10 Industry

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Contents

Executive Summary

An absolute reduction in emissions from the industry sector will require deployment of a broad set of mitigation options beyond energy efficiency measures (*medium evidence, high agreement*). In the last two to three decades there has been continued improvement in energy and process efficiency in industry, driven by the relatively high share of energy costs. In addition to energy efficiency, other strategies such as emissions efficiency (including e.g., fuel and feedstock switching, carbon dioxide capture and storage (CCS)), material use efficiency (e.g., less scrap, new product design), recycling and re-use of materials and products, product service efficiency (e.g., car sharing, maintaining buildings for longer, longer life for products), or demand reductions (e.g., less mobility services, less product demand) are required in parallel (*medium evidence, high agreement*). [Section 10.4, 10.7]

Industry-related greenhouse gas (GHG) emissions have continued to increase and are higher than GHG emissions from other end-use sectors (*high confidence*). Despite the declining share of industry in global gross domestic product (GDP), global industry and waste/wastewater GHG emissions grew from 10.4 $GtCO_2eq$ in 1990 to 13.0 $GtCO_2eq$ in 2005 to 15.4 $GtCO_2eq$ in 2010. Total global GHG emissions for industry and waste/wastewater in 2010, which nearly doubled since 1970, were comprised of direct energy-related CO_2 emissions of 5.3 $GtCO_2eq$, indirect CO_2 emissions from production of electricity and heat for industry of 5.2 $GtCO_2eq$, 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 $GtCO_2eq$. 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. Currently, emissions from industry are larger than the emissions from either the buildings or transport end-use sectors and represent just over 30 % of global GHG emissions in 2010 (just over 40 % if Agriculture, Forestry, and Other Land Use (AFOLU) emissions are not included). (*high confidence*) [10.2, 10.3]

Globally, industrial GHG emissions are dominated by the Asia region, which was also the region with the fastest emission growth between 2005 and 2010 (*high confidence*). In 2010, over half (52 %) of global direct GHG emissions from industry and waste/wastewater were from the Asia region (ASIA), followed by the member countries of the Organisation for Economic Co-operation and Development in 1990 (OECD-1990) (25 %), Economies in Transition (EIT) (9 %), Middle East and Africa (MAF) (8 %), and Latin America (LAM) (6 %). Between 2005 and 2010, GHG emissions from industry grew at an average annual rate of 3.5 % globally, comprised of 7 % average annual growth in the ASIA region, followed by MAF (4.4 %), LAM (2 %), and the EIT countries (0.1 %), but declined in the OECD-1990 countries (–1.1 %). [10.3]

The energy intensity of the sector could be reduced by approximately up to 25 % compared to current level through wide-scale upgrading, replacement and deployment of best available technologies, particularly in countries where these are not in practice and for non-energy intensive industries (*robust evidence, high agreement*). Despite long-standing attention to energy efficiency in industry, many options for improved energy efficiency remain. [10.4, 10.7]

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 (*limited evidence, medium agreement*). Barriers to implementing energy efficiency relate largely to the initial investment costs and lack of information. Information programmes are the most prevalent approach for promoting energy efficiency, followed by economic instruments, regulatory approaches, and voluntary actions. [10.4, 10.7, 10.9, 10.11]

Besides sector specific technologies, cross-cutting technologies and measures applicable in both large energy intensive industries and Small and Medium Enterprises (SMEs) can help to reduce GHG emissions (*robust evidence, high agreement*). Cross-cutting technologies such as efficient motors, electronic control systems, and cross-cutting measures such as reducing air or steam leaks help to optimize performance of industrial processes and improve plant efficiency cost-effectively with both energy savings and emissions benefits [10.4].

Long-term step-change options can include a shift to low carbon electricity, radical product innovations (e.g., alternatives to cement), or carbon dioxide capture and storage (CCS). Once demonstrated, sufficiently tested, cost-effective, and publicly accepted, these options may contribute to significant climate change mitigation in the future (*medium evidence, medium agreement*). [10.4]

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 (*medium evidence, high agreement*). Extending product life and using products more intensively could contribute to reduction of product demand without reducing the service. Absolute emission reductions can also come 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 activities in other sectors and adaptation measures may result in increased industrial product demand and corresponding emissions (*robust evidence, high agreement*). Production of mitigation technologies (e.g., insulation materials for buildings) or material demand for adaptation measures (e.g., infrastructure materials) contribute to industrial GHG emissions. [10.4, 10.6]

Systemic approaches and collaboration within and across industrial sectors at different levels, e.g., sharing of infrastructure, information, waste and waste management facilities, heating,

and cooling, may provide further mitigation potential in certain regions or industry types (*robust evidence, high agreement*). The formation of industrial clusters, industrial parks, and industrial symbiosis are emerging trends in many developing countries, especially with SMEs. [10.5]

Several emission-reducing options in the industrial sector are cost-effective and profitable (*medium evidence, medium agreement*). While options in cost ranges of 20–50, 0–20, and even below 0 USD_{2010}/tCO_2eq exist, to achieve near-zero emission intensity levels in the industry sector would require additional realization of long-term step-change options (e.g., CCS) associated with higher levelized costs of conserved carbon (LCCC) in the range of 50–150 USD_{2010}/tCO_2. However, mitigation costs vary regionally and depend on site-specific conditions. Similar estimates of costs for implementing material efficiency, product-service efficiency, and service demand reduction strategies are not available. [10.7]

Mitigation measures in the industry sector are often associated with co-benefits (*robust evidence, high agreement*). Co-benefits of mitigation measures could drive industrial decisions and policy choices. They include enhanced competitiveness through cost reductions, new business opportunities, better environmental compliance, health benefits through better local air and water quality and better work conditions, and reduced waste, all of which provide multiple indirect private and social benefits. [10.8]

Unless barriers to mitigation in industry are resolved, the pace and extent of mitigation in industry will be limited and even profitable measures will remain untapped (*robust evidence, high agreement*). There are a broad variety of barriers to implementing energy efficiency in the industry sector; for energy-intensive industry, the issue is largely initial investment costs for retrofits, while barriers for other industries include both cost and a lack of information. For material efficiency, product-service efficiency, and demand reduction, there is a lack of experience with implementation of mitigation measures and often there are no clear incentives for either the supplier or consumer. Barriers to material efficiency include lack of human and institutional capacities to encourage management decisions and public participation. [10.9]

There is no single policy that can address the full range of mitigation measures available for industry and overcome associated barriers (*robust evidence, high agreement*). In promoting energy efficiency, information programs are the most prevalent approach, followed by economic instruments, regulatory approaches and voluntary actions. To date, few policies have specifically pursued material or product service efficiency. [10.11]

While the largest mitigation potential in industry lies in reducing CO_2 emissions from fossil fuel use, there are also significant mitigation opportunities for non-CO_2 gases. Key opportunities comprise, for example, reduction of HFC emissions by leak repair, refrigerant recovery and recycling, and proper disposal and replacement by alternative refrigerants (ammonia, HC, CO_2). Nitrous oxide (N_2O) emissions from adipic and nitric acid production can be reduced through the implementation of thermal destruction and secondary catalysts. The reduction of non-CO_2 GHGs also faces numerous barriers. Lack of awareness, lack of economic incentives, and lack of commercially available technologies (e.g., for HFC recycling and incineration) are typical examples. [10.4, 10.7, 10.9]

Long-term scenarios for industry highlight improvements in emissions efficiency as an important future mitigation strategy (*robust evidence, high agreement*). Detailed industry sector scenarios fall within the range of more general long-term integrated scenarios. Improvements in emissions efficiency in the mitigation scenarios result from a shift from fossil fuels to electricity with low (or negative) CO_2 emissions and use of CCS for industry fossil fuel use and process emissions. The crude representation of materials, products, and demand in scenarios limits the evaluation of the relative importance of material efficiency, product-service efficiency, and demand reduction options. (*robust evidence, high agreement*) [6.8, 10.10]

The most effective option for mitigation in waste management is waste reduction, followed by re-use and recycling and energy recovery (*robust evidence, high agreement*) [10.4, 10.14]. Direct emissions from the waste sector almost doubled during the period from 1970 to 2010. Globally, approximately only 20 % of municipal solid waste (MSW) is recycled and approximately 13.5 % is treated with energy recovery while the rest is deposited in open dumpsites or landfills. Approximately 47 % of wastewater produced in the domestic and manufacturing sectors is still untreated. As the share of recycled or reused material is still low, waste treatment technologies and energy recovery can also result in significant emission reductions from waste disposal. Reducing emissions from landfilling through treatment of waste by anaerobic digestion has the largest cost range, going from negative cost to very high cost. Also, advanced wastewater treatment technologies may enhance GHG emissions reduction in the wastewater treatment but they tend to concentrate in the higher costs options (*medium evidence, medium agreement*). [10.14]

A key challenge for the industry sector is the uncertainty, incompleteness, and quality of data available in the public domain on energy use and costs for specific technologies on global and regional scales that can serve as a basis for assessing performance, mitigation potential, costs, and for developing policies and programmes with high confidence. Bottom-up information on cross-sector collaboration and demand reduction as well as their implications for mitigation in industry is particularly limited. Improved modelling of material flows in integrated models could lead to a better understanding of material efficiency and demand reduction strategies and the associated mitigation potentials. [10.12]

10.1 Introduction

This chapter provides an update to developments on mitigation in the industry sector since the IPCC (Intergovernmental Panel on Climate Change) Fourth Assessment Report (AR4) (IPCC, 2007), but has much wider coverage. Industrial activities create all the physical products (e.g., cars, agricultural equipment, fertilizers, textiles, etc.) whose use delivers the final services that satisfy current human needs. Compared to the industry chapter in AR4, this chapter analyzes industrial activities over the whole supply chain, from extraction of primary materials (e.g., ores) or recycling (of waste materials), through product manufacturing, to the demand for the products and their services. It includes a discussion of trends in activity and emissions, options for mitigation (technology, practices, and behavioural aspects), estimates of the mitigation potentials of some of these options and related costs, co-benefits, risks and barriers to their deployment, as well as industry-specific policy instruments. Findings of integrated models (long-term mitigation pathways) are also presented and discussed from the sector perspective. In addition, at the end of the chapter, the hierarchy in waste management and mitigation opportunities are synthesized, covering key waste-related issues that appear across all chapters in the Working Group III contribution to the IPCC Fifth Assessment Report (AR5).

Figure 10.1, which shows a breakdown of total global anthropogenic GHG emissions in 2010 based on Bajželj et al. (2013), illustrates the logic that has been used to distinguish the industry sector from other sectors discussed in this report. The figure shows how human demand for energy services, on the left, is provided by economic sectors, through the use of equipment in which devices create heat or work from final energy. In turn, the final energy has been created by processing a primary energy source. Combustion of carbon-based fuels leads to the release of GHG emissions as shown on the right. The remaining anthropogenic emissions arise from chemical reactions in industrial processes, from waste management and from the agriculture and land-use changes discussed in Chapter 11.

Mitigation options can be chosen to reduce GHG emissions at all stages in Figure 10.1, but caution is needed to avoid 'double counting'. The figure also demonstrates that care is needed when allocating emissions to specific products and services ('carbon footprints', for example) while ensuring that the sum of all 'footprints' adds to the sum of all emissions.

Emissions from industry (30 % of total global GHG emissions) arise mainly from material processing, i.e., the conversion of natural resources (ores, oil, biomass) or scrap into materials stocks which are then converted in manufacturing and construction into products. Pro-

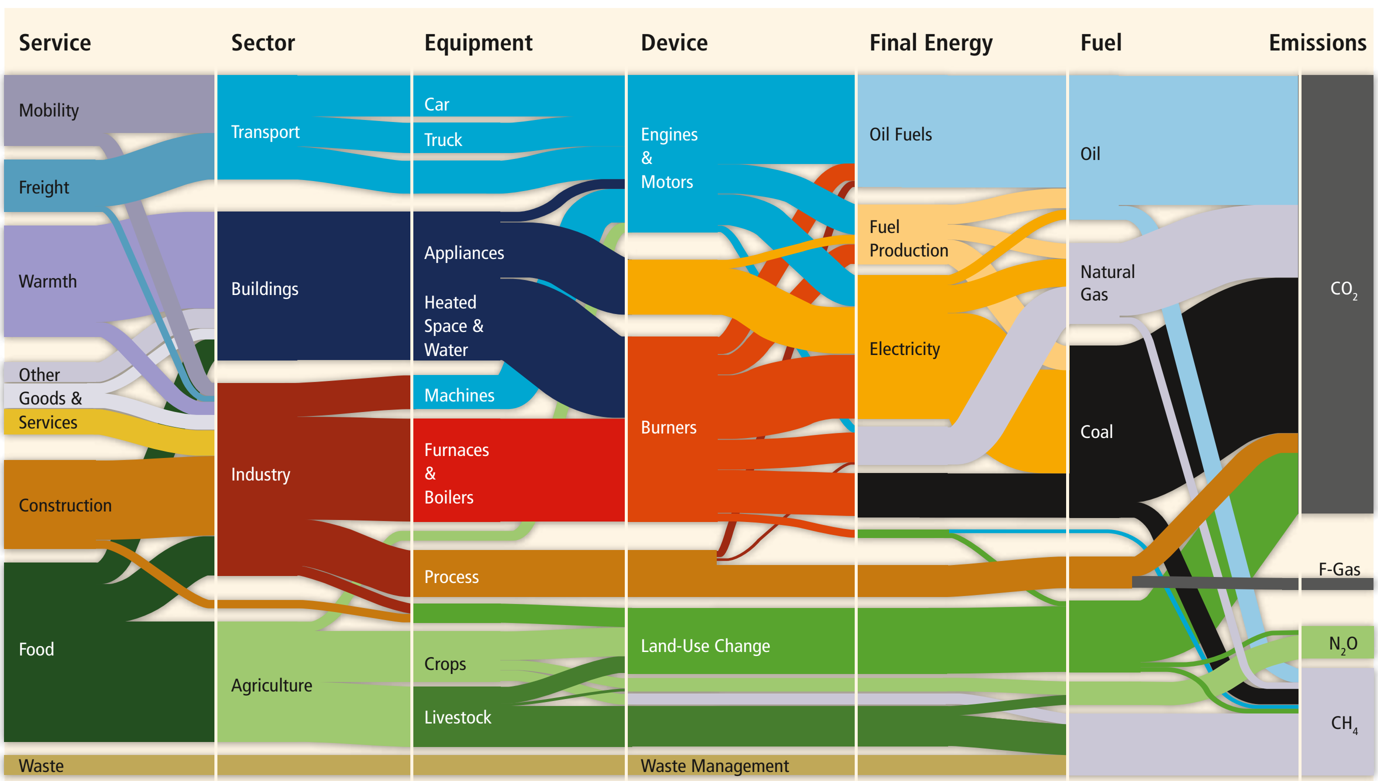


Figure 10.1 | A Sankey diagram showing the system boundaries of the industry sector and demonstrating how global anthropogenic emissions in 2010 arose from the chain of technologies and systems required to deliver final services triggered by human demand. The width of each line is proportional to GHG emissions released, and the sum of these widths along any vertical slice through the diagram is the same, representing all emissions in 2010 (Bajželj et al., 2013).

duction of just iron and steel and non-metallic minerals (predominately cement) results in 44 % of all carbon dioxide (CO_2) emissions (direct, indirect, and process-related) from industry. Other emission-intensive sectors are chemicals (including plastics) and fertilizers, pulp and paper, non-ferrous metals (in particular aluminium), food processing (food growing is covered in Chapter 11), and textiles.

Decompositions of GHG emissions have been used to analyze the different drivers of global industry-related emissions. An accurate decomposition for the industry sector would involve great complexity, so instead this chapter uses a simplified conceptual expression to identify the key mitigation opportunities available within the sector:

$$G = \frac{G}{E} \times \frac{E}{M} \times \frac{M}{P} \times \frac{P}{S} \times S$$

where G is the GHG emissions of the industrial sector within a specified time period (usually one year), E is industrial sector energy consumption and M is the total global production of materials in that period. P is stock of products created from these materials (including both consumables and durables added to existing stocks), and S is the services delivered in the time period through use of those products.

The expression is indicative only, but leads to the main mitigation strategies discussed in this chapter:

G/E is the *emissions intensity* of the sector expressed as a ratio to the energy used: the GHG emissions of industry arise largely from energy use (directly from combusting fossil fuels, and indirectly through purchasing electricity and steam), but emissions also arise from industrial chemical reactions. In particular, producing cement, chemicals, and non-ferrous metals leads to the inevitable release of significant 'process emissions' regardless of energy supply. We refer to reductions in G/E as *emissions efficiency* for the energy inputs and the processes.

E/M is the *energy intensity*: approximately three quarters of industrial energy use is required to create materials from ores, oil or biomass, with the remaining quarter used in the downstream manufacturing and construction sectors that convert materials to products. The energy required can in some cases (particularly for metals and paper) be reduced by production from recycled scrap, and can be further reduced by material re-use, or by exchange of waste heat and exchange of by-products between sectors. Reducing E/M is the goal of *energy efficiency.*

M/P is the *material intensity* of the sector: the amount of material required to create a product and maintain the stock of a product depends both on the design of the product and on the scrap discarded during its production. Both can be reduced by *material efficiency.*

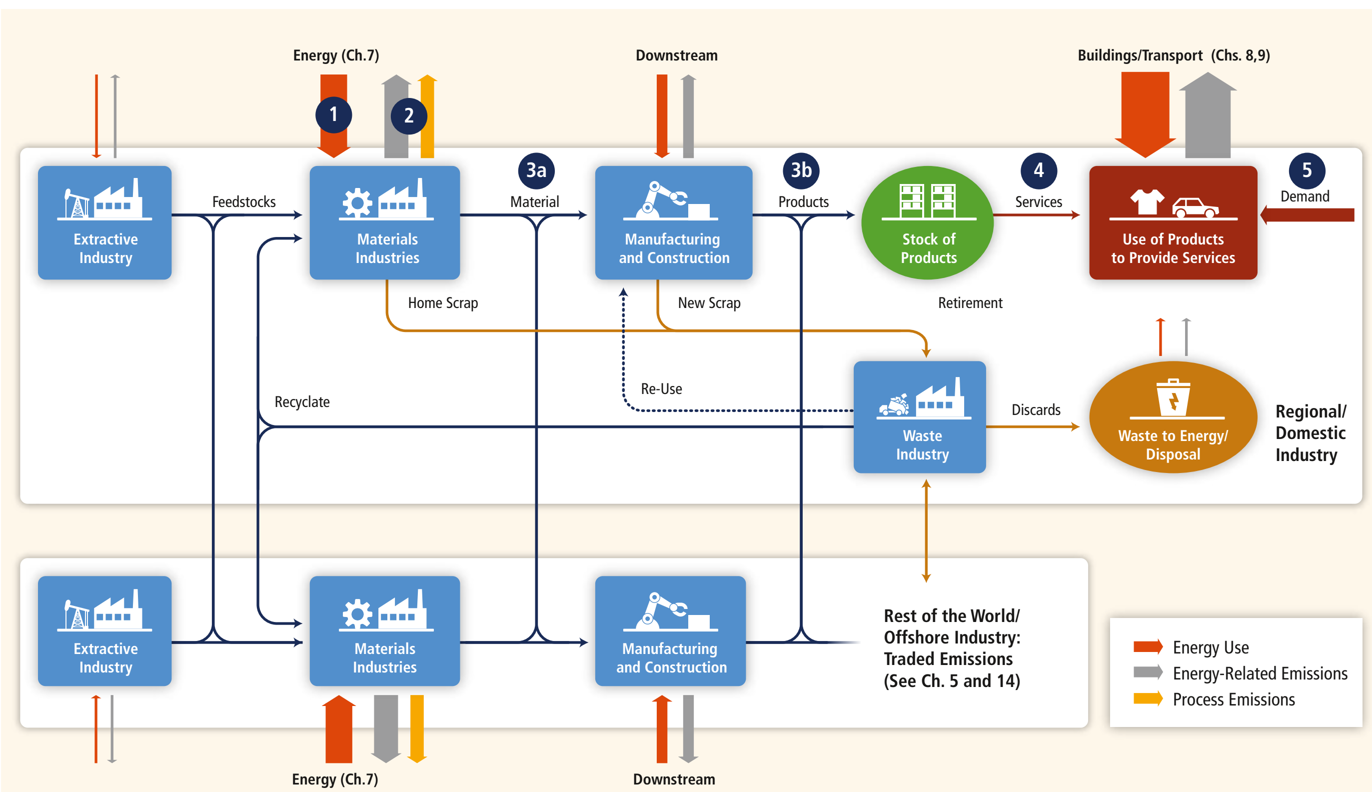


Figure 10.2 | A schematic illustration of industrial activity over the supply chain. Options for climate change mitigation in the industry sector are indicated by the circled numbers: (1) Energy efficiency (e. g., through furnace insulation, process coupling, or increased material recycling); (2) Emissions efficiency (e. g., from switching to non-fossil fuel electricity supply, or applying CCS to cement kilns); (3a) Material efficiency in manufacturing (e. g., through reducing yield losses in blanking and stamping sheet metal or re-using old structural steel without melting); (3b) Material efficiency in product design (e. g., through extended product life, light-weight design, or de-materialization); (4) Product-Service efficiency (e. g., through car sharing, or higher building occupancy); (5) Service demand reduction (e. g., switching from private to public transport).

P/S is the *product-service intensity*: the level of service provided by a product depends on its intensity of use. For consumables (e.g., food or detergent) that are used within the accounting period in which they are produced, service is provided solely by the production within that period. For durables that last for longer than the accounting period (e.g., clothing), services are provided by the stock of products in current use. In this case P is the flow of material required to replace retiring products and to meet demand for increases in total stock. Thus for consumables, P/S can be reduced by more precise use (for example using only recommended doses of detergents or applying fertilizer precisely) while for durables, P/S can be reduced both by using durable products for longer and by using them more intensively. We refer to reductions in P/S as *product-service efficiency*.

S: The total global demand for service is a function of population, wealth, lifestyle, and the whole social system of expectations and aspirations. If the total demand for service were to decrease, it would lead to a reduction in industrial emissions, and we refer to this as *demand reduction.*

Figure 10.2 expands on this simplified relationship to illustrate the main options for GHG emissions mitigation in industry (circled numbers). The figure also demonstrates how international trade of products leads to significant differences between 'production' and 'consumption' measures of national emissions, and demonstrates how the 'waste' industry, which includes material recycling as well as options like 'waste to energy' and disposal, has a significant potential for influencing future industrial emissions.

Figure 10.2 clarifies the terms used for key sectors in this chapter: 'Industry' refers to the totality of activities involving the physical transformation of materials within which 'extractive industry' supplies feedstock to the energy-intensive 'materials industries' which create refined materials. These are converted by 'manufacturing' into products and by 'construction' into buildings and infrastructure. 'Home scrap' from the materials processing industries, 'new scrap' from downstream construction and manufacturing, and products retiring at end-of-life are processed in the 'waste industry.' This 'waste' may be recycled (particularly bulk metals, paper, glass and some plastics), may be re-used to save the energy required for recycling, or may be discarded to landfills or incinerated (which can lead to further emissions on one hand and energy recovery on the other hand).

10.2 New developments in extractive mineral industries, manufacturing industries and services

World production trends of mineral extractive industries, manufacturing, and services, have grown steadily in the last 40 years (Figure 10.3). However, the service sector share in world GDP increased from 50 % in 1970 to 70 % in 2010; while the industry world GDP share decreased from 38.2 to 26.9 % (World Bank, 2013).

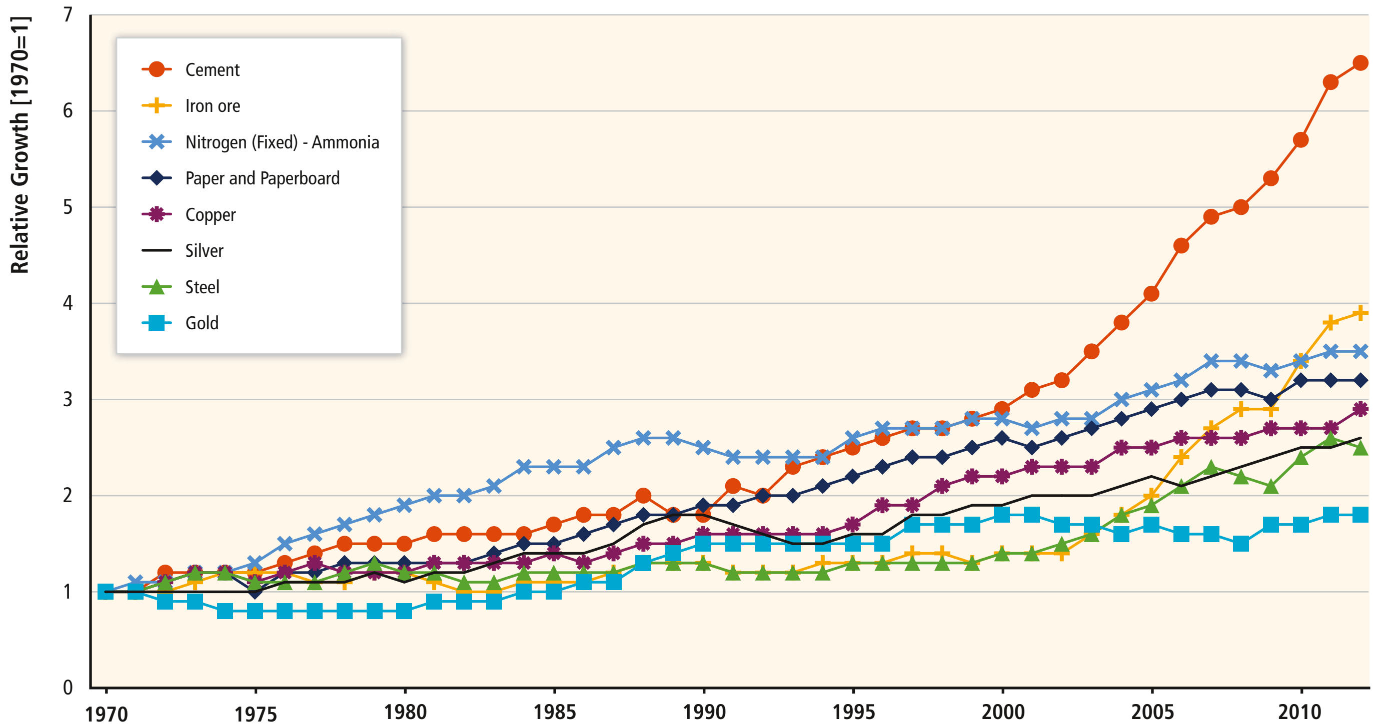


Figure 10.3 | World's growth of main minerals and manufacturing products (1970 = 1). Sources: (WSA, 2012a; FAO, 2013; Kelly and Matos, 2013).

Table 10.1 | Total production of energy-intensive industrial goods for the world top-5 producers of each commodity: 2005, 2012, and average annual growth rate (AAGR) (FAO, 2013; Kelly and Matos, 2013).

Commodity/Country	2005 [Mt]	2012 [Mt]	AAGR
Iron ore			
World	1540	3000	10 %
China	420	1300	18 %
Australia	262	525	10 %
Brazil	280	375	4 %
India	140	245	8 %
Russia	97	100	0.4 %
Steel			
World	1130	1500	4 %
China	349	720	11 %
Japan	113	108	−1 %
U.S.	95	91	−1 %
India	46	76	8 %
Russia	66	76	2 %
Cement			
World	2310	3400	6 %
China	1040	2150	11 %
India	145	250	8 %
U.S.	101	74	−4 %
Brazil	37	70	10 %
Iran	33	65	10 %
Ammonia			
World	121.0	137.0	2 %
China	37.8	44.0	2 %
India	10.8	12.0	2 %
Russia	10.0	10.0	0 %
U.S.	8.0	9.5	2 %
Trinidad & Tobago	4.2	5.5	4 %
Aluminium			
World	31.9	44.9	5 %
China	7.8	19.0	14 %
Russia	3.7	4.2	2 %
Canada	2.9	2.7	−1 %
U.S.	2.5	2.0	−3 %
Australia	1.9	1.9	0 %
Paper			
World	364.7	401.1	1 %
China	60.4	106.3	8 %
U.S.	83.7	75.5	−1 %
Japan	31.0	26.0	−2 %
Germany	21.7	22.6	1 %
Indonesia	7.2	11.5	7 %

Concerning extractive industries for metallic minerals, from 2005 to 2012 annual mining production of iron ore, gold, silver, and copper increased by 10 %, 1 %, 2 %, and 2 % respectively (Kelly and Matos, 2013). Most of the countries in Africa, Latin America, and the transition economies produce more than they use; whereas use is being driven mainly by consumption in China, India, and developed countries (UNCTAD, 2008)[1]. Extractive industries of rare earths are gaining importance because of their various uses in high-tech industry (Moldoveanu and Papangelakis, 2012). New mitigation technologies, such as hybrid and electric vehicles (EVs), electricity storage and renewable technologies, increase the demand for certain minerals, such as lithium, gallium, and phosphates (Bebbington and Bury, 2009). Concerns over depletion of these minerals have been raised, but important research on extraction methods as well as increasing recycling rates are leading to increasing reserve estimates for these materials (Graedel et al., 2011; Resnick Institute, 2011; Moldoveanu and Papangelakis, 2012; Eckelman et al., 2012). China accounts for 97 % of global rare earth extraction (130 Mt in 2010) (Kelly and Matos, 2013).

Regarding manufacturing production, the annual global production growth rate of steel, cement, ammonia, aluminium, and paper—the most energy-intensive industries—ranged from 2 % to 6 % between 2005 and 2012 (Table 10.1). Many trends are responsible for this development (e.g., urbanization significantly triggered demand on construction materials). Over the last decades, as a general trend, the world has witnessed decreasing industrial activity in developed countries with a major downturn in industrial production due to the economic recession in 2009 (Kelly and Matos, 2013). There is continued increase in industrial activity and trade of some developing countries. The increase in manufacturing production and consumption has occurred mostly in Asia. China is the largest producer of the main industrial outputs. In many middle-income countries industrialization has stagnated, and in general Africa and Least Developed Countries (LDCs) have remained marginalized (UNIDO, 2009; WSA, 2012a). In 2012, 1.5 billion tonnes of steel (212 kg/cap) were manufactured; 46 % was produced and consumed in mainland China (522 kg/cap). China also dominates global cement production, producing 2.2 billion tonnes (1,561 kg/cap) in 2012, followed by India with only 250 Mt (202 kg/cap) (Kelly and Matos, 2013; UNDESA, 2013). More subsector specific trends are in Section 10.4.

Globally large-scale production dominates energy-intensive industries; however small- and medium-sized enterprises are very important in many developing countries. This brings additional challenges for mitigation efforts (Worrell et al., 2009; Roy, 2010; Ghosh and Roy, 2011).

[1] For example, in 2008, China imported 50 % of the world's total iron ore exports and produced about 50 % of the world's pig iron (Kelly and Matos, 2013). India demanded 35 % of world's total gold production in 2011 (WGC, 2011), and the United States consumed 33 % of world's total silver production in 2011 (Kelly and Matos, 2013).

Another important change in the world´s industrial output over the last decades has been the rise in the proportion of international trade. Manufactured products are not only traded, but the production process is increasingly broken down into tasks that are themselves outsourced and/or traded; i.e., production is becoming less vertically integrated. In addition to other drivers such as population growth, urbanization, and income increase, the rise in the proportion of trade has been driving production increase for certain countries (Fisher-Vanden et al., 2004; Liu and Ang, 2007; Reddy and Ray, 2010; OECD, 2011). The economic recession of 2009 reduced industrial production worldwide because of consumption reduction, low optimism in credit market, and a decline in world trade (Nissanke, 2009). More discussion on GHG emissions embodied in trade is presented in Chapter 14. Similar to industry, the service sector is heterogeneous and has significant proportion of small and medium sized enterprises. The service sector covers activities such as public administration, finance, education, trade, hotels, restaurants, and health. Activity growth in developing countries and structural shift with rising income is driving service sector growth (Fisher-Vanden et al., 2004; Liu and Ang, 2007; Reddy and Ray, 2010; OECD, 2011). OECD countries are shifting from manufacturing towards service-oriented economies (Sun, 1998; Schäfer, 2005; US EIA, 2010), however, this is also true for some non-OECD countries. For example, India has almost 64 %–66 % of GDP contribution from service sector (World Bank, 2013).

10.3 New developments in emission trends and drivers

In 2010, the industry sector accounted for around 28 % of final energy use (IEA, 2013). Global industry and waste/wastewater GHG emissions grew from 10.37 $GtCO_2eq$ in 1990 to 13.04 $GtCO_2eq$ in 2005 to 15.44 $GtCO_2eq$ in 2010. These emissions are larger than the emissions from either the buildings or transport end-use sectors and represent just over 30 % of global GHG emissions in 2010 (just over 40 % if AFOLU emissions are not included). These total emissions are comprised of:

- Direct energy-related CO_2 emissions for industry[2]
- Indirect CO_2 emissions from production of electricity and heat for industry[3]
- Process CO_2 emissions
- Non-CO_2 GHG emissions
- Direct emissions for waste/wastewater

[2] This also includes CO_2 emissions from non-energy uses of fossil fuels.

[3] The methodology for calculating indirect CO_2 emissions is based on de la Rue du Can and Price (2008) and described in Annex II.5.

Figure 10.4 shows global industry and waste/wastewater direct and indirect GHG emissions by source from 1970 to 2010. Table 10.2 shows primary energy[4] and GHG emissions for industry by emission type (direct energy-related, indirect from electricity and heat production, process CO_2, and non-CO_2), and for waste/wastewater for five world regions and the world total.[5]

Figure 10.5 shows global industry and waste/wastewater direct and indirect GHG emissions by region from 1970 to 2010. This regional breakdown shows that:

- Over half (52 %) of global direct GHG emissions from industry and waste/wastewater are from the ASIA region, followed by OECD-1990 (25 %), EIT (9.4 %), MAF (7.6 %), and LAM (5.7 %).
- Between 2005 and 2010, GHG emissions from industry grew at an average annual rate of 3.5 % globally, comprised of 7.0 % average annual growth in the ASIA region, followed by MAF (4.4 %), LAM (2.0 %), and the EIT countries (0.1 %), but declined in the OECD-1990 countries (–1.1 %).

Regional trends are further discussed in Chapter 5, Section 5.2.1.

Table 10.3 provides 2010 direct and indirect GHG emissions by source and gas. 2010 direct and indirect emissions were dominated by CO_2 (85.1 %), followed by methane (CH_4) (8.6 %), hydrofluorocarbons (HFC) (3.5 %), nitrous oxide (N_2O) (2.0 %), Perfluorocarbons (PFC) (0.5 %) and sulphur hexafluoride (SF_6) (0.4 %) emissions.

10.3.1 Industrial CO_2 emissions

As shown in Table 10.3, industrial CO_2 emissions were 13.14 $GtCO_2$ in 2010. These emissions were comprised of 5.27 $GtCO_2$ direct energy-related emissions, 5.25 $GtCO_2$ indirect emissions from electricity and heat production, 2.59 $GtCO_2$ from process CO_2 emissions and 0.03 $GtCO_2$ from waste/wastewater. Process CO_2 emissions are comprised of process-related emissions of 1.352 $GtCO_2$ from cement production,[6] 0.477 $GtCO_2$ from production of chemicals, 0.242 $GtCO_2$ from lime production, 0.134 $GtCO_2$ from coke ovens, 0.074 $GtCO_2$ from non-ferrous metals production, 0.072 $GtCO_2$ from iron and steel production, 0.061 $GtCO_2$ from ferroalloy production, 0.060 $GtCO_2$ from limestone and dolomite use, 0.049 $GtCO_2$ from solvent and other product use, 0.042 $GtCO_2$ from production of other minerals and 0.024 $GtCO_2$ from non-energy use of lubricants/waxes (JRC/PBL, 2013). Total industrial CO_2 values include emissions from mining and quarrying, from manufacturing, and from construction.

[4] See Glossary in Annex I for definition of primary energy.

[5] The IEA also recently published CO_2 emissions with electricity and heat allocated to end-use sectors (IEA, 2012a). However, the methodology used in this report differs slightly from the IEA approach as explained in Annex II.5

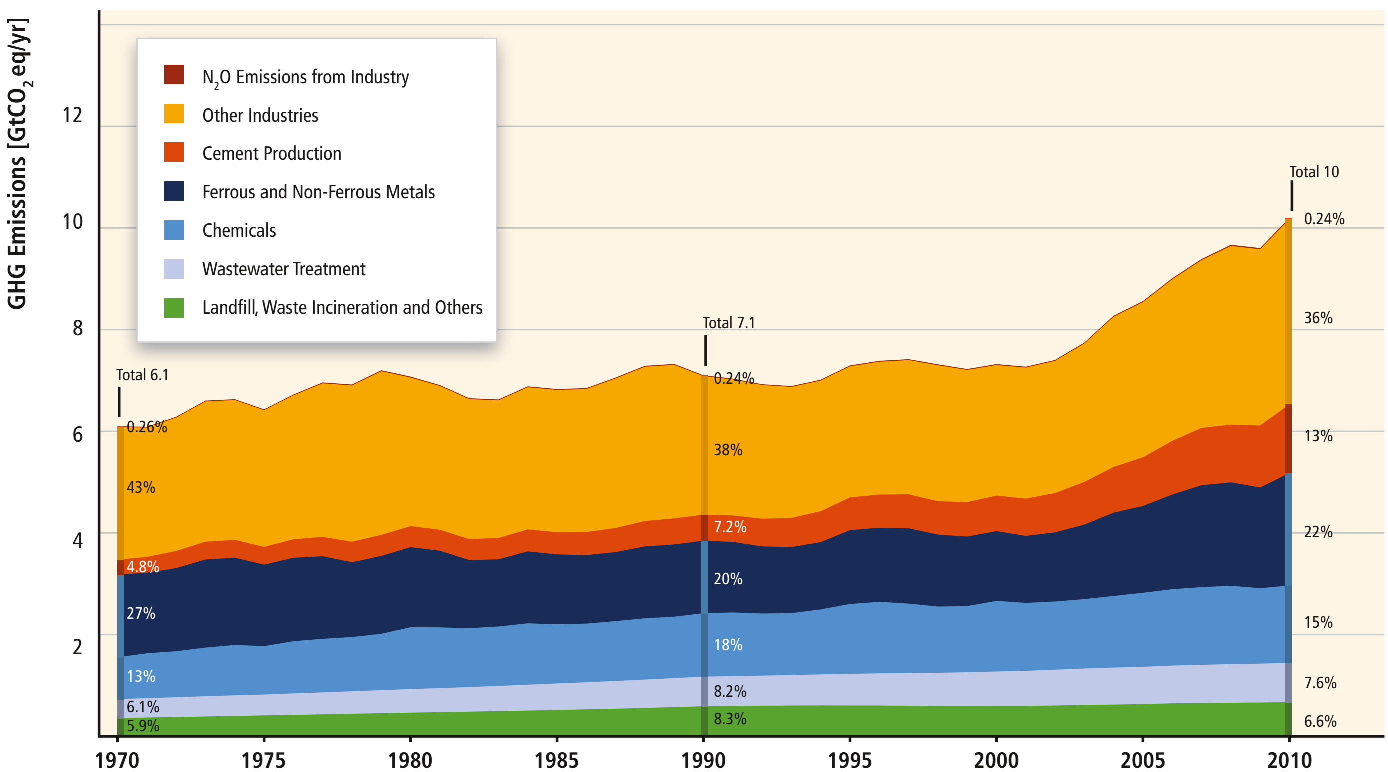
[6] Another source, Boden et al., 2013, indicates that cement process CO_2 emissions in 2010 were 1.65 $GtCO_2$.

Energy-intensive processes in the mining sector include excavation, mine operation, material transfer, mineral preparation, and separation. Energy consumption for mining[7] and quarrying, which is included in 'other industries' in Figure 10.4, represents about 2.7 % of worldwide industrial energy use, varying regionally, and a significant share of national industrial energy use in Botswana and Namibia (around 80 %), Chile (over 50 %), Canada (30 %), Zimbabwe (18.6 %), Mongolia (16.5 %), and South Africa (almost 15 %) in 2010 (IEA, 2012b; c).

[7] Discussion of extraction of energy carriers (e.g., coal, oil, and natural gas) takes place in Chapter 7.

Manufacturing is a subset of industry that includes production of all products (e.g., steel, cement, machinery, textiles) except for energy products, and does not include energy used for construction. Manufacturing is responsible for about 98 % of total direct CO_2 emissions from the industrial sector (IEA, 2012b; c). Most manufacturing CO_2 emissions arise due to chemical reactions and fossil fuel combustion largely used to provide the intense heat that is often required to bring about the physical and chemical transformations that convert raw materials into industrial products. These industries, which include production of chemicals and petrochemicals, iron and steel, cement, pulp and paper, and aluminium, usually account for most of the sector's

Direct Emissions



Indirect Emissions

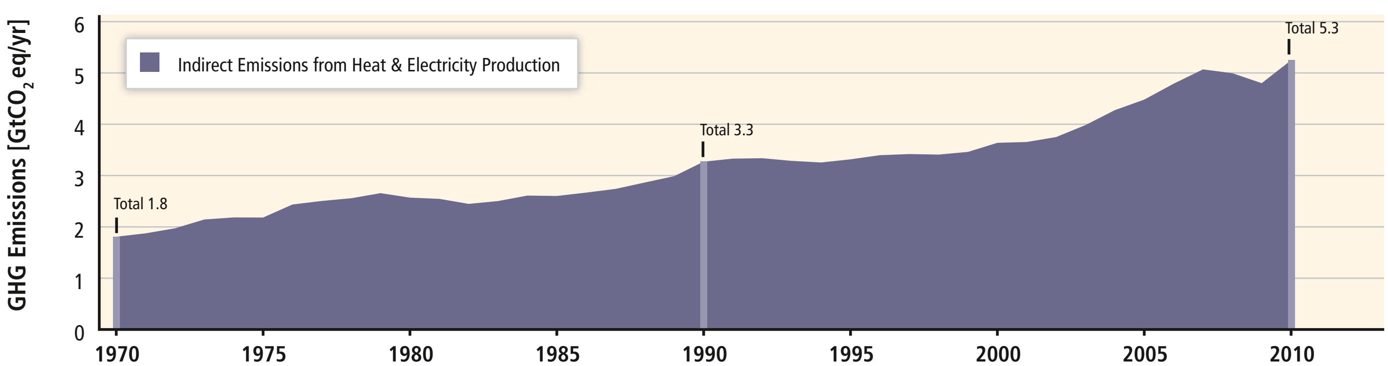


Figure 10.4 | Total global industry and waste/wastewater direct and indirect GHG emissions by source, 1970–2010 ($GtCO_2eq$/yr) (de la Rue du Can and Price, 2008; IEA, 2012a; JRC/PBL, 2013). See also Annex II.9, Annex II.5.

Note: For statistical reasons 'Cement production' only covers process CO_2 emissions (i.e., emissions from cement-forming reactions); energy-related direct emissions from cement production are included in 'other industries' CO_2 emissions.

energy consumption in many countries. In India, the share of energy use by energy-intensive manufacturing industries in total manufacturing energy consumption is 62 % (INCCA, 2010), while it is about 80 % in China (NBS, 2012).

Overall reductions in industrial energy use/manufacturing value-added were found to be greatest in developing economies during 1995–2008. Low-income developing economies had the highest industrial energy intensity values while developed economies had the lowest. Reductions in intensity were realized through technological changes (e.g., changes in product mix, adoption of energy-efficient technologies, etc.) and structural change in the share of energy-intensive industries in the economy. During 1995–2008, developing economies had greater reductions in energy intensity while developed economies had greater reductions through structural change (UNIDO, 2011).

The share of non-energy use of fossil fuels (e.g., the use of fossil fuels as a chemical industry feedstock, of refinery and coke oven products, and of solid carbon for the production of metals and inorganic chemicals) in total manufacturing final energy use has grown from 20 % in 2000 to 24 % in 2009 (IEA, 2012b; c). Fossil fuels used as raw materi-

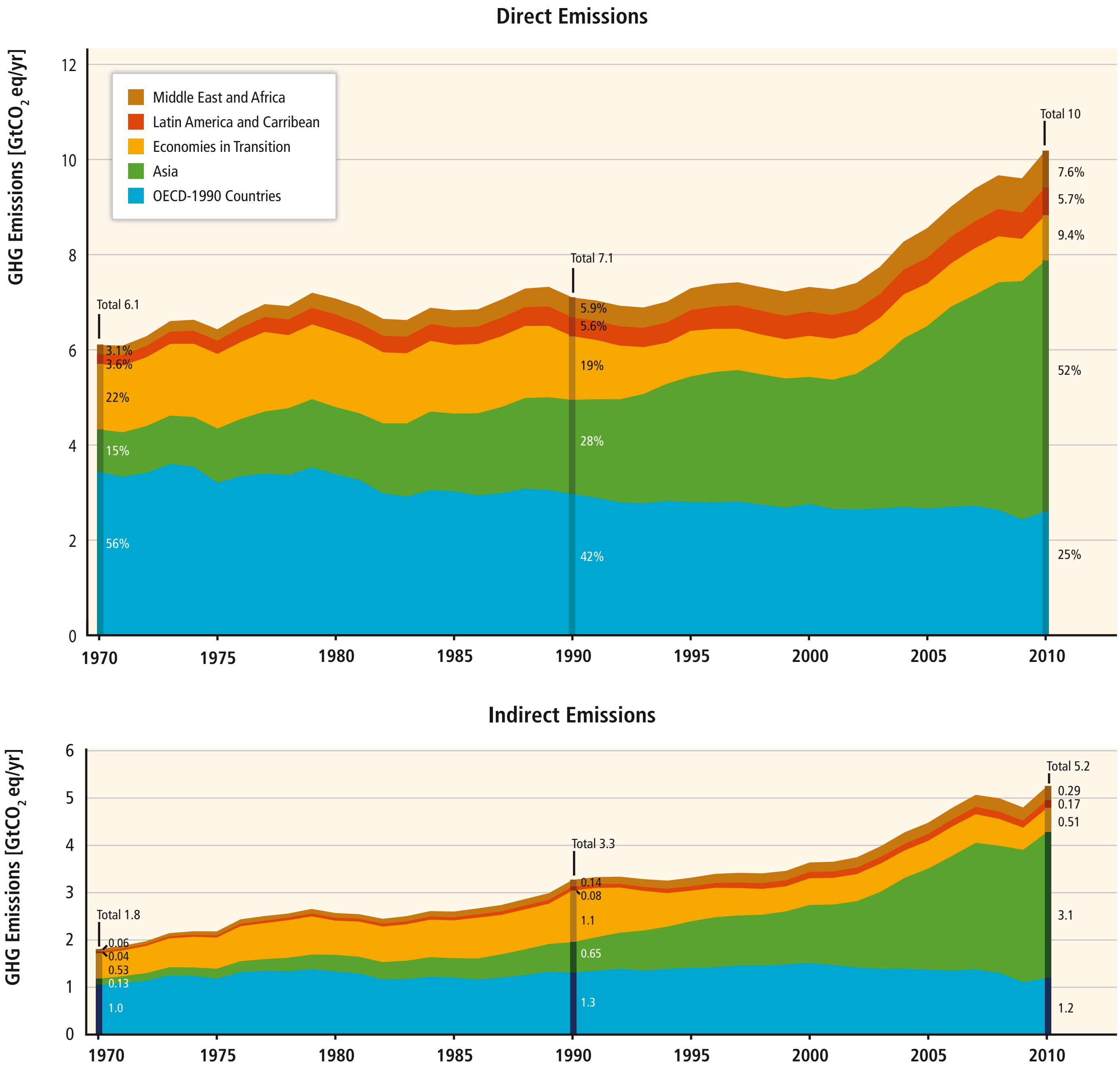


Figure 10.5 | Total global industry and waste/wastewater direct and indirect GHG emissions by region, 1970–2010 ($GtCO_2eq/yr$) (de la Rue du Can and Price, 2008; IEA, 2012a; JRC/PBL, 2013). See also Annex II.9, Annex II.5.

Table 10.2 | Industrial Primary Energy (EJ) and GHG emissions ($GtCO_2eq$) by emission type (direct energy-related, indirect from electricity and heat production, process CO_2, and non-CO_2), and waste/wastewater for five world regions and the world total (IEA, 2012a; b; c; JRC/PBL, 2013; see Annex II.9). For definitions of regions see Annex II.2.

		Primary Energy (EJ)			GHG Emissions ($GtCO_2eq$)		
		1990	2005	2010	1990	2005	2010
ASIA	Direct (energy-related)	20.89	42.83	56.80	1.21	2.08	2.92
	Indirect (electricity + heat)	5.25	15.11	24.38	0.65	2.14	3.08
	Process CO_2 emissions				0.36	0.96	1.49
	Non-CO_2 GHG emissions				0.05	0.25	0.27
	Waste/wastewater				0.35	0.54	0.60
	Total	**26.14**	**57.93**	**81.17**	**2.62**	**5.98**	**8.36**
EIT	Direct (energy-related)	21.98	13.47	13.68	0.79	0.41	0.45
	Indirect (electricity + heat)	6.84	4.10	3.42	1.09	0.59	0.51
	Process CO_2 emissions				0.32	0.23	0.23
	Non-CO_2 GHG emissions				0.11	0.12	0.12
	Waste/wastewater				0.12	0.13	0.15
	Total	**28.82**	**17.56**	**17.10**	**2.43**	**1.48**	**1.47**
LAM	Direct (energy-related)	5.85	8.64	9.45	0.19	0.26	0.28
	Indirect (electricity + heat)	0.97	1.67	1.93	0.08	0.15	0.17
	Process CO_2 emissions				0.08	0.11	0.13
	Non-CO_2 GHG emissions				0.03	0.03	0.03
	Waste/wastewater				0.10	0.14	0.14
	Total	**6.82**	**10.31**	**11.38**	**0.48**	**0.68**	**0.75**
MAF	Direct (energy-related)	5.59	8.91	11.43	0.22	0.30	0.37
	Indirect (electricity + heat)	1.12	1.99	2.58	0.14	0.24	0.29
	Process CO_2 emissions				0.08	0.15	0.21
	Non-CO_2 GHG emissions				0.02	0.02	0.02
	Waste/wastewater				0.10	0.16	0.17
	Total	**6.71**	**10.90**	**14.01**	**0.56**	**0.86**	**1.07**
OECD-1990	Direct (energy-related)	40.93	45.63	42.45	1.55	1.36	1.24
	Indirect (electricity + heat)	11.25	10.92	9.71	1.31	1.37	1.19
	Process CO_2 emissions				0.57	0.56	0.52
	Non-CO_2 GHG emissions				0.35	0.35	0.44
	Waste/wastewater				0.50	0.40	0.39
	Total	**52.18**	**56.55**	**52.16**	**4.28**	**4.04**	**3.79**
World	Direct (energy-related)	95.25	119.47	133.81	3.96	4.41	5.27
	Indirect (electricity + heat)	25.42	33.78	42.01	3.27	4.48	5.25
	Process CO_2 emissions				1.42	2.01	2.59
	Non-CO_2 GHG emissions				0.55	0.77	0.89
	Waste/wastewater				1.17	1.37	1.45
	Total	**120.67**	**153.25**	**175.82**	**10.37**	**13.04**	**15.44**

Note: Includes energy and non-energy use. Non-energy use covers those fuels that are used as raw materials in the different sectors and are not consumed as a fuel or transformed into another fuel. Also includes construction.

als/feedstocks in the chemical industry may result in CO_2 emissions at the end of their life-span in the disposal phase if they are not recovered or recycled (Patel et al., 2005). These emissions need to be accounted for in the waste disposal sector's emissions, although data on waste imports/exports and ultimate disposition are not consistently compiled or reliable (Masanet and Sathaye, 2009). Subsector specific details are also in Section 10.4.

Trade is an important factor that influences production choice decisions and hence CO_2 emissions at the country level. Emission inventories based on consumption rather than production reflect the fact that products produced and exported for consumption in developed countries are an important contributing factor of the emission increase for certain countries such as China, particularly since 2000 (Ahmad and Wyckoff, 2003; Wang and Watson, 2007; Peters and Hertwich, 2008;

Table 10.3 | Industry and waste/wastewater direct and indirect GHG emissions by source and gas, 2010 (in $MtCO_2eq$) (IEA, 2012a; JRC/PBL, 2013).

Source	Gas	2010 Emissions ($MtCO_2eq$)
Ferrous and non ferrous metals	CO_2	2,127
	CH_4	18.87
	SF_6	8.77
	PFC	52.45
	N_2O	4.27
Chemicals	CO_2	1,159
	HFC	206.9
	N_2O	139.71
	SF_6	11.86
	CH_4	4.91
Cement*	CO_2	1,352.35
Indirect (electricity + heat)	CO_2	5,246.79
Landfill, Waste Incineration and Others	CH_4	627.34
	CO_2	32.50
	N_2O	11.05
Wastewater treatment	CH_4	666.75
	N_2O	108.04
Other industries	CO_2	3,222.24
	SF_6	40.59
	N_2O	15.96
	CH_4	9.06
	PFC	20.48
	HFC	332.38
Indirect	N_2O	24.33

Gas		2010 Emissions ($MtCO_2eq$)
Carbon dioxide	CO_2	13,139
Methane	CH_4	1,326.93
Hydrofluorocarbons	HFC	539.28
Nitrous oxide	N_2O	303.35
Perfluorocarbons	PFC	72.93
Sulphur hexafluoride	SF_6	61.21
Carbon Dioxide Equivalent (total of all gases)	**CO_2eq**	**15,443**

Note: CO_2 emissions from cement-forming reactions only; cement energy-related direct emissions are included in 'other industries' CO_2 emissions.

Weber et al., 2008). Chapter 14 provides an in-depth discussion and review of the literature related to trade, embodied emissions, and consumption-based emissions inventories.

10.3.2 Industrial non-CO_2 GHG emissions

Table 10.4 provides emissions of non-CO_2 gases for some key industrial processes (JRC/PBL, 2013). N_2O emissions from adipic acid and nitric acid production and PFC emissions from aluminium production decreased while emissions from HFC-23 from HCFC-22 production increased from 0.075 $GtCO_2eq$ in 1990 to 0.207 $GtCO_2eq$ in 2010. In the period from 1990–2010, fluorinated gases (F-gases) and N_2O were the most important non-CO_2 GHG emissions in manufacturing industry. Most of the F-gases arise from the emissions from different processes including the production of aluminium and HCFC-22 and the manufacturing of flat panel displays, magnesium, photovoltaics, and semiconductors. The rest of the F-gases correspond mostly to HFCs that are used in refrigeration equipment used in industrial processes. Most of the N_2O emissions from the industrial sector are contributed by the chemical industry, particularly from the production of nitric and adipic acids (EPA, 2012a). A summary of the issues and trends that concern developing countries and Least Developed Countries (LDCs) in this chapter is found in Box 10.1.

Table 10.4 | Emissions of non-CO_2 GHGs for key industrial processes (JRC/PBL, 2013)[1]

Process	Emissions ($MtCO_2eq$)		
	1990	2005	2010
HFC-23 from HCFC-22 production	75	194	207
ODS substitutes (Industrial process refrigeration)[2]	0	13	21
PFC, SF_6, NF_3 from flat panel display manufacturing	0	4	6
N_2O from adipic acid and nitric acid production	232	153	104
PFCs and SF_6 from photovoltaic manufacturing	0	0	1
PFCs from aluminium production	107	70	52
SF_6 from manufacturing of electrical equipment	12	7	10
HFCs, PFCs, SF_6 and NF_3 from semiconductor manufacturing	7	21	17
SF_6 from magnesium manufacturing	12	9	8
CH_4 and N_2O from other industrial processes	3	5	6

Note:

1 the data from US EPA (EPA, 2012a) show emissions of roughly the same magnitude, but differ in total amounts per source as well as the growth trends. The differences are significant in some particular sources like HFC-23 from HCFC-22 production, PFCs from aluminium production and N_2O from adipic acid and nitric acid production.

2 Ozone depleting substances (ODS) substitutes values from EPA (2012a).

10.4 Mitigation technology options, practices and behavioural aspects

Figure 10.2, and its associated identity, define six options for climate change mitigation in industry.

- **Energy efficiency (E/M):** Energy is used in industry to drive chemical reactions, to create heat, and to perform mechanical work. The required chemical reactions are subject to thermodynamic limits. The history of industrial energy efficiency is one of innovating to

Box 10.1 | Issues regarding Developing and Least Developed Countries (LDCs)

Reductions in energy intensity (measured as final energy use per industrial GDP) from 1995 to 2008 were larger in developing economies than in developed economies (UNIDO, 2011). The shift from energy-intensive industries towards high-tech sectors (structural change) was the main driving force in developed economies, while the energy intensity reductions in large developing economies such as China, India, and Mexico and transition economies such as Azerbaijan and Ukraine were related to technological changes (Reddy and Ray, 2010; Price et al., 2011; UNIDO, 2011; Sheinbaum-Pardo et al., 2012; Roy et al., 2013). Brazil is a special case were industrial energy intensity increased (UNIDO, 2011; Sheinbaum et al., 2011). The potential for industrial energy efficiency is still very important for developing countries (see Sections 10.4 and 10.7), and possible industrialization development opens the opportunity for the installation of new plants with highly efficient energy and material technologies and processes (UNIDO, 2011).

Other strategies for mitigation in developing countries such as emissions efficiency (e.g., fuel switching) depend on the fuel mix and availability for each country. Product-service efficiency (e.g., using products more intensively) and reducing overall demand for product services must be accounted differently depending on the country's income and development levels. Demand reduction strategies are more relevant for developed countries because of higher levels of consumption. However, some strategies for material efficiency such as manufacturing lighter products (e.g., cars) and modal shifts in the transport sector that reduce energy consumption in industry can have an important role in future energy demand (see Chapter 8.4.2.2).

LDCs have to be treated separately because of their small manufacturing production base. The share of manufacturing value added (MVA) in the GDP of LDCs in 2011 was 9.7 % (7.2 % Africa LDCs; Asia and the Pacific LDCs 13.3 % and no data for Haiti), while it was 21.8 % in developing countries and 16.5 % in developed countries. The LDCs' contribution to world MVA represented only 0.46 % in 2010 (UNIDO, 2011; UN, 2013).

In most LDCs, the share of extractive industries has increased (in many cases with important economic, social, and environmental problems (Maconachie and Hilson, 2013)), while that of manufacturing either decreased in importance or stagnated, with the exceptions of Tanzania and Ethiopia where their relative share of agriculture decreased while manufacturing, services, and mining increased (UNCTAD, 2011; UN, 2013).

Developed and developing countries are changing their industrial structure, from low technology to medium and high technology products (level of technology in production process), but LDCs remain highly concentrated in low technology products. The share of low technology products in the years 1995 and 2009 in LDCs MVA was 68 % and 71 %, while in developing countries it was 38 % and 30 % and in developed countries 33 % and 21 %, respectively (UNIDO, 2011).

Among other development strategies, two alternative possible scenarios could be envisaged for the industrial sector in LDCs: (1) continuing with the present situation of concentration in labour intensive and resource intensive industries or (2) moving towards an increase in the production share of higher technology products (following the trend in developing countries). The future evolution of the industrial sector will be successful only if the technologies adopted are consistent with the resource endowments of LDCs. However, the heterogeneity of LDCs circumstances needs to be taken into account when analyzing major trends in the evolution of the group. A report prepared by the United Nations Framework Convention on Climate Change (UNFCCC) Secretariat summarizes the findings of 70 Technology Needs Assessments (TNA) submitted, including 24 from LDCs. Regarding the relationship between low carbon and sustainable development, the relevant technologies for most of the LDCs are related to poverty and hunger eradication, avoiding the loss of resources, time and capital. Almost 80 % of LDCs considered the industrial structure in their TNA, evidencing that they consider this sector as a key element in their development strategies. The technologies identified in the industrial sector and the proportion of countries selecting them are: fuel switching (42 %), energy efficiency (35 %), mining (30 %), high efficiency motors (25 %), and cement production (25 %) (UNFCCC SBSTA, 2009).

A low carbon development strategy facilitated by access to financial resources, technology transfer, technologies, and capacity building would contribute to make the deployment of national mitigation efforts politically viable. As adaptation is the priority in almost all LDCs, industrial development strategies and mitigation actions look for synergies with national adaptation strategies.

create 'best available technologies' and implementing these technologies at scale to define a reference 'best practice technology', and investing in and controlling installed equipment to raise 'average performance' nearer to 'best practice' (Dasgupta et al., 2012).

Energy efficiency has been an important strategy for industry for various reasons for a long time. Over the last four decades there has been continued improvement in energy efficiency in energy-intensive industries and 'best available technologies' are increasingly 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.

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.

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

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 generally 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.

[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.

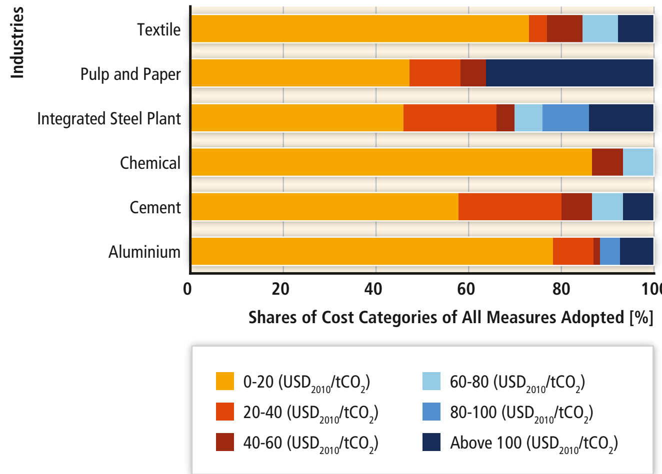


Figure 10.6 | Range of unit cost of avoided CO_2 emissions (USD_{2010}/tCO_2) in India. Source: Database of energy efficiency measures adopted by the winners of the National Awards on Energy Conservations for aluminium (26 measures), cement (42), chemicals (62), ISP: integrated steel plant (30), pulp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

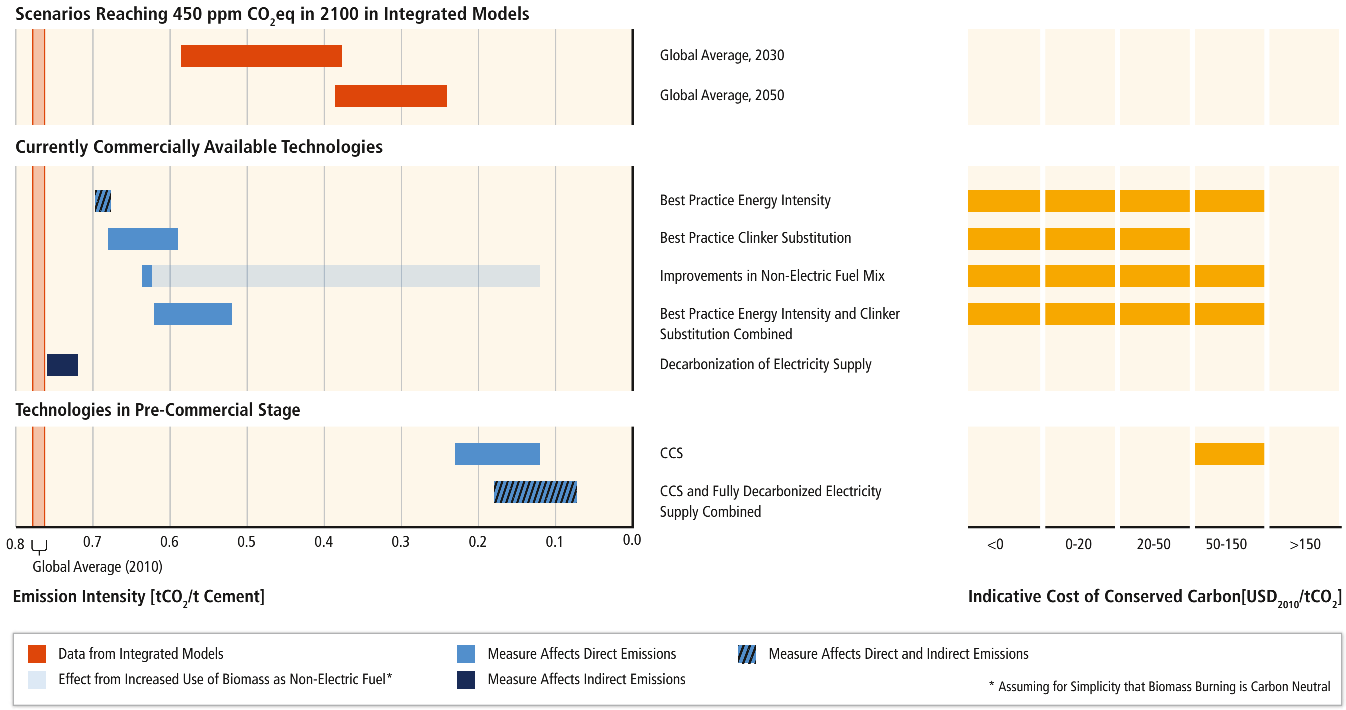


Figure 10.7 | Indicative CO_2 emission intensities and levelized cost of conserved carbon in cement production for various production practices/technologies and in 450 ppm scenarios of selected models (AIM, DNE21+, IEA ETP 2DS) (for data and methodology, see Annex III).

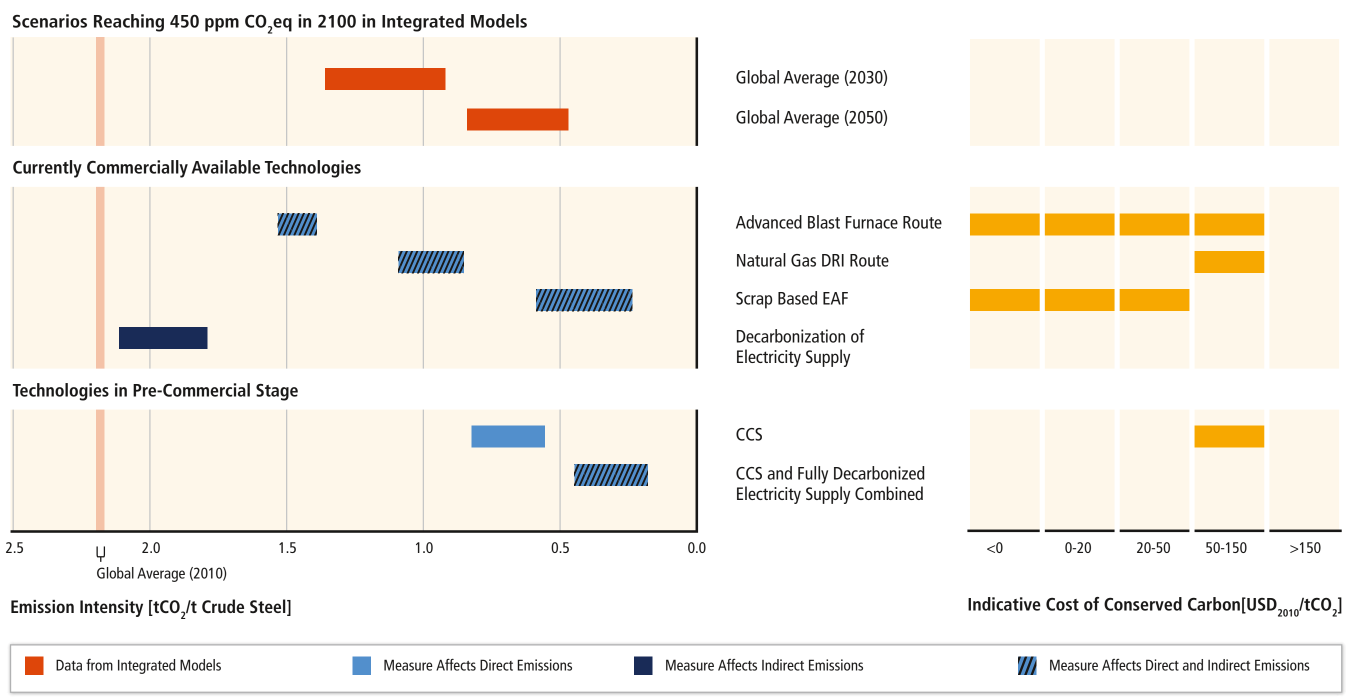


Figure 10.8 | Indicative CO_2 emission intensities and levelized cost of conserved carbon in steel production for various production practices/technologies and in 450 ppm scenarios of selected models (AIM, DNE21+, and IEA ETP 2DS) (for data and methodology, see Annex III).

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.

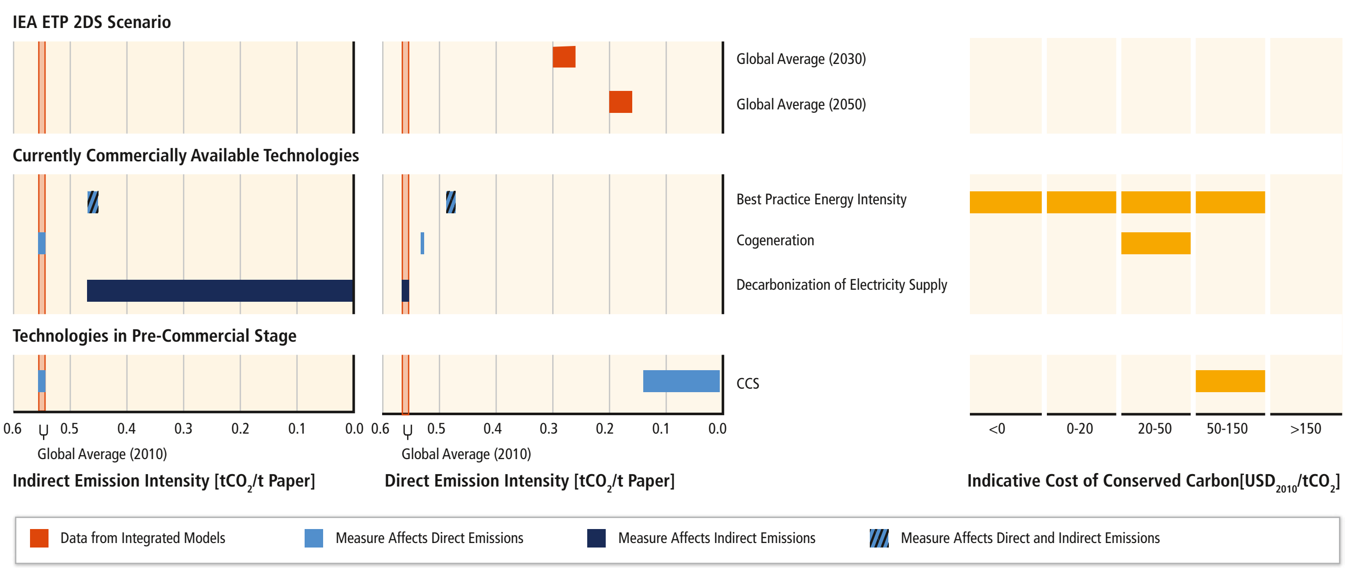


Figure 10.10 | Indicative global indirect (left) and direct (right) CO_2 emission intensities and levelized cost of conserved carbon in paper production for various production practices/technologies and in IEA ETP 2DS scenario (for data and methodology, see Annex III).

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.

[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.

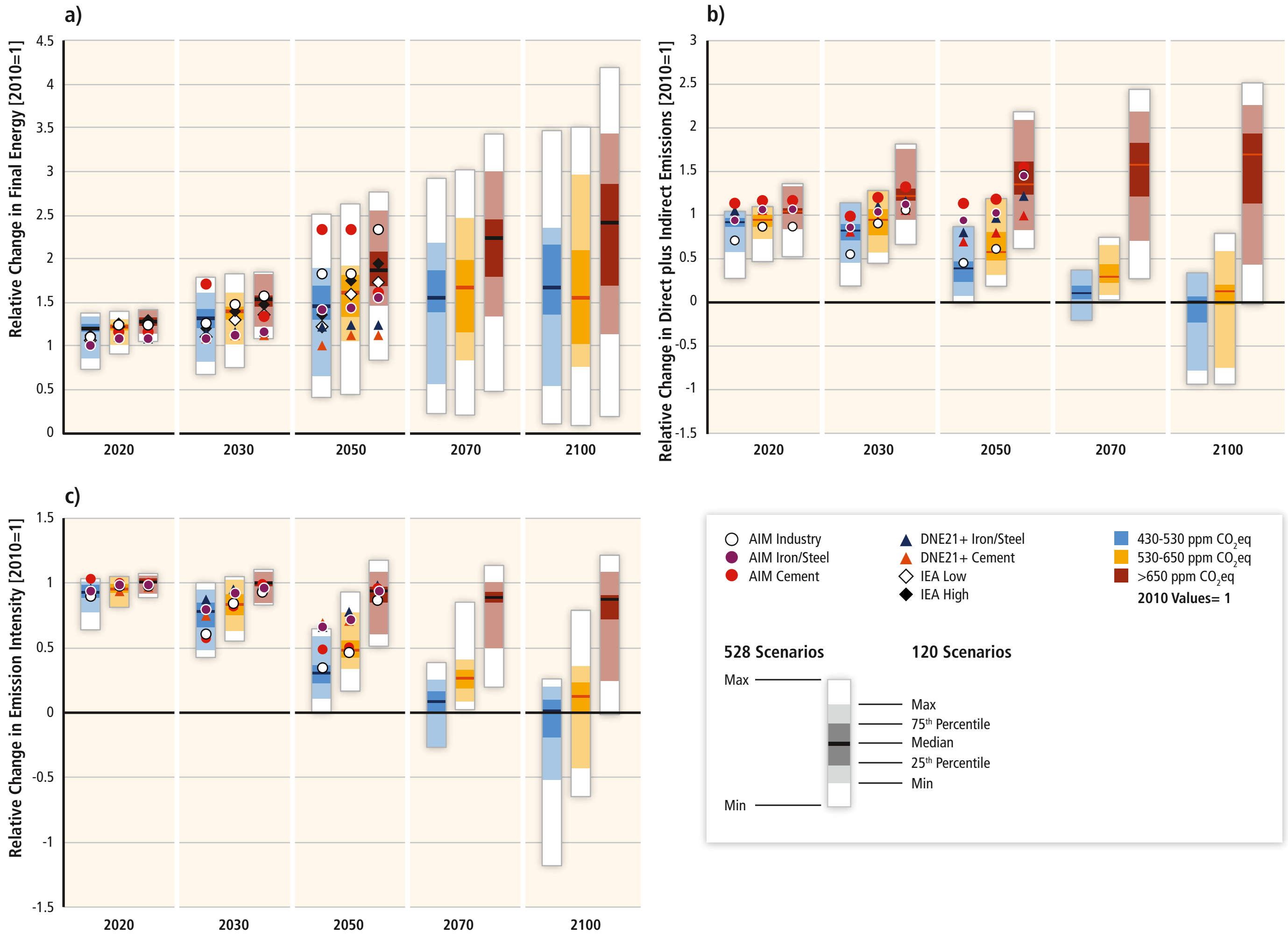


Figure 10.11 | Industry sector scenarios over the 21st century that lead to low (430–530 ppm CO_2eq), medium (530–650 ppm CO_2eq) and high (> 650 ppm CO_2eq) atmospheric CO_2eq concentrations in 2100 (see Table 6.3 for definitions of categories). All results are indexed relative to 2010 values for each 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

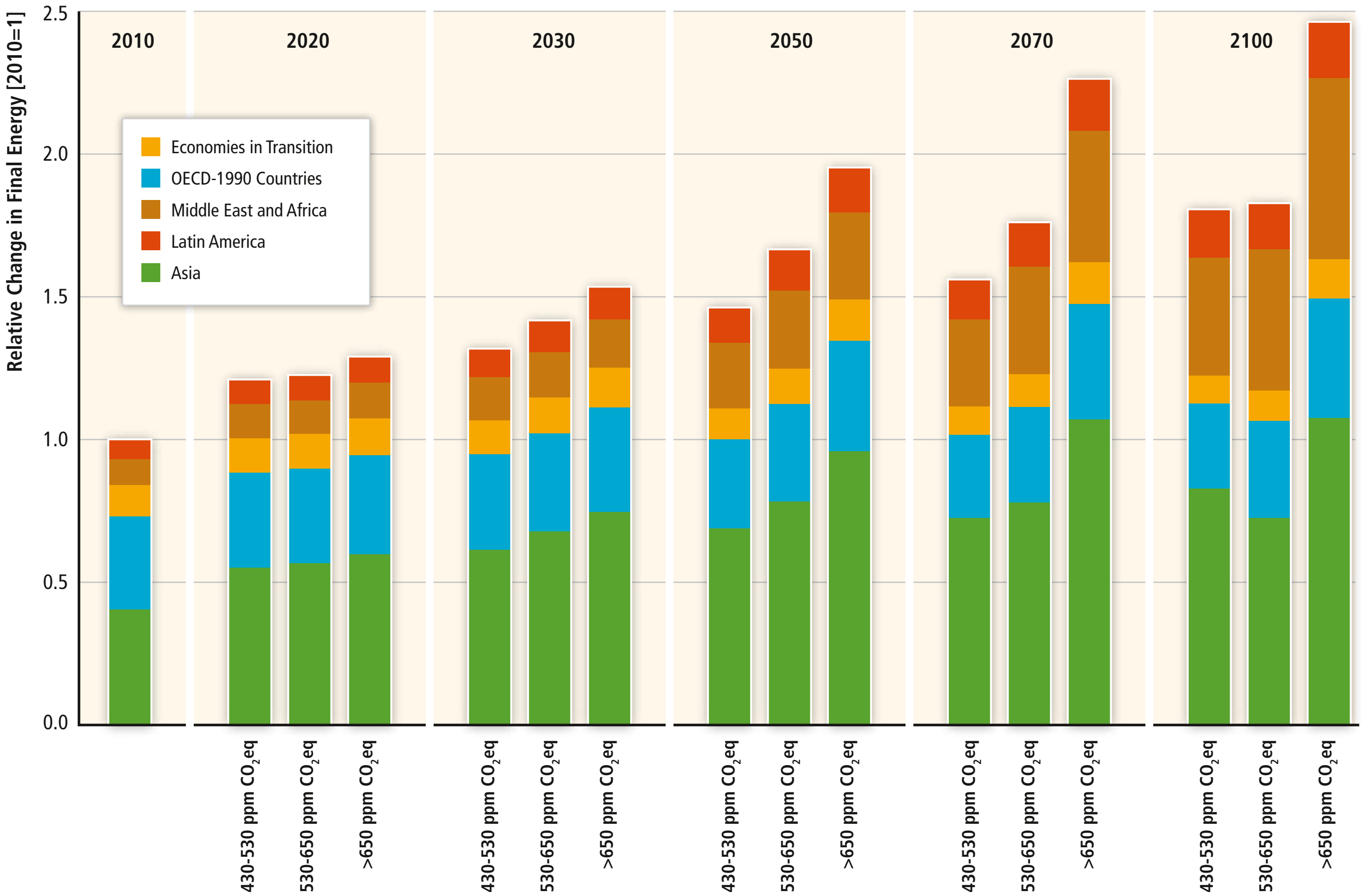


Figure 10.12 | Final energy demand from the industry sector shown for the RC5 regions (see Annex II.2 for definition) over the 21st century. Bars are compiled using information from 105 of the 120 scenarios assessed in Chapter 6, with model default technology assumptions that submitted detailed final 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.

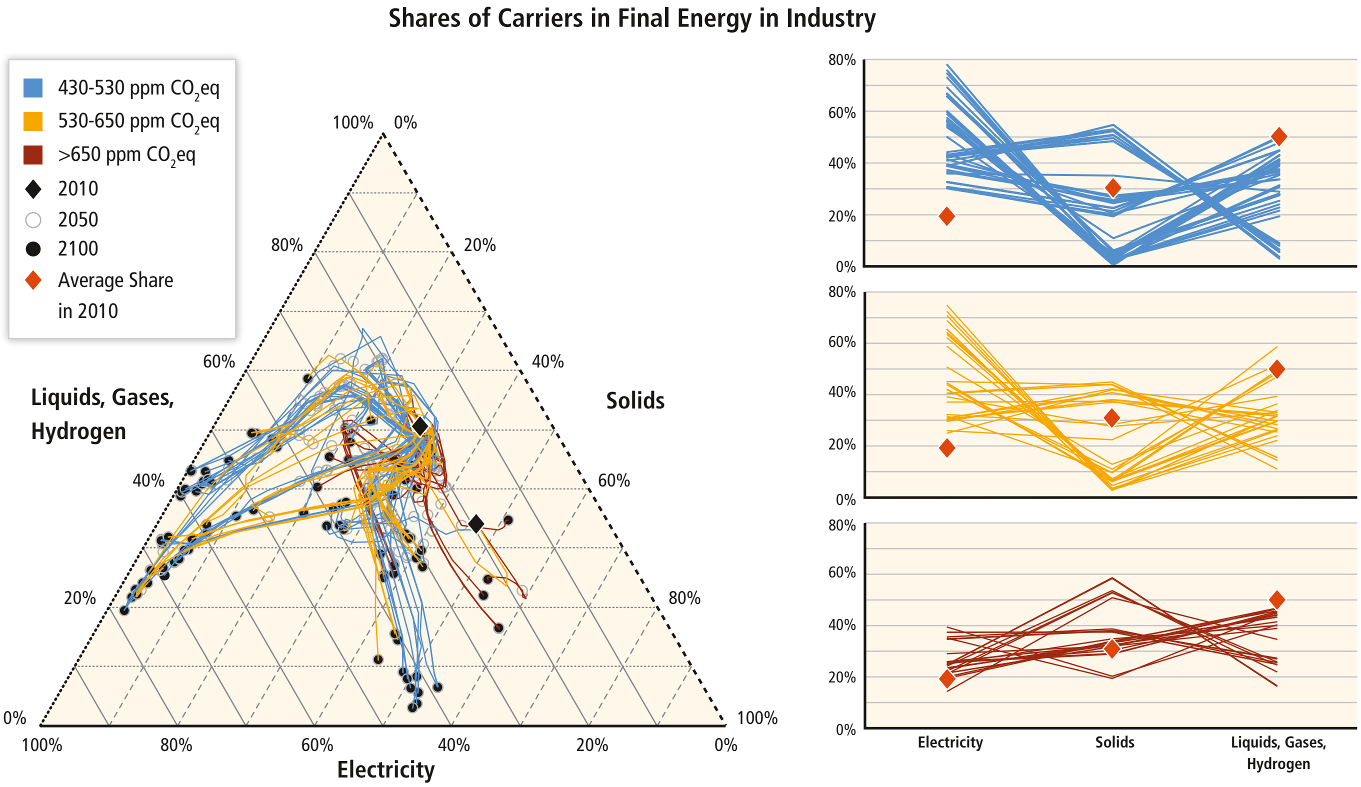


Figure 10.13 | The ternary panel on the left provides the industry final energy share trajectories across three groups of energy carriers: electricity, solids, and liquids-gases-hydrogen. The path of each scenario's trajectory is shown by a single line with symbols at the start in 2010 (the diamond towards 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-

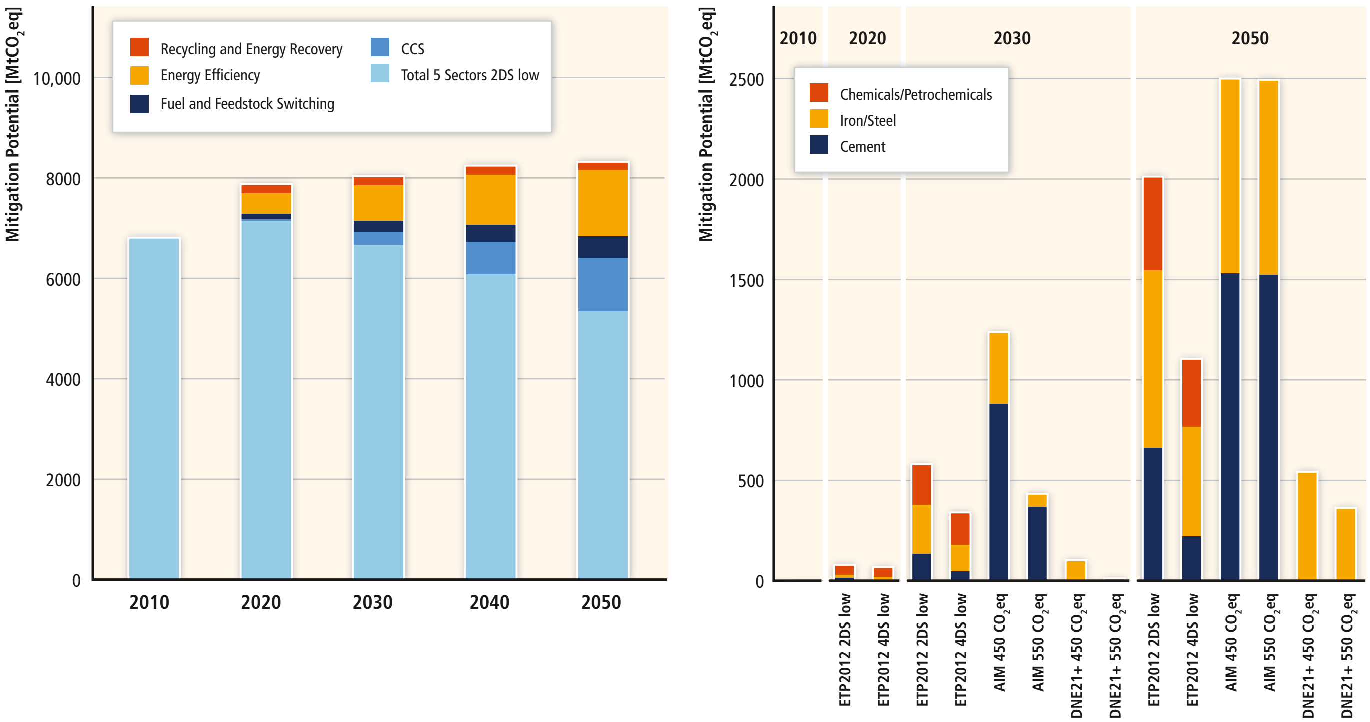


Figure 10.14 | Mitigation of direct CO_2eq annual emissions in five major industrial sectors: iron/steel, cement, chemicals/petrochemicals, pulp/paper, and aluminium. The left panel shows results from IEA scenarios (IEA, 2012d), broken down by mitigation option. The tops of the bars show the IEA 4DS 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.

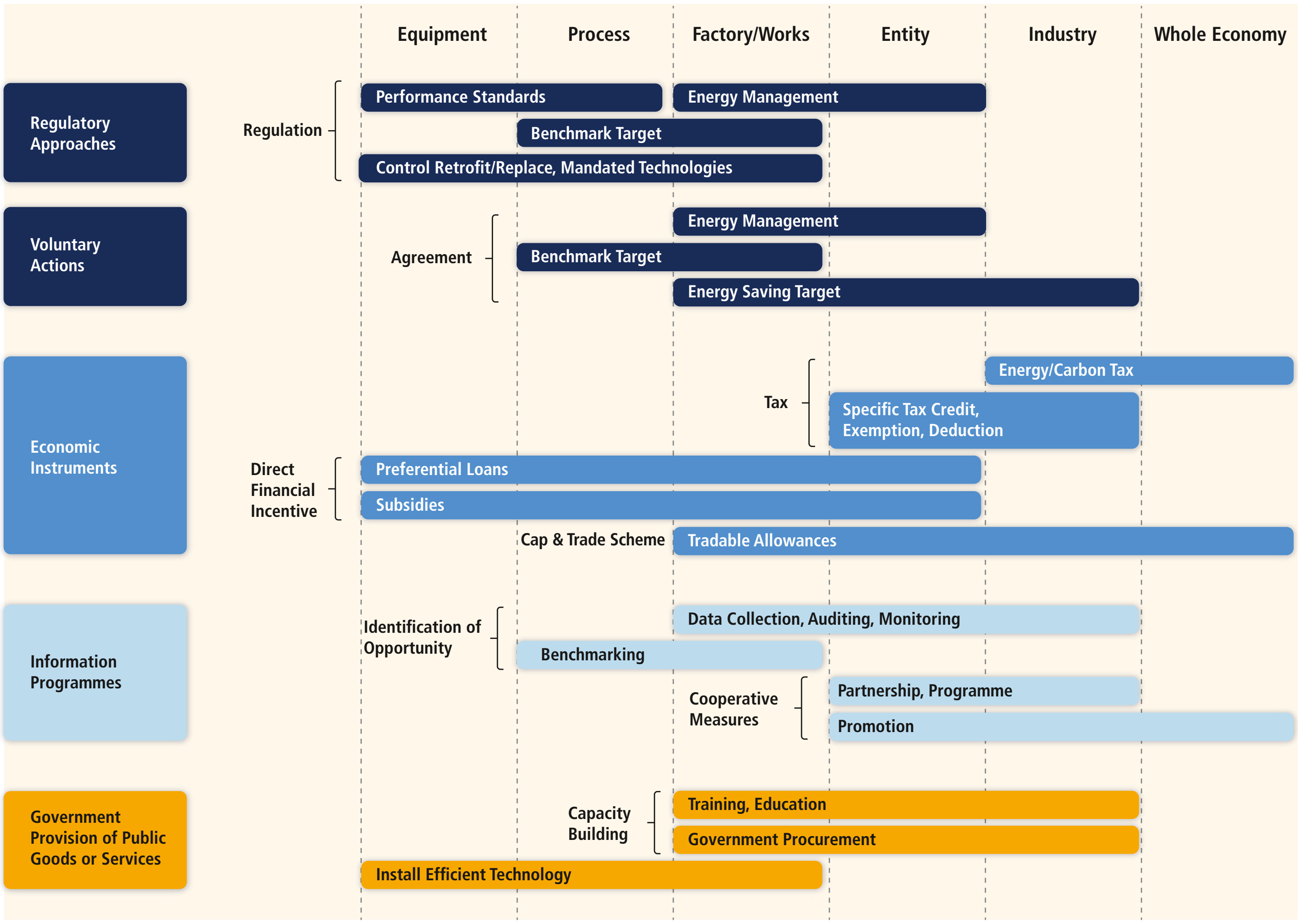


Figure 10.15 | Selected policies for energy efficiency in industry and their coverage (from Tanaka, 2011).

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.

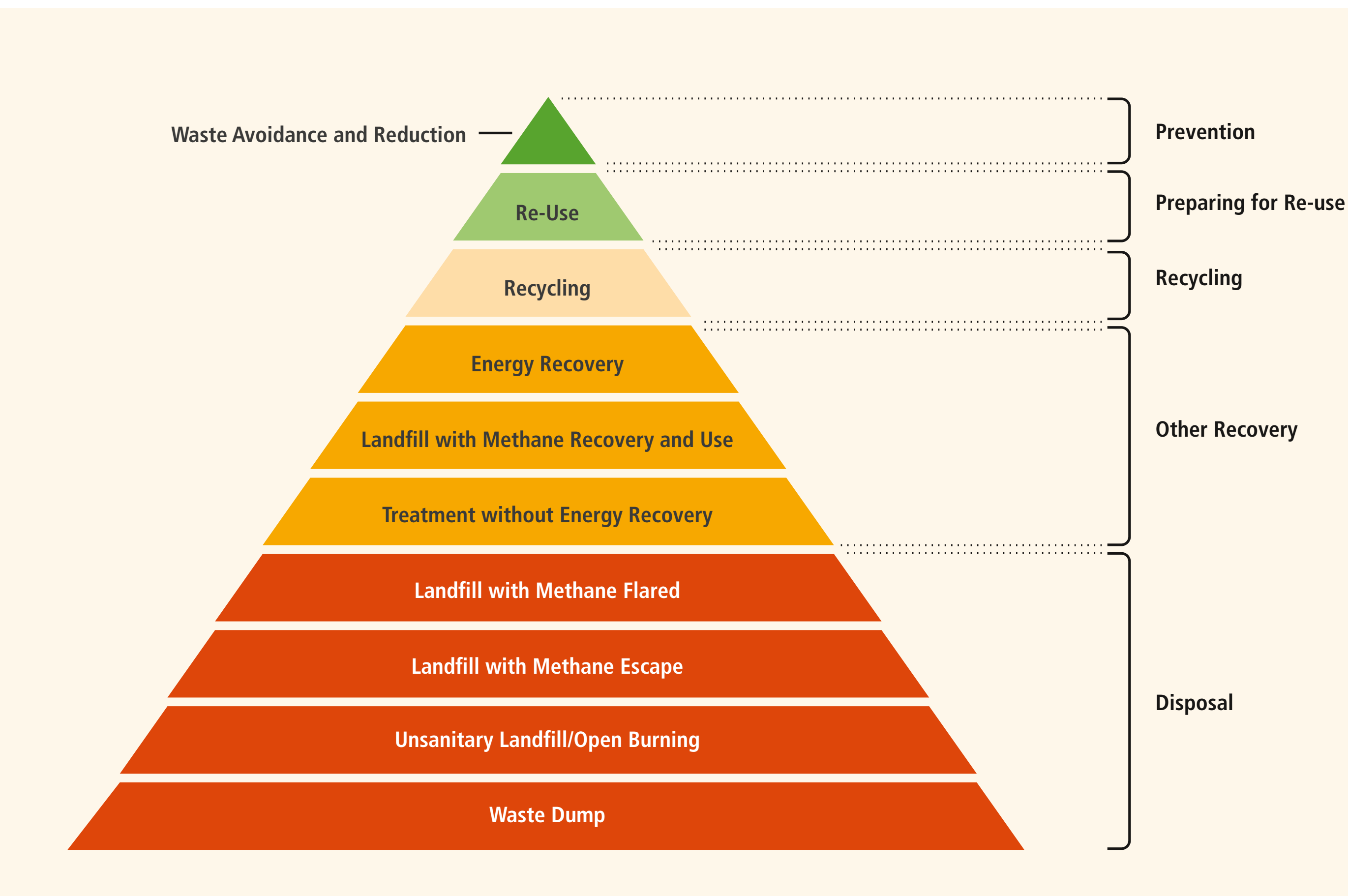


Figure 10.16 | The hierarchy of waste management. The priority order and colour coding is based on the five main groups of waste hierarchy classification (Prevention; Preparing for Re-Use; Recycling; Other Recovery e.g., Energy Recovery; and Disposal) outlined by the European Commission (EC, 2008b).

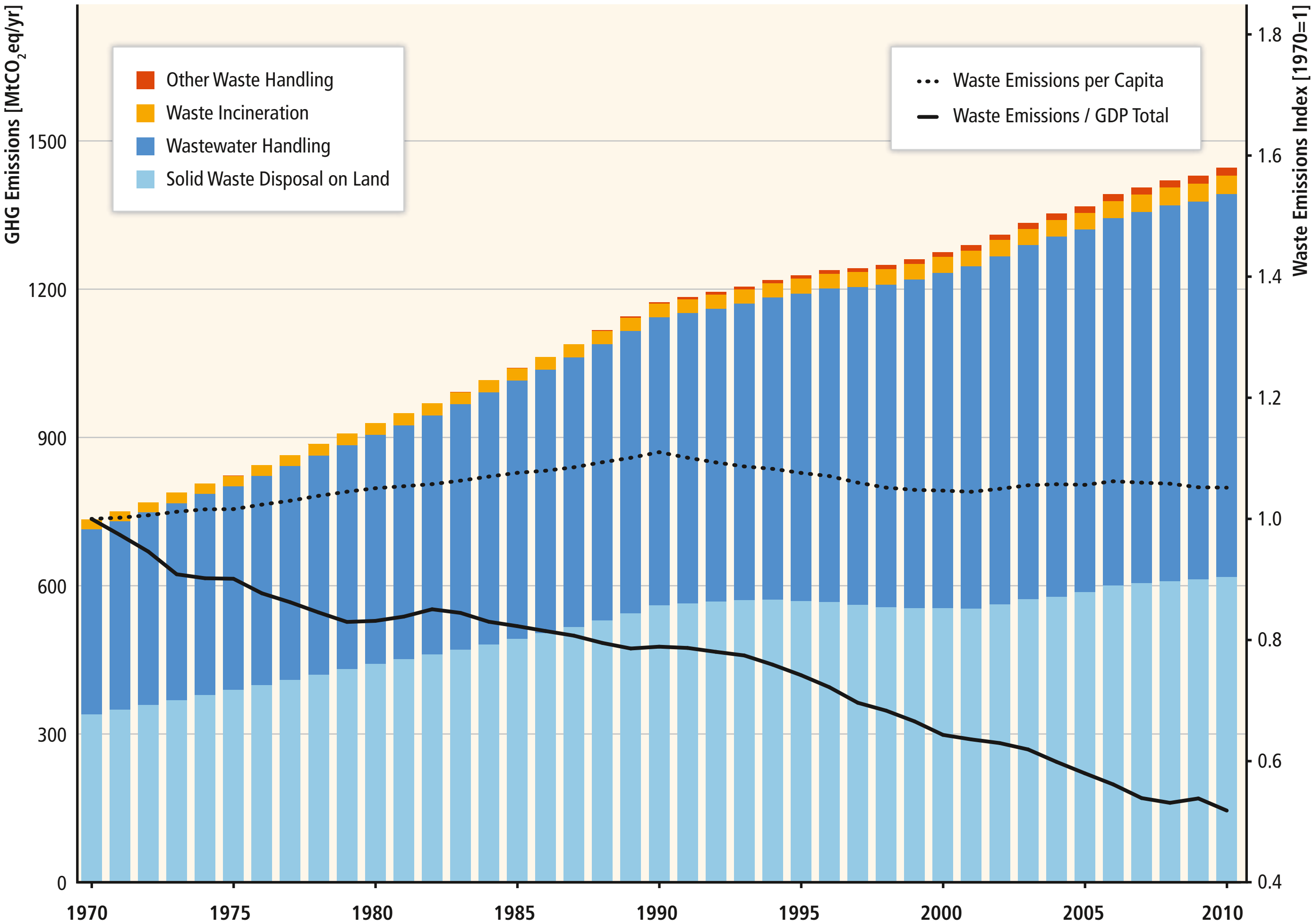


Figure 10.17 | Global waste emissions $MtCO_2eq$/year, global waste emissions per GDP and global waste emissions per capita referred to 1970 values. Based on JRC/PBL (2013), see Annex II.9.

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) .

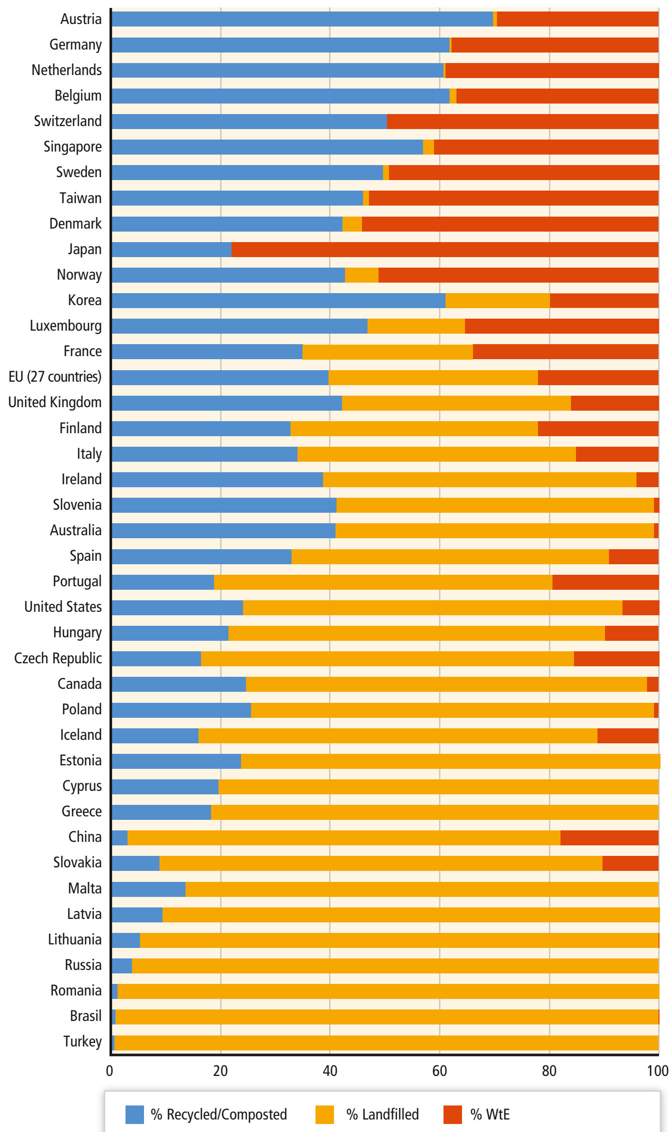


Figure 10.18 | Management practices concerning MSW in several nations (based on World Bank and national statistics, methodology described in Themelis and Bourtsalas (2013).

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

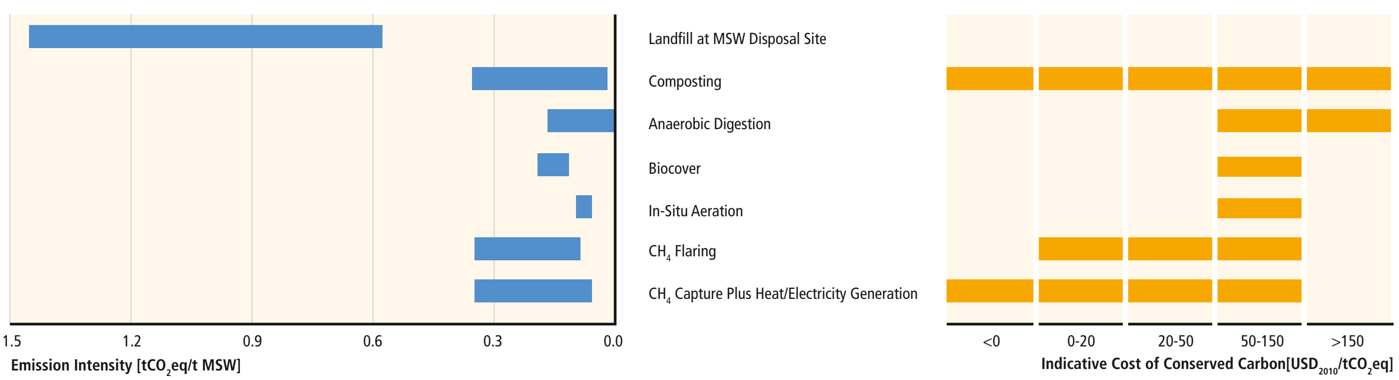


Figure 10.19 | Indicative CO_2eq emission intensities and levelized cost of conserved carbon of municipal solid waste disposal practices/technologies (for data and methodology, see Annex III).

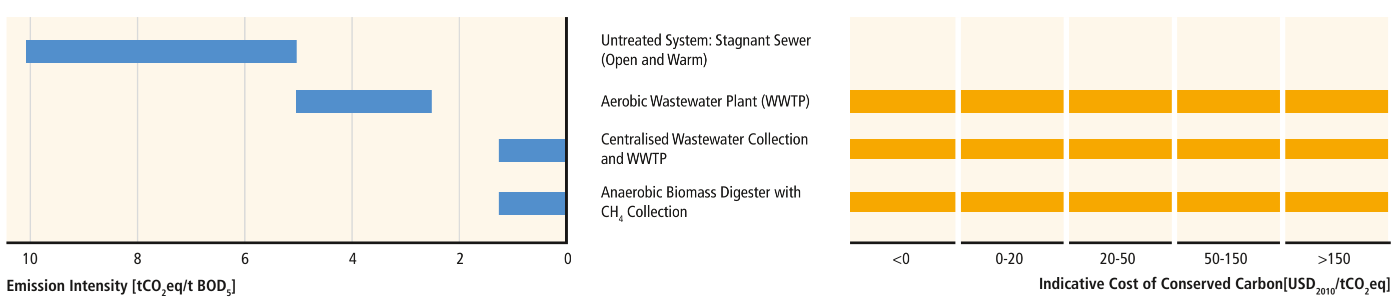


Figure 10.20 | Indicative CO_2eq emission intensities and levelized cost of conserved carbon of different wastewater treatments (for underlying data and methodology, see Annex III).

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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