ingly approaching technical limits. However, many options for energy efficiency improvement remain and there is still significant potential to reduce the gap between actual energy use and the best practice in many industries and in most countries. For all, but particularly for less energy intensive industries, there are still many energy efficiency options both for process and system-wide technologies and measures. Several detailed analyses related to particular sectors estimate the technical potential of energy efficiency measures in industry to be approximately up to 25 % (Schäfer, 2005; Allwood et al., 2010; UNIDO, 2011; Saygin et al., 2011b; Gutowski et al., 2013). Through innovation, additional reductions of approximately up to 20 % in energy intensity may potentially be realized before approaching technological limits in some energy-intensive industries (Allwood et al., 2010).

In industry, energy efficiency opportunities are found within sector-specific processes as well as in systems such as steam systems, process heating systems (furnaces and boilers), and electric motor systems (e.g., pumps, fans, air compressor, refrigerators, material handling). As a class of technology, electronic control systems help to optimize performance of motors, compressors, steam combustion, heating, etc. and improve plant efficiency cost-effectively with both energy savings and emissions benefits, especially for SMEs (Masanet, 2010).

Opportunities to improve heat management include better heat exchange between hot and cold gases and fluids, improved insulation, capture and use of heat in hot products, and use of exhaust heat for electricity generation or as an input to lower temperature processes (US DoE, 2004a, 2008). However, the value of these options is in many cases limited by the low temperature of 'waste heat'—industrial heat exchangers generally require a temperature difference of ~200 °C—and the difficulty of exchanging heat out of solid materials.

Recycling can also help to reduce energy demand, as it can be a strategy to create material with less energy. Recycling is already widely applied for bulk metals (steel, aluminium, and copper in particular), paper, and glass and leads to an energy saving when producing new material from old avoids the need for further energy intensive chemical reactions. Plastics recycling rates in Europe are currently around 25 % (Plastics Europe, 2012) due to the wide variety of compositions in common use in small products, and glass recycling saves little energy as the reaction energy is small compared to that needed for melting (Sardeshpande et al., 2007). Recycling is applied when it is cost effective, but in many cases leads to lower quality materials, is constrained by lack of supply because collection rates, while high for some materials (particularly steel), are not 100 %, and because with growing global demand for material, available supply of scrap lags total demand. Cement cannot be recycled, although concrete can be crushed and down-cycled into aggregates or engineering fill. However, although this saves on aggregate production, it may lead to increased emissions, due to energy used in concrete crushing and refinement and because more cement is required to achieve target properties (Dosho, 2008).

- **Emissions efficiency (G/E)**: In 2008, 42 % of industrial energy supply was from coal and oil, 20 % from gas, and the remainder from electricity and direct use of renewable energy sources. These shares are forecast to change to 30 % and 24 % respectively by 2035 (IEA, 2011a) resulting in lower emissions per unit of energy, as discussed in Chapter 7. Switching to natural gas also favours more efficient use of energy in industrial combined heat and power (CHP) installations (IEA, 2008, 2009a). For several renewable sources of energy, CHP (IEA, 2011b) offers useful load balancing opportunities if coupled with low-grade heat storage; this issue is discussed further in Chapter 7. The use of wastes and biomass in the energy industry is currently limited, but forecast to grow (IEA, 2009b). The cement industry incinerates (with due care for e.g., dioxins/furans) municipal solid waste and sewage sludge in kilns, providing ~17 % of the thermal energy required by European Union (EU) cement production in 2004 (IEA ETSAP, 2010). The European paper industry reports that over 50 % of its energy supply is from biomass (CEPI, 2012). If electricity generation is decarbonized, greater electrification, for example appropriate use of heat pumps instead of boilers (IEA, 2009b; HPTCJ, 2010), could also reduce emissions. Solar thermal energy for drying, washing, and evaporation may also be developed further (IEA, 2009c) although to date this has not been implemented widely (Sims et al., 2011).

The International Energy Agency (IEA) forecasts that a large part of emission reduction in industry will occur by carbon dioxide capture and storage (CCS) (up to 30 % in 2050) (IEA, 2009c). Carbon dioxide capture and storage is largely discussed in Chapter 7. In gas processing (Kuramochi et al., 2012a) and parts of the chemical industry (ammonia production without downstream use of CO_2), there might be early opportunities for application of CCS as the CO_2 in vented gas is already highly concentrated (up to 85 %), compared to cement or steel (up to 30 %). Industrial utilization of CO_2 was assessed in the IPCC Special Report on Carbon Dioxide Capture and Storage (SRCCS) (Mazzotti et al., 2005) and it was found that potential industrial use of CO_2 was rather small and the storage time of CO_2 in industrial products often short. Therefore industrial uses of CO_2 are unlikely to contribute to a great extent to climate change mitigation. However, currently CO_2 use is subject of various industrial RD&DD projects (Research and Development, Demonstration and Diffusion).

- In terms of non-CO_2-emissions from industry, HFC-23 emissions, which arise in HCFC-22 production, can be reduced by process optimization and by thermal destruction. N_2O emissions from adipic and nitric acid production have decreased almost by half between 1990 and 2010 (EPA, 2012a) due to the implementation of thermal destruction and secondary catalysts.

10

Box 10.2 | Service demand reduction and mitigation opportunities in industry sector:

Besides technological mitigation measures, an additional mitigation option (see Figure 10.2.) for the industry sector involves the end uses of industrial products that provide services to consumers (e.g., diet, mobility, shelter, clothing, amenities, health care and services, hygiene). Assessment of the mitigation potential associated with this option is nascent, however, and important knowledge gaps exist (for a more general review of sustainable consumption and production (SCP) policies, see Section 10.11.3 and 4.4.3). The nature of the linkage between service demand and the demand for industrial products is different and shown here through two examples representing both a direct and an indirect link:

- clothing demand, which is linked directly to the textile industry products (strong link)
- tourism demand, which is linked directly to mobility and shelter demand but also indirectly to industrial materials demand (weak link)

Clothing demand: Even in developed economies, consumers appear to have no absolute limit to their demand for clothing, and if prices fall, will continue to purchase more garments: during the period 2000–2005, the advent of 'fast fashion' in the UK led to a drop in prices, but an increase in sales equivalent to one third more garments per year per person with consequent increases in material production and hence industrial emissions (Allwood et al., 2008). This growth in demand relates to 'fashion' and 'conspicuous consumption' (Roy and Pal, 2009) rather than 'need', and has triggered a wave of interest in concepts like 'sustainable lifestyle/fashion'. While much of this interest is related to marketing new fabrics linked to environmental claims, authors such as Fletcher (2008) have examined the possibility that 'commodity' clothing, which can be discarded easily, would be used for longer and valued more, if given personal meaning by some shared activity or association.

Tourism demand: GHG emissions triggered by tourism significantly contribute to global anthropogenic CO_2 emissions. Estimates show a range between 3.9 % to 6 % of global emissions, with a best estimate of 4.9 % (UNWTO et al., 2008). Worldwide, three quarters (75 %) of tourism-related emissions are generated by transport and just over 20 % by accommodation (UNWTO et al., 2008). A minority of travellers (frequent travellers using the plane over long distances) (Gössling et al., 2009) are responsible for the greater share of these emissions (Gössling et al., 2005; TEC and DEEE, 2008; de Bruijn et al., 2010) (see Sections 8.1.2 and 8.2.1).

Mitigation options for tourism (Gössling, 2010; Becken and Hay, 2012) include technical, behavioural, and organizational aspects. Many mitigation options and potentials are the same as those identified in the transport and buildings chapters (see Chapters 8 and 9). However, the demand reduction of direct tourism-related products delivered by the industry in addition to products for buildings and other infrastructure e.g., snow-lifts and associated accessories, artificial snow, etc. can also impact the industry sector as they determine product and material demand of the sector. Thus, the industry sector has only limited influence on emissions from tourism (via reduction of the embodied emissions), but is affected by decisions in mitigation measures in tourism. For example, a sustainable lifestyle resulting in a lower demand for transportation can reduce demand for steel to manufacture cars and contribute to reducing emissions in the industry sector.

A business-as-usual (BAU) scenario (UNWTO et al., 2008) projects emissions from tourism to grow by 130 % from 2005 to 2035 globally; notably the emissions of air transport and accommodation will triple. Two alternative scenarios show that the contribution of technology is limited in terms of achievable mitigation potentials and that even when combining technological and behavioural potentials, no significant reduction can be achieved in 2035 compared to 2005. Insufficient technological mitigation potential and the need for drastic changes in the forms of tourism (e.g., reduction in long haul travel; UNWTO et al., 2008), in the place of tourism (Gössling et al., 2010; Peeters and Landré, 2011) and in the uses of leisure time, implying changes in lifestyles (Ceron and Dubois, 2005; Dubois et al., 2011) are the limiting factors.

Several studies show that for some countries (e.g., the UK) an unrestricted growth of tourism would consume the whole carbon budget compatible with the +2 °C target by 2050 (Bows et al., 2009; Scott et al., 2010). However, some authors also point out that by reducing demand in some small subsectors of tourism (e.g., long haul, cruises) effective emission reductions may be reached with a minimum of damage to the sector (Peeters and Dubois, 2010).

Tourism is an example of human activity where the discussion of mitigation is not only technology-driven, but strongly correlated with lifestyles. For many other activities, the question is how certain mitigation goals would result in consequences for the activity level with indirect implications for industry sector emissions.

Hydrofluorocarbons used as refrigerants can be replaced by alternatives (e.g., ammonia, hydrofluoro-olefins, HC, CO_2). Replacement is also an appropriate measure to reduce HFC emissions from foams (use of alternative blowing agents) or solvent uses. Emission reduction (in the case of refrigerants) is possible by leak repair, refrigerant recovery and recycling, and proper disposal. Emissions of PFCs, SF_6 and nitrogen trifluoride (NF_3) are growing rapidly due to flat panel display manufacturing. Ninety-eight percent of these emissions are in China (EPA, 2012a) and can be countered by fuelled combustion, plasma, and catalytic technologies.

- **Material efficiency in production (M/P):** Material efficiency—delivering services with less new material—is a significant opportunity for industrial emissions abatement, that has had relatively little attention to date (Allwood et al., 2012). Two key strategies would significantly improve material efficiency in manufacturing existing products:
  - *Reducing yield losses in materials production, manufacturing, and construction*. Approximately one-tenth of all paper, a quarter of all steel, and a half of all aluminium produced each year is scrapped (mainly in downstream manufacturing) and internally recycled—see Figure 10.2. This could be reduced by process innovations and new approaches to design (Milford et al., 2011).
  - *Re-using old material*. A detailed study (Allwood et al., 2012) on re-use of structural steel in construction concluded that there are no insurmountable technical barriers to re-use, that there is a profit opportunity, and that the potential supply is growing.
- **Material efficiency in product design (M/P):** Although new steels and production techniques have allowed relative light-weighting of cars, in practice cars continue to become heavier as they are larger and have more features. However, many products could be one-third lighter without loss of performance in use (Carruth et al., 2011) if design and production were optimized. At present, the high costs of labour relative to materials and other barriers inhibit this opportunity, except in industries such as aerospace where the cost of design and manufacture for lightness is paid back through reduced fuel use. Substitution of one material by another is often technically possible (Ashby, 2009), but options for material substitution as an abatement strategy are limited: global steel and cement production exceeds 200 and 380 (kg/cap)/yr respectively, and no other materials capable of delivering the same functions are available in comparable quantities; epoxy based composite materials and magnesium alloys have significantly higher embodied energy than steel or aluminium (Ashby, 2009) (although for vehicles this may be worthwhile if it allows significant savings in energy during use); wood is kiln dried, so in effect is energy intensive (Puettmann and Wilson, 2005); and blast furnace slag and fly ash from coal-fired power stations can substitute to some extent for cement clinker.
- **Using products more intensively (P/S):** Products, such as food, that are intended to be consumed in use are in many cases used inefficiently, and estimates show that up to one-third of all food in developed countries is wasted (Gustavsonn et al., 2011). This indicates the opportunity for behaviour change to reduce significantly the demand for industrial production of what currently becomes waste without any service provision. In contrast to these consumable products, most durable goods are owned in order to deliver a 'product service' rather than for their own sake, so potentially the same level of service could be delivered with fewer products. Using products for longer could reduce demand for replacement goods, and hence reduce industrial emissions (Allwood et al., 2012). New business models could foster dematerialization and more intense use of products. The ambition of the 'sustainable consumption' agenda and policies (see Sections 10.11 and 4.4.3) aims towards this goal, although evidence of its application in practice remains scarce.
- **Reducing overall demand for product services (S)** (see Box 10.2): Industrial emissions would be reduced if overall demand for product services were reduced (Kainuma et al., 2013)—if the population chose to travel less (e.g., through more domestic tourism or telecommuting), heat or cool buildings only to the degree required, or reduce unnecessary consumption or products. Clear evidence that, beyond some threshold of development, populations do not become 'happier' (as reflected in a wide range of socio-economic measures) with increasing wealth, suggests that reduced overall consumption might not be harmful in developed economies (Layard, 2011; Roy and Pal, 2009; GEA, 2012), and a literature questioning the ultimate policy target of GDP growth is growing, albeit without clear prescriptions about implementation (Jackson, 2011).

In the rest of this section, the application of these six strategies, where it exists, is reviewed for the major emitting industrial sectors.

10.4.1 Iron and steel

Steel continues to dominate global metal production, with total crude steel production of around 1,490 Mt in 2011. In 2011, China produced 46 % of the world's steel. Other significant producers include the EU-27 (12 %), the United States (8 %), Japan (7 %), India (5 %) and Russia (5 %) (WSA, 2012b). Seventy percent (70 %) of all steel is made from pig iron produced by reducing iron oxide in a blast furnace using coke or coal before reduction in an oxygen blown converter (WSA, 2011). Steel is also made from scrap (23 %) or from iron oxide reduced in solid state (direct reduced iron, 7 %) melted in electric-arc furnaces before refining. The specific energy intensity of steel production varies by technology and region. Global steel sector emissions were estimated to be 2.6 $GtCO_2$ in 2006, including direct and indirect emissions (IEA, 2009c; Oda et al., 2012).

Energy efficiency. The steel industry is pursuing: improved heat and energy recovery from process gases, products and waste streams; improved fuel delivery through pulverized coal injection; improved furnace designs and process controls; and reduced number of temperature cycles through better process coupling such as in Endless Strip Production (ESP) (Arvedi et al., 2008) and use of various energy efficiency technologies (Worrell et al., 2010; Xu et al., 2011a) including coke dry quenching and top pressure recovery turbines (LBNL and AISI, 2010). Efforts to promote energy efficiency and to reduce the production of hazardous wastes are the subject of both international guidelines on environmental monitoring (International Finance Corporation, 2007) and regional benchmarks on best practice techniques (EC, 2012a).

Emissions efficiency: The coal and coke used in conventional iron-making is emissions intensive; switching to gas-based direct reduced iron (DRI) and oil and natural gas injection has been used, where economic and practicable. However, DRI production currently occurs at smaller scale than large blast furnaces (Cullen et al., 2012), and any emissions benefit depends on the emissions associated with increased electricity use for the required electric arc furnace (EAF) process. Charcoal, another coke substitute, is currently used for iron-making, notably in Brazil (Taibi et al.; Henriques Jr. et al., 2010), and processing to improve charcoal's mechanical properties is another substitute under development, although extensive land area is required to produce wood for charcoal. Other substitutions include use of ferro-coke as a reductant (Takeda et al., 2011) and the use of biomass and waste plastics to displace coal (IEA, 2009c). The Ultra-Low CO_2 Steelmaking (ULCOS) programme has identified four production routes for further development: top-gas recycling applied to blast furnaces, HIsarna (a smelt reduction technology), advanced direct reduction, and electrolysis. The first three of these routes would require CCS (discussion of the costs, risks, deployment barriers and policy aspects of CCS can be found in Sections 7.8.2, 7.9, 7.10, and 7.12), and the fourth would reduce emissions only if powered by low carbon electricity. Hydrogen fuel might reduce emissions if a cost effective emissions free source of hydrogen were available at scale, but at present this is not the case. Hydrogen reduction is being investigated in the United States (Pinegar et al., 2011) and in Japan as Course 50 (Matsumiya, 2011). Course 50 aims to reduce CO_2 emissions by approximately 30 % by 2050 through capture, separation and recovery. Molten oxide electrolysis (Wang et al., 2011) could reduce emissions if a low or CO_2-free electricity source was available. However this technology is only at the very early stages of development and identifying a suitable anode material has proved difficult.

Material efficiency: Material efficiency offers significant potential for emissions reductions (Allwood et al., 2010) and cost savings (Roy et al., 2013) in the iron and steel sector. Milford et al. (2011) examined the impact of yield losses along the steel supply chain and found that 26 % of global liquid steel is lost as process scrap, so its elimination could have reduced sectoral CO_2 emissions by 16 % in 2008. Cooper et al. (2012) estimate that nearly 30 % of all steel produced in 2008 could be re-used in future. However, in many economies steel is relatively cheap in comparison to labour, and this difference is amplified by tax policy, so economic logic currently drives a preference for material inefficiency to reduce labour costs (Skelton and Allwood, 2013b).

Reduced product and service demand: Commercial buildings in developed economies are currently built with up to twice the steel required by safety codes, and are typically replaced after around 30–60 years (Michaelis and Jackson, 2000; Hatayama et al., 2010; Pauliuk et al., 2012). The same service (e.g., office space provision) could be achieved with one quarter of the steel, if safety codes were met accurately and buildings replaced not as frequently, but after 80 years. Similarly, there is a strong correlation between vehicle fuel consumption and vehicle mass. For example, in the UK, 4- or 5-seater cars are used for an average of around 4 hours per week by 1.6 people (DfT, 2011), so a move towards smaller, lighter fuel efficient vehicles (FEVs), used for more hours per week by more people could lead to a four-fold or more reduction in steel requirements, while providing a similar mobility service. There is a well-known tradeoff between the emissions embodied in producing goods and those generated during use, so product life extension strategies should account for different anticipated rates of improvement in embodied and use-phase emissions (Skelton and Allwood, 2013a).

10.4.2 Cement

Emissions in cement production arise from fuel combustion (to heat limestone, clay, and sand to 1450 °C) and from the calcination reaction. Fuel emissions (0.8 $GtCO_2$ (IEA, 2009d), around 40 % of the total) can be reduced through improvements in energy efficiency and fuel switching while process emissions (the calcination reaction, ~50 % of the total) are unavoidable, so can be reduced only through reduced demand, including through improved material efficiency. The remaining 10 % of CO_2 emissions arise from grinding and transport (Bosoaga et al., 2009).

Energy efficiency: Estimates of theoretical minimum primary energy consumption for thermal (fuel) energy use ranges between 1.6 and 1.85 GJ/t (Locher, 2006). For large new dry kilns, the 'best possible' energy efficiency is 2.7 GJ/t clinker with electricity consumption of 80 kWh/t clinker or lower (Muller and Harnish, 2008). 'International best practice' final energy ranges from 1.8 to 2.1 to 2.9 GJ/t cement and primary energy ranges from 2.15 to 2.5 to 3.4 GJ/t cement for production of blast furnace slag, fly ash, and Portland cement, respectively (Worrell et al., 2008b). Klee et al. (2011) shows that CO_2 emissions intensities have declined in most regions of the world, with a 2009 global average intensity (excluding emissions from the use of alternative fuels) of 633 kg CO_2 per tonne of cementitious product, a decline of 6 % since 2005 and 16 % since 1990. Many options still exist to improve the energy efficiency of cement manufacturing (Muller and Harnish, 2008; Worrell et al., 2008a; Worrell and Galitsky, 2008; APP, 2010).

Emissions efficiency and fuel switching: The majority of cement kilns burn coal (IEA/WBCSD, 2009), but fossil or biomass wastes can also be burned. While these alternatives have a lower CO_2 intensity depending on their exact composition (Sathaye et al., 2011) and can result in reduced overall CO_2 emissions from the cement industry (CEMBUREAU, 2009), their use can also increase overall energy use per tonne of clinker produced if the fuels require pre-treatment such as drying (Hand, 2007). Waste fuels have been used in cement production for the past 20 years in Europe, Japan, the United States, and Canada (GTZ/Holcim, 2006; Genon and Brizio, 2008); the Netherlands and Switzerland use 83 % and 48 % waste, respectively, as a cement fuel (WBCSD, 2005). It is important that wastes are burned in accordance with strict environmental guidelines as emissions resulting from such wastes can cause adverse environmental impacts such as extremely high concentrations of particulates in ambient air, ground-level ozone, acid rain, and water quality deterioration (Karstensen, 2007)[8].

Cement kilns can be fitted to harvest CO_2, which could then be stored, but this has yet to be piloted and "commercial-scale CCS in the cement industry is still far from deployment" (Naranjo et al., 2011). CCS potential in the cement sector has been investigated in several recent studies: IEAGHG, 2008; Barker et al., 2009; Croezen and Korteland, 2010; Bosoaga et al., 2009. A number of emerging technologies aim to reduce emissions and energy use in cement production (Hasanbeigi et al., 2012b), but there are regulatory, supply chain, product confidence and technical barriers to be overcome before such technologies (such as geopolymer cement) could be widely adopted (Van Deventer et al., 2012).

Material efficiency: Almost all cement is used in concrete to construct buildings and infrastructure (van Oss and Padovani, 2002). For concrete, which is formed by mixing cement, water, sand, and aggregates, two applicable material efficiency strategies are: using less cement initially and reusing concrete components at end of first product life (distinct from down-cycling of concrete into aggregate which is widely applied). Less cement can be used by placing concrete only where necessary, for example Orr et al. (2010) use curved fabric moulds to reduce concrete mass by 40 % compared with a standard, prismatic shape. By using higher-strength concrete, less material is needed; CO_2 savings of 40 % have been reported on specific projects using 'ultra-high-strength' concretes (Muller and Harnish, 2008). Portland cement comprises 95 % clinker and 5 % gypsum, but cement can be produced with lower ratios of clinker through use of additives such as blast furnace slag, fly ash from power plants, limestone, and natural or artificial pozzolans. The weighted average clinker-to-cement ratio for the companies participating in the WBCSD GNR project was 76 % in 2009 (WBCSD, 2011). In China, this ratio was 63 % in 2010 (NDRC, 2011a). In India the ratio is 80 % but computer optimization is improving this (India Planning Commission, 2007). Reusing continuous concrete elements is difficult because it requires elements to be broken up but remain undamaged. Concrete blocks can be reused, as masonry blocks and bricks are reused already, but to date there is little published literature in this area.

Reduced product and service demand: Cement, in concrete, is used in the construction of buildings and infrastructure. Reducing demand for these products can be achieved by extending their lifespans or using them more intensely. Buildings and infrastructure have lifetimes less than 80 years—less than 40 years in East Asia—(Hatayama et al., 2010), however their core structural elements (those that drive demand for concrete) could last over 200 years if well maintained. Reduced demand for building and infrastructure services could be achieved by human settlement design, increasing the number of people living and working in each building, or decreasing per-capita demand for utilities (water, electricity, waste), but has as yet had little attention.

10.4.3 Chemicals (plastics/fertilizers/others)

The chemicals industry produces a wide range of different products on scales ranging over several orders of magnitude. This results in methodological and data collection challenges, in contrast to other sectors such as iron and steel or cement (Saygin et al., 2011a). However, emissions in this sector are dominated by a relatively small number of key outputs: ethylene, ammonia, nitric acid, adipic acid and caprolactam used in producing plastics, fertilizer, and synthetic fibres. Emissions arise both from the use of energy in production and from the venting of by-products from the chemical processes. The synthesis of chlorine in chlor-alkali electrolysis is responsible for about 40 % of the electricity demand of the chemical industry.

Energy efficiency: Steam cracking for the production of light olefins, such as ethylene and propylene, is the most energy consuming process in the chemical industry, and the pyrolysis section of steam cracking consumes about 65 % of the total process energy (Ren et al., 2006). Upgrading all steam cracking plants to best practice technology could reduce energy intensity by 23 % (Saygin et al., 2011a; b) with a further 12 % saving possible with best available technology. Switching to a biomass-based route to avoid steam cracking could reduce CO_2 intensity (Ren and Patel, 2009) but at the cost of higher energy use, and with high land-use requirements. Fertilizer production accounts for around 1.2 % of world energy consumption (IFA, 2009), mostly to produce ammonia (NH_3). 22 % energy savings are possible (Saygin et al., 2011b) by upgrading all plants to best practice technology. Nitrous oxide (N_2O) is emitted during production of adipic and nitric acids. By 2020 annual emissions from these industries are estimated to be 125 $MtCO_2eq$ (EPA, 2012a). Many options exist to reduce emissions, depending on plant operating conditions (Reimer et al., 2000). A broad survey of options in the petrochemicals industry is given by Neelis et al. (2008). Plastics recycling saves energy, but to produce a high value recycled material, a relatively pure waste stream is required: impurities greatly degrade the properties of the recycled material. Some plastics can be produced from mixed waste streams, but gen-

8 See also: http://www2.epa.gov/enforcement/cement-manufacturing-enforcement-initiative

erally have a lower value than virgin material. A theoretical estimate suggest that increasing use of combined heat and power plants in the chemical and petrochemical sector from current levels of 10 to 25% up to 100% would result in energy savings up to 2 EJ for the activity level in 2006 (IEA, 2009e).

Emissions efficiency: There are limited opportunities for innovation in the current process of ammonia production via the Haber-Bosch process (Erisman et al., 2008). Possible improvements relate to the introduction of new N_2O emission reduction technologies in nitric acid production such as high-temperature catalytic N_2O decomposition (Melián-Cabrera et al., 2004) which has been shown to reduce N_2O emissions by up to 70–90% (BIS Production Partner, 2012; Yara, 2012). While implementation of this technology has been largely completed in regions pursuing carbon emission reduction (e.g., the EU through the Emissions Trading Scheme (ETS) or China and other developing countries through Clean Development Mechanism (CDM), the implementation of this technology still offers large mitigation potential in other regions like the former Soviet Union and the United States (Kollmus and Lazarus, 2010). Fuel switching can also lead to significant emission reductions and energy savings. For example, natural gas based ammonia production results in 36% emission reductions compared to naphtha, 47% compared to fuel oil and 58% compared to coal. The total potential mitigation arising from this fuel switching would amount to 27 $MtCO_2eq$/year GHG emissions savings (IFA, 2009).

Material efficiency: Many of the material efficiency measures identified above can be applied to the use of plastics, but this has had little attention to date, although Hekkert et al. (2000) anticipate a potential 51% saving in emissions associated with the use of plastic packaging in the Netherlands from application of a number of material efficiency strategies. More efficient use of fertilizer gives benefits both in reduced direct emissions of N_2O from the fertilizer itself and from reduced fertilizer production (Smith et al., 2008).

10.4.4 Pulp and paper

Global paper production has increased steadily during the last three decades (except for a minor production decline associated with the 2008 financial crisis) (FAO, 2013), with global demand expansion currently driven by developing nations. Fuel and energy use are the main sources of GHG emissions during the forestry, pulping, and manufacturing stages of paper production.

Energy efficiency: A broad range of energy efficiency technologies are available for this sector, reviewed by Kramer et al. (2009), and Laurijssen et al. (2012). Over half the energy used in paper making is to create heat for drying paper after it has been laid and Laurijssen et al. (2010) estimate that this could be reduced by ~32% by the use of additives, an increased dew point, and improved heat recovery. Energy savings may also be obtained from emerging technologies (Jacobs and IPST, 2006; Worrell et al., 2008b; Kong et al., 2012) such as black liquor gasification, which uses the by-product of the chemical pulping process to increase the energy efficiency of pulp and paper mills (Naqvi et al., 2010). With commercial maturity expected within the next decade (Eriksson and Harvey, 2004), black liquor gasification can be used as a waste-to-energy method with the potential to achieve higher overall energy efficiency (38% for electricity generation) than the conventional recovery boiler (9–14% efficiency) while generating an energy-rich syngas from the liquor (Naqvi et al., 2010). The syngas can also be utilized as a feedstock for production of renewable motor fuels such as bio-methanol, dimethyl ether, and FT-diesel or hydrogen (Pettersson and Harvey, 2012). Gasification combined cycle systems have potential disadvantages (Kramer et al., 2009), including high energy investments to concentrate sufficient black liquor solids and higher lime kiln and causticizer loads compared to Tomlinson systems. Paper recycling generally saves energy and may reduce emissions (although electricity in some primary paper making is derived from biomass-powered CHP plants) and rates can be increased (Laurijssen et al., 2010b). Paper recycling is also important as competition for biomass will increase with population growth and increased use of biomass for fuel.

Emissions efficiency: Direct CO_2 emissions from European pulp and paper production reduced from 0.57 to 0.34 $ktCO_2$ per kt of paper between 1990 and 2011, while indirect emissions reduced from 0.21 to 0.09 $ktCO_2$ per kt of paper (CEPI, 2012). Combined heat and power (CHP) accounted for 95% of total on-site electricity produced by EU paper makers in 2011, compared to 88% in 1990 (CEPI, 2012), so has little further potential in Europe, but may offer opportunities globally. The global pulp and paper industry usually has ready access to biomass resources and it generates approximately a third of its own energy needs from biomass (IEA, 2009c), 53% in the EU (CEPI, 2012). Paper recycling can have a positive impact on energy intensity and CO_2 emissions over the total lifecycle of paper production (Miner, 2010; Laurijssen et al., 2010). Recycling rates in Europe and North America reached 70% and 67% in 2011, respectively[9] (CEPI, 2012), leaving a small range for improvement when considering the limit of 81% estimated by CEPI (2006). In Europe, the share of recovered paper used in paper manufacturing has increased from roughly 33% in 1991 to around 44% in 2009 (CEPI, 2012). GHG fluxes from forestry are discussed in Section 11.2.3.

Material efficiency: Higher material efficiency could be achieved through increased use of duplex printing, print on demand, improved recycling yields and the manufacturing of lighter paper. Recycling yields could be improved by the design of easy to remove inks and adhesives and less harmful de-inking chemicals; paper weights for newspapers and office paper could be reduced from 45 and 80 g/m^2 to 42 and 70 g/m^2 respectively and might lead to a 37% saving in paper used for current service levels (Van den Reek, 1999; Hekkert et al., 2002).

[9] American Forest and Paper Association, Paper Recycles—Statistics—Paper & Paperboard Recovery http://www.paperrecycles.org/statistics/paper-paperboard-recovery.

Reduced demand: Opportunities to reduce demand for paper products in the future include printing on demand, removing print to allow paper re-use (Leal-Ayala et al., 2012), and substituting e-readers for paper. The latter has been the subject of substantial academic research (e.g., Gard and Keoleian, 2002; Reichart and Hischier, 2003) although the substitution of electronic media for paper has mixed environmental outcomes, with no clear statistics yet on whether such media reduces paper demand, or whether it leads to a net reduction in emissions.

10.4.5 Non-ferrous (aluminium/others)

Annual production of non-ferrous metals is small compared to steel, and is dominated by aluminium, with 56 Mt made globally in 2009, of which 18 Mt was through secondary (recycled) production. Production is expected to rise to 97 Mt by 2020 (IAI, 2009). Magnesium is also significant, but with global primary production of only 653 Kt in 2009 (IMA, 2009), is dwarfed by aluminium.

Energy efficiency: Aluminium production is particularly associated with high electricity demand. Indirect (electricity-related) emissions account for over 80% of total GHG emissions in aluminium production. The sector accounts for 3.5% of global electricity consumption (IEA 2008) and energy accounts for nearly 40% of aluminium production costs.

Aluminium can be made from raw materials (bauxite) or through recycling. Best practice primary aluminium production—from alumina production through ingot casting—consumes 174 GJ/t primary energy (accounting for electricity production, transmission, distribution losses) and 70.6 GJ/t final energy (Worrell et al., 2008b). Best practice for electrolysis—which consumes roughly 85% of the energy used for production of primary aluminium—is about 47 GJ/t final energy while the theoretical energy requirement is 22 GJ/t final energy (BCS Inc., 2007). Best practice for recycled aluminium production is 7.6 GJ/t primary energy and 2.5 GJ/t final energy (Worrell et al., 2008b), although in reality, recycling uses much more energy due to pre-processing of scrap, 'sweetening' with virgin aluminium and downstream processing after casting. The U.S. aluminium industry consumes almost three times the theoretical minimum energy level (BCS Inc., 2007). The options for new process development in aluminium production—multipolar electrolysis cells, inert anodes and carbothermic reactions—have not yet reached commercial scale (IEA, 2012d). The IEA estimates that application of best available technology can reduce energy use for aluminium production by about 10% compared with current levels (IEA, 2012d).

At present, post-consumer scrap makes up only 20% of total aluminium recycling (Cullen and Allwood, 2013), which is dominated by internal 'home' or 'new' scrap (see Figure 10.2). As per capita stock levels saturate in the 21st century, there could be a shift from primary to secondary aluminium production (Liu et al., 2012a) if recycling rates can be increased, and the accumulation of different alloying elements in the scrap stream can be controlled. These challenges will require improved end of life management and even new technologies for separating the different alloys (Liu et al., 2012a).

Emissions efficiency: Data on emissions intensities for a range of non-ferrous metals are given by (Sjardin, 2003). The aluminium industry alone contributed 3% of CO_2 emissions from industry in 2006 (Allwood et al., 2010). In addition to CO_2 emissions resulting from electrode and reductant use, the production of non-ferrous metals can result in the emission of high-global warming potential (GWP) GHGs, for example PFCs (such as CF_4) in aluminium or SF_6 in magnesium. PFCs result from carbon in the anode and fluorine in the cryolite. The reaction can be minimized by controlling the process to prevent a drop in alumina concentrations, which triggers the process[10].

Material efficiency: For aluminium, there are significant carbon abatement opportunities in the area of material efficiency and demand reduction. From liquid aluminium to final product, the yield in forming and fabrication is only 59%, which could be improved by near-net shape casting and blanking and stamping process innovation (Milford et al., 2011). For chip scrap produced from machining operations (in aluminium, for example (Tekkaya et al., 2009), or magnesium (Wu et al., 2010)) extrusion, processes are being developed to bond scrap in the solid state to form a relatively high quality product potentially offering energy savings of up to 95% compared to re-melting. Aluminium building components (window frames, curtain walls, and cladding) could be reused when a building is demolished (Cooper and Allwood, 2012) and more modular product designs would allow longer product lives and an overall reduction in demand for new materials (Cooper et al., 2012).

10.4.6 Food processing

The food industry as discussed in this chapter includes all processing beyond the farm gate, while everything before is in the agriculture industry and discussed in Chapter 11. In the developed world, the emissions released beyond the farm gate are approximately equal to those released before. Garnett (2011) suggests that provision of human food drives around 17.7 $GtCO_2eq$ in total.

Energy efficiency: The three largest uses of energy in the food industry in the United States are animal slaughtering and processing, wet corn milling, and fruit and vegetable preservation, accounting for 19%, 15%, and 14% of total use, respectively (US EIA, 2009). Increased use of heat exchanger networks or heat pumps (Fritzson and Berntsson, 2006; Sakamoto et al., 2011), combined heat and power, mechanical dewatering compared to rotary drying (Masanet et al., 2008), and thermal and mechanical vapour recompression in evaporation further enhanced by use of reverse osmosis can deliver energy use efficiency. Many of these technologies could also be used in cooking and drying in other parts of the food industry. Savings in energy for refrigeration

[10] http://www.aluminum.org/Content/NavigationMenu/TheIndustry/Environment/ReducingPFCEmissionsintheAluminumIndustry/default.html.

could be made with better insulation and reduced ventilation in fridges and freezers. Dairy processing is also among the most energy- and carbon-intensive activities within the global food production industry, with estimated annual emissions of over 128 $MtCO_2$ (Xu and Flapper, 2009, 2011). Within dairy processing, cheese production is the most energy intensive sector (Xu et al., 2009). Ramirez and Block (2006) report that EU dairy operations, having improved in the 1980s and 1990s, are now reaching a plateau of energy intensity, but Brush et al. (2011) provide a survey of best practice opportunities for energy efficiency in dairy operations.

Emissions efficiency: The most cost effective reduction in CO_2 emissions from food production is by switching from heavy fuel oil to natural gas. Other ways of improving emissions efficiency involve using lower-emission modes of transport (Garnett, 2011). In transporting food, there is a tradeoff between local sourcing and producing the food in areas where there are other environmental benefits (Sim et al., 2007; Edwards-Jones et al., 2008). Landfill emissions associated with food waste could be reduced by use of anaerobic digestion processes (Woods et al., 2010).

Demand reduction: Overall demand for food could be reduced without sacrificing well-being (GEA, 2012). Up to one-third of food produced for human consumption is wasted in either in the production/retailing stage, or by consumers (Gunders (2012) estimates 40 % waste in the United States). Gustavsonn et al. (2011) suggest that, in developed countries, consumer behaviour could be changed, and 'best-before-dates' reviewed. Increasing cooling demand, the globalization of the food system with corresponding transport distances, and the growing importance of processed convenience food are also important drivers (GEA, 2012). Globally, approximately 1.5 billion out of 5 billion people over the age of 20 are overweight and 500 million are obese (Beddington et al., 2011). Demand for high-emission food such as meat and dairy products could be replaced by demand for other, lower-emission foods. Meat and dairy products contribute to half of the emissions from food (when the emissions from the up-stream processes are included) according to Garnett (2009), while Stehfest et al. (2009) puts the figure at 18 % of global GHG emissions, and Wirsenius (2003) estimates that two-thirds of food-related phytomass is consumed by animals, which provide just 13 % of the gross energy of human diets. Furthermore, demand is set to double by 2050, as developing nations grow wealthier and eat more meat and dairy foods (Stehfest et al., 2009; Garnett, 2009). In order to maintain a constant total demand for meat and dairy, Garnett (2009) suggests that by 2050 average per capita consumption should be around 0.5 kg meat and 1 litre of milk per week, which is around the current averages in the developing world today.

10.4.7 Textiles and leather

In 2009, textiles and leather manufacturing consumed 2.15 EJ final energy globally. Global consumption is dominated by Asia, which was responsible for 65 % of total world energy use for textiles and leather manufacturing in 2009. In the United States, about 45 % of the final energy used for textile mills is natural gas, about 35 % is net electricity (site), and 14 % coal (US EIA, 2009). In China, final energy consumption for textiles production is dominated by coal (39 %) and site electricity (38 %) (NBS, 2012). In the US textile industry, motor driven systems and steam systems dominate energy end uses. Around 36 % of the energy input to the US textile industry is lost onsite, with motor driven systems responsible for 13 %, followed by energy distribution and boiler losses of 8 % and 7 %, respectively (US DoE, 2004b).

Energy and emissions efficiency: Numerous energy efficiency technologies and measures exist that are applicable to the textile industry (CIPEC, 2007; Hasanbeigi and Price, 2012). For Taiwan, Province of China, Hong et al. (2010) report energy savings of about 1 % in textile industry following the adoption of energy-saving measures in 303 firms (less than 10 % of the total number of local textile firms in 2005) (Chen Chiu, 2009). In India, CO_2 emissions reductions of at least 13 % were calculated based on implementation of operations and maintenance improvements, fuel switching, and adoption of five energy-efficient technologies (Velavan et al., 2009).

Demand reduction: see Box 10.2.

10.4.8 Mining

Energy efficiency: The energy requirements of mining are dominated by grinding (comminution) and the use of diesel-powered material handling equipment (US DoE, 2007; Haque and Norgate, 2013). The major area of energy usage—up to 40 % of the total—is in electricity for comminution (Smith, 2012). Underground mining requires more energy than surface mining due to greater requirements for hauling, ventilation, water pumping, and other operations (US DoE, 2007). Strategies for GHG mitigation are diverse. An overall scheme to reduce energy consumption is the implementation of strategies that upgrade the ore body concentration before crushing and grinding, through resource characterization by geo-metallurgical data and methods (Bye, 2005, 2007, 2011; CRC ORE, 2011; Smith, 2012). Selective blast design, combined with ore sorting and gangue rejection, significantly improve the grade of ore being fed to the crusher and grinding mill, by as much as 2.5 fold. This leads to large reductions of energy usage compared to business-as-usual (CRC ORE, 2011; Smith, 2012).

There is also a significant potential to save energy in comminution through the following options: more crushing, less grinding, using more energy-efficient crushing technologies, removing minerals and gangue from the crushing stage, optimizing the particle size feed for grinding mills from crushing mills, selecting target product size(s) at each stage of the circuit, using advanced flexible comminution circuits, using more efficient grinding equipment, and by improving the design of new comminution equipment (Smith, 2012).

Other important energy savings opportunities are in the following areas: a) separation processes—mixers, agitators and froth flotation cells, b) drying and dewatering in mineral processing, c) materials movement, d) air ventilation and conditioning opportunities, e) processing site energy demand management and waste heat recovery options, f) technology specific for lighting, motors, pumps and fans and air compressor systems, and g) improvement in energy efficiency of product transport from mine site to port (Rathmann, 2007; Raaz and Mentges, 2009; Daniel et al., 2010; Norgate and Haque, 2010; DRET, 2011; Smith, 2012).

Recycling represents an important source of world's metal supply and it can be increased as a means of waste reduction (see Section 10.14) and thus energy saving in metals production. In recent years, around 36 % of world's gold supply was from recycled scrap (WGC, 2011), 25 % of silver (SI and GFMS, 2013), and 35 % of copper (ICSG, 2012).

Emissions efficiency: Substitution of onsite fossil fuel electricity generators with renewable energy is an important mitigation strategy. Cost effectiveness depends on the characteristics of each site (Evans & Peck, 2011; Smith, 2012).

Material efficiency: In the extraction of metal ores, one of the greatest challenges for energy efficiency enhancement is that of the recovery ratio, which refers to the percentage of valuable ore within the total mine material. Lower grades inevitably require greater amounts of material to be moved per unit of product. The recovery ratio for metals averages about 4.5 % (US DoE, 2007). The 'grade' of recyclable materials is often greater than the one of ores being currently mined; for this reason, advancing recycling for mineral commodities would bring improvements in the overall energy efficiency (IIED, 2002).

10.5 Infrastructure and systemic perspectives

Improved understanding of interactions among different industries, and between industry and other economic sectors, is becoming more important in a mitigation and sustainable development context. Strategies adopted in other sectors may lead to increased (or decreased) emissions from the industry sector. Collaborative activities within and across the sector may enhance the outcome of climate change mitigation. Initiatives to adopt a system-wide view face a barrier as currently practiced system boundaries often pose a challenge. A systemic approach can be at different levels, namely, at the micro-level (within a single company, such as process integration and cleaner production), the meso-level (between three or more companies, such as eco-industrial parks) and the macro-level (cross-sectoral cooperation, such as urban symbiosis or regional eco-industrial network). Systemic collaborative activities can reduce the total consumption of materials and energy and can contribute to the reduction of GHG emissions. The rest of this section focuses mainly on the meso- and macro-levels as micro-level options have already been covered in Section 10.4.

10.5.1 Industrial clusters and parks (meso-level)

Small and medium enterprises (SMEs) often suffer not only from difficulties arising due to their size and lack of access to information, but also from being isolated while in operation (Sengenberger and Pyke, 1992). Clustering of SMEs usually in the form of industrial parks can facilitate growth and competitiveness (Schmitz, 1995). In terms of implementation of mitigation options, SMEs in clusters/parks can benefit from by-products exchange (including waste heat) and infrastructure sharing, as well as joint purchase (e.g., of energy efficient technologies). Cooperation in eco-industrial parks (EIPs) reduces the cumulative environmental impact of the whole industrial park (Geng and Doberstein, 2008). Such an initiative reduces the total consumption of virgin materials and final waste and improves the efficiency of companies and their competitiveness. Since the extraction and transformation of virgin materials is usually energy intensive, EIP efforts can abate industrial GHG emissions. For example, in order to encourage target-oriented cooperation, Chinese 'eco-industrial park standards' contain quantitative indicators for material reduction and recycling, as well as pollution control (Geng et al., 2009). Two pioneering eco-industrial parks in China achieved over 80 % solid waste reuse ratio and over 82 % industrial water reuse ratio during 2002–2005 (Geng et al., 2008). The Japanese eco-town project in Kawasaki achieved substitution of 513,000 tonnes of raw material, resulting in the avoidance of 1 % of the current total landfill in Japan during 1997–2006 (van Berkel et al., 2009).

In order to encourage industrial symbiosis[11] at the industrial cluster level, different kinds of technical infrastructure (e.g., pipelines) as well as non-technical infrastructure (e.g., information exchange platforms) are necessary so that both material and energy use can be optimized (Côté and Hall, 1995). Although additional investment for infrastructure building is unavoidable, such an investment can bring both economic and environmental benefits. In India there have been several instances where the government has taken proactive approaches to provide land and infrastructure, access to water, non-conventional (MSW-based) power to private sector industries (such as chemicals, textile, paper, pharmaceutical companies, cement) operating in clusters (IBEF, 2013). A case study in the Tianjin Economic Development Area in northern China indicates that the application of an integrated water optimization model (e.g., reuse of treated wastewater by other firms) can reduce the total water related costs by 10.4 %, fresh water consumption by 16.9 % and wastewater discharge by 45.6 % (Geng et al., 2007). As an additional consequence, due to the strong energy-water nexus, energy use and release of GHG emissions related to fresh water provision or wastewater treatment can be reduced.

[11] Note that industrial symbiosis is further covered in Chapter 4 (Sustainable Development and Equity), Section 4.4.3.3

10.5.2 Cross-sectoral cooperation (macro-level)

Besides inter-industry cooperation, opportunities arise from the geographic proximity of urban and industrial areas, leading to transfer of urban refuse as a resource to industrial applications, and vice versa (Geng et al., 2010a). For instance, the cement industry can accept as their inputs not only virgin materials such as limestone and coal, but also various wastes/industrial by-products (see Section 10.4), thus contributing up to 15–20 % CO_2 emission reduction (Morimoto et al., 2006; Hashimoto et al., 2010). In Northern Europe (e.g., Sweden, Finland, and Denmark), for example, both exhaust heat from industries and heat generated from burning municipal wastes are supplied to local municipal users through district heating (Holmgren and Gebremedhin, 2004). Industrial waste can also be used to reduce conventional fuel demand in other sectors. For example, the European bio-DME project[12] aims to supply heavy-duty trucks and industry with dimethyl-ether fuel made from black liquor produced by the pulp industry. However, careful design of regional recycling networks has to be undertaken because different types of waste have different characteristics and optimal collection and recycling boundaries and therefore need different infrastructure support (Chen et al., 2012).

The reuse of materials recovered from urban infrastructures can reduce the demand for primary products (e.g., ore) and thus contribute to climate change mitigation in extractive industries (Klinglmair and Fellner, 2010). So far, reuse of specific materials is only partly established and the potential for future urban mining is growing as the urban stock of materials still increases. While in the 2011 fiscal year in Japan only 5.79 Mt of steel scrap came from the building sector, 13.6 Mt were consumed by the building sector. In total, urban stock of steel is estimated to be 1.33 Gt in Japan where the total annual crude steel production was 0.106 Gt (NSSMC, 2013).

10.5.3 Cross-sectoral implications of mitigation efforts

Currently much attention is focused on improving energy efficiency within the industry sector (Yeo and Gabbai, 2011). However, many mitigation strategies adopted in other sectors significantly affect activities of the industrial sector and industry-related GHG emissions. For example, consumer preference for lightweight cars can incentivize material substitution for car manufacturing (e.g., potential lightweight materials: see Chapter 8), growing demand for rechargeable vehicle batteries (see Chapter 8) and the demand for new materials (e.g., innovative building structures or thermal insulation for buildings: see Chapter 9; high-temperature steel demand by power plants: see Chapter 7). These materials or products consume energy at the time of manufacturing, so changes outside the industry sector that lead to changes in demand for energy-saving products within the industry sector can be observed over a long period of time (ICCA, 2009). Thus, for a careful assessment of mitigation options, a lifecycle perspective is needed so that a holistic emission picture (including embodied emissions) can be presented. For instance, the increase in GHG emissions from increased aluminium production could under specific circumstances be larger than the GHG savings from vehicle weight reduction (Geyer, 2008). Kim et al. (2010) have, however, indicated that in about two decades, closed-loop recycling can significantly reduce the impacts of aluminium-intensive vehicles.

Increasing demand on end-use related mitigation technologies could contribute to potential material shortages. Moss et al. (2011) examined market and political risks for 14 metals that are used in significant quantities in the technologies of the EU's Strategic Energy Technology Plan (SET Plan) so that metal requirements and associated bottlenecks in green technologies, such as electric vehicles, low-carbon lighting, electricity storage and fuel cells and hydrogen, can be recognized.

Following a systemic perspective enables the identification of unexpected outcomes and even potential conflicts between different targets when implementing mitigation options. For example, the quality of many recycled metals is maintained solely through the addition of pure primary materials (Verhoef et al., 2004), thus perpetuating the use of these materials and creating a challenge for the set up of closed loop recycling (e.g., automotive aluminium; Kim et al., 2011). Additionally, due to product retention (the period of use) and growing demand, secondary materials needed for recycling are limited.

10.6 Climate change feedback and interaction with adaptation

There is currently a distinct lack of knowledge on how climate change feedbacks may impact mitigation options and potentials as well as costs in industry[13].

Insights into potential synergy effects (how adaptation options could reduce emissions in industry) or tradeoffs (how adaptation options could lead to additional emissions in industry) are also lacking. However, it can be expected that many adaptation options will generate additional industrial product demand and will lead to additional emissions in the sector. Improving flood defence, for example, in response to sea level rise may lead to a growing demand

[12] Production of DME from biomass and utilization of fuel for transport and industrial use. Project website at: http://www.biodme.eu.

[13] There is limited literature on the impacts of climate change on industry (e.g., availability of water for the food industry and in general for cooling and processing in many different industries), and these are dealt within WG 2 of AR 5, Chapter 10.

for materials for embankment and similar infrastructure. Manufacturers of textile products, machinery for agriculture or construction, and heating/cooling equipment may be affected by changing product requirements in both number and quality due to climate change. There is as yet no comprehensive assessment of these effects, nor any estimate on market effects resulting from changes in demand for products.

10.7 Costs and potentials

The six main categories of mitigation options discussed in Section 10.4 for manufacturing industries can deliver GHG emission reduction benefits at varying levels and at varying costs over varying time periods across subsectors and countries. There is not much comparable, comprehensive, detailed quantitative information and literature on costs and potentials associated with each of the mitigation options. Available mitigation potential assessments (e.g., UNIDO, 2011; IEA, 2012d) are not always supplemented by cost estimates. Also, available cost estimates (e.g., McKinsey&Company, 2009; Akashi et al., 2011) are not always comparable across studies due to differences in the treatment of costs and energy price estimates across regions. There are many mitigation potential assessments for individual industries (examples are included in Section 10.4) with varying time horizons; some studies report the mitigation potential of energy efficiency measures with associated initial investment costs which do not account for the full life time energy cost savings benefits of investments, while other studies report marginal abatement costs (MACs) based on selected technological options. Many sector- or system-specific mitigation potential studies use the concept of cost of conserved energy (CCE) that accounts for annualized initial investment costs, operation and maintenance (O&M) costs, and energy savings using either social or private discount rates (Hasanbeigi et al., 2010b). Those mitigation options with a CCE below the unit cost of energy are referred to as 'cost-effective'. Some studies (e.g., McKinsey&Company, 2009) identify 'negative abatement costs' by including the energy cost savings in the abatement cost calculation.

The sections below provide an assessment of option-specific potential and associated cost estimates using information available in the literature (including underlying databases used by some of such studies) and expert judgement (see Annex III, Technology-specific cost and performance parameters) and distinguish mitigation of CO_2 and non-CO_2 emissions. Generally, the assessment of costs is relatively more uncertain but some indicative results convey information about the wide cost range (costs per tonne of CO_2 reduction) within which various options can deliver GHG reduction benefit. The inclusion of additional multiple benefits of mitigation measures might change the cost-effectiveness of a technology completely, but are not included in this section. Co-benefits are discussed in Section 10.8.

10.7.1 CO_2 emissions

Quantitative assessments of CO_2 emission reduction potential for the industrial sector explored in this section are mainly based on: (1) studies with a global scope (e.g., IEA, UNIDO), (2) MAC studies and (3) various information sources on available technology at industrial units along with plant level and country specific data. IEA estimates a global mitigation potential for the overall industry sector of 5.5 to 7.5 $GtCO_2$ for the year 2050 (IEA, 2012d)[14]. The IEA report (2012d) shows a range of 50 % reduction in four key sectors (iron and steel, cement, chemicals, and paper) and in the range of 20 % for the aluminium sector. From a regional perspective, China and India comprise 44 % of this potential. In terms of how different options contribute to industry mitigation potential, with regard to CO_2 emissions reduction compared with 2007 values, the IEA (2009c) shows implementation of end use fuel efficiency can achieve 40 %, fuel and feedstock switching can achieve 21 %, recycling and energy recovery can achieve 9 %, and CCS can achieve 30 %. McKinsey (2009) provides a global mitigation potential estimate for the overall industry sector of 6.9 $GtCO_2$ for 2030. The potential is found to be the largest for iron and steel, followed by chemicals and cement at 2.4, 1.9 and 1.0 $GtCO_2$ for the year 2030, respectively (McKinsey&Company, 2010). The United Nations Industrial Development Organization (UNIDO) analyzed the potential of energy savings based on universal application of best available technologies. All the potential mitigation values are higher in developing countries (30 to 35 %) compared with developed countries (15 %) (UNIDO, 2011).

Other studies addressing the industrial sector as a whole found potential for future improvements in energy intensity of industrial production to be in the range of up to 25 % of current global industrial final energy consumption per unit output (Schäfer, 2005; Allwood et al., 2010; UNIDO, 2011; Saygin et al., 2011b; Gutowski et al., 2013) (see Section 10.4). Additional savings can be realized in the future through adoption of emerging technologies currently under development or that have not yet been fully commercialized (Kong et al., 2012; Hasanbeigi et al., 2012b, 2013a). Examples of industries from India show that specific energy consumption is steadily declining in all energy intensive sectors (Roy et al., 2013), and a wide variety of measures at varying costs have been adopted by the energy intensive industries (Figure 10.6). However, all sectors still have energy savings potential when compared to world best practice (Dasgupta et al., 2012).

Bottom-up country analyses provide energy savings estimates for specific industrial sub-sectors based on individual energy efficiency technologies and measures. Because results vary among studies, these estimates should not be considered as the upper bound of energy saving potential but rather should give an orientation about the general possibilities.

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[14] Expressed here in the form of a deployment potential (difference between the 6 °C and 2 °C scenarios, 6DS and 2DS) rather than the technical potential.

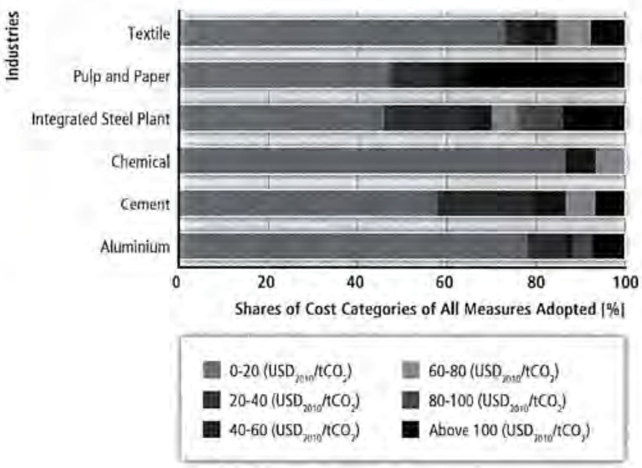

Figure 10.6 | Range of unit cost of avoided CO_2 emissions (USD_{2010}/tCO_2) in India. Source: Database of energy efficiency measures adopted by the winners of the National Awards on Energy Conservations for aluminium (26 measures), cement (42), chemicals (62), ISP: integrated steel plant (30), pulp and paper (46), and textile (75) industry in India during the period of 2007–2012 (BEE, 2012).

In the cement sector, global weighted average thermal energy intensity could drop to 3.2 GJ/t clinker and electric energy intensity to 90 kWh/t cement by 2050 (IEA/WBCSD, 2009). Emissions of 510 $MtCO_2$ would be saved if all current cement kilns used best available technology and increased use of clinker substitutes (IEA, 2009c). Oda et al. (2012) found large differences in regional thermal energy consumption for cement manufacture, with the least efficient region consuming 75 % more energy than the best in 2005. Even though processing alternative fuels requires additional electricity consumption (Oda et al., 2012), their use could reduce cement sector emissions by 0.16 $GtCO_2eq$ per year by 2030 (Vattenfall, 2007) although increasing costs may in due course limit uptake (IEA/WBCSD, 2009). Implementing commercial-scale CCS in the cement industry could contribute to climate change mitigation, but would increase cement production costs by 40–90 % (IEAGHG, 2008). From the cumulative energy savings potential for China's cement industry (2010 to 2030), 90 % is assessed as cost-effective using a discount rate of 15 % (Hasanbeigi et al., 2012a). Electricity and fuel savings of 6 and 1.5 times the total electricity and fuel use in the Indian cement industry in 2010, respectively, can be realized for the period 2010–2030, almost all of which is assessed as cost-effective using a discount rate of 15 % (Morrow III et al., 2013a). About 50 % of the electricity used by Thailand's cement industry in 2005 could have been saved (16 % cost-effectively), while about 20 % of the fuel use could have been reduced (80 % cost-effectively using a discount rate of 30 %) (Hasanbeigi et al., 2010a, 2011). Some subnational level information also shows negative CO_2 abatement costs associated with emissions reductions in the cement sector (e.g., CCAP, 2005).

Nearly 60 % of the estimated electricity savings and all of the fuel savings of the Chinese steel industry for the period 2010–2030 can be realized cost-effectively using a discount rate of 15 % (Hasanbeigi et al., 2013c). Total technical primary energy savings potential of the Indian steel industry from 2010–2030 is equal to around 87 % of total primary Indian steel industry energy use in 2007, of which 91 % of the electricity savings and 64 % of the fuel savings can be achieved cost-effectively using a discount rate of 15 % (Morrow III et al., 2013b). Akashi et al. (2011) indicate that the largest potential for CO_2 emissions savings for some energy-intensive industries remains in China and India. They also indicate that with associated costs under 100 USD/tCO_2 in 2030, the use of efficient blast furnaces in the steel industry in China and India can reduce total emissions by 186 $MtCO_2$ and 165 $MtCO_2$, respectively. This represents a combined total of 75 % of the global CO_2 emissions reduction potential for this technology.

Total technical electricity and fuel savings potential for China's pulp and paper industry in 2010 are estimated to be 4.3 % and 38 %, respectively. All of the electricity and 70 % of the fuel savings can be realized cost-effectively using a discount rate of 30 % (Kong et al., 2013). Fleiter et al. (2012a) found energy saving potentials for the German pulp and paper industry of 21 % and 16 % of fuel and electricity demand in 2035, respectively. The savings result in 3 $MtCO_2$ emissions reduction with two-thirds of this having negative private abatement cost (Fleiter et al., 2012a). Zafeiris (2010) estimates energy saving potential of 6.2 % of the global energy demand of the pulp and paper industry in year 2030. More than 90 % of the estimated savings potential can be realized at negative cost using a discount rate of 30 % (Zafeiris, 2010). The energy intensity of the European pulp and paper industry reduced from 16 to 13.5 GJ per tonne of paper between 1990 and 2008 (Allwood et al., 2012, p. 318; CEPI, 2012). However, energy intensity of the European pulp and paper industry has now stabilized, and few significant future efficiency improvements are forecasted.

In non-ferrous production (aluminium/others), energy accounts for nearly 40 % of aluminium production costs. The IEA forecasts a maximum possible 12 % future saving in energy requirements by future efficiencies. In food processing, reductions between 5 % and 35 % of total CO_2 emissions can be made by investing in increased heat exchanger networks or heat pumps (Fritzson and Berntsson, 2006). Combined heat and power can reduce energy demand by 20–30 %. Around 83 % of the energy used in wet corn milling is for dewatering, drying, and evaporation processes (Galitsky et al., 2003), while 60 % of that used in fruit and vegetable processing is in boilers (Masanet et al., 2008). Thermal and mechanical vapour recompression in drying allows for estimated 15–20 % total energy savings, which could be increased further by use of reverse osmosis (Galitsky et al., 2003). Cullen et al. (2011) suggest that about 88 % savings in energy for refrigeration could be made with better insulation, and reduced ventilation in refrigerators and freezers.

There is very little data available on mineral extractive industries in general. Some analyses reveal that investments in state-of-the-art equipment and further research could reduce energy consumption by almost 50 % (SWEEP, 2011; US DoE, 2007).

Allwood et al. (2010) assessed different strategies to achieve a 50% cut in the emissions of five sectors (cement, steel, paper, aluminium, and plastics) assuming doubling of demand by 2050. They found that gains in efficiency could result in emissions intensity reductions in the range of 21%–40%. Further reductions to reach the required 75% reduction in emissions intensity can only be achieved by implementing strategies at least partly going beyond the sectors boundaries: i.e., non destructive recycling, reducing demand through light weighting, product life extension, increasing intensity of product use or substitution for other materials, and radical process innovations, notwithstanding significant implementation barriers (see Section 10.9).

Mitigation options can also be analyzed from the perspective of some industry-wide technologies. Around two-thirds of electricity consumption in the industrial sector is used to drive motors (McKane and Hasanbeigi, 2011). Steam generation represents 30% of global final industrial energy use. Efficiency of motor systems and steam systems can be improved by 20–25% and 10%, respectively (GEA, 2012; Brown et al., 2012). Improvements in the design and especially the operation of motor systems, which include motors and associated system components in compressed air, pumping, and fan systems (McKane and Hasanbeigi, 2010, 2011; Saidur, 2010), have the potential to save 2.58 EJ in final energy use globally (IEA, 2007). McKane and Hasanbeigi (2011) developed energy efficiency supply curve models for the United States, Canada, the European Union, Thailand, Vietnam, and Brazil and found that the cost-effective potential for electricity savings in motor system energy use compared to the base year varied between 27% and 49% for pumping, 21% and 47% for compressed air, and 14% and 46% for fan systems. The total technical saving potential varied between 43% and 57% for pumping, 29% and 56% for compressed air, and 27% and 46% for fan systems. Ways to reduce emissions from many industries include more efficient operation of process heating systems (LBNL and RDC, 2007; Hasanuzzaman et al., 2012) and steam systems (NREL et al., 2012), minimized waste heat loss and waste heat recovery (US DoE, 2004a, 2008), advanced cooling systems, use of cogeneration (or combined heat and power) (Oland, 2004; Shipley et al., 2008; Brown et al., 2013), and use of renewable energy sources. Recent analysis show, for example, that recuperators can reduce furnace energy use by 25% while economizers can reduce boiler energy use by 10% to 20%, both with payback periods typically under two years (Hasanuzzaman et al., 2012).

According to data from McKinsey (2010) on MACs for cement, iron, and steel and chemical sectors, and from Akashi et al. (2011) for cement and iron and steel, around 40% mitigation potential in industry can be realized cost-effectively. Due to methodological reasons, MACs always have to be discussed with caution. It has to be considered that the information about the direct additional cost associated with additional reduction of CO_2 through technological options is limited. Moreover, system perspectives and system interdependencies are not typically taken into account for MACs (McKinsey&Company, 2010; Akashi et al., 2011).

Unless barriers to mitigation in industry are resolved, the pace and extent of mitigation in industry will be limited, and even cost-effective measures will remain untapped. Various barriers that block technology adoption despite low direct costs are often not appropriately accounted for in mitigation cost assessments. Such barriers are discussed in Section 10.9.

In the long term, however, it may be more relevant to look at radically new ways of producing energy-intensive products. Low-carbon cement and concrete might become relevant (Hasanbeigi et al., 2012b); however, from current perspective cost assessments for these technologies are connected with high uncertainties.

10.7.2 Non-CO_2 emissions

Emissions of non-CO_2 gases from different industrial sources are projected to be 0.70 $GtCO_2eq$ in the year 2030 (EPA, 2013), dominated by HFC-23 from HCFC-22 production (46%) and N_2O from nitric acid and from adipic acid (24%). In 2030, it is projected that HFC-23 emissions will be related mainly to the production of HCFC-22 for feedstock use, as its use as refrigerant will be phased out in 2035 (Miller and Kuijpers, 2011). The EPA (2013) provides MACs for all non-CO_2 emissions. Emissions resulting from the production of flat panel displays and from photovoltaic cell manufacturing are projected to be small (2 and 12 $MtCO_2eq$ respectively in 2030), but particularly uncertain due to limited information on emissions rates, use of fluorinated gases, and production growth rates.

10.7.3 Summary results on costs and potentials

Based on the available bottom-up information from literature and through expert consultation, a global picture of the four industrial key sub-sectors (cement, steel, chemicals, and pulp and paper) is assessed and presented in Figures 10.7 to 10.10 below. Detailed justification of the figures and description of the options are provided in Annex III. Globally, in 2010, these four selected sub-sectors contributed 5.3 $GtCO_2$ direct energy- and process-related CO_2 emissions (see Section 10.3): iron and steel 1.9 $GtCO_2$, non-metallic minerals (which includes cement) 2.6 $GtCO_2$, chemicals and petrochemicals 0.6 $GtCO_2$, and pulp and paper 0.2 $GtCO_2$. This amounts to 73% of all direct[15] energy- and process-related CO_2 emissions from the industry sector.

For each of the sub-sectors, only selected mitigation options are covered (for other feasible options in the industry sector refer to Section 10.4): energy efficiency, shift in raw material use to less carbon-intensive alternatives (e.g., reducing the clinker to cement ratio, recycling etc.), fuel mix options, end-of-pipe emission abatement options such

15 These values do not include indirect emissions from electricity and heat production.

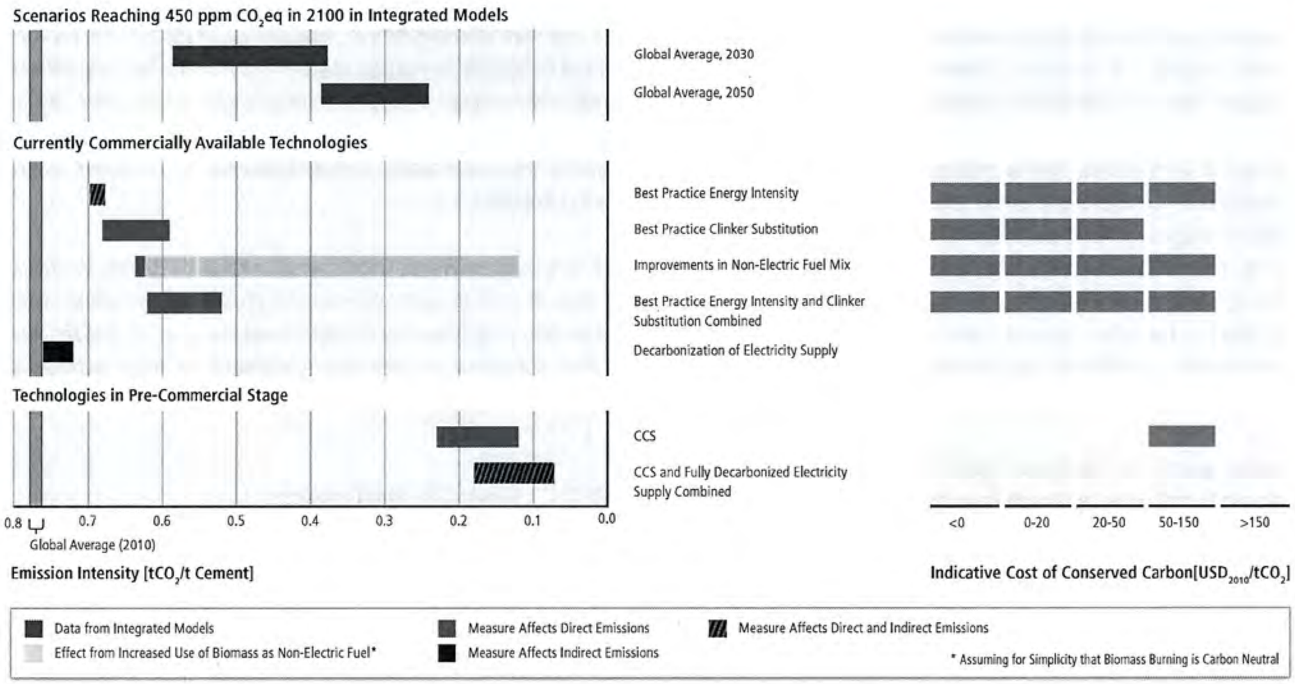

Figure 10.7 | Indicative CO_2 emission intensities and levelized cost of conserved carbon in cement production for various production practices/technologies and in 450 ppm scenarios of selected models (AIM, DNE21+, IEA ETP 2DS) (for data and methodology, see Annex III).

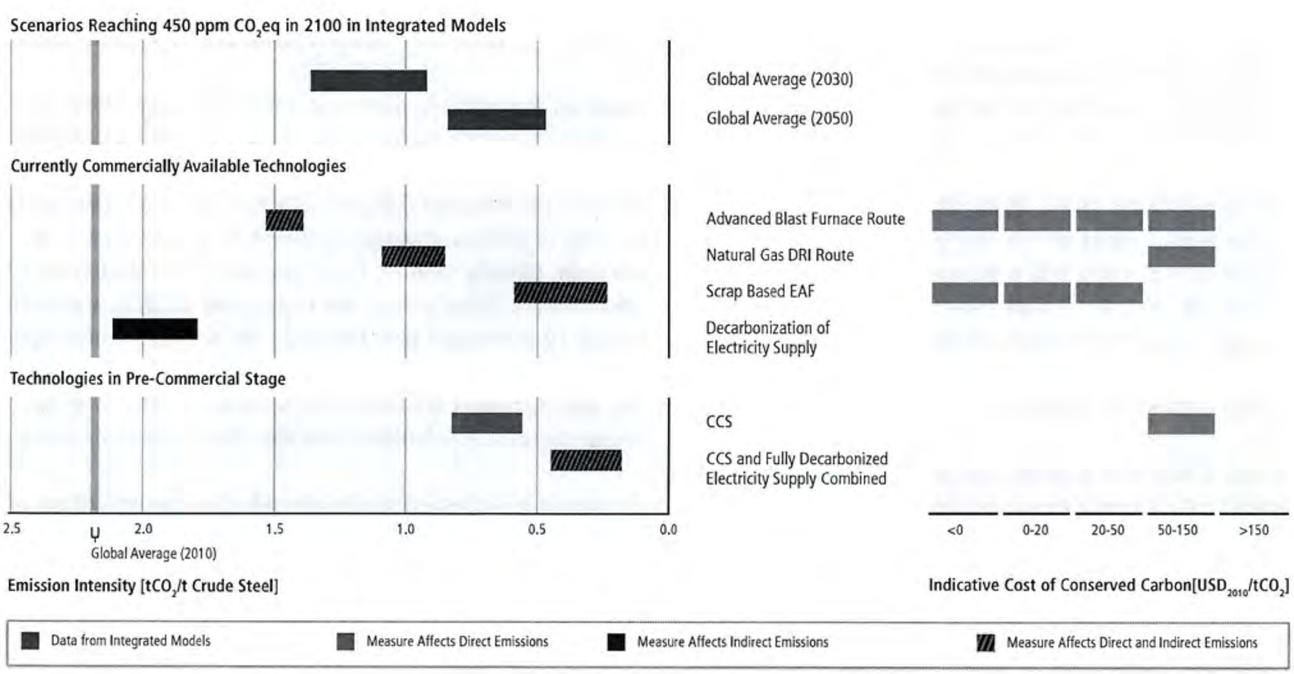

Figure 10.8 | Indicative CO_2 emission intensities and levelized cost of conserved carbon in steel production for various production practices/technologies and in 450 ppm scenarios of selected models (AIM, DNE21+, and IEA ETP 2DS) (for data and methodology, see Annex III).

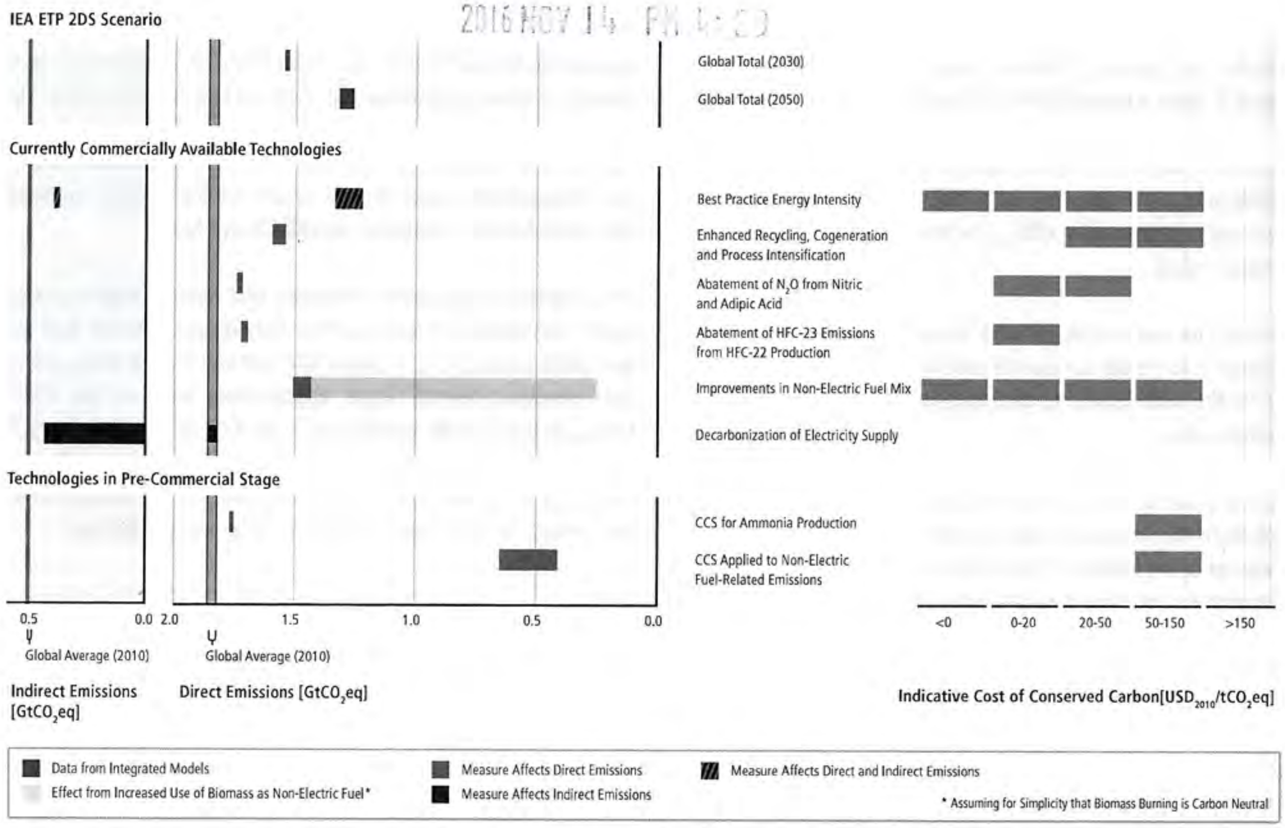

Figure 10.9 | Indicative global indirect (left) and direct (right) CO_2eq emissions and levelized cost of conserved carbon resulting from chemicals production for various production practices/technologies and CO_2 emissions in IEA ETP 2DS scenario (for data and methodology, see Annex III).

Notes: Graph includes energy-related emissions (including process emissions from ammonia production), N_2O emissions from nitric and adipic acid production and HFC-23 emissions from HFC-22 production. Costs for N_2O abatement from nitric/adipic acid production and for HFC-23 abatement in HFC-22 production based on EPA (2013) and Miller and Kuijpers (2011), respectively.

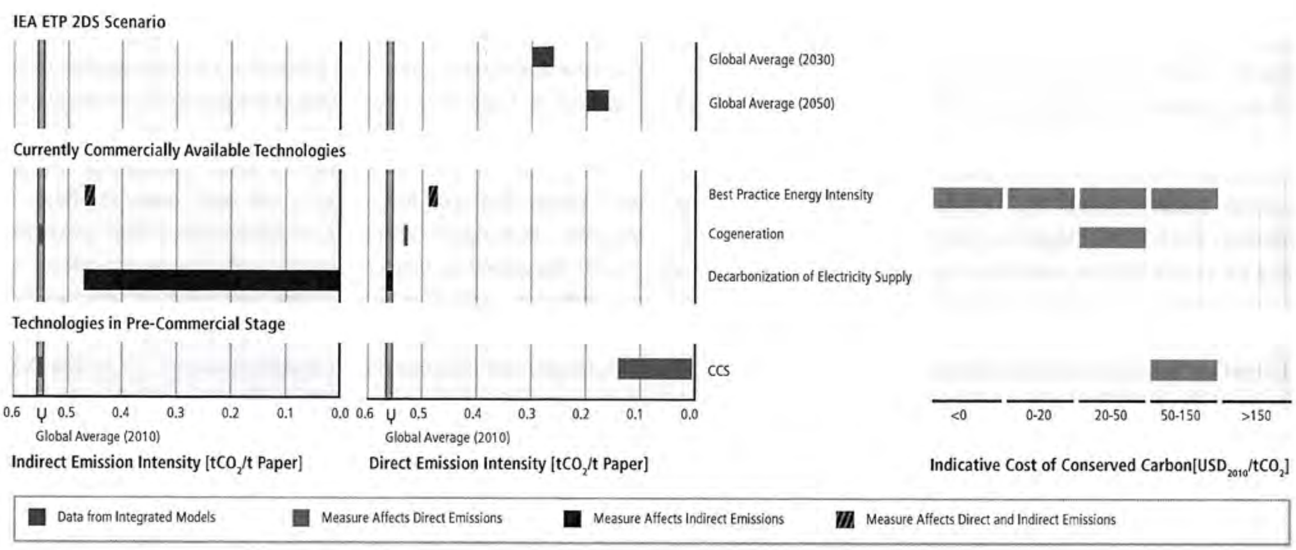

Figure 10.10 | Indicative global indirect (left) and direct (right) CO_2 emission intensities and levelized cost of conserved carbon in paper production for various production practices/technologies and in IEA ETP 2DS scenario (for data and methodology, see Annex III).

as carbon dioxide capture and storage (CCS), use of decarbonized electricity and options for the two most important current sources of non-CO_2 GHG emissions (HFC 23 emissions from HFC 22 production and N_2O emissions from nitric and adipic acid production) in the chemical industry. The potentials are given related to the 2010 emission intensity or absolute emissions. Cost estimates relate to the current costs (expressed in USD_{2010}) of the abatement options unless otherwise stated.

Potentials and costs to decarbonize the electricity sector are covered in Chapter 7. To ensure consistency with that chapter, no estimates are given for the costs related to decarbonizing the electricity mix for the industrial sector.

Costs and potentials are global averages, but based on region-specific information. The technology options are given relative to the global average emission intensity. Some options are not mutually exclusive and potentials can therefore not always be added. As such, none of the individual options can yield full GHG emission abatement, because of the multiple emission sources included (e.g., in the chemical sector CCS and fuel mix improvements cannot reduce N_2O emissions).

Costs relate to costs of abatement taking into account total incremental operational and capital costs. The figures give indicatively the costs of implementing different options; they also exclude options related to material efficiency (e.g., reduction of demand), but include some recycling options (although not in pulp and paper). Figure 10.7 about cement production includes process CO_2 emissions.

Emissions after implementing potential options to reduce the GHG emission intensity of cement, steel, pulp and paper sectors are presented in tCO_2/t product compared to 2010 global average respectively. Future relevant scenarios are also presented. However, for the chemical sector, due to its heterogeneity in terms of products and processes, the information is presented in terms of total emissions. This can be an under-representation of relatively higher mitigation potential in e.g., ammonia production. In addition, unknown/unexplored options such as hydrogen/electricity-based chemicals and fuels are not included, so it is worth noting that the options are exemplary. In the cement industry (Figure 10.7), the potential and costs for clinker substitution and fuel mix changes are dependent on regional availability and the price of clinker substitutes and alternative fuels. Negative cost options in cement manufacturing are in switching to best practice clinker-to-cement ratio. In the iron and steel industry (Figure 10.8), a shift from blast furnace based steelmaking to electric arc furnace steelmaking provides significant negative cost opportunities. However, this potential is highly dependent on scrap availability. The chemical sector (Figure 10.9) includes options related to energy efficiency improvements and options related to reduction of N_2O emissions from nitric and adipic acid production and HFC-23 emissions from HFC-22 production. In pulp and paper manufacturing (Figure 10.10), the estimates exclude increased recycling because the effect on CO_2 emissions is uncertain.

The costs of the abatement options shown in Figure 10.7 vary widely between individual regions and from plant to plant in the cement industry. Factors influencing the costs include typical capital stock turnover rates (some measures can only be applied when plants are replaced), relative energy costs, etc. For clinker substitution and fuel mix improvements, costs depend heavily on the regional availability and price of clinker substitutes and alternative fuels.

For all subsectors, negative abatement cost options exist to a certain extent for shifting to best practice technologies and for fuel shifting. While options in cost ranges of 0–20 and 20–50 USD_{2010}/tCO_2eq are somewhat limited, larger opportunities exist in the 50–150 USD_{2010}/tCO_2eq range (particularly since CCS is included here). The feasibility of CCS depends on global CCS developments. CCS is currently not yet applied (with some exceptions) at commercial scale in the cement, iron and steel, chemical, or pulp/paper industries.

10.8 Co-benefits, risks and spillovers

In addition to mitigation costs and potentials (see Section 10.7), the deployment of mitigation measures will depend on a variety of other factors that relate to broader economic, social, and environmental objectives that drive decisions in the industry sector and policy choices. The implementation of mitigation measures can have positive or negative effects on these other objectives. To the extent that these side-effects are positive, they can be deemed 'co-benefits'; if adverse and uncertain, they imply risks.[16] Co-benefits and adverse side-effects of mitigation measures (10.8.1), the associated technical risks and uncertainties (10.8.2) as well as their public perception (10.8.3) and technological spillovers (10.8.4), can significantly affect investment decisions, individual behaviour, and policymaker priorities. Table 10.5 provides an overview of the potential co-benefits and adverse side-effects of the mitigation measures that are assessed in this chapter. In accordance with the three sustainable development pillars described in Chapter 4, the table presents effects on objectives that may be economic, social, environmental, and health related. The extent to which co-benefits and adverse side-effects will materialize in practice as well as their net effect on social welfare differ greatly across regions, and is strongly dependent on local circumstances and implementation practices, as well as on the scale and pace of the deployment of the different mitigation measures (see Section 6.6).

[16] 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 (particularly Sections 2.4, 3.6.3, and 4.8) as well as to the glossary in Annex I for concepts and definitions.

10.8.1 Socio-economic and environmental effects

Social embedding of technologies depends on compatibility with existing systems, social acceptance, divisibility, eco-friendliness, relative advantage, etc. (Geels and Schot, 2010; Roy et al., 2013). A typical example is the tradeoff or the choice that is made between investing in mitigation in industry and adaptation in the absence of right incentives for mitigation action (Chakraborty and Roy, 2012a). Slow diffusion of mitigation options (UNIDO, 2011) can be overcome by focusing on, and explicit consideration of, non-direct cost-related characteristics of the technologies (Fleiter et al., 2012c). It is unanimously understood that maintaining competitiveness of industrial products in the market place is an important objective of industries, so implementation of mitigation measures will be a major favoured strategy for industries if they contribute to cost reduction (Bernstein et al., 2007; Winkler et al., 2007; Bassi et al., 2009). Increasing demand for energy in many countries has led to imports and increasing investment in high-cost reliable electric power generation capacity; so mitigation via implementation of energy efficiency measures help to reduce import dependency and investment pressure (Winkler et al., 2007). Labour unions are increasingly expressing their desire for policies to address climate change and support for a transition to 'green' jobs (Räthzel and Uzzell, 2012). Local air and water pollution in areas near industries have led to regulatory restrictions in almost all countries. In many countries, new industrial developments face increasing public resistance and litigation. If mitigation options deliver local air pollution benefits, they will have indirect value and greater acceptance.

The literature (cited in the following sections and in Table 10.5) documents that mitigation measures interact with multiple economic, social, and environmental objectives, although these associated impacts are not always quantified. In general, quantifying the corresponding welfare effects that a mitigation technology or practice entails is challenging, because they are very localized and different stakeholders may have different perspectives of the corresponding losses and gains (Fleiter et al., 2012c) (see Sections 2.4, 3.6.3, 4.2, and 6.6). It is important to note that co-benefits need to be assessed together with direct benefits to overcome barriers in implementation of the mitigation options (e.g., training requirements, losses during technology installation) (Worrell et al., 2003), which may appear otherwise larger for SMEs or isolated enterprises (Crichton, 2006; Zhang and Wang, 2008; Ghosh and Roy, 2011).

Energy efficiency (E/M): Energy efficiency includes a wide variety of measures that also achieve economic efficiency and natural/energy resource saving, which contribute to the achievement of environmental goals and other macro benefits (Roy et al., 2013). At the company level, the impact of energy efficient technology is often found to enhance productivity growth (Zuev et al., 1998; Boyd and Pang, 2000; Murphy, 2001; Worrell et al., 2003; Gallagher, 2006; Winkler et al., 2007; Zhang and Wang, 2008; May et al., 2013). Other benefits to companies, industry, and the economy as a whole come in the form of reduced fuel consumption requirements[17] and imports as well as reduced requirements for new electricity general capacity addition (Sarkar et al., 2003; Geller et al., 2006; Winkler et al., 2007; Sathaye and Gupta, 2010) which contribute to energy security (see Sections 6.6.2.2 and 7.9.1). Energy security in the industrial sector is primarily affected by concerns related to the sufficiency of resources to meet national energy demand at competitive and stable prices. Supply-side vulnerabilities in this sector arise if there is a high share of imported fuels in the industrial energy mix (Cherp et al., 2012a). Cherp et al. (2012a) estimate that the overall vulnerability of industrial energy consumption is lower than in the transport and residential and commercial (R&C) sectors in most countries. Nevertheless, since mitigation policies in industry would likely lead to higher energy efficiency, they may reduce exposure to energy supply and price shocks (Gnansounou, 2008; Kruyt et al., 2009; Sovacool and Brown, 2010; Cherp et al., 2012b).

Reduced fossil fuel burning brings associated reduced costs (Winkler et al., 2007), and reduced local impacts on ecosystems related to fossil fuel extraction and waste disposal liability (Liu and Diamond, 2005; Zhang and Wang, 2008; Chen et al., 2012; Ren et al., 2012; Hasanbeigi et al., 2013b; Lee and van de Meene, 2013; Xi et al., 2013; Liu et al., 2013) (see also Sections 7.9.2 and 7.9.3). In addition, other possible benefits of reduced reliance on fossil fuels include increases in employment and national income (Sathaye and Gupta, 2010) with new business opportunities (Winkler et al., 2007; Nidumolu et al., 2009; Wei et al., 2010; Horbach and Rennings, 2013).

There is wide consensus in the literature on local air pollution reduction benefits from energy efficiency measures in industries (Winkler et al., 2007; Bassi et al., 2009; Ren et al., 2012), such as positive health effects, increased safety and working conditions, and improved job satisfaction (Getzner, 2002; Worrell et al., 2003; Wei et al., 2010; Walz, 2011; Zhang et al., 2011; Horbach and Rennings, 2013) (see also Sections 7.9.2, 7.9.3 and WGII 11.9). Energy efficient technologies can also have positive impacts on employment (Getzner, 2002; Wei et al., 2010; UNIDO, 2011; OECD/IEA, 2012). Despite these multiple co-benefits, sometimes the relatively large initial investment required and the relatively long payback period of some energy efficiency measures can be a disincentive and an affordability issue, especially for SMEs, since the co-benefits are often not monetized (Brown, 2001; Thollander et al., 2007; Ghosh and Roy, 2011; UNIDO, 2011).

Emission efficiency (G/E): The literature documents well that increases in emissions efficiency can lead to multiple benefits (see Table 10.5). Local air pollution reduction is well documented as co-benefit of emissions efficiency measures (Winkler et al., 2007; Bassi et al., 2009; Ren et al., 2012). Associated health benefits (Aunan et al., 2004; Haines et al., 2009) and reduced ecosystem impacts (please refer to Section 7.9.2 for details) are society-wide benefits, while reduc-

[17] Please see Section 10.4 and references cited therein (e.g., Schäfer, 2005; Allwood et al., 2010; UNIDO, 2011; Saygin et al., 2011b; Gutowski et al., 2013).

tions in emission-related taxes or payment liabilities (Metcalf, 2009) are specific to industries, even though compliance costs might increase (Dasgupta et al., 2000; Mestl et al., 2005; Rivers, 2010). The net effect of these benefits and costs has not been studied comprehensively. Quantification of benefits is often done on a case-by-case basis. For example, Mestl et al. (2005) found that the environmental and health benefits of using electric arc furnaces for steel production in the city of Tiyuan (China) could potentially lead to higher benefits than other options, despite being the most costly option. For India, a detailed study (Chakraborty and Roy, 2012b) of 13 energy-intensive industrial units showed that several measures to reduce GHG emissions were adopted because the industries could realize positive effects on their own economic competitiveness, resource conservation such as water, and an enhanced reputation/public image for their commitment to corporate social responsibility towards a global cause.

If existing barriers (see Section 10.9) can be overcome, industrial applications of CCS deployed in the future could provide environmental co-benefits because CCS-enabled facilities have very low emissions rates for critical pollutants even without specific policies being in place for those emissions (Kuramochi et al., 2012b) (see Section 7.9.2 and Figure 7.8 for the air pollution effects of CCS deployment in power plants).

Mitigations options to reduce PFC emissions from aluminium production, N_2O emissions from adipic and nitric acid production (EPA, 2010a), and PFC emissions from semiconductor manufacturing (ISMI, 2005) have proven to enhance productivity and reduce the cost of production. Simultaneously, these measures provide health benefits and better working conditions for labour and local ambient air quality (Heijnes et al., 1999).[18]

Material efficiency (M/P): There is a wide range of benefits to be harnessed from implementing material efficiency options. Private benefits to industry in terms of cost reduction (Meyer et al., 2007) can enhance competitiveness, but national and subnational sales revenue might decline in the medium term due to reduction in demand for intermediate products used in manufacturing (Thomas, 2003). Material use efficiency increases can often be realized via cooperation in industrial clusters (see Section 10.5), while associated infrastructure development (new industrial parks) and associated cooperation schemes lead to additional societal gains (e.g., more efficient use of land through bundling activities) (Lowe, 1997; Chertow, 2000). With the reduction in need for virgin materials (Allwood et al., 2013; Stahel, 2013) and the prioritization of prevention in line with the waste management hierarchy (see Section 10.14.2, Figure 10.16), mining-related social conflicts can decrease (Germond-Duret, 2012), health and safety can be enhanced, recycling-related employment can increase, the amount of waste material (see Section 10.14.2.1 and Figure 10.17) going into landfills can decrease, and new business opportunities related to material efficiency can emerge (Clift and Wright, 2000; Rennings and Zwick, 2002; Widmer et al., 2005; Clift, 2006; Zhang and Wang, 2008; Walz, 2011; Allwood et al., 2011; Raghupathy and Chaturvedi, 2013; Menikpura et al., 2013).

Demand reductions (P/S and S): Demand reduction through adoption of new diverse lifestyles (see Section 10.4) (Roy and Pal, 2009; GEA, 2012; Kainuma et al., 2012; Allwood et al., 2013) and implementation of healthy eating (see Section 11.4.3) and sufficiency goals can result in multiple co-benefits related to health that enhance human well-being (GEA, 2012). Well-being indicators can be developed to evaluate industrial economic activities in terms of multiple effects of sustainable consumption on a range of policy objectives (GEA, 2012).

10.8.2 Technological risks and uncertainties

There are some specific risks and uncertainties with adoption of mitigation options in industry. Potential health, safety, and environmental risks could arise from additional mining activities as some mitigation technologies could substantially increase the need for specific materials (e.g., rare earths, see Section 7.9.2) and the exploitation of new extraction locations or methods. Industrial production is closely linked to extractive industry (see Figure 10.2) and there are risks associated with closing mines if post-closure measures for environmental protection are not adopted due to a lack of appropriate technology or resources. Carbon dioxide capture and storage for industry is an example of a technological option subject to several risks and uncertainties (see Sections 10.7, 7.5.5, 7.6.4 and 7.9.4 for more in-depth discussion on CO_2 storage, transport, and the public perception thereof, respectively).

Specific literature on accidents and technology failure related to mitigation measures in the industry sector is lacking. In general, industrial activities are subject to the main categories of risks and emergencies, namely natural disasters, malicious activities, and unexpected consequences arising from overly complex systems (Mitroff and Alpaslan, 2003; Olson and Wu, 2010). For example, process safety is still a major issue for the chemical industry. Future improvements in process safety will likely involve a holistic integration of complementary activities and be supported by several layers of detail (Pitblado, 2011).

10.8.3 Public perception

From a socio-constructivist perspective, the social response to industrial activity depends on three sets of factors related to: 1) the dynamics of regional development and the historical place of industry in the community, 2) the relationship between residents and the industry and local governance capacities, and 3) the social or socio-economic impacts experienced (Fortin and Gagnon, 2006). Public hearings and stakeholder participation—especially on environmental and social impact assessments—prior to issuance of permission to operate has become mandatory in almost all countries,

[18] See also EPA Voluntary Aluminum Industrial Partnership: http://www.epa.gov/highgwp/aluminum-pfc/faq.html.

and industry expenditures for social corporate responsibility are now often disclosed. Mitigation measures in the industry sector might be considered socially acceptable if associated with co-benefits, such as reducing GHG emissions while also improving local environmental quality as a whole (e.g., energy efficiency measures that reduce local emissions). Public perception related to mitigation actions can be influenced by national political positions in international negotiations and media.

Research on public perception and acceptance with regard to industrial applications of CCS is lacking (for the general discussion of CCS see Chapter 7). To date, broad evidence related to whether public perception of CCS for industrial applications will be significantly different from CCS in power generation units is not available, since CCS is not yet in place in the industry sector (Section 10.7).

Mining activities have generated social conflicts in different parts of the world (Martinez-Alier, 2001; World Bank, 2007; Germond-Duret, 2012; Guha, 2013). The Observatory of Mining Conflicts in Latin America (OMCLA) reported more than 150 active mining conflicts in the region, most of which started in the 2000s[19]. Besides this general experience, the potential for interactions between social tensions and mitigation initiatives in this sector are unknown.

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[19] Observatorio de Conflictos Mineros de América Latina. Available at: http://www.conflictosmineros.net.

Table 10.5 | Overview of potential co-benefits (green arrows) and adverse side-effects (orange arrows) of the main mitigation measures in the industry sector. Arrows pointing up/down denote positive/negative effect on the respective objective or concern. Co-benefits and adverse side-effects depend on local circumstances as well as on the implementation practice, pace, and scale (see Section 6.6). For possible upstream effects of low-carbon energy supply (incl. CCS), see Section 7.9. For possible upstream effects of biomass supply, see Sections 11.7 and 11.13.6. 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. Numbers correspond to references below the table.

Mitigation measures	Effect on additional objectives/concerns		
	Economic	Social (including health)	Environmental
Technical energy efficiency improvements via new processes and technologies	↑ Energy security (via reduced energy intensity) [1, 2, 3, 4, 13, 29, 57]; ↑ Employment impact [14, 15, 19, 28] ↑ Competitiveness and Productivity [4, 5, 6, 7, 8, 9, 10, 11, 12] ↑ Technological spillovers in DCs (due to supply chain linkages) [59, 60, 61]	↓ Health impact via reduced local pollution [16] ↑ New business opportunities [4, 17–20] ↑ Water availability and quality [26] ↑ Safety, working conditions and job satisfaction [5, 19, 20]	Ecosystem impact via ↓ Fossil fuel extraction [21] ↓ Local pollution [11, 22–24, 25] and ↓ Waste [11, 27]
CO_2 and non-CO_2 GHG emissions intensity reduction	↑ Competitiveness [31, 55] and productivity [52, 53]	↓ Health impact via reduced local air pollution [30, 31, 32, 33, 53] and better work conditions (for PFCs from aluminium) [58]	Ecosystem impact via ↓ Local air pollution [4, 25, 30, 31, 34, 52] ↓ Water pollution [54] ↑ Water conservation [56]
Material efficiency of goods, recycling	↓ National sales tax revenue in medium term [35] ↑ Employment impact in waste recycling market [44, 45] ↑ New infrastructure for industrial clusters [36, 37] ↑ Competitiveness in manufacturing [38]	↑ New business opportunities [11, 39–43] ↓ Local conflicts (reduced resource extraction) [58] ↓ Health impacts and safety concerns [49]	↓ Ecosystem impact via reduced local air and water pollution and waste material disposal [42, 46] ↓ Use of raw/virgin materials and natural resources implying reduced unsustainable resource mining [47, 48]
Product demand reductions	↓ National sales tax revenue in medium term [35]	↑ Wellbeing via new diverse lifestyle choices [48, 50, 51]	↓ Post consumption waste [48]

[1] Sovacool and Brown, 2010; [2] Geller et al., 2006; [3] Gnansounou, 2008; [4] Winkler et al., 2007; [5] Worrell et al., 2003; [6] Boyd and Pang, 2000; [7] May et al., 2013; [8] Goldemberg, 1998; [9] Murphy, 2001; [10] Gallagher, 2006; [11] Zhang and Wang, 2008; [12] Roy et al., 2013; [13] see Section 10.4 and references cited therein; [14] UNIDO, 2011; [15] OECD/IEA, 2012; [16] Zhang et al., 2011; [17] Nidumolu et al., 2009; [18] Horbach and Rennings, 2013; [19] Getzner, 2002; [20] Wei et al., 2010; [21] Liu and Diamond, 2005; [22] Hasanbeigi et al., 2013a; [23] Xi et al., 2013; [24] Chen et al., 2012; [25] Ren et al., 2012; [26] Zhelev, 2005; [27] Lee and van de Meene, 2013; [28] Sathaye and Gupta, 2010; [29] Sathaye and Gupta, 2010; [30] Mestl et al., 2005; [31] Chakraborty and Roy, 2012a; [32] Haines et al., 2009; [33] Aunan et al., 2004; [34] Bassi et al., 2009; [35] Thomas, 2003; [36] Lowe, 1997; [37] Chertow, 2000; [38] Meyer et al., 2007; [39] Widmer et al., 2005; [40] Raghupathy and Chaturvedi, 2013; [41] Clift and Wright, 2000; [42] Allwood et al., 2011; [43] Clift, 2006; [44] Walz, 2011; [45] Rennings and Zwick, 2002; [46] Menikpura et al., 2013; [47] Stahel, 2013; [48] Allwood et al., 2013; [49] GEA, 2012; [50] Kainuma et al., 2012; [51] Roy and Pal, 2009; [52] EPA, 2010b; [53] ISMI, 2005; [54] Heijnes et al., 1999; [55] Rivers, 2010; [56] Chakraborty and Roy, 2012b; [57] Sarkar et al., 2003; [58] Germond-Duret, 2012; [59] Kugler, 2006; [60] Bitzer and Kerekes, 2008; [61] Zhao et al., 2010.

10.8.4 Technological spillovers

Spillovers are difficult to measure, but existing studies (Bouoiyour and Akhawayn, 2005) show that a technology gap is one of the conditions for positive spillovers. Sections 10.4 and 10.7 have already shown that there is gap between the world best practices in energy efficiency and industrial practices in many countries. As such, cross-country investment in mitigation technologies can enhance positive spillovers in host countries. In the industrial technology context, multinational companies try to minimize imitation probability and technology leakage, but studies show that spillover works faster through supply chain linkage inter-industry (Kugler, 2006; Bitzer and Kerekes, 2008; Zhao et al., 2010). In general, studies suggest that technology spillovers in the mitigation context depend on additional technology policies besides direct investment (Gillingham et al., 2009; Le and Pomfret, 2011; Wang et al., 2012a; Costantini et al., 2013; Jeon et al., 2013). These results are relevant for investments on industrial mitigation technologies as well.

10.9 Barriers and opportunities

Besides uncertainties in financial costs of mitigation options assessed in 10.7, a number of non-financial barriers and opportunities assessed in this section hinder or facilitate implementation of measures to reduce GHG emissions in industry. Barriers must be overcome to allow implementation (see Flannery and Kheshgi, 2005), however, in general they are not sufficiently captured in integrated model studies and scenarios (see Section 10.10). Barriers that are often common across sectors are given in Chapter 3. Table 10.6 summarizes barriers and opportunities for the major mitigation options listed in Section 10.4.

Typically, barriers and opportunities can be distinguished into the following categories:

- Technology: includes maturity, reliability, safety, performance, cost of technology options and systems, and gaps in information
- Physical: includes availability of infrastructure, geography, and space available
- Institutional and legal: includes regulatory frameworks and institutions that may enable investment
- Cultural: includes public acceptance, workforce capacity (e.g., education, training, and knowledge), and cultural norms.

10.9.1 Energy efficiency for reducing energy requirements

Even though energy consumption can be a significant cost for industry, a number of barriers limit industrial sector steps to minimize energy use via energy efficiency measures. These barriers include: failure to recognize the positive impact of energy efficiency on profitability, short investment payback thresholds (two to eight years; IEA, 2012e), industrial organizational and behavioural barriers to implementing change; limited access to capital; impact of non-energy policies on energy efficiency; public acceptance of unconventional manufacturing processes; and a wide range of market failures (Bailey et al., 2009; IEA, 2009d). While large energy-intensive industries—such as iron and steel, and mineral processing—are often aware of potential cost savings and consider energy efficiency in investment decisions, this is less common in the commercial and service sectors where the energy cost share is usually low, or for smaller companies where overhead costs for energy management and training personnel can be prohibitive (UNIDO, 2011; Ghosh and Roy, 2011; Schleich and Gruber, 2008; Fleiter et al., 2012d; Hasanbeigi et al., 2009). Of course, investment decisions also consider investment risks, which are generally not reflected in the cost estimates assessed in Section 10.7. The importance of barriers depends on specific circumstances. For example, by surveying the Swedish foundry industry, Rohdin et al. (2007) found that access to capital was reported to be the largest barrier, followed by technical risk and other barriers.

Cogeneration, or combined heat and power (CHP), is an energy efficiency option that can not only reduce GHG emissions by improving system energy efficiency, but can also reduce system cost and decrease dependence on grid power. For industry, however, (IEA, 2009d) CHP faces a complex set of economic, regulatory, social, and political barriers that restrain its wider use including: market restriction securing a fair market value for electricity exported to the grid; high upfront costs compared to large power plants; difficulty concentrating suitable heat loads and lack of integrated planning; grid access; non-transparent and technically demanding interconnection procedures; lack of consumer and policymaker knowledge about CHP energy, cost and emission savings; and industry perceptions that CHP is an investment outside their core business. Regulatory barriers can stem from taxes, tariffs, or permits. For a cogeneration project of an existing facility, the electricity price paid to a cogeneration facility is the most important variable in determining the project's success—more so than capital costs, operating and maintenance cost, and even fuel costs (Meidel, 2005). Prices are affected by rules for electricity markets, which differ from region to region, and which can form either incentives or barriers for cogeneration (Meidel, 2005).

10.9.2 Emissions efficiency, fuel switching, and carbon dioxide capture and storage

There are a number of challenges associated with feedstock and energy substitution in industry. Waste materials and biomass as fuel and feedstock substitutes are limited by their availability, and hence competition could drive up prices and make industrial applications less attractive (IEA, 2009b). A decarbonized power sector would offer new opportunities to reduce CO_2 intensity of some industrial processes via use of electricity, however, decarbonization of power also has barriers (assessed in Section 7.10).

Table 10.6 | Barriers (–) and opportunities (+) for GHG emission reduction options in industry. References and discussion appear in respective sub-sections of 10.9.

	Energy efficiency for reducing energy requirements	Emissions efficiency, fuel switching and CCS	Material efficiency	Product demand reduction	Non-CO_2 GHGs
Technological Aspects: Technology	+ many options available – technical risk + cogeneration mature in heavy industry – non-transparent and technically demanding interconnection procedures for cogeneration	+ fuels and technologies readily available – retrofit challenges + large potential scope for CCS in cement production, iron and steel, and petrochemicals – limited CCS technology development, demonstration and maturity for industry applications	+ options available	– slower technology turnover can slow technology improvement and operational emission reduction	+/– approaches and technologies available for some sources – lack of lower cost technology for PFC emission reduction in existing aluminium production plants
Technological Aspects: Physical	+ less energy and fuel use, lower cooling needs, smaller size – concentrating suitable heat loads for cogeneration – retrofit constraints on cogeneration	– lack of sufficient feedstock to meet demand – CCS retrofit constraints – lack of CO_2 pipeline infrastructure – limited scope and lifetime for industrial CO_2 utilization	+ reduction in raw and waste materials – transport infrastructure and industry proximity for material/waste reuse	+ reduction in raw materials and disposed products	– lack of control of HFC leakage in refrigeration systems
Institutional and Legal	– impact of non-energy policies + energy efficiency policies (10.11) – market barriers – regulatory, tax/tariff and permitting of cogeneration +/– grid access for cogeneration		– fragmented and weak institutions	– regulatory and legal instruments generally do not take account of externalities	– lack of certification of refrigeration systems – regulatory barriers to HFC alternatives in aerosols
Cultural	– lack of trained personnel +/– attention to energy efficiency – lack of acceptance of unconventional manufacturing processes – cogeneration outside core business – lack of consumer and policymaker knowledge of cogeneration	– social acceptance of CCS	+/– public participation – human capacity for management decisions	+/– user preferences drive demand	– lack of information/education about solvent replacements – lack of awareness of alternative refrigerants
Financial	– access to capital and short investment payback requirements – high overhead costs for small or less energy intensive industries +/– factoring in efficiency into investment decisions (e.g., energy management) + cogeneration economic in many cases +/– market value of grid power for cogeneration – high capital cost for cogeneration	– lack of sufficient financial incentive for widespread CCS deployment – liability risk for CCS – high CCS capital cost and long project development times	– upfront cost and potentially longer payback period + reduced production costs	– businesses, governments, and labour favour increased production	– recycled HFCs not cost competitive with new HFCs – cost of HFC incineration

The application of CCS to the industries covered in this chapter share many of the barriers to its application to power generation (see Section 7.10). Barriers for application of CCS in industry include space constraints when applied in retrofit situations (Concawe, 2011); high capital costs and long project development times; investment risk associated with poorly defined liability; the trade-exposed nature of many industries, which can limit viable CCS business models; current lack in general of financial incentives to offset the additional cost of CCS; and the immaturity of CO_2 capture technology for cement, iron and steel, and petrochemical industries (Kheshgi et al., 2012).

10.9.3 Material efficiency

There are technically feasible opportunities to improve material efficiency in industry (Allwood et al., 2011). One opportunity is a circular economy, which is a growing model across various countries and which aims to systematically fulfil the hierarchy principles of material efficiency "reduce, re-use, recycle" (see Section 10.14). This approach however, has barriers which include a lack of human and institutional capacities to encourage management decisions and public participation (Geng and Doberstein, 2008), as well as fragmented and weak institutions (Geng et al., 2010b). Improving material efficiency by integrating different industries (see Section 10.5) is often limited by specific local conditions, infrastructure requirements (e.g., pipelines) and the complexity of multiple users (Geng et al., 2010b).

10.9.4 Product demand reduction

Improved product design or material properties, respectively, can help to extend the product's lifetime and can lead to lower product demand. However it has to be considered that extended lifetime may not actually satisfy current user preferences, and the user may choose to replace an older, functioning product with a new one (van Nes and Cramer, 2006; Allwood et al., 2011). In addition, continually providing newer products may result in lower operational emissions (e.g., improved energy efficiency). In this case, longer product lifetimes might not automatically lead to lower overall emissions. For example, from a lifecycle balance point of view, it may be better to replace specific energy-intensive products such as washing machines, before their end-of-life to make use of more efficient substitutes (Scholl et al., 2010; Intlekofer et al., 2010; Fischer et al., 2012; Agrawal et al., 2012).

Businesses are rewarded for growing sales volumes and can prefer process innovation over product innovation (e.g., EIO 2011; 2012). Existing markets generally do not take into account negative externalities associated with resource use nor do they adequately incorporate the risks of resource-related conflicts (Bleischwitz et al., 2012; Transatlantic Academy, 2012), yet existing national accounting systems based on GDP indicators also support the pursuit of actions and policies that aim to increase demand spending for more products (Jackson, 2009; Roy and Pal, 2009). Labour unions often have an ambivalent position in terms of environmental policies and partly see environmental goals as a threat for their livelihood (Räthzel and Uzzell, 2012).

10.9.5 Non-CO_2 greenhouse gases

Non-CO_2 greenhouse gas emissions are an important contributor to industry process emissions (note that emissions of CO_2 from calcination are another important contributor: for barriers to controlling these emissions by CO_2 capture and storage see Section 10.9.2). Barriers to preventing or avoiding the release of HFCs, CFCs, HCFCs, PFC, and SF_6 in industry and from its products include: lack of awareness of alternative refrigerants and lack of guidance as to their use in a given or new system (UNEP and EC, 2010); lack of certification and control of leakage of HFCs from refrigeration (Heijnes et al., 1999); cost of recycled HFCs in markets where there is direct competition from newly produced HFCs (Heijnes et al., 1999); lack of information and communication and education about solvent replacements (Heijnes et al., 1999; IPCC/TEAP, 2005); cost of adaptation of existing aluminium production for PFC emission reduction and the absence of lower cost technologies in such situations (Heijnes et al., 1999); cost of incineration of HFCs emitted in HCFC production (Heijnes et al., 1999); regulatory barriers to alternatives to some HFC use in aerosols (IPCC/TEAP, 2005). UNEP (2010) found that there are technically and economically feasible substitutes for HCFCs, however, transitional costs remain a barrier for smaller enterprises.

10.10 Sectoral implications of transformation pathways and sustainable development

This section assesses transformation pathways for the industry sector over the 21st century by examining a wide range of published scenarios. This section builds upon scenarios which were collated by Chapter 6 in the WG III AR5 Scenario Database (see Annex II.10), which span a wide range of possible energy future pathways and which rely on a wide range of assumptions (e.g., population, economic growth, policies, and technology development and its acceptance). Against that background, scenarios for the industrial sector over the 21st century associated with different atmospheric CO_2eq concentrations in 2100 are assessed in Section 10.10.1, and corresponding implications for sustainable development and investment are assessed in Section 10.10.2 from a sector perspective.

10.10.1 Industry transformation pathways

The different possible trajectories for industry final energy demand (globally and for different regions), emissions, and carbon intensity under a wide range of CO_2eq concentrations over the 21st century are shown in Figure 10.11, Figure 10.12 and Figure 10.13[20]. These scenarios exhibit economic growth in general over the 21st century as well as growth specifically in the industry sector. Detailed scenarios of the industry sector extend to 2050 and exhibit increasing material production, e.g., iron/steel and cement (Sano et al., 2013; IEA, 2009b; Akashi et al., 2013). Scenarios generated by general equilibrium models, which include economic feedbacks (see Table 6.1), implicitly include changes in material flow due to, for example, changes in prices that may be driven by a price on carbon; however, these models do not generally provide detailed subsectoral material flows. Options for reducing material demand and inter-input substitution elasticities (Roy et al.,

[20] This section builds upon emissions scenarios which were collated by Chapter 6 in the WGIII AR5 scenario database (see Section 6.2.2), and compares them to detailed scenarios for industry referenced in this section. The scenarios included both baseline and mitigation scenarios. As described in more detail in Section 6.3.2, the scenarios shown in this section are categorized into bins based on 2100 concentrations: between 430–530 ppm CO_2eq, 530–650 ppm CO_2eq, and > 650 ppm CO_2eq by 2100. The relation between these bins of emission scenarios and the increase in global mean temperature since pre-industrial times is reviewed in Section 6.3.2.

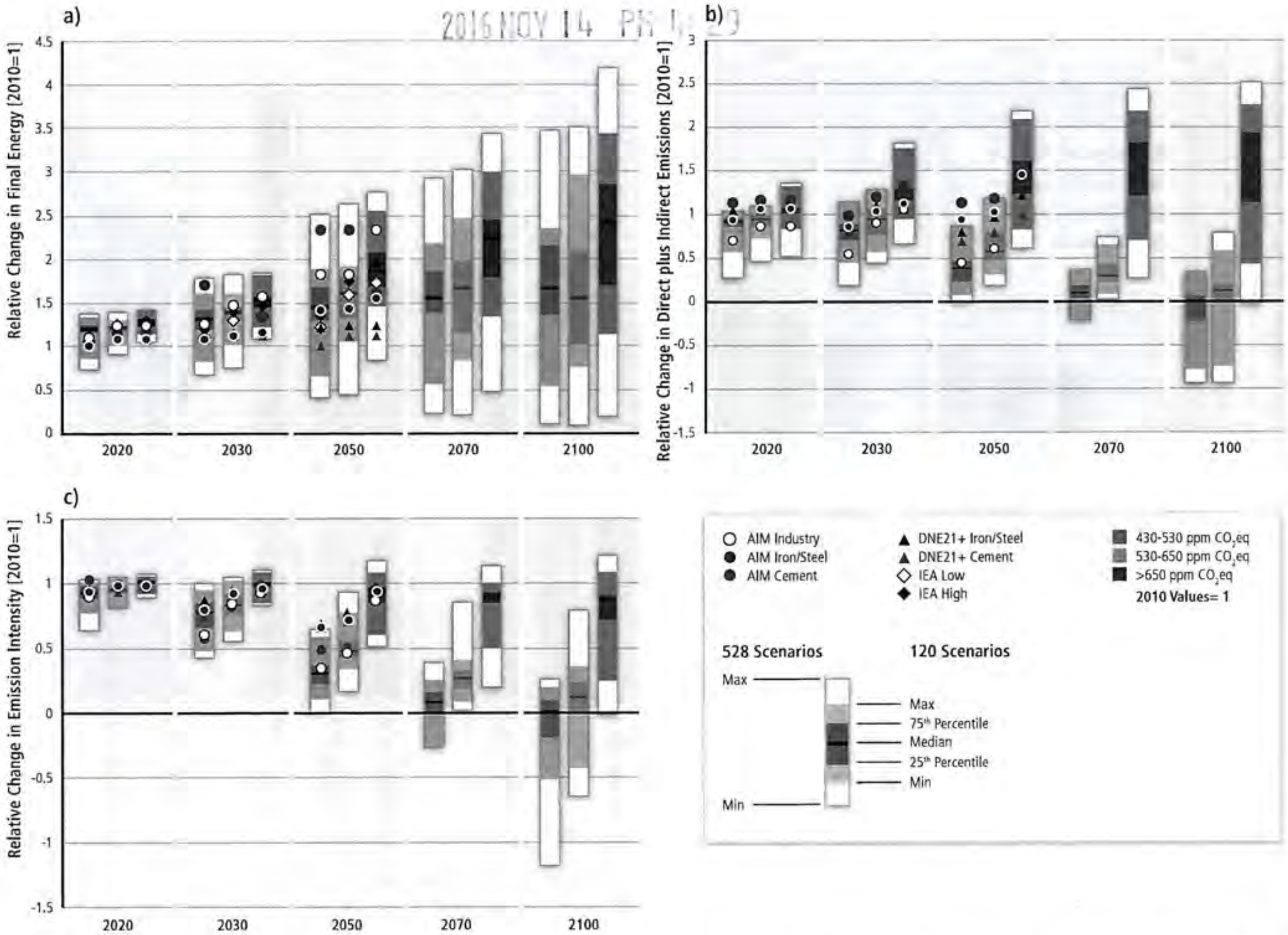

Figure 10.11 | Industry sector scenarios over the 21st century that lead to low (430–530 ppm CO_2eq), medium (530–650 ppm CO_2eq) and high (> 650 ppm CO_2eq) atmospheric CO_2eq concentrations in 2100 (see Table 6.3 for definitions of categories). All results are indexed relative to 2010 values for each scenario. Panels show: (a) final energy demand; (b) direct plus indirect CO_2eq emissions; (c) emission intensity (emissions from (b) divided by energy from (a)). Indirect emissions are emissions from industrial electricity demand. The median scenario (horizontal line symbol) surrounded by the darker colour bar (inner quartiles of scenarios) and lighter bar (full range) represent those 120 scenarios assessed in Chapter 6 with model default technology assumptions which submitted detailed final energy and emissions data for the industrial sector; white bars show the full range of scenarios including an additional 408, with alternate economic, resource, and technology assumptions (e.g., altering the economic and population growth rates, excluding some technology options or increasing response of energy efficiency improvement) (Source: WG III AR5 Scenario Database, see Annex II.10). Symbols are provided for selected scenarios for industry and industry sub-sectors (iron and steel, cement) for the IEA ETP (IEA, 2012d), AIM Enduse (Akashi et al., 2013 and Table A.II.14) and DNE21+ models (Sano et al., 2013a, b; and Table A.II.14) for their baseline, 550 ppm and 450 ppm CO_2eq scenarios to 2050.

2006; Sanstad et al., 2006) are used with various assumptions in the models that can better be characterized as gaps in integrated models currently in use.

Final energy (FE) demand from industry increases in most scenarios, as seen in Figure 10.11(a) driven by the growth of the industry sector; however, FE is weakly dependent on the 2100 CO_2eq concentration in the scenarios, and the range of FE demand spanned by the scenarios becomes wide in the latter half of the century (compare also Figure 6.37). In these scenarios, energy productivity improvements help to limit the increase in FE. For example, results of the DNE21+ and AIM models include a 56 % and 114 % increase in steel produced from 2010 to 2050 and a decrease in FE per unit production of 20–22 % and 28–34 % (these are the ranges spanned by the reference, 550 and 450 ppm CO_2eq scenarios for each model), respectively (Akashi et al., 2013; Sano et al., 2013). While energy efficiency of industry improves with time, the growth of CCS in some scenarios leads to increases in FE demand. Growth of final energy for cement production to 2050, for example, is seen in Figure 10.11(a) due to energy required for CCS in the cement industry mitigation scenarios (i.e., going from AIM cement > 650 ppm CO_2eq scenario to the < 650 ppm CO_2eq scenarios).

After 2050, emissions from industry, including indirect emissions resulting from industrial electricity demand become very low, and in some scenarios even negative as seen in Figure 10.11(b). The emission intensity of FE shown in Figure 10.11(c) decreases in most scenarios over the century, and decreases more strongly for low CO_2eq concentration levels. A decrease in emission intensity is generally the dominant mechanism for decrease in direct plus indirect emissions in the < 650 ppm CO_2eq scenarios shown in Figure 10.11. In scenarios

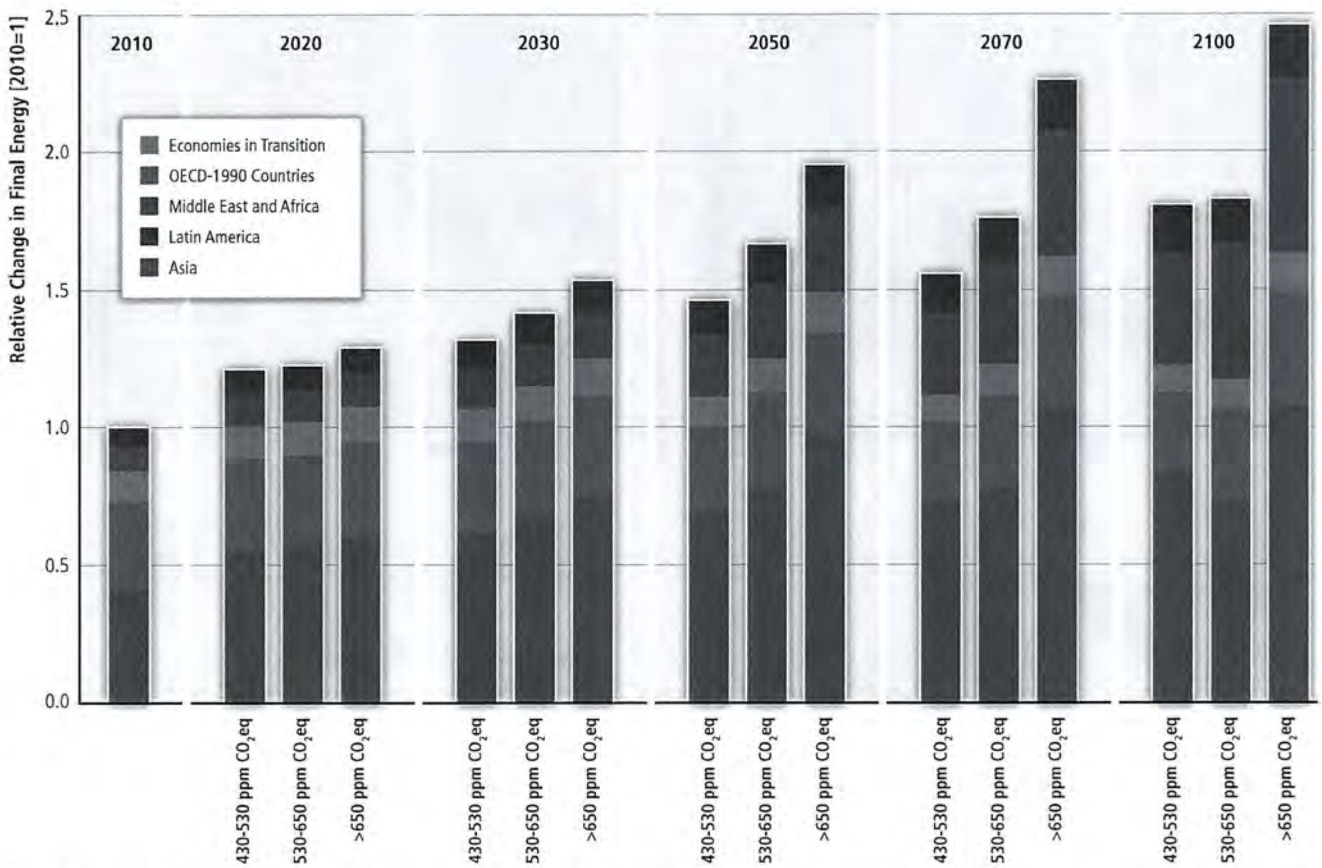

Figure 10.12 | Final energy demand from the industry sector shown for the RC5 regions (see Annex II.2 for definition) over the 21st century. Bars are compiled using information from 105 of the 120 scenarios assessed in Chapter 6, with model default technology assumptions that submitted detailed final energy and emissions data for the industrial sector. Bar height corresponds to the median scenario with respect to final energy demand relative to 2010; breakdown fractions correspond to the mean of scenarios. Source: WG III AR5 Scenario Database (see Annex II.10)

with strong decreases in emission intensity, this is generally due to some combination of application of CCS to direct industry emissions, and a shift to a lower-carbon carrier of energy—for example, a shift to low- or negative-carbon sources of electricity. Low carbon electricity is assessed in Chapter 7 and bioenergy with CCS—which could in theory result in net CO_2 removal from the atmosphere—is assessed in Chapter 11, Section 11.13.

Figure 10.12 shows the regional breakdown of final energy demand by world regions for different scenarios for the industrial sector. Over the 21st century, scenarios indicate that the growth of industry FE demand continues to be greatest in Asia, followed by the Middle East and Africa, although at a slower growth rate than seen over the last decade (see Section 10.3). The OECD-1990, Latin America, and Reforming Economies regions are expected to comprise a decreasing fraction of the world's industrial FE.

Figure 10.13 shows the projected changes in the shares of industry sector energy carriers—electricity, solids (primarily coal), and liquids, gases and hydrogen—from 2010 to 2100 for 120 scenarios (compare also Figure 6.38 with low carbon fuel shares in industrial final energy). Scenarios for all CO_2eq concentration levels show an increase in the share of electricity in 2100 compared to 2010, and generally show a decrease in the share of liquids/gases/hydrogen. Some of the < 650 ppm CO_2eq scenarios show an increase in the share of solids in 2100 compared to 2010 and some show a decrease. For the > 650 ppm CO_2eq scenarios, the change in shares from 2010 to 2100 is generally smaller than the change in shares for the < 650 ppm CO_2eq scenarios. A shift towards solids could lead to reduced emissions if the scenarios include the application of CCS to the emissions from solids. A shift towards electricity could lead to reduced emissions if the electricity generation is from low emission energy sources. The strong decrease in indirect emissions from electricity demand in most 430–530 ppm CO_2eq scenarios is shown in Figure 6.34 (see Section 6.8), with electricity emissions already negative in some scenarios by 2050. Each pathway implies some degree of lock-in of technology types and their supporting infrastructure, which has important implications; e.g., iron/steel in the basic oxygen furnace (BOF) route might follow a pathway with a higher solid fuel share but with CCS for direct emissions reduction by the industry. A decarbonized power sector provides the means to reduce the emission intensity of electricity use in the industrial sector, but barriers, such as a lack of a sufficient carbon price, exist (IEA, 2009b; Bassi et al., 2009). Barriers to decarbonization of electricity are discussed in more detail in Section 7.10.

Shares of Carriers in Final Energy in Industry

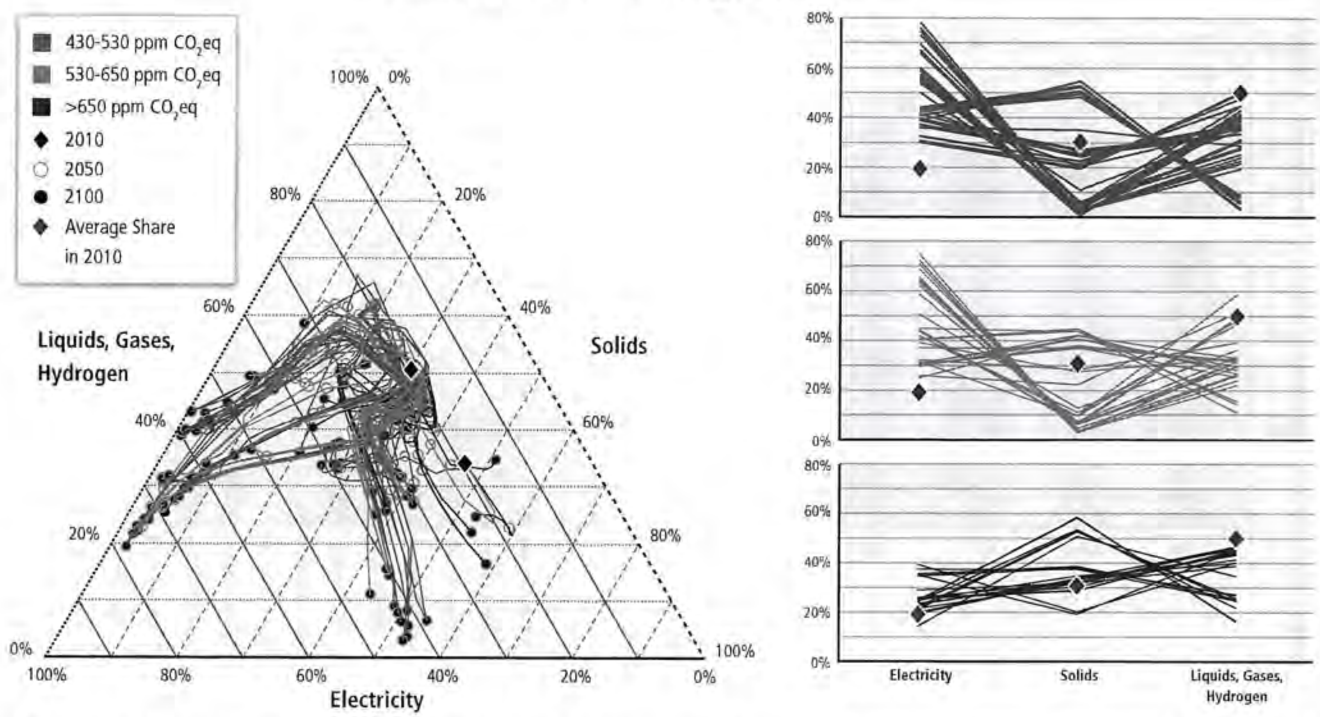

Figure 10.13 | The ternary panel on the left provides the industry final energy share trajectories across three groups of energy carriers: electricity, solids, and liquids-gases-hydrogen. The path of each scenario's trajectory is shown by a single line with symbols at the start in 2010 (the diamond towards the lower right accounts for 3 of 120 trajectories generated from one model that start in 2010 at a higher solids and lower liquids, gases, hydrogen share than the remainder of the trajectories which start at the upper diamond), in 2050 and at the end in 2100. The lines in the three panels on the right show the shares of energy carriers for each of the trajectories in the ternary diagram in 2100; the diamonds show the average share across a panel's models in 2010. Results are shown for those 120 scenarios assessed in Chapter 6, with model default technology assumptions that submitted detailed final energy and emissions data for the industrial sector. Source: WG III AR5 Scenario Database (see Annex II.10)

The IEA (2012d) 2DS scenario (Figure 10.14) shows a primary contribution to mitigation in 2050 from energy efficiency followed by recycling and energy recovery, fuel and feedstock switching, and a strong application of CCS to direct emissions. Carbon dioxide capture and storage has limited application before 2030, since CO_2 capture has yet to be applied at commercial scale in major industries such as cement or iron/steel and faces various barriers (see Section 10.9). Increased application of CCS is a precondition for rapid transitions and associated high levels of technology development and investment as well as social acceptance. The AIM 450 CO_2eq scenario (Akashi et al., 2013) has, for example, a stronger contribution from CCS than the IEA 2DS from 2030 onward, whereas the DNE21+ 450 ppm CO_2eq scenario (Sano et al., 2013) has a weaker contribution as shown in Figure 10.14. These more detailed industry sector scenarios fall within the range of the full set of scenarios shown in Figure 10.11.

10.10.2 Transition, sustainable development, and investment

Transitions in industry will require significant investment and offer opportunities for sustainable development (e.g., employment). Investment and development opportunities may be greatest in regions where industry is growing, particularly because investment in new facilities provides the opportunity to 'leapfrog', or avoid the use of less-efficient higher emissions technologies present in existing facilities, thus offering the opportunity for more sustainable development (for discussion of co-benefits and adverse side-effects when implementing mitigation options, see Section 10.8).

The wide range of scenarios implies that there will be massive investments in the industry sector over the 21st century. Mitigation scenarios generally imply an even greater investment in industry with shifts in investment focus. For example, due to an intensive use of mitigation technologies in the IEA's Blue Scenarios (IEA, 2009d), global investments in industry are 2–2.5 trillion USD higher by the middle of the century than in the reference case; successfully deploying these technologies requires not only consideration of competing investment options, but also removal of barriers and seizing of new opportunities (see Section 10.9).

The stringent mitigation scenarios discussed in Section 10.10.1 envisage emission intensity reductions, in particular due to deployment of CCS. However, public acceptance of widespread diffusion of CCS might hinder the realization of such scenarios. Taking the potential resistance into account, some alternative mitigation scenarios may require reduc-

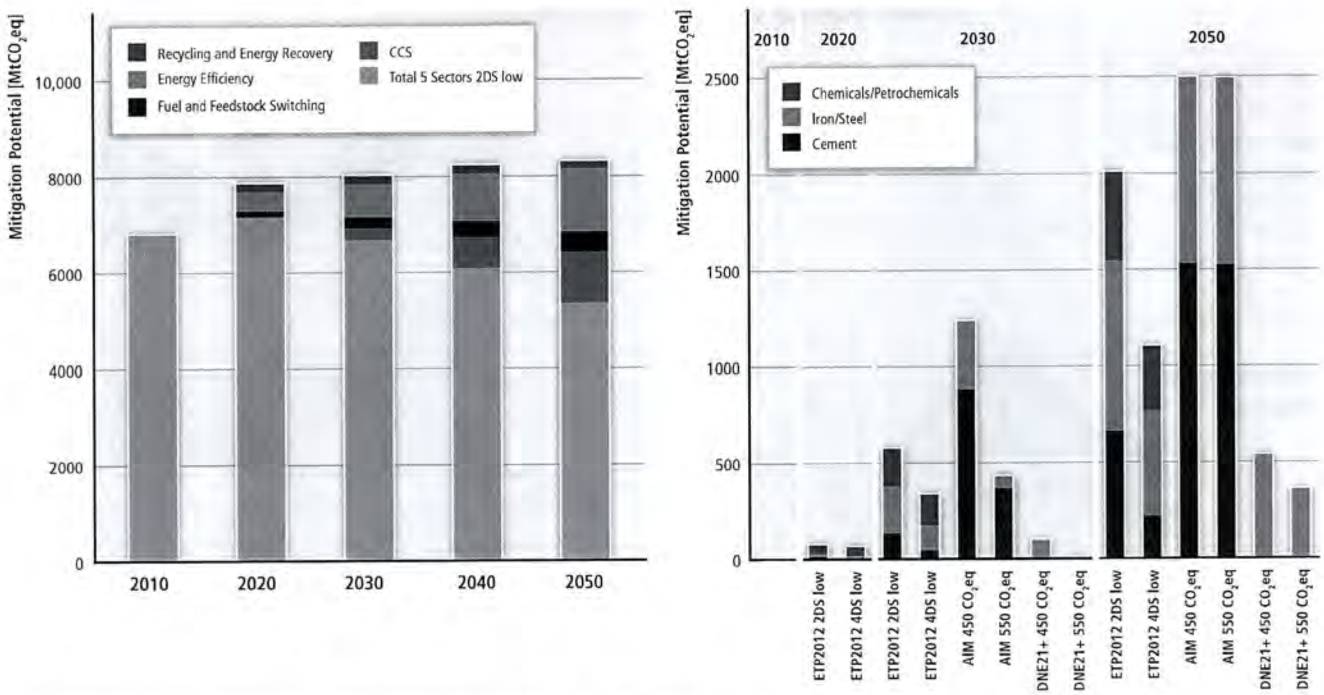

Figure 10.14 | Mitigation of direct CO_2eq annual emissions in five major industrial sectors: iron/steel, cement, chemicals/petrochemicals, pulp/paper, and aluminium. The left panel shows results from IEA scenarios (IEA, 2012d), broken down by mitigation option. The tops of the bars show the IEA 4DS low demand scenario, the light blue bars show the 2DS low demand scenario. The bar layers show the mitigation options that contribute to the emission difference from the 4DS to the 2DS low demand scenario. The right panel shows mitigation by CCS of direct industrial emissions in IEA, AIM Enduse (Akashi et al., 2013 and Table A.II.14) and DNE21+ (Sano et al., 2013a, b; and Table A.II.14) models. Scenarios are shown for those subsectors where CCS was reported.

tion of energy service demand (Kainuma et al., 2013). For the industry sector, options to reduce material demand or reduce demand for products become important as the latter do not rely on investment challenges, although they face a different set of barriers and can have high transaction costs (see Section 10.9).

Industry-related climate change mitigation options vary widely and may positively or negatively affect employment. Identifying mitigation options that enhance positive effects (e.g., due to some energy efficiency improvements) and minimize the negative outcomes is therefore critical. Some studies have argued that climate change mitigation policies can lead to unemployment and economic downturn (e.g. Babiker and Eckaus, 2007; Chateau et al., 2011) because such policies can threaten labour demand (e.g. Martinez-Fernandez et al., 2010) and can be regressive (Timilsina, 2009). Alternatively, other studies suggest that environmental regulation could stimulate eco-innovation and investment in more efficient production techniques and result in increased employment (OECD, 2009). Particularly, deployment of efficient energy technologies can lead to higher employment (Wei et al., 2010; UNIDO, 2011) depending on how redistribution of investment funds takes place within an economy (Sathaye and Gupta, 2010).

10.11 Sectoral policies

It is important to note that there is no single policy that can address the full variety of mitigation options for the industry sector. In addition to overarching policies (see Chapter 15 in particular, and Chapters 14 and 16), combinations of sectoral policies are needed. The diverse and relatively even mix of policy types in the industrial sector reflects the fact that there are numerous barriers to energy and material efficiency in the sector (see Section 10.9), and that industry is quite heterogeneous. In addition, the level of energy efficiency of industrial facilities varies significantly, both within subsectors and within and across regions. Most countries or regions use a mix of policy instruments, many of which interact. For example, energy audits for energy-intensive manufacturing firms are regularly combined with voluntary/negotiated agreements and energy management schemes (Anderson and Newell, 2004; Price and Lu, 2011; Rezessy and Bertoldi, 2011; Stenqvist and Nilsson, 2012). Tax exemptions are often combined with an obligation to conduct energy audits (Tanaka, 2011). Current practice acknowledges the importance of policy portfolios (e.g., Brown et al., 2011), as well as the necessity to consider national contexts and unintended behaviour of industrial companies. In terms of the latter, carbon leakage is relevant in the discussion of policies for industry (for a more in-depth analysis of carbon leakage see Chapter 5).

So far, only a few national governments have evaluated their industry-specific policy mixes (Reinaud and Goldberg, 2011). For the UK, Barker et al. (2007) modelled the impact of the UK Climate Change Agreements (CCAs) and estimated that from 2000 to 2010 they would result in a reduction of total final demand for energy of 2.6 % and a reduction in CO_2 emissions of 3.3 %. The CCAs established targets for industrial energy-efficiency improvements in energy-intensive industrial sectors; firms that met the targets qualified for a reduction of 80 % on the Climate Change Levy (CCL) rates on energy use in these sectors. Barker et al. (2007) also show that the macro-economic effect on the UK economy from the policies was positive.

In addition to dedicated sector-specific mitigation policies, co-benefits (see Section 10.8 and this report's framing chapters) should be considered. For example, local air quality standards have an indirect effect on mitigation as they set incentives for substitution of inefficient production technologies. Given the priorities of many governments, these indirect policies have played a relatively more effective role than climate policies, e.g. in India (Roy, 2010).

10.11.1 Energy efficiency

The use of energy efficiency policy in industry has increased appreciably in many IEA countries as well as major developing countries since the late 1990s (Roy, 2007; Worrell et al., 2009; Tanaka, 2011; Halsnæs et al., 2014). A review of 575 policy measures found that, as of 2010, information programmes are the most prevalent (40 %), followed by economic instruments (35 %), and measures such as regulatory approaches and voluntary actions (24 %) (Tanaka, 2011). Identification of energy efficiency opportunities through energy audits is the most popular measure, followed by subsidies, regulations for equipment efficiency, and voluntary/negotiated agreements. A classification of the various types of policies and their coverage are shown in Figure 10.15 and experiences in a range of these policies are analyzed below.

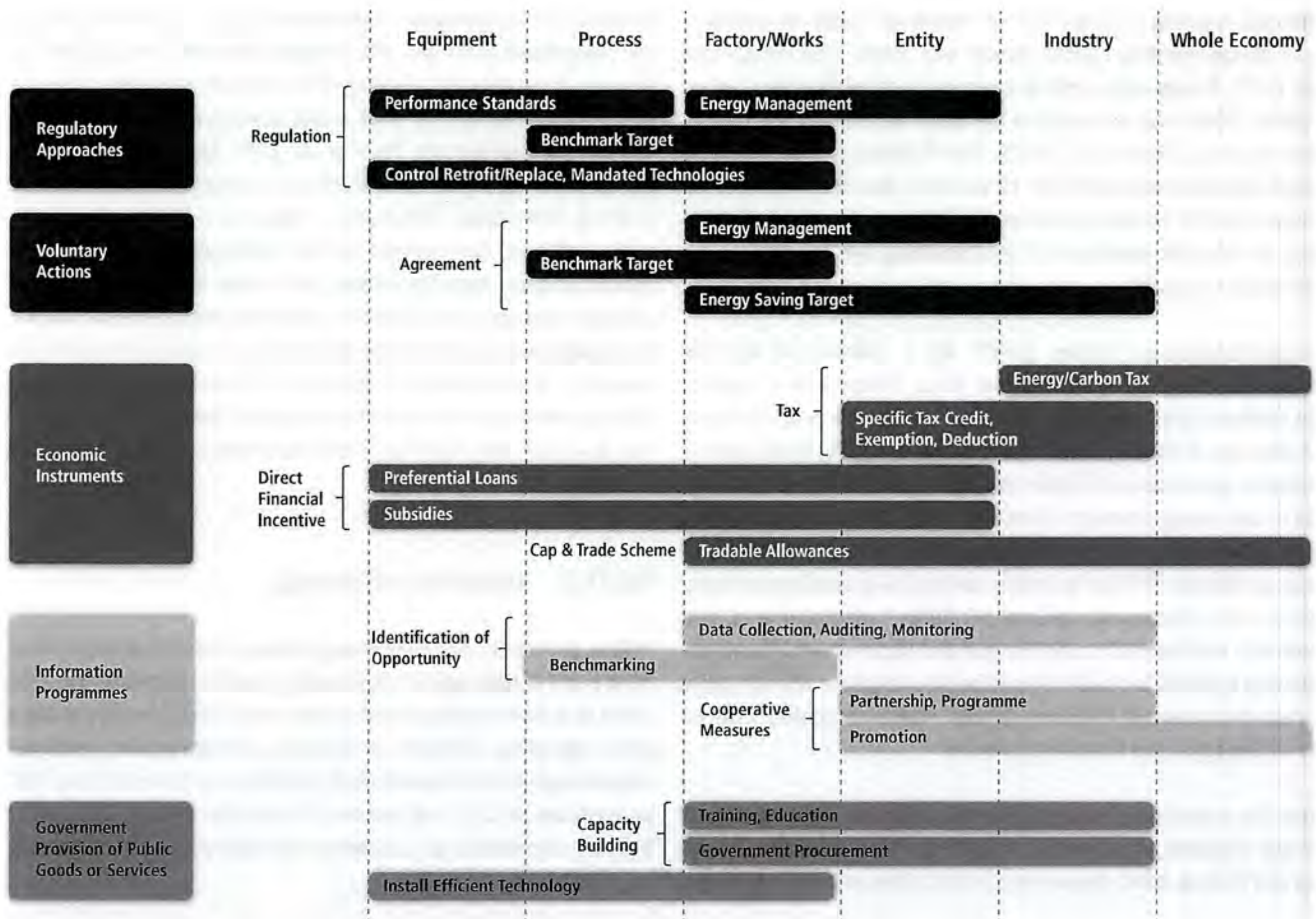

Figure 10.15 | Selected policies for energy efficiency in industry and their coverage (from Tanaka, 2011).

Greenhouse gas cap-and-trade and carbon tax schemes that aim to enhance energy efficiency in energy-intensive industry have been established in developed countries, particularly in the last decade, and are recently emerging in some developing countries. The largest example of these economic instruments by far is the European Emissions Trading Scheme (ETS). A more in-depth analysis of these overarching mechanisms is provided in Chapter 15.

Among regulatory approaches, regulations and energy efficiency standards for equipment have increased dramatically since 1992 (Tanaka, 2011). With regards to target-driven policies, one of the key initiatives for realizing the energy intensity reduction goals in China was the Top-1,000 Energy-Consuming Enterprises programme that required the establishment of energy-saving targets, energy use reporting systems and energy conservation plans, adoption of incentives and investments, and audits and training. The programme resulted in avoided CO_2 emissions of approximately 400 $MtCO_2$ compared to a business-as-usual baseline, and has been expanded to include more facilities under the new Top-10,000 enterprise programme. (Lin et al., 2011; Price et al., 2011; NDRC, 2011b)

Many firms (in particular SMEs) with rather low energy costs as a share of their revenue allocate fewer resources to improving energy efficiency, resulting in a low level of knowledge about the availability of energy-efficiency options (Gruber and Brand, 1991; Ghosh and Roy, 2011). Energy audits help to overcome such information barriers (Schleich, 2004) and can result in the faster adoption of energy-efficient measures (Fleiter et al., 2012b). The effectiveness of 22 industrial energy auditing programmes in 15 countries has been reviewed by Price and Lu (2011), who give recommendations on the success factors (e.g., use of public databases for benchmarking, use of incentives for participation in audits).

Energy Management Systems (EnMS) are a collection of business processes, carried out at plants and firms, designed to encourage and facilitate systematic improvement in energy efficiency. The typical elements of EnMS include maintenance checklists, measurement processes, performance indicators and benchmarks, progress reporting, and on-site energy managers (McKane, 2007). The adoption of EnMS schemes in industry can be mandatory, as in Japan, Italy, Turkey, or Portugal (Tanaka, 2011) or voluntary, and can be guided by standards, such as the international standard ISO 50001[21]. Backlund et al. and Thollander and Palm (2012; 2013) argue that improvement in practices identified by EnMS and audits should be given a greater role in studies of potential for energy efficiency, as most studies concentrate only on the technological and economic potentials.

There are a number of case studies that argue for the environmental and economic effectiveness of EnMS and energy audits (Anderson and Newell, 2004; Ogawa et al., 2011; Shen et al., 2012). Some studies report very quick payback for energy efficiency investments identified during such assessments (Price et al., 2008). For example, a programme in Germany offering partial subsidies to SMEs for energy audits was found to have saved energy at a cost to the German government of 2.4–5.7 USD_{2010}/tCO_2 (Fleiter et al., 2012b). In another case, the energy audit program by the Energy Conservation Centre of Japan (ECCJ), was found to provide positive net benefits for society, defined as the net benefit to private firms minus the costs to government, of 65 USD_{2010}/tCO_2 (Kimura and Noda, 2010). On the other hand, there are also studies that report mixed results of some mandatory EMS and energy audits, where some companies did not achieve any energy efficiency improvements (Kimura and Noda, 2010).

Many countries use benchmarking to compare energy use among different facilities within a particular sector (Tanaka, 2008; Price and McKane, 2009). In the Netherlands, for example, the Benchmarking Covenants encourage companies to compare themselves to others and to commit to becoming among the most energy-efficient in the world. However, in many countries high-quality energy efficiency data for benchmarking is lacking (Saygin et al., 2011b).

Negotiated, or voluntary agreements (VAs), have been found in various assessments to be effective and cost-efficient (Rezessy and Bertoldi, 2011). Agreement programmes (e.g., in Ireland, France, the Netherlands, Denmark, UK, Sweden) were often responsible for increasing the adoption of energy-efficiency and mitigation technologies by industries beyond what would have been otherwise adopted without the programmes (Price et al., 2010; Stenqvist and Nilsson, 2012). Some key factors contributing to successful VAs appear to be a strong institutional framework, a robust and independent monitoring and evaluation system, credible mechanisms for dealing with non-compliance, capacity-building and—very importantly—accompanying measures such as free or subsidized energy audits, mandatory energy management plans, technical assistance, information and financing for implementation (Rezessy and Bertoldi, 2011), as well as dialogue between industry and government (Yamaguchi, 2012). Further discussion and examples of the effectiveness of VAs can be found in Chapter 15.

10.11.2 Emissions efficiency

Policies directed at increasing energy efficiency (discussed above) most often result in reduction of CO_2 intensity as well, in particular when the aim is to make the policy part of a wider policy mix addressing multiple policy objectives. Examples of emissions efficiency policy strategies include support schemes and fiscal incentives for fuel switching, R&D programmes for CCS, and inclusion of reduction of non-CO_2 gases in voluntary agreements (e.g., Japanese voluntary action plan Keidanren, see Chapter 15).

Regarding gases with a relatively high GWP such as HFCs, PFCs, and SF_6, successful policy examples exist for capture in the power

[21] http://www.iso.org/iso/home/standards/management-standards/iso50001.htm.

sector (e.g., Japan; Nishimura and Sugiyama, 2008), but there is not much experience in the industry sector. The CDM has driven abatement of the industrial gases HFC-23 and N_2O in developing countries because of monetary incentives (Michaelowa and Buen, 2012)[22]. Including high GWP emissions within the same cap and trade programme (and therefore prices) as energy-related emissions may draw opposition from the industries concerned, so special programmes for these gases could be a better alternative (Hall, 2007). Another option suggested is to charge an upfront fee that would then be refunded when the gases are later captured and destroyed (Hall, 2007).

10.11.3 Material efficiency

Policy instruments for material or resource use efficiency in general are only just starting to be promoted for mitigation of GHG emissions in industry; consequently, effective communication to industry on the need and potential for an integrated approach is still lacking (Lettenmeier et al., 2009). Similarly, waste management policies are still not driven by climate concerns, although the potential for GHG emission reductions through waste management is increasingly recognized and accounted for (see Section 10.14, e.g., Worrell and van Sluisveld, 2013). Several economic instruments (e.g., taxes and charges) related to waste disposal have been shown to be effective in preventing waste, although they do not necessarily lead to improved design measures being taken further upstream (Hogg et al., 2011).

A number of policy packages are directly and indirectly aimed at reducing material input per unit of product or unit of service demand. Some examples are the European Action Plan on Sustainable Consumption and Production (SCP) and Sustainable Industry (EC, 2008a), the EU's resource efficiency strategy and roadmap (EC, 2011, 2012b), and Germany's resource efficiency programme, ProgRess (BMU, 2012). SCP policies[23] include both voluntary and regulatory instruments, such as the EU Eco-design Directive, as well as the Green Public Procurement policies. Aside from setting a framework and long-term goals for future legislation and setting up networks and knowledge bases, these packages include few specific policies and, most importantly, do not set quantitative targets nor explicitly address the link between material efficiency and GHG emission reductions.

Some single policies (as opposed to policy packages) related to material efficiency do include an assessment of their impacts in terms of GHG emissions. For example, the UK's National Industrial Symbiosis Programme (NISP) brokers the exchange of resources between companies (for an explanation of industrial symbiosis, see Section 10.5). An assessment of the savings through the NISP estimated that over 6 $MtCO_2eq$ were saved over the first five years (Laybourn and Morrissey, 2009). The PIUS-Check initiative by the German state of North Rhine-Westphalia (NRW) offers audits to companies where the relevant material flows are analyzed and recommendations for improvements are made. These PIUS-checks have been particularly successful in metal processing industries, and it is estimated that they have saved 20 thousand tonnes of CO_2 (EC, 2009).

In the Asia and Pacific region there are a number of region-specific policy instruments for climate change mitigation through SCP, such as the China Refrigerator Project, which realized emissions reductions of about 11 $MtCO_2$ between 1999 and 2005 by combining several practices including sustainable product design, technological innovation, eco-labelling, and awareness raising of consumers and retailers (SWITCH-Asia Network Facility, 2009). However, there is still a lack of solid ex-post assessments on SCP policy impacts.

Besides industry-specific policies there are policies with a different sector focus that influence industrial activity indirectly, by reducing the need for products (e.g., car pooling incentive schemes can lead to the production of less cars) or industrial materials (e.g., vehicle fuel economy targets can incentivize the design of lighter vehicles). A strategic approach in order to reflect the economy-wide resource use and the global risks may consist of national accounting systems beyond GDP[24] (Jackson, 2009; Roy and Pal, 2009; Arrow et al., 2010; GEA, 2012), including systems to account for increasing resource productivity (OECD, 2008; Bringezu and Bleischwitz, 2009) and of new international initiatives to spur systemic eco-innovations in key areas such as cement and steel production, light-weight cars, resource efficient construction, and reducing food waste.

10.12 Gaps in knowledge and data

The key challenge for making an assessment of the industry sector is the diversity in practices, which results in uncertainty, lack of comparability, incompleteness, and quality of data available in the public domain on process and technology specific energy use and costs. This diversity makes assessment of mitigation potential with high confidence at global and regional scales extremely difficult. Sector data are generally collected by industry/trade associations (international or national), are highly aggregated, and generally give little information about individual processes. The enormous variety of processes and technologies adds to the complexity of assessment (Tanaka, 2008, 2012; Siitonen et al., 2010).

[22] For a more in-depth analysis of CDM as a policy instrument, see Chapter 13, Sections 13.7.2 and 13.13.1.2.

[23] SCP policies are also covered in Chapter 4 (Sustainable Development and Equity, Section 4.4.3.1 SCP policies and programmes)

[24] For example, the EU's "Beyond GDP Initiative": http://www.beyond-gdp.eu/

Other major gaps in knowledge identified are:

- A systematic approach and underlying methodologies to avoid double counting due to the many different ways of attributing emissions (10.1).
- An in-depth assessment of mitigation potential and associated costs achievable particularly through material efficiency and demand-side options (10.4).
- Analysis of climate change impacts on industry and industry-specific mitigation options, as well as options for adaptation (10.6).
- Comprehensive information on sector and sub-sector specific option-based mitigation potential and associated costs based on a comparable methodology and transparent assumptions (10.7).
- Effect on long-term scenarios of demand reduction strategies through an improved modelling of material flows, inclusion of regional producer behaviour model parameters in integrated models (10.10).
- Understanding of the net impacts of different types of policies, the mitigation potential of linked policies e.g., resource efficiency/energy efficiency policies, as well as policy as drivers of carbon leakage effects (10.11).

10.13 Frequently Asked Questions

FAQ 10.1 How much does the industry sector contribute to GHG emissions?

Global industrial GHG emissions accounted for just over 30 % of global GHG emissions in 2010. Global industry and waste/wastewater GHG emissions grew from 10 $GtCO_2eq$ in 1990 to 13 $GtCO_2eq$ in 2005 to 15 $GtCO_2eq$ in 2010. Over half (52 %) of global direct GHG emissions from industry and waste/wastewater are from the ASIA region, followed by OECD-1990 (25 %), EIT (9 %), MAF (8 %), and LAM (6 %). GHG emissions from industry grew at an average annual rate of 3.5 % globally between 2005 and 2010. This included 7 % average annual growth in the ASIA region, followed by MAF (4.4 %) and LAM (2 %), and the EIT countries (0.1 %), but declined in the OECD-1990 countries (−1.1 %). (10.3)

In 2010, industrial GHG emissions were comprised of direct energy-related CO_2 emissions of 5.3 $GtCO_2eq$, 5.2 $GtCO_2eq$ indirect CO_2 emissions from production of electricity and heat for industry, process CO_2 emissions of 2.6 $GtCO_2eq$, non-CO_2 GHG emissions of 0.9 $GtCO_2eq$, and waste/wastewater emissions of 1.4 $GtCO_2eq$. (10.3)

2010 direct and indirect emissions were dominated by CO_2 (85.1 %) followed by CH_4 (8.6 %), HFC (3.5 %), N_2O (2.0 %), PFC (0.5 %) and SF_6 (0.4 %) emissions. Between 1990 and 2010, N_2O emissions from adipic acid and nitric acid production and PFC emissions from aluminium production decreased while HFC-23 emissions from HCFC-22 production increased. In the period 1990–2005, fluorinated gases (F-gases) were the most important non-CO_2 GHG source in manufacturing industry. (10.3)

FAQ 10.2 What are the main mitigation options in the industry sector and what is the potential for reducing GHG emissions?

Most industry sector scenarios indicate that demand for materials (steel, cement, etc.) will increase by between 45 % to 60 % by 2050 relative to 2010 production levels. To achieve an absolute reduction in emissions from the industry sector will require a broad set of mitigation options going beyond current practices. Options for mitigation of GHG emissions from industry fall into the following categories: energy efficiency, emissions efficiency (including fuel and feedstock switching, carbon dioxide capture and storage), material efficiency (for example through reduced yield losses in production), re-use of materials and recycling of products, more intensive and longer use of products, and reduced demand for product services. (10.4, 10.10)

In the last two to three decades there have been strong improvements in energy and process efficiency in industry, driven by the relatively high share of energy costs. Many options for energy efficiency improvement still remain, and there is still potential to reduce the gap between actual energy use and the best practice in many industries. Based on broad deployment of best available technologies, the GHG emissions intensity of the sector could be reduced through energy efficiency by approximately 25 %. Through innovation, additional reductions of approximately 20 % in energy intensity may potentially be realized before approaching technological limits in some energy intensive industries. (10.4, 10.7)

In addition to energy efficiency, material efficiency—using less new material to provide the same final service—is an important and promising option for GHG reductions that has had little attention to date. Long-term step-change options, including a shift to low carbon electricity or radical product innovations (e.g., alternatives to cement), may have the potential to contribute to significant mitigation in the future. (10.4)

FAQ 10.3 How will the level of product demand, interactions with other sectors, and collaboration within the industry sector affect emissions from industry?

The level of demand for new and replacement products has a significant effect on the activity level and resulting GHG emissions in the industry sector. Extending product life and using products more intensively could contribute to reduction of product demand without reducing the service. However, assessment of such strategies needs a careful net-balance (including calculation of energy demand in the production process and associated GHG emissions). Absolute emission reductions can also come about through changes in lifestyle and their corresponding demand levels, be it directly (e.g., for food, textiles) or indirectly (e.g., for product/service demand related to tourism). (10.4)

Mitigation strategies in other sectors may lead to increased emissions in industry if they require enhanced use of energy intensive materials (e.g., higher production of solar cells (PV) and insulation materials for buildings). Moreover, collaborative interactions within the industry sector and between the industry sector and other economic sectors have significant potential for mitigation (e.g., heat cascading). In addition, inter-sectoral cooperation, i.e., collaborative interactions among industries in industrial parks or with regional eco-industrial networks, can contribute to mitigation. (10.5)

FAQ 10.4 What are the barriers to reducing emissions in industry and how can these be overcome? Are there any co-benefits associated with mitigation actions in industry?

Implementation of mitigation measures in industry faces a variety of barriers. Typical examples include: the expectation of high return on investment (short payback period); high capital costs and long project development times for some measures; lack of access to capital for energy efficiency improvements and feedstock/fuel change; fair market value for cogenerated electricity to the grid; and costs/lack of awareness of need for control of HFC leakage. In addition, businesses today are mainly rewarded for growing sales volumes and can prefer process innovation over product innovation. Existing national accounting systems based on GDP indicators also support the pursuit of actions and policies that aim to increase demand for products and do not trigger product demand reduction strategies. (10.9)

Addressing the causes of investment risk, and better provisioning of user demand in the pursuit of human well-being could enable the reduction of industry emissions. Improvements in technologies, efficient sector specific policies (e.g., economic instruments, regulatory approaches and voluntary agreements), and information and energy management programmes could all contribute to overcome technological, financial, institutional, legal, and cultural barriers. (10.9, 10.11)

Implementation of mitigation measures in industries and related policies might gain momentum if co-benefits (10.8) are considered along with direct economic costs and benefits (10.7). Mitigation actions can improve cost competitiveness, lead to new market opportunities, and enhance corporate reputation through indirect social and environmental benefits at the local level. Associated positive health effects can enhance public acceptance. Mitigation can also lead to job creation and wider environmental gains such as reduced air and water pollution and reduced extraction of raw materials which in turn leads to reduced GHG emissions. (10.8)

10.14 Appendix: Waste

10.14.1 Introduction

Waste generation and reuse is an integral part of human activity. Figure 10.2 and Section 10.4 have shown how industries enhance resource use efficiency through recycling or reuse before discarding resources to landfills, which follows the waste hierarchy shown in Figure 10.16. Several mitigation options exist at the pre-consumer stage. Most important is reduction in waste during production processes. With regard to post-consumer waste, associated GHG emissions heavily depend on how waste is treated.

This section provides a summary of knowledge on current emissions from wastes generated from various economic activities (focusing on solid waste and wastewater) and discusses the mitigation options to reduce emissions and recover materials and energy from solid wastes.

10.14.2 Emissions trends

10.14.2.1 Solid waste disposal

The 'hierarchy of waste management' as shown in Figure 10.16, places waste reduction at the top, followed by re-use, recycling, energy recovery (including anaerobic digestion), treatment without energy recovery (including incineration and composting) and four types of landfills ranging from modern sanitary landfills that treat liquid effluents and also attempt to capture and use the generated biogas, through to traditional non-sanitary landfills (waste designated sites that lack controlled measures) and open burning. Finally, at the bottom of the pyramid are crude disposal methods in the form of waste dumps (designated or non-designated waste disposal sites without any kind of treatment) that are still dominant in many parts of the world. The hierarchy shown in Figure 10.16 provides general guidance. However, lifecycle assessment of the overall impacts of a waste management strategy for specific waste composition and local circumstances may change the priority order (EC, 2008b).

Municipal solid wastes (MSW) are the most visible and troublesome residues of human society. The total amount of MSW generated globally has been estimated at about 1.5 Gt per year (Themelis, 2007) and it is expected to increase to approximately 2.2 Gt by 2025 (Hoornweg and Bhada-Tata, 2012). Of the current amount, approximately 300 Mt are recycled, 200 Mt are treated with energy recovery, another 200 Mt are disposed in sanitary landfills, and the remaining 800 Mt are discarded in non-sanitary landfills or dumps. Thus, much of the recoverable matter in MSW is dispersed through mixing with other materials and exposure to reactive environmental conditions. The implications for GHG and other emissions are related not only to the direct emissions from waste management, but also to the emissions from production of materials to replace those lost in the waste.

Figure 10.17 presents global emissions from waste from 1970 until 2010 based on EDGAR version 4.2. Methane emissions from solid waste disposal almost doubled between 1970 and 2010. The drop in CH_4 emissions from solid waste disposal sites (SWDS) starting around 1990 is most likely related to the decrease in such emissions in Europe and the United States. However, it is important to note that the First Order Decay (FOD) model used in estimating emissions from solid waste disposal sites in the EDGAR database does not account for climate and soil micro-climate conditions like California Landfill Methane Inventory Model (CALMIM) (see Spokas et al., 2011; Spokas and Bogner, 2011; Bogner et al., 2011).

Global waste emissions per unit of GDP decreased 27 % from 1970 to 1990 and 34 % from 1990 to 2010, with a decrease of 48 % for the entire period (1970–2010). Global waste emissions per capita increased 10 % between 1970 and 1990, decreased 5 % from 1990 to 2010, with a net increase of 5 % for the entire period 1970–2010 (Figure 10.17). Several reasons may explain these trends: GHG emissions from waste in EU, mainly from solid waste disposal on land and wastewater handling decreased by 19.4 % in the decade 2000–2009; the decline is notable when compared to total EU27 emissions over the same period, which decreased by 9.3 %[25]. Energy production from waste in the EU in 2009 was more than double that generated in 2000, while biogas has experienced a 270 % increase in the same period. With the introduction of the Landfill Directive 10 1999/31/EC, the EU has established a powerful tool to reduce the amount of biodegradable municipal waste disposed in landfills (Blodgett and Parker, 2010). Moreover, methane emissions from landfills in the United States

[25] Eurostat 2013, available at http://epp.eurostat.ec.europa.eu/statistics_explained/index.php/Climate_change_-_driving_forces.

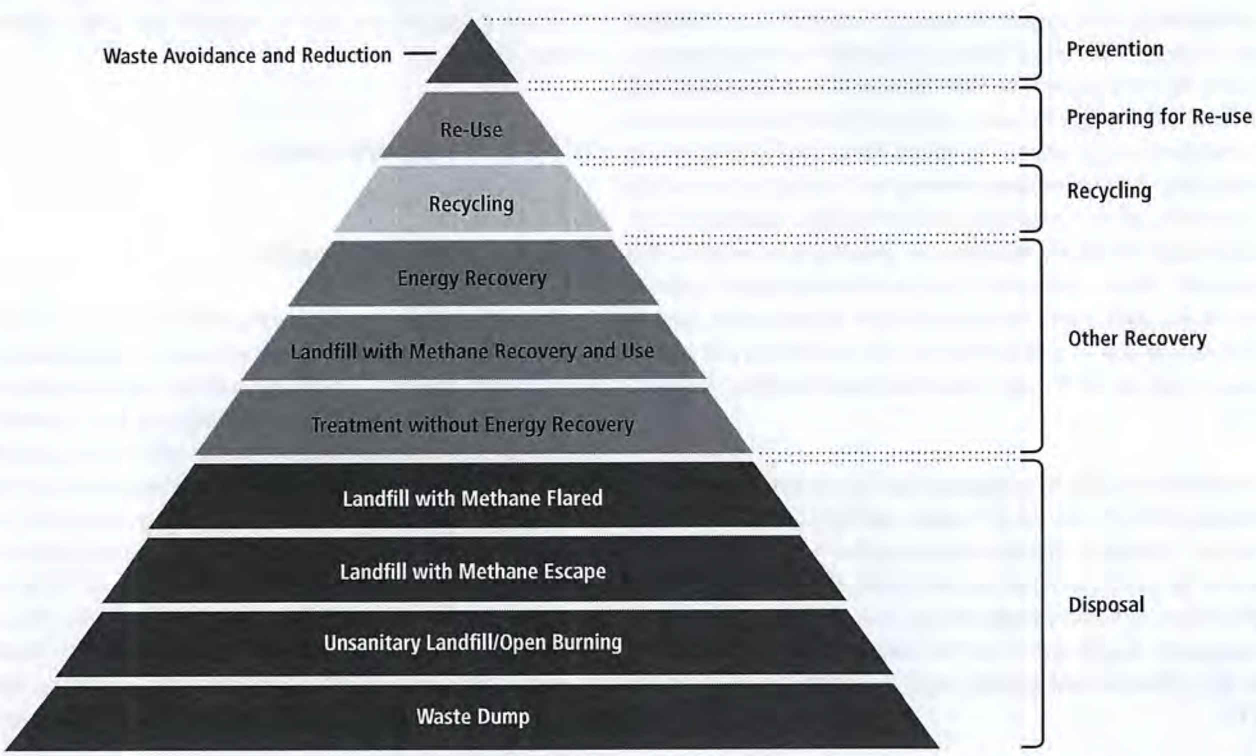

Figure 10.16 | The hierarchy of waste management. The priority order and colour coding is based on the five main groups of waste hierarchy classification (Prevention; Preparing for Re-Use; Recycling; Other Recovery e.g., Energy Recovery; and Disposal) outlined by the European Commission (EC, 2008b).

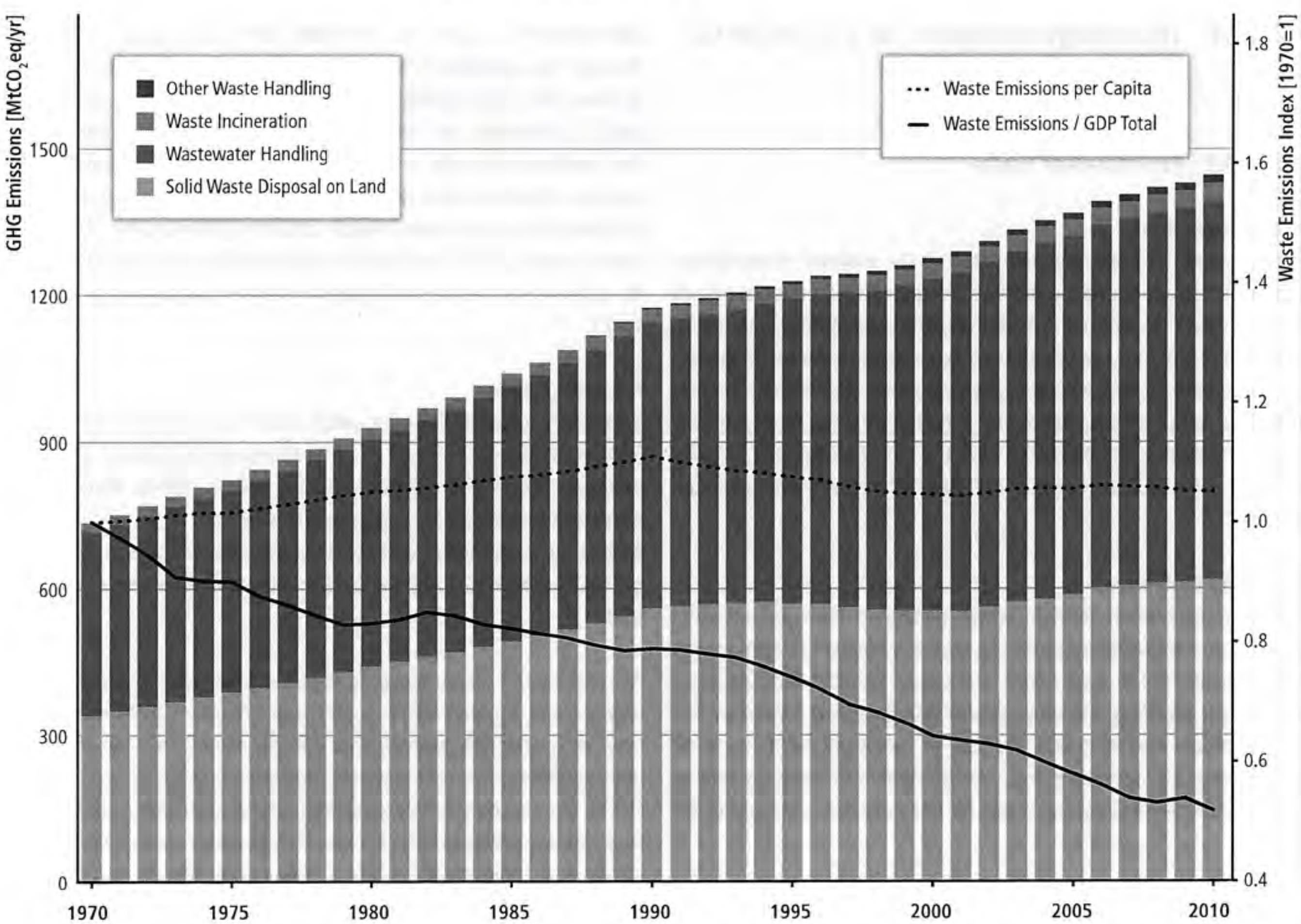

Figure 10.17 | Global waste emissions $MtCO_2eq$/year, global waste emissions per GDP and global waste emissions per capita referred to 1970 values. Based on JRC/PBL (2013), see Annex II.9.

decreased by approximately 27 % from 1990 to 2010. This net emissions decrease can be attributed to many factors, including changes in waste composition, an increase in the amount of landfill gas collected and combusted, a higher frequency of composting, and increased rates of recovery of degradable materials for recycling, e.g., paper and paperboard (EPA, 2012b).

China's GHG emissions in the waste sector increased rapidly in the 1981 to 2009 period, along with the growing scale of waste generation by industries as well as households in urban and rural areas (Qu and Yang, 2011). A 79 % increase in landfill methane emissions was estimated between 1990 (2.4 Mt) and 2000 (4.4 Mt) due to changes in both the amount and composition of municipal waste generated (Streets et al., 2001) and emission of China's waste sector will peak at 33.2 $MtCO_2eq$ in 2024 (Qu and Yang, 2011). In India (INCCA, 2010), the waste sector contributed 3 % of total national CO_2 emission equivalent of which 22 % is from municipal solid waste and the rest are from domestic wastewater (40 %) and industrial wastewater (38 %). Domestic wastewater is the dominant source of CH_4 in India. The decrease of GHG emissions in the waste sector in the EU and the United States from 1990 to 2009 has not been enough to compensate for the increase of emissions in other regions resulting in an overall increasing trend of total waste-related GHG emissions in that period.

10.14.2.2 Wastewater

Methane and nitrous oxide emissions from wastewater steadily increased during the last decades reaching 667 and 108 $MtCO_2eq$ in 2010, respectively. Methane emissions from domestic/commercial and industrial categories are responsible for 86 % of wastewater GHG emissions during the period 1970–2010, while the domestic/commercial sector was responsible for approximately 80 % of the methane emissions from wastewater category.

10.14.3 Technological options for mitigation of emissions from waste

10.14.3.1 Pre-consumer waste

Waste reduction

Pre-consumer (or post-industrial) waste is the material diverted from the waste stream during a manufacturing process that does not reach the end user. This does not include the reutilization of materials generated in a process that can be re-used as a substitute for raw materials (10.4) without being modified in any way. Waste reduction at the pre-consumer stage can be achieved by optimizing the use of raw materials, e.g., arranging the pattern of pieces to be cut on a length of fabric or metal sheet enable maximum utilization of material with minimum of waste.

Recycling and reuse

Material substitution through waste generated from an industrial process or manufacturing chain can lead to reduction in total energy requirements (10.4) and hence emissions. Section 10.4 discusses options for recycling and reuse in the manufacturing industries. The same section also discusses the use of municipal solid waste as energy source or feedstock, e.g., for the cement industry, as well as the possible use of industrial waste for mineralization approaches for CCS.

10.14.3.2 Post-consumer waste

Pre-consumer (or post-industrial) waste is the material resulting from a manufacturing process, which joins the waste stream and does not reach the end use. The top priority of the post-consumer waste management is reduction followed by re-use and recycling.

Waste reduction

To a certain extent, the amount of post-consumer waste is related to lifestyle. On a per capita basis, Japan and the EU have about 60 % of the US waste generation rates based significantly on different consumer behavior and regulations. Globally, a visionary goal of 'zero waste' assists countries in designing waste reduction strategies, technologies, and practices, keeping in mind other resource availability like land. Home composting has been successfully used in some regions, which reduces municipal waste generation rates (Favoino and Hogg, 2008; Andersen et al., 2010).

Non-technological behavioural strategies aim to avoid or reduce waste, for instance by decoupling waste generation from economic activity levels such as GDP (Mazzanti and Zoboli, 2008). In addition, strategies are in place that aim to enhance the use of materials and products that are easy to recycle, reuse, and recover (Sections 10.4, 10.11) in close proximity facilities.

Post-consumer waste can be linked with pre-consumer material through the principle of Extended Producer Responsibility in order to divert the waste going to landfills. This principle or policy is the explicit attribution of responsibility to the waste-generating parties, preferably already in the pre-consumer phase. In Germany, for example, the principle of producer responsibility for their products in the post-consuming phase is made concrete by the issuing of regulations (de Jong, 1997). Sustainable consumption and production and its influence on waste minimization are discussed also in Section 10.11.

Recycling/reuse

If reduction of post-consumer waste cannot be achieved, reuse and recycling is the next priority in order to reduce the amount of waste produced and to divert it from disposal (Valerio, 2010). Recycling of post-consumer waste can be achieved with high economic value to protect the environment and conserve the natural resources (El-Haggar, 2010). Section 10.4 discusses this in the context of reuse in industries.

As cities have become hotspots of material flows and stock density (Baccini and Brunner, 2012, p. 31) (see Chapter 12), MSW can be seen as a material reservoir that can be mined. This can be done not only through current recycling and/or energy recovery processes (10.4), but also by properly depositing and concentrating substances (e.g., metals, paper, plastic) in order to make their recuperation technically and economically viable in the future. The current amount of materials accumulated mainly in old/mature settlements, for the most part located in developed countries (Graedel, 2010), exceeds the amount of waste currently produced (Baccini and Brunner, 2012, p. 50).

With a high degree of agreement, it has been suggested that urban mining (as a contribution towards a zero waste scenario) could reduce important energy inputs of material future demands in contrast to domestically produced and, even more important for some countries, imported materials, while contributing to future material accessibility.

Landfilling and methane capture from landfills

It has been estimated (Themelis and Ulloa, 2007) that annually about 50 Mt of methane is generated in global landfills, 6 Mt of which are captured at sanitary landfills. Sanitary landfills that are equipped to capture methane at best capture 50 % of the methane generated; however, significantly higher collection efficiencies have been demonstrated at certain well designed and operated landfills with final caps/covers of up to 95 %.

The capital investment needed to build a sanitary landfill is less than 30 % of a waste-to-energy (WTE) plant of the same daily capacity. However, because of the higher production of electricity (average of 0.55 MWh of electricity per metric tonne of MSW in the U.S. vs 0.1

MWh for a sanitary landfill), a WTE plant is usually more economic over its lifetime of 30 years or more (Themelis and Ulloa, 2007). In other regions, however, the production of methane from landfills may be lower due to the reduction of biodegradable fraction entering the landfills or operating costs may be lower. Therefore, economics of both options may be different in such cases.

Landfill aeration

Landfill aeration can be considered as an effective method for GHG emissions reduction in the future (Ritzkowski and Stegmann, 2010). In situ aeration is one technology that introduces ambient air into MSW landfills to enhance biological processes and to inhibit methane production (Chai et al., 2013). Ambient air is introduced in the landfill via a system of gas wells, which results in accelerated aerobic stabilization of deposited waste. The resulting gas is collected and treated (Heyer et al., 2005; Prantl et al., 2006). Biological stabilization of the waste using in-situ aeration provides the possibility to reduce both the actual emissions and the emission potential of the waste material (Prantl et al., 2006).

Landfill aeration, which is not widely applied yet, is a promising technology for treating the residual methane from landfills utilizing landfill gas for energy when energy recovery becomes economically unattractive (Heyer et al., 2005; Ritzkowski et al., 2006; Rich et al., 2008). In the absence of mandatory environmental regulations that require the collection and flaring of landfill gas, landfill aeration might be applied to closed landfills or landfill cells without prior gas collection and disposal or utilization. For an in situ aerated landfill in northern Germany, for example, landfill aeration achieved a reduction in methane emissions by 83 % to 95 % under strictly controlled conditions (Ritzkowski and Stegmann, 2010). Pinjing et al. (2011) show that landfill aeration is associated with increased N_2O emissions.

Composting and anaerobic digestion

Municipal solid waste (MSW) contains 'green' wastes such as leaves, grass, and other garden and park residues, and also food wastes. Generally, green wastes are source-separated and composted aerobically (i.e., in presence of oxygen) in windrows. However, food wastes contain meat and other substances that, when composted in windrows, emit unpleasant odours. Therefore, food wastes need to be anaerobically digested in closed biochemical reactors. The methane generated in these reactors can be used in a gas engine to produce electricity, or for heating purposes. Source separation, collection, and anaerobic digestion of food wastes are costly and so far have been applied to small quantities of food wastes in a few cities (e.g., Barcelona, Toronto, Vienna; Arsova, 2010), except in cases where some food wastes are co-digested with agricultural residues. In contrast, windrow composting is practiced widely; for example, 62 % of the U.S. green wastes (22.7 million tonnes) were composted aerobically in 2006 (Arsova et al., 2008), while only 0.68 million tonnes of food wastes (i.e., 2.2 % of total food wastes; EPA, 2006a) were recovered.

Energy recovery from waste

With the exception of metals, glass, and other inorganic materials, MSW consists of biogenic and petrochemical compounds made of carbon and hydrogen atoms.

The energy contained in solid wastes can be recovered by means of several thermal treatment technologies including combustion of as-received solid wastes on a moving grate, shredding of MSW and combustion on a grate or fluidized bed, mechanical-biological treatment (MBT) of MSW into compost, refuse-derived fuel (RDF) or biogas from anaerobic digestion, partial combustion and gasification to a synthetic gas that is then combusted in a second chamber, and pyrolysis of source-separated plastic wastes to a synthetic oil. At this time, an estimated 90 % of the world's WTE capacity (i.e., about 180 Mt per year) is based on combustion of as-received MSW on a moving grate; the same is true of the nearly 120 new WTE plants that were built worldwide in the period of 2000–2007 (Themelis, 2007).

WTE plants require sophisticated Air Pollution Control (APC) systems that constitute a large part of the plant. In the last twenty years, because of the elaborate and costly APC systems, modern WTE plants have become one of the cleanest high temperature industrial processes (Nzihou et al., 2012). Source separation of high moisture organic wastes from the MSW increases the thermal efficiency of WTE plants.

Most of the mitigation options mentioned above require expenditures and, therefore, are more prevalent in developed countries with higher GDP levels. A notable exception to this general rule is China, where government policy has encouraged the construction of over 100 WTE plants during the first decade of the 21st century (Dong, 2011). Figure 10.18 shows the share of different management practices concerning the MSW generated in several nations (Themelis and Bourtsalas, 2013). China, with 18 % WTE and less than 3 % recycling, is at the level of Slovakia.

The average chemical energy stored in MSW is about 10 MJ/kg (lower heating value, LHV), corresponding to about 2.8 MWh per tonne. The average net thermal efficiency of U.S. WTE plants (i.e., electricity to the grid) is 20 %, which corresponds to 0.56 MWh per tonne of MSW. However, additional energy can be recovered from the exhaust steam of the turbine generator. For example, some plants in Denmark and elsewhere recover 0.5 MWh of electricity plus 1 MWh of district heating. A full discussion of the R1 factor, used in the EU for defining overall thermal efficiency of a WTE plant can be found in Themelis et al. (2013).

Studies of the biogenic and fossil-based carbon based on C14-C12 measurements on stack gas of nearly forty WTE plants in the United States have shown that about 65 % of the carbon content of MSW is biogenic (i.e., from paper, food wastes, wood, etc.) (Themelis et al., 2013) .

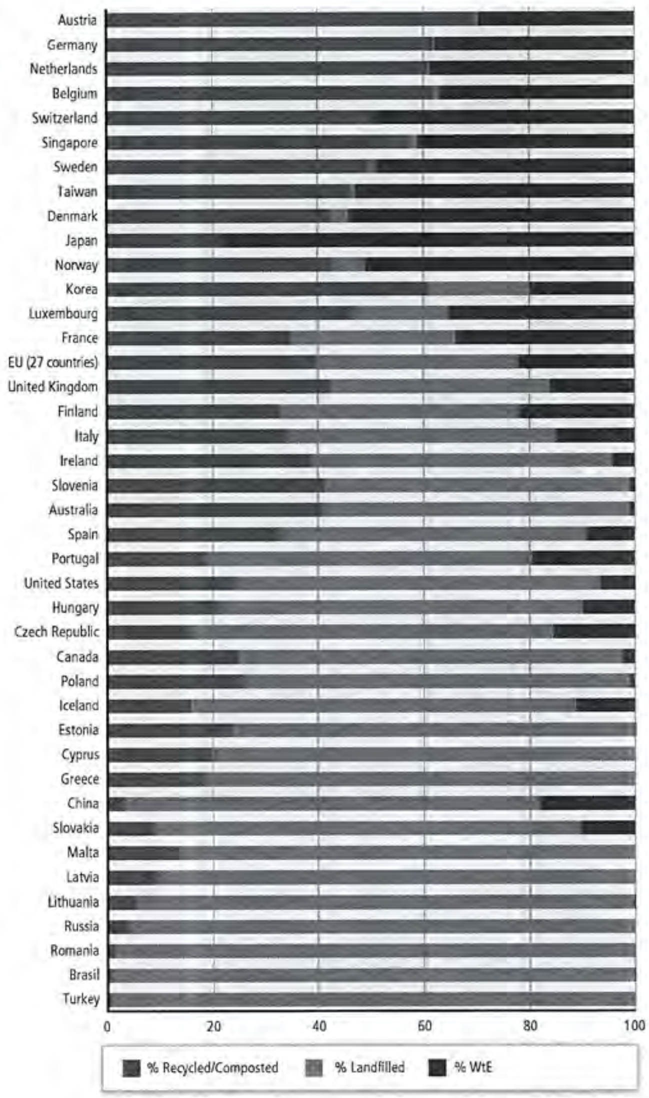

Figure 10.18 | Management practices concerning MSW in several nations (based on World Bank and national statistics, methodology described in Themelis and Bourtsalas (2013).

10.14.3.3 Wastewater

As a preventive measure, primary and secondary aerobic and land treatment help reduce CH_4 emissions during wastewater treatment. Alternatively, CH_4 emissions from wastewater, including sludge treatment under anaerobic conditions, can be captured and used as an energy source (Karakurt et al., 2012). Nitrous oxide is mainly released during biological nitrogen removal in wastewater treatment plants, primarily in aerated zones thus improved plant design and operational strategies (availability of dissolved oxygen, chemical oxygen demand and nitrogen ratio COD/N) have to be achieved in order to avoid the stripping of nitrous emissions (Kampschreur et al., 2009; Law et al., 2012).

Most developed countries rely on centralized aerobic/anaerobic wastewater treatment plants to handle their municipal wastewater. In developing countries, there is little or no collection and treatment of wastewater, anaerobic systems such as latrines, open sewers, or lagoons (Karakurt et al., 2012). Approximately 47 % of wastewater produced in the domestic and manufacturing sectors is untreated, particularly in South and Southeast Asia, but also in Northern Africa as well as Central and South America (Flörke et al., 2013). Wastewater treatment plants are highly capital-intensive but inflexible to adapt to growing demands, especially in rapidly expanding cities. Therefore, innovations related to decentralized wastewater infrastructure are becoming promising. These innovations include satellite systems, actions to achieve reduced wastewater flows, recovery and utilization of the energy content present in wastewater, recovery of nutrients, and the production of water for recycling, which will be needed to address the impacts of population growth and climate change (Larsen et al., 2013).

Industrial wastewater from the food industry usually has both high biochemical and chemical oxygen demand and suspended solid concentrations of organic origin that induce a higher GHG production per volume of wastewater treated compared to municipal wastewater treatment. The characteristics of the wastewater and the off-site GHG emissions have a significant impact on the total GHG emissions attributed to the wastewater treatment plants (Bani Shahabadi et al., 2009). For example, in the food processing industry with aerobic/anaerobic/hybrid process, the biological processes in the treatment plant made for the highest contribution to GHG emissions in the aerobic treatment system, while off-site emissions are mainly due to material usage and represent the highest emissions in anaerobic and hybrid treatment systems. Industrial cluster development in developing countries like China and India are enhancing wastewater treatment and recycling (see also Section 10.5).

Regional variation in wastewater quality matters in terms of performance of technological options. Conventional systems may be technologically inadequate to handle the locally produced sewage in arid areas like the Middle East. In these areas, domestic wastewater are up to five times more concentrated in the amount of biochemical and/or chemical oxygen demand per volume of sewage in comparison with United States and Europe, causing large amounts of sludge production. In these cases, choosing an appropriate treatment technology for the community could be a sustainable solution for wastewater management and emissions control. Example solutions include upflow anaerobic sludge blanket, hybrid reactors, soil aquifer treatment, approaches based on pathogens treatment, and reuse of the treated effluent for agricultural reuse (Bdour et al., 2009).

Wetlands can be a sustainable solution for municipal wastewater treatment due to their low cost, simple operation and maintenance, minimal secondary pollution, favourable environmental appearance, and other ecosystem service benefits (Mukherjee, 1999; Chen et al., 2008, 2011; Mukherjee and Gupta, 2011). It has been demonstrated that wetlands are a less energy intensive option than conventional wastewater treatment systems despite differences in costs across technologies and socio-economic contexts (Gao et al., 2012), but such systems are facing challenges in urban areas from demand for land for other economic activities (Mukherjee, 1999).

It has been highlighted that wastewater treatment with anaerobic sludge digestion and methane recovery and use for energy purposes reduces methane emissions (Bani Shahabadi et al., 2009; Foley et al., 2010; Massé et al., 2011; Fine and Hadas, 2012; Abbasi et al., 2012; Liu et al., 2012b; Wang et al., 2012b). Anaerobic digestion also provides an efficient means to reduce pollutant loads when high-strength organic wastewater (food waste, brewery, animal manure) have to be treated (Shin et al., 2011), although adequate regulatory policy incentives are needed for widespread implementation in developed and developing countries (Massé et al., 2011).

Advanced treatment technologies such as membrane filtration, ozonation, aeration efficiency, bacteria mix, and engineered nanomaterials (Xu et al., 2011b; Brame et al., 2011) may enhance GHG emissions reduction in wastewater treatment, and some such technologies, for example membranes, have increased the competitiveness and decentralization (Fane, 2007; Libralato et al., 2012).

The existence of a shared location and infrastructure can also facilitate the identification and implementation of more synergy opportunities to reduce industrial water provision and wastewater treatment, therefore abating GHG emissions from industry. The concept of eco-industrial parks is discussed in Section 10.5.

10.14.4 Summary results on costs and potentials

Figure 10.19 and Figure 10.20 present the potentials and costs of selected mitigation options to reduce the GHG emissions of the two waste sectors that represent 90 % of waste related emissions: solid waste disposal (0.67 $GtCO_2eq$) and domestic wastewater (0.77 $GtCO_2eq$) emissions (JRC/PBL, 2013). For solid waste, potentials are presented in tCO_2eq/t solid waste and for wastewater and in tCO_2eq/t BOD_5 as % compared to current global average.

Six mitigation options for solid waste and three mitigation options for wastewater are assessed and presented in the figures. The reference case and the basis for mitigation potentials were derived from IPCC 2006 guidelines. Abatement costs and potentials are based on EPA (2006b; 2013).

The actual costs and potentials of the abatement options vary widely across regions and design of a treatment methodology. Given that technology options to reduce emissions from industrial and municipal waste are the same, it is not further distinguished in the approach. Furthermore, the potential of reductions from emissions from landfills are directly related to climatic conditions as well as to the age and amount of landfill, both of which are not included in the chosen approach. Emission factors are global annual averages (derived from IPCC 2006 guideline aggregated regional averages). The actual emission factor differs between types of waste, climatic regions, and age of

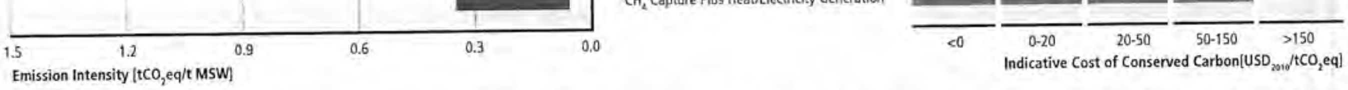

Figure 10.19 | Indicative CO_2eq emission intensities and levelized cost of conserved carbon of municipal solid waste disposal practices/technologies (for data and methodology, see Annex III).

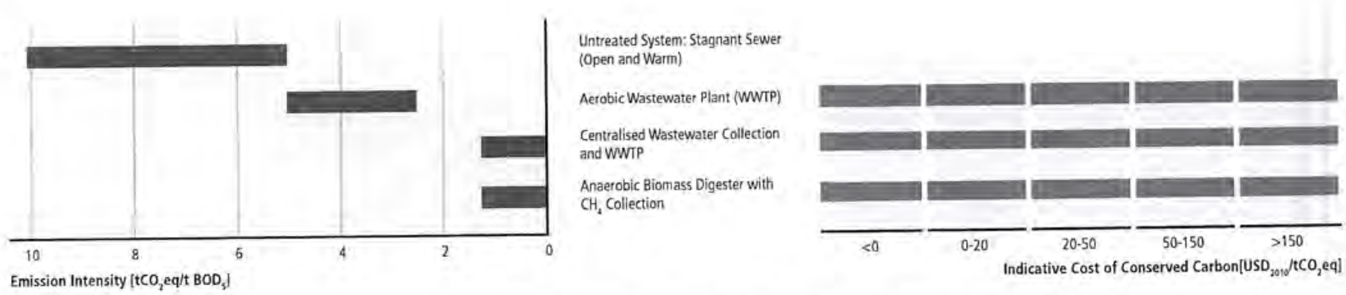

Figure 10.20 | Indicative CO_2eq emission intensities and levelized cost of conserved carbon of different wastewater treatments (for underlying data and methodology, see Annex III).

the landfill, explaining the wide range for each technology. The mitigation potential for waste is derived by comparing the emission range from a reference technology (e.g., a landfill) with the emission range for a chosen technology. The GHG coverage for solid waste is focused on methane, which is the most significant emission from landfilling; other GHG gases such as N_2O only play a minor role in the landfill solid waste sector and are neglected in this study (except for composting).

In the case of landfills, the top five emitting countries account for 27 % of the total abatement potential in the sector (United States 2 %, China 6 %, Mexico 9 %, Malaysia 3 %, and Russia 2 %). The distribution of the remaining potential per region is: Africa 16 %, Central and South America 9 %, Middle East 9 %, Europe 19 %, Eurasia 2 %, Asia 15 %, and North America 4 % (EPA, 2013).

In the case of wastewater, 58 % of the abatement potential is concentrated in the top five emitting countries (United States 7 %, Indonesia 9 %, Mexico 10 %, Nigeria 10 %, and China 23 %). The distribution of the remaining potential per region is: Africa 5 %, Central and South America 5 %, Middle East 14 %, Europe 5 %, Eurasia 4 %, and Asia 10 % (EPA, 2013).

The United States EPA has produced two studies with cost estimates of abatement in the solid waste sector (EPA, 2006b, 2013) which found a large range for options to reduce landfill (e.g., incineration, anaerobic digestion, and composting) of up to 590 USD_{2010}/tCO_2eq if the technology is only implemented for the sake of GHG emission reduction. However, the studies highlight that there are significant opportunities for CH_4 reductions in the landfill sector at carbon prices below 20 USD_{2010}. Improving landfill practices mainly by flaring and CH_4 utilization are low cost options, as both generate costs in the lower range (0—50 USD_{2010}/tCO_2eq).

The costs of the abatement options shown vary widely between individual regions and from plant to plant. The cost estimates should, for that reason, be regarded as indicative only and depend on a number of factors including capital stock turnover, relative energy costs, regional climate conditions, waste fee structures, etc. Furthermore, the method does not reflect the time variation in solid waste disposal and the degradation process as it assumes that all potential methane is released the year the solid waste is disposed.

The unit tonne biological oxygen demand (t BOD) stands for the organic content of wastewater ('loading') and represents the oxygen consumed by wastewater during decomposition. The average for domestic wastewater is in a range of 110–400 mg/l and is directly connected to climate conditions. Costs and potentials are global averages, but based on region-specific information. Options that are more often used in developing countries are not considered since data availability is limited. However, options like septic tanks, open sewers, and lagoons are low cost options with an impact of reducing GHG emission compared to untreated wastewater that is stored in a stagnant sewer under open and warm conditions.

The methane correction factor applied is based on the IPCC guidelines and gives an indication of the amount of methane that is released by applying the technology; furthermore emissions from N_2O have not been included as they play an insignificant role in domestic wastewater. Except in countries with advanced centralized wastewater treatment plants with nitrification and denitrification steps (IPCC, 2006), establishing a structured collection system for wastewater will always have an impact on GHG emissions in the waste sector.

Cost estimates of abatement in the domestic wastewater are provided in EPA (2006b; 2013), which find a large range for the options of 0 to 530 USD_{2010}/tCO_2eq with almost no variation across options. The actual costs of the abatement options shown vary widely between individual regions and from the design set up of a treatment methodology. Especially for wastewater treatment, the cost ranges largely depend on national circumstances like climate conditions (chemical process will be accelerated under warm conditions), economic development, and cultural aspects. The data availability for domestic wastewater options, especially on costs, is very low and would result in large ranges, which imply large uncertainties for each of the option. Mitigation potentials for landfills (in terms of % of potential above emissions for 2030) is double compared with wastewater (EPA, 2013). The mitigation potential for wastewater tends to concentrate in the higher costs options due to the significant costs of constructing public wastewater collection systems and centralized treatment facilities.

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11

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Contents

11

11

Executive Summary

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

The AFOLU sector is responsible for just under a quarter (~10–12 $GtCO_2eq/yr$) of anthropogenic GHG emissions mainly from deforestation and agricultural emissions from livestock, soil and nutrient management (*robust evidence; high agreement*) [11.2]. Anthropogenic forest degradation and biomass burning (forest fires and agricultural burning) also represent relevant contributions. Annual GHG emissions from agricultural production in 2000–2010 were estimated at 5.0–5.8 $GtCO_2eq/yr$ while annual GHG flux from land use and land-use change activities accounted for approximately 4.3–5.5 $GtCO_2eq/yr$. Leveraging the mitigation potential in the sector is extremely important in meeting emission reduction targets (*robust evidence; high agreement*) [11.9]. Since publication of the IPCC Fourth Assessment Report (AR4), emissions from the AFOLU sector have remained similar but the share of anthropogenic emissions has decreased to 24 % (in 2010), largely due to increases in emissions in the energy sector (*robust evidence, high agreement*). In spite of a large range across global Forestry and Other Land Use (FOLU) flux estimates, most approaches indicate a decline in FOLU carbon dioxide (CO_2) emissions over the most recent years, largely due to decreasing deforestation rates and increased afforestation (*limited evidence, medium agreement*). As in AR4, most projections suggest declining annual net CO_2 emissions in the long run. In part, this is driven by technological change, as well as projected declining rates of agriculture area expansion, which, in turn, is related to the expected slowing in population growth. However, unlike AR4, none of the more recent scenarios projects growth in the near-term [11.9].

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

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

AFOLU emissions could change substantially in transformation pathways, with significant mitigation potential from agriculture, forestry, and bioenergy mitigation measures (*medium evidence; high agreement*). Recent multi-model comparisons of idealized implementation transformation scenarios find land emissions (nitrous oxide, N_2O; methane, CH_4; CO_2) changing by −4 to 99 % through 2030, and 7 to 76 % through 2100, with the potential for increased emissions from land carbon stocks. Land-related mitigation, including bioenergy, could contribute 20 to 60 % of total cumulative abatement to 2030, and 15 to 40 % to 2100. However, policy coordination and implementation issues are challenges to realizing this potential [11.9]. Large-scale biomass supply for energy, or carbon sequestration in the AFOLU sector provide flexibility for the development of mitigation technologies in the energy supply and energy end-use sectors, as many technologies already exist and some of them are commercial (*limited evidence; medium agreement*) [11.3], but there are potential implications for biodiversity, food security, and other services provided by land (*medium evidence, high* *agreement*) [11.7]. Implementation challenges, including institutional barriers and inertia related to governance issues, make the costs and net emission reduction potential of near-term mitigation uncertain. In mitigation scenarios with idealized comprehensive climate policies, agriculture, forestry, and bioenergy contribute substantially to the reduction of global CO_2, CH_4, and N_2O emissions, and to the energy system, thereby reducing policy costs (*medium evidence; high agreement*) [11.9]. More realistic partial and delayed policies for global land mitigation have potentially significant spatial and temporal leakage, and economic implications, but could still be cost-effectively deployed (*limited evidence; limited agreement*) [11.9].

Economic mitigation potential of supply-side measures in the AFOLU sector is estimated to be 7.18 to 10.60 (full range: 0.49–10.60) $GtCO_2eq/yr$ in 2030 for mitigation efforts consistent with carbon prices up to 100 USD/tCO_2eq, about a third of which can be achieved at < 20 USD/tCO_2eq (*medium evidence; medium agreement*) [11.6]. These estimates are based on studies that cover both forestry and agriculture and that include agricultural soil carbon sequestration. Estimates from agricultural sector-only studies range from 0.3 to 4.6 $GtCO_2eq/yr$ at prices up to 100 USD/tCO_2eq, and estimates from forestry sector-only studies from 0.2 to 13.8 $GtCO_2eq/yr$ at prices up to 100 USD/tCO_2eq (*medium evidence; medium agreement*) [11.6]. The large range in the estimates arises due to widely different collections of options considered in each study, and because not all GHGs are considered in all of the studies. The composition of the agricultural mitigation portfolio varies with the carbon price, with the restoration of organic soils having the greatest potential at higher carbon prices (100 USD/tCO_2eq) and cropland and grazing land management at lower (20 USD/tCO_2eq). In forestry there is less difference between measures at different carbon prices, but there are significant differences between regions, with reduced deforestation dominating the forestry mitigation potential in Latin America and Caribbean (LAM) and Middle East and Africa (MAF), but very little potential in the member countries of the Organisation for Economic Co-operation and Development (OECD-1990) and Economies in Transition (EIT). Forest management, followed by afforestation, dominate in OECD-1990, EIT, and Asia (*medium evidence, strong agreement*) [11.6]. Among demand-side measures, which are under-researched compared to supply-side measures, changes in diet and reductions of losses in the food supply chain can have a significant, but uncertain, potential to reduce GHG emissions from food production (0.76–8.55 $GtCO_2eq/yr$ by 2050), with the range being determined by assumptions about how the freed land is used (*limited evidence; medium agreement*) [11.4]. More research into demand-side mitigation options is merited. There are significant regional differences in terms of mitigation potential, costs, and applicability, due to differing local biophysical, socioeconomic, and cultural circumstances, for instance between developed and developing regions, and among developing regions (*medium evidence; high agreement*) [11.6].

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

Bioenergy can play a critical role for mitigation, but there are issues to consider, such as the sustainability of practices and the efficiency of bioenergy systems (*robust evidence, medium agreement*) [11.4.4, Box 11.5, 11.13.6, 11.13.7]. Barriers to large-scale deployment of bioenergy include concerns about GHG emissions from land, food security, water resources, biodiversity conservation and livelihoods. The scientific debate about the overall climate impact related to land use competition effects of specific bioenergy pathways remains unresolved (*robust evidence, high agreement*) [11.4.4, 11.13]. Bioenergy technologies are diverse and span a wide range of options and technology pathways. Evidence suggests that options with low lifecycle emissions (e.g., sugar cane, Miscanthus, fast growing tree species, and sustainable use of biomass residues), some already available, can reduce GHG emissions; outcomes are site-specific and rely on efficient integrated 'biomass-to-bioenergy systems', and sustainable land-use management and governance. In some regions, specific bioenergy options, such as improved cookstoves, and small-scale biogas and biopower production, could reduce GHG emissions and improve livelihoods and health in the context of sustainable development (*medium evidence, medium agreement*) [11.13].

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

11.1 Introduction

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

The main mitigation options within AFOLU involve one or more of three strategies: *reduction/prevention* of emissions to the atmosphere by conserving existing carbon pools in soils or vegetation that would otherwise be lost or by reducing emissions of CH_4 and N_2O (Section 11.3); *sequestration*—enhancing the uptake of carbon in terrestrial reservoirs, and thereby removing CO_2 from the atmosphere (Section 11.3); and reducing CO_2 emissions by *substitution* of biological products for fossil fuels (Appendix 1) or energy-intensive products (Section 11.4). Demand-side options (e.g., by lifestyle changes, reducing losses and wastes of food, changes in human diet, changes in wood consumption), though known to be difficult to implement, may also play a role (Section 11.4).

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

In the Intergovernmental Panel on Climate Change (IPCC) Second Assessment Report (SAR) (IPCC, 1996) and in the IPCC Fourth Assessment Report (AR4) (IPCC, 2007a), agricultural and forestry mitigation were dealt with in separate chapters. In the IPCC Third Assessment Report (TAR) (IPCC, 2001), there were no separate sectoral chapters on either agriculture or forestry. In the IPCC Fifth Assessment Report (AR5), for the first time, the vast majority of the terrestrial land surface, comprising agriculture, forestry and other land use (AFOLU) (IPCC, 2006), is considered together in a single chapter, though settlements (which are important, with urban areas forecasted to triple in size from 2000 global extent by 2030; Section 12.2), are dealt with in Chapter 12. This approach ensures that all land-based mitigation options can be considered together; it minimizes the risk of double counting or inconsistent treatment (e.g., different assumptions about available land) between different land categories, and allows the consideration of systemic feedbacks between mitigation options related to the land surface (Section 11.4). Considering AFOLU in a single chapter allows phenomena common across land-use types, such as competition for land (Smith et al., 2010; Lambin and Meyfroidt, 2011) and water (e.g., Jackson et al., 2007), co-benefits (Sandor et al., 2002; Venter et al., 2009), adverse side-effects (Section 11.7) and interactions between mitigation and adaptation (Section 11.5) to be considered consistently. The complex nature of land presents a unique range of barriers and opportunities (Section 11.8), and policies to promote mitigation in the AFOLU sector (Section 11.10) need to take account of this complexity.

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

This chapter deals with AFOLU in an integrated way with respect to the underlying scenario projections of population growth, economic growth, dietary change, land-use change (LUC), and cost of mitigation. We draw evidence from both 'bottom-up' studies that estimate mitigation potentials at small scales or for individual options or technologies and then scale up, and multi-sectoral 'top-down' studies that consider AFOLU as just one component of a total multi-sector system response (Section 11.9). In this chapter, we provide updates on emissions trends and changes in drivers and pressures in the AFOLU sector (Section 11.2), describe the practices available in the AFOLU sector (Section 11.3), and provide refined estimates of mitigation costs and potentials for the AFOLU sector, by synthesising studies that have become available since AR4 (Section 11.6). We conclude the chapter by identifying gaps in knowledge and data (Section 11.11), providing a selection of Frequently Asked Questions (Section 11.12), and presenting an Appendix on bioenergy to update the IPCC Special Report on Renewable Energy Sources and Climate Change Mitigation (SRREN) (IPCC, 2011; see Section 11.13).

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

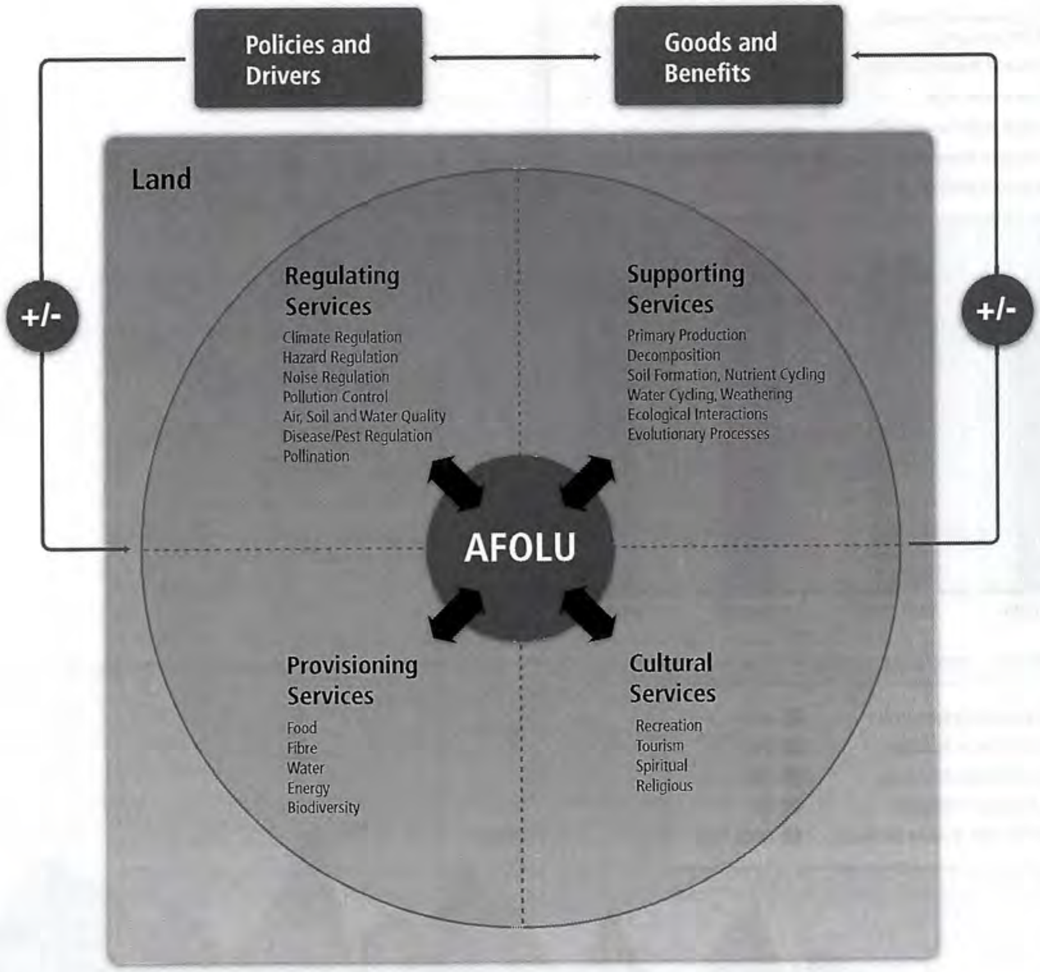

Figure 11.1 | Multiple ecosystem services, goods and benefits provided by land (after MEA, 2005; UNEP-WCMC, 2011). Mitigation actions aim to enhance climate regulation, but this is only one of the many functions fulfilled by land.

11.2 New developments in emission trends and drivers

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

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

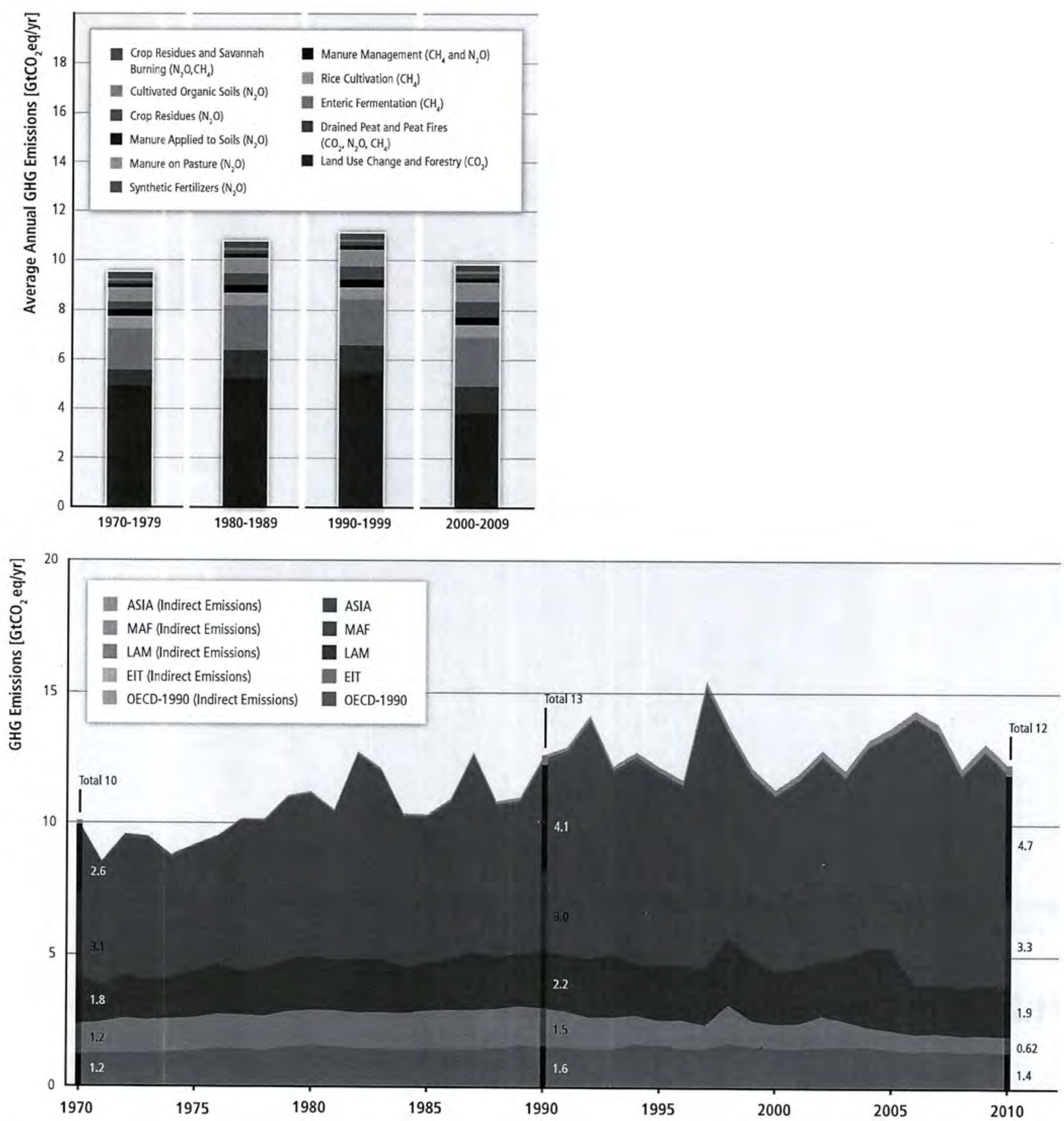

Figure 11.2 | Top: AFOLU emissions for the last four decades. For the agricultural sub-sectors emissions are shown for separate categories, based on FAOSTAT, (2013). Emissions from crop residues, manure applied to soils, manure left on pasture, cultivated organic soils, and synthetic fertilizers are typically aggregated to the category 'agricultural soils' for IPCC reporting. For the Forestry and Other Land Use (FOLU) sub-sector data are from the Houghton bookkeeping model results (Houghton et al., 2012). Emissions from drained peat and peat fires are, for the 1970s and the 1980s, from JRC/PBL (2013), derived from Hooijer et al. (2010) and van der Werf et al. (2006) and for the 1990s and the 2000s, from FAOSTAT, 2013. Bottom: Emissions from AFOLU for each RC5 region (see Annex II.2) using data from JRC/PBL (2013), with emissions from energy end-use in the AFOLU sector from IEA (2012a) included in a single aggregated category, see Annex II.9, used in the AFOLU section of Chapter 5.7.4 for cross-sectoral comparisons. The direct emission data from JRC/PBL (2013; see Annex II.9) represents land-based CO_2 emissions from forest and peat fires and decay that approximate to CO_2 flux from anthopogenic emission sources in the FOLU sub-sector. Differences between FAOSTAT/Houghton data and JRC/PBL (2013) are discussed in the text. See Figures 11.4 and 11.6 for the range of differences among available databases for AFOLU emissions.

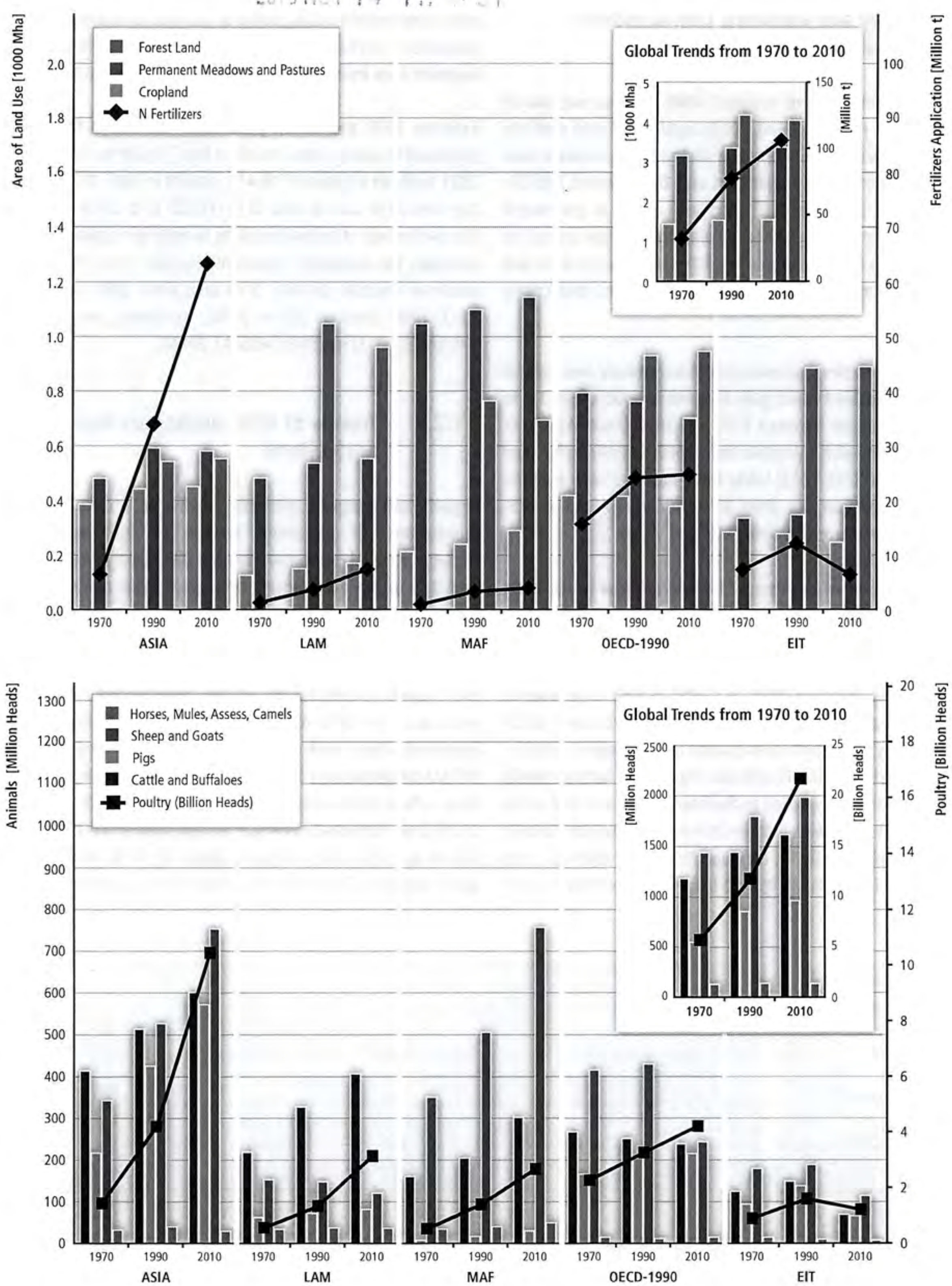

Figure 11.3 | Global trends from 1971 to 2010 in (top) area of land use (forest land—available only from 1990; 1000 Mha) and amount of N fertilizer use (million tonnes), and (bottom) number of livestock (million heads) and poultry (billion heads). Data presented by regions: 1) Asia, 2) LAM, 3) MAF, 4)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

11.2.1 Supply and consumption trends in agriculture and forestry

In 2010 world agricultural land occupied 4889 Mha, an increase of 7 % (311 Mha) since 1970 (FAOSTAT, 2013). Agricultural land area has decreased by 53 Mha since 2000 due to a decline of the cropland area (Organisation for Economic Co-operation and Development (OECD)-1990, Economies in Transition (EIT)) and a decrease in permanent meadows and pastures (OECD-1990 and Asia). The average amount of cropland and pasture land per capita in 1970 was 0.4 and 0.8 ha and by 2010 this had decreased to 0.2 and 0.5 ha per capita, respectively (FAOSTAT, 2013).

Changing land-use practices, technological advancement and varietal improvement have enabled world grain harvests to double from 1.2 to 2.5 billion tonnes per year between 1970 and 2010 (FAOSTAT, 2012). Average world cereal yields increased from 1600 to 3030 kg/ha over the same period (FAOSTAT, 2012) while there has also been a 233 % increase in global fertilizer use from 32 to 106 Mt/yr, and a 73 % increase in the irrigated cropland area (FAOSTAT, 2013).

Globally, since 1970, there has been a 1.4-fold increase in the numbers of cattle and buffalo, sheep and goats (which is closely linked to the trend of CH_4 emissions in the sector; Section 11.2.2), and increases of 1.6- and 3.7-fold for pigs and poultry, respectively (FAOSTAT, 2013). Major regional trends between 1970 and 2010 include a decrease in the total number of animals in Economies in Transition (EIT) and OECD-1990 (except poultry), and continuous growth in other regions, particularly Middle East and Africa (MAF) and Asia (Figure 11.3, bottom panel). The soaring demand for fish has led to the intensification of freshwater and marine fisheries worldwide, and an increased freshwater fisheries catch that topped 11 Mt in 2010, although the marine fisheries catch has slowly declined (78 Mt in 2010; FAOSTAT, 2013). The latter is, however, compensated in international markets by tremendous growth of aquaculture production to 60 Mt wet weight in 2010, of which 37 Mt originate from freshwater, overwhelmingly in Asia (FAOSTAT, 2013).

Between 1970 and 2010, global daily per capita food availability, expressed in energy units, has risen from 10,008 to 11,850 kJ (2391 to 2831 kcal), an increase of 18.4 %; growth in MAF (10,716 kJ in 2010) has been 22 %, and in Asia, 32 % (11,327 kJ in 2010; FAOSTAT, 2013). The percentage of animal products in daily per capita total food consumption has increased consistently in Asia since 1970 (7 to 16 %), remained constant in MAF (8 %) and, since 1985, has decreased in OECD-1990 countries (32 to 28 %), comprising, respectively, 1,790, 870 and 3,800 kJ in 2010 (FAOSTAT, 2013).

11.2.2 Trends of GHG emissions from agriculture

Organic and inorganic material provided as inputs or output in the management of agricultural systems are typically broken down through bacterial processes, releasing significant amounts of CO_2, CH_4, and N_2O to the atmosphere. Only agricultural non-CO_2 sources are reported as anthropogenic GHG emissions, however. The CO_2 emitted is considered neutral, being associated to annual cycles of carbon fixation and oxidation through photosynthesis. The agricultural sector is the largest contributor to global anthropogenic non-CO_2 GHGs, accounting for 56 % of emissions in 2005 (U.S. EPA, 2011). Other important, albeit much smaller non-CO_2 emissions sources from other AFOLU categories, and thus not treated here, include fertilizer applications in forests. Annual total non-CO_2 GHG emissions from agriculture in 2010 are estimated to be 5.2–5.8 $GtCO_2eq$/yr (FAOSTAT, 2013; Tubiello et al., 2013) and comprised about 10–12 % of global anthropogenic emissions. Fossil fuel CO_2 emissions on croplands added another

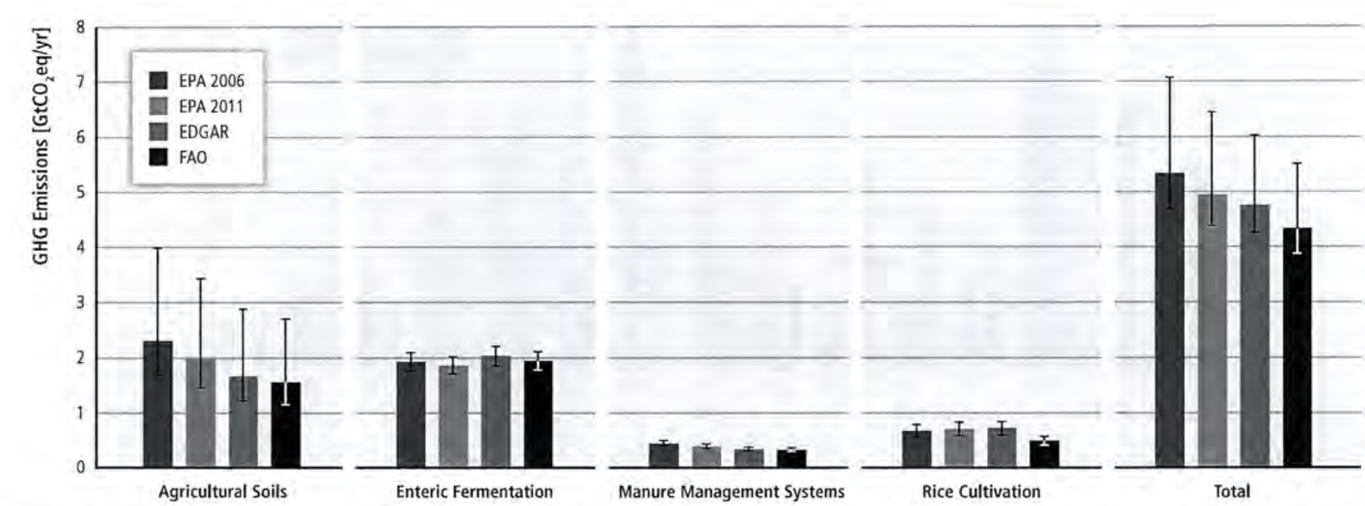

Figure 11.4 | Data comparison between FAOSTAT (2013), U.S. EPA (2006), and EDGAR (JRC/PBL, 2013) databases for key agricultural emission categories, grouped as agricultural soils, enteric fermentation, manure management systems, and rice cultivation, for 2005 | Whiskers represent 95 % confidence intervals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

0.4–0.6 $GtCO_2eq$/yr in 2010 from agricultural use in machinery, such as tractors, irrigation pumps, etc. (Ceschia et al., 2010; FAOSTAT, 2013), but these emissions are accounted for in the energy sector rather than the AFOLU sector. Between 1990 and 2010, agricultural non-CO_2 emissions grew by 0.9 %/yr, with a slight increase in growth rates after 2005 (Tubiello et al., 2013).

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

The following analyses refer to annual total non-CO_2 emissions by all categories. All three databases agree that that enteric fermentation and agricultural soils represent together about 70 % of total emissions, followed by paddy rice cultivation (9–11 %), biomass burning (6–12 %) and manure management (7–8 %). If all emission categories are disaggregated, both EDGAR and FAOSTAT agree that the largest emitting categories after enteric fermentation (32–40 % of total agriculture emissions) are manure deposited on pasture (15 %) and synthetic fertilizer (12 %), both contributing to emissions from agricultural soils. Paddy rice cultivation (11 %) is a major source of global CH_4 emissions, which in 2010 were estimated to be 493–723 $MtCO_2eq$/yr. The lower end of the range corresponds to estimates by FAO (FAOSTAT, 2013), with EDGAR and US EPA data at the higher end. Independent analyses suggest that emissions from rice may be at the lower end of the estimated range (Yan et al., 2009).

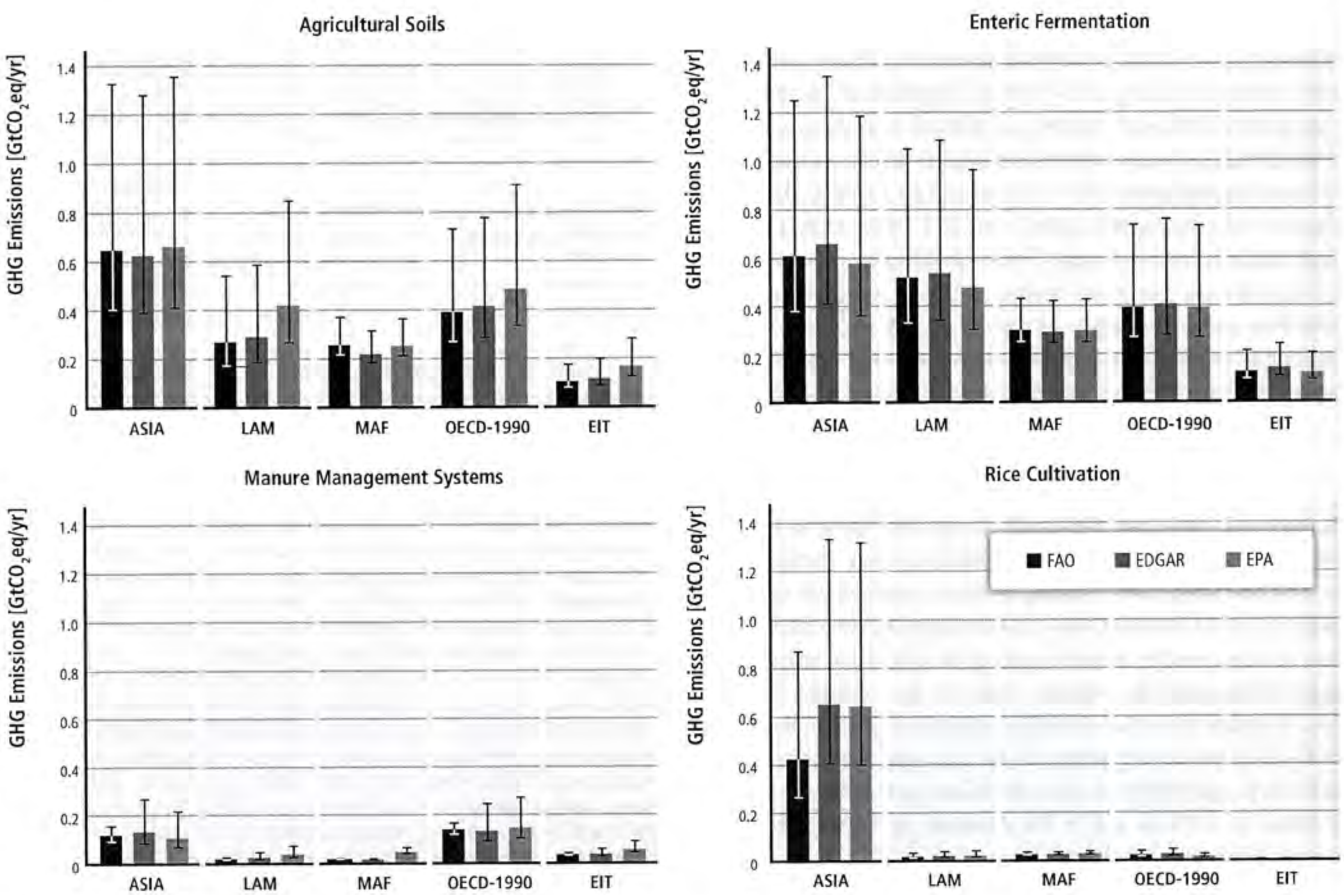

Figure 11.5 | Regional data comparisons for key agricultural emission categories in 2010 | Whiskers represent 95 % confidence intervals computed using IPCC guidelines (IPCC, 2006; Tubiello et al., 2013). The data show that most of the differences between regions and databases are of the same magnitude as the underlying emission uncertainties. [FAOSTAT, 2013; JRC/PBL, 2013; U.S. EPA, 2013]

Enteric Fermentation. Global emissions of this important category grew from 1.4 to 2.1 $GtCO_2eq/yr$ between 1961 and 2010, with average annual growth rates of 0.70 % (FAOSTAT, 2013). Emission growth slowed during the 1990s compared to the long-term average, but became faster again after the year 2000. In 2010, 1.0–1.5 $GtCO_2eq/yr$ (75 % of the total emissions), were estimated to come from developing countries (FAOSTAT, 2013). Over the period 2000–2010, Asia and the Americas contributed most, followed by Africa and Europe (FAOSTAT, 2013); see Figure 11.5). Emissions have grown most in Africa, on average 2.4 %/yr. In both Asia (2.0 %/yr) and the Americas (1.1 %/yr), emissions grew more slowly, and decreased in Europe (–1.7 %/yr). From 2000 to 2010, cattle contributed the largest share (75 % of the total), followed by buffalo, sheep and goats (FAOSTAT, 2013).

Manure. Global emissions from manure, as either organic fertilizer on cropland or manure deposited on pasture, grew between 1961 and 2010 from 0.57 to 0.99 $GtCO_2eq/yr$. Emissions grew by 1.1 %/yr on average. Manure deposited on pasture led to far larger emissions than manure applied to soils as organic fertilizer, with 80 % of emissions from deposited manures coming from developing countries (FAOSTAT, 2013; Herrero et al., 2013). The highest emitting regions from 2000–2010 were the Americas, Asia and Africa. Growth over the same period was most pronounced in Africa, with an average of 2.5 %/yr, followed by Asia (2.3 %/yr), and the Americas (1.2 %/yr), while there was a decrease in Europe of –1.2 %/yr. Two-thirds of the total came from grazing cattle, with smaller contributions from sheep and goats. In this decade, emissions from manure applied to soils as organic fertilizer were greatest in Asia, then in Europe and the Americas. Though the continent with the highest growth rates of 3.4 %/yr, Africa's share in total emissions remained small. In this sub-category, swine and cattle contributed more than three quarters (77 %) of the emissions. Emissions from manure management grew from 0.25 to 0.36 $GtCO_2eq/yr$, resulting in average annual growth rates of only 0.6 %/yr during the period 1961–2010. From 2000–2010 most emissions came from Asia, then Europe, and the Americas (Figure 11.5).

Synthetic Fertilizer. Emissions from synthetic fertilizers grew at an average rate of 3.9 %/yr from 1961 to 2010, with absolute values increasing more than 9-fold, from 0.07 to 0.68 $GtCO_2eq/yr$ (Tubiello et al., 2013). Considering current trends, synthetic fertilizers will become a larger source of emissions than manure deposited on pasture in less than 10 years and the second largest of all agricultural emission categories after enteric fermentation. Close to three quarters (70 %) of these emissions were from developing countries in 2010. In the decade 2000–2010, the largest emitter by far was Asia, then the Americas and then Europe (FAOSTAT, 2012). Emissions grew in Asia by 5.3 %/yr, in Africa by 2.0 %/yr, and in the Americas by 1.5 %/yr. Emissions decreased in Europe (–1.8 %/yr).

Rice. Emissions from rice are limited to paddy rice cultivation. From 1961 to 2010, global emissions increased with average annual growth rates of 0.4 %/yr (FAOSTAT, 2013) from 0.37 to 0.52 $GtCO_2eq/yr$. The growth in global emissions has slowed in recent decades, consistent with trends in rice cultivated area. During 2000–2010, the largest share of emissions (94 %) came from developing countries, with Asia being responsible for almost 90 % of the total (Figure 11.5). The largest growth of emissions took place in in Africa (2.7 %/yr), followed by Europe (1.4 %/yr). Growth rates in Asia and the Americas were much smaller over the same period (0.4–0.7 %/yr).

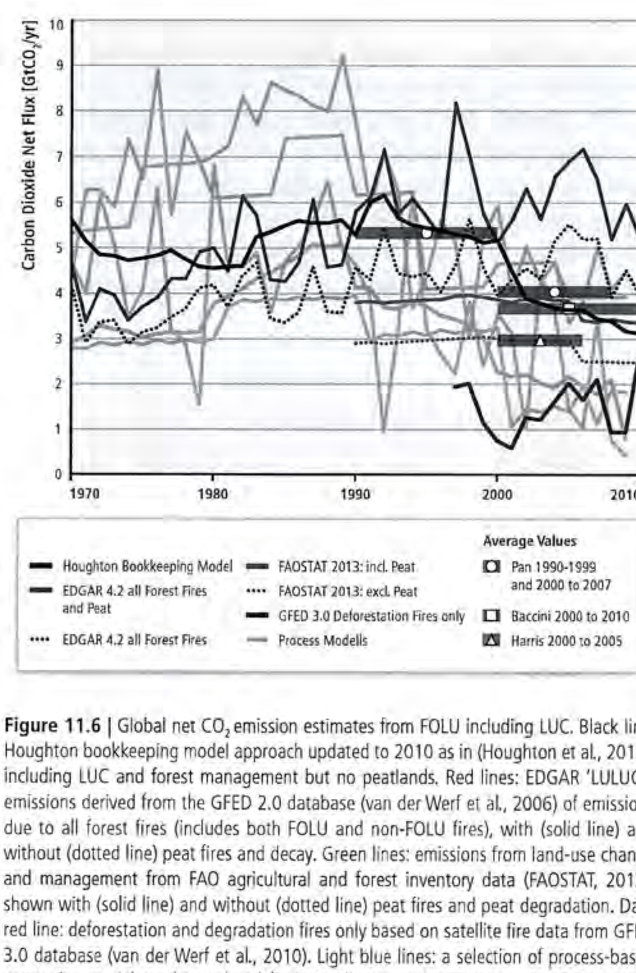

Figure 11.6 | Global net CO_2 emission estimates from FOLU including LUC. Black line: Houghton bookkeeping model approach updated to 2010 as in (Houghton et al., 2012), including LUC and forest management but no peatlands. Red lines: EDGAR 'LULUCF' emissions derived from the GFED 2.0 database (van der Werf et al., 2006) of emissions due to all forest fires (includes both FOLU and non-FOLU fires), with (solid line) and without (dotted line) peat fires and decay. Green lines: emissions from land-use change and management from FAO agricultural and forest inventory data (FAOSTAT, 2013), shown with (solid line) and without (dotted line) peat fires and peat degradation. Dark red line: deforestation and degradation fires only based on satellite fire data from GFED 3.0 database (van der Werf et al., 2010). Light blue lines: a selection of process-based vegetation model results, updated for WGI Chapter 6; (Le Quéré et al., 2013) include LUC, some include forest management, none include peatlands. LPJ-wsl: (Poulter et al., 2010); BernCC: (Stocker et al., 2011); VISIT: (Kato et al., 2011); ISAM: (Jain et al., 2013), IMAGE 2.4 (Van Minnen et al., 2009, deforestation only). The symbols and transparent rectangles represent mean values for the tropics only. Circles: tropical deforestation and forest management (Pan et al., 2011), using the Houghton (2003) bookkeeping model approach and FAO data. Triangle: tropical deforestation only, based on satellite forest area and biomass data (Baccini et al., 2012; Harris et al., 2012). Square: tropical deforestation and forest management, based on satellite forest area and biomass data and FAO data using bookkeeping model (Baccini et al., 2012; Harris et al., 2012).

Table 11.1 | Net global CO_2 flux from AFOLU.

	1750–2011 Cumulative $GtCO_2$			1980–1989 $GtCO_2$/yr			1990–1999 $GtCO_2$/yr			2000–2009 $GtCO_2$/yr		
IPCC WGI Carbon Budget, Table 6.1[a]:												
Net AFOLU CO_2 flux[b]	660	±	293	5.13	±	2.93	5.87	±	2.93	4.03	±	2.93
Residual terrestrial sink[c]	−550	±	330	−5.50	±	4.03	−9.90	±	4.40	−9.53	±	4.40
Fossil fuel combustions and cement production[d]	1338	±	110	20.17	±	1.47	23.47	±	1.83	28.60	±	2.20
Meta-analyses of net AFOLU CO_2 flux:												
WGI, Table 6.2[e]				4.77	±	2.57	4.40	±	2.20	2.93	±	2.20
Houghton et al., 2012[f]				4.18	±	1.83	4.14	±	1.83	4.03	±	1.83

Notes: Positive fluxes represent net emissions and negative fluxes represent net sinks.

(a) Selected components of the carbon budget in IPCC WGI AR5, Chapter 6, Table 6.1.

(b) From the bookkeeping model accounting method of Houghton (2003) updated in Houghton et al., (2012), uncertainty based on expert judgement; 90 % confidence uncertainty interval.

(c) Calculated as residual of other terms in the carbon budget.

(d) Fossil fuel flux shown for comparison (Boden et al., 2011).

(e) Average of estimates from 12 process models, only 5 were updated to 2009 and included in the 2000–2009 mean. Uncertainty based on standard deviation across models, 90 % confidence uncertainty interval (WGI Chapter 6).

(f) Average of 13 estimates including process models, bookkeeping model and satellite/model approaches, only four were updated to 2009 and included in the 2000–2009 mean. Uncertainty based on expert judgment.

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

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

FOLU accounted for about a third of anthropogenic CO_2 emissions from 1750 to 2011 and 12 % of emissions in 2000 to 2009 (Table 11.1). At the same time, atmospheric measurements indicate the land as a whole was a net sink for CO_2, implying a 'residual' terrestrial sink offsetting FOLU emissions (Table 11.1). This sink is confirmed by inventory measurements in both managed and unmanaged forests in temperate and tropical regions (Phillips et al., 1998; Luyssaert et al., 2008; Lewis et al., 2009; Pan et al., 2011). A sink of the right order of magnitude has been accounted for in models as a result of the indirect effects of human activity on ecosystems, i.e., the fertilizing effects of increased levels of CO_2 and N in the atmosphere and the effects of climate change (WGI Chapter 6; (Le Quéré et al., 2013), although some of it may be due to direct AFOLU activities not accounted for in current estimates (Erb et al., 2013). This sink capacity of forests is relevant to AFOLU mitigation through forest protection.

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

Regional trends in FOLU CO_2 emissions are shown in Figure 11.7. Model results indicate FOLU emissions peaked in the 1980s in Asia and LAM regions and declined thereafter. This is consistent with a reduced rate of deforestation, most notably in Brazil[4], and some areas of afforestation, the latter most notably in China, Vietnam and India (FAOSTAT, 2013). In MAF the picture is mixed, with the Houghton model (Houghton et al., 2012) showing a continuing increase from the 1970s to the 2000s, while the VISIT model (Kato et al., 2011) indicates a small sink in the 2000s. The results for temperate and boreal areas represented by OECD and EIT regions are very mixed ranging from large net sources (ISAM) to small net sinks. The general picture in temperate and boreal regions is of declining emissions and/or increasing sinks. These regions include large areas of managed forests subjected to harvest and regrowth, and areas of reforestation (e.g., following cropland abandonment in the United States and Europe). Thus results are sensitive to whether and how the models include forest management and environmental effects on regrowing forests.

[3] The term 'forestry and other land use' used here, is consistent with AFOLU in the (IPCC, 2006) Guidelines and consistent with LULUCF (IPCC, 2003).

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

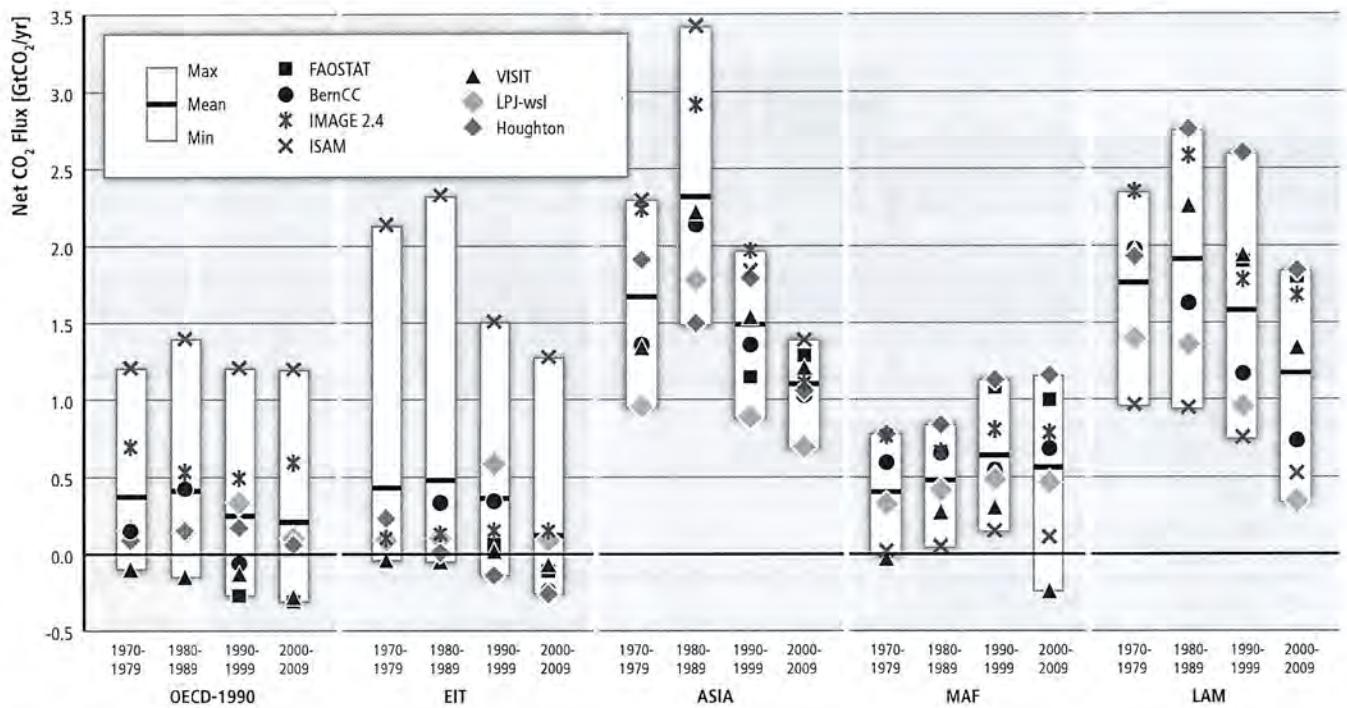

Figure 11.7 | Regional trends in net CO_2 fluxes from FOLU (including LUC). Houghton bookkeeping model approach updated to 2010 as in Houghton et al., (2012) and five process-based vegetation models updated to 2010 for WGI Chapter 6; (Le Quéré et al., 2013): LPJ-wsl: (Poulter et al., 2010); BernCC: (Stocker et al., 2011); VISIT: (Kato et al., 2011); ISAM: (Jain et al., 2013), IMAGE 2.4: ((Van Minnen et al., 2009), deforestation only). Only the FAO estimates (FAOSTAT, 2013) include peatlands.

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

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

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

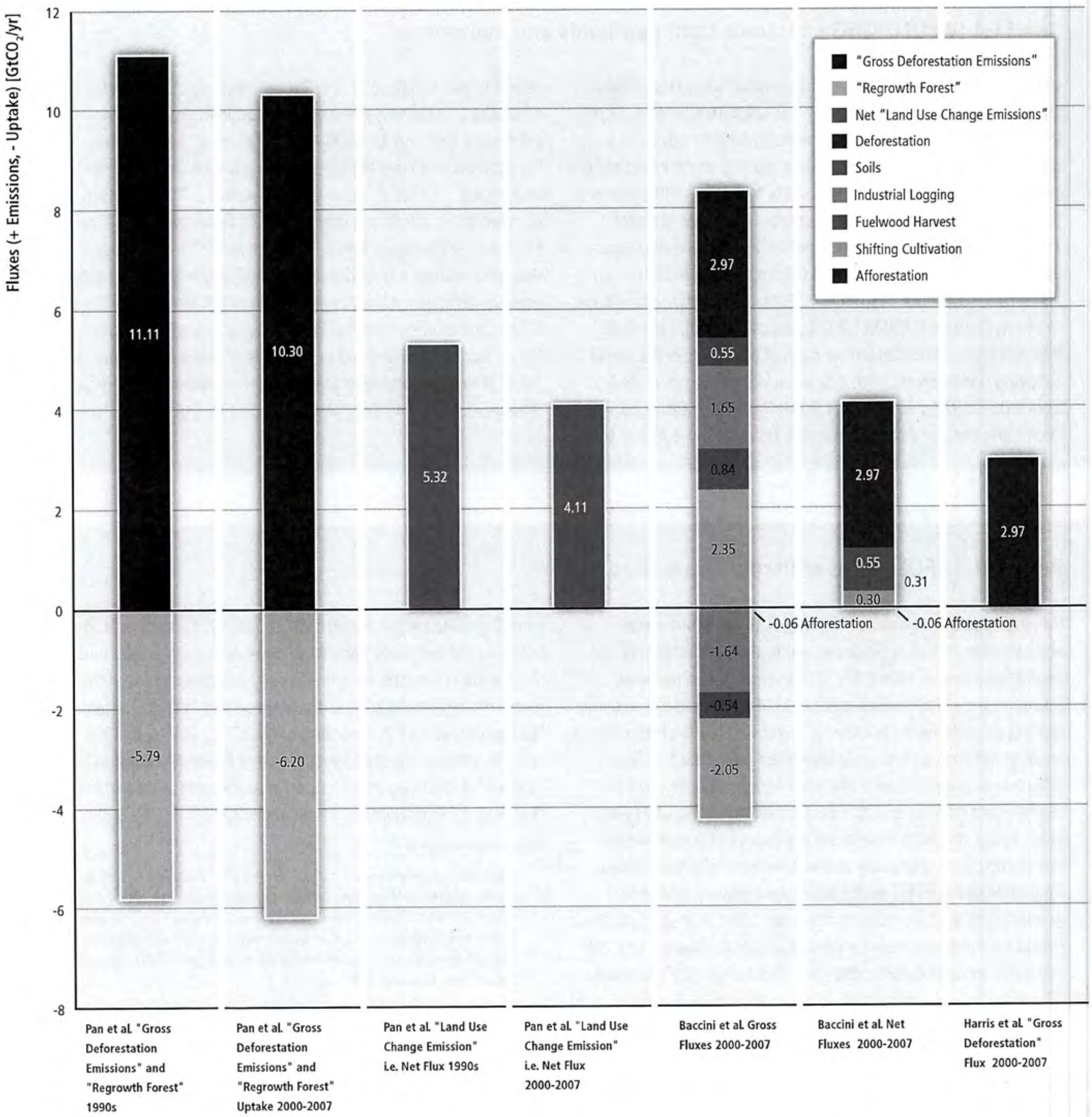

Figure 11.8 | Breakdown of mean annual CO_2 fluxes from deforestation and forest management in tropical countries ($GtCO_2$/yr). Pan et al. (2011) estimates are based on FAO data and the Houghton bookkeeping model (Houghton, 2003). Baccini et al. (2012) estimates are based on satellite land cover change and biomass data with FAO data, and the Houghton (2003) bookkeeping model, with the detailed breakdown of these results shown in Houghton, (2013). Harris et al. (2012) estimates are based on satellite land cover change and biomass data.

11

Box 11.1 | AFOLU GHG emissions from peatlands and mangroves

Undisturbed waterlogged peatlands (organic soils) store a large amount of carbon and act as small net sinks (Hooijer et al., 2010). Drainage of peatlands for agriculture and forestry results in a rapid increase in decomposition rates, leading to increased emissions of CO_2, and N_2O, and vulnerability to further GHG emissions through fire. The FAO emissions database estimates globally 250,000 km² of drained organic soils under cropland and grassland, with total GHG emissions of 0.9 $GtCO_2eq/yr$ in 2010—with the largest contributions from Asia (0.44 $GtCO_2eq/yr$) and Europe (0.18 $GtCO_2eq/yr$) (FAOSTAT, 2013). Joosten (2010), estimated that there are > 500,000 km² of drained peatlands in the world including under forests, with CO_2 emissions having increased from 1.06 $GtCO_2/yr$ in 1990 to 1.30 $GtCO_2/yr$ in 2008, despite a decreasing trend in Annex I countries, from 0.65 to 0.49 $GtCO_2/yr$, primarily due to natural and artificial rewetting of peatlands. In Southeast Asia, CO_2 emissions from drained peatlands in 2006 were 0.61 ± 0.25 $GtCO_2/yr$ (Hooijer et al., 2010). Satellite estimates indicate that peat fires in equatorial Asia emitted on average 0.39 $GtCO_2$ eq/yr over the period 1997–2009 (van der Werf et al., 2010), but only 0.2 $GtCO_2$ eq/yr over the period 1998–2009. This lower figure is consistent with recent independent FAO estimates over the same period and region. Mangrove ecosystems have declined in area by 20 % (36 Mha) since 1980, although the rate of loss has been slowing in recent years, reflecting an increased awareness of the value of these ecosystems (FAO, 2007). A recent study estimated that deforestation of mangroves released 0.07 to 0.42 $GtCO_2/yr$ (Donato et al., 2011).

Box 11.2. | AFOLU GHG emissions from fires

Burning vegetation releases CO_2, CH_4, N_2O, ozone-precursors and aerosols (including black carbon) to the atmosphere. When vegetation regrows after a fire, it takes up CO_2 and nitrogen. Anthropogenic land management or land conversion fire activities leading to permanent clearance or increasing levels of disturbance result in net emissions to the atmosphere over time. Satellite-detection of fire occurrence and persistence has been used to estimate fire emissions (e.g., GFED 2.0 database; (van der Werf et al., 2006). It is hard to separate the causes of fire as natural or anthropogenic, especially as the drivers are often combined. An update of the GFED methodology now distinguishes FOLU deforestation and degradation fires from other management fires (GFED 3.0 database; (van der Werf et al., 2010); Figure 11.6). The estimated tropical deforestation and degradation fire emissions were 1.39 $GtCO_2eq/yr$ during 1997 to 2009 (total carbon including CO_2, CH_4, CO and black carbon), 20 % of all fire emissions. Carbon dioxide FOLU fire emissions are already included as part of the global models results such as those presented in Table 1.1 and Figures 11.6 and 11.7. According to (FAOSTAT, 2013)[1], in 2010 the non-CO_2 component of deforestation and forest degradation fires totalled 0.1 $GtCO_2eq/yr$, with forest management and peatland fires (Box 11.1) responsible for an additional 0.2 $GtCO_2eq/yr$.

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

(2011) separated 'gross emissions' from deforestation and forest management on the one hand, from uptake in regrowing vegetation on the other. Deforestation emissions decline from the 1990s to 2000–2007, and uptake in regrowing vegetation increases, both contributing to the decline in net tropical CO_2 emissions.

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

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

FAO estimates AFOLU GHG emissions (FAOSTAT, 2013)[6] based on IPCC Tier 1 methodology[7]. With reference to the decade 2001–2010, total GHG FOLU emissions were 3.2 $GtCO_2eq$/yr including deforestation (3.8 $GtCO_2eq$/yr), forest degradation and forest management (−1.8 $GtCO_2eq$/yr), biomass fires including peatland fires (0.3 $GtCO_2eq$/yr), and drained peatlands (0.9 $GtCO_2eq$/yr). The FAO estimated total mean net GHG FOLU flux to the atmosphere decreased from 3.9 $GtCO_2eq$/yr in 1991–2000 to 3.2 $GtCO_2eq$/yr in 2001–2010 (FAOSTAT, 2013).

11.3 Mitigation technology options and practices, and behavioural aspects

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

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

- Reductions in CH_4 or N_2O emissions from croplands, grazing lands, and livestock.
- Conservation of existing carbon stocks, e.g., conservation of forest biomass, peatlands, and soil carbon that would otherwise be lost.
- Reductions of carbon losses from biota and soils, e.g., through management changes within the same land-use type (e.g., reducing soil carbon loss by switching from tillage to no-till cropping) or by reducing losses of carbon-rich ecosystems, e.g., reduced deforestation, rewetting of drained peatlands.
- Enhancement of carbon sequestration in soils, biota, and long-lived products through increases in the area of carbon-rich ecosystems such as forests (afforestation, reforestation), increased carbon storage per unit area, e.g., increased stocking density in forests, carbon sequestration in soils, and wood use in construction activities.
- Changes in albedo resulting from land-use and land-cover change that increase reflection of visible light.
- Provision of products with low GHG emissions that can replace products with higher GHG emissions for delivering the same service (e.g., replacement of concrete and steel in buildings with wood, some bioenergy options; see Section 11.13).
- Reductions of direct (e.g., agricultural machinery, pumps, fishing craft) or indirect (e.g., production of fertilizers, emissions resulting from fossil energy use in agriculture, fisheries, aquaculture, and forestry or from production of inputs); though indirect emission reductions are accounted for in the energy end-use sectors (buildings, industry, energy generation, transport) so are not discussed further in detail in this chapter.

11.3.1 Supply-side mitigation options

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

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

6 http://faostat.fao.org/

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

Table 11.2 | Summary of supply-side mitigation options in the AFOLU sector. Technical Mitigation Potential: Area = $(tCO_2eq/ha)/yr$; Animal = percent reduction of enteric emissions. Low = < 1; < 5 % (white), Medium = 1–10; 5–15 % (light grey), High = > 10, > 15 % (grey); Ease of Implementation (acceptance or adoption by land manager): Difficult (white), Medium (light grey), Easy, i. e., universal applicability (grey); Timescale for Implementation: Long-term (at research and development stage; white), Mid-term (trials in place, within 5–10 years; light grey), Immediate (technology available now, grey).

Categories	Practices and Impacts	Technical Mitigation Potential	Ease of Implementation	Timescale for implementation	References
Forestry					
Reducing deforestation	C: Conservation of existing C pools in forest vegetation and soil by controlling deforestation protecting forest in reserves, and controlling other anthropogenic disturbances such as fire and pest outbreaks. Reducing slash and burn agriculture, reducing forest fires.				1
	CH_4, N_2O: Protection of peatland forest, reduction of wildfires.				2
Afforestation/Reforestation	C: Improved biomass stocks by planting trees on non-forested agricultural lands. This can include either monocultures or mixed species plantings. These activities may also provide a range of other social, economic, and environmental benefits.				3, 4, 5
Forest management	C: Management of forests for sustainable timber production including extending rotation cycles, reducing damage to remaining trees, reducing logging waste, implementing soil conservation practices, fertilization, and using wood in a more efficient way, sustainable extortion of wood energy				6, 7, 8, 9
	CH_4, N_2O: Wildfire behaviour modification.				10, 11, 12
Forest restoration	C: Protecting secondary forests and other degraded forests whose biomass and soil C densities are less than their maximum value and allowing them to sequester C by natural or artificial regeneration, rehabilitation of degraded lands, long-term fallows.				13, 14
	CH_4, N_2O : Wildfire behaviour modification.				
Land-based agriculture					
Cropland management					
Croplands—plant management	C: High input carbon practices, e.g., improved crop varieties, crop rotation, use of cover crops, perennial cropping systems, agricultural biotechnology.				15, 16, 17
	N_2O: Improved N use efficiency.				18
Croplands—nutrient management	C: Fertilizer input to increase yields and residue inputs (especially important in low-yielding agriculture).				19, 20
	N_2O: Changing N fertilizer application rate, fertilizer type, timing, precision application, inhibitors.				21, 22, 23, 24, 25, 105, 106
Croplands—tillage/residues management	C: Reduced tillage intensity; residue retention.				17, 24, 26, 27
	N_2O:				28, 96, 97
	CH_4:				96
Croplands—water management	C: Improved water availability in cropland including water harvesting and application.				29
	CH_4: Decomposition of plant residues.				
	N_2O: Drainage management to reduce emissions, reduce N runoff leaching.				
Croplands—rice management	C: Straw retention.				30
	CH_4: Water management, mid-season paddy drainage.				31, 32, 98
	N_2O: Water management, N fertilizer application rate, fertilizer type, timing, precision application.				32, 98, 99
Rewet peatlands drained for agriculture	C: Ongoing CO_2 emissions from reduced drainage (but CH_4 emissions may increase).				33
Croplands—set-aside and LUC	C: Replanting to native grasses and trees. Increase C sequestration.				34, 35, 36, 37, 38
	N_2O: N inputs decreased resulting in reduced N_2O.				
Biochar application	C: Soil amendment to increase biomass productivity, and sequester C (biochar was not covered in AR4 so is described in Box 11.3).				39, 40, 41
	N_2O: Reduced N inputs will reduce emissions.				39, 42

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Categories	Practices and Impacts	Technical Mitigation Potential	Ease of Implementation	Timescale for implementation	References
Grazing Land Management					
Grazing lands—plant management	C: Improved grass varieties/sward composition, e.g., deep rooting grasses, increased productivity, and nutrient management. Appropriate stocking densities, carrying capacity, fodder banks, and improved grazing management.				43, 44, 45
	N_2O				46
Grazing lands—animal management	C: Appropriate stocking densities, carrying capacity management, fodder banks and improved grazing management, fodder production, and fodder diversification.				43, 47
	CH_4				
	N_2O: Stocking density, animal waste management.				
Grazing land—fire management	C: Improved use of fire for sustainable grassland management. Fire prevention and improved prescribed burning.				
Revegetation					
Revegetation	C: The establishment of vegetation that does not meet the definitions of afforestation and reforestation (e.g., *Atriplex* spp.).				48
	CH_4: Increased grazing by ruminants may increase net emissions.				
	N_2O: Reduced N inputs will reduce emissions.				
Other					
Organic soils—restoration	C: Soil carbon restoration on peatlands; and avoided net soil carbon emissions using improved land management.				49
	CH_4: May increase.				
Degraded soils—restoration	Land reclamation (afforestation, soil fertility management, water conservation soil nutrients enhancement, improved fallow).				100, 101, 102, 103, 104
Biosolid applications	C: Use of animal manures and other biosolids for improved management of nitrogen; integrated livestock agriculture techniques.				26
	N_2O				
Livestock					
Livestock—feeding	CH_4: Improved feed and dietary additives to reduce emissions from enteric fermentation; including improved forage, dietary additives (bioactive compounds, fats), ionophores/antibiotics, propionate enhancers, archaea inhibitors, nitrate and sulphate supplements.				50, 51, 52, 53, 54, 55, 56, 57, 58, 59
Livestock—breeding and other long-term management	CH_4: Improved breeds with higher productivity (so lower emissions per unit of product) or with reduced emissions from enteric fermentation; microbial technology such as archaeal vaccines, methanotrophs, acetogens, defaunation of the rumen, bacteriophages and probiotics; improved fertility.				54, 55, 56, 58, 60, 61, 62, 63, 64, 65, 66, 67, 68, 69, 70, 71
Manure management	CH_4: Manipulate bedding and storage conditions, anaerobic digesters; biofilters, dietary additives.				56, 58, 72, 73
	N_2O: Manipulate livestock diets to reduce N excreta, soil applied and animal fed nitrification inhibitors, urease inhibitors, fertilizer type, rate and timing, manipulate manure application practices, grazing management.				56, 58, 72, 74, 75, 76, 77, 78
Integrated systems					
Agroforestry (including agropastoral and agrosilvopastoral systems)	C: Mixed production systems can increase land productivity and efficiency in the use of water and other resources and protect against soil erosion as well as serve carbon sequestration objectives.				79, 80, 81, 82, 83, 84, 85, 86, 87, 88
	N_2O: Reduced N inputs will reduce emissions.				
Other mixed biomass production systems	C: Mixed production systems such as double-cropping systems and mixed crop-livestock systems can increase land productivity and efficiency in the use of water and other resources as well as serve carbon sequestration objectives. Perennial grasses (e.g., bamboo) can in the same way as woody plants be cultivated in shelter belts and riparian zones/buffer strips provide environmental services and supports C sequestration and biomass production.				82, 89, 90
	N_2O: Reduced N inputs will reduce emissions.				

⇒

Categories	Practices and Impacts	Technical Mitigation Potential	Ease of Implementation	Timescale for implementation	References
Integration of biomass production with subsequent processing in food and bioenergy sectors	**C**: Integrating feedstock production with conversion, typically producing animal feed that can reduce demand for cultivated feed such as soy and corn and can also reduce grazing requirements. Using agricultural and forestry residues for energy production.				91, 92, 93, 94, 95
	N_2O: Reduced N inputs will reduce emissions.				
Bioenergy (see Box 11.5 and Section 11.13)					

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

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

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

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

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

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

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

Saturation. Substitution of fossil fuel and material with biomass, and energy-intensive building materials with wood can continue in perpetuity. In contrast, it is often considered that carbon sequestration in soils (Guldea et al., 2008) or vegetation cannot continue indefinitely. The carbon stored in soils and vegetation reaches a new equilibrium (as the trees mature or as the soil carbon stock saturates). As the

Box 11.3 | Biochar

This box summarizes the mitigation potential for biochar technologies, which were not considered in AR4. Biomass C stabilization could be combined with (or substitute) bioenergy capture as part of a land-based mitigation strategy (Lehmann, 2007). Heating biomass with air excluded (pyrolysis) generates energy-containing volatiles and gases. Hydrogen and O are preferentially eliminated, creating a stable (biologically recalcitrant) C-rich co-product (char). By adding char to soil as 'biochar' a system can be established that may have a higher carbon abatement than typical bioenergy alternatives (Woolf et al., 2010). The gain is probably highest where efficient bioenergy is constrained by a remote, seasonal, or diffuse biomass resource (Shackley et al., 2012). The benefit of pyrolysis-biochar systems (PBS) is increased considerably if allowance is made for the indirect effects of using biochar *via* the soil. These effects include increased crop and biomass production and decreased N_2O and CH_4 emissions. Realizing the mitigation potential for biochar technologies will be constrained by the need for sustainable feedstock acquisition, competing biomass use options are an important influence of the production process on biochar properties. Considering sustainable feedstock production and targeting biochar deployment on less fertile land, Woolf et al. (2010) calculated maximum global abatement of 6.6 $GtCO_2eq/yr$ from 2.27 Gt biomass C. Allowing for competition for virgin non-waste biomass the value was lower (3.67 $GtCO_2eq/yr$ from 1.01 Gt biomass C), accruing 240–480 $GtCO_2eq$ abatement within 100 years.

Meta-analysis shows that in experimental situations crop productivity has, on average, been enhanced by circa 15 % near-term, but with a wide range of effects (Jeffery et al., 2011; Biederman and Harpole, 2013). This range is probably explained by the nature and extent of pre-existing soil constraints. The Woolf et al. (2010) analysis accordingly assumed crop yield increases of 0–90 % (relative). Relaxing this assumption by one-half decreased projected abatement by 10 %. Decreasing an assumed 25 % suppression on soil N_2O flux by the same proportion had a smaller impact. Beneficial interactions of biochar and the soil N cycle are beginning to be understood with effects on mineralization, nitrification, denitrification, immobilization and adsorption persisting at least for days and months after biochar addition (Nelissen et al., 2012; Clough et al., 2013). Although the often large suppression of soil N_2O flux observed under laboratory conditions can be increasingly explained (Cayuela et al., 2013), this effect is not yet predictable and there has been only limited validation of N_2O suppression by biochar in planted field soils (Liu et al., 2012; Van Zwieten et al., 2013) or over longer timeframes (Spokas, 2013). The potential to gain enhanced mitigation using biochar by tackling gaseous emissions from manures and fertilizers before and after application to soil are less well-explored (Steiner et al., 2010; Angst et al., 2013). The abatement potential for PBS remains most sensitive to the absolute stability of the C stored in biochar. Estimates of 'half-life' have been inferred from wildfire charcoal (Lehmann, 2007) or extrapolated from direct short-term observation. These give values that range from < 50 to > 10,000 years, but predominantly between 100–1000 years (Singh et al., 2012; Spokas, 2013). Nonetheless, the assumption made by Woolf et al. (2010) for the proportion of biochar C that is stable long-term (85 %) is subject to refinement and field validation.

Demonstration of the equipment and infrastructure required for effective use of energy products from biomass pyrolysis is still limited, especially across large and small unit scales. Preliminary analyses shows, however, that the break-even cost of biochar production is likely to be location- and feedstock-specific (Shackley et al., 2012; Field et al., 2013). Until economic incentives are established for the stabilization of C, biochar adoption will depend on predictable, positive effects on crop production. This requires more research on the use of biochar as a regular low-dose soil input, rather than single applications at rates > 10t/ha, which have so far been the norm (Sohi, 2012). Product standards are also required, to ensure that biochar is produced in a way that does not create or conserve problematic concentrations of toxic contaminants, and to support regulated deployment strategies (IBI Biochar, 2012; Downie et al., 2012).

soils/vegetation approach the new equilibrium, the annual removal (sometimes referred to as the sink strength) decreases until it becomes zero at equilibrium. This process is called saturation (Smith, 2005; Körner, 2006, 2009; Johnston et al., 2009b) , and the uncertainty associated with saturation has been estimated (Kim and McCarl, 2009). An alternative view is that saturation does not occur, with studies from old-growth forests, for example, showing that they can continue to sequester C in soil and dead organic matter even if net living biomass increment is near zero (e.g., Luyssaert et al., 2008). Peatlands are unlikely to saturate in carbon storage, but the rate of C uptake may be very slow (see Box 11.1).

Human and natural impacts. Soil and vegetation carbon sinks can be impacted upon by direct human-induced, indirect human-induced, and natural changes (Smith, 2005). All of the mitigation practices discussed in Section 11.3.1 arise from direct human-induced impacts (deliberate

Box 11.4 | Aquaculture

Aquaculture is defined as the farming of fish, shellfish, and aquatic plants (Hu et al., 2013). Although it is an ancient practice in some parts of world, this sector of the food system is growing rapidly. Since the mid-1970s, total aquaculture production has grown at an average rate of 8.3 % per year (1970–2008; (Hu et al., 2013). The estimated aquaculture production in 2009 was 55.10 Mt, which accounts for approximately 47 % of all the fish consumed by humans (Hu et al., 2013). The sector is diverse, being dominated by shellfish and herbivorous and omnivorous pond fish, either entirely or partly utilizing natural productivity, but globalizing trade and favourable economic conditions are driving intensive farming at larger scales (Bostock et al., 2010). Potential impacts of aquaculture, in terms emissions of N_2O, have recently been considered (Williams and Crutzen, 2010; Hu et al., 2012). Global N_2O emissions from aquaculture in 2009 were estimated to be 93 ktN_2O-N (~43 MtCO_2eq), and will increase to 383 ktN_2O-N (~178 MtCO_2eq) by 2030, which could account for 5.7 % of anthropogenic N_2O-N emissions if aquaculture continues to grow at the present growth rate (~7.1 %/yr; Hu et al., 2012).

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

management). Both sink processes and carbon stocks can be affected by natural factors such as soil and hydrological conditions. Indirect human-induced changes can impact carbon sinks and are influenced by human activity, but are not directly related to the management of that piece of land; examples include climate change and atmospheric nitrogen deposition. For some tree species, rising concentrations of tropospheric ozone caused by human activities may counteract the effects of increased atmospheric CO_2 or N deposition on tree growth (Sitch et al., 2008; Matyssek et al., 2010). Natural changes that threaten to impact the efficacy of mitigation measures are discussed in Section 11.5.

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

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

Box 11.5 | Bioenergy

Bioenergy deployment offers significant potential for climate change mitigation, but also carries considerable risks. The SRREN (IPCC, 2011) suggested potential bioenergy deployment levels to be between 100–300 EJ. This assessment agrees on a technical bioenergy potential of around 100 EJ, and possibly 300 EJ and higher. Integrated models project between 15–245 EJ/yr deployment in 2050, excluding traditional bioenergy. Achieving high deployment levels would require, amongst others, extensive use of agricultural residues and second-generation biofuels to mitigate adverse impacts on land use and food production, and the co-processing of biomass with coal or natural gas with carbon dioxide capture and storage (CCS) to produce low net GHG-emitting transportation fuels and/or electricity. Integration of crucial sectoral research (albedo effects, evaporation, counterfactual land carbon sink assumptions) into transformation pathways research, and exploration of risks of imperfect policy settings (for example, in absence of a global CO_2 price on land carbon) is subject of further research (Sections 11.9, 11.13.2, 11.13.4). Small-scale bioenergy systems aimed at meeting rural energy needs synergistically provide mitigation and energy access benefits. Decentralized deployment of biomass for energy, in combination with improved cookstoves, biogas, and small-scale biopower, could improve livelihoods and health of around 2.6 billion people. Both mitigation potential and sustainability hinges crucially on the protection of land carbon (high-density carbon ecosystems), careful fertilizer application, interaction with food markets, and good land and water management. Sustainability and livelihood concerns might constrain beneficial deployment of dedicated biomass plantations to lower values (Sections 11.13.3, 11.13.5, 11.13.7).

Lifecycle assessments for bioenergy options demonstrate a plethora of pathways, site-specific conditions and technologies that produce a wide range of climate-relevant effects. Specifically, LUC emissions, N_2O emissions from soil and fertilizers, co-products, process design and process fuel use, end-use technology, and reference system can all influence the total attributional lifecycle emissions of bioenergy use. The large variance for specific pathways points to the importance of management decisions in reducing the lifecycle emissions of bioenergy use. The total marginal global warming impact of bioenergy can only be evaluated in a comprehensive setting that also addresses equilibrium effects, e.g., indirect land-use change (iLUC) emissions, actual fossil fuel substitution, and other effects. Structural uncertainty in modelling decisions renders such evaluation exercises uncertain. Available data suggest a differentiation between options that offer low lifecycle emissions under good land-use management (e.g., sugarcane, Miscanthus, and fast-growing tree species) and those that are unlikely to contribute to climate change mitigation (e.g., corn and soybean), pending new insights from more comprehensive consequential analyses (Sections 8.7, 11.13.4).

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

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

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

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

11

11.4 Infrastructure and systemic perspectives

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

11.4.1 Land: a complex, integrated system

Mitigation in the AFOLU sector is embedded in the complex interactions between socioeconomic and natural factors simultaneously affecting land systems (Turner et al., 2007). Land is used for a variety of purposes, including housing and infrastructure (Chapter 12), production of goods and services through agriculture, aquaculture and forestry, and absorption or deposition of wastes and emissions (Dunlap and Catton, Jr., 2002). Agriculture and forestry are important for rural livelihoods and employment (Coelho et al., 2012), while aquaculture and fisheries can be regionally important (FAO, 2012). More than half of the planet's total land area

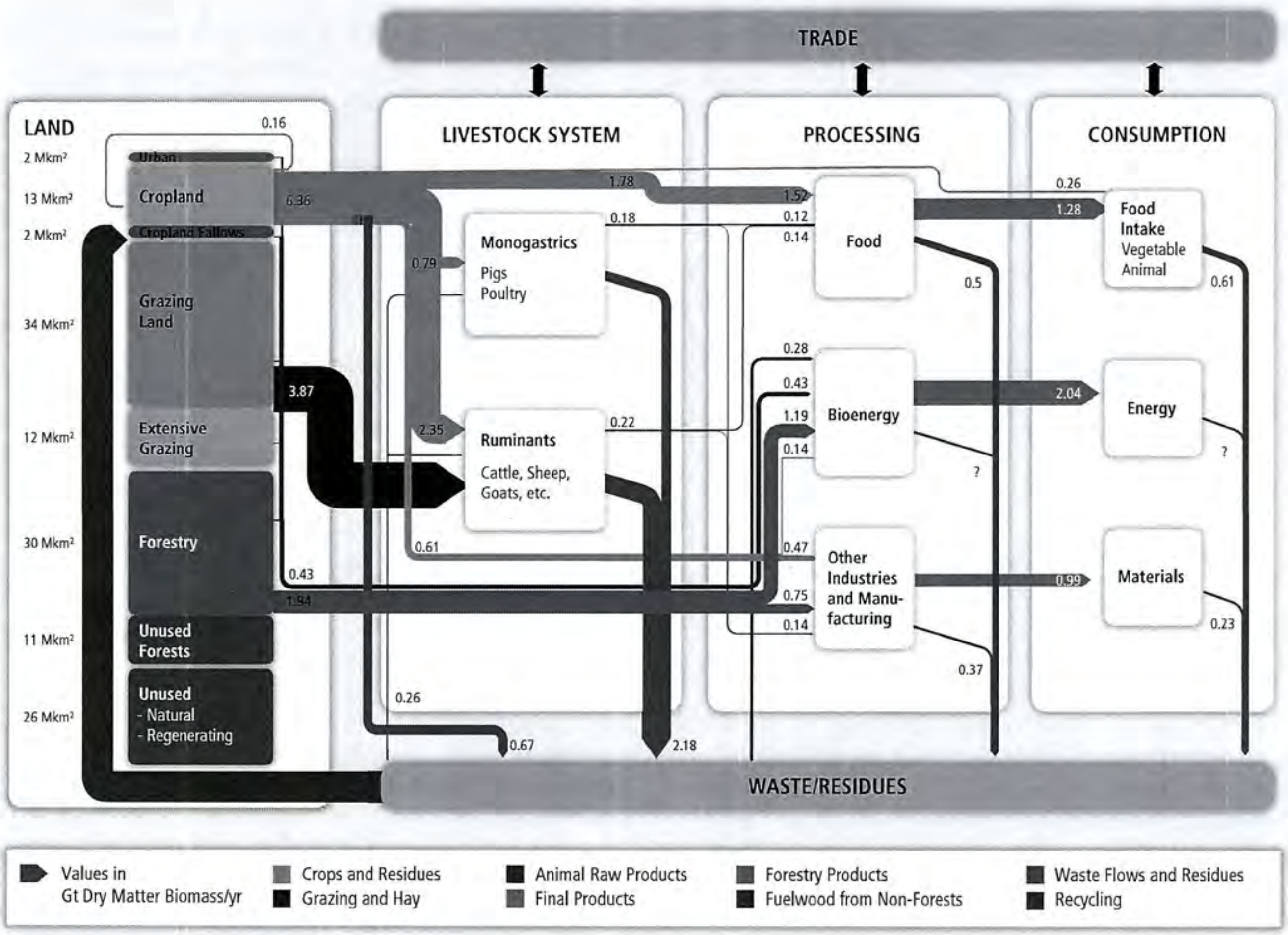

Figure 11.9 | Global land use and biomass flows arising from human economic activity in 2000 from the cradle to the grave. Values in Gt dry matter biomass/yr. Figure source: (Smith et al., 2013b). If a source reported biomass flows in energy units, the numbers were converted to dry matter assuming a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

(134 Mkm²) is used for urban and infrastructure land, agriculture, and forestry. Less than one quarter shows relatively minor signs of direct human use (Erb et al., 2007; Ellis et al., 2010; Figure 11.9). Some of the latter areas are inhabited by indigenous populations, which depend on the land for the supply of vitally important resources (Read et al., 2010).

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

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

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

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

11.4.2 Mitigation in AFOLU—feedbacks with land-use competition

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

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

- **Optimization of biomass-flow cascades**; that is, increased use of residues and by-products, recycling of biogenic materials and energetic use of wastes (WBGU, 2009). Such options increase resource use efficiency and may reduce competition, but there may also be tradeoffs. For example, using crop residues for bioenergy or roughage supply may leave less C and nutrients on cropland, reduce soil quality and C storage in soils, and increase the risk of losses of carbon through soil erosion. Residues are also often used as forage, particularly in the tropics. Forest residues are currently also used for other purposes, e.g., chipboard manufacture, pulp and paper production (González-Estrada et al., 2008; Blanco-Canqui and Lal, 2009; Muller, 2009; Ceschia et al., 2010).
- **Increases in yields** of cropland (Burney et al., 2010; Foley et al., 2011; Tilman et al., 2011; Mueller et al., 2012; Lobell et al., 2013), grazing land or forestry and improved livestock feeding efficiency (Steinfeld et al., 2010; Thornton and Herrero, 2010) can reduce land competition if yield increases relative to any additional inputs and the emission intensity (i.e., GHG emissions per unit of product) decreases. This may result in tradeoffs with other ecological, social, and economic costs (IAASTD, 2009) although these can to some extent be mitigated if intensification is sustainable (Tilman et al., 2011). Another caveat is that increases in yields may result in rebound effects that increase consumption (Lambin and Meyfroidt, 2011; Erb, 2012) or provide incentives to farm more land (Matson

11

and Vitousek, 2006), and hence may fail to spare land (Section 11.10).

- **Land-demanding options** reduce GHG emissions by harnessing the potential of the land for either C sequestration or growing energy crops (including food crops used as feedstocks for bioenergy production). These options result in competition for land (and sometimes other resources such as water) that may have substantial social, economic, and ecological effects (positive or negative; (UNEP, 2009; WBGU, 2009; Chum et al., 2011; Coelho et al., 2012). Such options may increase pressures on ecosystems (e.g., forests) and GHG emissions related to direct and indirect LUC, contribute to price increases of agricultural products, or negatively affect livelihoods of rural populations. These possible impacts need to be balanced against possible positive effects such as GHG reduction, improved water quality (Townsend et al., 2012), restoration of degraded land (Harper et al., 2007), biodiversity protection (Swingland et al., 2002), and job creation (Chum et al., 2011; Coelho et al., 2012).

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

11.4.3 Demand-side options for reducing GHG emissions from AFOLU

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

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

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

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

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

Losses ranging from 20 % in Sub-Saharan Africa to more than 30 % in the industrialized countries were calculated (Gustavsson et al., 2011).

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

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

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

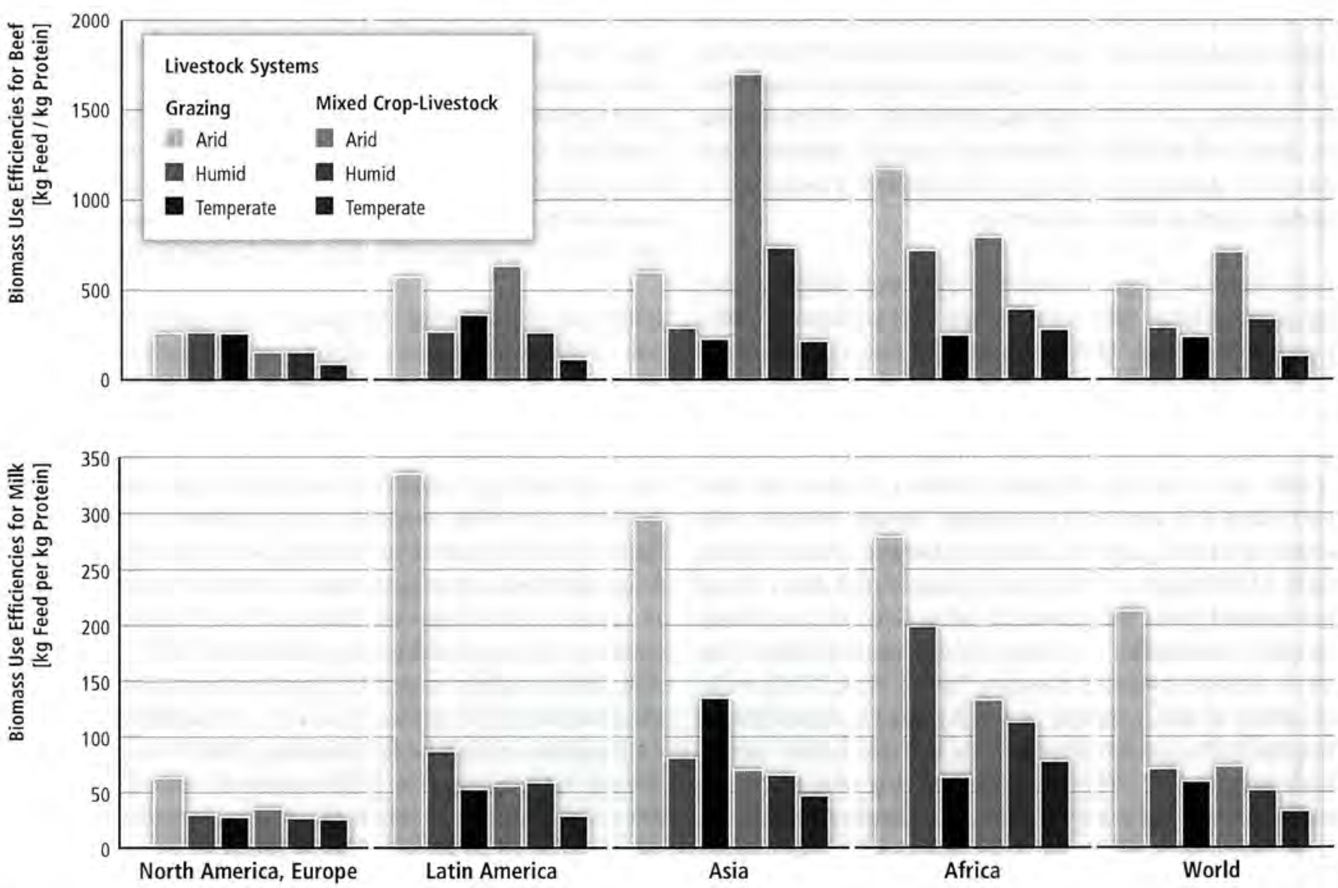

Figure 11.10 | Biomass use efficiencies for the production of edible protein from (top) beef and (bottom) milk for different production systems and regions of the world (Herrero et al., 2013).

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

Cases	Food crop area	Livestock grazing area	C sink on farmland[1]	Afforestation of spare land[2,3]	Biomass for bioenergy on spare land[2,4]	Total mitigation potential	Difference in mitigation from reference case
	[Gha]		$GtCO_2eq/yr$				
Reference	1.60	4.07	3.5	6.1	1.2–9.4	4.6–12.9	0
Diet change	1.38	3.87	3.2	11.0	2.1–17.0	5.3–20.2	0.7–7.3
Yield growth	1.49	4.06	3.4	7.3	1.4–11.4	4.8–14.8	0.2–1.9
Feeding efficiency	1.53	4.04	3.4	7.2	1.4–11.1	4.8–14.5	0.2–1.6
Waste reduction	1.50	3.82	3.3	10.1	1.9–15.6	5.2–18.9	0.6–6.0
Combined	1.21	3.58	2.9	16.5	3.2–25.6	6.1–28.5	1.5–15.6

Notes:

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

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

[3] Assuming 11.8 $(tCO_2eq/ha)/yr$ (Smith et al., 2000).

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

vary largely between different regions of the world due to differences in agro-ecology, diet quality, and intensity of production (Herrero et al., 2013). In overall terms, Europe and North America have lower emissions intensities per kg of protein than Africa, Asia, and Latin America. This shows that the highest potential for improving emissions intensities lies in developing countries, if intensification strategies can be matched to local resources and contexts.

Studies based on integrated modelling show that changes in diets strongly affect future GHG emissions from food production (Stehfest et al., 2009; Popp et al., 2010; Davidson, 2012). Popp et al. (2010) estimated that agricultural non-CO_2 emissions (CH_4 and N_2O) would triple by 2055 to 15.3 $GtCO_2eq/yr$ if current dietary trends and population growth were to continue. Technical mitigation options on the supply side, such as improved cropland or livestock management, alone could reduce that value to 9.8 $GtCO_2eq/yr$, whereas emissions were reduced to 4.3 $GtCO_2eq/yr$ in a 'decreased livestock product' scenario and to 2.5 $GtCO_2eq/yr$ if both technical mitigation and dietary change were assumed. Hence, the potential to reduce GHG emissions through changes in consumption was found to be substantially higher than that of technical mitigation measures. Stehfest et al., (2009) evaluated effects of dietary changes on CO_2 (including C sources/sinks of ecosystems), CH_4, and N_2O emissions. In a 'business-as-usual' scenario largely based on FAO (2006), total GHG emissions were projected to reach 11.9 $GtCO_2eq/yr$ in 2050. The following changes were evaluated: no ruminant meat, no meat, and a diet without any animal products. Changed diets resulted in GHG emission savings of 34–64 % compared to the 'business-as-usual' scenario; a switch to a 'healthy diet' recommended by the Harvard Medical School would save 4.3 $GtCO_2eq/yr$ (–36 %). Adoption of the 'healthy diet' (which includes a meat, fish and egg consumption of 90 g/cap/day) would reduce global GHG abatement costs to reach a 450 ppm CO_2eq concentration target by ~50 % compared to the reference case (Stehfest et al., 2009). The analysis assumed nutritionally sufficient diets; reduced supply of animal protein was compensated by plant products (soy, pulses, etc.). Considerable cultural and social barriers against a widespread adoption of dietary changes to low-GHG food may be expected (Davidson, 2012; Smith et al., 2013, 11.4.5).

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

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

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

11.4.4 Feedbacks of changes in land demand

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

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

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

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

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

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

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

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

11.4.5 Sustainable development and behavioural aspects

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

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

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

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

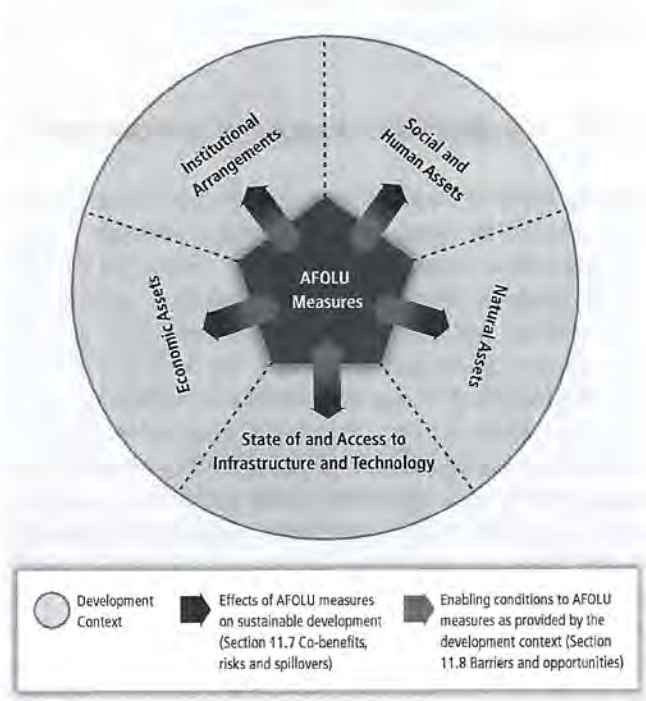

Figure 11.11 | Dynamic interactions between the development context and AFOLU.

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

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

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

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

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

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

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

Table 11.5 | Issues related to AFOLU measures and sustainable development.

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

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

11

Table 11.6 | Characterization of social actors in AFOLU.

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

Table 11.7 | Emerging behavioural aspects relevant for AFOLU mitigation measures.

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

[1] Stehfest et al. (2009); [2]Roy et al. (2012); [3]González et al. (2011); [4]Popp et al. (2010); [5]Schneider et al. (2011); [6]Cotula (2012); [7]Messerli et al. (2013); [8]German et al. (2013); [9]Muys et al. (2014); [10]MacMillan Uribe et al. (2012); [11]Chakrabarti (2010); [12]Karipidis et al. (2010); [13]Auld et al. (2008); [14]Diaz-Chavez (2011); [15]Calegari et al. (2008); [16]Deal et al. (2012); [17]DeFries and Rosenzweig (2010); [18]Hein and van der Meer (2012);[19] Lippke et al. (2003).

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

11.5.1 Feedbacks between ALOFU and climate change

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

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

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

Projections of the global carbon cycle to 2100 using 'Coupled Model Intercomparison Project Phase 5 (CMIP5) Earth System Models' (WGI Chapter 6; Le Quéré et al., 2013) that represent a wider range of complex interactions between the carbon cycle and the physical climate system consistently estimate a positive feedback between climate and the carbon cycle, i.e., reduced natural sinks or increased natural CO_2 sources in response to future climate change. Implications of climate change on terrestrial carbon pools biomes and mitigation potential of forests.

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

Experimental studies and observations suggest that predicted changes in temperature, rainfall regimes, and hydrology may promote the die-back of tropical forests (e.g., Nepstad et al., 2007). The prolonged drought conditions in the Amazon region during 2005 contributed to a decline in above-ground biomass and triggered a release of 4.40 to 5.87 $GtCO_2$ (Phillips et al., 2009). Earlier model studies suggested Amazon die-back in the future (Cox et al., 2013; Huntingford et al., 2013). However, recent model estimates suggest that rainforests may be more resilient to climate change, projecting a moderate risk of tropical forest reduction in South America and even lower risk for African and Asian tropical forests (Gumpenberger et al., 2010; Cox et al., 2013; Huntingford et al., 2013).

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

11.5.3 Implications of climate change on peatlands, grasslands, and croplands

Peatlands: Wetlands, peatlands, and permafrost soils contain higher carbon densities relative to mineral soils, and together they comprise extremely large stocks of carbon globally (Davidson and Janssens, 2006). Peatlands cover approximately 3% of the Earth's land area and are estimated to contain 350–550 Gt of carbon, roughly between 20 to 25% of the world's soil organic carbon stock (Gorham, 1991; Fenner et al., 2011). Peatlands can lose CO_2 through plant respiration and aerobic peat decomposition (Clair et al., 2002) and with the onset of climate change, may become a source of CO_2 (Koehler et al., 2010). Large carbon losses are likely from deep burning fires in boreal peatlands under future projections of climate warming and drying (Flannigan et al., 2009). A study by Fenner et al. (2011) suggests that climate change is expected to increase the frequency and severity of drought in many of the world's peatlands which, in turn, will release far more GHG emissions than thought previously. Climate change is projected to have a severe impact on the peatlands in northern regions where most of the perennially frozen peatlands are found (Tarnocai, 2006). According to Schuur et al. (2008), the thawing permafrost and consequent microbial decomposition of previously frozen organic carbon, is one of the most significant potential feedbacks from terrestrial ecosystems to the atmosphere in a changing climate. Large areas of permafrost will experience thawing (WGI Chapter 12), but uncertainty over the magnitude of frozen carbon losses through CO_2 or CH_4 emissions to the atmosphere is large, ranging between 180 and 920 $GtCO_2$ by the end of the 21st century under the Representative Concentration Pathways (RCP) 8.5 scenario (WGI Chapter 6; Le Quéré et al., 2013).

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

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

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

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

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

11.5.5 Mitigation and adaptation synergies and tradeoffs

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

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

11.6 Costs and potentials

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

Economic (as well as market) mitigation potentials tend to be context-specific and are likely to vary across spatial and temporal scales. Unless otherwise stated, in the rest of this section, economic potentials are expressed in million tonnes (Mt) of mitigation in carbon dioxide equivalent (CO_2eq) terms, that can arise from an individual mitigation option or from an AFOLU sub-sector at a given cost per tonne of CO_2eq. (USD/tCO_2eq) over a given period to 2030, which is 'additional' to the corresponding baseline or reference case levels.

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

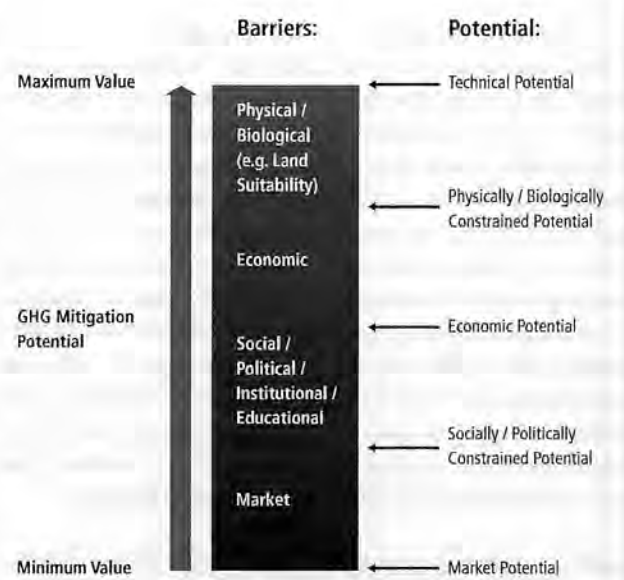

Figure 11.12 | Relationship between technical, economic, and market potential (based on Smith, 2012).

global mitigation costs and potentials published since AR4, and (c) provide a regional disaggregation of the potentials to show how potential, and the portfolio of available options, varies in different world regions.

11.6.1 Approaches to estimating economic mitigation potentials

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

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

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

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

11.6.2 Global estimates of costs and potentials in the AFOLU sector

Through combination of forestry and agriculture potentials from AR4, total mitigation potentials for the AFOLU sector are estimated to be ~3 to ~7.2 $GtCO_2eq/yr$ in 2030 at 20 and 100 USD/tCO_2eq, respectively (Figure 11.13), including only supply-side options in agriculture (Smith et al., 2007) and a combination of supply- and demand-side options for forestry (Nabuurs et al., 2007).

Estimates of global economic mitigation potentials in the AFOLU sector published since AR4 are shown in Figure 11.14, with AR4 estimates shown for comparison (IPCC, 2007a).

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

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

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

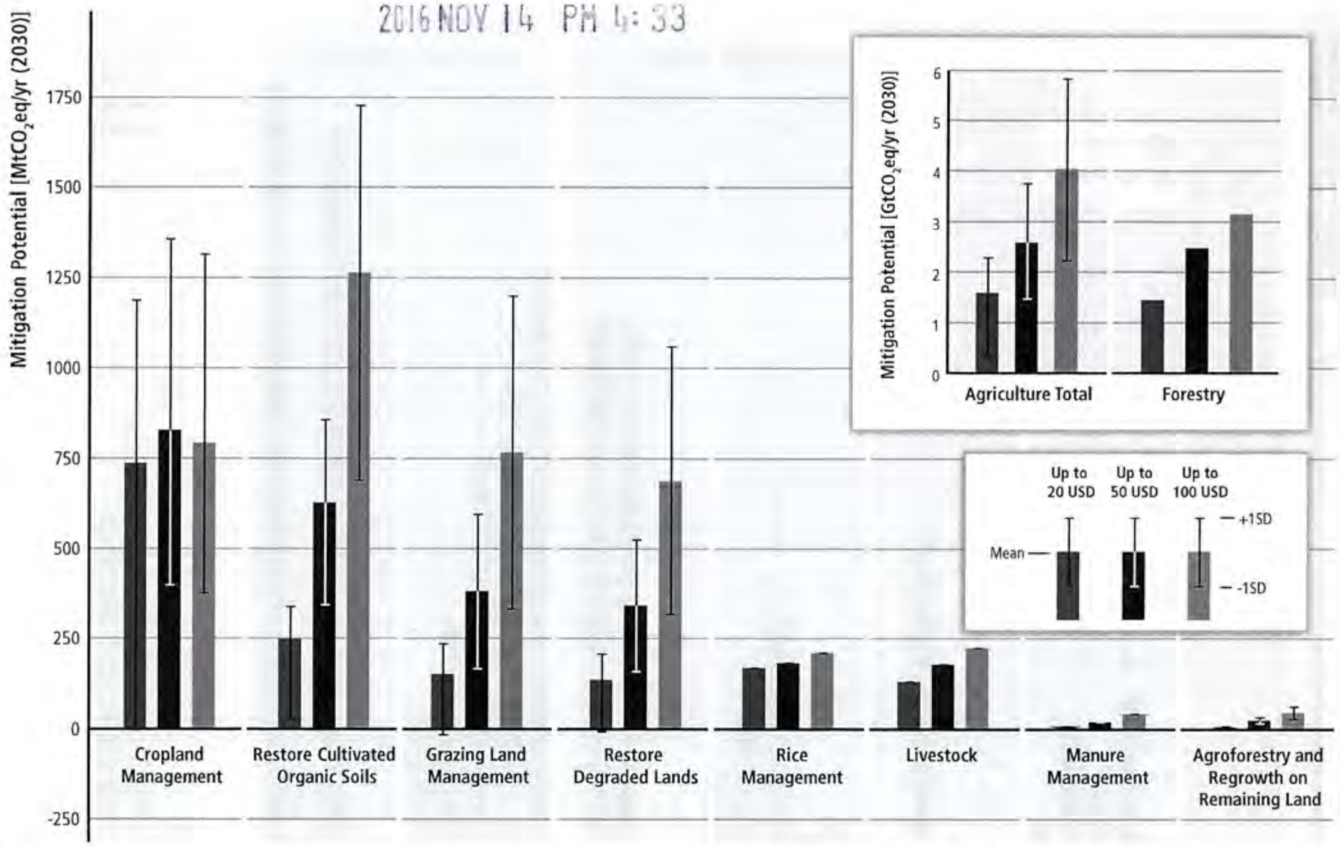

Figure 11.13 | Mitigation potential for the AFOLU sector, plotted using data from AR4 (Nabuurs et al., 2007; Smith et al., 2007). Whiskers show the range of estimates (+/- 1 standard deviation) for agricultural options for which estimates are available.

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

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

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

Figure 11.17 shows the economically viable mitigation opportunities in AFOLU in 2030 by region and by main mitigation option at carbon prices of up to 20, 50, and 100 USD/tCO_2eq. The composition of the agricultural mitigation portfolio varies greatly with the carbon price (Smith, 2012), with low cost options such as cropland management being favoured at low carbon prices, but higher cost options such as restoration of cultivated organic soils being more cost-effective at higher prices. Figure 11.17 also reveals some very large differences in mitigation potential, and different ranking of most effective options, between regions. Across all AFOLU options, Asia has the largest mitigation potential, with the largest mitigation in both forestry and agriculture, followed by LAM, OECD-1990, MAF, and EIT.