to global	See 11.5
	N and P cycles	Impacts on N and P cycles in water (↓/↑) especially from monocultures or large agricultural areas	Local to trans-boundary	Agriculture (19, 24, 31, 36); livestock (2, 3, 31)
	Water resources	Monocultures and /or short rotations can have negative impacts on water availability (↓). Potential water depletion due to irrigation (↓). Some management practices can support regulation of the hydrological cycle and protection of watersheds (↑)	Local to trans-boundary	Forestry (1, 21, 19, 28); land based agriculture (31, 44); integrated systems (2, 31, 44)
	Soil	Soil conservation (↑) and improvement of soil quality and fertility (↑). Reduction of erosion. Positive or negative carbon mineralization priming effect (↑/↓)	Local	Forestry (45, 46) Land- based agriculture (13, 19, 24, 29, 31), integrated systems biochar (40, 41)
	New products	Increase (↑) or decrease (↓) on fibre availability as well as non-timber/non-wood products output	Local to national	Forestry (18,19, 43); agriculture (7, 18, 19,15, 24, 29, 31); integrated systems (18, 19)
	Ecosystem resilience	Increase (↑) or reduction (↓) of resilience, reduction of disaster risks (↓)	Local to trans-boundary	Forestry, integrated systems (see 11.5)
Technology	Infrastructure	Increase (↑) or decrease (↓) in availability of and access to infrastructure. Competition for infrastructure for agriculture (↑), can increase social conflicts	Local	Agriculture (21, 47, 48, 49)
	Technology innovation and transfer	Promote (↑) or delay (↓) technology development and transfer	Local to global	Forestry (7, 13, 26); agriculture (24), livestock (2, 3)
	Technology acceptance	Can facilitate acceptance of sustainable technologies (↑)	Local to national	Forestry (7, 13, 26); livestock (2, 3, 36)

Notes: AFOLU mitigation measures are grouped following the structure given in table 11.2

Sources: [1]Trabucco et al. (2008); [2]Steinfeld et al. (2010); [3]Gerber et al. (2010); [4]Sikor et al. (2010); [5]Rosemary (2011); [6]Pettenella and Brotto (2011); [7]Jackson and Baker (2010); [8]Corbera and Schroeder (2011); [9]Colfer (2011); [10]Blom et al. (2010); [11]Halsnæs and Verhagen (2007); [12]Larson (2011); [13]Lichtfouse et al. (2009); [14]Thompson et al. (2011); [15]Graham-Rowe (2011); [16]Tubiello et al. (2009); [17]Barrow (2012); [18]Godfray et al. (2010); [19]Foley et al. (2005); [20]Halsnæs and Verhagen (2007) ; [21]Madlener et al. (2006); [22]Strassburg et al. (2012) ; [23]Canadell and Raupach (2008) ; [24]Pretty (2008); [25]Galinato et al. (2011); [26]Macauley and Sedjo (2011); [27]Jeffery et al. (2011); [28]Benayas et al. (2009); [29]Foley et al. (2011); [30]Haberl et al. (2013); [31]Smith et al. (2013a); [32]Stehfest et al. (2009); [33]Chhatre et al. (2012); [34]Seppälä et al. (2009); [35]Murdiyarso et al. (2012); [36]de Boer et al. (2011); [37]McMichael et al. (2007); [38]Koknaroglu and Akunal (2013); [39]Kehlbacher et al. (2012); [40]Zimmerman et al. (2011); [41]Luo et al. (2011); [42]Mirle (2012); [43]Albers and Robinson (2013); [44]Smith et al. (2013b); [45]Chatterjee and Lal (2009); [46]Smith (2008); [47]Ziv et al. (2012); [48]Beringer et al. (2011); [49]Douglas et al. (2009)

AFOLU mitigation measures can have several ***impacts on soil***. Increasing or maintaining carbon stocks in living biomass (e.g. through forest or agroforestry systems) will reduce wind erosion by acting as wind breaks and may increase crop production; and reforestation, conservation, forest management, agricultural systems or bioenergy systems can be used to restore degraded or abandoned land (Smith, 2008; Stickler et al., 2009; Chatterjee and Lal, 2009; Wicke et al., 2011b; Sochacki et al., 2012). Silvo-pastoral systems can help to reverse land degradation while providing food (Steinfeld et al., 2008, 2010; Janzen, 2011). Depending on the soil type, production temperature regimes, the specific placement and the feedstock tree species, biochar can have positive or negative carbon mineralization priming effects over time (Zimmerman et al., 2011; Luo et al., 2011).

AFOLU mitigation options can promote innovation, and many technological supply-side mitigation options outlined in 11.3 also increase agricultural and silvicultural efficiency. At any given level of demand for agricultural products, intensification increases output per unit area and would therefore, if all else were equal, allow the reduction in farmland area which would in turn free land for C sequestration and/or bioenergy production (11.4). For example, a recent study calculated potentially large GHG reductions from global agricultural intensification by comparing the past trajectory of agriculture (with substantial yield improvements), with a hypothetical trajectory with constant technology (Burney et al., 2010). However, in real-world situations increases in yield may result in feedbacks such as increased consumption (“rebound effects”; see Section 11.4; Lambin and Meyfroidt, 2011; Erb, 2012).

11.7.3 Public perception

Mitigation measures which support sustainable development are likely to be viewed positively in terms of public perception, but a large scale drive towards mitigation without inclusion of key stakeholder communities involved would likely not be greeted favourably (Smith and Wollenberg, 2012). However, there are concerns about competition between food and AFOLU outcomes, either because of an increasing use of land for biofuel plantations (Fargione et al., 2008a; Alves Finco and Doppler, 2010), or afforestation/reforestation (Mitchell et al., 2012), or by blocking the transformation of forest land into agricultural land (Harvey and Pilgrim, 2011).

Further, lack of clarity regarding the architecture of the future international climate regime and the role of AFOLU mitigation measures is perceived as a potential threat for long-term planning and long-term investments (Streck, 2012; Visseren-Hamakers et al., 2012). Certain technologies, such as animal feed additives and genetically modified organisms are banned in some jurisdictions due to perceived health and/or environmental risks. Public perception is often as important as scientific evidence of hazard / risk in considering government policy regarding such technologies (Royal Society, 2009; Smith and Wollenberg, 2012).

11.7.4 Spillover effects

Emerging knowledge on the importance of ecosystems services as a means for addressing climate change mitigation and adaptation have brought attention to the role of ecosystem management for achieving several development goals, beyond climate change adaptation and mitigation. This knowledge has enhanced the creation of ecosystem markets (11.10). In some jurisdictions ecosystem markets are developing (MEA, 2005; Engel et al., 2008; Deal and White, 2012; Wünscher and Engel, 2012) and these allow valuation of various components of land-use changes, in addition to carbon mitigation (Mayrand and Paquin, 2004; Barbier, 2007). Different approaches are used; in some cases the individual components (both co-benefits and adverse side-effects) are considered singly (bundled), in other situations they are considered together (stacked); (Deal and White, 2012). Ecosystem market approaches can serve as a framework to assess the benefits of mitigation actions from project, to regional and national level (Farley and Costanza, 2010). Furthermore, designing ecosystem market approaches yields methodologies for the evaluation of individual components (e.g. water quality response to reforestation, timber yield), and other types of ecosystem service (e.g. biodiversity, social amenity; Bryan et al., 2013).

11.8 Barriers and opportunities

Barriers and opportunities refer to the conditions provided by the development context (see 11.4.5). These conditions can enable and facilitate (opportunities) or hinder (barriers) the full use of AFOLU mitigation measures. AFOLU programmes and policies can help to overcome barriers, but countries being affected by many barriers will need time, financing and capacity support. In some cases, international negotiations have recognised these different circumstances among countries and have proposed corresponding approaches (e.g. a phased approach in the REDD+, Green Climate Fund; 11.10). Corresponding to the development framework presented in 11.4.5, the following types of barriers and benefits are discussed: socio-economic, environmental, institutional, technological and infrastructural.

11.8.1 Socio-economic barriers and opportunities

The ***design and coverage of the financing mechanisms*** is key to successfully use the AFOLU mitigation potential (see 11.10 and chapter 16). Questions remain over which costs will be covered by such mechanisms. If financing mechanisms fail to cover at least transaction and monitoring costs, they will become a barrier to the full implementation of AFOLU mitigation. According to some studies, opportunity costs also need to be fully covered by any financing mechanism for the AFOLU sector, especially in developing countries, as otherwise AFOLU mitigation measures would be less attractive compared to returns from other land uses (Angelsen, 2008; Cattaneo et al., 2010; Böttcher et al., 2012). Conversely, if financing mechanisms are designed to modify economic activity, they could provide an opportunity to leverage a larger proportion of AFOLU mitigation potential.

Scale of financing sources can become either a barrier (if a relevant financial volume is not secured) or create an opportunity (if financial sources for AFOLU suffice) for using AFOLU mitigation potential (Streck, 2012); see chapter 16). Another element is the ***accessibility to AFOLU financing*** for farmers and forest stakeholders (Tubiello et al., 2009; Havemann, 2011; Colfer, 2011). Financial concerns, including reduced access to loan and credits, high transaction costs or reduced income due to price changes of carbon credits over the project duration, are potential risks for AFOLU measures, especially in developing countries, and when land holders use market mechanisms (e.g. A/R CDM; (Madlener et al., 2006).

Poverty is characterized not only by low income, but also by insufficient food availability in terms of quantity and/or quality, limited access to decision making and social organization, low levels of education and reduced access to resources (e.g. land or technology; UNDP International Poverty Centre, 2006). High levels of poverty can limit the possibilities for using AFOLU mitigation options, because of short-term priorities and lacking reources. In addition, poor communities have limited skills and sometimes lack of social organization that can limit the use, and scaling up of, AFOLU mitigation options, and can increase the risk of displacement, with other potential adverse side-effects (Smith and Wollenberg, 2012; Huettner, 2012). This is especially relevant when forest land sparing competes with other development needs e.g. increasing land for agriculture or promoting some types of mining (Forner et al., 2006), or when large scale bioenergy compromises food security (Nonhebel, 2005) and 11.13.

Cultural values and social acceptance can determine the feasibility of AFOLU measures, becoming a barrier or an opportunity depending of the specific circumstances (de Boer et al., 2011).

11.8.2 Institutional barriers and opportunities

Transparent and accountable governance and swift institutional establishment are very important for a sustainable implementation of AFOLU mitigation measures. This includes the need to have

clear land tenure and land use rights regulations and a certain level of enforcement, as well as clarity about carbon ownership (see 11.4.5 and 11.10, and Chapters 14 and 15; Palmer, 2011; Thompson et al., 2011; Markus, 2011; Rosendal and Andresen, 2011; Murdiyarso et al., 2012).

Lack of institutional capacity (as a means for securing creation of equal institutions among social groups and individuals) can reduce feasibility of AFOLU mitigation measures in the near future, especially in areas where small-scale farmers or forest users are the main stakeholders (Laitner et al., 2000; Madlener et al., 2006; Thompson et al., 2011). ***Lack of an international agreement*** that supports a wide implementation of AFOLU measures can become a major barrier for realizing the mitigation potential from the sector globally (see 11.10 and chapter 13).

11.8.3 Ecological barriers and opportunities

Mitigation potential in the agricultural sector is highly site-specific, even within the same region or cropping system (Baker et al., 2007; Chatterjee and Lal, 2009). ***Availability of land and water*** for different uses need to be balanced, considering short- and long-term priorities, and global differences in resource use. Consequently, limited resources can become an ecological barrier and the decision of how to use them needs to balance ecological integrity and societal needs (Jackson, 2009).

At the local level, the ***specific soil conditions, water availability, GHG emission reduction potential as well as natural variability and resilience*** to specific systems will determine the level of relaisation of mitigation potential of each AFOLU measure (Baker et al., 2007; Halvorson et al., 2011). Frequent droughts in Africa and changes in the hydro-meteorological events in Asia and Central and South America are important in defining the specific regional potential (Bradley et al., 2006; Rotenberg and Yakir, 2010). Ecological saturation (e.g. soil carbon or yield) means that some AFOLU mitigation options have their own limits (11.5). The fact that many ***AFOLU measures can provide adaptation benefits*** provides an opportunity for increasing ecological efficiency (Guariguata et al., 2008; van Vuuren et al., 2009; Robledo et al., 2011; 11.5).

11.8.4 Technological barriers and opportunities

Technological barriers refer to the limitations in generating, procuring and applying science and technology to identify and solve an environmental problem. Some mitigation technologies are already applied now (e.g. afforestation, cropland and grazing land management, improved livestock breeds and diets) so there are no technological barriers for these options, but others (e.g. some livestock dietary additives, crop trait manipulation) are still at the development stage (see Table 11.2). The ***ability to manage and re-use knowledge assets*** for scientific communication, technical documentation and learning is lacking in many areas where mitigation could take place. Future developments present opportunities for additional mitigation to be realised if efforts to deliver ease-of-use and range-of-use are guaranteed. There is also a need to adapt technology to local needs by focussing on existing local opportunities (Kandji et al., 2006), as proposed in Nationally Appropriate Mitigation Actions (NAMAs; see 11.10).

Barriers and opportunities related to ***monitoring, reporting and verification*** of the progress of AFOLU mitigation measures also need be considered. Monitoring activities, aimed at reducing uncertainties, provide the opportunity of increasing credibility in the AFOLU sector. However there are technical challenges. For instance, monitoring carbon in forests with high spatial variability in species composition and tree density can pose a technical barrier to the implementation of some AFOLU activities (e.g. REDD+; (Baker et al., 2010); see 11.10). The IPCC National Greenhouse Gas Inventory Guidelines (Paustian et al., 2006) also provide an opportunity, because they offer standard scientific methods that countries already use to report AFOLU emissions and removals under the UNFCCC. Also, field research in high-biomass forests (Gonzalez et al., 2010) shows that remote sensing data and Monte Carlo quantification of uncertainty offer a technical opportunity for implementing REDD+ (11.10). Expoliting the existing ***human skills*** within a country is essential for realising full AFOLU potential. A lack of trained people can therefore become a barrier to implementation of appropriate technologies (Herold and Johns, 2007).

Technology improvement and technology transfer are two crucial components for the sustainable increase of agricultural production in developed and developing regions with positive impacts in terms of mitigation, soil and biodiversity conservation (Tilman et al., 2011). International and national policy instruments are relevant to foster technology transfer and to support research and development (see 11.10.4), overcoming technological barriers.

11.9 Sectoral implications of transformation pathways and sustainable development

Some climate change management objectives require large-scale transformations in human societies, in particular in the production and consumption of energy and the use of the land resource. Chapter 6 describes alternative “transformation pathways” of societies over time from now into the future, consistent with different climate change outcomes. Many pathways that foresee large efforts in mitigation will have implications for sustainable development, and corrective actions to move toward sustainability may be possible. However, impacts on development are context specific and depend upon scale and institutional agreements of the AFOLU options, and not merely on the type of option (see 11.4 for development context and systemic view, 11.7 for potential co-benefits and adverse effects, and 11.8 for opportunities and challenges). To evaluate sectoral implications of transformation pathways, it is useful to first characterise the pathways in terms of mitigation technologies and policy assumptions.

11.9.1 Characterisation of transformation pathways

Uncertainty about reference AFOLU emissions is significant both historically (see 11.2) and in projections (see 6.3.1.3). The transformation projections of the energy system, AFOLU emissions and land-use are characterized by the reference scenario, as well as the abatement policy assumptions regarding eligible abatement options, regions covered, and technology costs over time. Many transformation scenarios suggest a substantial cost-effective mitigation role for land related mitigation assuming idealized policy implementation, with immediate, global, and comprehensive availability of land-related mitigation options. However, policy implementation of large-scale land-based mitigation will be challenging. In addition, the transformation pathways often ignore, or only partially cover, important mitigation risks, costs and benefits (e.g. transaction costs or Monitoring Reporting and Verification [MRV] costs), and other developmental issues including intergenerational debt or non-monetary benefits (Ackerman et al., 2009; Lubowski and Rose, 2013).

In recent idealized implementation scenarios from a model comparison study, land-related mitigation represents a significant share of emissions reductions (Table 11.10). In these scenarios, as described in 6.3.2 and 6.3.5, models assume an explicit terrestrial carbon stock incentive, or a global forest protection policy, as well as an immediate global mitigation policy in general. Bioenergy is consistently deployed (because it is considered to reduce net GHG emissions over time; see Section 6.3.5), and agricultural emissions are priced. The largest land emission reductions occur in net CO_2 emissions, which also have the greatest variability across models. Some models exhibit increasing land CO_2 emissions under mitigation, as bioenergy feedstock production leads to land-use change, while other models exhibit significant reductions with protection of existing terrestrial carbon stocks and planting of new trees to increase carbon stocks. Land-related CO_2 and N_2O mitigation is more important in the nearer-term for some models. Land-related N_2O and CH_4 reductions are a significant part of total N_2O and CH_4 reductions, but only a small fraction of baseline emissions, suggesting that models have cost-effective reasons to keep N_2O and CH_4 emissions. Emissions reductions from land increase only slightly with the stringency of the atmospheric concentration goal, as energy and industry emission reductions increase faster with target stringency. This result is consistent with previous studies (Rose et al., 2012). Land-based CO_2 reductions can be over 100% of baseline emissions, from the expansion of managed and unmanaged forests for sequestration.

Emissions reductions from individual land-related technologies, especially bioenergy, are not generally reported in transformation pathway studies. In part, this is due to emphasis on the energy system, but also other factors that make it difficult to uniquely quantify mitigation by technology. An exception is (Rose et al., 2012) who reported agriculture, forest carbon, and bioenergy abatement levels for various atmospheric concentration goals. Cumulatively, over the century, bioenergy was the dominant strategy, followed by forestry, and then agriculture. Bioenergy cummulatively generated approximately 5 to 52 and 113 to 749 Gt CO_2eq mitigation by 2050 and 2100, respectively.

Within models, there is a positive correlation between emissions reductions and GHG prices. However, across models, it is less clear, as some estimate large reductions with a low GHG price, while others estimate low reductions despite a high GHG price (Rose et al., 2012). For the most part, these divergent views are due to differences in model assumptions and are difficult to disentangle. Overall, while a tighter target and higher carbon price results in a decrease in land-use emissions, emissions decline at a decreasing rate. This is indicative of the rising relative cost of land mitigation, the increasing demand for bioenergy, and subsequent increasing need for overall energy system GHG abatement and energy consumption reductions.

Table 11.10: Cumulative land-related emissions reductions, land reduction share of global reductions, and percent of baseline land emissions reduced for CH_4, CO_2, and N_2O in idealized implementation 550 and 450 CO_2-eq ppm scenarios. The number of scenarios is indicated for each GHG and atmospheric concentration goal. Negative values represent increases in emissions. (Kriegler et al., 2013b).

			550ppm CO_2eq			**450ppm CO_2eq**		
			2010-2030	**2010-2050**	**2010-2100**	**2010-2030**	**2010-2050**	**2010-2100**
Cumulative global land-related emissions reductions [$GtCO_2eq$]	**CH_4**	**min**	3.5	17.5	51.4	0.0	4.5	52.3
	(n=5/5)	**max**	9.8	46.0	201.7	12.7	50.5	208.6
	CO_2	**min**	-20.2	-43.2	-129.8	-20.3	-50.8	-153.9
	(n=11/10)	**max**	280.9	543.0	733.4	286.6	550.5	744.6
	N_2O	**min**	3.1	8.4	25.5	3.1	8.4	25.5
	(n=4/4)	**max**	8.2	27.7	96.6	9.7	29.3	96.8
Land reductions share of total global emissions reductions	**CH_4**	**min**	25%	20%	20%	22%	20%	16%
		max	37%	40%	42%	30%	31%	36%
	CO_2	**min**	-43%	-12%	-4%	-20%	-8%	-4%
		max	74%	48%	17%	73%	47%	15%
	N_2O	**min**	52%	61%	65%	53%	61%	65%
		max	95%	90%	87%	78%	83%	85%
Percent of baseline land emissions reduced	**CH_4**	**min**	3%	8%	10%	0%	2%	10%
		max	8%	16%	28%	10%	18%	30%
	CO_2	**min**	-42%	-89%	0%	-42%	-104%	0%
		max	373%	417%	504%	381%	423%	512%
	N_2O	**min**	4%	6%	8%	4%	6%	8%
		max	10%	16%	22%	12%	17%	22%

Models project increased deployment of, and dependence on, modern bioenergy (i.e. non-traditional bioenergy that is produced centrally to service communities rather than individual household production for heat and cooking), with some models projecting up to 95 EJ per year by 2030, and up to 245 EJ per year by 2050. Models universally project that the majority of agriculture and forestry mitigation, and bioenergy primary energy, will occur in developing and transitional economies (6.3.5).

More recently, the literature has begun analysing more realistic policy contexts. This work has identified a number of policy coordination and implementation issues. There are many dimensions to policy coordination: technologies, sectors, regions, climate and non-climate policies, and timing. There are three prominent issues. First, there is coordination between mitigation activities. For instance, increased bioenergy incentives without global terrestrial carbon stock incentives or global forest protection policy, could result in substantial land conversion and emissions with large-scale deployment of energy crops. The projected emissions come primarily from the displacement of pasture, grassland, and natural forest (see 6.3.5 and 11.4.3). Energy cropland expansion also results in non-energy cropland conversion. These studies find that ignoring land conversion emissions with energy crop expansion, results in the need for deeper emissions reductions in the fossil and industrial sectors, and increased total mitigation costs. However, illustrative scenarios by (Calvin et al., 2013a) suggest that extensive forest protection policies may be needed for managing bioenergy driven deforestation. Note that providing energy crops, especially while protecting terrestrial carbon stocks, could result in a significant increase in food prices, potentially further exacerbated if also expanding forests (Wise et al., 2009; Popp et al., 2011a; Reilly et al., 2012; Calvin et al., 2013; see also Sections 11.4.3 and 11.13.7). In addition to competition between energy crops and forest carbon strategies, there is also competition between avoided deforestation and afforestation mitigation strategies, but synergies between forest management and afforestation (Rose and Sohngen, 2011). Bioenergy sustainability policies across sectors also need to be coordinated (Frank et al., 2013).

The second major concern is coordination of mitigation activity over time. The analyses noted in the previous paragraph assume the ability to globally protect or incentivize all, or a portion, of forest carbon stocks. A few studies to date have evaluated the implications of staggered forest carbon incentives—across regions and forest carbon activities. For instance, (Calvin et al., 2009) estimate land CO_2 emissions increases of 4 and 6 Gt CO_2/year in 2030 and 2050, respectively, from scenarios with staggered global regional climate policies that include forest carbon incentives. And, (Rose and Sohngen, 2011) find that fragmented or delayed forest carbon policy could accelerate deforestation. For example, (Rose and Sohngen, 2011) project 60-100 Gt CO_2 of leakage by 2025 with a carbon price of $15/t CO_2 that rises at 5% per year. Regional agriculture and forestry mitigation supply costs are also affected by regional participation/non-participation, with non-participating regions potentially increasing the mitigation costs for participating regions (Golub et al., 2009). Staggered adoption of land mitigation policies will likely have institutional and socioeconomic implications as well (Madlener et al., 2006). Institutional issues, especially clarification of land tenure and property rights and equity issues (11.7), will also be critical for successful land mitigation in forestry over time (Palmer, 2011; Karsenty et al., 2012; Gupta, 2012).

Finally, the type of incentive structure has implications. International land-related mitigation projects are currently regarded as high risk carbon market investments, which may affect market appeal. Also, transformation scenarios assume that all emissions and sequestration changes are priced (similar to capping all emissions). However, mitigation, especially in agriculture and forestry, may be sought through voluntary markets, where mitigation suppliers choose whether to participate (11.10). For instance, Rose et al. (2013) estimated reduced mitigation potential, as well as over-crediting, for US agriculture and forestry with voluntary mitigation supply incentives, e.g. mitigation decreased 25-55% at $15/t CO_2eq due to non-participant leakage and non-additional crediting.

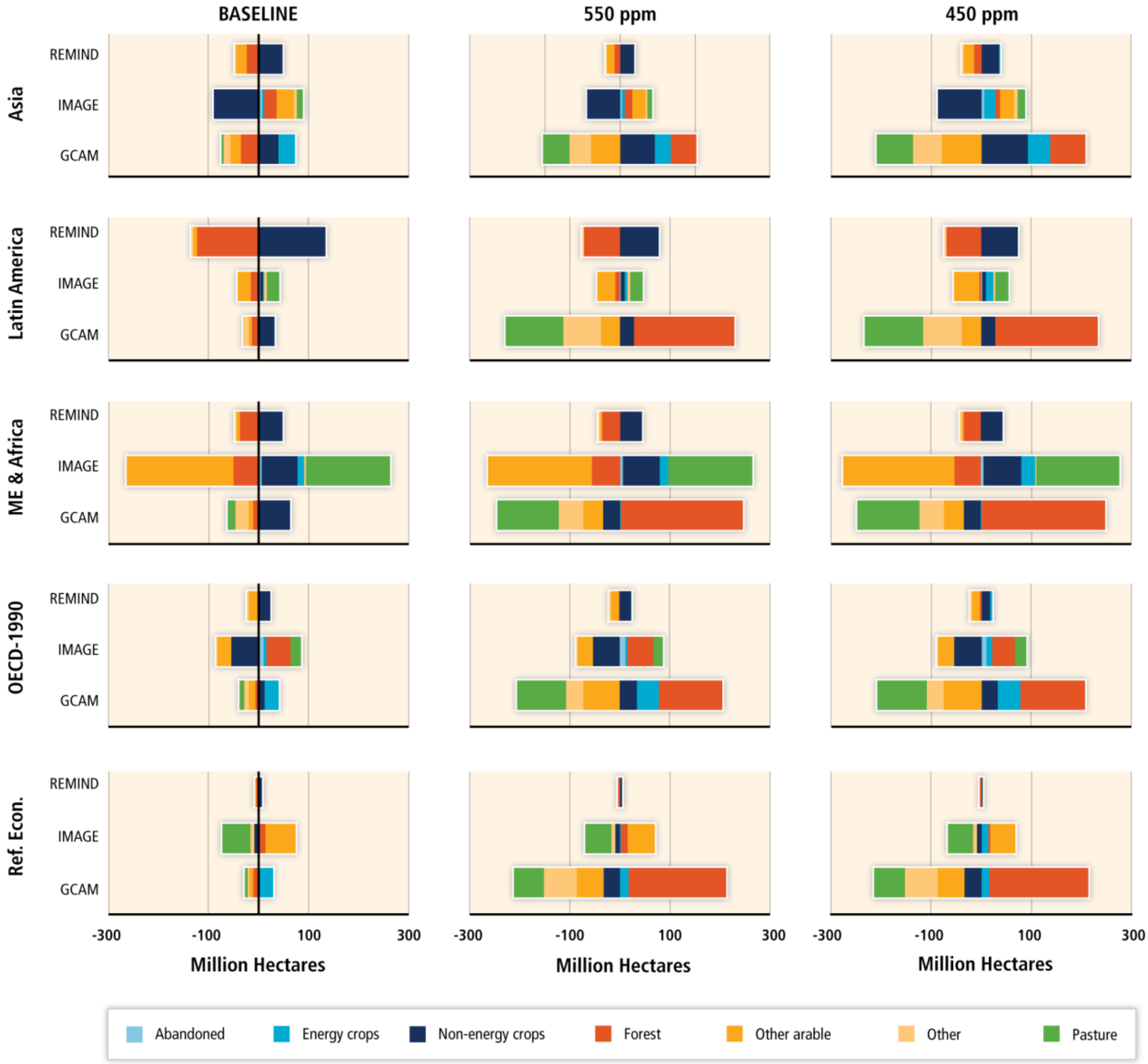

Figure 11.19. Regional land cover change by 2030 from 2005 from three models for baseline (left) and idealized policy implementation 550 CO_2eq ppm (centre) and 450 CO_2eq ppm (right) scenarios. Key: abandoned land (blue), non-energy crops (red), energy crops (green), forest (black), other arable land (land that could be used for crops or pasture that is not used - yellow), other land (orange), and pasture (light blue) (Popp et al., 2013).

11.9.2 Implications of transformation pathways for the AFOLU sector

Transformation pathways indicate that a combination of forces can result in very different projected landscapes relative to today, even in baseline scenarios (6.3.5). For instance, Popp et al. (2013) evaluate three models, and show that projected 2030 baseline changes from today alone vary sharply across models in all regions (Figure 11.19). See Section 6.3.5 for global land cover change results for a broader set of studies and policy contexts. In the examples in Figure 11.19, projections exhibit growth and reductions in both non-energy cropland (e.g. Asia), and energy cropland (e.g., Asia, OECD90, REF). Furthermore, different kinds of land are converted when baseline cropland expands (e.g. MAF). Mitigation generally induces greater land cover changes than in baseline scenarios, but there are very different potential transformation visions. Overall, it is difficult to generalize on regional land cover effects of mitigation. For the same atmospheric concentration goal, some models convert significant area, some do not. There is energy cropland expansion in many regions that supports the production of bioenergy. Less consistent is the response of forest land, primarily due to differences in the land carbon options/policies modelled (see 6.3.5). Finally, there is relatively modest additional land conversion in the 450 ppm, compared to the 550 ppm, scenarios, which is consistent with the declining role of land-related mitigation with policy stringency.

The implications of transformation pathway scenarios with large regional expansion of forest cover for carbon sequestration, depends in part on how the forest area increases (Figure 11.19; Popp et al., 2013). If forest areas increase through the expansion of natural vegetation, biodiversity and a range of other ecosystem services provided by forests could be enhanced. If afforestation occurs through large scale plantation, however, some negative impacts on biodiversity, water and other ecosystem services could arise, depending on what land cover the plantation replaces and the rotation time (11.7). Similar issues arise with large scale bioenergy, and environmental impacts of energy crop plantations, which largely depend upon where, how, and at what scale they are implemented, and how they are managed (Davis et al., 2013; see Section 11.13.6). Not surprisingly, the realistic policy coordination and implementation issues discussed in 11.9.1 could have significant land use consequences, and additional policy design research is essential to better characterize mitigation costs, net emissions, and other social implications.

11.9.3 Implications of transformation pathways for sustainable development

The implications of the transformation pathways on sustainable development are context and time specific. A detailed discussion of the implications of large scale land use change, competition between different demands for land, and the feedbacks between land use change and other services provided by land is provided in 11.4, potential co-benefits and adverse side-effects are discussed in 11.7 and 6.6 compares potential co-benefits and adverse side-effects across sectors, while 11.8 presents the opportunities and barriers for promoting AFOLU mitigation activities in the future. Finally 11.13 discusses the specific implications of increasing bioenergy crops.

11.10 Sectoral policies

Climate change and different policy and management choices interact. The interrelations are particularly strong in agriculture and forestry: climate has a strong influence on these sectors which also constitute sources of greenhouse gases as well as sinks (Golub et al., 2009). The land provides a multitude of ecosystem services, greenhouse gas mitigation being just one of many services that are vital to human wellbeing. The nature of the sector means that there are, potentially, many barriers and opportunities as well as a wide range of potential impacts related to the implementation of AFOLU mitigation options (11.7 and 11.8). Successful mitigation policies need to consider how to address the multi-functionality of the sector. Furthermore, physical environmental limitations are central for the implementation of mitigation options and associated policies (Pretty, 2013). The cost-effectiveness of different measures is hapmered by regional variability. National and international agricultural and forest climate policies have the potential to redefine the opportunity costs of international land-use in ways that either complement or hinder the attainment of climate change mitigation goals (Golub et al., 2009). Policy interactions could be synergistic (e.g. research and development investments and economic incentives for integrated production systems) or conflicting (e.g. policies promoting land conversion *vs*. conservation policies) across the sector (see Table 11.11). Additionally, adequate policies are needed to orient practices in agriculture and in forestry toward global sharing of innovative technologies for the efficient use of land resources, to support effective mitigation options (see Table 11.2).

Forty-three countries in total (as of December 2010) have proposed Nationally Appropriate Mitigation Actions (NAMAs) to the UNFCCC. Agriculture and forestry activities were considered as ways to reduce their GHG emissions in 59% and 94% of the proposed NAMAs. For the least developed countries, the forestry sector was quoted in all the NAMAs, while the agricultural sector was represented in 70% of the NAMAs (Bockel et al., 2010). Policies related to the AFOLU sector that affect mitigation are discussed below according to the instruments through which they may be implemented (economic incentives, regulatory and control approaches, information, communication and outreach, research and development). Economic incentives (e.g. special credit lines for low carbon agriculture, sustainable agriculture and forestry practices, tradable credits, payment for ecosystem services) and regulatory approaches (e.g. enforcement of environmental law to reduce deforestation, set-aside policies, air and water pollution control reducing nitrate load and N_2O emissions) have been effective in different cases. Investments in research, development and diffusion (e.g. improved fertilizer use effieciency, livestock improvement, better forestry management practices) could result in positive and synergistic impacts for adaptation and mitigation (11.5). Emphasis is given to REDD+, considering its development in recent years, and relevance for the discussion of mitigation policies in the forestry sector.

11.10.1 Economic Incentives

Emissions trading: Carbon markets occur under both compliance schemes and as voluntary programmes. A review of existing offset programmes was provided by (Kollmuss et al., 2010). More details are also presented in 15.5.3. Compliance markets (Kyoto offset mechanisms, mandatory cap-and-trade systems and other mandatory GHG systems) are created and regulated by mandatory national, regional or international carbon reduction regimes (Kollmuss et al., 2010). The three Kyoto Protocol mechanisms are very important for the regulatory market: Clean Development Mechanism (CDM), Joint Implementation (JI) and the Emissions Trading System (ETS). Currently, AFOLU projects in CDM only include specific types of projects: for agriculture - methane avoidance (manure management), biogas projects, agricultural residues for biomass energy; for forestry – reforestation and afforestation. By June 2013, the total number of registered CDM projects was 6989, 0.6 and 2.5% of this total being related to afforestation/reforestation and agriculture, respectively (http://cdm.unfccc.int/Statistics/); so finance streams coming from A/R CDM Projects are marginal from the global perspective. An analysis of A/R CDM projects suggests crucial factors for the performance of these projects are initial funding support, design and implementation guided by large organizations with technical expertise, occurrence on private land (land with secured property rights attached), and that most revenue from Certified Emission Reductions (CERs) is directed back to local communities (Thomas et al., 2010).

There are compliance schemes outside the scope of the Kyoto Protocol, but these are carried out exclusively at the national level, with no relation to the Protocol. In 2011, Australia started the Carbon Farming Initiative (CFI) that allows farmers and investors to generate tradable carbon offsets from farmland and forestry projects. This followed several years of State-based and voluntary activity that resulted in 65,000 ha of A/R projects (Mitchell et al., 2012). Another example is The Western Arnhem Land Fire Abatement Project (WALFA), a fire management project in Australia initiated in 2006 that produces a tradable carbon offset through the application of improved fire management using traditional management practices of indigenous land owners (Whitehead et al., 2008; Bradstock et al., 2012). Alberta's offset credit system is a compliance mechanism for entities regulated under the province's mandatory GHG emission intensity-based regulatory system (Kollmuss et al., 2010). In the case of N_2O emissions from agriculture, the Alberta Quantification Protocol for Agricultural N_2O Emissions Reductions issues C offset credits for on-farm reductions of N_2O emissions and fuel use associated with the management of fertilizer, manure, and crop residues for each crop type grown. Other N_2O emission reduction protocols (e.g., Millar et al., 2010) are being considered for the Verified Carbon Standard, the American Carbon Registry, and the Climate Action Reserve (Robertson et al., 2012).

Agriculture and Forestry activities are not covered by the European Union Emissions Trading Scheme (EU ETS), which is by far the largest existing carbon market. Forestry entered the New Zealand Kyoto Protocol compliant ETS in 2008, and mandatory reporting for agriculture began in 2012, although full entry of agriculture into the scheme has been delayed indefinitely. Agricultural participants include meat processors, dairy processors, nitrogen fertiliser manufacturers and importers, and live animal exporters, although some exemptions apply (www.climatechange.govt.nz). California's Cap-and-Trade Regulation took effect on January 1, 2012, with amendments to the Regulation effective September 1, 2012. The enforceable compliance obligation began on January 1, 2013. Four types of projects were approved as eligible to generate carbon credits to regulated emitters in California: avoidance of methane emissions from installation of anaerobic digesters on farms, carbon sequestration in urban and rural forestry, and destruction of ozone depleting substances (http://www.arb.ca.gov).

Voluntary carbon markets operate outside of the compliance markets. By enabling businesses, governments, NGOs, and individuals to purchase offsets that were created either in the voluntary market or through the CDM, they can offset their emissions (Verified or Voluntary Emissions Reductions - VERs). The voluntary offset market includes a wide range of programs, entities, standards and protocols (e.g. Community & Biodiversity Standards, Gold Standard, Plan Vivo among others) to improve the quality and credibility of voluntary offsets. The most common incentives for the quantity buyers of carbon credits in the private sector are corporate social responsibility and public relations. Forest projects are increasing in the voluntary markets. Transactions of carbon credits from this sector totalled UD$133M in 2010, 95% of them in voluntary markets (Peters-Stanley et al., 2011).

Reducing emissions from deforestation; reducing emissions from forest degradation; conservation of forest carbon stocks; sustainable management of forests; and enhancement of forest carbon stocks (REDD+): REDD+ consists of forest-related activities implemented voluntarily by developing countries that may, in isolation or jointly lead to significant climate change mitigation[11]. REDD+ was introduced in the agenda of the Climate Change Convention (UNFCCC) in 2005, and has since evolved to an improved understanding of the potential positive and negative impacts, methodological issues, safeguards, and financial aspects associated with REDD+ implementation. Here, we first address the REDD+ discussions under the UNFCCC, but also introduce other REDD+-related initiatives. The novel aspects of REDD+ under the Convention, relative to previous forest-related mitigation efforts by developing countries under the UNFCCC are its national and broader coverage, in contrast to project-based mitigation activities[12] (e.g. under the Clean Development Mechanism of the Kyoto Protocol). Its main innovation is its results-based approach, in which payments are done *ex post* in relation to a mitigation outcome already achieved, as opposed to project-based activities, where financing is provided *ex ante* in relation to expected outcomes. A phased approach to REDD+ was agreed at the UNFCCC, building from the development of national strategies or action plans, policies and measures, and evolving into results-based actions that should be fully measured, reported and verified – MRV (UNFCCC Dec. 1/16). REDD+ payments are expected for results-based actions, and

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[11] Decision 1/CP.16 (FCCC/CP/2010/7/Add.1 , paragraph 70) "Encourages developing countries to contribute to mitigation actions in the forest sector by undertaking the following activities, as deemed appropriate by each Party and in accordance with their respective capabilities and national circumstances - reducing emissions from deforestation; reducing emissions from forest degradation; conservation of forest carbon stocks; sustainable management of forests; and enhancement of forest carbon stocks".

[12] Decision 1/CP.16 (FCCC/CP/2010/7/Add.1 , paragraph 73) *Decides* that the activities undertaken by Parties referred to in paragraph 70 above should be implemented in phases, beginning with the development of national strategies or action plans, policies and measures, and capacity-building, followed by the implementation of national policies and measures and national strategies or action plans that could involve further capacity-building, technology development and transfer and results-based demonstration activities, and evolving into results-based actions that should be fully measured, reported and verified"

although the UNFCCC has already identified potential ways to pay for these[13], the financing architecture for the REDD+ mechanism is still under negotiation under the UNFCCC.

Meanwhile, and as a result to the explicit request from the UNFCCC for early actions in REDD+, different regional and global programmes and partnerships address forest management and conservation and readiness for REDD+ (Table 11.11), while some REDD+ strategies have started in countries with significant forest cover (see Box 11.7 for examples). Initiatives include multilateral activities (e.g. UN-REDD Programme, Forest Carbon Partnership Facility, Forest Investment Program), bilateral activities (e.g. Tanzania-Norway, Indonesia-Norway), country driven initiatives (in addition to 16 UN-REDD Programme countries, the Programme also supports 31 other partner countries across Africa, Asia-Pacific and Latin America and the Caribbean – (http://www.un-redd.org/AboutUNREDDProgramme/NationalProgrammes/).

REDD+ can be a very cost effective option for mitigating climate change and could supply a large share of global abatement of emissions from the AFOLU sector from the extensive margin of forestry, especially through reducing deforestation in tropical regions (Golub et al., 2009). Issues of concern for REDD+ implementation have been captured under REDD+ safeguards in line with the UNFCCC Cancun Agreement. In order to respond to the requirements outlined in the UNFCCC agreement, a number of steps need to be considered in the development of country-level safeguard information systems for REDD+ including: defining social and environmental objectives, assessing potential benefits and risks from REDD+, assessing current safeguard systems, drafting a strategic plan or policy, and establishing a governance system.

A growing body of literature has analyzed different aspects related to the implementation, effectiveness and scale of REDD+, as well as the interactions with other social and environmental co-benefits (e.g., Angelsen et al., 2008; Levin et al., 2008; Larson, 2011; Gardner et al., 2012). Results-based REDD+ actions, which are entitled to results-based finance, require internationally agreed rules for measuring, reporting and verification (MRV). Measuring and monitoring the results will most likely rely on a combination of remotely-sensed data with ground-based inventories. The design of a REDD policy framework (and specifically its rules) can have a significant impact on monitoring costs (Angelsen et al., 2008; Böttcher et al., 2009). Forest governance is another central aspect in recent studies, including debate on decentralization of forest management, logging concessions in public owned commercially valuable forests, and timber certification, primarily in temperate forests (Agrawal et al., 2008). Although the majority of forests continue to be formally owned by governments, there are indications that the effectiveness of forest governance is increasingly independent of formal ownership (Agrawal et al., 2008). However, there are widespread concerns that REDD+ will increase costs on forest-dependent peoples and in this context, stakeholders rights, including rights to continue sustainable traditional land use practices, appear as a precondition for REDD development (Phelps et al., 2010b).

Some studies have addressed the potential displacement of emissions (i.e. a reduction of emissions in one place resulting in an increase of emissions elsewhere (or leakage: 11.3.2; Santilli et al., 2005; Forner et al., 2006; Nabuurs et al., 2007; Strassburg et al., 2008, 2009). The national coverage of REDD+ might ameliorate the issue of emissions displacement, a major drawback of project-based approaches (Herold and Skutsch, 2011). To minimize transnational displacement of emissions, REDD+ needs to stimulate the largest number of developing countries to engage voluntarily. There are also concerns about the impacts of REDD+ design and implementation options on biodiversity conservation, as areas of high C content and high biodiversity are not necessarily coincident. Some aspects of REDD+ implementation that might affect biodiversity include site selection, management

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[13] Decision 2/CP.17 (FCCC/CP/2011/9/Add.1, paragraph 65) "*Agrees* that results-based finance provided to developing country Parties that is new, additional and predictable may come from a wide variety of sources, public and private, bilateral and multilateral, including alternative sources".

strategies and stakeholder engagement (Harvey et al., 2010). From a conservation biology perspective, it is also relevant where the displacement occurs, as deforestation and exploitation of natural resources could move from areas of low conservation value to those of higher conservation value, or to other natural ecosystems, threatening species native to these ecosystems (Harvey et al., 2010). Additionally, transnational displacement could cause deforestation to move into relatively intact areas of high biodiversity value, or into countries which currently have little deforestation (Putz and Redford, 2009).

Box 11.7 Examples of REDD+ initiatives at national scale in different regions with significant extension of forest cover

Amazon Fund - The Amazon Fund in Brazil was officially created in 2008 by a presidential decree. The Brazilian Economic and Social Development Bank (BNDES) was given the responsibility of managing it. The Norwegian government played a key role in creating the fund by donating funds to the initiative in 2009. Since then, the Amazon Fund has received funds from two more donors: the Federal Republic of Germany and Petrobrás, Brazil's largest oil company. As of February 2013, USD 1.03 billion has been pledged, with USD 227 million approved for activities (www.amazonfund.gov.br).

UN-REDD Democratic Republic of Congo - The Congo Basin rainforests are the second largest after Amazonia. In 2009, Democratic Republic of the Congo (DRC), with support of UN-REDD Programme and Forest Carbon Partnership Facility (FCPC), started planning the implementation stages of REDD+ readiness. The initial DRC National Programme transitioned into the full National Programme (Readiness Plan) after it was approved by the UN-REDD Programme Policy Board in 2010 (www.un-redd.org/UNREDDProgramme/). The budget comprises US $5.5 million and timeframe is: 2010 – 2013.

Indonesia-Norway REDD+ Partnership - In 2010, the Indonesia-Norway REDD+ Partnership was established through an agreement between governments of the two countries. The objective was to "support Indonesia's efforts to reduce emissions from deforestation and degradation of forests and peatlands. Indonesia agreed to take systematic and decisive action to reduce its forest and peat related GHG emissions, whereas Norway agreed to support those efforts by making available up to US $1 billion, exclusively on a payment-for-results basis over the next few years" (www.un-redd.org/UNREDDProgramme/). In 2013, Indonesia's government has extended the moratorium on new forest concessions for a further two years, protecting an additional 14.5 Mha of forest.

Taxes, charges, subsidies: Financial regulations are another approach to pollution control. A range of instruments can be used: pollution charges; taxes on emission; taxes on inputs, and subsidies (Jakobsson et al., 2002). Nitrogen (N) taxes are one possible instrument, since agricultural emissions of N_2O mainly derive from the use of nitrogenous fertilizers. An analysis of the tax on the nitrogen content of synthetic fertilizers in Sweden indicated that direct N_2O emissions from agricultural soils in Sweden (the tax abolished in 2010) would have been on average 160 tons or 2% higher without the tax (Mohlin, 2012). Additionally, the study showed that removal of the N tax could completely counteract the decreases in CO_2 emissions expected from the future tax increase on agricultural CO_2. The emission mitigation potential of GHG weighted consumption taxes on animal food products was estimated for the EU using a model of food consumption (Wirsenius et al., 2011). A 7% reduction of current GHG emission in EU agriculture was estimated with a GHG weighted tax on animal food products of 60 €/t CO_2eq. Low-interest loans can also support the transition to sustainable agricultural practices as currently implemented in Brazil, the second largest food exporter, through the national program Low Carbon Agriculture (launched in 2010) (http://www.agricultura.gov.br/desenvolvimento-sustentavel/plano-abc).

11.10.2 Regulatory and Control Approaches

Deforestation control and land planning (protected areas and land sparing / set-aside policies): The rate of deforestation in the tropics and relative contribution to anthropogenic carbon emissions has been declining (Houghton, 2012; see section 11.2 for details). Public policies have had a significant impact by reducing deforestation rates in some tropical countries (see e.g. Box 11.8).

Box 11.8 Deforestation control in Brazil

The Brazilian Action Plan for the Prevention and Control of Deforestation in the Legal Amazon (PPCDAm) includes: coordinated efforts among federal, state, and municipal governments, and civil organizations, remote-sensing monitoring, significant increase of new protected areas (Soares-Filho et al., 2010), and combination of economic and regulatory approaches (for example, since 2008 federal government imposed sanctions to municipalities with very high deforestation rates, subsidies were cut and new credit policies made rural credit dependent on compliance with environmental legislation; Macedo et al., 2012; Nolte et al., 2013).

Since agricultural expansion is one of the drivers of deforestation (especially in tropical regions), one central question is if intensification of agriculture reduces cultivated areas and results in land sparing by concentrating production on other land. Land sparing would allow released lands to sequester carbon, provide other environmental services, and protect biodiversity (Fischer et al., 2008). In the United States, over 13 Mha of former cropland are enrolled in the US Conservation Reserve Program (CRP), with biodiversity, water quality, and carbon sequestration benefits (Gelfand et al., 2011). In 1999, China launched the Grain for Green Program or Sloping Land Conversion Program as a national measure to increase vegetation cover and reduce erosion. Cropland and barren land were targeted and over 20 Mha of land were converted into mostly tree-based plantations. Over its first 10 years between ~800 to 1700 Mt CO_2eq (Moberg, 2011) were sequestered.

Environmental regulation (GHG and their precursors emissions control): In many developed countries, environmental concerns related to water and air pollution since the mid-1990s led to the adoption of laws and regulations that now mandate improved agricultural nutrient management planning (Jakobsson et al., 2002). Some policy initiatives deal indirectly with N leakages and thus promote the reduction of N_2O emissions. The EU Nitrates Directive (1991) sets limits on the use of fertilizer N and animal manure N in nitrate-vulnerable zones. Across the 27 EU Member States, 39.6% of territory is subject to related action programmes. However, in terms of the effectiveness of environmental policies and agriculture, there has been considerable progress in controlling point pollution, but efforts to control non-point pollution of nutrients have been less successful, and potential synergies from various soil-management strategies could be better exploited. Emission targets for the AFOLU sector were also introduced by different countries (e.g. Climate Change Acts in UK and Scotland; European Union).

Bioenergy targets: Many countries worldwide, by 2012, have set targets or mandates or both for bioenergy, to deliver to multiple policy objectives, such as climate change mitigation, energy security, and rural development. The bulk of mandates continue to come from the EU-27 but 13 countries in the Americas, 12 in Asia-Pacific, and 8 in Africa have mandates or targets in place (Petersen, 2008; www.biofuelsdigest.com). For the sustainability of biofuels implementation, land use planning and governance are central (Tilman et al., 2009), as related policy and legislation, e.g. in agriculture, forestry, environment and trade, can strongly influence the development of bioenergy programmes (Jull et al., 2007). A recent study analysed the consequences of renewable targets of EU member states on the CO_2 sink of EU forests, and indicated a decrease in the forest sink by 4–11% (Böttcher et al., 2012). Another possible trade-off of biofuel targets is related to international trade. Global trade in biofuels might have a major impact on other commodity markets (e.g. vegetable oils or animal fodder) and has already caused a number of trade disputes, because of subsidies and non-tariff barriers (Oosterveer and Mol, 2010).

11.10.3 Information Schemes

Acceptability by land managers and practicability of mitigation measures (Table 11.2) need to be considered, because the efficiency of a policy is determined by the cost of achieving a given goal (see sections 11.4.5 and 11.7). Therefore, costs related to education and communication of policies should be taken into account (Jakobsson et al., 2002). Organizations created to foster the use of science in environmental policy, management, and education can facilitate the flow of information from science to society, increasing awareness of environmental problems (Osmond et al., 2010). In the agriculture sector, non-profit conservation organizations (e.g. The Sustainable Agriculture Network - SAN) and governments (e.g. Farming for a Better Climate, Scotland) promote the social and environmental sustainability of activities by developing standards and educational campaigns.

Certification schemes also support sustainable agricultural practices (see section 11.4.5 and 11.7). Climate-friendly criteria reinforce existing certification criteria and provide additional value. Different certification systems also consider improvements in forest management, reduced deforestation and carbon uptake by regrowth, reforestation, agroforestry and sustainable agriculture. In the last 20 years, forest certification has been developed as an instrument for promoting sustainable forest management. Certification schemes encompass all forest types, but there is a concentration in temperate forests (Durst et al., 2006). Approximately 8% of global forest area has been certified under a variety of schemes and 25% of global industrial roundwood comes from certified forests (FAO, 2009b). Less than 2% of forest area in African, Asian and tropical American forests are certified, and most certified forests (82%) are large and managed by the private sector (ITTO, 2008). In the forestry sector, many governments have worked towards a common understanding of sustainable forest management (Auld et al., 2008). Certification bodies certify that farms or groups comply with standards and policies (e.g. Rainforest Alliance Certified). In some, specific voluntary climate change adaptation and mitigation criteria are included.

Forest certification as an instrument to promote sustainable forest management (SFM) and biodiversity maintenance was evaluated by (Rametsteiner and Simula, 2003); they indicated that standards used for issuing certificates upon compliance are diverse, but often include elements that set higher than mimimum standards.

Further, independent audits are an incentive for improving forest management. In spite of many difficulties, forest certification was considered successful in raising awareness, disseminating knowledge on the SFM concept worldwide, and providing a tool for a range of applications other than the assessment of sustainability, e.g. verifying carbon sinks. Another evaluation of certification schemes for conserving biodiversity (Harvey et al., 2008) indicated some constraints that probably also apply to climate-friendly certification: weakness of compliance or enforcement of standards, transaction costs and paperwork often limit participation, and incentives are insufficient to attract high levels of participation. Biofuel certification is a specific case as there are multiple actors and several successive segments of biofuel production pathways: feedstock production, conversion of the feedstock to biofuels, wholesale trade, retail, and use of biofuels in engines (Gnansounou, 2011). Because of the length and the complexity of biofuel supply chains assessing sustainability is challenging (Kaphengst et al., 2009).

Table 11.11: Some regional and global programs and partnerships related to illegal logging, forest management and conservation and REDD+

Program / Institution/Source	Context	Objectives and Strategies
Forest Law Enforcement and Governance (FLEG) / World Bank/ www.worldbank.org/eapfleg	Illegal logging and lack of appropriate forest governance are major obstacle to countries to alleviate poverty, to develop their natural resources and to protect global and local environmental services and values	Support regional forest law enforcement and governance
Improving Forest Law Enforcement and Governance in the European Neighbourhood Policy East Countries and Russia (ENPI-FLEG) / EU/ www.enpi-fleg.org	Regional cooperation in the European Neighbourhood Policy Initiative East Countries (Armenia, Azerbaijan, Belarus, Georgia, Moldova and Ukraine), and Russia following up on the St Petersburg Declaration	Support governments, civil society, and the private sector in participating countries in the development of sound and sustainable forest management practices, including reducing the incidence of illegal forestry activities
Forest Law Enforcement, Governance and Trade (FLEGT) / European Union/ www.euflegt.efi.int/	Illegal Logging has a devastating impact on some of the world's most valuable forests. It can have not only serious environmental, but also economic and social consequences	Exclude illegal timber from markets, to improve the supply of legal timber and to increase the demand for responsible wood products. Central elements are trade accords to ensure legal timber trade and support good forest governance in the partner countries. There is a number of countries in Africa, Asia, South and Central America currently negotiating FLEGT Voluntary Partnership Agreements (VPAs) with the European Union.
Program on Forests (PROFOR) / multiple donors including the European Union, European countries, Japan and the World Bank/ www.profor.info	Well-managed forests have the potential to reduce poverty, spur economic development and contribute to a healthy local and global environment	Provide in-depth analysis and technical assistance on key forest questions related to livelihoods, governance, financing and cross-sectoral issues. PROFOR activities comprise analytical and knowledge generating work that support the strategy's objectives of enhancing forests' contribution to poverty reduction, sustainable development and the protection of environmental services.
UN-REDD Programme / United Nations/ www.un-redd.org	The UN collaborative initiative on Reducing Emissions from Deforestation and forest Degradation (REDD) in developing countries was launched in 2008 and builds on the convening role and technical expertise of the FAO, UNDP and the UNEP.	The Programme supports national REDD+ readiness efforts in 46 partner countries (Africa, Asia-Pacific and Latin America) through: (i) direct support to the design and implementation of REDD+ National Programmes; and (ii) complementary support to national REDD+ action (common approaches, analyses, methodologies, tools, data and best practices).
REDD+ Partnership / International effort (50 different countries)/ www.reddpluspartnership.org	The UNFCCC has encouraged the Parties to coordinate their efforts to reduce emissions from deforestation and forest degradation. As a response, countries attending the March 2010 International Conference on the Major Forest Basins, hosted by the Government of France, agreed on the need to forge a strong international partnership on REDD+.	The REDD+ Partnership serves as an interim platform for its partner countries to scale up actions and finance for REDD+ initiatives in developing countries (including improving the effectiveness, efficiency, transparency and coordination of REDD+ and financial instruments), to facilitate knowledge transfer, capacity enhancement, mitigation actions and technology development and transfer among others.
Forest Investment Program (FIP) / Strategic Climate Fund (a multi-donor Trust Fund within the Climate Investment Funds) www.climateinvestmentfunds.org/cif/	Reduction of deforestation and forest degradation and promotion of sustainable forest management, leading to emission reductions and the protection of carbon terrestrial sinks.	Support developing countries' efforts to REDD and promote sustainable forest management by providing scaled-up financing to developing countries for readiness reforms and public and private investments, identified through national REDD readiness or equivalent strategies.
Forest Carbon Partnership (FCPF) / World Bank/ www.forestcarbonpartnership.org	Assistance to developing countries to implement REDD+ by providing value to standing forests.	Builds the capacity of developing countries to reduce emissions from deforestation and forest degradation and to tap into any future system of REDD+.
Indonesia-Australia Forest Carbon Partnership/ www.iafcp.or.id	Australia's assistance on climate change and builds on long-term practical cooperation between Indonesia and Australia.	The Partnership supports strategic policy dialogue on climate change, the development of Indonesia's National Carbon Accounting System, and implementing demonstration activities in Central Kalimantan.

11.10.4 Voluntary Actions and Agreements

Innovative agricultural practices and technologies can play a central role in climate change mitigation and adaptation, with policy and institutional changes needed to encourage the innovation and diffusion of these practices and technologies to developing countries. Under the UNFCCC, the 2007 Bali Action Plan identified technology development and transfer as a priority area. A Technology Mechanism was established by Parties at the COP16 in 2010 "to facilitate the implementation of enhanced action on technology development and transfer, to support action on mitigation and adaptation, in order to achieve the full implementation of the Convention" (http://unfccc.int). For agriculture, Burney et al. (2010) indicated that investment in yield improvements compared favourably with other commonly proposed mitigation strategies.

Additionally, adaptation measures in agriculture can also generate significant mitigation effects. (Lobell et al., 2013) investigated the co-benefits of adaptation measures on farm level that reduced GHG, emissions from land-use change. The study focused on investments in research for developing and deploying new technologies (e.g. disease resistant or drought tolerant crops, or soil management techniques). It concluded that broad-based efforts to adapt agriculture to climate change have mitigation co-benefits that are associated with lower costs than many activities focussing on mitigation, especially in developed countries.

11.11 Gaps in knowledge and data

Data and knowledge gaps include:

- Improved global high resolution data sets of crop production systems (including crop rotations, variety selection, fertilization practices and tillage practices), grazing areas (including quality, intensity of use, management), and freshwater fisheries and aquaculture, also comprising subsistence farming.
- Globally standardized and homogenized data on soil as well as forest degradation and a better understanding of the effects of degradation on carbon balances and productivity.
- Improved understanding of the mitigation potential, interplay, costs as well as environmental and socio-economic consequences of land use based mitigation options such as improved agricultural management, forest conservation, bioenergy production and afforestation on the national, regional and global scale.
- Better understanding of the effect of changes in climate parameters, rising CO_2 concentrations and N deposition on productivity and carbon stocks of different types of ecosystems, and the related consequences for land based climate change mitigation potentials.

11.12 Frequently Asked Questions

FAQ 11.1 How much does AFOLU contribute to GHG emissions and how is this changing?

Agriculture and land use change, mainly deforestation of tropical forests, contribute greatly to anthropogenic greenhouse gas emissions and are expected to remain important during the 21st century. Annual GHG emissions (mainly CH_4 and N_2O) from agricultural production in 2000-2010 were estimated at 5.0-5.8 Gt CO_2eq/yr, comprising about 10-12% of global anthropogenic emissions. Annual GHG flux from land use and land use change activities accounted for approximately 4.3-5.5 Gt CO_2eq/yr, or about 9-11% of total anthropogenic greenhouse gas emissions. The total contribution of the AFOLU sector to anthropogenic emissions is therefore around one quarter of the global anthropogenic total.

FAQ 11.2 How will mitigation actions in AFOLU affect GHG emissions over different timescales?

There are many mitigation options in the AFOLU sector which are already being implemented e.g. afforestation, reducing deforestation, cropland and grazing land management, fire management and improved livestock breeds and diets. These can be implemented now. Others (such as some forms of biotechnology and livestock dietary additives) are still in development and may not be applicable for a number of years. In terms of the mode of action of the options, in common with other sectors, non-CO_2 greenhouse gas emission reduction is immediate and permanent. However, a large portion of the mitigation potential in the AFOLU sector is carbon sequestration in soils and vegetation. This mitigation potential differs, in that the options are time-limited (the potential saturates), and the enhanced carbon stocks created are reversible and non-permanent. There is, therefore, a significant time component in the realisation and the duration of much of the mitigation potential available in the AFOLU sector.

FAQ 11.3 What is the potential of the main mitigation options in AFOLU for reducing GHG emissions?

In general, available top-down estimates of costs and potentials suggest that AFOLU mitigation will be an important part of a global cost-effective abatement strategy. However, potentials and costs of these mitigation options differ greatly by activity, regions, system boundaries and the time horizon. Especially, forestry mitigation options - including reduced deforestation, forest management, afforestation, and agro-forestry - are estimated to contribute 0.2-13.8 Gt CO_2/yr of economically viable abatement in 2030 at carbon prices up to 100 USD/t CO_2eq. Global economic mitigation potentials in agriculture in 2030 are estimated to be up to 0.49-10.6 Gt CO_2eq/yr. Besides supply side based mitigation, demand side mitigation options can have a significant impact on GHG emissions from food production. Changes in diet towards plant-based, and hence less GHG intensive, food can result in GHG emission savings of 0.7-7.3 Gt CO_2eq/yr in 2050, depending on which GHGs and diets considered. Reducing food losses and waste in the supply chain from harvest to consumption can reduce GHG emissions by 0.6-6.0 Gt CO_2eq/yr.

FAQ 11.4 Are there any co-benefits associated with mitigation actions in AFOLU?

In several cases, the implementation of AFOLU mitigation measures may result in an improvement in land management and therefore have socio-economic, health and environmental benefits: For example, reducing deforestation, reforestation and afforestation can improve local climatic conditions, water quality, biodiversity conservation and help to restore degraded or abandoned land. Soil management to increase soil carbon sequestration may also reduce the amount of wind and water erosion due to an increase in surface cover. Further considerations on economic co-benefits are related to the access to carbon payments either within or outside the UNFCCC agreements and new income opportunities especially in developing countries (especially for labour intensive mitigation options such as afforestation).

FAQ 11.5 What are the barriers to reducing emissions in AFOLU and how can these be overcome?

There are many barriers to emission reduction. Firstly, mitigation practices may not be implemented for economic reasons (e.g. market failures, need for capital investment to realise recurrent savings), or a range of factors including risk-related, political/bureaucratic, logistical and educational/societal barriers. Technological barriers can be overcome by research and development; logistical and political / bureaucratic barriers can be overcome by better governance and institutions; education barriers can be overcome through better education and extension networks; and risk-related barriers can be overcome, for example, through clarification of land tenure uncertainties.

11.13 Appendix Bioenergy: Climate effects, mitigation options, potential and sustainability implications

11.13.1 Introduction

The recent IPCC report on renewables (SRREN) provided a comprehensive overview on bioenergy (Chum et al. 2011). However, a specific bioenergy Appendix in the context of the AR5 report is necessary because: a) many of the more stringent mitigation scenarios (resulting in 450 ppm, but also 550 ppm CO_2eq concentration by 2100, see Section 11.9.1) heavily rely on a large scale deployment of bioenergy with CO_2 capture and storage (CCS) called BECCS technologies; b) there has been a large body of literature published since SRREN, which complement and update the analysis presented in this last report; c) bioenergy is important for many chapters (Ch. 6, 7, 8, 10, 11), and also policy chapters,which makes it more useful to treat it in a single section instead of in many scattered chapter sections throughout the report. Chapter 11 is the appropriate location for the Appendix, as bioenergy analysis relies crucially on land-use assessments.

Bioenergy is energy derived from biomass, which can be deployed as solid, liquid and gaseous fuels for a wide range of uses, including transport, heating, electricity production, and cooking (Chum et al. 2011). Bioenergy has a significant GHG mitigation potential, but there are issues to consider, such as the sustainability of practices and the efficiency of bioenergy systems (Chum et al. 2011). Bioenergy systems can cause both positive and negative effects and their deployment needs to balance a range of environmental, social and economic objectives that are not always fully compatible. The consequences of bioenergy implementation depend on a) the technology used; b) the location, scales and pace of implementation; and c) the land category used (forest, grassland, marginal lands and crop lands) and d) the business models and practices adopted - including how these integrate with or displace the existing land use.

11.13.2 Technical primary biomass potential for bioenergy

The technical primary biomass potential for bioenergy –from here on referred to as “technical bioenergy potential”- is the fraction of the theoretical potential (i.e., the theoretical maximum amount of biomass constrained only by biophysical limits) available with current technology. Unfortunately there is no standard methodology to estimate the technical bioenergy potential, which leads to diverging estimates. Also, most of the recent studies estimating technical bioenergy potentials assume a ‘food/fibre first principle’ and exclude deforestation, eventually resulting in an estimate of the ‘environmentally sustainable bioenergy potential’ when a comprehensive range of environmental constraints is considered (Batidzirai et al., 2012).

Recently published estimates that are based in this extended definition of global technical primary biomass potentials in 2050 span a range of almost three orders of magnitude, from <50 EJ/yr to >1,000 EJ/yr (Smeets et al., 2007b; Field et al., 2008; Haberl et al., 2010; Batidzirai et al., 2012). For example, the SRREN reported global technical bioenergy potentials of 50-500 EJ/yr for the year 2050 (Chum et al., 2011) and the Global Energy Assessment gave a range of 160-270 EJ/yr (Johansson et al., 2012). The discussion following the publication of these global reports has not resulted in a consensus on the magnitude of the future global technical bioenergy potential, but has helped to better understand some of its many structural determinants (Wirsenius et al., 2010; Erb et al., 2012a; Berndes et al., 2012). How much biomass for energy is technically available in the future depends on the evolution of a multitude of social, political and economic factors, e.g., land tenure and regulation, trade and technology (Dornburg et al., 2010).

Figure 11.20 shows estimates of the global technical bioenergy potential in 2050 by resource categories. Ranges were obtained from assessing a large number of studies based on a food/fibre first principle and various restrictions regarding resource limitations and environmental concerns but no explicit cost considerations (Hoogwijk et al., 2005, 2009; Smeets and Faaij, 2007; Smeets et al., 2007b; van Vuuren et al., 2009b; Hakala et al., 2009; Dornburg et al., 2010; Gregg and Smith, 2010; Haberl et al., 2010, 2011b; Chum et al., 2011; GEA, 2012; Rogner et al., 2012b). Most studies agree that the technical bioenergy in 2050 is at least approximately 100 EJ/yr with some modelling assumptions leading to estimates exceeding 500 EJ/yr (Smeets et al., 2007b). As stated, different views about sustainability and socio-ecological constraints lead to very different estimates, with some studies reporting much lower figures.

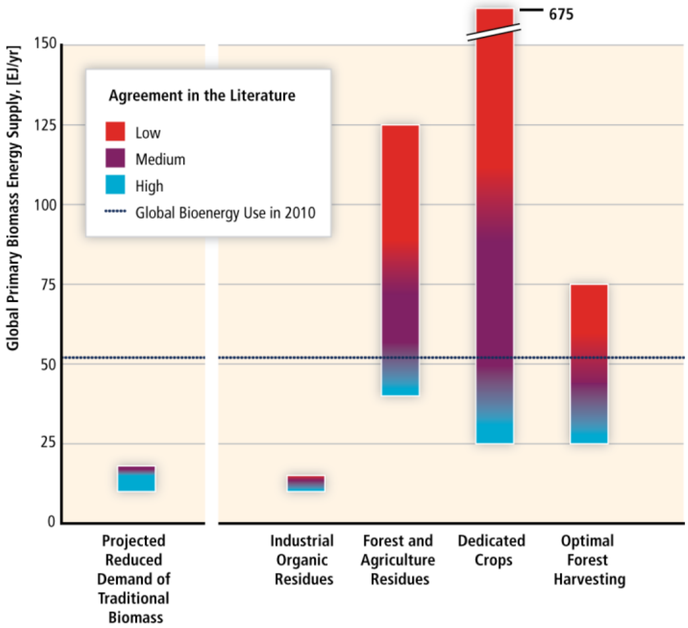

Figure 11.20. Global Technical Primary Biomass Potential for Bioenergy by Main Resource Category for the year 2050. The Figure shows the ranges in the estimates by major resource category of the global technical primary biomass potential for bioenergy. The color grading is intended to show qualitatively the degree of agreement in the estimates, from blue (large agreement in the literature) to purple (medium agreeement) to red (small agreement). In addition, reducing traditional biomass demand by increasing its use efficiency could release the saved biomass for other energy purposes with large benefits from a sustainable development perspective.

As shown in Figure 11.20, the total technical bioenergy potential is composed of several resource categories that differ in terms of their absolute potential, the span of the ranges –which also reflect the relative agreement/disagreement in the literature- and the implications of utilizing them. Regional differences –which are not addressed here- are also important as the relative size of each biomass resource within the total potential and its absolute magnitude vary widely across countries and world regions.

Forest and Agriculture residues. Forest residues (Smeets and Faaij, 2007; Smeets et al., 2007b; Dornburg et al., 2010; Gregg and Smith, 2010; Haberl et al., 2010; Rogner et al., 2012b): include residues from silvicultural thinning and logging; wood processing residues such as sawdust, bark and black liquor; dead wood from natural disturbances, such as storms and insect outbreaks (irregular source). The use of these resources is in general beneficial. Adverse side-effects can be mitigated by controlling residue removal rates considering biodiversity, climate, topography, and soil factors. There is a near term trade-off, particularly within temperate and boreal regions, in that organic matter retains organic C for longer if residues are left to decompose slowly instead of being used for energy. Agricultural residues (Smeets et al., 2007b; Hakala et al., 2009; Gregg and Smith, 2010; Haberl et al., 2010, 2011b; Chum et al., 2011; Rogner et al., 2012b) include manure, harvest residues (e.g., straw) and processing residues (e.g., rice husks from rice milling) and are also in general beneficial. However, mitigating potential adverse side-effects –such as the loss of soil C- associated to harvesting agriculture residues is more complex as they depend on the different crops, climate and soil conditions (Kochsiek and Knops, 2012; Repo et al., 2012). Alternative uses of residues (bedding, use as fertilizer) need to be considered. Residues have varying collection and processing costs (in both agriculture and forestry) depending on residue quality and dispersal, with secondary residues often having the benefits of not being dispersed and having relatively constant quality. Densification and storage technologies would enable cost effective collections over larger areas. Optimisation of crop rotation for food and bioenergy output and the use of residues in biogas plants may result in higher bioenergy yields from residues without food-energy competition.

Optimal forest harvesting is defined as the fraction of sustainable harvest levels (often set equal to net annual increment) in forests available for wood extraction, which is additional to the projected biomass demand for producing other forest products. This includes both biomass suitable for other uses (e.g., pulp and paper production) and biomass that is not used commercially (Smeets and Faaij, 2007; Chum et al., 2011). The resource potential depends on both environmental and socio-economic factors. For example, the change in forest management and harvesting regimes due to bioenergy demand depends on forest ownership and the structure of the associated forest industry. Also, the forest productivity -and C stock- response to changes in forest management and harvesting depend on the character of the forest ecosystem, as shaped by historic forest management and events such as fires, storms and insect outbreaks, but also on the management scheme (e.g. including replanting after harvest, soil protection, recycling of nutrients and soil types; (Jonker et al., 2013; Lamers et al., 2013). In particular, optimizing forest management for mitigation is a complex issue with many uncertainties and still subject to scientific debate. Intensive forest management activities of the early- to mid-20th century as well as other factors such as recovery from past overuse, have led to strong forest C-sinks in many OECD regions (Erb et al., 2013, Loudermilk et al., 2013, Nabuurs et al., 2013, Pan et al., 2011). However, the capacity of these sinks is being reduced as forests approach saturation (Gulde et al., 2008, Körner, 2006, Smith, 2005, Smith & Bustamante, 2013; Nabuurs et al., 2013). Active forest management, including management for bioenergy, is therefore important for sustaining the strength of the forest carbon sink well into the future (Canadell & Raupach, 2008, Ciais et al., 2008, Nabuurs et al., 2013, Nabuurs et al., 2007), although countries should realize that for some old forest areas, conserving carbon stocks may be preferential, and that the actively managed forests may for some time (decades) act as sources.

Organic wastes include waste from households and restaurants, discarded wood products such as paper, construction, and demolition wood waste, and waste waters suitable for anaerobic biogas production (Gregg and Smith, 2010; Haberl et al., 2010). Organic waste may be dispersed and also heterogeneous in quality but the health and environmental gains from collection and proper management through combustion or anaerobic digestion can be significant. Competition with alternative uses of the wastes may limit this resource potential.

Dedicated biomass plantations include annual (cereals, oil- and sugar crops) and perennial plants (e.g., switchgrass, Miscanthus) and tree plantations (both coppice and single-stem plantations (e.g., willow, poplar, eucalyptus, pine; Hoogwijk et al., 2005, 2009; Smeets et al., 2007b; van Vuuren et al., 2009b; Dornburg et al., 2010; Wicke et al., 2011c; Haberl et al., 2011a). The range of estimates of technical bioenergy potentials from that resource in 2050 is particularly large (<50 to >500 EJ/yr). Technical bioenergy potentials from dedicated biomass plantations are generally calculated by multiplying (i) the area deemed available for energy crops by (ii) the yield per unit area and year (Batidzirai et al., 2012; Coelho et al., 2012). Some studies have identified a sizable technical potential (up to 100 EJ) for bioenergy production using marginal and degraded lands (e.g. saline land) that are currently not in use for food production or grazing (Nijsen et al., 2012b). However, how much land is really unused and available is contested (Erb et al., 2007; Haberl et al., 2010; Coelho et al., 2012). Contrasting views on future technical bioenergy potentials from dedicated biomass plantations can be explained by differences in assumptions regarding feasible future agricultural crop yields, livestock feeding efficiency, land availability for energy crops and yields of energy crops (Dornburg et al., 2010; Batidzirai et al., 2012; Erb et al., 2012a). Most scientists agree that increases in food crop yields and higher feeding efficiencies and lower consumption of animal-products results in higher technical bioenergy potential. Also, there is a large agreement that careful policies for implementation focused on land-use zoning approaches (including nature conservation and biodiversity protection), multifunctional land use, integration of food and energy production, avoidance of detrimental livelihood impacts e.g. on livestock grazing and subsistence farming, and consideration of equity issues and sound management of impacts on water systems are crucial for sustainable solutions.

Reduced Traditional Biomass Demand. A substantial quantity of biomass will become available for modern applications by improving the end-use efficiency of traditional biomass consumption for energy, mostly in households but also within small industries (such as charcoal kilns, brick kilns, etc.). Traditional bioenergy represents approximately 15% of total global energy use and 80% of current bioenergy use (≈35 EJ/yr) and helps meeting the cooking and heating needs of ~2.7 billion people (Chum et al, 2011, WHO, 2012). Traditional bioenergy use covers several end-uses including cooking, water and space heating, and small-industries (such as brick and pottery kilns, bakeries, and many others). Cooking is the dominant end-use; it is mostly done in open fires and rudimentary stoves, with approximately 10%-20% conversion efficiency, leading to very high primary energy consumption. Advanced woodburning and biogas stoves can potentially reduce biomass fuel consumption by 60% or more (Jetter et al 2012) and further lower the atmospheric radiative forcing, reducing CO_2 emissions, and in many cases black carbon emissions, by up to 90% (Anenberg et al., 2013). Assuming that actual savings reach on average from 30%-60% of current consumption, the total bioenergy potential from reducing traditional bioenergy demand can be estimated at 8-18 EJ/yr. An unknown fraction of global traditional biomass is consumed in a non-environmentally sustainable way, leading to forest degradation and deforestation. Detailed country studies have estimated the fraction of non-renewable biomass from traditional bioenergy use to vary widely –e.g. from 1.6% for the Democratic Republic of Congo to 73% for Burundi (UNFCCC-CDM, 2012)- with most countries in the range between 10-30% (i.e. meaning that 70%-90% of total traditional bioenergy use is managed sustainably). Thus a fraction of the traditional biomass saved through better technology, should not be actually used for other energy purposes but simply not consumed to help restore the local ecosystems.

11.13.3 Bioenergy conversion: technologies, and management practices

Numerous conversion technologies can transform biomass to heat, power, liquid and gaseous fuels for use in the residential, industrial, transport and power sectors (see H. Chum et al., 2011 and GEA, 2012, for a comprehensive coverage of each alternative, and Figure 11.21 for the pathways concerning liquid and gaseous fuels). Since SRREN, the major advances in the large-scale production of bioenergy include the increasing use of hybrid biomass-fossil fuel systems. For example, the use of current commercial coal and biomass co-combustion technologies are the lowest cost technology to implement renewable energy policies, enabled by the large-scale pelletized feedstocks trade (Junginger et al., 2014; REN21, 2013). Direct biopower use is also increasing commercially globally (REN21, 2013, p. 21). In fact, using biomass for electricity and heat, e.g., co-firing of woody biomass with coal in the near term and large heating systems coupled with networks for district heating, and biochemical processing of waste biomass, are among the most cost-efficient and effective biomass applications for GHG emission reduction in modern pathways (Sterner and Fritsche, 2011).

Integrated gasification combined cycle (IGCC) technologies for coproduction of electricity and liquid fuels from coal and biomass with higher efficiency than current commercial processes are in demonstration phase to reduce cost (GEA, 2012; Larson, E. D. et al., 2012; Williams et al., 2011). Coupling of biomass and natural gas for fuels is another option for liquid fuels (Baliban et al., 2013) as the biomass gasification technology development progresses. Simulations suggest that integrated gasification facilities are technically feasible (with up to 50% biomass input; Meerman et al., 2011) and economically attractive with a CO_2 price of about 50€/tCO_2 (Meerman et al., 2012). Many gasification technology developments around the world are in pilot, demonstration, operating first commercial scale for a variety of applications (see examples in Bacovsky, Nikolaus, Monica, & Manfred, 2013; Balan, Chiaramonti, & Kumar, 2013).

Many pathways and feedstocks (Fig. 11.21) can lead to biofuels for aviation; the development of biofuel standards started and enabled commercial domestic and transatlantic flights testing of 50% biofuel in jet fuel by consortia of governments, aviation industry, associations (REN21, 2012; IEA, 2010a); (REN21, 2013, p. 21). Advanced 'drop in' fuels, such as iso-butanol, synthetic aviation kerosene from biomass gasification or upgrading of pyrolysis liquids, can be derived through a number of possible conversion routes such as hydro treatment of vegetable oils, iso-butanol, and Fischer-Tropsch synthesis from gasification of biomass (Hamelinck and Faaij, 2006; Bacovsky et al., 2010; Meerman et al., 2011, 2012; Rosillo-Calle et al., 2012; see also Ch. 8). In specific cases, powering electric cars with electricity from biomass has higher land-use efficiency and lower GWP effects than the usage of bioethanol from biofuel crops for road transport across a range of feedstocks, conversion technologies, and vehicle classes (Campbell et al., 2009; Schmidt et al., 2011)[14], though costs are likely to remain prohibitive for considerable time (van Vliet et al., 2011; Van Vliet et al., 2011; Schmidt et al., 2011).

The number of routes from biomass to a broad range of biofuels, shown in Figure 11.21, includes hydrocarbons connecting today's fossil fuels industry in familiar thermal/catalytic routes such as gasification (Larson, E. D. et al., 2012; Williams et al., 2011) and pyrolysis (Bridgwater, 2012; Elliott, 2013; Meier et al., 2013; Brown, 2011). In addition, advances in genomic technology, the emphasis in systems approach, and the integration between engineering, physics, chemistry, and biology bring together many new approaches to biomass conversion (Liao and Messing, 2012) such as: (a) biomolecular engineering (Li et al., 2010; Peralta-Yahya et al., 2012; Favaro et al., 2013; Lee et al., 2013; Yoon et al., 2013) deconstruction of lignocellulosic biomass through combinations of mild thermal and biochemical routes in multiple sequential or consolidated steps using similar biomolecular engineering tools (Rubin, 2008; Chundawat et al., 2011; Beckham et al., 2012; Olson et al., 2012; Tracy et al., 2012; Kataeva et al., 2013; Saddler and Kumar, 2013); (b) advances in (bio) catalysis and basic understanding of the synthesis of cellulose are leading to routes for many fuels and chemicals under mild conditions (Serrano-Ruiz et al., 2010; Carpita, 2012; Triantafyllidis et al., 2013; Yoon et al., 2013; Shen et al., 2013). Fundamental understanding of biofuels production increased for microbial genomes by forward engineering of cyanobacteria, microalgae, aiming to arrive at minimum genomes for synthesis of biofuels or chemicals (Chen and Blankenship, 2011; Eckert et al., 2012; Ungerer et al., 2012; Jones and Mayfield, 2012; Kontur et al., 2012; Lee et al., 2013).

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[14] Biomass can be used for electric transport and biofuels within one pathway (Macedo et al., 2008).

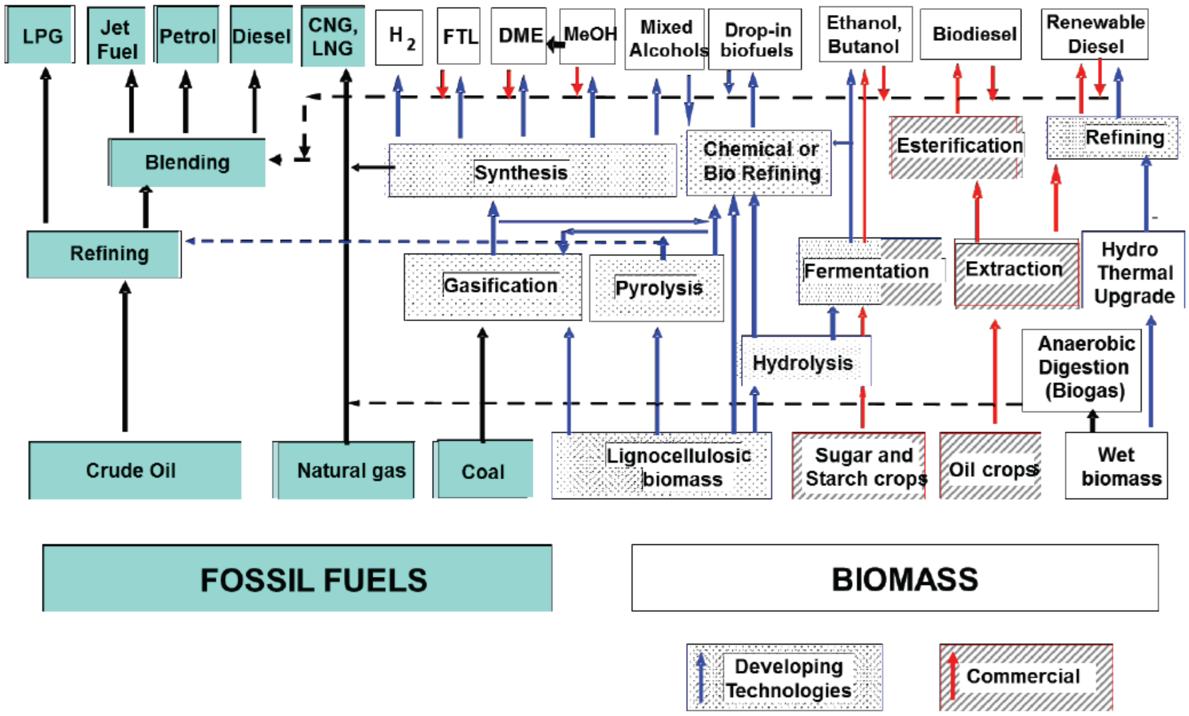

Figure 11.21. Production pathways to liquid and gaseous fuels from biomass and, for comparison from fossil fuels (adapted from Turkenburg et al., 2012, GEA, 2012, Chapter 11).

Bioenergy coupled with CO_2 Capture and Storage (BECCS; Spath and Mann, 2004; Liu et al., 2010, 2011) Is seen as an option to mitigate climate change through negative emissions if CCS can be successfully deployed (Cao and Caldeira 2010; Lenton and Vaughan 2009). BECCS features prominently in long-run mitigation scenarios (6.3.2 and 6.3.5) for two reasons: 1) The potential for negative emissions may allow shifting emissions in time; 2) In scenarios, negative emissions from BECCS compensate for residual emissions in other sectors (most importantly transport) in the second half of the 21st century. As illustrated in Figure 11.22, BECCS is markedly different than fossil CCS because it not only reduces CO_2 emissions by storing C in long term geological sinks, but it continually sequesters CO_2 from the air through regeneration of the biomass resource feedstock.

BECCS deployment is in the development and exploration stages. The most relevant BECCS project is the Illinois Basin – Decatur Project (IBDP) that is projected to inject 1MtCO_2/yr (Gollakota and McDonald, 2012; Senel and Chugunov, 2013). In the US, two ethanol fuel production by fermentation facilities are currently integrated commercially with carbon dioxide capture, pipeline transport, and use in enhanced oil recovery in nearby facilities at a rate of about 0.2 Mt CO_2/year (DiPietro et al., 2012). Altogether there are 16 global BECCS projects in exploration stage (Karlsson and Byström, 2011).

Critical to overall CO_2 storage is the realization of a lignocellulosic biomass supply infrastructure for large-scale commodity feedstock production and efficient advanced conversion technologies at scale; both benefit from cost reductions and technological learning as does the integrated system with CCS, with financial and institutional conditions that minimize the risks of investment and facilitate dissemination (Eranki and Dale, 2011; IEA, 2012, 2013). Integrated analysis is needed to capture system and knock-on effects for bioenergy potentials. A nascent feedstock infrastructure for densified biomass trading globally could indicate decreased pressure on the need for closely co-located storage and production (IEA, 2011; Junginger et al., 2014).

The overall technical potential is estimated to be around 10Gt CO_2 storage per year for both IGCC-CCS co-firing (Integrated Gasification Combined Cycle with co-gasification of biomass), and BIGCC-CCS dedicated (Biomass Integrated Gasification Combined Cycle), and around 6Gt CO_2 storage for FT diesel (Biodiesel based on gasification and Fischer–Tropsch synthesis), and 2.7 Gt CO_2 for biomethane production (Koornneef et al., 2012, 2013). Another study estimates the potential capacity (similar to technical potential) to be between 2.4 and 10 Gt CO_2 per year for 2030-2050 (McLaren, 2012). The economic potential, at a CO_2 price of around 70$/t is estimated to be around 3.3 Gt CO_2, 3.5 Gt CO_2, 3.1 Gt CO_2 and 0.8 Gt CO_2 in the corresponding four cases, judged to be those with highest economic potential (Koornneef et al., 2012, 2013). Potentials are assessed on a route-by-route basis and cannot simply be added, as they may compete and substitute each other. Practical figures might be not much higher than 2.4 Gt CO_2 per year at $70-250/t$CO_2$ (McLaren, 2012). Altogether, until 2050 the economic potential is anywhere between 2-10 GtCO_2 per year. Some climate stabilization scenarios see considerable higher deployment towards the end of the century, even in some 580-650ppm scenarios, operating under different time scales, socioeconomic assumptions, technology portfolios, CO_2 prices, and interpreting BECCS as part of an overall mitigation framework (e.g. Rose et al., 2012; Kriegler et al., 2013; Tavoni and Socolow, 2013).

Possible climate risks of BECCS relate to reduction of land carbon stock, feasible scales of biomass production and increased N_2O emissions, and potential leakage of CO_2, which has been stored in deep geologic reservoirs (Rhodes and Keith, 2008). The assumptions of sufficient spatially appropriate CCS capture, pipeline and storage infrastructure are uncertain. The literature highlights that BECCS as well as CCS deployment is dependent on strong financial incentives, as they are not cost competitive otherwise (see Sections 7.5.5, 7.6.4., 7.9 and 7.12).

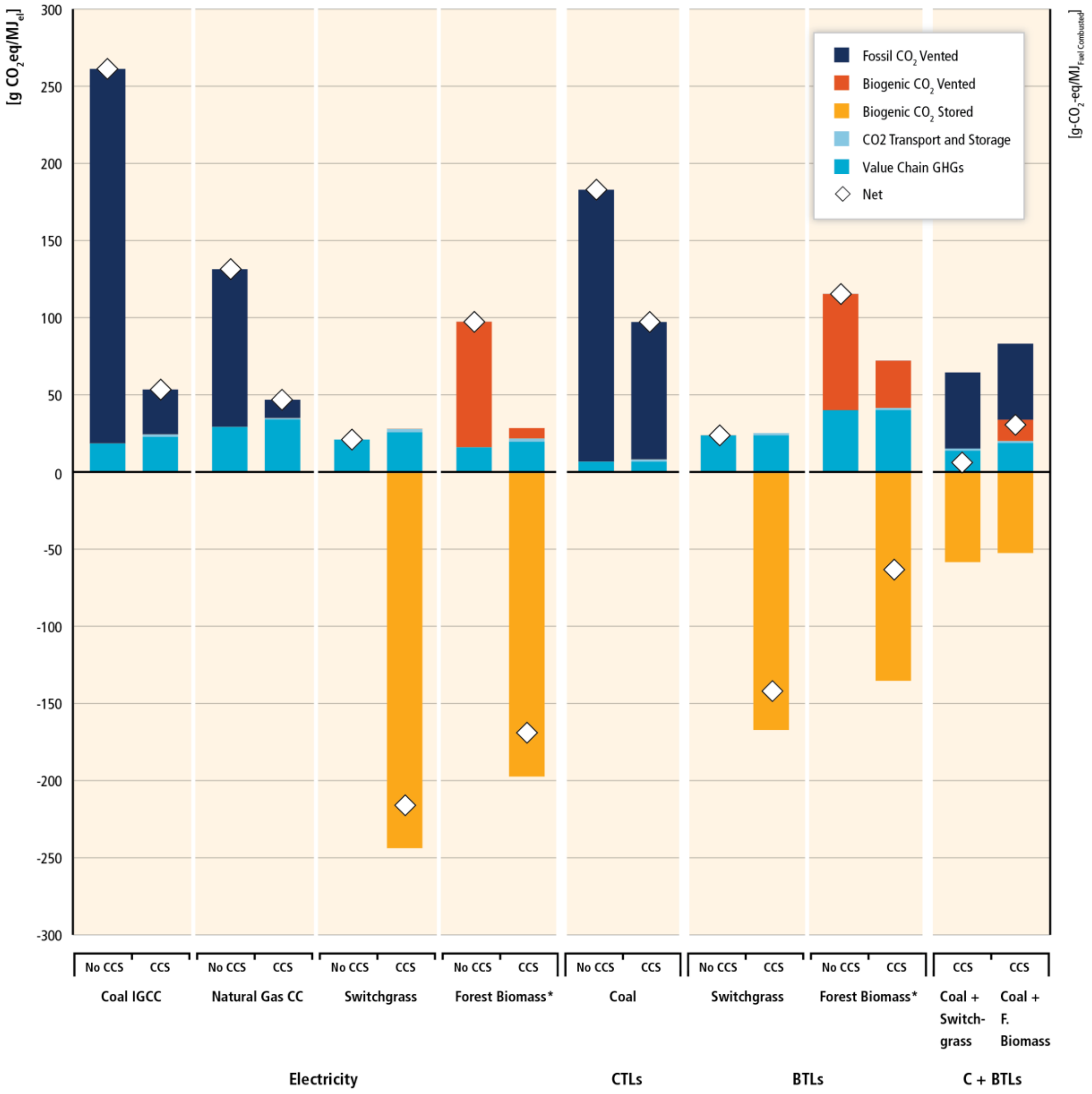

Figure 11.22. Illustration of the sum of CO_2-equivalent (GWP100)[15] emissions from the process chain of alternative transport and power generation technologies both with and without CCS. (*Differences in C-density between forest biomass and switchgrass are taken into account but not calorific values (balance-of-plant data are for switchgrass, ref. Larson et al., 2012) Specific emissions vary with biomass feedstock and conversion technology combinations, as well as lifecycle GHG calculation boundaries. For policy relevant purposes, counterfactual and market-mediated aspects (e.g., ILUC), changes in soil organic carbon, or changes in surface albedo need also to be considered, possibly leading to quantitatively significantly different outcomes (Section 11A4, Figures 11A4 and 11A5). Unit: g CO_2eq. MJ_{El} (left y-axis, electricity); g-CO_2-eq./MJ combusted (right y-axis, transport fuels). Direct CO_2 emissions from energy conversion (“vented” and “stored”) are adapted from the mean values in Tables 12.7, 12.8, and 12.15 of ref. [1], which are based on the work of refs. [2, 3], and characterized with the emission metrics in ref. [4]. Impacts upstream in the supply chain associated with feedstock procurement (i.e., sum of GHGs from mining/cultivation, transport, etc.) are adapted from refs. [5, 6] and Figure 11.23 (median values).

[1]Larson, et al. (2012); [2]Woods, et al., (2007); [3]Liu et al. (2010); [4]Guest et al. (2013); [5]Turconi et al. (2013); [6]Jaramillo et al. (2008)

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[15] Global Warming Potential over 100 years. See Glossary and Section 1.2.5.

Figure 11.22 illustrates some GHG effects associated with BECCS pathways. Trade-offs between CO_2 capture rate and feedstock conversion efficiency are possible. Depicted are pathways with the highest removal rate but not necessarily with the highest feedstock conversion rate. Among all BECCS pathways, those based on integrated gasification combined cycle produce most significant geologic storage potential from biomass, alone (shown in Fig 11.23, electricity) or coupled with coal. Fischer-Tropsch diesel fuel production with biomass as feedstock and CCS attached to plant facilities could enable BECCS for transport; uncertainties in input factors and output metrics warrant further research (Van Vliet et al., 2009); Fischer-Tropsch diesel would also allow net removal but at lower rates than BIGCC.

Economics of scale in power plant size are crucial to improve economic viability of envisaged BECCS projects. Increasing power plant size requires higher logistic challenges in delivering biomass.

Scales of 4,000 to 10,000 Mg/day needed for >600 MW power plants could become feasible as the biomass feedstock supply logistic development with manageable logistic costs if biomass is derived from high-yield monocrops; logistical costs are more challenging when biomass is derived from residues (e.g., Argo et al., 2013; Junginger et al., 2014). Large-scale biomass production with flexible integrated polygeneration facilities for fuels and/or power can improve the techno-economic performance, currently above market prices to become more economically competitive over time (Meerman et al., 2011). In the future, increased operating experience of BECCS IGCC-CCS through technological improvements and learning could enable carbon neutral electricity and, in combination with CCS, could result in net removal of CO_2 (Figure 11.22). BECCS is among the lowest cost CCS options for a number of key industrial sectors (Meerman et al., 2013). It should be noted that primary empiric cost and performance data for dedicated bioenergy plants are not yet available and needed for comprehensively assessing BECCS. The current status of CCS and on-going research issues are discussed in Sections 7.5.5. and 7.6.4. Social concerns constitute a major barrier for implement demonstration and deployment projects.

Integrated bio-refineries continue to be developed; for instance, 10% of the ethanol or corresponding sugar stream goes into bio-products in Brazil (REN21, 2012) including making ethylene for polymers (IEA-ETSAP and IRENA, 2013). Multi product bio-refineries could produce a wider variety of co-products to enhance the economics of the overall process, facilitating learning in the new industry (IEA, 2011); LCAs for these systems are complex (Pawelzik et al., 2013).

Microalgae offers an alternative to land-based bioenergy. Its high-end technical potential might be compromised by water supply, if produced in arid land, or by its impact on ocean ecosystems. To make algae cost competitive, maximizing algal lipid content (and then maximizing growth rate) require essential technological breakthroughs (Davis et al., 2011; Sun et al., 2011; Jonker and Faaij, 2013). Its market potential depends on the co-use of products for food, fodder, higher value products, and fuel markets (Chum et al., 2011). Similarly, lignocellulosic feedstocks produced from waste or residues, or grown on land unsupportive of food production (e.g., contaminated land for remediation as in previously mined land) have been suggested to reduce socio-environmental impact. Reforestation schemes have potential to restore soil quality and increase soil carbon stocks over time (Wicke et al., 2013). In addition, lignocellulosic feedstocks can be bred specifically for energy purposes, and can be harvested by coupling collection and pre-processing (densification and others) in depots prior to final conversion, which could enable delivery of more uniform feedstocks throughout the year (Eranki and Dale, 2011; U.S. DOE, 2011; Argo et al., 2013). Various conversion pathways are in R&D, near commercialization, or in early deployment stages in several countries (see 2.6.3 in H. Chum et al., 2011). More productive land is also more economically attractive for cellulosic feedstocks, in which case competition with food production is more likely. Depending on the feedstock, conversion process, prior land use, and land demand, lignocellulosic bioenergy can be associated with high or low GHG emissions (e.g. Davis et al., 2012). Improving agricultural lands and reducing non-point pollution emissions to watersheds remediate nitrogen run off and increase overall ecosystems' health (Van Dam et al., 2009a; b; Gopalakrishnan et al., 2012). Also regeneration of saline lands by salt tolerant tree and grass species can have a large potential on global scale as demonstrated by (Wicke et al., 2011c).

A range of agro-ecological options to improve agricultural practices such as no/low tillage conservation, agroforestry, etc., have potential to increase yields (e.g. in sub-Saharan Africa), while also providing a range of co-benefits such as increased soil organic matter. Such options require a much lower level of investment and inputs and are thus more readily applicable in developing countries, while also holding a low risk of increased GHG emissions (Keating, B.A. et al., 2013). Substantial progress has also been achieved in the last four years in small-scale bioenergy applications in the areas of technology innovation, impact evaluation and monitoring and in large-scale implementation programs. Regarding technology, advanced combustion biomass cookstoves which reduce fuel use by more than 60% and hazardous pollutant as well as short-lived climate pollutants by up to 90% are now in the last demonstration stages or commercial (Kar et al., 2012; Anenberg et al., 2013). Innovative designs include micro-gasifiers, stoves with thermoelectric generators to improve combustion efficiency and provide electricity to charge led lamps while cooking, stoves with advanced combustion chamber designs and multi-use stoves (e.g. cooking and water heating for bathing; Ürge-Vorsatz et al., 2012; Anenberg et al., 2013). Biogas stoves, in addition to provide clean combustion, help reduce the health risks associated to the disposal of organic wastes. There has also been a boost in cookstove dissemination efforts ranging from regional (multi-country) initiatives (Wang et al., 2013a) to national, and project level interventions. In total more than 200 cookstove large-scale projects are in place worldwide, with several million efficient cookstoves installed each year (Cordes, 2011). A Global Alliance for Clean Cook stoves has been launched that is promoting the adoption of 100 million clean and efficient cookstoves per year by 2030 and several countries have launched National Cookstove Programs in recent years (e.g., Mexico, Peru, Honduras, and others). Many cookstove models are now manufactured in large-scale industrial facilities using state-of the art materials and combustion design technology. Significant efforts are also in place to develop international standards and regional stove testing facilities. In addition to providing tangible local health and other sustainable benefits, replacing traditional open fires with efficient biomass cookstoves has a global mitigation potential estimated in between 0.6 and 2.4 $GtCO_2eq$/yr (Ürge-Vorsatz et al., 2012). Small scale decentralized biomass power generation systems based on biomass combustion and gasification and biogas production systems have the potential to meet the electricity needs of rural communities in the developing countries. The biomass feedstocks for these small-scale systems could come from residues of crops and forests, wastes from livestock production and/or from small-scale energy plantations (Faaij, 2006).

11.13.4 GHG emission estimates of bioenergy production systems

The combustion of biomass generates gross GHG emissions roughly equivalent to the combustion of fossil fuels. If bioenergy production is to generate a net reduction in emissions, it must do so by offsetting those emissions through increased net carbon uptake of biota and soils. The appropriate comparison is then between the net biosphere flux in the absence of bioenergy compared to the net biosphere flux in the presence of bioenergy production. Direct and indirect effects need to be considered in calculating these fluxes.

Bioenergy systems directly influence local and global climate through: (i) GHG emissions from fossil fuels associated with biomass production, harvest, transport, and conversion to secondary energy carriers (van der Voet et al. 2010; von Blottnitz and Curran 2007); (ii) CO_2 and other GHG emissions from biomass or biofuel combustion (Fernandes et al. 2011; Cherubini et al. 2011), (iii) atmosphere-ecosystem exchanges of CO_2 following land disturbance (Berndes et al. 2013; Haberl 2013; Amiro et al. 2010); (iv) climate forcing resulting from emissions of short-lived GHGs like black carbon and other chemically active gases (NO_x, CO, etc.) (Tsao et al. 2012; Jetter et al. 2012); (v) climate forcing resulting from alteration of biophysical properties of the land surface affecting the surface energy balance (e.g., from changes in surface albedo, heat and water fluxes, surface roughness, etc.; Bonan

2008; West et al. 2010; Pielke Sr. et al. 2011); and (vi) GHGs from land management and perturbations to soil biogeochemistry, e.g. N_2O from fertilizers, CH_4, etc (Cai et al. 2001; Allen et al. 2009). Indirect effects include the partial or complete substitution of fossil fuels and the indirect transformation of land use by equilibrium effects. Hence, the total climate forcing of bioenergy depends on feedstock, site-specific climate and ecosystems, management conditions, production pathway, end use, and on the interdependencies with energy and land markets.

In contrast, bioenergy systems have often been assessed (e.g., in LCA studies, integrated assessment models, policy directives, etc.) under the assumption that the CO_2 emitted from biomass combustion is climate neutral[16] because the carbon that was previously sequestered from the atmosphere will be re-sequestered if the bioenergy system is managed sustainably (Creutzig et al., 2012; Chum et al., 2011). The shortcomings of this assumption have been extensively discussed in environmental impact studies and emission accounting mechanisms (Cherubini et al. 2011; Searchinger et al. 2009; Searchinger et al. 2010; Haberl 2013),

Studies also call for a consistent and case-specific carbon stock/flux change accounting that integrates the biomass system with the global carbon cycle (Mackey et al. 2013). As shown in Chapter 8 of WGI (Myhre and Shindell, 2013) and elsewhere (Plattner et al., 2009; Fuglestvedt et al., 2010), the climate impacts can be quantified at different points along a cause-effect chain, from emissions to changes in temperature and sea level rise. While a simple sum of the net CO_2 fluxes over time can inform about the skewed time distribution between sources and sinks (“C debt”; Bernier and Paré, 2013; Fargione et al., 2008c; Marland and Schlamadinger, 1995), understanding the climate implications as it relates to policy targets (e.g., limiting warming to 2°C) requires models and/or metrics that also includes temperature effects and climate consequence (Smith et al., 2012b; Tanaka et al., 2013). While the warming from fossil fuels is nearly permanent as it persists for thousands of years, direct impacts from renewable bioenergy systems cause a perturbation in global temperature that is temporary and even at times cooling if terrestrial carbon stocks are not depleted (Cherubini et al., 2013; House et al., 2002; Joos et al., 2013; Mackey et al., 2013). The direct, physical climate effects at various end-points need to be fully understood and characterized– despite the measurement challenges that some climate forcing mechanisms can entail (Anderson-Teixeira et al., 2012; West et al., 2010b), and coherently embedded in mitigation policy scenarios along with the possible counterfactual effects. For example, in the specific case of existing forests that may continue to grow if not used for bioenergy, some studies employing counterfactual baselines show that forest bioenergy systems can temporarily have higher cumulative CO_2 emissions than a fossil reference system (for a time period ranging from few decades up to several centuries; Holtsmark, 2013; Pingoud et al., 2012a; Helin et al., 2012; Guest et al., 2013; Repo et al., 2011; Bernier and Paré, 2013; Mitchell et al., 2012)

In some cases, cooling contributions from changes in surface albedo can mitigate or offset these effects (Hallgren et al., in review; Anderson-Teixeira et al., 2012; O’Halloran et al., 2012; Arora and Montenegro, 2011).

Accounting always depends on the time horizon adopted when assessing climate change impacts, and the assumed baseline, and hence includes value judgements (Schwietzke et al., 2011; Cherubini et al., 2013; Kløverpris and Mueller, 2013a).

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[16] The neutrality perception is linked to a misunderstanding of the guidelines for GHG inventories, e.g., IPCC – Land Use, Land-Use Change and Forestry (2000) states “Biomass fuels are included in the national energy and carbon dioxide emissions accounts for informational purposes only. Within the energy module biomass consumption is assumed to equal its regrowth. Any departures from this hypothesis are counted within the Land Use Change and Forestry Model.” Carbon neutrality is valid if the countries account for LUC in their inventories for self-produced bioenergy.

Two specific contributions to the climate forcing of bioenergy, not addressed in detail in SRREN include: nitrous oxide and biogeophysical factors.

Nitrous oxide (N_2O) emissions: for first-generation crop-based biofuels, as with food crops (see Chapter 11), emissions of N_2O from agricultural soils is the single largest contributor to direct life cycle GHG emissions, and one of the largest contributors across many biofuel production cycles (Smeets et al., 2009a; Hsu et al., 2010). Emission rates can vary by as much as 700% between different crop types for the same site, fertilization rate and measurement period (Kaiser and Ruser, 2000; Don et al., 2012a; Yang et al., 2012). Increased estimates of N_2O emissions alone can convert some biofuel systems from apparent net sinks to net sources (Crutzen et al. 2007; Smith et al. 2012). Improvements in nitrogen use efficiency and nitrogen inhibitors can substantially reduce emissions of N_2O (Robertson and Vitousek 2009). For some specific crops, such as sugarcane, N_2O emissions can be low (Macedo et al., 2008; Seabra et al., 2011a) or high (Lisboa et al., 2011). Other bioenergy crops require minimal or zero N fertilization and can reduce GHG emissions relative to the former land use where they replace conventional food crops (Clair et al., 2008).

Biogeophysical factors: Land cover changes or land-use disturbances of the surface energy balance, such as surface albedo, surface roughness, and evapotranspiration influence the climate system (Betts 2007; Betts 2001; Bonan 2008; Jackson et al. 2008; Marland et al. 2003; Mahmood et al., 2013). Perturbations to these can lead to both direct and indirect climate forcings whose impacts can differ in spatial extent (global and/or local) (Bala et al. 2007; Davin et al. 2007). Surface albedo is found to be the dominant direct biogeophysical climate impact mechanism linked to land cover change at the global scale, especially in areas with seasonal snow cover (Bathiany et al. 2010; Claussen et al. 2001), with radiative forcing effects possibly stronger than those of the co-occuring C-cycle changes (Bright et al. 2011; Randerson et al. 2006; O'Halloran et al. 2012; Lohila et al. 2010; Cherubini et al. 2012a). Land cover changes can also affect other biogeophysical factors like evapotranspiration and surface roughness, which can have important local (Loarie et al. 2011; Georgescu et al. 2011) and global climatic consequences (Swann et al. 2010; Swann et al. 2011; Bala et al. 2007). Biogeophysical climate impacts from changes in land use are site specific and show variations in magnitude across different geographic regions and biomes (Anderson et al. 2010; Anderson-Teixeira et al. 2012; Bonan 2008; Pielke Sr. et al. 2011). Biogeophysical impacts should be considered in climate impact assessments and in the design of land use policies in order to adequately assess the net impacts of land-use mitigation options (Betts 2011; Jackson et al. 2008; Arora and Montenegro 2011) as their size may be comparable to impacts from changes to the C cycle.

Figure 11.23. illustrates the range of life-cycle global direct climate impact (in g CO_2 equivalents per MJ, after characterization with GWP time horizon=100 years) attributed to major global bioenergy products reported in the peer-reviewed literature after 2010. Results are broadly comparable to those of Chapter 2 in SRREN (Figure 2.10 and 2.11 in SRREN; those figures displayed negative emissions, resulting from crediting emission reduction due to substitution effects; this appendix refrains from allocating credits to feedstocks to avoid double accounting). Significant variation in the results reflects the wide range of conversion technologies and their reported performances in addition to analyst assumptions affecting system boundary completeness, emission inventory completeness, and choice of allocation method (among others). Additional “site-specific” land use considerations such as changes in soil organic carbon stocks (“ΔSOC”), changes in surface albedo (“Δalbedo”), and the skewed time distribution of terrestrial biogenic CO_2 fluxes can either reduce or compound land use impacts and are presented to exemplify that, for some bioenergy systems, these impacts can be greater in magnitude than life-cycle impacts from feedstock cultivation and bioenergy product conversion. “Site-specific” land-use considerations are geographically explicit and highly sensitive to background climate conditions, soil properties, biomass yields, and land management regimes. The figure reveals that studies find very different values depending on the boundaries of analysis chosen, site-specific effects and management methods. Nonetheless, it is

clear that fuels from sugarcane, perennial grasses, crop residues and waste cooking oil are more beneficial than other fuels (land use change emissions can still be relevant, see Fig 11.23). Another important result is that albedo effects and site-specific CO_2 fluxes are highly variable for different forest systems and environmental conditions and determine the total climate forcing of bioenergy from forestry.

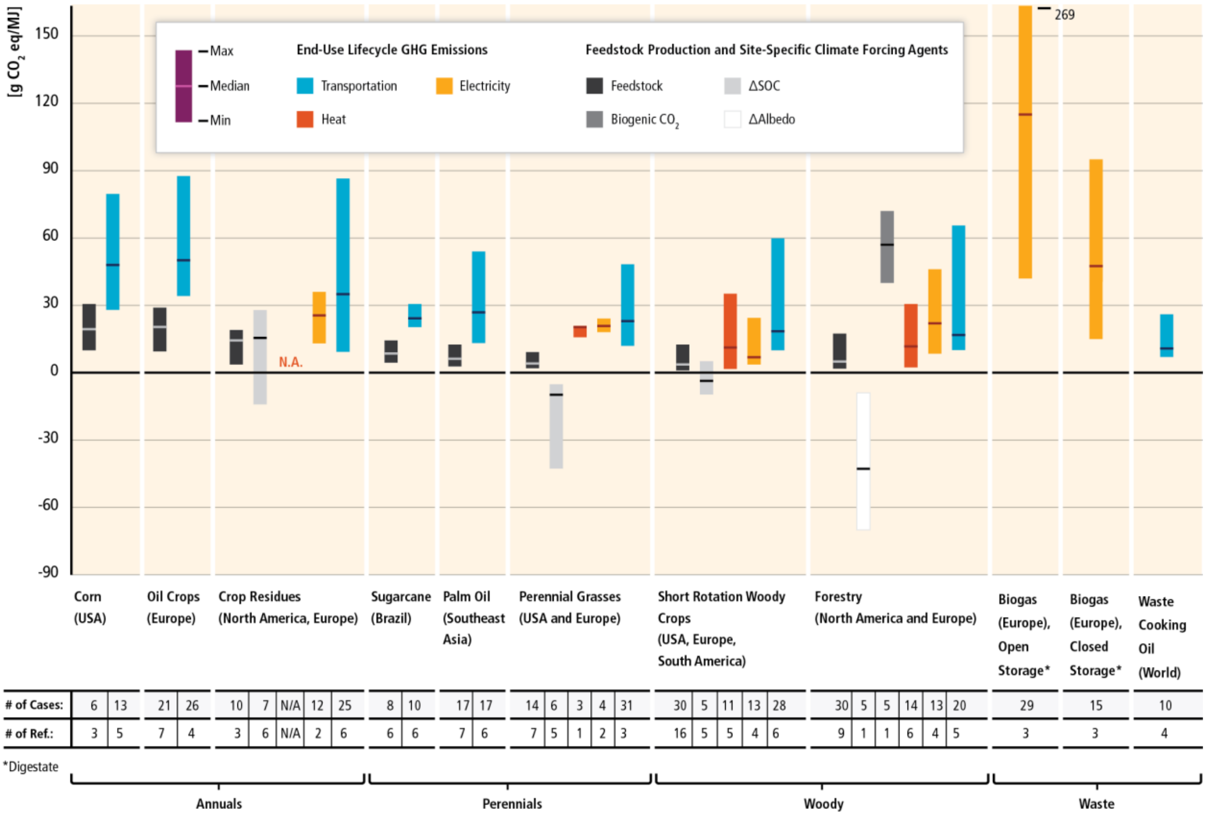

Figure 11.23. Direct CO_2-equivalent (GWP100) emissions from the process chain or land use disturbances of major bioenergy product systems, not including impacts from land use change (see Figure 11.24). The interpretation of values depends also on baseline assumption about the land carbon sink when appropriate and the intertemporal accounting frame chosen, and should also consider information from Figure 11.24. The lower and upper bounds of the bars represent the minimum and the maximum value reported in the literature. Whenever possible, only peer reviewed scientific literature published post SRREN is used (but results are comparable). Note that narrow ranges may be an artefact of the number of studies for a given case. Results are disaggregated in a manner showing the impact of *Feedstock* production (in g CO_2-eq./MJ LHV of feedstock) and the contributions from end product/conversion technology. Results from conversion into final energy products *Heat*, *Power*, and *Transport fuels* include the contribution from *Feedstock* production and are shown in g CO_2-eq./MJ of final product. For some pathways, additional site-specific climate forcing agents apply and are presented as separate values to be added or subtracted from the value indicated by the median in the *Feedstock* bar (dark grey). Final products are also affected by these factors, but this is not displayed here. References for: Corn 1-7; Oil crops 1, 8, 8-12; Crop residues 1, 4, 13-24; Sugarcane 2, 3, 5, 6, 25-27; Palm Oil 2, 3, 10, 28-31; Perennial grasses 1, 3, 11, 18, 22, 32-40; Short Rotation Woody Crops 1, 3, 6, 12, 22, 33, 35, 37, 38, 41-53; Forestry 5, 6, 38, 49, 54-66; Biogas, open storage: 67-69; Biogas, closed storage 69-71; Waste cooking oil: 22, 72-74. Note that the biofuels technologies for transport from lignocellulosic feedstocks, short rotation woody crops, and crop residues, including collection and delivery, are developing so larger ranges are expected than for more mature commercial technologies such as sugarcane ethanol and WCO biodiesel. The biogas electricity bar represents scenarios using LCAs to explore treating mixtures of a variety of lignocellulosic feedstocks (e.g., ensiled grain or agricultural residues or perennial grasses) with more easily biodegradable wastes (e.g., from animal husbandry), to optimize multiple outputs. Some of the scenarios assume CH_4 leakage, which leads to very high life-cycle emissions.

[1] Gelfand et al. (2013a); [2] Nemecek et al. (2012); [3] Hoefnagels et al. (2010); [4] Kaufman et al. (2010); [5] Cherubini et al. (2009) [6] Cherubini (2012a); [7] Wang et al. (2011b); [8] Milazzo et al. (2013); [9] Goglio et al. (2012); [10] Stratton et al. (2011); [11] Fazio and Monti (2011); [12] Börjesson and Tufvesson (2011); [13] Cherubini and Ulgiati (2010); [14] Li et al. (2012); [15] Luo et al. (2009); [16] Gabrielle and Gagnaire (2008); [17] Smith et al. (2012a); [18] Anderson-Teixeira et al. (2009); [19] Nguyen et al. (2013); [20] Searcy and Flynn (2008); [21] Giuntoli et al. (2013); [22] Whitaker et al. (2010); [23] Wang et al. (2013b); [24] Patrizi et al. (2013); [25] Souza et al. (2012a); [26] Seabra et al. (2011b); [27] Walter et al. (2011); [28] Choo et al. (2011); [29] Harsono et al. (2012); [30] Siangjaeo et al. (2011); [31] Silalertruksa and Gheewala (2012); [32] Smeets et al. (2009b); [33] Tiwary and Colls (2010); [34] Wilson et al. (2011); [35] Brandão et al. (2011a); [36] Cherubini and Jungmeier (2010); [37] Don et al. (2012b); [38] Pucker et al. (2012); [39] Monti et al. (2011); [40] Bai et al. (2010); [41] Bacenetti et al. (2012); [42] Budsberg et al. (2012); [43] González-García et al. (2013); [44] González-García (2012a); [45] Stephenson et al. (2010); [46] Hennig and Gawor (2012); [47] Buonocore et al. (2012); [48] Gabrielle et al. (2013); [49] Dias and Arroja (2012); [50] González-García et al. (2012b); [51] Roedl (2010); [52] Djomo et al. (2011); [53] Njakou Djomo et al. (2013); [54] McKechnie et al. (2011a); [55] Pa et al. (2012); [56] Puettmann et al. (2010); [57] Guest et al. (2011); [58] Valente et al. (2011); [59] Whittaker et al. (2011); [60] Bright and Strømman (2009); [61] Felder and Dones (2007); [62] Solli et al. (2009); [63] Lindholm et al. (2011); [64] Mallia and Lewis (2013); [65] Bright et al. (2010); [66] Bright and Strømman (2010); [67] Rehl et al. (2012); [68] Blengini et al. (2011); [69] Boulamanti et al. (2013); [70] Lansche and Müller (2012); [71] De Meester et al. (2012); [72] Sunde et al. (2011); [73] Thamsiriroj and Murphy (2011); [74] Talens Peiró et al. (2010)

Direct and indirect land use change: direct land use change (LUC) occurs when bioenergy crops displace other crops or pastures or forests, while ILUC results from bioenergy deployment triggering the conversion to cropland of lands, somewhere on the globe, to replace some portion of the displaced crops (Searchinger et al. 2008; Kloverpris et al. 2008; Hertel et al. 2010; Delucchi 2010). Direct LUC to establish biomass cropping systems can increase the net GHG emissions, for example if carbon rich ecosystems such as wetlands, forests or natural grasslands are brought into cultivation (Gibbs et al., 2008;UNEP, 2009; Chum et al., 2011). Biospheric C losses associated with LUC from some bioenergy schemes can be, in some cases, more than hundred times larger than the annual GHG savings from the assumed fossil fuel replacement (Gibbs et al. 2008; Chum et al. 2011). Impacts have been shown to be significantly reduced when a dynamic baseline includes future trends in global agricultural land use (Kløverpris and Mueller, 2013b). Albeit at lower magnitude, beneficial LUC effects can also be observed, for example when some semi-perennial crops, perennial grasses or woody plants replace annual crops grown with high fertilizer levels, or where such plants are produced on lands with carbon-poor soils (Tilman et al., 2006; Harper et al., 2009; Sochacki et al., 2012; Sterner and Fritsche, 2011). In particular, Miscanthus improves soil organic carbon reducing overall GHG emissions (Brandão et al., 2011b); degraded US Midwest land for economic agriculture, over a 20-year period, shows successional perennial crops without the initial carbon debt and indirect land-use costs associated with food-based biofuels (Gelfand et al., 2013b). Palm oil, when grown on more marginal grasslands, can deliver a good GHG balance and net carbon storage in soil (Wicke et al., 2008). Such lands represent a substantial potential for palm oil expansion in Indonesia without deforestation and draining peat lands (Wicke et al., 2011a).

In long-term rotation forests, the increased removal of biomass for bioenergy may be beneficial or not depending on the site-specific forest conditions (Cherubini et al., 2012b). For long-term rotation biomass, the carbon debt (increased cumulative CO_2 emissions for a duration in the order of a rotation cycle or longer) becomes increasingly important (Marland and Schlamadinger, 1997; Fargione et al., 2008b; Hudiburg et al., 2011; Schlamadinger and Marland, 1996; Hudiburg et al., 2011; McKechnie et al., 2011b). Calculations of specific GHG emissions from long-term rotation forests need to account for the foregone CO_2-accumulation (Holtsmark, 2012; Searchinger, 2010; Haberl et al., 2012; Pingoud et al., 2012b).

If part of a larger forest is used as a feedstock for bioenergy while the overall forest carbon stock increases (the so-called landscape perspective), then the overall mitigation effects is positive, in particular over several harvesting cycles making use of the faster carbon sequestration rates of younger forests (Daigneault et al., 2012; Latta et al., 2013; Lamers and Junginger, 2013; Ximenes et al., 2012). Nabuurs et al., 2013 observe first signs of a carbon sink saturation in European forest biomass and suggest to focus less on the forest biomass sink strength but to consider a mitigation strategy that maximizes the sum of all the possible components: a) carbon sequestration in forest biomass; b) soil and wood products; and c) the effects of material and energy substitution of woody biomass. In general, the use of easily decomposable residues and wastes for bioenergy can produce GHG benefits (Zanchi et al 2011), similarly to increasing the biomass outtake from forests affected by high mortality rates (Lamers et al., 2013), whereas the removal of slowly decomposing residues reduces soil carbon accumulation at a site and results in net emissions (Repo et al. 2011). The anticipation of future bioenergy markets may promote optimized forest management practices or afforestation of marginal land areas to establish managed plantations, so contributing to increased forest carbon stocks (Sedjo and Tian, 2012). Rather than leading to wide-scale loss of forest lands, growing markets for tree products can provide incentives for maintaining or increasing forest stocks and land covers, and improving forest health through management (Eisenbies et al., 2009; Dale et al., 2013). If managed to maximize CO_2 storage rate over the long-term, long-term rotation forests offer a low-cost mitigation options, in particular when woody products keep carbon within the human built environment over long time scales (e.g. wood substituting for steel joist; Lippke et al., 2011).

Indirect land-use change is difficult to ascertain because the magnitude of these effects must be modelled (Nassar et al., 2011) raising important questions about model validity and uncertainty (Liska and Perrin, 2009; Plevin et al., 2010a; Gawel and Ludwig, 2011; Khanna et al., 2011; Wicke et al., 2012) and policy implications (DeCicco, 2013; Finkbeiner, 2013; Plevin et al., 2013). Available model-based studies have consistently found positive and, in some cases, high emissions from LUC and ILUC, mostly of first-generation biofuels (Fig. 11.23), albeit with high variability and uncertainty in results (Warner, Zhang, Inman, & Heath, 2013; see also Hertel et al. 2010; Dumortier et al. 2011; Havlík et al. 2011; Taheripour et al. 2011; Chen & Khanna 2012; Timilsina et al. 2012). Causes of the great uncertainty include: incomplete knowledge on global economic dynamics (trade patterns, land use productivity, diets, use of by-products, fuel prices and elasticities); selection of specific policies modelled; and the treatment of emissions over time (O'Hare et al., 2009; Khanna et al., 2011; Wicke et al., 2012). In addition, LUC modelling philosophies and model structures and features (e.g. dynamic *vs*. static model) differ among studies. Variations in estimated GHG emissions from biofuel-induced LUC are also driven by differences in scenarios assessed, varying assumptions, inconsistent definitions across models (e.g. LUC, land type), specific selection of reference scenarios against which (marginal) LUC is quantified, and disparities in data availability and quality. The general lack of thorough sensitivity and uncertainty analysis hampers the evaluation of plausible ranges of estimates of GHG emissions from LUC.

(Wicke et al., 2012) identified the need to incorporate the impacts of ILUC prevention or mitigation strategies in future modelling efforts, including the impact of zoning and protection of carbon stocks, selective sourcing from low risk areas, policies and investments to improve agricultural productivity, double cropping, agroforestry schemes and the (improved) use of degraded and marginal lands (see Box 7.1). ILUC is mostly avoided in the modelled mitigation pathways in Chapter 6. The relatively limited fuel coverage in the literature precludes a complete set of direct comparisons across alternative and conventional fuels sought by regulatory bodies and researchers.

GHG emissions from LUC can be reduced, for instance through production of bioenergy co-products that displace additional feedstock requirements thus decreasing the net area needed (e.g., for corn, Wang et al. 2011; for wheat, Berndes et al. 2011). Proper management of lifestock and agriculture can lead to improved resource efficiency, lower GHG emissions and lower land use while releasing land for bioenergy production as demonstrated for Europe (de Wit et al., 2013a) and Mozambique (van der Hilst et al., 2012b). For land transport, cellulosic biomass, such as Miscanthus, has been suggested as a relatively low-carbon source for bioethanol that could be produced at scale, but only if ILUC can be avoided by not displacing food and other commodities and if a comprehensive national land management strategies are developed (e.g., Dornburg et al., 2010; Scown et al., 2012).

Negative ILUC values are theoretically possible (RFA, 2008). Producing biofuels from wastes and sustainably harvested residues, and replacing first generation biofuel feedstocks with lignocellulosic crops (e.g. grasses) would induce little or no ILUC (Davis et al. 2012; Scown et al. 2012). While ILUC quantifications remain uncertain, lower agricultural yields, land-intensive diets, and livestock feeding efficiencies, stronger climate impacts and higher energy crop production levels can result in higher LUC-related GHG emissions. Strong global and regional governance (forest protection, zoning), technological change in agriculture and biobased options, and high-yield bioenergy crops and use of residues and degraded land (if available) could all reduce ILUC (Van Dam et al., 2009a; b; Wicke et al., 2009; Fischer et al., 2010; De Wit et al., 2011; van der Hilst et al., 2012b; Rose et al., 2013b; de Wit et al., 2013a). As with any other renewable fuel, bioenergy can replace or complement fossil fuel. The fossil fuel replacement effect, relevant when a global cap on CO_2 emissions is absent, is discussed in Chapter 8.7. Indirect effects are not restricted to indirect GHG effects of production of biomass in agricultural systems; there are also indirect (market mediated) effects of wood energy, but also effects in terms of biodiversity threats, environmental degradation, and external social costs, which are not considered here.

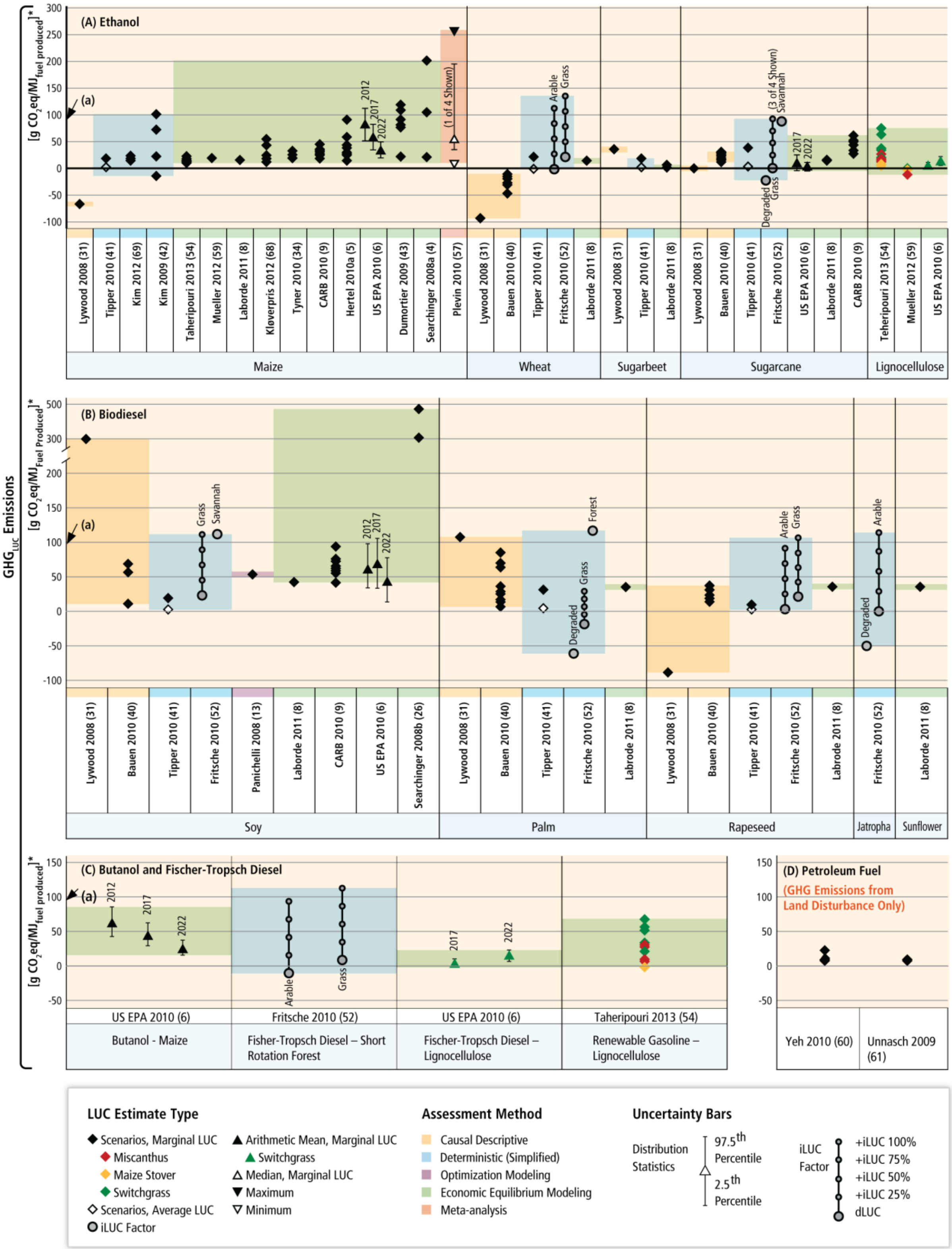

Figure 11.24. Estimates of GHG_{LUC} emissions - GHG emissions from biofuel production-induced LUC (as g CO_2eq/$MJ_{\text{fuel produced}}$) over a 30 year time horizon organized by fuel(s), feedstock, and study. Assessment methods, LUC estimate types and uncertainty metrics are portrayed to demonstrate the diversity in approaches and differences in results within and across any given category. Points labeled “a” on the Y axis represent a commonly used estimate of life cycle GHG emissions associated with the direct supply chain of petroleum gasoline (frame A) and diesel (frame B). These emissions are not directly comparable to GHG_{LUC} because the emission sources considered are different, but are potentially of interest for scaling comparison. Based on (Warner et al., 2013). Please note: These estimates of global LUC are highly uncertain, unobservable,

unverifiable, and dependent on assumed policy, economic contexts, and inputs used in the modeling. All entries are not equally valid nor do they attempt to measure the same metric despite the use of similar naming conventions (e.g., ILUC). In addition, many different approaches to estimating GHG_{LUC} have been used. Therefore, each paper has its own interpretation and any comparisons should be made only after careful consideration. *CO_2eq includes studies both with and without CH_4 and N_2O accounting.

11.13.5 Aggregate future potential deployment in Integrated Models

In the IPCC SRREN scenarios, bioenergy is projected to contribute 80 to 190 EJ/yr to global primary energy supply by 2050 for 50% of the scenarios in the two climate mitigation levels modelled. The min to max ranges were 20 – 265 EJ/yr for the less stringent scenarios and 25 to 300 EJ for the tight climate mitigation scenarios (<440ppm). Many of these scenarios coupled bioenergy with CCS. The GEA (GEA, 2012) scenarios project 80–140 EJ by 2050, including extensive use of agricultural residues and second-generation bioenergy to try to reduce the adverse impacts on land use and food production, and the co-processing of biomass with coal or natural gas with CCS to make low net GHG-emitting transport fuels and or electricity.

Traditional biomass demand is steady or declines in most scenarios from 34 EJ/yr. The transport sector increases nearly tenfold from 2008 to 18-20 EJ/yr while modern uses for heat, power, combinations, and industry increase by factors of 2-4 from 18 EJ in 2008 (Fischedick et al., 2011). The 2010 IEA model projects a contribution of 12 EJ/yr (11%) by 2035 to the transport sector, including 60 % of advanced biofuels for road and aviation. Bioenergy supplies 5% of global power generation in 2035, up from 1% in 2008. Modern heat and industry doubles their contributions from 2008 (IEA, 2010b). The future potential deployment level varies at the global and national level depending on the technological developments, land availability, financial viability and mitigation policies.

The AR5 transformation pathway studies suggest that modern bioenergy could play a significant role within the energy system (6.3.5) providing 5 to 95 EJ/yr in 2030, 10 to 245 EJ/yr in 2050 and 105 to 325 EJ/yr in 2100 under idealized full implementation scenarios (see also Figure 7.12), with immediate, global, and comprehensive incentives for land related mitigation options. The scenarios project increasing deployment of bioenergy with tighter climate change targets, both in a given year as well as earlier in time (see Figure 6.20). Models project increased dependence, as well as increased deployment, of modern bioenergy, with some models projecting 35% of total primary energy from bioenergy in 2050, and as much as 50% of total primary energy from modern bioenergy in 2100. Bioenergy’s share of regional total electricity and liquid fuels could be significant—up to 35% of global regional electricity from biopower by 2050, and up to 70% of global regional liquid fuels from biofuels by 2050. However, the cost-effective allocation of bioenergy within the energy system varies across models. Several sectoral studies, focusing on biophysical constraints, model assumptions (e.g. estimated increase in crop yields over large areas), current observations, suggest to focus on the lower half the ranges reported above (Field et al., 2008; Campbell et al., 2008b; Johnston et al., 2009a, 2011; Haberl et al., 2013b).

BECCS features prominently in many transformation scenarios. BECCS is deployed in greater quantities and earlier in time the more stringent the climate policy (6.3.5). Whether BECCS is essential for mitigation, or even sufficient, is unclear. In addition, the likelihood of BECCS deployment is difficult to evaluate and depends on safety confirmations, affordability and public acceptance (see 11.13.3 for details). BECCS may also affect the cost-effective emissions trajectory (Richels et al., In Review; Rose et al., 2013b).

Some integrated models are cost-effectively trading-off lower land carbon stocks and increased land N_2O emissions for the long-run mitigation benefits of bioenergy (Popp et al., 2013; Rose et al., 2013b). The models find that bioenergy could contribute effectively to climate change mitigation despite land conversion and intensification emissions. However, as discussed below and in Section 11.9, policy implementation and coordination are factors to consider. In these models, constraining bioenergy has a cost. For instance, limiting global bioenergy availability to 100 EJ/year tripled marginal abatement costs and doubled consumption losses associated with transformation pathways (Rose et al., 2013b). Overall outcomes may depend strongly on governance of land use and deployment of best practices in agricultural production (see sections above). Progressive developments in governance of land and modernization of agriculture and livestock and effective sustainability frameworks can help realize large parts of the technical bioenergy potential with low associated GHG emissions.

With increasing scarcity of productive land, the growing demand for food and bioenergy could induce substantial LUC causing high GHG emissions and/or increased agricultural intensification and higher N_2O emissions unless wise integration of bioenergy into agriculture and forestry landscapes occurs (Delucchi, 2010). Consideration of LUC emissions in integrated assessment models show that valuing or protecting global terrestrial carbon stocks reduces the potential LUC-related GHG emissions of energy crop deployment, and could lower the cost of achieving climate change objectives, but could exacerbate increases in agricultural commodity prices (Popp et al., 2011a; Reilly et al., 2012b). Many integrated models are investigating idealized policy implementation pathways, assuming global prices on greenhouse gases (including the terrestrial land carbon stock); if such conditions cannot be realized, certain types of bioenergy could lead to additional GHG emissions. More specifically, if the global terrestrial land carbon stock remains unprotected, large GHG emissions from bioenergy related land use change alone are possible (Melillo et al., 2009; Wise et al., 2009a; Creutzig et al., 2012; Calvin et al., 2013b).

In summary, recent integrated model scenarios project between 10-245 EJ/yr modern bioenergy deployment in 2050. Good governance and favourable conditions for bioenergy development may facilitate higher bioenergy deployment while sustainability and livelihood concerns might constrain deployment of bioenergy scenarios to low values (see 11.13.6).

11.13.6 Bioenergy and sustainable development

The nature and extent of the impacts of implementing bioenergy depend on the specific system, the development context and on the size of the intervention (11.4.5). The effects on livelihoods have not yet been systematically evaluated in integrated assessments (Creutzig et al., 2012; Davis et al., 2013a; Muys et al., 2013), even if human geography studies have shown that bioenergy deployment can have strong distributional impacts (Davis et al., 2013; Muys et al., 2013). The total effects on livelihoods will be mediated by global market dynamics, including policy regulations and incentives, the production model and deployment scale, and place-specific factors such as governance, land tenure security, labour and financial capabilities, among others (Creutzig et al., 2013).
Bioenergy projects can be economically beneficial, e.g. by raising and diversifying farm incomes and increasing rural employment through the production of biofuels for domestic (Gohin, 2008) or export (Arndt et al. 2011) markets (Wicke et al., 2009).

Box 11.9 Some reported examples of co-benefits from biofuel production

Brazilian sugar cane ethanol production provides six times more jobs than the Brazilian petroleum sector and spreads income benefits across numerous municipalities (de Moraes et al., 2010). Worker income is higher than in nearly all other agricultural sectors (de Moraes et al., 2010; Satolo and Bacchi, 2013) and several sustainability standards have been adopted (Viana and Perez, 2013). When substituting gasoline, ethanol from sugarcane also eliminates lead compounds and reduces noxious emissions (Goldemberg et al., 2008). Broader strategic planning, understanding of cumulative impacts, and credible and collaborative decision making processes can help to enhance biodiversity and reverse ecological fragmentation, address direct and indirect land use change, improve the quality and durability of livelihoods, and other sustainability issues (Duarte et al., 2013).

Co-benefits of palm oil production have been reported in the major producer countries, Malaysia and Indonesia (Sumathi et al., 2008; Lam et al., 2009) as well as from new producer countries (Garcia-Ulloa et al., 2012). Palm oil production results in employment creation as well as in increments state and individual income (Sumathi et al., 2008; Tan et al., 2009; Lam et al., 2009; Sayer et al., 2012; von Geibler, 2013). When combined with agroforestry palm oil plantations can increase food production locally and have a positive impact on biodiversity (Lam et al., 2009; Garcia-Ulloa et al., 2012) and when palm oil plantations are installed on degraded land further co-benefits on biodiversity and carbon enhancement (Sumathi et al., 2008; Garcia-Ulloa et al., 2012; Sayer et al., 2012). Further, due to its high productivity palm oil plantations can produce the same bioenergy input using less land than other bio-energy crops (Sumathi et al., 2008; Tan et al., 2009). Certification in palm oil production can become a means for increasing sustainable production of biofuels (Tan et al., 2009; “Inaugural report shows impressive uptake of RSPO-certified sustainable palm oil in 2011,” 2012; von Geibler, 2013).

Similarly, co-benefits from the production of Jatropha as a biofuel crop in developing countries have been reported, mainly when Jatropha is planted on degraded land. These include increases in individuals income (Garg et al., 2011c; Arndt et al., 2012), improvement in energy security at the local level (Muys et al., 2013; von Maltitz and Setzkorn, 2013), and reducing soil erosion (Garg et al., 2011c).

The establishment of large-scale biofuels feedstock production can also cause smallholders, tenants and herders to lose access to productive land, while other social groups such as workers, investors, company owners, biofuels consumers, and populations who are more responsible for GHG emission reductions enjoy the benefits of this production (van der Horst and Vermeylen, 2011). This is particularly relevant where large areas of land are still unregistered or are being claimed and under dispute by several users and ethnic groups (Dauvergne and Neville, 2010). Furthermore, increasing demand for first-generation biofuels is partly driving the expansion of crops like soy and oil palm, which in turn contribute to promote large-scale agribusinesses at the expense of family and community-based agriculture, in some cases (Wilkinson and Herrera, 2010). Biofuels deployment can also translate into reductions of time invested in on-farm subsistence and community-based activities, thus translating into lower productivity rates of subsistence crops and an increase in intra-community conflicts as a result of the uneven share of collective responsibilities (Mingorría et al., 2010).

Bioenergy deployment is more beneficial when it is not an additional land use activity expanding over the landscape, but rather integrates into existing land uses and influences the way farmers and forest owners use their land. Various studies indicate the ecosystem services and values that perennial crops have in restoring degraded lands, via agroforestry systems, controlling erosion and even in regional climate effects such as improved water retention and precipitation (Faaij, 2006; Wicke et al., 2011d; van der Hilst et al., 2012b; Immerzeel et al., 2013). Examples include adjustments in agriculture practices where farmers, for instance, change their manure treatment to produce biogas, reduce methane losses and reduce N losses. Changes in management practice may swing the net GHG balance of options and also have clear sustainable development implications (Davis et al., 2012).

Small-scale bioenergy options can provide cost-effective alternatives for mitigating climate change, at the same time helping advance sustainable development priorities, particularly in rural areas of developing countries. IEA (2011) estimates that 2.7 billion people worldwide depend on traditional biomass for cooking and heating, while 84% of these belong to rural communities. Use of low quality fuels and inefficient cooking and heating devices leads to pollution resulting in nearly 4 million premature deaths every year, and a range of chronic illnesses and other health problems (Lim et al., 2012; see Section 9.7.3.1). Modern small-scale bioenergy technologies such as advanced/efficient cook stoves, biogas for cooking and village electrification, biomass gasifiers and bagasse based co-generation systems for decentralized power generation, can provide energy for rural communities with energy services that also promote rural development (IEA, 2011). Such bioenergy systems reduce CO_2 emissions from unsustainable biomass harvesting and short-lived climate pollutants, e.g. black carbon, from cleaner combustion (FAO, 2010b; Chung et al., 2012). Scaling up clean cookstove initiatives could not only save 2 million lives a year, but also significantly reduce GHG emissions (Section 11.13.3). Efficient biomass cook stoves and biogas stoves at the same time provide multiple benefits: reduce pressure on forests and biodiversity, reduce exposure to smoke related health hazards, reduce drudgery for women in collecting fuelwood and save money if purchasing fuels (Martin et al., 2011). Benefits from the dissemination of improved cookstoves outweigh their costs by 7 fold, when their health, economic, and environmental benefits are accounted for (Garcia-Frapolli et al., 2010).

Table 11.12 presents the implications of bioenergy options in the light of social, institutional, environmental, economic and technological conditions. The relationship between bioenergy and these conditions is complex and there could be negative or positive implications, depending on the type of bioenergy option, the scale of the production system and the local context. While biofuels can allow the reduction of fossil fuel use and of greenhouse gas emissions, they often shift environmental burdens towards land use-related impacts (i.e. eutrophication, acidification, water depletion, ecotoxicity; EMPA, 2012; Smith and Torn, 2013; Tavoni and Socolow, 2013b). Co-benefits and adverse side-effects do not necessarily overlap, neither geographically nor socially (Dauvergne and Neville, 2010; Wilkinson and Herrera, 2010; van der Horst and Vermeylen, 2011). The main potential co-benefits are related to access to energy and impacts on the economy and wellbeing, jobs creation and improvement of local resilience (Walter et al., 2011; Creutzig et al., 2013). Main risks of crop-based bioenergy for sustainable development and livelihoods include competition on arable land (Haberl et al., 2013b) and consequent impact on food security, tenure arrangements, displacement of communities and economic activities, creation of a driver of deforestation, impacts on biodiversity, water and soil or increment in vulnerability to climate change, and unequal distribution of benefits (Sala et al., 2000; Thompson et al., 2011; Hall et al., 2009; German et al., 2011; SREX, 2012).

Good governance is an essential component of a sustainable energy system. Integrated studies that compare impacts of bioenergy production between different crops and land management strategies show that the overall impact (both ecological and socio-economic) depends strongly on the governance of land use and design of the bioenergy system (see van der Hilst et al. (2012a) in the European context and J. Van Dam et al. (2009a, 2009b) for different crops and scenarios in Argentina). Van Eijck et al. (2012) show similar differences in impacts between the production and use of Jatropha based on smallholder production versus plantation models. This implies that governance and planning have a strong impact on the ultimate result and impact of large-scale bioenergy deployment. Legislation and regulation of bioenergy as well as voluntary certification schemes are required to guide bioenergy production system deployment so that the resources and feedstocks be put to best use, and that (positive and negative) socioeconomic and environmental issues are addressed as production grows (van Dam et al., 2010). There are different options, from voluntary to legal and global agreements, to improve governance of biomass markets and land use that still require much further attention (Verdonk et al., 2007). The integration of bioenergy systems into agriculture and forest landscapes can improve land and water use efficiency and help address concerns about environmental impacts of present land use (Berndes et al., 2004, 2008; Börjesson and Berndes, 2006; Sparovek et al., 2007b; Gopalakrishnan et al., 2009, 2011a; b, 2012; Dimitriou et al., 2009, 2011; Dornburg et al., 2010; Garg et al., 2011a; Batidzirai et al., 2012; Parish et al., 2012; Baum et al., 2012; Busch, 2012), but the global potentials of such systems are difficult to determine (Berndes and Börjesson, 2007; Dale and Kline, 2013). Similarly, existing and emerging guiding principles and governance systems influence biomass resources availability (Stupak, Lattimore, Titus, & Smith, 2011). Certification approaches can be useful, but they should be accompanied by effective territorial policy frameworks (Hunsberger et al., 2012).

Table 11.12: Potential institutional, social, environmental, economic and technological implications of bioenergy options at local to global scale

Institutional		Scale
May contribute to energy independence (+), especially at the local level (reduce dependency on fossil fuels) (2, 20, 32, 39,50)	+	Local to national
Can improve (+) or decrease (-) land tenure and use rights for local stakeholders (2, 17, 38, 50)	+/-	Local
Cross-sectoral coordination (+) or conflicts (-) between forestry, agriculture, energy and/or mining (2, 13, 26, 31, 60)	+/-	Local to national
Impacts on labor rights among the value chain (2, 6, 17)	+/-	Local to national
Promoting of participative mechanisms for small scale producers (14, 15)	+	Local to national
Social		**Scale**
Competition with food security including food availability (through reduced food production at the local level), food access (due to price volatility) use usage (as food crops can be diverted towards biofuel production) and consequently to food stability. Bio-energy derived from residues, wastes or by-products is an exception (1,2, 7, 9, 12, 18, 23)	–	Local to global
Integrated systems (including agroforestry) can improve food production at the local level creating a positive impact towards food security (51, 52, 53, 69, 74,75).) Further, biomass production combined with improved agricultural management can avoid such competition and bring investment in agricultural production systems with overall improvements of management as a result (as observed in Brazil). (60, 63 66, 67, 70, 71)	+	Local
Increasing (+) or decreasing (-) existing conflicts or social tension (9, 14, 19, 26)	+/-	Local to national
Impacts on traditional practices: using local knowledge in production and treatment of bioenergy crops (+) or discouraging local knowledge and practices (-) (2, 50)	+/-	Local
Displacement of small-scale farmers (14, 15, 19). Bioenergy alternatives can also empower local farmers by creating local income opportunities.	+/_	Local
Promote capacity building and new skills (3, 15, 50)	+	Local
Gender impacts (2, 4, 14, 15, 27)	+/-	Local to national
Efficient biomass techniques for cooking (e.g. biomass cook-stoves) can have positive impacts on health specially for women and children in developing countries (42, 43, 44)	+	Local to national
Environmental		**Scale**
Biofuel plantations can promote deforestation and/or forest degradation, under weak or no regulation (1, 8, 22).	_	Local to global
When used on degraded lands, perennial crops offer large-scale potential to improve soil carbon and structure, abate erosion and salinity problems. Agroforestry schemes can have multiple benefits including increased overall biomass production, increase biodiversity and higher resilience to climate changes. (59, 64, 65, 69, 74)	+	Local to global
Some large-scale bio-energy crops can have negative impacts on soil quality, water pollution and biodiversity. Similarly potential adverse side effects can be a consequence of increments in use of fertilizers for increasing productivity (7, 12, 26, 30). Experience with sugarcane plantations has shown that they can maintain soil structure (56) and application of pesticides can be substituted by the use of natural predators and parasitoids (57, 71).	-/+	Local to transboundary
Can displace activities or other land uses (8, 26)	_	Local to global
Smart modernization and intensification can lead to lower environmental impacts and more efficient land use (76, 77)..	+	Local to transboundary
Creating bio-energy plantations on degraded land can have positive impacts on soil and biodiversity (12)	+	Local to transboundary
There can be trade-offs between different land uses, reducing land availability for local stakeholders (45, 46, 47,48, 49). Multicropping system provide bioenergy while better maintaining ecological diversity and reducing land use competition (58).	-/+	Local to national
Ethanol utilization leads to the phase-out of lead addititives and MBTE and reduces sulfur, particulate matter and carbon monoxide emissions (55)	+	Local to global
Economic		**Scale**
Increase in economic activity, income generation and income diversification (1, 2, 3, 12, 20, 21, 27, 54)	+	Local
Increase (+) or decrease (-) market opportunities (16, 27, 31)	+/-	Local to national

Contribute to the changes in prices of feedstock (2, 3, 5, 21)	+/-	Local to global
May promote concentration of income and /or increase poverty if sustainability criteria and strong governance is not in place (2, 16, 26)	_	Local to regional
Using waste and residues may create socio-economic benefits with little environmental risks (2, 41, 36)	+	Local to regional
Uncertainty about mid- and long term revenues (6, 30)	-	National
Employment creation (3, 14, 15)	+	Local to regional
Technological		**Scale**
Can promote technology development and/or facilitate technology transfer (2, 27, 31)	+	Local to global
Increasing infrastructure coverage (+). However if access to infrastructure and/or technology is reduced to few social groups it can increase marginalization (-) (27, 28, 29)	+/-	Local
Bioenergy options for generating local power or to use residues may increase labor demand, creating new job opportunities. Participatory technology development also increases acceptance and appropriation (6, 8, 10, 37, 40)	+	Local
Technology might reduce labor demand (-). High dependent of tech. transfer and/or acceptance	-	Local

[1]Alves Finco and Doppler (2010); [2]Amigun et al. (2011); [3]Arndt et al. (2012); [4]Arndt et al. (2011a); [5]Arndt et al. (2011c); [6]Awudu and Zhang (2012); [7]Beringer et al. (2011); [8]Borzoni (2011); [9]Bringezu et al. (2012); [10]Cacciatore et al. (2012); [11]Cançado et al. (2006); [12]Danielsen et al. (2009); [13]Diaz-Chavez (2011b); [14]Duvenage et al. (2013); [15]Ewing and Msangi (2009); [16]Gasparatos et al. (2011); [17]German and Schoneveld (2012); [18]Haberl et al. (2011a); [19]Hall et al. (2009); [20]Hanff et al. (2011); [21]Huang et al. (2012); [22]Koh and Wilcove (2008); [23]Koizumi (2013); [24]Kyu et al. (2010); [25]Madlener et al. (2006); [26]Martinelli and Filoso (2008); [27]Mwakaje (2012); [28]Oberling et al. (2012); [29]Schut et al. (2010); [30]Selfa et al. (2011); [31]Steenblik (2007); [32]Stromberg and Gasparatos (2012); [33]Searchinger et al. (2009); [34]Searchinger et al. (2008b); [35]Smith and Searchinger (2012); [36]Tilman et al. (2009); [37]Van de Velde et al. (2009); [38]von Maltitz and Setzkorn (2013); [39]Wu and Lin (2009); [40]Zhang et al. (2011); [41]Fargione et al. (2008a); [42]Jerneck and Olsson (2012); [43]Gurung and Oh (2013); [44]O'Shaughnessy et al. (2013); [45]German et al. (2013); [46]Cotula (2012); [47]Mwakaje (2012); [48]Scheidel and Sorman (2012); [49]Haberl et al. (2013a); [50]Muys et al. (2013); [51]Egeskog et al. (2011); [52]Diaz-Chavez (2012); [53]Ewing and Msangi (2009); [54]de Moraes et al. (2010); [55]Goldemberg (2007); [56]Walter et al. (2011); [57]Macedo (2005); [58]Langeveld et al. (2013); [59]Van Dam et al. (2009a; b); [60]van Dam et al. (2010); [61]van Eijck et al. (2012); [62]Eijck et al. (2013a; b); [63]Martínez et al. (2013); [64]van der Hilst et al. (2010); [65]van der Hilst et al. (2012a); [66]van Der Hilst and Faaij (2012); [67]van der Hilst et al. (2012c); [68]Hoefnagels et al. (2013); [69]Immerzeel et al. (2013); [70]Lynd et al. (2011); [71]Smeets et al. (2008); [72]Smeets and Faaij (2010); [73]Wicke et al. (2011a); [74]Wicke et al. (2013); [75]Wiskerke et al. (2010); [76]De Wit et al. (2011); [77]de Wit et al. (2013b)

11.13.7 Trade offs and Synergies with Land, Water, Food and Biodiversity

This section summarizes results from integrated models (models that have a global aggregate view, but cannot disaggregate place-specific effects in biodiversity and livelihoods discussed above) on land, water, food and biodiversity. In these models, at any level of future bioenergy supply, land demand for bioenergy depends on (1) the share of bioenergy derived from wastes and residues (Rogner et al., 2012a); (2) the extent to which bioenergy production can be integrated with food or fiber production, which ideally results in synergies (Garg et al., 2011b; Sochacki et al., 2012) or at least mitigates land-use competition (Berndes et al., 2013); (3) the extent to which bioenergy can be grown on areas with little current or future production, taking into account growing land demand for food (Nijsen et al., 2012a); and (4) the volume of dedicated energy crops and their yields (Haberl et al., 2010; Batidzirai et al., 2012; Smith et al., 2012). Energy crop yields per unit area may differ by factors of >10 depending on differences in natural fertility (soils, climate), energy crop plants, previous land use, management and technology (Johnston et al., 2009b; Lal, 2010; Beringer et al., 2011; Pacca and Moreira, 2011; Smith et al., 2012d), (Erb et al., 2012a)Assumptions on energy crop yields are one of the main reasons for the large differences in estimates of future area demand of energy crops (Popp et al., 2013). Likewise, assumptions on yields, strategies and governance on future food/feed crops have large implications for assessments of the degree of land competition between biofuels and these land uses (Batidzirai et al., 2012; de Wit et al., 2013b)

However, across models, there are very different potential landscape transformation visions in all regions (6.3.5 and 11.9.). Overall, it is difficult to generalize on regional land cover effects of mitigation. Some models assume significant land conversion while other models do not. In idealized implementation scenarios, there is expansion of energy cropland and forest land in many regions, with some models exhibiting very strong forest land expansion and others very little by 2030. Land conversion is increased in the 450 ppm scenarios compared to the 550 ppm scenarios, but at a declining share, a result consistent with a declining land-related mitigation rate with policy stringency. The results of these integrated model studies need to be interpreted with caution, as not all GHG emissions and biogeophysical or socio-economic effects of bioenergy deployment are incorporated into these models, and as not all relevant technologies are represented (e.g. cascade utilization).

Large-scale bioenergy production from dedicated crops may affect water availability and quality (see Section 6.6.2.6.), which are highly dependent on (1) type and quantity of local freshwater resources; (2) necessary water quality; (3) competition for multiple uses (agricultural, urban, industrial, power generation) and (4) efficiency in all sector end-uses (Gerbens-Leenes et al., 2009; Coelho et al., 2012). In many regions, additional irrigation of energy crops could further intensify existing pressures on water resources (Popp, Dietrich, et al., 2011). Studies indicate that an exclusion of severe water scarce areas for bioenergy production (mainly to be found in the Middle East, parts of Asia and western USA) would reduce global technical bioenergy potentials by 17 % until 2050 (van Vuuren et al., 2009a). A model comparison study with five global economic models shows that the aggregate food price effect of large-scale ligno-cellulosic bioenergy deployment (i.e. 100 EJ globally by the year 2050) is significantly lower (+5% on average across models) than the potential price effects induced by climate impacts on crop yields (+25% on average across models (Lotze-Campen, Hermann et al., 2013). Possibly hence, ambitious climate change mitigation need not drive up global food prices much, if the extra land required for bioenergy production is accessible or if the feedstock, e.g. from forests, does not directly compete for agricultural land. Effective land-use planning and strict adherence to sustainability criteria need to be integrated to large-scale bioenergy projects to minimize competitions for water (for example, by excluding the establishment of biofuel projects in irrigated areas). If bioenergy is not managed properly, additional land demand and associated land use change may put pressures on biodiversity (Groom et al. 2008; see Section 6.6.2.5). However, implementing appropriate management, such as establishing bioenergy crops in degraded areas represents an opportunity where bioenergy can be used to achieve positive environmental outcomes (Nijsen et al., 2012b).

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