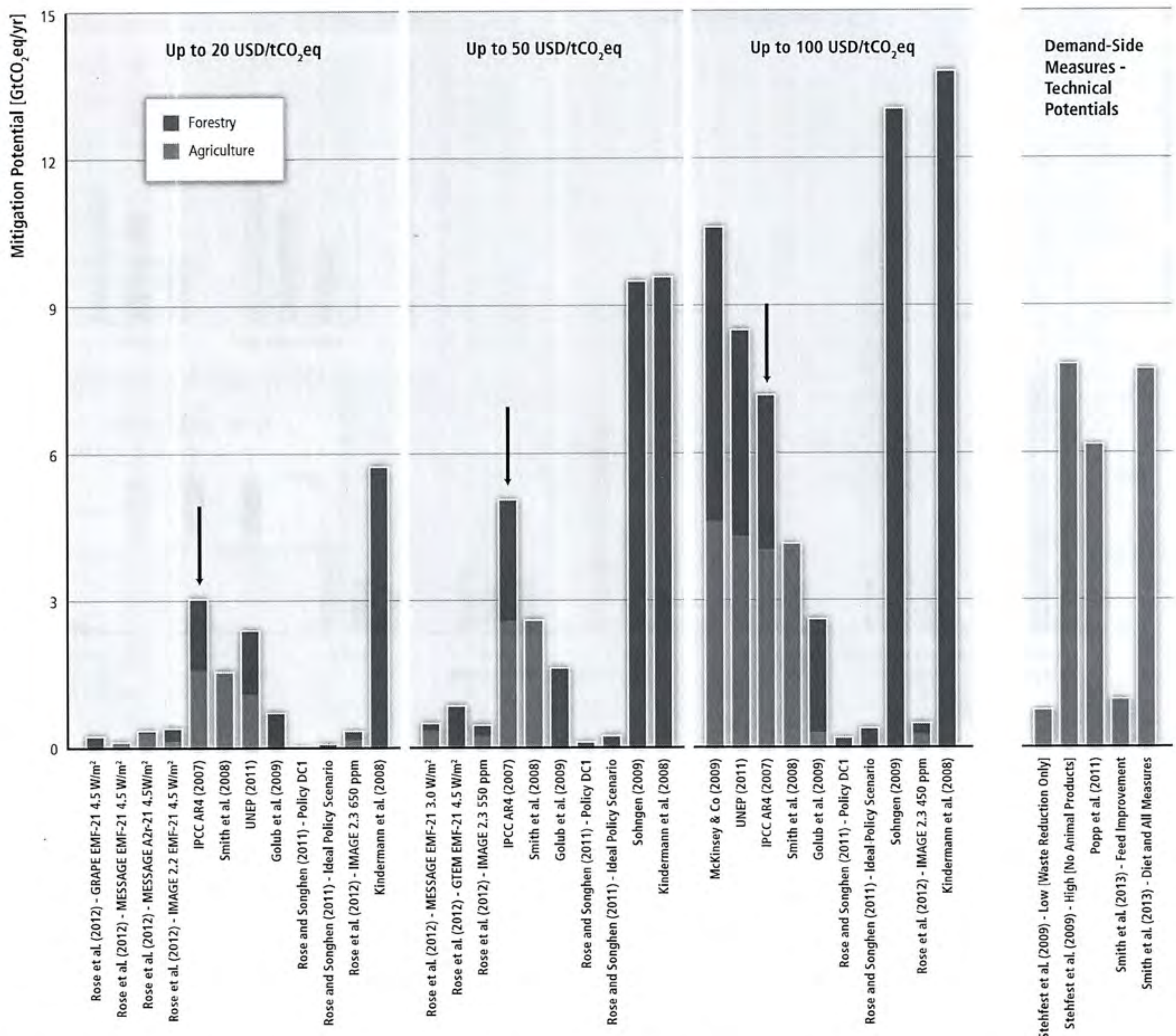


Figure 11.14 | Estimates of economic mitigation potentials in the AFOLU sector published since AR4, (AR4 estimates shown for comparison, denoted by arrows), including bottom-up, sectoral studies, and top-down, multi-sector studies. Some studies estimate potential for agriculture and forestry, others 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 carbon prices of up to ~20 USD/tCO$_2$eq (actual range 1.64–21.45), up to ~50 USD/tCO$_2$eq (actual range 31.39–50.00), and up to ~100 USD/tCO$_2$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.

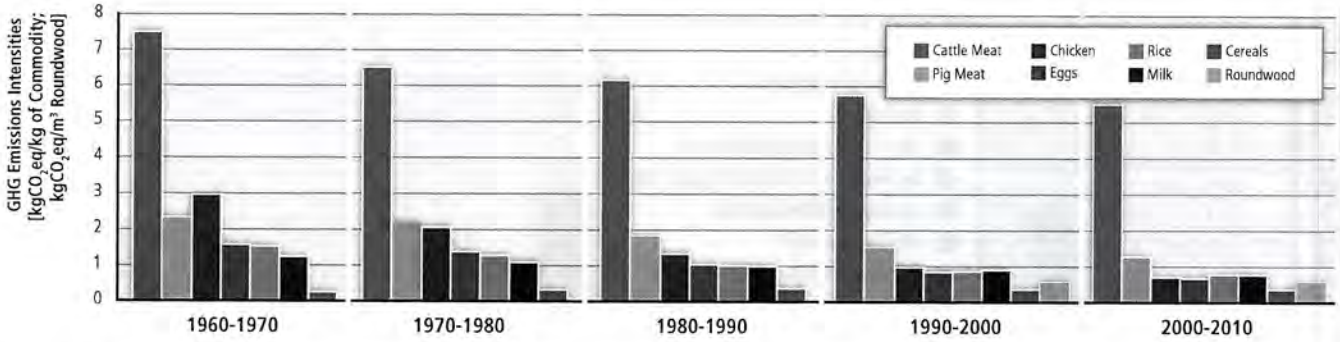


Figure 11.15 | GHG emissions intensities of selected major AFOLU commodities for decades 1960s–2000s, based on (Tubiello et al., 2012). i) Cattle meat, defined as GHG (enteric fermentation + manure management of cattle, dairy and non-dairy)/meat produced; ii) Pig meat, defined as GHG (enteric fermentation + 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).

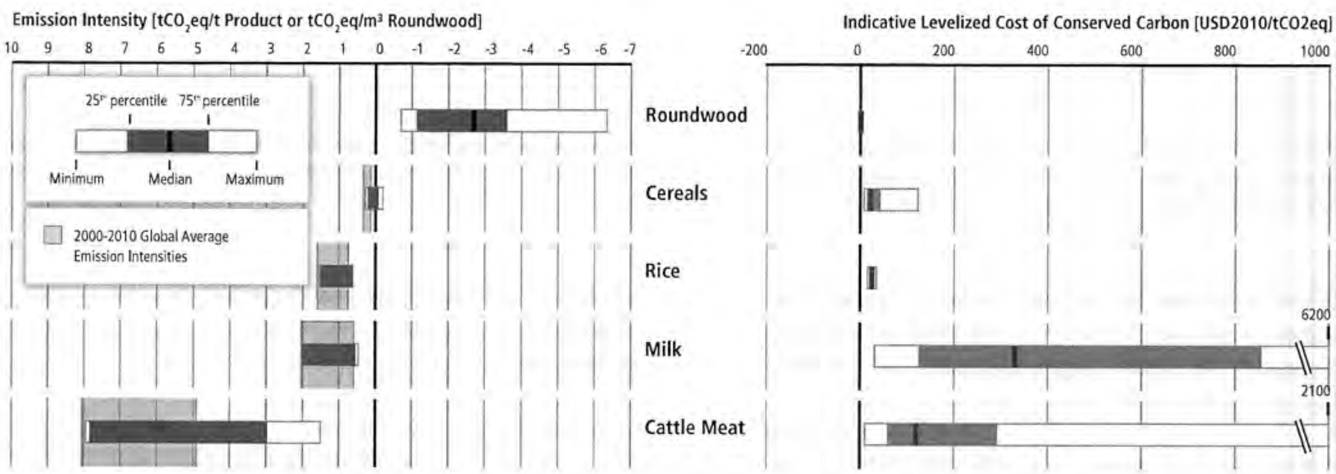


Figure 11.16 | Potential changes of emission intensities of major AFOLU commodities through implementation of commodity-specific mitigation measures (left panel) and related mitigation costs (right panel). Commodities and GHG emission sources are defined as in Figure 11.15, except for roundwood, expressed 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 (2006), and IPCC (2006); maximum and minimum values are defined by the highest and lowest values for 10 geographical regions. The right panel shows the mitigation costs (in USD/tCO$_2$eq) of commodity-specific mitigation measures (25th to 75th percentile range).

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

	up to 20 USD/tCO$_2$eq	up to 50 USD/tCO$_2$eq	up to 100 USD/tCO$_2$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	-	-	-	0.76–8.55

Notes:

[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.

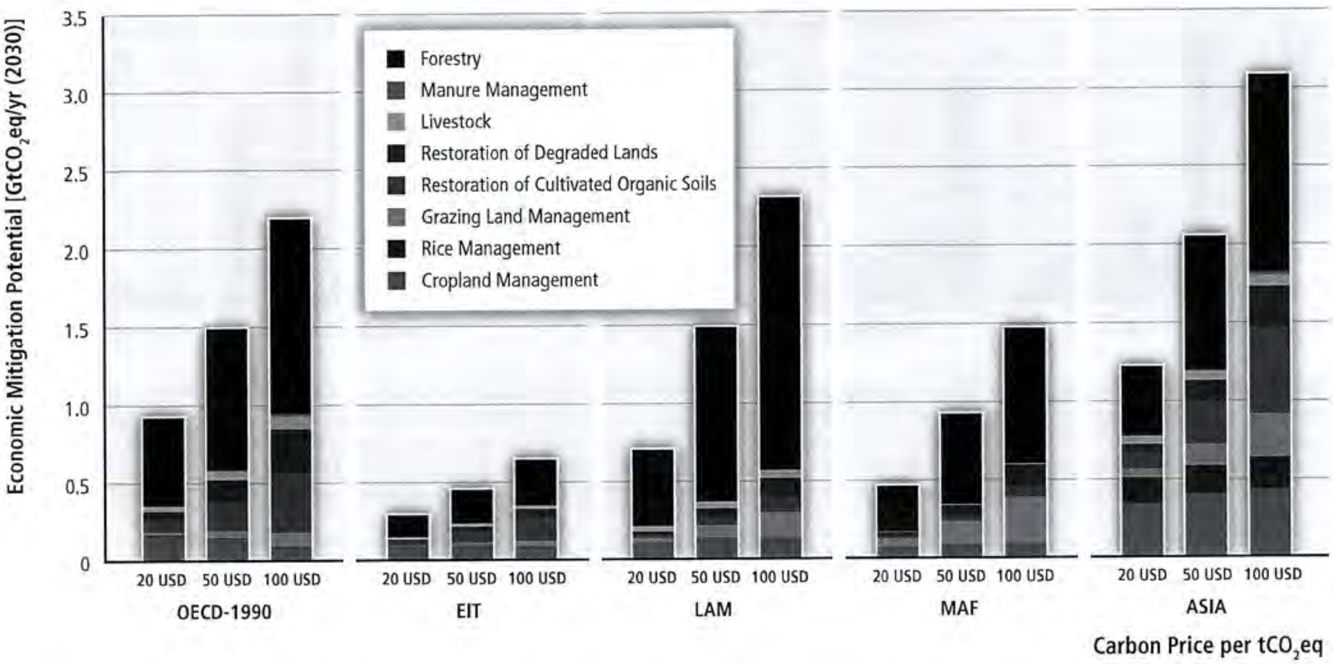


Figure 11.17 | Economic mitigation potentials in the AFOLU sector by region. Agriculture values are from Smith et al. (2007). Forestry values are from Nabuurs et al. (2007). For forestry, 20 USD values correspond to 'low', and 100 USD values correspond to 'high' values from Nabuurs et al. (2007). Values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

Differences between the most effective forestry 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 OECD-1990 and EIT. Forest management, followed by afforestation, dominate in OECD-1990, EIT, 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 Southeast Asian peats), and OECD-1990 (due to cultivated northern peatlands; Figure 11.18).

11.7 Co-benefits, risks, and spillovers

Implementation of AFOLU mitigation measures (Section 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.[8] 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, p. 201; Colfer, 2011; Davis et al., 2013; Albers and Robinson, 2013; Muys et al., 2014). 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 (Section 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 that allow an integrated assessment of multiple outcomes at landscape (Bryant et al., 2011), 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 the text presents the most relevant co-benefits and potential adverse side-effects from the recent literature.

Maximizing 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 actions may also contribute to a broader global sus-

[8] Co-benefits and adverse side-effects describe effects in non-monetary 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 Sections 2.4, 3.6.3, and 4.8.

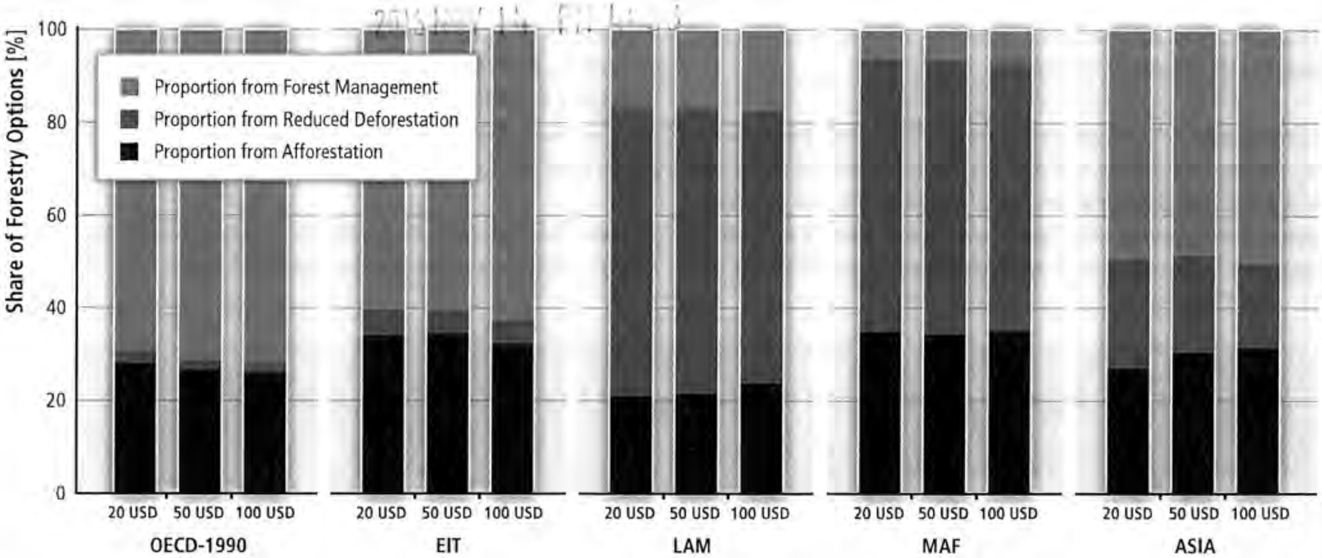


Figure 11.18 | Regional differences in forestry options, shown as a proportion of total potential available in forestry in each region. Global forestry activities (annual amount sequestered or emissions avoided above the baseline for forest management, reduced deforestation and afforestation), at carbon prices up to 100 USD/tCO_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).

tainability 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).

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 (Section 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; 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 covers a forest area, where agricultural land is promoted by another policy (Madlener et al., 2006; Halsnæs and Verhagen, 2007; Smith et al., 2007; Beach et al., 2009; Alig et al., 2010; Jackson and Baker, 2010; DeFries and Rosenzweig, 2010; Pettenella and Brotto, 2011; Section 11.10).

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 secure and nutritionally sufficient food production. 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; Graham-Rowe, 2011; Jeffery et al., 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* (Sections 3.3; 4.2; 4.7;

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 (Section 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; Section 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; Sections 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; Section 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; Sections 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 (Section 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 mitigation potential, provided that the resources are developed sustainably and that bioenergy systems are efficient (Chum et al., 2011; Section 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 (Beringer et al., 2011; Popp et al., 2011; Box 7.1). Agricultural and forestry residues can provide low-carbon and low-cost 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., 2011; Section 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.

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 and Franzel, 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 to 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; Section 11.8). The realization 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 Section 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; Section 11.10). Some authors propose that carbon payments can be complemented with biodiversity payments as an option for reducing tradeoffs 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 tradeoffs 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., ammonia (NH_3^+), nitrogen oxides (NO_x))[9], soil nitrate (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 (Section 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, for-

9 Please see Section 7.9.2 and WGII Section 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 Section 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, 12, 20)
	Sectoral policies	Promoting (↑) or contradicting (↓) the enforcement of sectoral (forest and/or agriculture) policies	National	Forestry (2, 5, 6, 9, 20); land-based agriculture (7, 11, 20)
	Cross-sectoral policies	Cross-sectoral coordination (↑) or clashes (↓) between forestry, agriculture, energy, and/or mining policies	Local to national	Forestry (7, 20); agriculture (7, 11, 20)
	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, 14, 20); agriculture (20, 32); integrated systems (20, 34)
	Benefit sharing mechanisms	Creation/use of benefits-sharing mechanisms (↑) from AFOLU mitigation measures	Local to national	Forestry (4, 5, 6, 8, 20)
Social	Food security	Increase (↑) or decrease (↓) on food availability and access	Local to national	Forestry (18, 19); agriculture (7, 15, 18, 19, 23, 28, 30); livestock (2, 3, 19, 35, 36); integrated systems (18, 19); biochar (17, 26)
	Local/traditional knowledge	Recognition (↑) or denial (↓) of indigenous and local knowledge in managing (forest/agricultural) land	Local/sub-national	Forestry (4, 5, 6, 8, 20); agriculture (20, 28); integrated systems (2); livestock (2, 3, 35); 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 (2, 31, 35, 37, 38)
	Cultural values	Respect and value cultural habitat and traditions (↑), reduce (↓), or increase (↑) existing conflicts or social discomfort (4, 5, 6, 20, 8)	Local to trans-boundary	Forestry (4, 5, 6, 9, 20)
	Human health	Impacts on health due to dietary changes, especially in societies with a high consumption of animal protein (↓)	Local to global	Changes in demand patterns (31, 36)
	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, 8, 9, 10, 20); agriculture (11, 23, 32)
Economic	Income	Increase (↑) or decrease (↓) in income. There are concerns regarding income distribution (↑)	Local	Forestry (6, 7, 8, 16, 20, 21, 22); agriculture (16, 19, 20, 23, 28); livestock (2, 3); integrated systems (7, 20); biochar (24); changes in demand patterns (2)
	Employment	Employment creation (↑) or reduction of employment (especially for small farmers or local communities) (↓)	Local	Forestry (8, 20); agriculture (20, 23); livestock (2, 3); integrated systems (7, 20)
	Financing mechanisms	Access (↑) or lack of access (↓) to new financing schemes	Local to global	Forestry (6, 8, 16, 20); agriculture (16, 20); livestock (2, 3)
	Economic activity	Diversification and increase in economic activity (↑) while concerns on equity (↑)	Local	Forestry (6, 7, 8, 20); land-based agriculture (16, 19, 20, 23, 28); 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, 20, 29, 30); livestock (2, 3, 29, 40)
	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, 19, 20, 27); on conservation and forest management (1, 19, 21, 27, 30); agriculture and integrated systems (15, 19, 20, 28, 30)
	Albedo	Positive impacts (↑) on albedo and evaporation and interactions with ozone	Local to global	See Section 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, 23, 30, 35); livestock (2, 3, 30)
	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, 19, 20, 27); land-based agriculture (30, 44); integrated systems (2, 30, 44)
	Soil	Soil conservation (↑) and improvement of soil quality and fertility (↑). Reduction of erosion. Positive or negative carbon mineralization priming effect (↑/↓)	Local	Forestry (44, 45); land-based agriculture (13, 19, 23, 28, 30); integrated systems biochar (39, 40)
	New products	Increase (↑) or decrease (↓) on fibre availability as well as non-timber/non-wood products output	Local to national	Forestry (18, 19, 41, 42); agriculture (7, 15, 18, 19, 23, 28, 30); integrated systems (18, 19)
	Ecosystem resilience	Increase (↑) or reduction (↓) of resilience, reduction of disaster risks (↓)	Local to trans-boundary	Forestry, integrated systems (11, 33; see Section 11.5)

⇒

	Issue	Potential co-benefit or adverse side-effect	Scale	AFOLU mitigation measure
Technology	Infrastructure	Increase (↑) or decrease (↓) in availability of and access to infrastructure. Competition for infrastructure for agriculture (↑), can increase social conflicts	Local	Agriculture (20, 46, 47)
	Technology innovation and transfer	Promote (↑) or delay (↓) technology development and transfer	Local to global	Forestry (7, 13, 25); agriculture (23); livestock (2, 3)
	Technology Acceptance	Can facilitate acceptance of sustainable technologies (↑)	Local to national	Forestry (7, 13, 25); livestock (2, 3, 35)

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]Madlener et al., 2006; [21]Strassburg et al., 2012; [22]Canadell and Raupach, 2008; [23]Pretty, 2008; [24]Galinato et al., 2011; [25]Macauley and Sedjo, 2011; [26]Jeffery et al., 2011; [27]Benayas et al., 2009; [28]Foley et al., 2011; [29]Haberl et al., 2013; [30]Smith et al., 2013; [31]Stehfest et al., 2009; [32]Chhatre et al., 2012; [33]Seppälä et al., 2009; [34]Murdiyarso et al., 2012; [35]de Boer et al., 2011; [36]McMichael et al., 2007; [37]Koknaroglu and Akunal, 2013; [38]Kehlbacher et al., 2012; [39]Zimmerman et al., 2011; [40]Luo et al., 2011; [41]Mirle, 2012; [42]Albers and Robinson, 2013; [43]Smith et al., 2013a; [44]Chatterjee and Lal, 2009; [45]Smith, 2008; [46]Ziv et al., 2012; [47]Beringer et al., 2011; [48]Douglas et al., 2009

est 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 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 stabilized (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, 2011). 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).

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 et al., 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 Section 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 (Section 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 that 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., 2008; 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 Spillovers

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 (Section 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 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 (Section 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 recognized these different circumstances among countries and have proposed corresponding approaches (e.g., a phased approach in the REDD+, Green Climate Fund; Section 11.10). Corresponding to the development framework presented in Section 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 successful use of the AFOLU mitigation potential (Section 11.10; 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; Chapter 16). Another element is the *accessibility to AFOLU financing* for farmers and forest stakeholders (Tubiello et al., 2009, p. 200; 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., Afforestation and Reforestation (A/R) Clean Development Mechanism (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 resources. 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; Section 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 (Palmer, 2011; Thompson et al., 2011; Markus, 2011; Rosendal and Andresen, 2011; Murdiyarso et al., 2012 Sections 11.4.5; 11.10; Chapters 14; 15).

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., 2011a). *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 (Section 11.10; 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 realization 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 (Section 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; Section 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 realized 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 focusing on existing local opportunities (Kandji et al., 2006), as proposed in Nationally Appropriate Mitigation Actions (NAMAs) (Section 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; Section 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+ (Section 11.10). Exploiting the existing *human skills* within a country is essential for realizing 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 (Section 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 Sections 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 characterize the pathways in terms of mitigation technologies and policy assumptions.

11.9.1 Characterization of transformation pathways

Uncertainty about reference AFOLU emissions is significant both historically (Section 11.2) and in projections (Section 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 mitigation 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 changes can represent a significant share of emissions reductions (Table 11.10). In these scenarios, 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. Note that bioenergy related mitigation is not captured in Table 11.10. 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 LUC, 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

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 ppm CO_2eq scenarios. The number of scenarios is indicated for each GHG and atmospheric concentration goal. Negative values represent increases in emissions (Kriegler et al., 2013). Bioenergy-related mitigation is not captured in the table.

			550 ppm			450 ppm		
			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
	Sum	min	−8.7	2.5	53.9	−3.7	5.6	69.7
	(n = 4/4)	max	295.2	587.7	903.5	301.4	596.9	940.3
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 %
	Sum	min	-11 %	0 %	1 %	-2 %	1 %	1 %
		max	70 %	47 %	19 %	69 %	46 %	17 %
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 %
	Sum	min	-4 %	1 %	7 %	-2 %	1 %	8 %
		max	97 %	100 %	73 %	99 %	101 %	76 %

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 cumulatively generated approximately 5 to 52 $GtCO_2eq$ and 113 to 749 $GtCO_2eq$ mitigation by 2050 and 2100, respectively. In total, land-related strategies contributed 20 to 60% of total cumulative abatement to 2030, 15 to 70% to 2050, and 15 to 40% to 2100.

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. For additional discussion of land's potential role in transformation pathways, especially regarding physical land-use and bioenergy, see sections 6.3.2.4 and 6.3.5.

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 (Section 6.3.5).

More recently, the literature has begun analyzing 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 (Sections 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., 2011; Reilly et al., 2012; Calvin et al., 2013a; 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 $GtCO_2$/yr 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. They project 60–100 $GtCO_2$ of leakage by 2025 with a carbon price of 15 USD_{2010}/tCO_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 (Section 11.7), will also be critical for successful land mitigation in forestry over time (Palmer, 2011; Gupta, 2012; Karsenty et al., 2014).

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, mitigation 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 (Section 11.10). For instance, Rose et al. (2013) estimate reduced mitigation potential, as well as over-crediting, for United States agriculture and forestry with voluntary mitigation supply incentives, e.g., mitigation decreased 25–55% at 15 USD_{2010}/tCO_2eq due to non-participant leakage and non-additional crediting.

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 (Section 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, OECD-1990, EIT). 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 (Section 6.3.5). Finally, there is relatively modest additional land conversion in the 450ppm, compared to the 550ppm, 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 (Section 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 Section 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 LUC, competition between different demands for land, and the feedbacks between LUC and other services provided by land is provided in Section 11.4, potential co-benefits and adverse side-effects are discussed in Section 11.7, and Section 6.6 compares potential co-benefits and adverse side-effects across sectors, while Section 11.8 presents the opportunities and barriers for promoting AFOLU mitigation activities in the future. Finally, Section 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 that also constitute sources of GHG as well as sinks (Golub et al., 2009). The land provides a multitude of ecosystem services, climate change mitigation being just one of many services that are vital to human well-being. 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 (Sections 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 hampered 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 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 agricul-

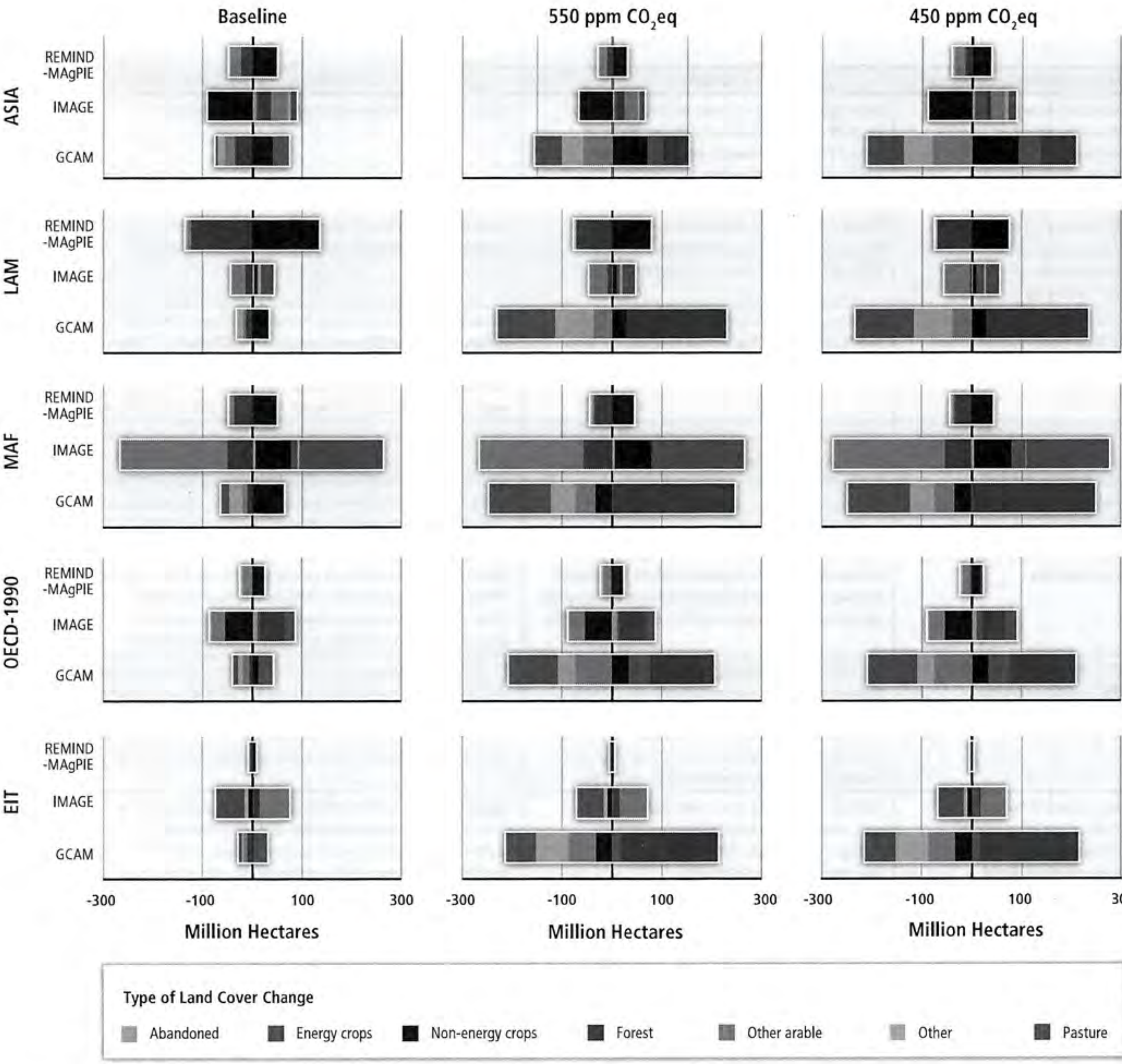


Figure 11.19 | Regional land cover change by 2030 from 2005 from three models for baseline (left) and idealized policy implementation 550 ppm CO_2eq (centre) and 450 ppm CO_2eq (right) scenarios. (Popp et al., 2013).

tural 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 efficiency, livestock improvement, better forestry management practices) could result in positive and synergistic impacts for adaptation and mitigation (Section 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.

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

Programme/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 (FLEG)
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 are 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.

11.10.1 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 Section 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: 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 (UNFCCC—CDM); therefore, 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., 2013).

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 fertilizer manufacturers and importers, and live animal exporters, although some exemptions apply (Government of New Zealand). 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 (California Environmental Protection Agency).

Voluntary carbon markets operate outside of the compliance markets. By enabling businesses, governments, non-governmental organizations (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 programmes, 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 133 million USD 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[10]. REDD+ was introduced in the agenda of the 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[11] (e.g., under the CDM 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 although the UNFCCC has already identified potential ways to pay for these[12], 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

[10] 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".

[11] 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".

[12] 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".

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; UN-REDD Programme—Support to Partner Countries).

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. 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 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) (Santilli et al., 2005; Forner et al., 2006; Nabuurs et al., 2007; Strassburg et al., 2008, 2009; Section 11.3.2). 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 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

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 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, 1.03 billion USD has been pledged, with 227 million USD approved for activities (Amazon Fund).

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 (UN-REDD Programme). The budget comprises 5.5 million USD_{2010} 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 1 billion USD_{2010}, exclusively on a payment-for-results basis over the next few years' (UN-REDD Programme). 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.

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 that currently have little deforestation (Putz and Redford, 2009).

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 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, 2013). 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 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 European Union (EU) agriculture was estimated with a GHG-weighted tax on animal food products of 79 USD_{2010}/tCO_2eq (60 EUR_{2010}/tCO_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 programme (launched in 2010; 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).

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 $MtCO_2eq$ (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 analyzed 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 tradeoff 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).

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).

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 (Sections 11.4.5; 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 (Sections 11.4.5; 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 minimum 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).

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" (UNFCCC). 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 LUC. 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 focusing 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, and 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 $GtCO_2eq/yr$, 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 $GtCO_2eq/yr$, 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 that 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 realization 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 $GtCO_2/yr$ of economically viable abatement in 2030 at carbon prices up to 100 USD/tCO_2eq. Global economic mitigation potentials in agriculture in 2030 are estimated to be up to 0.5–10.6 $GtCO_2eq/yr$. 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 $GtCO_2eq/yr$ in 2050, depending on which GHGs and diets are considered. Reducing food losses and waste in the supply chain from harvest to consumption can reduce GHG emissions by 0.6–6.0 $GtCO_2eq/yr$.

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 (particularly 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 realize 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

SRREN (IPCC, 2011) provided a comprehensive overview on bioenergy (Chum et al., 2011). However, a specific bioenergy Appendix in the context of the WGIII AR5 contribution is necessary because (1) 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 carbon dioxide capture and storage (BECCS); (2) there has been a large body of literature published since SRREN, which complements and updates the analysis presented in this last report; (3) bioenergy is important for many chapters (Chapters 6; 7; 8; 10; 11), 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 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 (1) the technology used; (2) the location, scales, and pace of implementation; (3) the land category used (forest, grassland, marginal lands, and crop lands); and (4) the business models and practices adopted—including how these integrate with or displace the existing land use.

As an update to the SRREN, this report presents (1) a more fine-grained assessment of the technical bioenergy potential reflecting diverse perspectives in the literature; (2) recent potential estimates on technological solutions such as BECCS; (3) an in-depth description of different lifecycle emission accounting methods and their results; (4) a small increase in uncertainty on the future economic bioenergy potential; (5) a comprehensive assessment of diverse livelihood and sustainability effects of bioenergy deployment, identifying the need for systematic aggregation.

11.13.2 Technical bioenergy potential

The technical bioenergy potential, also known as the technical primary biomass potential for bioenergy, is the amount of the theoretical bioenergy output obtainable by full implementation of demonstrated technologies or practices (IPCC, 2011). Unfortunately there is no standard methodology to estimate the technical bioenergy potential, which leads to diverging estimates. 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 bioenergy potentials in 2050 span a range of almost three orders of magnitude, from < 50 EJ/yr to > 1,000 EJ/yr (Smeets et al., 2007; Field et al., 2008; Haberl et al., 2010; Batidzirai et al., 2012). For example, Chum et al. reported global technical bioenergy potentials of 50–500 EJ/yr for the year 2050 (IPCC, 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., 2011; Berndes, 2012; Erb et al., 2012a). 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; Smeets et al., 2007; Smeets and Faaij, 2007; van Vuuren et al., 2009; Hakala et al., 2009; Dornburg et al., 2010; Haberl et al., 2010, 2011a; Gregg and Smith, 2010; Chum et al., 2011; GEA, 2012; Rogner et al., 2012). Most studies agree that the technical bioenergy potential in 2050 is at least approximately 100 EJ/yr with some modelling assumptions leading to estimates exceeding 500 EJ/yr (Smeets et al., 2007). As stated, different views about sustainability and socio-ecological constraints lead to very different estimates, with some studies reporting much lower figures.

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., 2007; Dornburg et al., 2010; Haberl et al., 2010; Gregg and Smith, 2010; Rogner et al., 2012) include residues from silvicultural thinning and logging; wood processing residues such as sawdust, bark, and black liquor; and dead wood from natural disturbances, such as storms and insect outbreaks (irregular source). The use of these resources is in general beneficial and any adverse side-effects can be mitigated by controlling residue removal rates considering biodiversity, climate, topography, and soil factors. There is a near-term tradeoff, 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., 2007; Hakala et al., 2009; Haberl et al., 2010, 2011a; Gregg and Smith, 2010; Chum et al., 2011; Rogner et al., 2012) 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

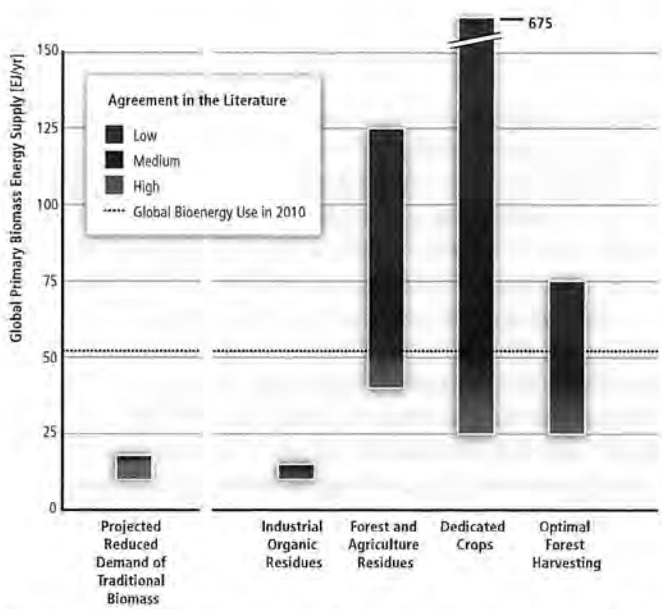


Figure 11.20 | Global Technical Bioenergy Potential by main resource category for the year 2050 | The figure shows the ranges in the estimates by major resource category of the global technical bioenergy potential. The color grading is intended to show qualitatively the degree of agreement in the estimates, from blue (large agreement in the literature) to purple (medium agreement) 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.

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. Optimization 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 depends 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-twentieth century as well as other factors such as recovery from past overuse, have led to strong forest C sinks in many OECD regions (Pan et al., 2011; Loudermilk et al., 2013; Nabuurs et al., 2013; Erb et al., 2013). However, the capacity of these sinks is being reduced as forests approach saturation (Smith, 2005; Körner, 2006; Guldea et al., 2008; Nabuurs et al., 2013; Sections 11.2.3, 11.3.2). Active forest management, including management for bioenergy, is therefore important for sustaining the strength of the forest carbon sink well into the future (Nabuurs et al., 2007, 2013; Canadell and Raupach, 2008; Ciais et al., 2008), 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 (Haberl et al., 2010; Gregg and Smith, 2010). Organic waste may be dispersed and also heterogeneous in quality but the health and environmental gains from collection and proper management through

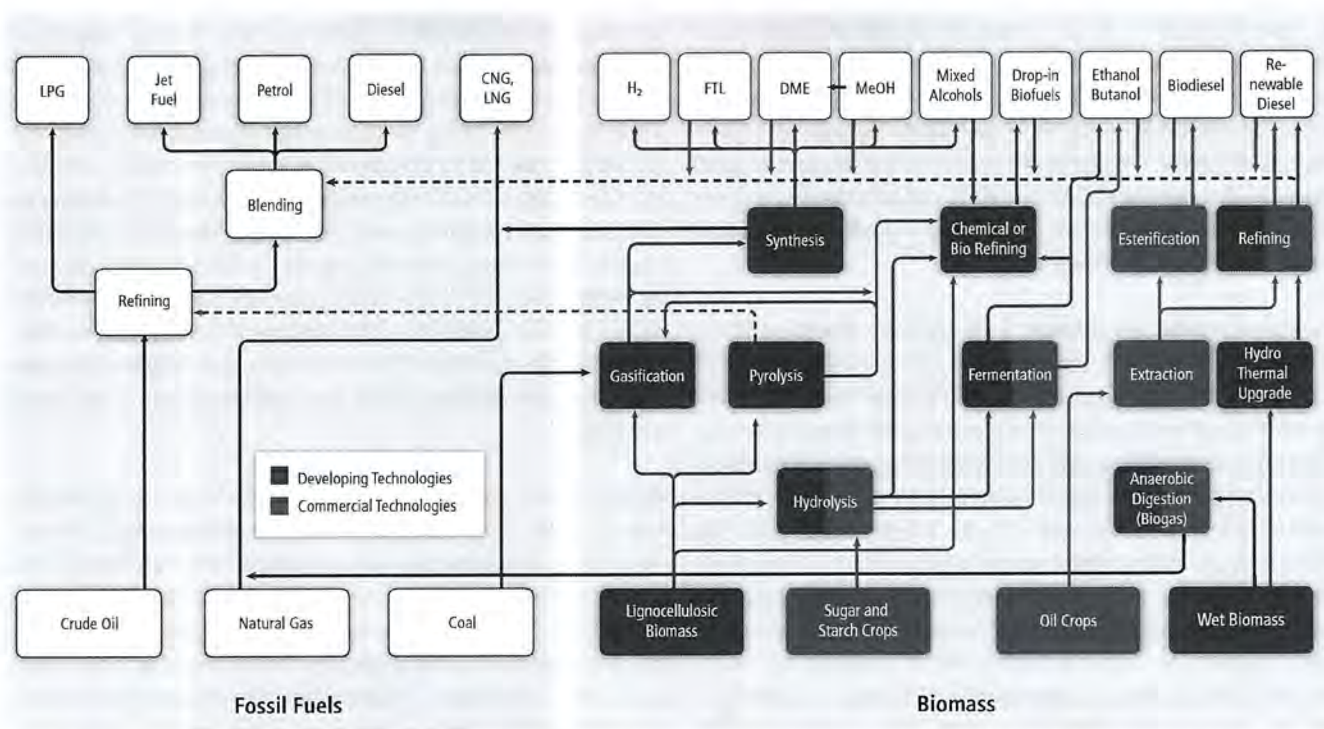


Figure 11.21 | Production pathways to liquid and gaseous fuels from biomass and, for comparison from fossil fuels (adapted from GEA, 2012; Turkenburg et al., 2012).

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., 2007; van Vuuren et al., 2009; Dornburg et al., 2010; Wicke et al., 2011b; 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 (1) the area deemed available for energy crops by (2) 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., 2012). 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 needs of ~2.6 billion people (Chum et al., 2011; IEA, 2012b). 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 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 (CDM-SSC WG, 2011) 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 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 Chum et al., 2011; 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, current commercial coal and biomass co-combustion technologies are the lowest-cost technologies for implementing renewable energy policies, enabled by the large-scale pelletized feedstocks trade (REN21, 2013; Junginger et al., 2014). Direct biopower use is also increasing commercially on a global scale (REN21, 2013, p. 21). In fact, using biomass for electricity and heat, for example, 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 co-production of electricity and liquid fuels from coal and biomass with higher efficiency than current commercial processes are in demonstration phase to reduce cost (Williams et al., 2011; GEA, 2012; Larson et al., 2012). 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 66 USD_{2010}/tCO_2 (50 EUR_{2010}/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 et al., 2013; Balan et al., 2013).

Many pathways and feedstocks (Figure 11.21) can lead to biofuels for aviation. The development of biofuel standards started and enabled testing of 50% biofuel in jet fuel for commercial domestic and trans-atlantic flights by consortia of governments, aviation industry, and associations (IEA, 2010; REN21, 2013). 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 Chapter 8). In specific cases, powering electric cars with electricity from biomass has higher land-use efficiency and lower global-warming potential (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)[13], though costs are likely to remain prohibitive for considerable time (van Vliet et al., 2011a; b; 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 (Williams et al., 2011; Larson et al., 2012) and pyrolysis (Brown et al., 2011; Bridgwater, 2012; Elliott, 2013; Meier et al., 2013). 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 (1) biomolecular engineering (Li et al., 2010; Favaro et al., 2012; Peralta-Yahya et al., 2012; Lee et al., 2013; Yoon et al., 2013); (2) 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; Saddler and Kumar, 2013; Kataeva et al., 2013); and (3) 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; Shen et al., 2013; Triantafyllidis et al., 2013; Yoon et al., 2013). Fundamental understanding of biofuel 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).

Bioenergy coupled with CCS (Spath and Mann, 2004; Liu et al., 2010) 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 (Sections 6.3.2 and 6.3.5) for two reasons: (1) The potential for negative emissions may allow shifting emissions in time; and (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 continu-

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

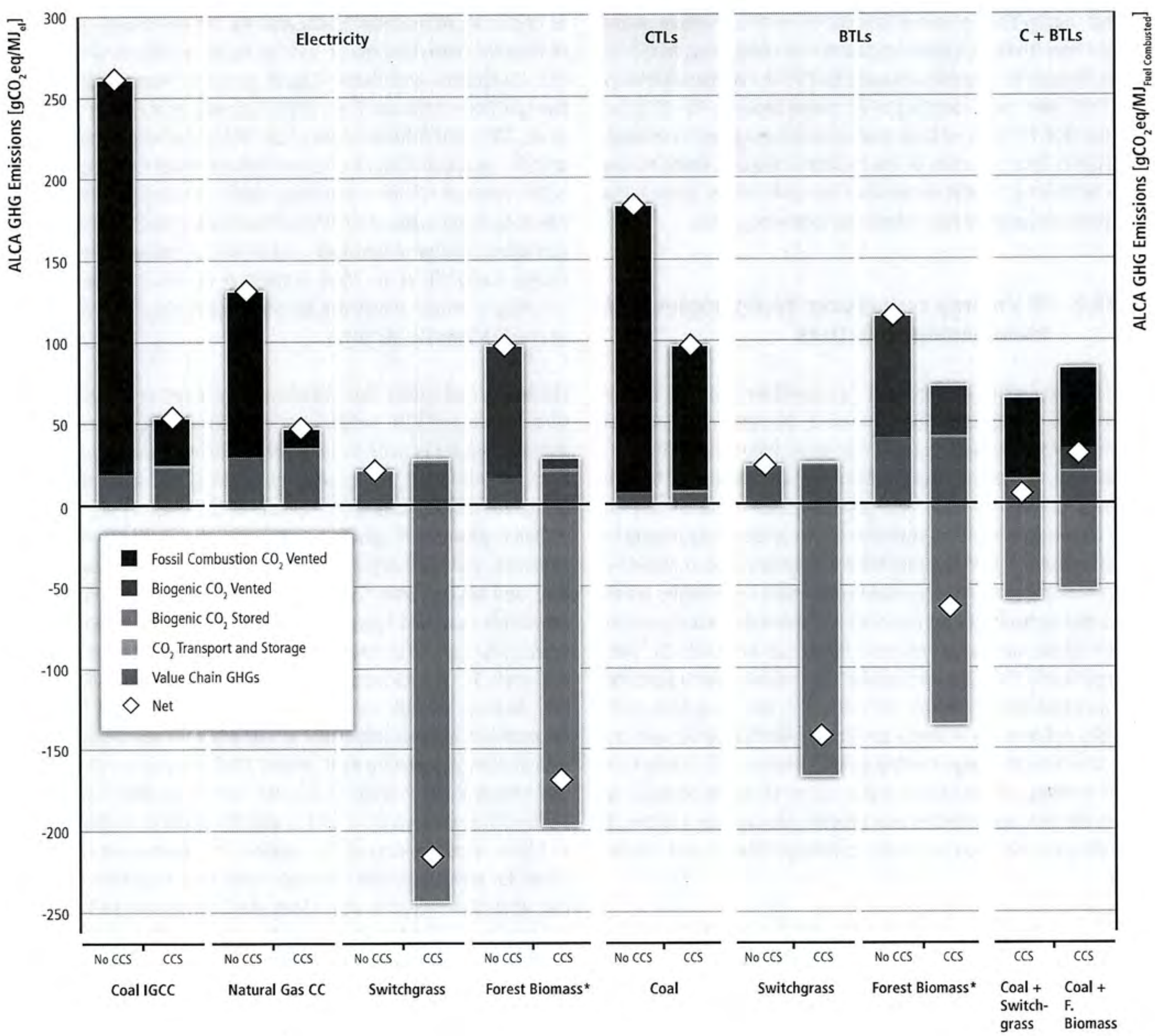


Figure 11.22 | Illustration of the sum of CO_2eq (GWP_{100})* emissions from the process chain of alternative transport and power generation technologies both with and without CCS. (*Differences in C-density between forest biomass and switchgrass are taken into account but not calorific values (balance-of-plant 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 significantly different outcomes, quantitatively (Section 11.13.4, Figures 11.23 and 11.24). Unit: gCO_2eq/MJ_{El} (left y-axis, electricity); gCO_2eq/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)

Notes:

* Global Warming Potential over 100 years. See Glossary and Section 1.2.5.

ally 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' that is projected to inject 1 $MtCO_2$/yr (Gollakota and McDonald, 2012; Senel and Chugunov, 2013). In the United States, 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 $MtCO_2$/yr (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, 2012c, 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 10 $GtCO_2$ storage per year for both Integrated Gasification Combined Cycle (IGCC)-CCS co-firing (IGCC with co-gasification of biomass), and Biomass Integrated Gasification Combined Cycle (BIGCC)-CCS dedicated, and around 6 $GtCO_2$ storage for biodiesel based on gasification and Fischer-Tropsch synthesis (FT diesel), and 2.7 $GtCO_2$ 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 $GtCO_2$ per year for 2030–2050 (McLaren, 2012). The economic potential, at a CO_2 price of around 70 USD/t is estimated to be around 3.3 $GtCO_2$, 3.5 $GtCO_2$, 3.1 $GtCO_2$ and 0.8 $GtCO_2$ 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 $GtCO_2$ per year at 70–250 USD/tCO_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 580–650 ppm 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 (Sections 7.5.5; 7.6.4; 7.9; 7.12).

Figure 11.22 illustrates some GHG effects associated with BECCS pathways. Tradeoffs 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 Figure 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); Lifecycle Analyses (LCAs) for these systems are complex (Pawelzik et al., 2013).

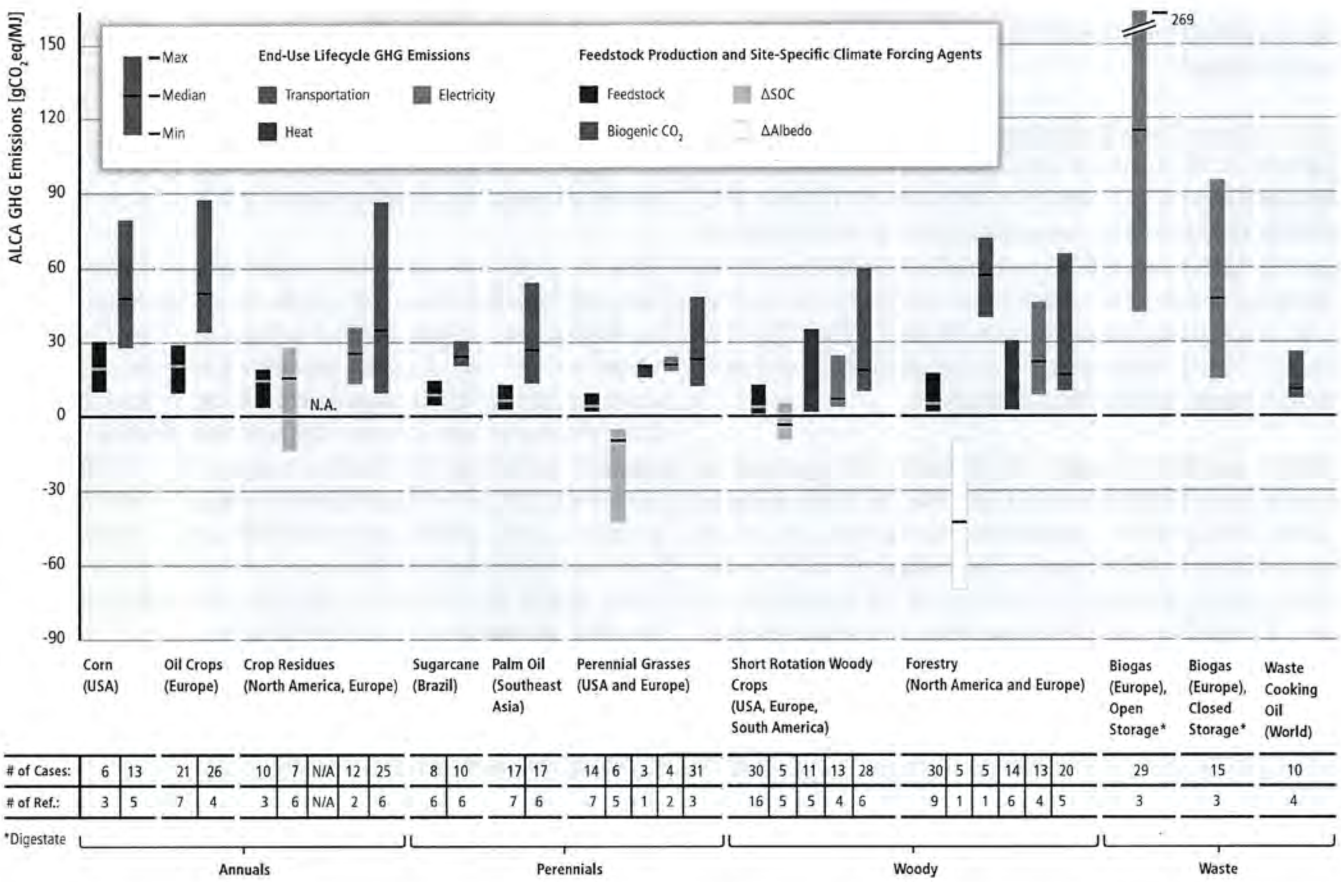


Figure 11.23 | Direct CO_2eq (GWP_{100}) emissions from the process chain or land-use disturbances of major bioenergy product systems, not including impacts from LUC (see Figure 11.24). The interpretation of values depends also on baseline assumption about the land carbon sink when appropriate and the 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, 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 gCO_2eq/MJ lower heating value (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 gCO_2eq/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: 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 waste cooking oil (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 lifecycle emissions.

[1]Gelfand et al. (2013); [2]Nemecek et al. (2012); [3]Hoefnagels et al. (2010); [4]Kaufman et al. (2010); [5]Cherubini et al. (2009); [6] Cherubini (2012); [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. (2012b); [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. (2013a); [24]Patrizi et al. (2013); [25]Souza et al. (2012a); [26]Seabra et al. (2011); [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. (2011); [36]Cherubini and Jungmeier (2010); [37]Don et al. (2012); [38]Pucker et al. (2012); [39]Monti et al. (2012); [40]Bai et al. (2010); [41]Bacenetti et al. (2012); [42]Budsberg et al. (2012); [43]González-García et al. (2012a); [44]González-García (2012b) ; [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. (2011); [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)

There are alternatives to land-based bioenergy. Microalgae, for example, offer a high-end technical potential. However, it might be compromised by water supply, if produced in arid land, or by impacts on ocean ecosystems. To make microalgae cost competitive, maximizing algal lipid content (and then maximizing growth rate) requires technological breakthroughs (Davis et al., 2011a; Sun et al., 2011; Jonker and Faaij, 2013). The 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. 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 research and development (R&D), near commercialization, or in early deployment stages in several countries (see Section 2.6.3 in 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., 2011b). 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. (2011).

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 et al., 2013).

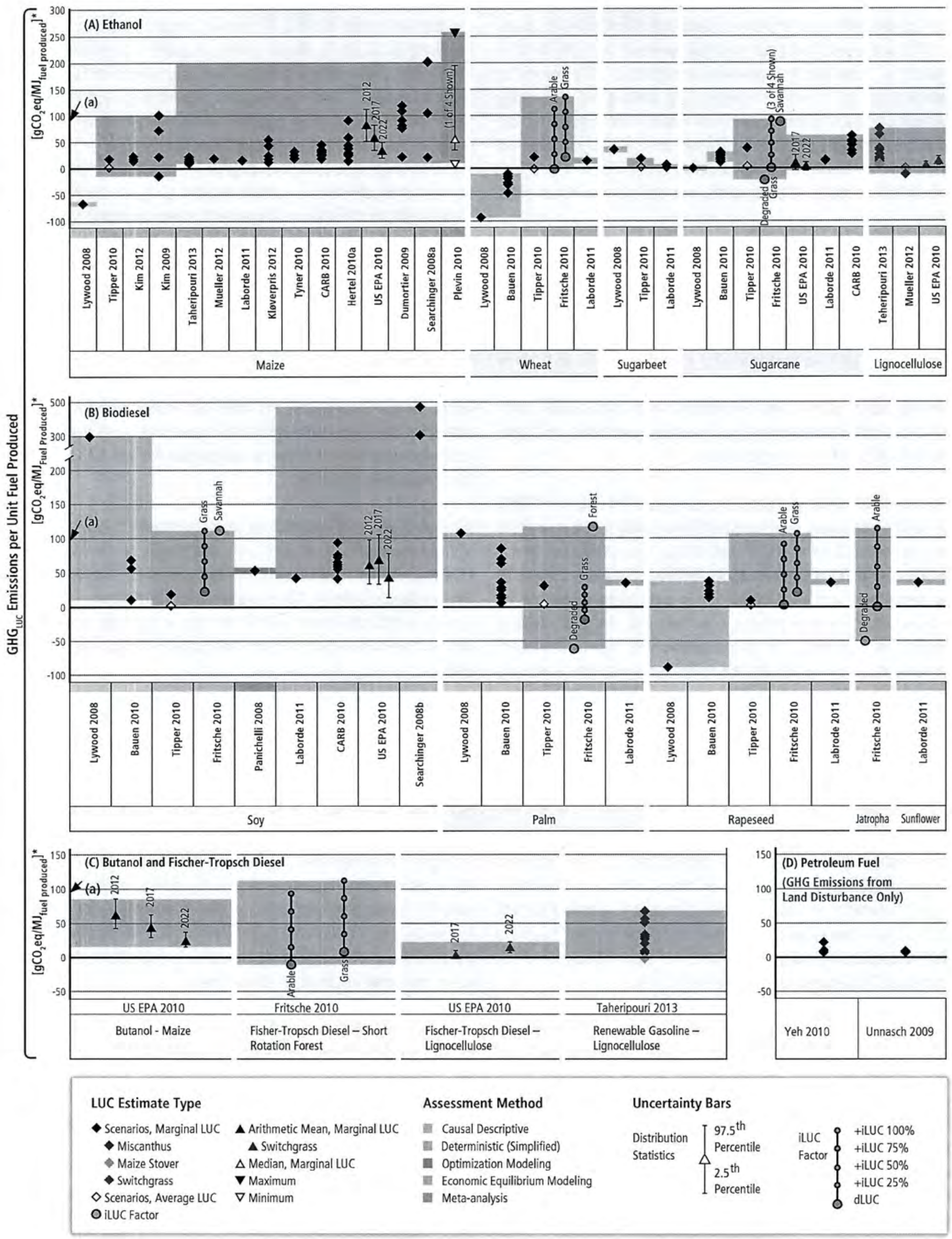
Substantial progress has also been achieved in the last four years in small-scale bioenergy applications in the areas of technology innovation, impact evaluation and monitoring, and in large-scale implementation programmes. For example, advanced combustion biomass cookstoves, which reduce fuel use by 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 providing clean combustion, help reduce the health risks associated with the disposal of organic wastes. There has also been a boost in cookstove dissemination efforts ranging from regional (multi-country) initiatives (Wang et al., 2013b) to national, and project-level interventions. In total, more than 200 large-scale cookstove projects are in place worldwide, with several million efficient cookstoves installed each year (Cordes, 2011). A Global Alliance for Clean Cookstoves 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 to be 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 (von Blottnitz and Curran, 2007; van der Voet et al., 2010); (ii) CO_2 and other GHG emissions from biomass or biofuel combustion (Cherubini et al., 2011); (iii) atmosphere-ecosystem exchanges of CO_2 following land disturbance (Berndes et al., 2013; Haberl, 2013); (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


GHG$_{LUC}$ Emissions per Unit Fuel Produced
[gCO$_2$eq/MJ$_{fuel\ produced}$]*
(A) Ethanol
(a)
Maize
Lywood 2008
Tipper 2010
Kim 2012
Kim 2009
Taheripouri 2013
Mueller 2012
Laborde 2011
Kløverpris 2012
Tyner 2010
CARB 2010
Hertel 2010a
US EPA 2010
Dumortier 2009
Searchinger 2008a
Plevin 2010
(1 of 4 Shown)
2012
2017
2022
Wheat
Lywood 2008
Bauen 2010
Tipper 2010
Fritsche 2010
Laborde 2011
Arable
Grass
Sugarbeet
Lywood 2008
Tipper 2010
Laborde 2011
Sugarcane
Lywood 2008
Bauen 2010
Tipper 2010
Fritsche 2010
US EPA 2010
Laborde 2011
CARB 2010
(3 of 4 Shown)
Savannah
Degraded
Grass
Lignocellulose
Teheripouri 2013
Mueller 2012
US EPA 2010
(B) Biodiesel
Soy
Lywood 2008
Bauen 2010
Tipper 2010
Fritsche 2010
Panichelli 2008
Laborde 2011
CARB 2010
US EPA 2010
Searchinger 2008b
Grass
Savannah
Palm
Lywood 2008
Bauen 2010
Tipper 2010
Fritsche 2010
Labrode 2011
Forest
Grass
Degraded
Rapeseed
Lywood 2008
Bauen 2010
Tipper 2010
Fritsche 2010
Labrode 2011
Arable
Grass
Jatropha
Fritsche 2010
Arable
Degraded
Sunflower
Laborde 2011
(C) Butanol and Fischer-Tropsch Diesel
US EPA 2010
Butanol - Maize
Fritsche 2010
Fisher-Tropsch Diesel – Short Rotation Forest
Arable
Grass
US EPA 2010
Fischer-Tropsch Diesel – Lignocellulose
Taheripouri 2013
Renewable Gasoline – Lignocellulose
(D) Petroleum Fuel
(GHG Emissions from Land Disturbance Only)
Yeh 2010
Unnasch 2009
LUC Estimate Type
Scenarios, Marginal LUC
Miscanthus
Maize Stover
Switchgrass
Scenarios, Average LUC
iLUC Factor
Arithmetic Mean, Marginal LUC
Switchgrass
Median, Marginal LUC
Maximum
Minimum
Assessment Method
Causal Descriptive
Deterministic (Simplified)
Optimization Modeling
Economic Equilibrium Modeling
Meta-analysis
Uncertainty Bars
Distribution Statistics
97.5th Percentile
2.5th Percentile
iLUC Factor
+iLUC 100%
+iLUC 75%
+iLUC 50%
+iLUC 25%
dLUC

⇒

Figure 11.24 | Estimates of GHG_{LUC} emissions—GHG emissions from biofuel production-induced LUC (as $gCO_2eq/MJ_{fuel\ produced}$) 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 lifecycle 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 modelling. 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.

(e.g., from changes in surface albedo, heat and water fluxes, surface roughness, etc.; (Bonan, 2008; West et al., 2010a; 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, 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 pathways, 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 models, policy directives, etc.) under the assumption that the CO_2 emitted from biomass combustion is climate neutral[14] because the carbon that was previously sequestered from the atmosphere will be re-sequestered if the bioenergy system is managed sustainably (Chum et al., 2011; Creutzig et al., 2012a; b). The shortcomings of this assumption have been extensively discussed in environmental impact studies and emission accounting mechanisms (Searchinger et al., 2009; Searchinger, 2010; Cherubini et al., 2011; 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 (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'; Marland and Schlamadinger, 1995; Fargione et al., 2008; Bernier and Paré, 2013), understanding the climate implication as it relates to policy targets (e.g., limiting warming to 2 °C) requires models and/or metrics that also include temperature effects and climate consequence (Smith et al., 2012c; 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 (House et al., 2002; Cherubini et al., 2013; 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 (West et al., 2010b; Anderson-Teixeira et al., 2012), 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 a few decades up to several centuries; (Repo et al., 2011; Mitchell et al., 2012; Pingoud et al., 2012; Bernier and Paré, 2013; Guest et al., 2013; Helin et al., 2013; Holtsmark, 2013).

In some cases, cooling contributions from changes in surface albedo can mitigate or offset these effects (Arora and Montenegro, 2011; O'Halloran et al., 2012; Anderson-Teixeira et al., 2012; Hallgren et al., 2013).

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, 2013).

Two specific contributions to the climate forcing of bioenergy, not addressed in detail in SRREN include N_2O and biogeophysical factors.

Nitrous oxide 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 lifecycle 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., 2012; 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., 2012c). 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., 2011) 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).

[14] 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.

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, 2001; Marland et al., 2003; Betts et al., 2007; Bonan, 2008; Jackson et al., 2008; 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 (Claussen et al., 2001; Bathiany et al., 2010), with radiative forcing effects possibly stronger than those of the co-occuring C-cycle changes (Randerson et al., 2006; Lohila et al., 2010; Bright et al., 2011; Cherubini et al., 2012; O'Halloran et al., 2012). 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 (Bala et al., 2007; Swann et al., 2010, 2011). Biogeophysical climate impacts from changes in land use are site-specific and show variations in magnitude across different geographic regions and biomes (Bonan, 2008; Anderson, 2010; Pielke Sr. et al., 2011; Anderson-Teixeira et al., 2012). Biogeophysical impacts should be considered in climate impact assessments and in the design of land-use policies to adequately assess the net impacts of land-use mitigation options (Jackson et al., 2008; Betts, 2011; 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 lifecycle 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 (Figures 2.10 and 2.11 in SRREN; Chum et al., 2011) 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 the 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 lifecycle 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 (LUC emissions can still be relevant, see Figure 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.

Direct and indirect land-use change: Direct land-use change 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; Kløverpris et al., 2008; Delucchi, 2010; Hertel et al., 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, p. 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, 2013). 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., 2010; Sterner and Fritsche, 2011; Sochacki et al., 2012). In particular, Miscanthus improves soil organic carbon reducing overall GHG emissions (Brandão et al., 2011); degraded USA 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., 2013). 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 (Schlamadinger and Marland, 1996; Marland and Schlamadinger, 1997; Fargione et al., 2008; McKechnie et al., 2011; Hudiburg et al., 2011). Calculations of specific GHG emissions from long-term rotation forests need to account for the foregone CO_2-accumulation (Searchinger, 2010; Holtsmark, 2012; Pingoud et al., 2012; Haberl et al., 2012).

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 are positive, in particular over several harvesting cycles making use of the faster carbon sequestration rates of younger forests (Daigneault et al., 2012; Ximenes et al., 2012; Lamers and Junginger, 2013; Latta et al., 2013). 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: (1) carbon sequestration in forest biomass; (2) soil and wood products; and (3) 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., 2012), 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, thus 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 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., 2010; Khanna et al., 2011; Gawel and Ludwig, 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 (Figure 11.23), albeit with high variability and uncertainty in results (Hertel et al., 2010; Taheripour et al., 2011; Dumortier et al., 2011; Havlík et al., 2011; Chen et al., 2012; Timilsina et al., 2012; Warner et al., 2014). 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). Indirect land-use change 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., 2011a; for wheat, Berndes et al., 2011). Proper management of livestock 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., 2013) 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 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., 2011b; 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, 2013; van der Hilst et al., 2012a; Rose et al., 2013). 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.

11.13.5 Aggregate future potential deployment in integrated models

In SRREN scenarios (IPCC, 2011), bioenergy is projected to contribute 80–190 EJ/yr to global primary energy supply by 2050 for 50 % of the scenarios in the two mitigation levels modelled. The min to max ranges were 20–265 EJ/yr for the less stringent scenarios and 25–300 EJ for the tight mitigation scenarios (< 440 ppm). Many of these scenarios coupled bioenergy with CCS. The Global Energy Assessment (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 ten-fold 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 International Energy Agency (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, 2010). 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 WGIII AR5 transformation pathway studies suggest that modern bioenergy could play a significant role within the energy system (Section 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) and current observations, suggest to focus on the lower half of the ranges reported above (Field et al., 2008; Campbell et al., 2008; Johnston et al., 2009a, 2011; Haberl et al., 2013b).

BECCS features prominently in many mitigation scenarios. BECCS is deployed in greater quantities and earlier in time the more stringent the climate policy (Section 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 con-

Box 11.9 | 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 sugar cane 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 iLUC, 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 increment 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; Edser, 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' incomes (Garg et al., 2011; Arndt et al., 2012), improvement in energy security at the local level (von Maltitz and Setzkorn, 2013; Muys et al., 2014), and reducing soil erosion (Garg et al., 2011).

firmations, affordability and public acceptance (see Section 11.13.3 for details). BECCS may also affect the cost-effective emissions trajectory (Richels et al., In Review; Rose et al., 2013).

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 (Rose et al., 2013; Popp et al., 2013). 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., 2013). 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 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., 2011; Reilly et al., 2012). Many integrated models are investigating idealized policy implementation pathways, assuming global prices on GHG (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 LUC alone are possible (Melillo et al., 2009; Wise et al., 2009; Creutzig et al., 2012a; 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 Section 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 (Section 11.4.5). The effects on livelihoods have not yet been systematically evaluated in integrated models (Davis et al., 2013; Creutzig et al., 2012b; Creutzig et al., 2013; Muys et al., 2014), even if human geography studies have shown that bioenergy deployment can have strong distributional impacts (Davis et al., 2013; Muys et al., 2014). 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 use (Gohin, 2008) or export markets (Wicke et al., 2009; Arndt et al., 2011).

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., 2011c; Immerzeel et al., 2013). Examples include adjustments in agriculture practices where farmers, for instance, change their manure treatment to produce biogas, reduce methane and N losses. Changes in management practice may swing the net GHG balance of options and also have clear sustainable development implications (Davis et al., 2013).

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 (2012b) estimates that 2.6 billion people world-

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), 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, 73, 74). 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 cookstoves) can have positive impacts on health, especially 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, 73)	+	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 (75, 76)	+	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 tradeoffs 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 phaseout of lead addititives and methyl tertiary-butyl ether (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

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[1]Alves Finco and Doppler (2010); [2]Amigun et al. (2011); [3]Arndt et al. (2012); [4]Arndt et al. (2011); [5]Arndt et al.(2012); [6]Awudu and Zhang (2012); [7]Beringer et al. (2011); [8]Borzoni (2012); [9]Bringezu et al. (2012); [10]Cacciatore et al. (2012); [11]Cançado et al. (2006); [12]Danielsen et al. (2009); [13]Diaz-Chavez (2011); [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. (2008); [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. (2008); [42]Jerneck and Olsson (2013); [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.(2013b); [50]Muys et al. (2014); [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]van Eijck et al. (2014); [63]Martínez et al. (2013); [64]van der Hilst et al. (2010); [65]van der Hilst et al. (2012); [66]van der Hilst and Faaij (2012); [67]van der Hilst et al. (2012b); [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. (2013); [74]Wiskerke et al. (2010); [75]De Wit et al. (2011); [76]de Wit et al. (2013)

wide depend on traditional biomass for cooking, 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 (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: They reduce the pressure on forests and biodiversity; they reduce exposure to smoke-related health hazards; they reduce drudgery for women in collection fuelwood; and they save money if fuel needs to be purchased (Martin et al., 2011). Benefits from the dissemination of improved cookstoves outweigh their costs by sevenfold, 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 GHG 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, 2013). 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 well-being, 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 for 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; Hall et al., 2009; German et al., 2011; Thompson et al., 2011b; IPCC, 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. (2012) in the European context, and Van Dam et al. (2009a; b) 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., 2007; Gopalakrishnan et al., 2009, 2011a; b, 2012; Dimitriou et al., 2009, 2011; Dornburg et al., 2010; 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 et al., 2011). Certification approaches can be useful, but they should be accompanied by effective territorial policy frameworks (Hunsberger et al., 2012).

11.13.7 Tradeoffs 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., 2012); (2) the extent to which bioenergy production can be integrated with food or fiber production, which ideally results in synergies (Garg et al., 2011; Sochacki et al., 2013) 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., 2012); and (4) the volume of dedicated energy crops and their yields (Haberl et al., 2010; Batidzirai et al., 2012; Smith et al., 2012d). 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., 2009a; Lal, 2010; Beringer et al., 2011; Pacca and Moreira, 2011; Smith et al., 2012a; 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., 2013).

However, across models, there are very different potential landscape transformation visions in all regions (Sections 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 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 United States) would reduce global technical bioenergy potentials by 17 % until 2050 (van Vuuren et al., 2009). A model comparison study with five global economic models shows that the aggregate food price effect of large-scale lignocellulosic 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 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 into 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 LUC 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., 2012).

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Contents

Executive Summary

The shift from rural to more urban societies is a global trend with significant consequences for greenhouse gas (GHG) emissions and climate change mitigation. Across multiple dimensions, the scale and speed of urbanization is unprecedented: more than half of the world population live in urban areas and each week the global urban population increases by 1.3 million. Today there are nearly 1000 urban agglomerations with populations of 500,000 or greater; by 2050, the global urban population is expected to increase by between 2.5 to 3 billion, corresponding to 64 % to 69 % of the world population (*robust evidence, high agreement*). Expansion of urban areas is on average twice as fast as urban population growth, and the expected increase in urban land cover during the first three decades of the 21st century will be greater than the cumulative urban expansion in all of human history (*medium evidence, high agreement*). Urban areas generate around 80 % of global Gross Domestic Product (GDP) (*medium evidence, medium agreement*). Urbanization is associated with increases in income, and higher urban incomes are correlated with higher consumption of energy use and GHG emissions (*medium evidence, high agreement*) [Sections 12.1, 12.2, 12.3].

Current and future urbanization trends are significantly different from the past (*robust evidence, high agreement*). Urbanization is taking place at lower levels of economic development and the majority of future urban population growth will take place in small- to medium-sized urban areas in developing countries. Expansion of urban areas is on average twice as fast as urban population growth, and the expected increase in urban land cover during the first three decades of the 21st century will be greater than the cumulative urban expansion in all of human history (*robust evidence, high agreement*). [12.1, 12.2]

Urban areas account for between 71 % and 76 % of CO_2 emissions from global final energy use and between 67–76 % of global energy use (*medium evidence, medium agreement*). There are very few studies that have examined the contribution of all urban areas to global GHG emissions. The fraction of global CO_2 emissions from urban areas depends on the spatial and functional boundary definitions of urban and the choice of emissions accounting method. Estimates for urban energy related CO_2 emissions range from 71 % for 2006 to between 53 % and 87 % (central estimate, 76 %) of CO_2 emissions from global final energy use (*medium evidence, medium agreement*). There is only one attempt in the literature that examines the total GHG (CO_2, CH_4, N_2O and SF_6) contribution of urban areas globally, estimated at between 37 % and 49 % of global GHG emissions for the year 2000. Using Scope1 accounting, urban share of global CO_2 emissions is about 44 % (*limited evidence, medium agreement*). [12.2]

No single factor explains variations in per-capita emissions across cities, and there are significant differences in per capita GHG emissions between cities within a single country (*robust evidence, high agreement*). Urban GHG emissions are influenced by a variety of physical, economic and social factors, development levels, and urbanization histories specific to each city. Key influences on urban GHG emissions include income, population dynamics, urban form, locational factors, economic structure, and market failures. There is a prevalence for cities in Annex I countries to have lower per capita final energy use and GHG emissions than national averages, and for per capita final energy use and GHG emissions of cities in non-Annex I countries tend to be higher than national averages (*robust evidence, high agreement*) [12.3].

The anticipated growth in urban population will require a massive build-up of urban infrastructure, which is a key driver of emissions across multiple sectors (*limited evidence, high agreement*). If the global population increases to 9.3 billion by 2050 and developing countries expand their built environment and infrastructure to current global average levels using available technology of today, the production of infrastructure materials alone would generate approximately 470 Gt of CO_2 emissions. Currently, average per capita CO_2 emissions embodied in the infrastructure of industrialized countries is five times larger than those in developing countries. The continued expansion of fossil fuel-based infrastructure would produce cumulative emissions of 2,986–7,402 $GtCO_2$ during the remainder of the 21st century (*limited evidence, high agreement*). [12.2, 12.3]

The existing infrastructure stock of the average Annex I resident is three times that of the world average and about five times higher than that of the average non-Annex I resident (*medium evidence, medium agreement*). The long life of infrastructure and the built environment, make them particularly prone to lock-in of energy and emissions pathways, lifestyles and consumption patterns that are difficult to change. The committed emissions from energy and transportation infrastructures are especially high, with respective ranges of 127–336 and 63–132 Gt, respectively (*medium evidence, medium agreement*). [12.3, 12.4]

Infrastructure and urban form are strongly linked, especially among transportation infrastructure provision, travel demand and vehicle kilometres travelled (*robust evidence, high agreement*). In developing countries in particular, the growth of transport infrastructure and ensuing urban forms will play important roles in affecting long-run emissions trajectories. Urban form and structure significantly affect direct (operational) and indirect (embodied) GHG emissions, and are strongly linked to the throughput of materials and energy in a city, the wastes that it generates, and system efficiencies of a city. (*robust evidence, high agreement*) [12.4, 12.5]

Key urban form drivers of energy and GHG emissions are density, land use mix, connectivity, and accessibility (*medium evidence, high agreement*). These factors are interrelated and interdependent. Pursuing one of them in isolation is insufficient for lower emissions. Connectivity and accessibility are tightly related: highly connected places are accessible. While individual measures of urban form

have relatively small effects on vehicle miles travelled, they become more effective when combined. There is consistent evidence that co-locating higher residential densities with higher employment densities, coupled with significant public transit improvements, higher land use mixes, and other supportive demand management measures can lead to greater emissions savings in the long run. Highly accessible communities are typically characterized by low daily commuting distances and travel times, enabled by multiple modes of transportation (*robust evidence, high agreement*). [12.5]

Urban mitigation options vary across urbanization trajectories and are expected to be most effective when policy instruments are bundled (*robust evidence, high agreement*). For rapidly developing cities, options include shaping their urbanization and infrastructure development towards more sustainable and low carbon pathways. In mature or established cities, options are constrained by existing urban forms and infrastructure and the potential for refurbishing existing systems and infrastructures. Key mitigation strategies include co-locating high residential with high employment densities, achieving high land use mixes, increasing accessibility and investing in public transit and other supportive demand management measures. Bundling these strategies can reduce emissions in the short term and generate even higher emissions savings in the long term (*robust evidence, high agreement*). [12.5]

Successful implementation of mitigation strategies at local scales requires that there be in place the institutional capacity and political will to align the right policy instruments to specific spatial planning strategies (*robust evidence, high agreement*). Integrated land-use and transportation planning provides the opportunity to envision and articulate future settlement patterns, backed by zoning ordinances, subdivision regulations, and capital improvements programmes to implement the vision. While smaller scale spatial planning may not have the energy conservation or emissions reduction benefits of larger scale ones, development tends to occur parcel by parcel and urbanized areas are ultimately the products of thousands of individual site-level development and design decisions (*robust evidence, high agreement*). [12.5, 12.6]

The largest opportunities for future urban GHG emissions reduction are in rapidly urbanizing areas where urban form and infrastructure are not locked-in, but where there are often limited governance, technical, financial, and institutional capacities (*robust evidence, high agreement*). The bulk of future infrastructure and urban growth is expected in small- to medium-size cities in developing countries, where these capacities are often limited or weak (*robust evidence, high agreement*). [12.4, 12.5, 12.6, 12.7]

Thousands of cities are undertaking climate action plans, but their aggregate impact on urban emissions is uncertain (*robust evidence, high agreement*). Local governments and institutions possess unique opportunities to engage in urban mitigation activities and local mitigation efforts have expanded rapidly. However, there has been little systematic assessment regarding the overall extent to which cities are implementing mitigation policies and emission reduction targets are being achieved, or emissions reduced. Climate action plans include a range of measures across sectors, largely focused on energy efficiency rather than broader land-use planning strategies and cross-sectoral measures to reduce sprawl and promote transit-oriented development. The majority of these targets have been developed for Annex I countries and reflect neither their mitigation potential nor implementation. Few targets have been established for non-Annex I country cities, and it is in these places where reliable city-level GHG emissions inventory may not exist (*robust evidence, high agreement*). [12.6, 12.7, 12.9]

The feasibility of spatial planning instruments for climate change mitigation is highly dependent on a city's financial and governance capability (*robust evidence, high agreement*). Drivers of urban GHG emissions are interrelated and can be addressed by a number of regulatory, management, and market-based instruments. Many of these instruments are applicable to cities in both developed and developing countries, but the degree to which they can be implemented varies. In addition, each instrument varies in its potential to generate public revenues or require government expenditures, and the administrative scale at which it can be applied. A bundling of instruments and a high level of coordination across institutions can increase the likelihood of achieving emissions reductions and avoiding unintended outcomes (*robust evidence, high agreement*). [12.6, 12.7]

For designing and implementing climate policies effectively, institutional arrangements, governance mechanisms, and financial resources should be aligned with the goals of reducing urban GHG emissions (*robust evidence, high agreement*). These goals will reflect the specific challenges facing individual cities and local governments. The following have been identified as key factors: (1) institutional arrangements that facilitate the integration of mitigation with other high-priority urban agendas; (2) a multilevel governance context that empowers cities to promote urban transformations; (3) spatial planning competencies and political will to support integrated land-use and transportation planning; and (4) sufficient financial flows and incentives to adequately support mitigation strategies (*robust evidence, high agreement*). [12.6, 12.7]

Successful implementation of urban climate change mitigation strategies can provide co-benefits (*robust evidence, high agreement*). Urban areas throughout the world continue to struggle with challenges, including ensuring access to energy, limiting air and water pollution, and maintaining employment opportunities and competitiveness. Action on urban-scale mitigation often depends on the ability to relate climate change mitigation efforts to local co-benefits. The co-benefits of local climate change mitigation can include public savings, air quality and associated health benefits, and productivity increases in urban centres, providing additional motivation for undertaking mitigation activities (*robust evidence, high agreement*). [12.5, 12.6, 12.7, 12.8]

This assessment highlights a number of key knowledge gaps. First, there is lack of consistent and comparable emissions data at local scales, making it particularly challenging to assess the urban share of global GHG emissions as well as develop urbanization typologies and their emissions pathways. Second, there is little scientific understanding of the magnitude of the emissions reduction from altering urban form, and the emissions savings from integrated infrastructure and land use planning. Third, there is a lack of consistency and thus comparability on local emissions accounting methods, making cross-city comparisons of emissions or climate action plans difficult. Fourth, there are few evaluations of urban climate action plans and their effectiveness. Fifth, there is lack of scientific understanding of how cities can prioritize mitigation strategies, local actions, investments, and policy responses that are locally relevant. Sixth, there are large uncertainties about future urbanization trajectories, although urban form and infrastructure will play large roles in determining emissions pathways. [12.9]

12.1 Introduction

Urbanization is a global phenomenon that is transforming human settlements. The shift from primarily rural to more urban societies is evident through the transformation of places, populations, economies, and the built environment. In each of these dimensions, urbanization is unprecedented for its speed and scale: massive urbanization is a megatrend of the 21st century. With disorienting speed, villages and towns are being absorbed by, or coalescing into, larger urban conurbations and agglomerations. This rapid transformation is occurring throughout the world, and in many places it is accelerating.

Today, more than half of the global population is urban, compared to only 13 % in 1900 (UN DESA, 2012). There are nearly 1,000 urban agglomerations with populations of 500,000 or more, three-quarters of which are in developing countries (UN DESA, 2012). By 2050, the global urban population is expected to increase between 2.5 to 3 billion, corresponding to 64 % to 69 % of the world population (Grubler et al., 2007; IIASA, 2009; UN DESA, 2012). Put differently, each week the urban population is increasing by approximately 1.3 million.

Future trends in the levels, patterns, and regional variation of urbanization will be significantly different from those of the past. Most of the urban population growth will take place in small- to medium-sized urban areas. Nearly all of the future population growth will be absorbed by urban areas in developing countries (IIASA, 2009; UN DESA, 2012). In many developing countries, infrastructure and urban growth will be greatest, but technical capacities are limited, and governance, financial, and economic institutional capacities are weak (Bräutigam and Knack, 2004; Rodrik et al., 2004). The kinds of towns, cities, and urban agglomerations that ultimately emerge over the coming decades will have a critical impact on energy use and carbon emissions.

The Fourth Assessment Report (AR4) of the Intergovernmental Panel on Climate Change (IPCC) did not have a chapter on human settlements or urban areas. Urban areas were addressed through the lens of individual sector chapters. Since the publication of AR4, there has been a growing recognition of the significant contribution of urban areas to GHG emissions, their potential role in mitigating them, and a multi-fold increase in the corresponding scientific literature. This chapter provides an assessment of this literature and the key mitigation options that are available at the local level. The majority of this literature has focused on urban areas and cities in developed countries. With the exception of China, there are few studies on the mitigation potential or GHG emissions of urban areas in developing countries. This assessment reflects these geographic limitations in the published literature.

Urbanization is a process that involves simultaneous transitions and transformations across multiple dimensions, including demographic, economic, and physical changes in the landscape. Each of these dimensions presents different indicators and definitions of urbanization. The chapter begins with a brief discussion of the multiple dimensions and definitions of urbanization, including implications for GHG emissions accounting, and then continues with an assessment of historical, current, and future trends across different dimensions of urbanization in the context of GHG emissions (12.2). It then discusses GHG accounting approaches and challenges specific to urban areas and human settlements.

In Section 12.3, the chapter assesses the drivers of urban GHG emissions in a systemic fashion, and examines the impacts of drivers on individuals sectors as well as the interaction and interdependence of drivers. In this section, the relative magnitude of each driver's impact on urban GHG emissions is discussed both qualitatively and quantitatively, and provides the context for a more detailed assessment of how urban form and infrastructure affect urban GHG emissions (12.4). Here, the section discusses the individual urban form drivers such as density, connectivity, and land use mix, as well as their interactions with each other. Section 12.4 also examines the links between infrastructure and urban form, as well as their combined and interacting effects on GHG emissions.

Section 12.5 identifies spatial planning strategies and policy instruments that can affect multiple drivers, and Section 12.6 examines the institutional, governance, and financial requirements to implement such policies. Of particular importance with regard to mitigation potential at the urban or local scale is a discussion of the geographic and administrative scales for which policies are implemented, overlapping, and/or in conflict. The chapter then identifies the scale and range of mitigation actions currently planned and/or implemented by local governments, and assesses the evidence of successful implementation of the plans, as well as barriers to further implementation (12.7). Next, the chapter discusses major co-benefits and adverse side-effects of mitigation at the local scale, including opportunities for sustainable development (12.8). The chapter concludes with a discussion of the major gaps in knowledge with respect to mitigation of climate change in urban areas (12.9).

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12.2 Human settlements and GHG emissions

This section assesses past, current, and future trends in human settlements in the context of GHG emissions. It aims to provide a multidimensional perspective on the scale of the urbanization process. This section includes a discussion of the development trends of urban areas, including population size, land use, and density. Section 12.2.1 outlines historic urbanization dynamics in multiple dimensions as drivers of GHG emissions. Section 12.2.2 focuses on current GHG emissions. Finally, Section 12.2.3 assesses future scenarios of urbanization in order to frame the GHG emissions challenges to come.

12.2.1 The role of cities and urban areas in energy use and GHG emissions

Worldwide, 3.3 billion people live in rural areas, the majority of whom, about 92 %, live in rural areas in developing countries (UN DESA, 2012). In general, rural populations have lower per capita energy consumption compared with urban populations in developing countries (IEA, 2008). Globally, 32 % of the rural population lack access to electricity and other modern energy sources, compared to only 5.3 % of the urban population (IEA, 2010). Hence, energy use and GHG emissions from human settlements is mainly from urban areas rather than rural areas, and the role of cities and urban areas in global climate change has become increasingly important over time.

Table 12.1 | Arithmetic growth of human settlement classes for five periods between 1950–2050. Number of human settlements by size class at four points in time.

Population	Average annual growth [%]					Number of cities			
	1950–1970	1970–1990	1990–2010	1950–2010	2010–2050	1950	1970	1990	2010
10,000,000 and more	2.60	6.72	4.11	4.46	2.13	2	2	10	23
5,000,000—10,000,000	7.55	1.34	2.53	3.77	1.22	4	15	19	38
1,000,000—5,000,000	3.27	3.17	2.70	3.05	1.36	69	128	237	388
100,000–1,000,000	2.86	2.48	1.87	2.40	0.70	Not Available			
Less than 100,000	2.54	2.37	1.71	2.21	1.95				
Rural	1.38	1.23	0.61	1.07	-0.50				

Source: (UN DESA, 2012).

Box 12.1 | What is urban? The system boundary problem

Any empirical analysis of urban and rural areas, as well as human settlements, requires clear delineation of physical boundaries. However, it is not a trivial or unambiguous task to determine where a city, an urban area, or human settlement physically begins and ends. In the literature, there are a number of methods to establish the boundaries of a city or urban area (Elliot, 1987; Buisseret, 1998; Churchill, 2004). Three common types of boundaries include:

1. **Administrative boundaries**, which refer to the territorial or political boundaries of a city (Hartshorne, 1933; Aguilar and Ward, 2003).
2. **Functional boundaries**, which are delineated according to connections or interactions between areas, such as economic activity, per capita income, or commuting zone (Brown and Holmes, 1971; Douglass, 2000; Hidle et al., 2009).
3. **Morphological boundaries**, which are based on the form or structure of land use, land cover, or the built environment. This is the dominant approach when satellite images are used to delineate urban areas (Benediktsson et al., 2003; Rashed et al., 2003).

What approach is chosen will often depend on the particular research question under consideration. The choice of the physical boundaries can have a substantial influence on the results of the analysis. For example, the Global Energy Assessment (GEA) (GEA, 2012) estimates global urban energy consumption between 180–250 EJ/yr depending on the particular choice of the physical delineation between rural and urban areas. Similarly, depending on the choice of different administrative, morphological, and functional boundaries, between 37 % and 86 % in buildings and industry, and 37 % to 77 % of mobile diesel and gasoline consumption can be attributed in urban areas (Parshall et al., 2010). Thus any empirical evidence presented in this chapter is dependent on the particular boundary choice made in the respective analysis.

Urbanization involves change across multiple dimensions and accordingly is defined differently by different disciplines. Demographers define urbanization as a demographic transition that involves a population becoming urbanized through the increase in the urban proportion of the total population (Montgomery, 2008; Dorélien et al., 2013). Geographers and planners describe urbanization as a land change process that includes the expansion of the urban land cover and growth in built-up areas and infrastructure (Berry et al., 1970; Blanco et al., 2011; Seto et al., 2011). Economists characterize urbanization as a structural shift from primary economic activities such as agriculture and forestry to manufacturing and services (Davis and Henderson, 2003; Henderson, 2003). Sociologists, political scientists, and other social scientists describe urbanization as cultural change, including change in social interactions and the growing complexity of political, social, and economic institutions (Weber, 1966; Berry, 1973). The next sections describe urbanization trends across the first three of these four dimensions and point to the increasing and unprecedented speed and scale of urbanization.

12.2.1.1 Urban population dynamics

In the absence of any other independent data source with global coverage, assessments of historic urban and rural population are commonly based on statistics provided by the United Nations Department for Economic and Social Affairs (UN DESA). The *World Urbanization Prospects* is published every two years by UN DESA and provides projections of key demographic and urbanization indicators for all countries in the world. Even within this dataset, there is no single definition of urban or rural areas that is uniformly applied across the data. Rather, each country develops its own definition of urban, often based on a combination of population size or density, and other criteria such as the percentage of population not employed in agriculture; the availability of electricity, piped water, or other infrastructure; and characteristics of the built environment such as dwellings and built structures (UN DESA, 2012). The large variation in criteria gives rise to significant differences in national definitions. However, the underlying variations in the data do not seriously affect an assessment of urbanization dynamics as long as the national definitions are sufficiently consistent over time (GEA, 2012; UN DESA, 2012). Irrespective of definition, the underlying assumption in all the definitions is that urban areas provide a higher standard of living than rural areas (UN DESA, 2013). A comprehensive assessment of urban and rural population dynamics is provided in the Global Energy Assessment (2012). Here, only key developments are briefly summarized.

For most of human history, the world population mostly lived in rural areas and in small urban settlements, and growth in global urban population occurred slowly. In 1800, when the world population was around one billion, only 3 % of the total population lived in urban areas and only one city—Beijing—had had a population greater than one million (Davis, 1955; Chandler, 1987; Satterthwaite, 2007). Over the next one hundred years, the global share of urban population

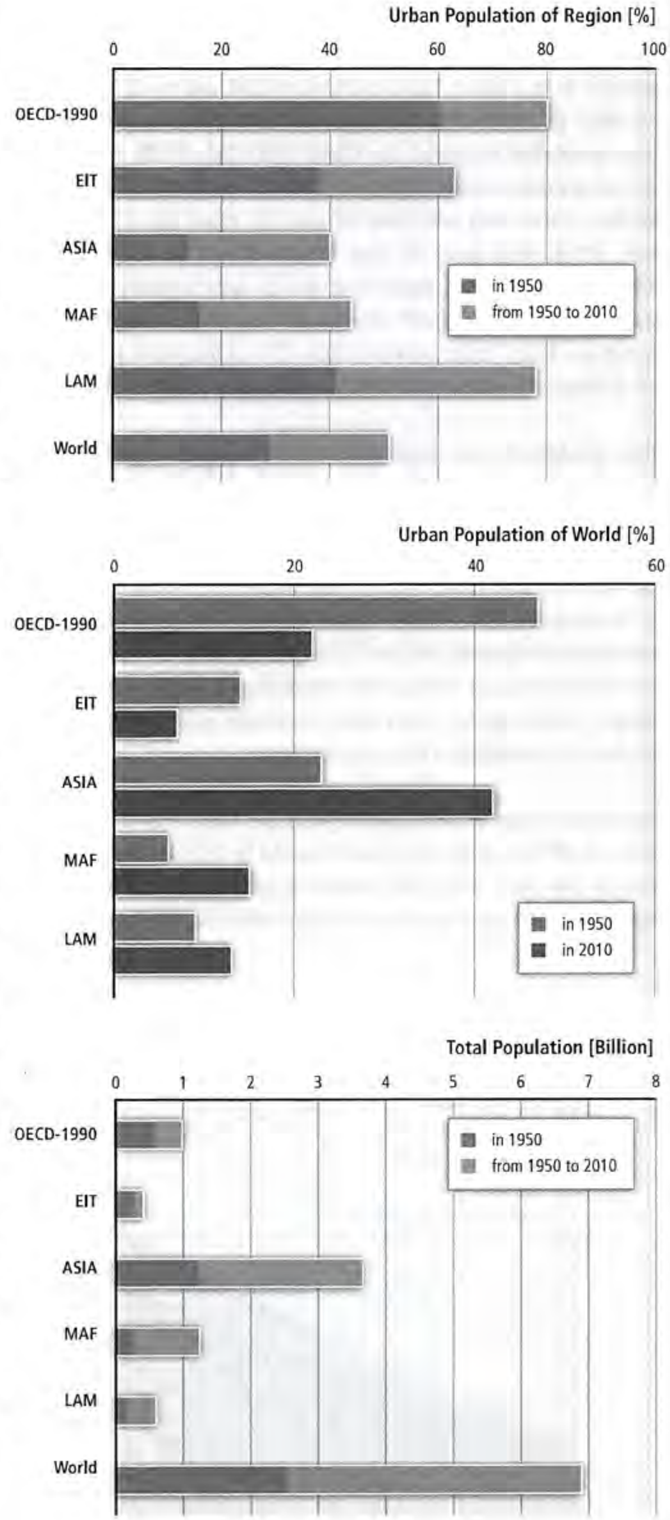


Figure 12.1 | Urban population as percentage of regional and world populations and in absolute numbers for RC5 regions (see Annex II.2), 1950–2010 Source: UN DESA (2012).

increased to 13 % in 1900. The second half of the 20th century experienced rapid urbanization. The proportion of world urban population increased from 13 % in 1900, to 29 % in 1950, and to 52 % in 2011 (UN DESA, 2012). In 1960, the world reached a milestone when global urban population surpassed one billion (UN DESA, 2012). Although it took all previous human history to 1960 to reach one billion urban dwellers, it took only additional 26 years to reach two billion (Seto et al., 2010). Since then, the time interval to add an additional one billion urban dwellers is decreasing, and by approximately 2030, the world urban population will increase by one billion every 13 years (Seto et al., 2010). Today, approximately 52 % of the global population, or 3.6 billion, are estimated to live in urban areas (UN DESA, 2012).

While urbanization has been occurring in all major regions of the world (Table 12.1) since 1950, there is great variability in urban transitions across regions and settlement types. This variability is shaped by multiple factors, including history (Melosi, 2000), migration patterns (Harris and Todaro, 1970; Keyfitz, 1980; Chen et al., 1998), technological development (Tarr, 1984), culture (Wirth, 1938; Inglehart, 1997), governance institutions (National Research Council, 2003), as well as environmental factors such as the availability of energy (Jones, 2004; Dredge, 2008). Together, these factors partially account for the large variations in urbanization levels across regions.

Urbanization rates in developed regions are high, between 73 % in Europe to 89 % in North America, compared to 45 % in Asia and 40 % in Africa (UN DESA, 2012).The majority of urbanization in the future is expected to take place primarily in Africa and Asia, and will occur at lower levels of economic development than the urban transitions that occurred in Europe and North America. While its urbanization rate is still lower than that of Europe and the Americas, the urban population in Asia increased by 2.3 billion between 1950 and 2010 (Figure 12.1).

Overall, urbanization has led to the growth of cities of all sizes (Figure 12.2). Although mega-cities (those with populations of 10 million or greater) receive a lot of attention in the literature, urban population growth has been dominated by cities of smaller sizes. About one-third of the growth in urban population between 1950 and 2010 (1.16 billion) occurred in settlements with populations fewer than 100 thousand. Currently, approximately 10 % of the 3.6 billion urban dwellers live in mega-cities of 10 million or greater (UN DESA, 2012). Within regions and countries, there are large variations in development levels, urbanization processes, and urban transitions. While the dominant global urbanization trend is growth, some regions are experiencing significant urban population declines. Urban shrinkage is not a new phenomenon, and most cities undergo cycles of growth and decline, which is argued to correspond to waves of economic growth and recession (Kondratieff and Stolper, 1935). There are few systematic analyses on the scale and prevalence of shrinking cities (UN-Habitat, 2008). A recent assessment by the United Nations (UN) (UN DESA, 2012) indicates that about 11 % of 3,552 cities with populations of 100,000 or more in 2005 experienced total population declines of 10.4 million between 1990 and 2005. These 'shrinking cities' are distributed globally but concentrated mainly in Eastern Europe (Bontje, 2005; Bernt, 2009) and the rust belt in the United States (Martinez-Fernandez et al., 2012), where de-urbanization is strongly tied with de-industrialization.

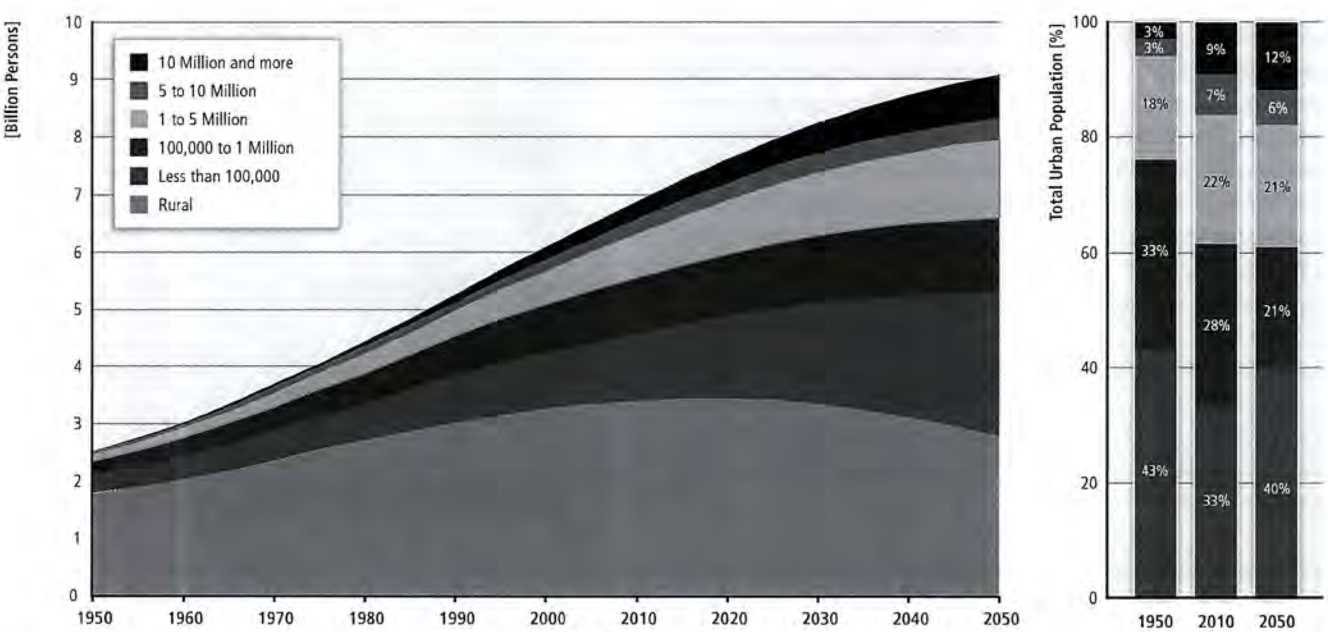


Figure 12.2 | Population by settlement size using historical (1950–2010) and projected data to 2050. Source: UN DESA (2010), Grubler et al. (2012). Note: rounded population percentages displayed across size classes sum do not sum to 100 % for year 2010 due to rounding. Urbanization results in not only in growth in urban population, but also changes in household structures and dynamics. As societies industrialize and urbanize, there is often a decline in household size, as traditional complex households become more simple and less extended (Bongaarts, 2001; Jiang and O'Neill, 2007; O'Neill et al., 2010). This trend has been observed in Europe and North America, where household size has declined from between four to six in the mid 1800s to between two and three today (Bongaarts, 2001).

12.2.1.2 Urban land use

Another key dimension of urbanization is the increase in built-up area and urban land cover. Worldwide, urban land cover occupies a small fraction of global land surface, with estimates ranging between 0.28 to 3.5 million km^2, or between 0.2 % to 2.7 % of ice free terrestrial land (Schneider et al., 2009). Although the urban share of global land cover is negligible, urban land use at the local scale shows trends of declining densities and outward expansion.

Analyses of 120 global cities show significant variation in densities across world regions, but the dominant trend is one of declining built-up and population densities across all income levels and city sizes (Figure 12.3) (Angel et al., 2010). For this sample of cities, built-up area densities have declined significantly between 1990 and 2000, at an average annual rate of 2.0±0.4 % (Angel et al., 2010). On average, urban population densities are four times higher in low-income countries (11,850 persons/km^2 in 2000) than in high-income countries (2,855 persons/km^2 in 2000). Urban areas in Asia experienced the largest decline in population densities during the 1990s. Urban population densities in East Asia and Southeast Asia declined 4.9 % and 4.2 %, respectively, between 1990 and 2000 (World Bank, 2005). These urban population densities are still higher than those in Europe, North America, and Australia, where densities are on average 2,835 persons/km^2. As the urban transition continues in Asia and Africa, it is expected that their urban population densities will continue to decline. Although urban population densities are decreasing, the amount of built-up area per person is increasing (Seto et al., 2010; Angel et al., 2011). A meta-analysis of 326 studies using satellite data shows a minimum global increase in urban land area of 58,000km^2 between 1970 and 2000, or roughly 9 % of the 2000 urban extent (Seto et al., 2011). At current rates of declining densities among developing country cities, a doubling of the urban population over the next 30 years will require a tripling of built-up areas (Angel et al., 2010). For a discussion on drivers of declining densities, see Box 12.4.

12.2.1.3 Urban economies and GDP

Urban areas are engines of economic activities and growth. Further, the transition from a largely agrarian and rural society to an industrial and consumption-based society is largely coincident with a country's level of industrialization and economic development (Tisdale, 1942; Jones, 2004), and reflects changes in the relative share of GDP by both sector and the proportion of the labour force employed in these sectors (Satterthwaite, 2007; World Bank, 2009). The concentration and scale of people, activities, and resources in urban areas fosters economic growth (Henderson et al., 1995; Fujita and Thisse, 1996; Duranton and Puga, 2004; Puga, 2010), innovation (Feldman and Audretsch, 1999; Bettencourt et al., 2007; Arbesman et al., 2009), and an increase of economic and resource use efficiencies (Kahn, 2009; Glaeser and Kahn, 2010). The agglomeration economies made possible by the concentration of individuals and firms make cities ideal settings for innovation, job, and wealth creation (Rosenthal and Strange, 2004; Carlino et al., 2007; Knudsen et al., 2008; Puga, 2010).

A precise estimate of the contribution of all urban areas to global GDP is not available. However, a downscaling of global GDP during the Global Energy Assessment (Grubler et al., 2007; GEA, 2012) showed that urban areas contribute about 80 % of global GDP. Other studies show that urban economies generate more than 90 % of global gross value (Gutman, 2007; United Nations, 2011). In OECD countries, more than 80 % of the patents filed are in cities (OECD, 2006a). Not many cities report city-level GDP but recent attempts have been made by the Metropolitan Policy Program of the Brookings Institute, PriceWaterhouseCoopers (PWC), and the McKinsey Global Institute to provide such estimates. The PWC report shows that key 27 key global cities[1] accounted for 8 % of world GDP for 2012 but only 2.5 % of the global population (PwC and Partnership for New York City, 2012).

In a compilation by UN-Habitat, big cities are shown to have disproportionately high share of national GDP compared to their population (UN-Habitat, 2012). The importance of big cities is further underscored in a recent report that shows that 600 cities generated 60 % of global GDP in 2007 (McKinsey Global Institute, 2011). This same report shows that the largest 380 cities in developed countries account for half of the global GDP. More than 20 % of global GDP comes from 190 North American cities alone (McKinsey Global Institute, 2011). In contrast, the 220 largest cities in developing countries contribute to only 10 % global of GDP, while 23 global megacities generated 14 % of global GDP in 2007. The prevalence of economic concentration in big cities highlights their importance but does not undermine the role of small and medium size cities. Although top-down and bottom-up estimates suggest a large urban contribution to global GDP, challenges remain in estimating the size of this, given large uncertainties in the downscaled GDP, incomplete urban coverage, sample bias, methodological ambiguities, and limitations of the city-based estimations in the existing studies.

12.2.2 GHG emission estimates from human settlements

Most of the literature on human settlements and climate change is rather recent.[2] Since AR4, there has been a considerable growth in scientific evidence on energy consumption and GHG emissions from human settlements. However, there are very few studies that have examined the contribution of all urban areas to global GHG emissions.

[1] Paris, Hong Kong, Sydney, San Francisco, Singapore, Toronto, Berlin, Stockholm, London, Chicago, Los Angeles, New York, Tokyo, Abu Dhabi, Madrid, Kuala Lumpur, Milan, Moscow, São Paulo, Beijing, Buenos Aires, Johannesburg, Mexico City, Shanghai, Seoul, Istanbul, and Mumbai.

[2] A search on the ISI Web of Science database for keywords "urban AND climate change" for the years 1900–2007 yielded over 700 English language publications. The same search for the period from 2007 to present yielded nearly 2800 English language publications.

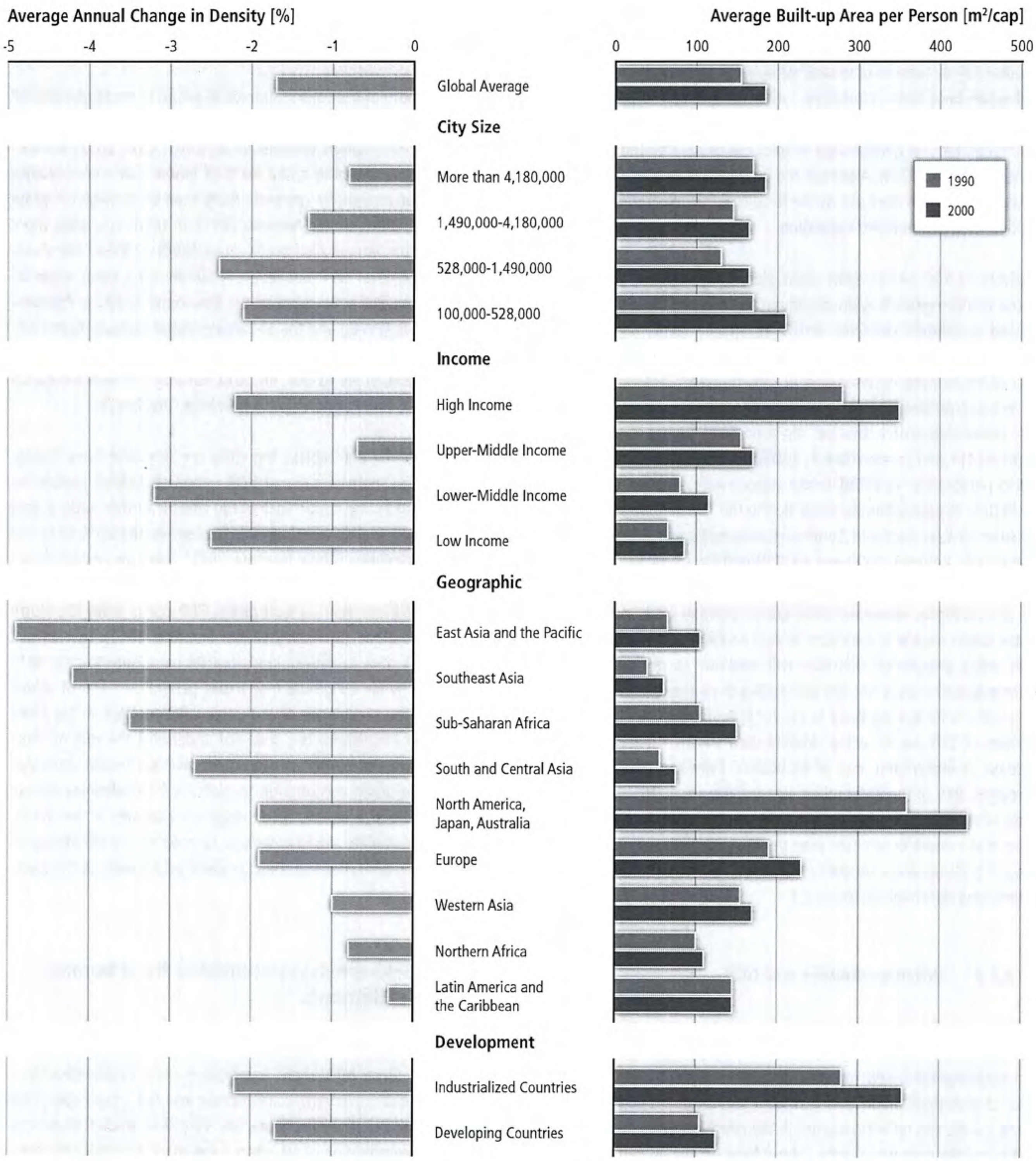


Figure 12.3 | Left: Average annual percent change in density between 1990 and 2010 (light blue). Right: Average built-up area per person (m²) in 1990 (yellow) and 2000 (blue). Data from 120 cities. Source: Angel et al. (2005).

The few studies that do exist will be discussed in Section 12.2.2.1. In contrast, a larger number of studies have quantified GHG emissions for individual cities and other human settlements. These will be assessed in Section 12.2.2.2.

12.2.2.1 Estimates of the urban share of global emissions

There are very few studies that estimate the relative urban and rural shares of global GHG emissions. One challenge is that of boundary definitions and delineation: it is difficult to consistently define and delineate rural and urban areas globally (see Box 12.1). Another challenge is that of severe data constraints about GHG emissions. There is no comprehensive statistical database on urban or rural GHG emissions. Available global estimates of urban and rural emission shares are either derived bottom-up or top-down. Bottom-up, or up-scaling studies, use a representative sample of estimates from regions or countries and scale these up to develop world totals (see IEA, 2008). Top-down studies use global or national datasets and downscale these to local grid cells. Urban and rural emissions contributions are then estimated based on additional spatial information such as the extent of urban areas or the location of emission point sources (GEA, 2012). In the absence of a more substantive body of evidence, large uncertainties remain surrounding the estimates and their sensitivities (Grubler et al., 2012).

The *World Energy Outlook 2008* estimates urban energy related CO_2 emissions at 19.8 Gt, or 71 % of the global total for the year 2006 (IEA, 2008). This corresponds to 330 EJ of primary energy, of which urban final energy use is estimated to be at 222 EJ. The Global Energy Assessment provides a range of final urban energy use between 180 and 250 EJ with a central estimate of 240 EJ for the year 2005. This is equivalent to an urban share between 56 % and 78 % (central estimate, 76 %) of global final energy use. Converting the GEA estimates on urban final energy (Grubler et al., 2012) into CO_2 emissions (see Methodology and Metrics Annex) results in global urban energy related CO_2 emissions of 8.8—14.3 Gt (central estimate, 12.5Gt) which is between 53 % and 87 % (central estimate, 76 %) of CO_2 emissions from global final energy use and between 30 % and 56 % (central estimate, 43 %) of global primary energy related CO_2 emissions (CO_2 includes flaring and cement emissions which are small). Urban CO_2 emission estimates refer to commercial final energy fuel use only and exclude upstream emissions from energy conversion.

Aside from these global assessments, there is only one attempt in the literature to estimate the total GHG (CO_2, CH_4, N_2O and SF_6) contribution of urban areas globally (Marcotullio et al., 2013). Estimates are provided in ranges where the lower end provides an estimate of the direct emissions from urban areas only and the higher end provides an estimate that assigns all emissions from electricity consumption to the consuming (urban) areas. Using this methodology, the estimated total GHG emission contribution of all urban areas is lower than other approaches, and ranges from 12.8 $GtCO_{2eq}$ to 16.9 $GtCO_{2eq}$, or between 37 % and 49 % of global GHG emissions in the year 2000. The estimated urban share of energy related CO_2 emissions in 2000 is slightly lower than the GEA and IEA estimate, at 72 % using Scope 2 accounting and 44 % using Scope 1 accounting (see Figure 12.4). The urban GHG emissions (CO_2, N_2O, CH_4, and SF_6) from the energy share of total energy GHGs is between 42 % and 66 %. Hence, while the sparse evidence available suggests that urban areas dominate final energy consumption and associated CO_2 emissions, the contribution to total global GHG emissions may be more modest as the large majority of CO_2 emissions from land-use change, N_2O emissions, and CH_4 emissions take place outside urban areas.

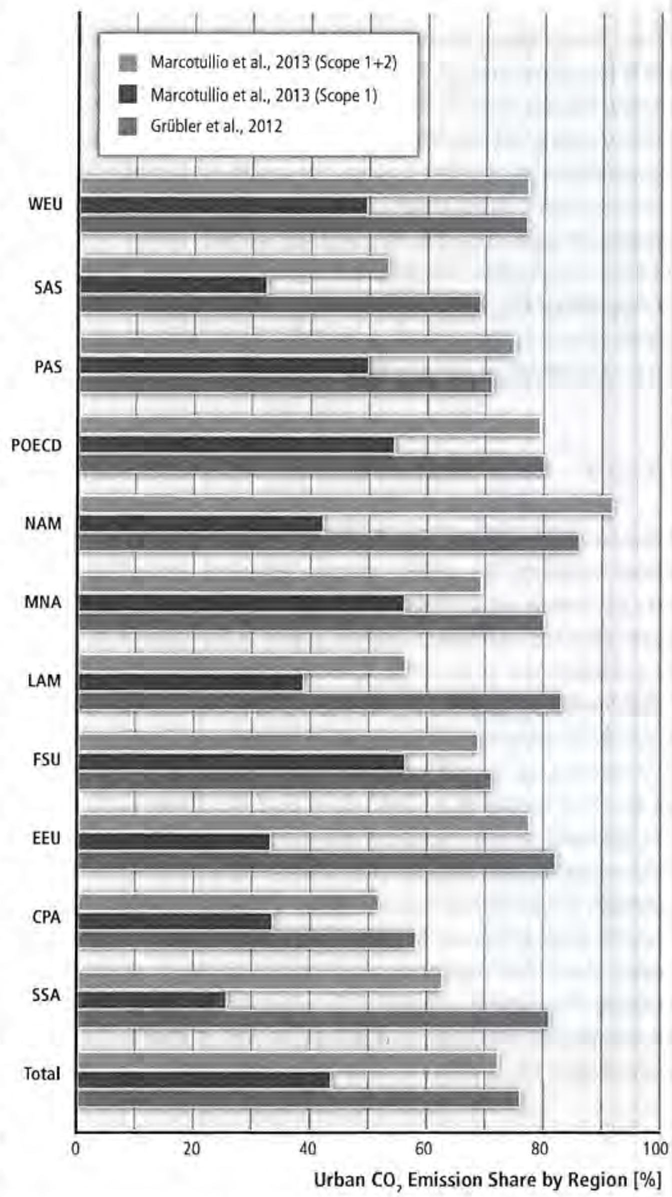


Figure 12.4 | Estimates of urban CO_2 emissions shares of total emissions across world regions. Grubler et al. (2012) estimates are based on estimates of final urban and total final energy use in 2005. Marcotullio et al. (2013) estimates are based on emissions attributed to urban areas as share of regional totals reported by EDGAR. Scope 2 emissions allocate all emissions from thermal power plants to urban areas.

Figure 12.4 shows CO_2 estimates derived from Grubler et al. (2012) and Marcotullio et al. (2013). It highlights that there are large variations in the share of urban CO_2 emissions across world regions. For example, urban emission shares of final energy related CO_2 emissions range from 58 % in China and Central Pacific Asia to 86 % in North America. Ranges are from 31 % to 57 % in South Asia, if urban final energy related CO_2 emissions are taken relative to primary energy related CO_2 emissions in the respective region.

Although differences in definitions make it challenging to compare across regional studies, there is consistent evidence that large variations exist (Parshall et al., 2010; Marcotullio et al., 2011, 2012). For example, the International Energy Agency (IEA) (2008) estimates of the urban primary energy related CO_2 emission shares are 69 % for the EU (69 % for primary energy), 80 % for the United States (85 % for primary energy, see also (Parshall et al., 2010), and 86 % for China (75 % for primary energy, see also Dhakal, 2009). Marcotullio et al. (2013) highlight that non-energy related sectors can lead to substantially different urban emissions shares under consideration of a broader selection of greenhouse gases (CO_2, CH_4, N_2O, SF_6). For example, while Africa tends to have a high urban CO_2 emissions share (64 %–74 %) in terms of energy related CO_2 emissions, the overall contribution of urban areas across all sectors and gases is estimated to range between 21 % and 30 % of all emissions (Marcotullio et al., 2013).

12.2.2.2 Emissions accounting for human settlements

Whereas the previous section discussed the urban proportion of total global emissions, this section assesses emissions accounting methods for human settlements. A variety of emission estimates have been published by different research groups in the scientific literature (e.g.,Ramaswami et al., 2008; Kennedy et al., 2009, 2011; Dhakal, 2009; World Bank, 2010; Hillman and Ramaswami, 2010; Glaeser and Kahn, 2010; Sovacool and Brown, 2010; Heinonen and Junnila, 2011a, c; Hoornweg et al., 2011; Chavez and Ramaswami, 2011; Chavez et al., 2012; Grubler et al., 2012; Yu et al., 2012; Chong et al., 2012). The estimates of GHG emissions and energy consumption for human settlements are very diverse. Comparable estimates are usually only available across small samples of human settlements, which currently limit the insights that can be gained from an assessment of these estimates. The limited number of comparable estimates is rooted in the absence of commonly accepted GHG accounting standards and a lack of transparency over data availabilities, as well as choices that have been made in the compilation of particular estimates:

- **Choice of physical urban boundaries.** Human settlements are open systems with porous boundaries. Depending on how physical boundaries are defined, estimates of energy consumption and GHG emissions can vary significantly (see Box 12.1).

- **Choice of accounting approach/reporting scopes.** There is widespread acknowledgement in the literature for the need to report beyond the direct GHG emissions released from within a settlement's territory. Complementary accounting approaches have therefore been proposed to characterize different aspects of the GHG performance of human settlements (see Box 12.2). Cities and other human settlements are increasingly adopting dual approaches (Baynes et al., 2011; Ramaswami et al., 2011; ICLEI et al., 2012; Carbon Disclosure Project, 2013; Chavez and Ramaswami, 2013).

- **Choice of calculation methods.** There are differences in the methods used for calculating emissions, including differences in emission factors used, methods for imputing missing data, and methods for calculating indirect emissions (Heijungs and Suh, 2010; Ibrahim et al., 2012).

A number of organizations have started working towards standardization protocols for emissions accounting (Carney et al., 2009; ICLEI, 2009; Covenant of Mayors, 2010; UNEP et al., 2010; Arikan, 2011). Further progress has been achieved recently when several key efforts joined forces to create a more broadly supported reporting framework (ICLEI et al., 2012). Ibrahim et al. (2012) show that the differences across reporting standards explains significant cross-sectional variability in reported emission estimates. However, while high degrees of cross-sectional comparability are crucial in order to gain further insight into the emission patterns of human settlements across the world, many applications at the settlement level do not require this. Cities and other localities often compile these data to track their own performance in reducing energy consumption and/or greenhouse gas emissions (see Section 12.7). This makes a substantial body of evidence difficult to use for scientific inquiries.

Beyond the restricted comparability of the available GHG estimates, six other limitations of the available literature remain. First, the growth in publications is restricted to the analysis of energy consumption and GHG emissions from a limited set of comparable emission estimates. New estimates do not emerge at the same pace. Second, available evidence is particularly scarce for medium and small cities as well as rural settlements (Grubler et al., 2012). Third, there is a regional bias in the evidence. Most studies focus on emissions from cities in developed countries with limited evidence from a few large cities in the developing world (Kennedy et al., 2009, 2011; Hoornweg et al., 2011; Sugar et al., 2012). Much of the most recent literature provides Chinese evidence (Dhakal, 2009; Ru et al., 2010; Chun et al., 2011; Wang et al., 2012a, b; Chong et al., 2012; Yu et al., 2012; Guo et al., 2013; Lin et al., 2013; Vause et al., 2013; Lu et al., 2013), but only limited new emission estimates are emerging from that. Evidence on human settlements in least developed countries is almost non-existent with some notable exceptions in the non peer-reviewed literature (Lwasa, 2013). Fourth, most of the available emission estimates are focusing on energy related CO_2 rather than all GHG emissions. Fifth, while there is a considerable amount of evidence for territorial emissions, studies that include Scope 2 and 3 emission components are growing but remain limited (Ramaswami et al., 2008, 2012b; Kennedy et al.,

Box 12.2 | Emission accounting at the local scale

Three broad approaches have emerged for GHG emissions accounting for human settlements, each of which uses different boundaries and units of analysis.

1) Territorial or production-based emissions accounting includes all GHG emissions from activities within a city or settlement's territory (see Box 12.1). This is also referred to as Scope 1 accounting (Kennedy et al., 2010; ICLEI et al., 2012). Territorial emissions accounting is, for example, commonly applied by national statistical offices and used by countries under the United Nations Framework Convention on Climate Change (UNFCCC) for emission reporting (Ganson, 2008; DeShazo and Matute, 2012; ICLEI et al., 2012).

However, human settlements are typically smaller than the infrastructure in which they are embedded, and important emission sources may therefore be located outside the city's territorial boundary. Moreover, human settlements trade goods and services that are often produced in one settlement but are consumed elsewhere, thus creating GHG emissions at different geographic locations associated with the production process of these consumable items. Two further approaches have thus been developed in the literature, as noted below.

2) Territorial plus supply chain accounting approaches start with territorial emissions and then add a well defined set of indirect emissions which take place outside the settlement's territory. These include indirect emissions from (1) the consumption of purchased electricity, heat and steam (Scope 2 emissions), and (2) any other activity (Scope 3 emissions). The simplest and most frequently used territorial plus supply chain accounting approach includes Scope 2 emissions (Hillman and Ramaswami, 2010; Kennedy et al., 2010; Baynes et al., 2011; ICLEI et al., 2012).

3) Consumption-based accounting approaches include all direct and indirect emissions from final consumption activities associated with the settlement, which usually include consumption by residents and government (Larsen and Hertwich, 2009, 2010a, b; Heinonen and Junnila, 2011a, b; Jones and Kammen, 2011; Minx et al., 2013). This approach excludes all emissions from the production of exports in the settlement territory and includes all indirect emissions occurring outside the settlement territory in the production of the final consumption items.

2009; Larsen and Hertwich, 2009, 2010a, b; Hillman and Ramaswami, 2010; White et al., 2010; Petsch et al., 2011; Heinonen and Junnila, 2011a, b; Heinonen et al., 2011; Chavez et al., 2012; Paloheimo and Salmi, 2013; Minx et al., 2013). Finally, the comparability of available evidence of GHG emissions at the city scale is usually restricted across studies. There prevails marked differences in terms of the accounting methods, scope of covered sectors, sector definition, greenhouse gas covered, and data sources used (Bader and Bleischwitz, 2009; Kennedy et al., 2010; Chavez and Ramaswami, 2011; Grubler et al., 2012; Ibrahim et al., 2012).

Across cities, existing studies point to a large variation in the magnitude of total and per capita emissions. For this assessment, emission estimates for several hundred individual cities were reviewed. Reported emission estimates for cities and other human settlements in the literature range from 0.5 tCO_2/cap to more than 190 tCO_2/cap (Carney et al., 2009; Kennedy et al., 2009; Dhakal, 2009; Heinonen and Junnila, 2011a, c; Wright et al., 2011; Sugar et al., 2012; Ibrahim et al., 2012; Ramaswami et al., 2012b; Carbon Disclosure Project, 2013; Chavez and Ramaswami, 2013; Department of Energy & Climate Change, 2013). Local emission inventories in the UK for 2005–2011 show that end use activities and industrial processes of both rural and urban localities vary from below 3 to 190 tCO_2/cap and more (Department of Energy & Climate Change, 2013). The total CO_2 emissions from end use activities for ten global cities range (reference year ranges 2003–2006) between 4.2 and 21.5 tCO_2eq/cap (Kennedy et al., 2009; Sugar et al., 2012), while there is variation reported in GHG estimates from 18 European city regions from 3.5 to 30 tCO_2eq/cap in 2005 (Carney et al., 2009).

In many cases, a large part of the observed variability will be related to the underlying drivers of emissions such as urban economic structures (balance of manufacturing versus service sector), local climate and geography, stage of economic development, energy mix, state of public transport, urban form and density, and many others (Carney et al., 2009; Kennedy et al., 2009, 2011; Dhakal, 2009, 2010; Glaeser and Kahn, 2010; Shrestha and Rajbhandari, 2010; Gomi et al., 2010; Parshall et al., 2010; Rosenzweig et al., 2011; Sugar et al., 2012; Grubler et al., 2012; Wiedenhofer et al., 2013). Normalizing aggregate city-level emissions by population therefore does not necessarily result in robust cross-city comparisons, since each city's economic function, trade typology, and imports-exports balance can differ widely. Hence, using different emissions accounting methods can lead to substantial differences in reported emissions (see Figure 12.4). Therefore, understanding differences in accounting approaches is essential in order to draw meaningful conclusions from cross-city comparisons of emissions.

Evidence from developed countries such as the United States, Finland, or the United Kingdom suggests that consumption-based emission estimates for cities and other human settlements tend to be higher than their territorial emissions. However, in some cases,

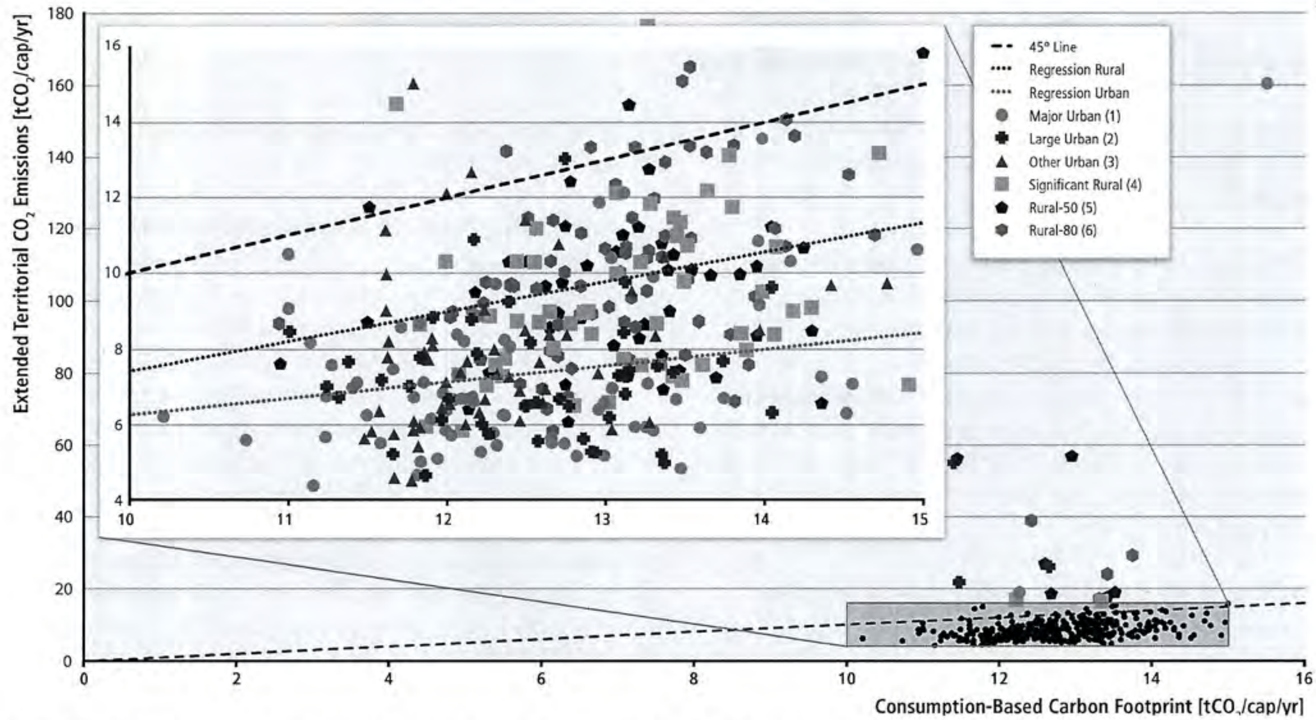


Figure 12.5 | Extended territorial and consumption-based per capita CO_2 emissions for 354 urban (yellow/orange/red) and rural (blue) municipalities in England in 2004. The extended territorial CO_2 emissions accounts assign CO_2 emissions from electricity consumption to each municipality's energy use. The consumption-based carbon footprint accounts assign all emissions from the production of goods and services in the global supply chain to the municipality where final consumption takes place. At the 45° line, per capita extended territorial and consumption-based CO_2 emissions are of equal size. Below the 45° line, consumption-based CO_2 emission estimates are larger than extended territorial emissions. Above the 45° line, estimates of extended territorial CO_2 emissions are larger than consumption-based CO_2 emissions. Robust regression lines are shown for the rural (blue) and urban (yellow/orange/red) sub-samples. In the inset, the x-axis shows 10–15 tonnes of CO_2 emissions per capita and the y-axis shows 4–16 tonnes of CO_2 emissions per capita. Source: Minx et al. (2013).

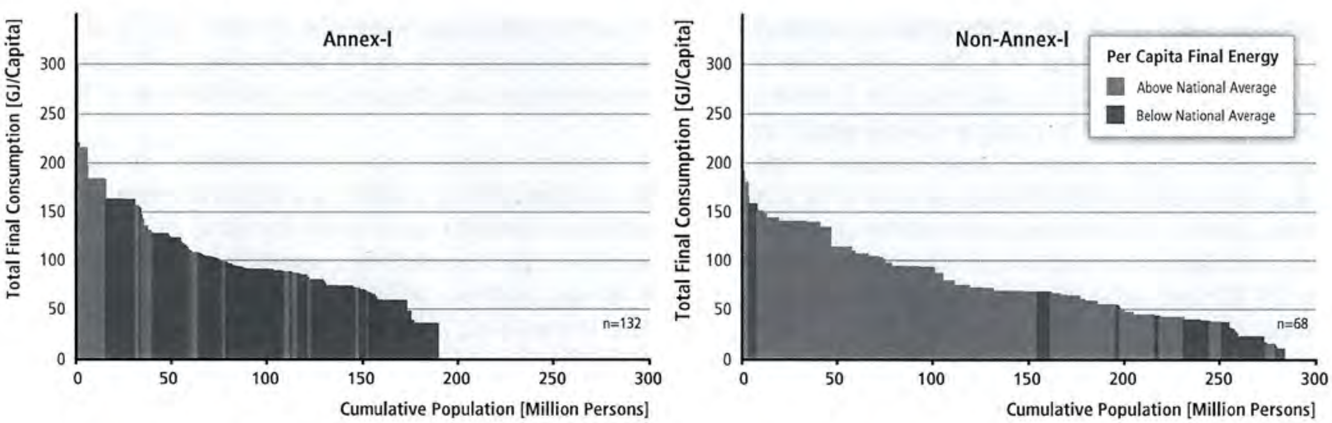


Figure 12.6 | Per capita (direct) total final consumption (TFC) of energy (GJ) versus cumulative population (millions) in urban areas. Source: Grubler et al. (2012).

territorial or extended territorial emission estimates (Scope 1 and Scope 2 emissions) can be substantially higher. This is mainly due to the large fluctuations in territorial emission estimates that are highly dependent on a city's economic structure and trade typology. Consumption-based estimates tend to be more homogenous (see Figure 12.5).

Based on a global sample of 198 cities by the Global Energy Assessment, Grubler et al. (2012) found that two out of three cities in Annex I countries have a lower per capita final energy use than national levels. In contrast, per capita final energy use for more than two out of three cities in non-Annex I countries have higher than national averages (see Figure 12.6). There is not sufficient comparable evidence available for this assessment to confirm this finding for energy related CO_2 emissions, but this pattern is suggested by the close relationship between final energy use and energy related CO_2 emissions. Individual studies for 35 cities in China, Bangkok, and 10 global cities provide additional evidence of these trends (Dhakal, 2009; Aumnad, 2010; Kennedy et al., 2010; Sovacool and Brown, 2010). Moreover, the literature suggests that differences in per capita energy consumption and CO_2 emission patterns of cities in Annex I and non-Annex I countries have converged more than their national emissions (Sovacool and Brown, 2010; Sugar et al., 2012). For consumption-based CO_2 emissions, initial evidence suggests that urban areas tend to have much higher emissions than rural areas in non-Annex I countries, but the evidence is limited to a few studies on India and China (Parikh and Shukla, 1995; Guan et al., 2008, 2009; Pachauri and Jiang, 2008; Minx et al., 2011). For Annex I countries, studies suggest that using consumption based CO_2 emission accounting, urban areas can, but do not always, have higher emissions than rural settlements (Lenzen et al., 2006; Heinonen and Junnila, 2011c; Minx et al., 2013).

There are only a few downscaled estimates of CO_2 emissions from human settlements and urban as well as rural areas, mostly at regional and national scales for the EU, United States, China, and India (Parshall et al., 2010; Raupach et al., 2010; Marcotullio et al., 2011, 2012; Gurney et al., 2012). However, these studies provide little to no representation of intra-urban features and therefore cannot be substitutes for place-based emission studies from cities. Recent studies have begun to combine downscaled estimates of CO_2 emissions with local urban energy consumption information to generate fine-scale maps of urban emissions (see Figure 12.7 and Gurney et al., 2012). Similarly, geographic-demographic approaches have been used for downscaling consumption-based estimates (Druckman and Jackson, 2008; Minx et al., 2013). Such studies may allow more detailed analyses of the drivers of urban energy consumption and emissions in the future.

12.2.3 Future trends in urbanization and GHG emissions from human settlements

This section addresses two issues concerning future scenarios of urbanization. It summarizes projected future urbanization dynamics in multiple dimensions. It assesses and contextualizes scenarios of urban population growth, urban expansion, and urban emissions.

12.2.3.1 Dimension 1: Urban population

Worldwide, populations will increasingly live in urban settlements. By the middle of the century, the global urban population is expected to reach between 5.6 to 7.1 billion, with trends growth varying substantially across regions (Table 12.2). While highly urbanized North America, Europe, Oceania, and Latin America will continue to urbanize, the increase in urbanization levels in these regions is relatively small. Urbanization will be much more significant in Asia and Africa where

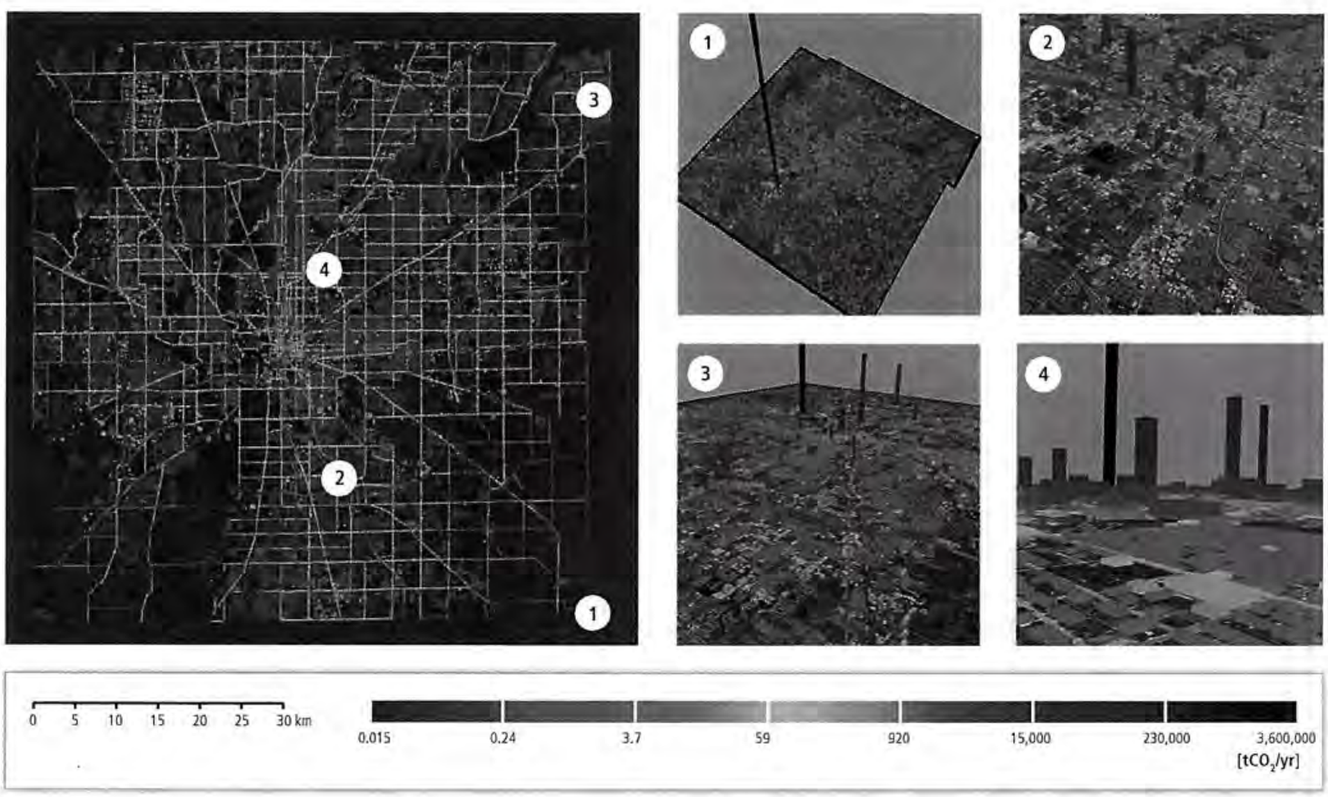


Figure 12.7 | Total fossil fuel emissions of Marion County, Indiana, USA, for the year 2002. Left map: Top-down view with numbered zones. Right four panels: Blow ups of numbered zones. Box height units: Linear. Source: Gurney et al. (2012).

Table 12.2 | Global urban population in 2050 (mid-year)

Source	Total Pop. in billions	% Urban	Urban Pop. in billions
IIASA Greenhouse Gas Index, A2R Scenario	10.245	69	7.069
World Bank	9.417	67	6.308
United Nations	9.306	67	6.252
IIASA Greenhouse Gas Index, B2 Scenario	9.367	66	6.182
IIASA Greenhouse Gas Index, B1 Scenario	8.721	64	5.581

Sources: IIASA (2009), UN DESA (2012), World Bank (2013).

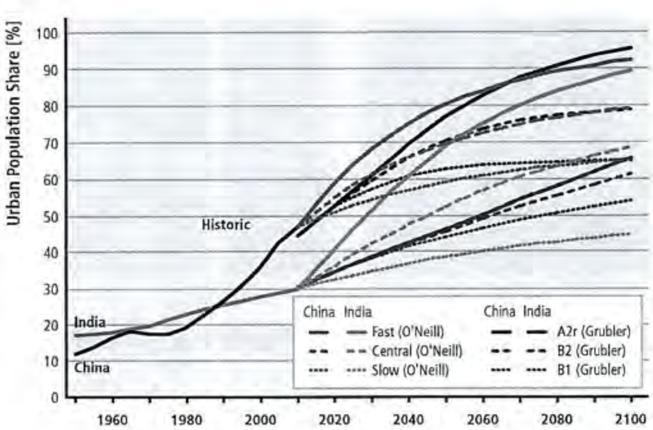
the majority of the population is still rural. Urban population growth will also largely occur in the less developed Africa, Asia, and Latin America. The proportion of rural population in the developed regions have declined from about 60% in 1950 to less than 30% in 2010, and will continue to decline to less than 20% by 2050.

Uncertainties in future global urbanization trends are large, due in part to different trajectories in economic development and population growth. While the United Nations Development Programme (UNPD) produces a single urbanization scenario for each country through 2050, studies suggests that urbanization processes in different countries and different periods of time vary remarkably. Moreover, past UN urbanization projections have contained large errors and have tended to overestimate urban growth, especially for countries at low and middle urbanization levels (Bocquier, 2005; Montgomery, 2008; Alkema et al., 2011).

Given these limitations, recent studies have begun to explore a range of urban population growth scenarios. A study undertaken at International Institute for Applied Systems Analysis (IIASA) extrapolates UN scenarios to 2100 and develops three alternative scenarios by making assumptions about long-term maximum urbanization levels (Grubler et al., 2007). However, missing from these scenarios is the full range of uncertainty over the next twenty to thirty years, the period when the majority of developing countries will undergo significant urban transitions. For instance, variation across different urbanization scenarios before 2030 is negligible (0.3%) for India and also very small (< 4%) for China (see Figure 12.8, dashed lines). By 2050, urbanization levels could realistically reach between 38–69% in India, and 55–78% in China (O'Neill et al., 2012). In other words, there are large uncertainties in urbanization trajectories for both countries. The *speed* (fast or slow) as well as the *nature* (an increase in industrialization) of urbanization could lead to significant effects on future urban energy use and emissions.

12.2.3.2 Dimension 2: Urban land cover

Recently, global forecasts of urban expansion that take into account population and economic factors have become available (Nelson et al., 2010; Angel et al., 2011; Seto et al., 2011, 2012). These studies vary in their baseline urban extent, model inputs, assumptions about future trends in densities, economic and population growth, and modelling methods. They forecast that between 2000 and 2030, urban areas will expand between 0.3 million to 2.3 million km², corresponding to an increase between 56% to 310% (see Table 12.3 and Angel et al., 2011; Seto et al., 2011, 2012). It is important to note that these studies forecast changes in urban land cover (features of Earth's surface) and not changes in the built environment and infrastructure (e.g., buildings, roads). However, these forecasts of urban land cover can be useful to project infrastructure development and associated emissions. Given worldwide trends of declining densities, the zero population density decline scenario and associated urban growth forecast (0.3 million) is unlikely, as is the *Special Report on Emissions Scenarios* (SRES) A1 scenario of very rapid economic growth and a peak in global population mid-century. According to the studies, the most likely scenarios are SRES B2 (Seto et al., 2011), > 75% probability (Seto et al., 2012), and 2% decline (Angel et al., 2011), which reduces the range of forecast estimates to between 1.1 to 1.5 million km² of new urban land. This corresponds to an increase in urban land cover between 110% to 210% over the 2000 global urban extent. Hurtt et al. (2011) report



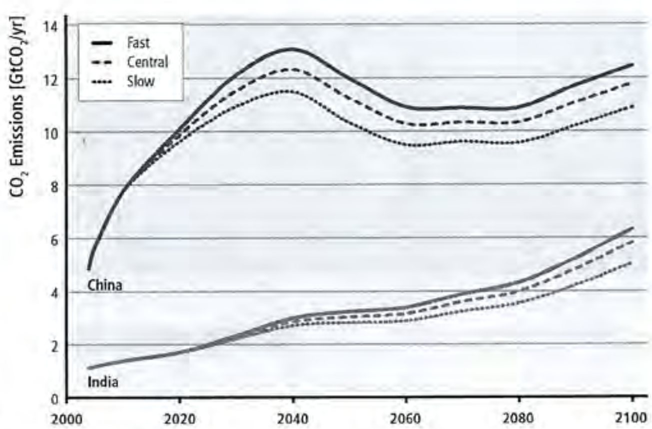


Figure 12.8 | Projected urban population growth for India and China under fast, central, and slow growth scenarios (left) and associated growth in CO_2 emissions (right). Sources: O'Neill et al. (2012), Grubler et al. (2007).

projected land-use transitions including urbanization, out to 2100, for the intended use in Earth System Models (ESMs). However, they do not give a detailed account of the projected urban expansion in different parts of the world.

Depending on the scenario and forecast, 55 % of the total urban land in 2030 is expected to be built in the first three decades of the 21st century. Nearly half of the global growth in urban land cover is forecasted to occur in Asia, and 55 % of the regional growth will take place in China and India (Seto et al., 2012). China's urban land area is expected to expand by almost 220,000 km² by 2030, and account for 18 % of the global increase in urban land cover (Seto et al., 2012). These forecasts provide first-order estimates of the likelihood that expansion of urban areas will occur in areas of increasing vulnerability to extreme climate events including floods, storm surges, sea level rise, droughts, and heat waves (see WGII AR5 Chapter 8). Urban expansion and associated land clearing and loss of aboveground biomass carbon in the pan-tropics is expected to be 1.38 PgC between 2000 and 2030, or 0.05 PgC/yr (Seto et al., 2012).

12.2.3.3 Dimension 3: GHG emissions

Recent developments in integrated models are beginning to capture the interdependence among urban population, urban land cover, and GHG emissions. Some integrated models have found that changes in urbanization in China and India have a less than proportional effect on aggregate emissions and energy use (O'Neill et al., 2012). These studies find that income effects due to economic growth and urbanization result in household consumption shifts toward cleaner cooking fuels (O'Neill et al., 2012). In India, the urbanization level in 2050 will be 16 percentage points lower under the slow urbanization scenario than under the central scenario, or 15 percentage points higher under the fast scenario than under the central scenario. However, these large differences in potential urbanization levels in India lead to relatively small differences in emissions: 7 % between the slow and central urbanization scenarios, and 6 % between the fast and central urbanization scenarios (O'Neill et al., 2012). The relatively small effect of urbanization on emissions is likely due to relatively small differences in per capita income between rural and urban areas (O'Neill et al., 2012). In contrast, large differences in per capita income between urban and rural areas in China result in significant differences in household consumption, including for energy (O'Neill et al., 2012). Differences in urbanization pathways also reflect different speeds of transition away from the use of traditional fuels toward modern fuels such as electricity and natural gas (Krey et al., 2012). Slower rates of urbanization result in slower transitions away from traditional to modern fuels (Jiang and O'Neill, 2004; Pachauri and Jiang, 2008). A large share of solid fuels or traditional biomass in the final energy mix can have adverse health impacts due to indoor air pollution (Bailis et al., 2005; Venkataraman et al., 2010).

Accounting for uncertainties in urban population growth, the scenarios show that urbanization as a demographic process does not lead to a

Table 12.3 | Forecasts of global urban land expansion to 2030. Sources: Angel et al. (2011), Seto et al. (2011, 2012).

Study	Scenario	Projected Urban Expansion to 2030 (km²)								% of projected urban land in 2030 to be built between 2000–2030
		Urban Land 2000 (km²)	Africa	Asia	Europe	Latin America	North America	Oceania	Total (% increase from 2000)	
Seto et al. (2011)	SRES A1	726,943	107,551	1,354,001	296,638	407,214	73,176	16,996	2,255,576 (310)	76
	SRES A2	726,943	113,423	702,772	162,179	122,438	49,487	15,486	1,165,785 (160)	62
	SRES B1	726,943	107,551	1,238,267	232,625	230,559	86,165	18,106	1,913,273 (263)	72
	SRES B2	726,943	136,419	989,198	180,265	131,016	74,572	15,334	1,526,805 (210)	68
Seto et al. (2012)	> 75 % probability	652,825	244,475	585,475	77,575	175,075	118,175	9,700	1,210,475	65
		Urban Land 2000 (km²)	Africa	Asia	East Asia and the Pacific	Europe and Japan	Latin America and the Caribbean	Land Rich Developed Countries	Total (% increase from 2000)	
Angel et al. (2011)	0 % density decline	602,864	58,132	120,757	43,092	9,772	49,348	54,801	335,902 (56)	36
	1 % density decline	602,864	92,002	203,949	75,674	74,290	98,554	119,868	664,337 (110)	52
	2 % density decline	602,846	137,722	316,248	119,654	161,379	164,975	207,699	1,107,677 (184)	65

corresponding growth in emissions and energy use (Figure 12.8b). In China, for example, under the central scenario (similar to UN projections) the country will reach 70 % urban population by 2050 and the total carbon emissions will reach 11 GtC/yr. Under the slow urbanization scenario, the urbanization level is 13 % lower than the central urbanization scenario, but results in emissions that are 9 % lower than under the central urbanization scenario. Similarly, the fast urbanization scenario results in emissions that are 7 % higher than under the central scenario, but with urbanization levels that are 11 % higher.

Studies of the effects of demographic change on GHG emissions come to contradicting conclusions (Dalton et al., 2008; Kronenberg, 2009). Many of the forecasts on urbanization also do not explicitly account for the infrastructure for which there is a separate set of forecasts (Davis et al., 2010; Kennedy and Corfee-Morlot, 2013; Müller et al., 2013) including those developed by the IEA (IEA, 2013) and the Organisation for Economic Co-operation and Development (OECD) (OECD, 2006b, 2007). However these infrastructure forecasts, typically by region or country, do not specify the portion of the forecasted infrastructure in urban areas and other settlements. One study finds that both ageing and urbanization can have substantial impacts on emissions in certain world regions such as the United States, the EU, China, and India. Globally, a 16–29 % reduction in the emissions by 2050 (1.4–2.5 GtC/yr) could be achieved through slowing population growth (O'Neill et al., 2010).

12.3 Urban systems: Activities, resources, and performance

How does urbanization influence global or regional CO_2 emissions? This section discusses drivers of urban GHG emissions, how they affect different sectors, and their interaction and interdependence. The magnitude of their impact on urban GHG emissions is also discussed qualitatively and quantitatively to provide context for a more detailed assessment of urban form and infrastructure (12.4) and spatial planning (12.5).

12.3.1 Overview of drivers of urban GHG emissions

Urban areas and nations share some common drivers of GHG emissions. Other drivers of urban GHG emissions are distinct from national drivers and are locally specific. The previous section discussed important accounting issues that affect the estimation of urban-scale GHG emissions. (For a more comprehensive review, see Kennedy et al., 2009; ICLEI et al., 2012; Ramaswami et al., 2012b; Steinberger and Weisz, 2013). Another characteristic of urban areas is that their physical form and structure in terms of land-use mix and patterns, density, and spatial configuration of infrastructure can strongly influence GHG emissions (see discussion below and in 12.4). The basic constituent elements of cities such as streets, public spaces, buildings, and their design, placement, and function reflect their socio-political, economic, and technological histories (Kostof, 1992; Morris, 1994; Kostof and Tobias, 1999). Hence, cities often portray features of 'path dependency' (Arthur, 1989), a historical contingency that is compounded by the extent of pre-existing policies and market failures that have lasting impacts on emissions (see Section 12.6 below).

The following sections group and discuss urban GHG emission drivers into four clusters that reflect both the specificity of urban scale emissions as well as their commonality with national-scale drivers of GHG emissions addressed in the other chapters of this assessment:

- Economic geography and income
- Socio-demographic factors
- Technology
- Infrastructure and urban form

Economic geography refers to the function of a human settlement within the global hierarchy of places and the international division of labour, as well as the resulting trade flows of raw materials, energy, manufactured goods, and services. Income refers to the scale of economic activity, often expressed through measures of Gross Regional Product (GRP) (i.e., the GDP equivalent at the scale of human settlements), calculated either as an urban (or settlement) total, or normalized on a per capita basis.

Socio-demographic drivers of urban GHG emissions include population structure and dynamics (e.g., population size, age distribution, and household characteristics) (O'Neill et al., 2010) as well as cultural norms (e.g., consumption and lifestyle choices) and distributional and equity factors (e.g., access or lack thereof to basic urban infrastructure). Unequal access to housing and electricity is a significant social problem in many rapidly growing cities of the Global South (Grubler and Schulz, 2013) and shapes patterns of urban development. Here, 'technology' refers to macro-level drivers such as the technology of manufacturing and commercial activities. 'Infrastructure' and 'urban form' refer to the patterns and spatial arrangements of land use, transportation systems, and urban design elements (Lynch, 1981; Handy, 1996) and are discussed in greater detail in Section 12.4.

12.3.1.1 Emission drivers decomposition via IPAT

Explaining GHG emission growth trends via decomposition analysis is a widely used technique in the scientific literature and within IPCC assessments ever since Kaya (1990). The so-called IPAT identity (for a review, see Chertow, 2000) is a multiplicative identity in which Impacts (e.g., emissions) are described as being the product of Population *x* Affluence *x* Technology. First derivatives (growth rates) of the components of this identity become additive, thus allowing a first analysis on the relative weight of different drivers. The IPAT identity is

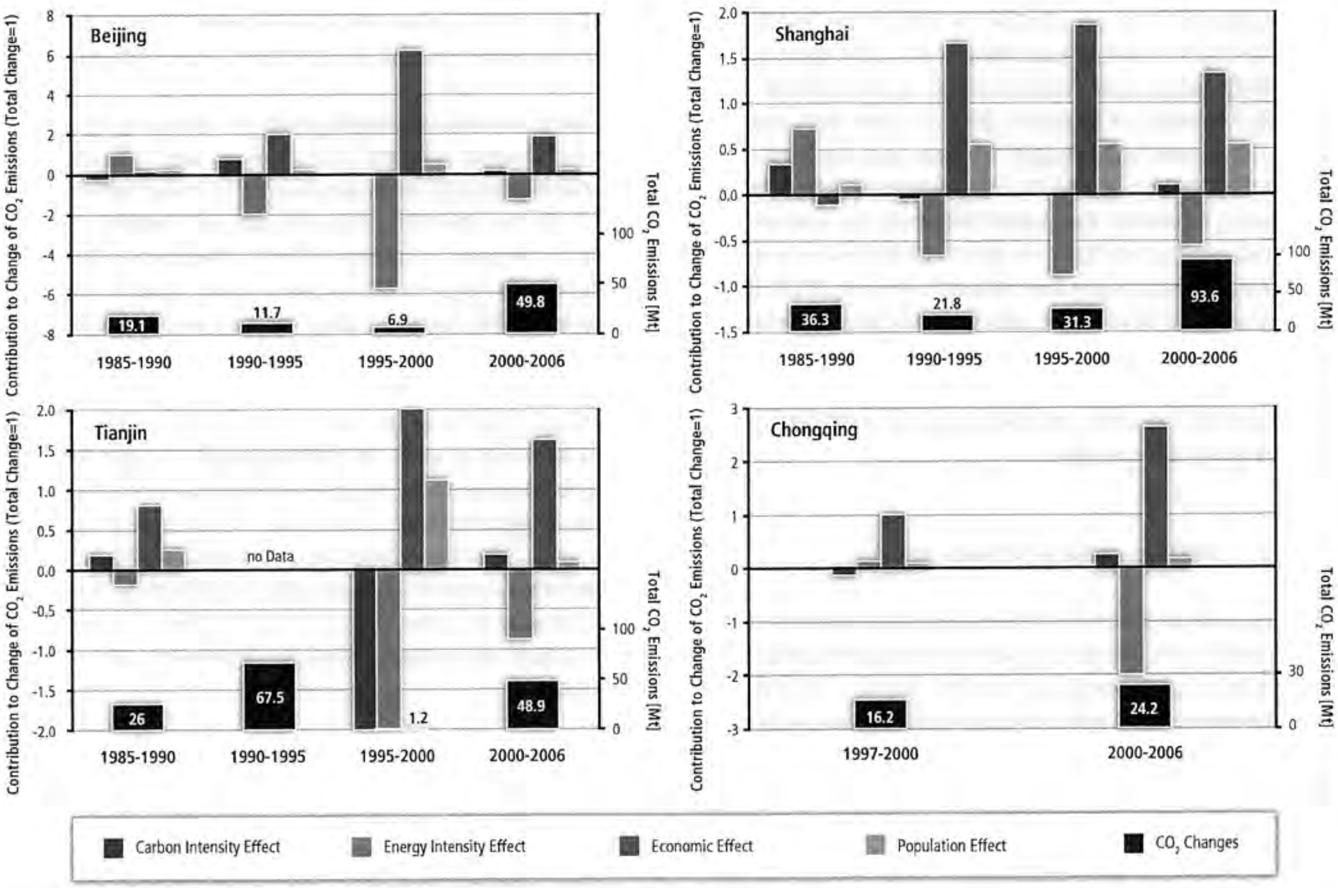


Figure 12.9 | Decomposition of urban-scale CO_2 emissions (absolute difference over time period specified (dark blue) and renormalized to index 1 (other colours)) for four Chinese cities 1985 to 2006. Source: Grubler et al. (2012) based on Dhakal (2009). Note the 'economic effect' in the graph corresponds to an income effect as discussed in the text. For comparison, per capita CO_2 emissions for these four cities range between 11.7 (Shanghai), 11.1 (Tianjin), 10.1 (Beijing), and 3.7 (Chongqing) tCO_2/cap (Hoornweg et al., 2011).

a growth accounting framework and does not lend itself to explaining differences between urban settlements in terms of absolute GHG emission levels and their driving forces (see discussion below).

There is great interest in understanding the drivers of China's urban GHG emissions, which has resulted in a large literature on the decomposition of GHG emissions for Chinese megacities. With approximately 10 tonnes of CO_2 per urban capita—three times the national average—China approaches and in some cases, surpasses levels for Annex-I countries and cities (Dhakal, 2009). Studies have used national emission inventory methods following the IPCC/OECD guidelines (Dhakal, 2009; Chong et al., 2012) or input-output techniques (Wang et al., 2013) and thus have used both production and consumption accounting perspectives. Studies have also gone beyond the simple IPAT accounting framework, such as using index decomposition (Donglan et al., 2010). Together, these studies show considerable variation in per capita GHG emissions across Chinese cities (see, for example, Figure 12.9). Although the relative contribution of different drivers of emissions varies across cities and time periods, one study of several Chinese cities found that income is the most important driver of increases in urban carbon emissions, far surpassing population growth, with improvements in energy efficiency serving as a critical counterbalancing factor to income growth (Dhakal, 2009). The importance of economic growth as a driver of urban CO_2 emissions in China has been consistently corroborated in other studies, including those that examine relatively smaller cities and with the use of alternative types of data and methods (Li et al., 2010; Liu et al., 2012; Chong et al., 2012; Jiang and Lin, 2012).

However, the evidence on whether the gains in efficiency can counterbalance the scale of infrastructure construction and income growth in China is less conclusive. Several studies implemented at different spatial scales have found that the scale of urbanization and associated consumption growth in China have outpaced gains from improvements in efficiency (Peters et al., 2007; Feng et al., 2012; Güneralp and Seto, 2012). Other studies have found that improvements in efficiency offset the increase in consumption (Liu et al., 2007; Zhang et al., 2009; Minx et al., 2011).

The literature on drivers of urban GHG emissions in other non-Annex I countries is more sparse, often focusing on emission drivers at the sectoral level such as transport (Mraihi et al., 2013) or household energy use (Ekholm et al., 2010). In these sectoral studies, income and other factors (that are highly correlated with income) such as vehicle ownership and household discount rates, are also shown as important determining variables.