Chapter 11: Industry

Coordinating Lead Authors: Igor Alexeyevich Bashmakov (the Russian Federation) and Lars J. Nilsson (Sweden)

Lead Authors: Adolf Acquaye (United Kingdom), Christopher Bataille (Canada), Jonathan Michael Cullen (United Kingdom), Stéphane de la Rue du Can (the United States of America), Manfred Fischedick (Germany), Yong Geng (China), Kanako Tanaka (Japan)

Contributing Authors: Daniel Perczyk (Argentina), Fredric Bauer (Sweden), Anna Myshak (the Russian Federation), Cedric Philibert (France), Sascha Samadi (Germany), Peter Levi (United Kingdom)

Review Editors: Nick Campbell (France), Ramón Pichs-Madruga (Cuba)

Chapter Scientist: Siyue Guo (China)

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Executive summary

The Paris Agreement, the Sustainable Development Goals (SDGs) and the COVID-19 pandemic provide a new context for the development of industry *(high confidence)*. This chapter is focused on what is new since AR5. It emphasises the energy and emissions intensive basic materials industries and key strategies for reaching net-zero emissions. {11.1.1}

Net-zero emissions from the industrial sector is possible *(high confidence)*. It requires provision of greenhouse gas (GHG) emissions free electricity and high temperature heat, and other low carbon energy carriers, low carbon feedstocks, and a combination of energy efficiency, reduced materials demand, improved materials efficiency, a more circular economy, electrification and hydrogen use, as well as carbon capture and use (CCU) and storage (CCS). It also requires substantial scaling up of electricity, gas, hydrogen, recycling, and other infrastructure, as well as phase-out of, for example, blast furnaces in steelmaking and conversion of chemical industries to low GHG feedstocks and fuels. {11.3, 11.4}

Whatever metric is used, industrial emissions have been growing faster since 2000 than emissions in any other sector, driven by increased basic materials extraction and production *(high confidence)*. GHG emissions attributed to the industrial sector originate from fuels combustion, processes emissions, product use and waste which jointly accounted for 23% of all direct anthropogenic emissions in 2018, second behind the energy sectors. Industry becomes a leading GHG emitter (19.3 $GtCO_2$-eq (31%) in 2018) if indirect emissions from power and heat generation are included. {11.2.2}

In the 21st century there is a clear trend that industry accounts for an increasing share of overall GHG emissions *(medium confidence)*. The transition to net-zero emissions in industry may take longer due to lock-in long-lived capital stock; the early stage of development and commercialisation for many technologies with deep emissions reduction potential; material demand from other sectors when decarbonising, and high sensitivity to carbon price driven increases in production costs for globally traded commodities. {11.2.2, 11.5.1, 11.5.2}

In contrast to global energy intensity, which is declining, material intensity (in-use stock of manufactured capital per unit of gross domestic product (GDP)) is increasing *(high confidence)*. In-use stock of manufactured capital per capita has been growing faster than GDP per capita since 2000. Total global in-use stock of manufactured capital grew at an annual growth rate of 3.8% in 1971–2000 and 3.5% in 2000–2017. At the same time, per capita material stocks in several developed countries have stopped growing, showing a relative decoupling from GDP per capita. {11.2.1, 11.3.1}

The emissions share by source as part of total emissions is changing in industry *(high confidence)*. The share of emissions originating from industrial fuel combustion is declining whereas the contribution is growing from process emissions, electrification and indirect emissions, and waste. Developing emissions free new industrial processes and decarbonising electricity supplies are therefore key strategies for reaching net-zero emissions in industry. {11.2.2}

Plastic is the material for which demand has been growing the strongest since 1970 *(high confidence)*. The current >99% reliance on fossil feedstock, very low recycling, and high emissions from petrochemical processes is a challenge for reaching net-zero emissions. At the same time, plastics is important for reducing emissions elsewhere, for example, as an insulation material in power cables or for preserving food. There are yet no shared visions for fossil-free plastics, but several possibilities. {Box 11.2}

Scenario analyses show that significant cuts in GHG emissions and even close to net zero emissions from energy intensive industry (e.g., steel, plastics, ammonia, and cement) can be achieved by 2050 by deploying multiple and available options *(medium confidence)*. It requires a reorientation from the historic focus on incremental improvements (e.g., through energy efficiency) to transformational changes in energy and feedstock sourcing, materials efficiency, and more circular material flows. This involves structural changes, including the establishment of new infrastructures, appropriate planning methods and public acceptance, as well as sequencing of the various mitigation options. {11.3, 11.4}

Key climate mitigation options such as materials efficiency, circular material flows and new primary processes, with some progress in recent years, are still highly underrepresented in climate change scenario modelling and integrated assessment models *(high confidence)*. The character of these interventions (e.g., appearing in many forms across complex value chains, making cost estimates difficult) and the novelty of understanding new fossil free primary processes help explain why they are less represented than, for example, CCS in models. As a result, effective mitigation options are missed while overall mitigation costs and the need for CCS may be overestimated. {11.4.2}

Electrification is emerging as a key mitigation option for industry in general and for primary and secondary basic material production *(high confidence)*. Electricity is a versatile and carbon free energy carrier, potentially produced from very large potential renewable energy sources (e.g., wind and solar) or other low carbon options (e.g. biomass, nuclear or fossil fuels with >90% capture CCS); regional resources and preferences will vary. Using electricity directly, or indirectly via hydrogen from electrolysis for high temperature processes and other applications offer several options to reduce emissions, as well as for providing substantial grid balancing services, for example, through electrolysis and storage of hydrogen for process use. {11.3.5}

Carbon is a key building block in organic chemicals, fuels and materials and will remain important *(high confidence)*. In order to reach net zero emissions, it is important to close the loops on carbon and carbon dioxide through increased circularity with mechanical and chemical recycling, use of biomass feedstock or direct air capture, for the carbon needed in society (e.g., plastics, wood, aviation fuels, solvents, etc.). Addition of hydrogen can become important for maximising the potential of low GHG carbon sources (e.g. biomass) for fuels and feedstocks. {11.3.6, 11.4.1.3}

The technological capacity for very low to zero emissions industrial materials exists. Costs will be high for primary producers but low for final consumers and the general economy *(medium confidence)*. Costs and emissions reductions potential in industry, and especially heavy industry, are highly contingent on innovation, commercialisation, and uptake policy. Technologies exist to take all sectors to very low or zero emissions, but require 5–15 years of intensive innovation, commercialisation, and policy to ensure uptake. Directs costs are in the rough range of USD50–150 $tCO_2\text{-eq}^{-1}$, with wide variation within and outside this band. This affects competitiveness and requires supporting policy. These focussed costs on producers translate to much smaller increases for intermediate manufacturers and final consumers. For example, it will likely cost 20–40% more for virgin green steel, 5–10% for steel parts, but will add below 1–2% on the price for a new car or a new house, based on higher costs for steel and cement respectively. {11.3.7, 11.4.1.5}

There are several technological options for very low to zero emissions steel, but their uptake will require intensive, integrated material efficiency, recycling, and production decarbonisation policies *(high confidence)*. Material efficiency can potentially reduce steel demand by up to 40% based on design for less steel use, long life, reuse, constructability, and low contamination recycling. Secondary production through high quality recycling must be maximised. Production decarbonisation will also be required, starting with the retrofitting of existing facilities for partial fuel switching (e.g. to hydrogen), CCU and CCS, followed by very low and zero emissions production based on CCS or direct hydrogen or electrolytic iron ore reduction followed by an electric arc furnace. {11.4.1.1}

There are several current and near horizon options to greatly reduce cement and concrete emissions. Producer, user and regulator education is required for the former, and intensive innovation and commercialisation policy for the latter *(medium confidence)*. Cement and concrete are currently overused because they are inexpensive, durable, and ubiquitous. Basic material efficiency efforts to use only well-made cement (e.g. with well sized and mixed aggregates) where it is needed could reduce emissions by 24–50%. CCS will likely be essential for eliminating the roughly 60% of emissions from the limestone calcination process to make clinker, the glue in concrete. The remaining process heat emissions can be reduced via fuel switching to biomass, fossil-free waste fuel, or potentially electricity and hydrogen. Eliminating cement emissions will require new chemistries that will likely take decades to be commercialised and accepted in markets. {11.4.1.2}

While several technological options for decarbonising the main feedstock chemicals and their derivatives exist, the costs vary widely *(high confidence)*, partly because fossil fuel based feedstocks are very inexpensive, and their biomass and electricity based replacements will likely be more constrained and expensive. The chemical industry consumes large amounts of hydrogen, ammonia, methanol, carbon monoxide, ethylene, propylene, benzene, toluene, and mixed xylenes & aromatics; and produces tens of thousands of derivative end-use chemicals. Hydrogen, biocarbon and collected plastic waste for the primary feedstocks for all the above can greatly reduce total emissions, but biogenic carbon is likely to be limited in low carbon world with competition for land for food production, biodiversity, and land use negative carbon sinks. {11.4.1.3, 11.3.5, 11.3.6}

Light industry and manufacturing can be largely decarbonised through switching to low GHG fuels (e.g., biofuels and hydrogen) and electricity (e.g., for electrothermal heating and heat pumps) *(high confidence).* {11.4.1.4}

The pulp and paper industry has good potentials for reducing emissions through close access to biomass and its use of process heat at low to medium temperatures *(high confidence)*. The pulp and paper industry is energy intensive but not a large direct emitter if it uses sustainably sourced feedstock and bioenergy rather than fossil fuels. Competition for feedstock will increase if wood substitutes for building materials and petrochemicals feedstock. {11.4.1.4}

In the context of industry, climate change mitigation and fostering sustainable development are two mutually reinforcing sides of the same coin *(medium confidence)*. Climate action is key to any progress that can be made towards enhancing sustainable development and so it is unsurprising that many of the SDGs are strongly linked to climate mitigation. The character of these links is different across various regional contexts and local conditions but particularly salient in relation to SDG 8 (Decent work and economic growth), SDG 9 (Industry innovation and infrastructure), and SDG 12 (Responsible consumption and production). {11.5.3}

The geographical distribution of renewable resources has implications for industry *(medium confidence)***.** The potential for "green" hydrogen from electrolysis powered by solar and wind, or hydrogen from other zero emission sources may reshape where heavy industry is located, how value chains are organised, and what gets transported in international shipping. Regions with bountiful solar and wind resources, or methane co-located with CCS geology, may become exporters of hydrogen or hydrogen carriers such as methanol and ammonia, or home to the production of iron and steel, organic platform chemicals, and other energy intensive basic materials. This in turn may generate new trade patterns and needs for bulk transport. {Box 11.1}

The level of maturity and policy experience varies widely across the mitigation options. Energy efficiency is a well-established policy field with decades of experience from voluntary and negotiated agreements, regulations, energy audit and demand side-management (DSM) programs, etc (see AR5). In contrast, demand management and materials efficiency are not well understood from a policy perspective. Barriers to recycling that policy could address are often specific to the different material loops (e.g., copper contamination for steel and lack of technologies or poor economics for plastics). For electrification and fuel switching the focus has so far been mainly on innovation and developing technical supply-side solutions rather than creating market demand. {11.6}

Industry has so far largely been sheltered from the impacts of climate policy due to concerns for competitiveness and carbon leakage *(high confidence)*. New industrial development policy approaches will be needed for realising a transition to net zero emissions. The transition requires a clear direction towards net-zero, technology development, market demand for green materials and products, governance capacity and learning, as well as international coordination of climate and trade policies. It requires broad and sequential industrial policy strategies with governance at different levels (from international to local) and across traditional policy domains (e.g., from climate policy to welfare and education). {11.5.2, 11.6}

11.1 Introduction and new developments

11.1.1 About this chapter

AR5 was published in 2014. The Paris Agreement (PA) and the 17 SDGs were adopted in 2015. An increasing number of countries have since announced ambitions to be carbon neutral by 2045-2060. The COVID-19 pandemic shocked the global economy in 2020 and motivated economic stimulus with demands for green recovery and concerns for economic security. All this has created a new context and a growing recognition that all industry, including the energy and emissions intensive industries, need to reach net zero emissions. There is an ongoing mind shift around the opportunities to do so, with electrification and hydrogen emerging as key mitigation options as a result of renewable electricity costs falling rapidly. On the demand side there has been renewed attention to end-use demand, material efficiency and more and better-quality recycling measures. This chapter takes its starting point in this new context and emphasises the need for deploying innovative processes and practices in order to limit the global warming to 1.5ºC or 2ºC (IPCC 2018).

The industrial sector GHG emissions include direct and indirect fuel combustion related emissions, emissions from industrial processes and some products use, as well as from waste. This chapter is focused on the energy and emissions intensive basic materials industries that account for 60% of direct and total direct and indirect combustion and processes related industrial emissions (waste excluded). The transition to zero emissions requires supplementing the traditional toolkit of energy and process efficiency, fuel switch, electrification, and decarbonisation of power with material end-use demand management and efficiency, circular economy, fossil-free feedstocks, CCU, and CCS. Energy efficiency was extensively treated in AR5 and remains a key mitigation option. This chapter is focused mainly on new options and developments since AR5, highlighting measures along the whole value chains that are required to approach zero emissions in primary materials production.

11.1.2 Approach to understanding industrial emissions

The Kaya-identity offers a useful tool of decomposing emission sources and their drivers, as well as of weighing the mitigation options. The one presented below builds on the previous assessments (see (IPCC 2014, 2018)), and reflect a material stock-driven services-oriented vision to better highlight the growing importance of industrial processes (dominated emissions increments in 2010-2018), products use and waste in driving emissions. Services delivery (nutrition, shelter, mobility, education, etc., see Chapter 5 for more detail) not only requires energy and materials flows (fuels, food, feed, fertilisers, packaging, etc.), but also material stocks (buildings, roads, vehicles, machinery, etc.), the weight of which has already exceeded 1,000 Gt (Wiedenhofer et al. 2019). As material efficiency appears to be an important mitigation option, material intensity or productivity (material extraction or consumption versus GDP (IRP 2020, 2019)) is reflected in the identity with two dimensions: as material stock intensity of GDP (tons per dollar) and material intensity of building and operating accumulated in-use stock.[1] For sub-global analysis the ratio of domestically used materials to total material production becomes important to reflect outsourced materials production and distinguish between territorial and consumption-based emissions. The identity for industry differs much from that for sectors with combustion emissions dominance. Recent progress in data availability that allows it to integrate major

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[1] FOOTNOTE Accumulated material stock initially was introduced in the analysis of past trends (Wiedenhofer et al. 2019; Krausmann et al. 2018), but recently it was incorporated in different forms in the long-term projections for the whole economy (Krausmann et al. 2020) and for some sectors (buildings and cars in (IRP 2020)) with a steadily improving regional resolution (Krausmann et al. 2020).

emission sources along with socio-economic metabolism, material flows and stocks analysis enriches the identity for industry reflecting impacts of the following components (Equation 11.1):

$$GHG = POP \cdot \frac{GDP}{POP} \cdot \frac{MStock}{GDP} \cdot \frac{MPR + MSE}{MStock} \cdot Dm \cdot \left(\frac{E}{(MPR + MSE)} \cdot \frac{(GHGed + GHGeind)}{E} + \frac{GHGoth}{MPR + MSE} \right)$$

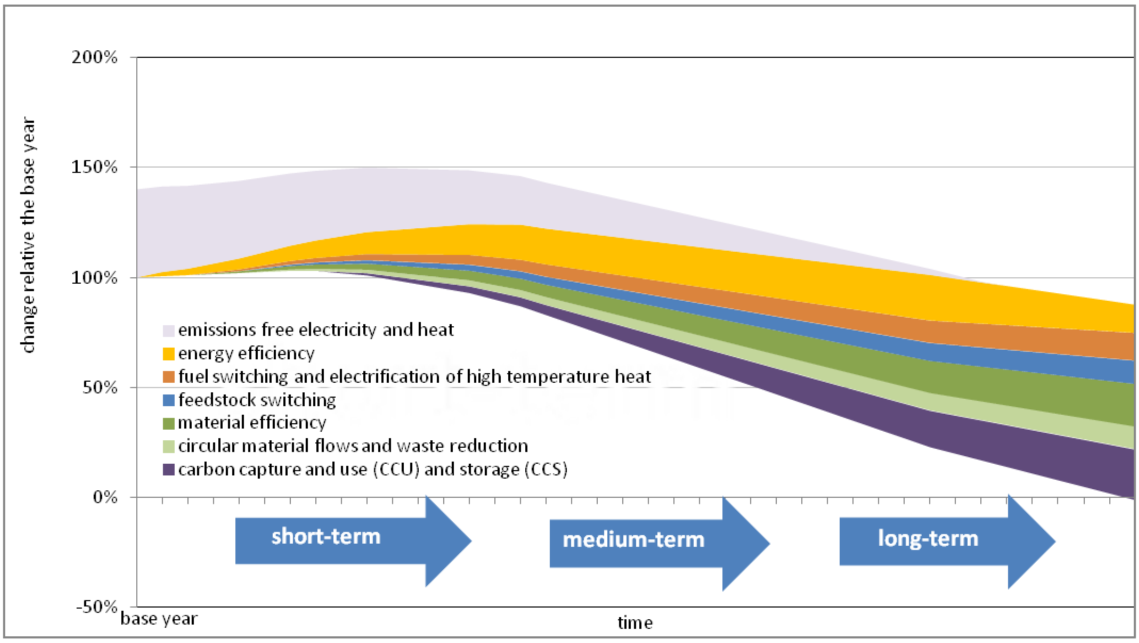
Equation 11.1

Factors	population	services (expressed via GDP – final consumption and investments needed to maintain and expand stock) per capita	material stock (accumulated in-use stocks of materials embodied in manufactured fixed capital) intensity of GDP	material inputs (both virgin (*MPR*) and recycled (*MSE*)) per unit of in-use material stock	share of allocated emissions - (valid only for sub-global levels)*	energy use for basic material production (E_m), processing and other operational industrial energy use (E_{oind}) per unit of material inputs	direct (GHG_{ed}) and indirect (GHG_{eind}) combustion-related industrial emissions per unit of energy	emissions from industrial processes and product use, waste, F-gases, indirect nitrogen emissions per unit of produced materials
Policies	population control policies	sufficiency and demand management (reduction)	material stock efficiency improvement	material efficiency, substitution and circular economy	trade policies including carbon leakage issues (localization versus globalization)	energy efficiency of basic materials production and other industrial processes	electrification, fuel switching, and energy decarbonisation (hydrogen, CCUS-fuels)	feedstock decarbonisation (hydrogen), CCUS-industrial processes, waste and F-gases management
Demand decarbonisation						Production decarbonisation		

* *Dm*=1, when territorial emission is considered, and *Dm* equals the ratio of domestically used materials to total material production for the consumption-based emission accounting)

Factors in (Equation 11.1) are interconnected by either positive (synergy) or negative feedbacks: scrap-based production or light weighing improves operational energy efficiency, while growing application of carbon capture, use and storage (CCUS) brings it down (Hertwich et al. 2019; IEA 2020a). There are different ways to disaggregate Equation 11.1: by industrial subsectors; by reservoirs of material stock (buildings, infrastructure, vehicles, machinery and appliances, packaging, etc.); by regions and countries (carbon leakage comes out to the stage); by products and production chains (material extraction, production of basic materials, basic materials processing, production of final industrial products); by traditional and low carbon technologies used; by stages of products' lives including recycling.

Industrial transition to net-zero emissions is possible when the three last multipliers in Equation 11.1 are approaching zero. Contributions from different drivers (energy efficiency, low carbon electricity and heat, material efficiency, switching to low carbon feedstock and CCUS) to this evolution vary with time. Energy efficiency dominate in the short- and medium-term and potentially long-term (in the range of 10-40% by 2050)(IPCC 2018; Crijns-Graus et al. 2020; IEA 2020a), but for deep decarbonisation trajectories contributions from other drivers steadily grow, as the share of non-energy sources in industrial emissions escalates and new technologies to address mitigation from these sources mature (IEA 2020a; IRP 2020; Hertwich et al. 2019; Material Economics 2019; CAMBUREAU 2020; BPS 2020) (Figure 11.1).



Mitigation options

Decarbonising production	Reduction of indirect emissions via lower carbon electricity and heat supply	Provision of emissions free electricity and high temperature heat
	Energy efficiency improvements to best available technologies	Energy efficiency approaching thermodynamic minimums
	Fuel switching, biomass and electricity use for high temperature process heat	Deep low carbon electrification, green hydrogen use
	Partial substitution of high carbon feedstock	Zero emissions feedstock (green hydrogen, biomass) for basic materials production
	Pilot CCUS	Large-scale CCUS
Decarbonising demand	Material efficiency and substitution	Eco-design, material efficiency, demand reduction
	Increasing recycling rates	Circular material flows and effective industrial waste management

Figure 11.1 Stylised composition and contributions from different drivers to the transition of industry to net-zero emissions (contributions from the factors are only illustrative)

11.2 New trends in emissions and industrial development

11.2.1 Major drivers

The use of materials is deeply coupled with economic development and growth. For centuries, the mankind has been producing and using hundreds of materials (Ashby 2012), the diversity of which skyrocketed in the recent half-century to achieve the desired performance and functionality of multiple products (density; hardness; compressive strength; melting point, resistance to mechanical and thermal shocks and to corrosion; transparency; heat- or electricity conductivity; chemical neutrality or activity, to name a few). New functions drive the growth of material complexity of products; for example, a modern computer chip embodies over 60 different elements (Graedel et al. 2015).

Key driving factors behind the industrial GHG emissions evolution since 1900 include: on the growth side – population and per capita GDP;[2] on the reduction side – energy efficiency and non-combustion GHG emissions intensity (from industrial processes and waste). Material efficiency factors – material stock intensity of GDP and ratio of extraction, processing and recycling of materials per unit of built capital along with combustion-related emissions intensity factors – were cyclically switching their contributions with relatively limited overall impact. Recycling allowed for replacing some energy intensive virgin materials and thus contributed to mitigation. In 2010–2018, and especially in 2014–2018, a combination of these drivers allowed for a slowdown in industrial GHG emissions (Figure 11.2 and Table 11.1).

There are two major concepts of **material efficiency** (*ME*). The broader one highlights demand reduction via policies promoting more intensive use, assuming sufficient (excl. luxury) living space or car ownership providing appropriate service levels – housing days or miles driven and life-time extension (Hertwich et al. 2019; IRP 2020). This approach focuses on the dematerialisation of society (Lechtenböhmer and Fischedick 2020) and limits both material stock and GDP growth, as fewer activities are required to build and operate the stock to deliver sufficient services. According to IRP (2020), reducing floor space demand by 20% via shared and smaller housing compared to the reference scenario would decrease Group of Seven (G7) countries GHG emissions from the material-cycle of residential construction by over 70% in 2050. The narrower concept ignores demand and sufficiency aspects and focuses on supply chains considering *ME* as smaller materials use to produce a certain final product –a car or 1 m^2 of living space (IEA 2020a; OECD 2019a). No matter if the broader or the narrower concept of *ME* is applied, in 1970-2018 it did not contribute much to the decoupling of industrial emissions from GDP. This is expected to change in the future (Figure 11.1).

FOOTNOTE [2] In 2020 this factor played on the reduction side as the Covid-19 crisis led to a global decline in demand for basic materials, respective energy use and emissions by 3-5 % (IEA 2020a).

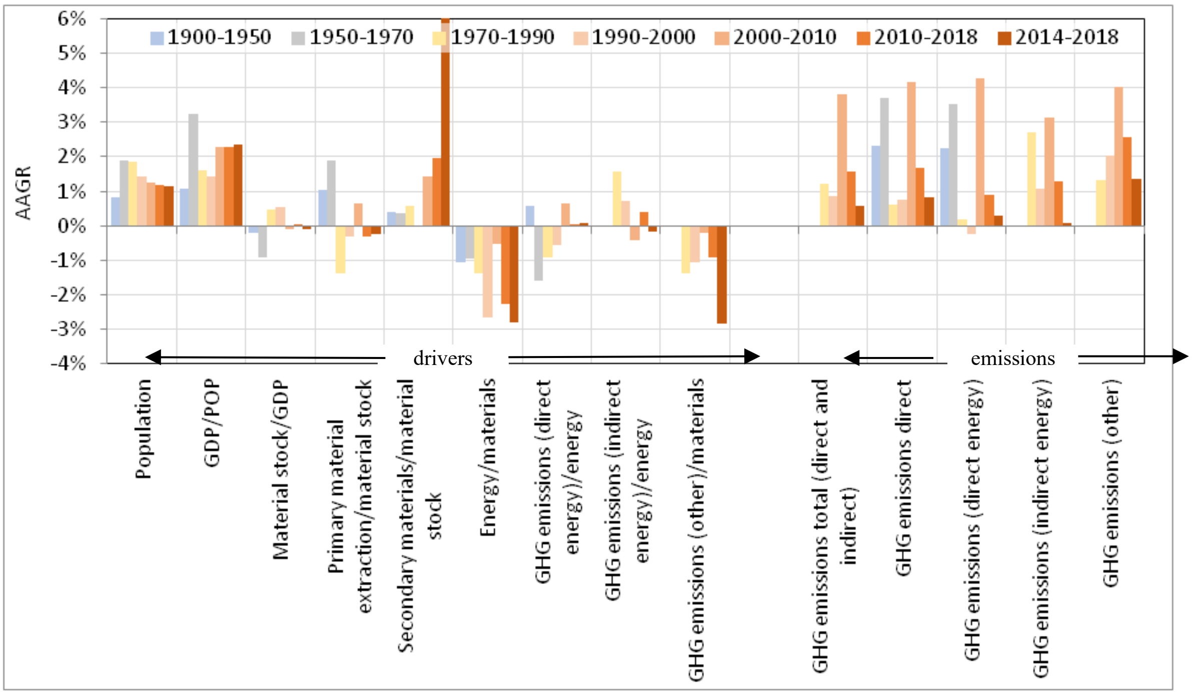


Figure 11.2 Average annual growth rates of industrial sector GHG emissions and drivers. 1900–2018. Before 1970, GHG emission (other) is limited to that from cement production. Primary material extraction excludes fuels and biomass.

Sources: Population before 1950 and GDP before 1960: Maddison Project (2018); Population from 1950 to 1970: UN (2015); GDP for 1960–1970: World Bank (2019a); Data on material stock, extraction, and use of secondary materials: Wiedenhofer et al. (2019); Data on material extraction: UNEP and IRP (2020); Industrial energy use for 1900–1970: IIASA (2018), for 1971–2018: IEA (2019a); Data on emissions for 1900–1970: CDIAC (2017); Population, GDP, and industrial GHG emissions for 1970–2018: Crippa et al. (2019).

Material efficiency analysis mostly uses material intensity or productivity indicators, which compare material extraction or consumption with GDP (IRP 2019, 2020). Those indicators are functions of **material stock intensity of GDP** (tons per dollar) and material intensity of building and operating accumulated in-use stock. Coupling services or GDP with the built stock allows for a better evaluation of demand for primary basic materials (Wiedenhofer et al. 2019; Cao et al. 2017; Liu et al. 2013; Liu and Müller 2013; Müller et al. 2011; Pauliuk et al. 2013a; IRP 2020; Krausmann et al. 2020). Since 1970 material stock growth slightly exceeded that of GDP and there was no decoupling,[3] so in Kaya-like identities material stock may effectively replace GDP. There are different methods to estimate the former (see reviews in (Pauliuk et al. 2015, 2019; Wiedenhofer et al. 2019)), the results of which are presented for major basic materials with some geographical resolution (Pauliuk et al. 2013a; Liu and Müller 2013) or globally (UNEP 2011; Wiedenhofer et al. 2019; Krausmann et al. 2018; Pauliuk et al. 2019; Geyer et al. 2017a). For a subset of materials, such as solid wood, paper, plastics, iron/steel, aluminium, copper, other metals/minerals, concrete, asphalt, bricks, aggregate, and glass, total in-use stock escalated from 36 Gt back in 1900 to 186 Gt in 1970, 572 Gt in 2000, and 960 Gt in 2015, and by 2018 it approached 1,060 Gt, or 136 t per capita (Wiedenhofer et al. 2019; Krausmann et al. 2018, 2020). Therefore, in 1900–2018, the stock grew up 30-fold, which is strongly coupled with GDP growth (37-fold). As the UK experience shows, material stock intensity of GDP may ultimately decline after

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FOOTNOTE [3] This conclusion is also valid separately for developed countries, rest of the world, and for China, when adjusted GDP for this country is used (Krausmann et al. 2020).

services fully dominate GDP, and this allows for material productivity improvements to achieve absolute reduction in material use, as stock expansion slows down (Streeck et al. 2020). While the composition of basic materials within the stock of manufactured capital was evolving significantly, overall stock use associated with a unit of GDP has been evolving over the last half-century in a quite narrow range of 7.7–8.6 t per USD1000 (2017 PPP) showing neither signs of decoupling from GDP, nor saturation yet. Mineral building materials (concrete, asphalt, bricks, aggregate, and glass) dominate the stock volume by mass (94.6% of the whole stock, with the share of concrete alone standing at 43.5%), followed by metals (3.5%) and solid wood (1.4%). The largest part of in-use stock of our 'cementing societies' Cao et al. (2017) is constituted by concrete: about 417 Gt in 2015 Krausmann et al. (2018) extrapolated to 478 Gt (65 tons per capita) in 2018, which contains about 88 Gt of cement. Plastics stock reached 2.5-3.2 Gt (Geyer et al. 2017a; Wiedenhofer et al. 2019) and aluminium stock exceeded 1 Gt (World Aluminum Institute 2019), or just 0.1% of the total. In sharp contrast to global energy intensity, which has more than halved since 1900 (Bashmakov 2019), in 2018 material stock intensity (in-use stock of manufactured capital per GDP) was only 15% below the 1900 level, but 15% above the 1970 level. In-use stock per capita has been growing even faster, than GDP per capita, since 2000 (

Figure 11.3). Total in-use stock of manufactured capital growth rate was 3.8% in 1971-2000 and 3.5% in 2000-2018, or 32-35 Gt·yr^{-1}, to which concrete and aggregates contributed 88%. Recent demand for stock building materials was 51-54 Gt·yr^{-1}, to which recycled materials recently contributed only about 10% of material input leaving the rest (about 46-49 Gt·yr^{-1}) to virgin stock-building inputs, which after accounting for processing waste and short-lived products (over 8 Gt·yr^{-1}) scale up to 54-58 Gt·yr^{-1} of primary stock building resources extraction (Krausmann et al. 2017, 2018; UNEP and IRP 2020), and thus articulating a circularity gap and challenge.

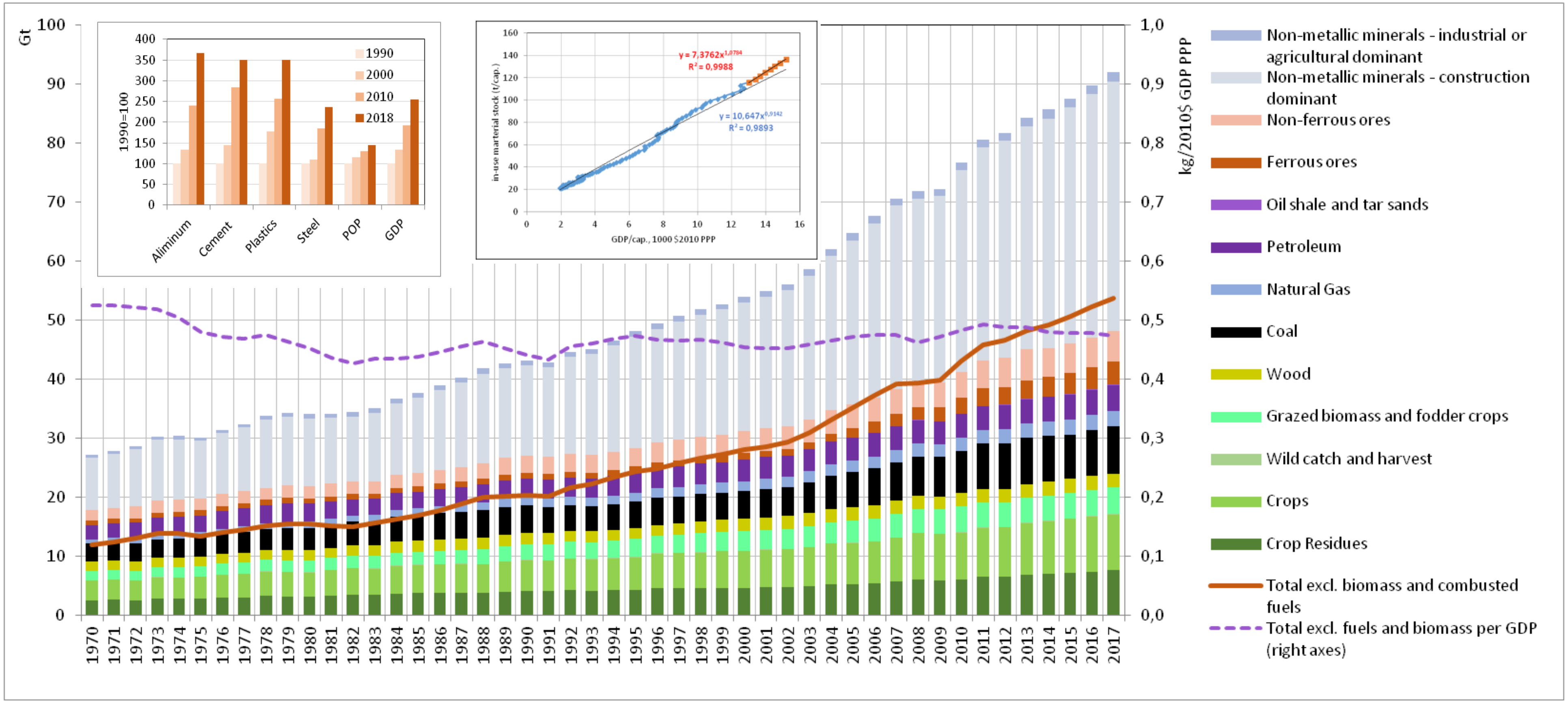

Figure 11.3 Raw natural materials extraction since 1970. In windows: production of major basic materials in 1990-2018 and in-use stock per capita versus income level 1900–2018 (brown dots are for 2000–2018).

Source: Developed based on (Wiedenhofer et al. 2019; Maddison Project 2018; UNEP and IRP 2020; IEA 2020b).

Total **extraction of all basic materials** (including biomass and fuels) in 2017 reached 92 Gt·yr^{-1}, which is 13 times above the 1900 level (Figure 11.3).[4] When recycled resources are added, total material inputs exceed 100 Gt (Circle Economy 2020). In Equation 11.1 *MRP* represents only material inputs to the stock, excluding dissipative use –biomass (food and feed) and fuels. Total extraction of stock building materials (metal ores and non-metallic minerals) in 2017 reached 55 Gt Gt·yr^{-1}.[5] In 1970-2018, it grew 4.3-fold and the ratio of *MRP* to accumulated in-use capital has nearly been constant since 1990 along with ratio to GDP (Figure 11.3).

End-of-life waste from accumulated stocks and (re)-manufacturing and construction waste is assessed at 16 Gt·yr^{-1} in 2014 and can be extrapolated in 2018 to 19 Gt·yr^{-1} (Wiedenhofer et al. 2019; Krausmann et al. 2018), or 1.8% from stock of manufactured capital. Less than 6 Gt·yr^{-1} was recycled and used to build the stock (about 10% of inputs). While the circularity gap is still large, and limited circularity was designed into accumulated stocks,[6] **material recycling and down cycling** mitigated some GHG emissions by replacing energy intensive virgin materials.[7] When the stock saturates, in closed material loops the end-of-life materials waste has to be equal to material input, and primary production therefore has to be equal to end-of-life waste multiplied by unity minus recycling rate. When the latter grows, as the liner metabolism is replaced with the circular one, the share of primary materials production in total material input declines.

Recycling rates for metals are higher, than for other materials: old scrap ratio for 13 metals is over 50%, and stays in the range 25–50% for another ten, but even for metals recycling flows fail to match the required inputs (UNEP 2011). Globally, despite overall recycling rates being at 85%, old scrap ratio for steel production was down from 29% in 2000 to 20% in 2012 and stabilised close to that level afterwards[8] as a result of limited scrap availability (IEA 2020c). For aluminium, the share of scrap-based production grew from 15% in 1962 to 33% in 2004 and stabilised at this level till 2018; while the share of old scrap grew from 1.5% in 1962 to nearly 20% in 2018 (World Aluminium Institute 2020a). The global recycling rate for plastics is only 9% (Geyer et al. 2017a) and that for paper progressed from 34% in 1990 to 44% in 2000 and to over 50% in 2014-2018 (IEA 2020b).

The limited impacts of material efficiency factors on industrial GHG emission trends reflect the lack of integration of material efficiency in energy and climate policies which partly results from the inadequacy of monitored indicators to inform policy debates and set targets;[9] lack of high-level political focus and industrial lobbing; uncoordinated policy across institutions and sequential nature of decision-making along supply chains; carbon pricing policy lock-in with upstream sectors failing to pass carbon costs on to downstream sectors (due to compensation mechanisms to reduce carbon leakage) and so have no incentives to exploit such options as light-weighting, reusing, re-manufacturing, recycling, diverting scrap, extending product lives, using products more intensely, improving process yields, and substituting materials (Skelton and Allwood 2017; Gonzalez Hernandez et al. 2018c). Poor progress with material efficiency is part of the reason why industrial GHG emissions are perceived and falsely designated as 'hard to abate', and many industrial low-carbon trajectories to 2050 leave up to 40% of emissions in place (Material Economics, 2019). The importance of this factor activation rises as in-use

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FOOTNOTE [4] IRP (2020) estimate 2017 material extraction at 94 Gt·yr^{-1}.

FOOTNOTE [5] It approaches 60 Gt·yr^{-1} after construction and furniture wood and feedstock fuels are added (Krausmann et al. 2018; Wiedenhofer et al. 2019; UNEP and IRP 2020).

FOOTNOTE [6] According to Circle Economy (2020) 8.6 Gt·year^{-1} or 8.6% of total inputs for all resources.

FOOTNOTE [7] Environmental impacts of secondary materials are an order of magnitude lower compared to primary materials (OECD 2019a).

FOOTNOTE [8]According to Gielen et al. (2020), the post-consumer scrap contributed about a quarter of finished steel production in 2015.

FOOTNOTE [9] Significant progress with data and indicators was reached in recent years with the development of several global coverage material flows datasets (IRP 2019).

material stock is expected to scale up by a factor of 2.2-2.7 to reach 2215-2720 Gt by 2050 (Krausmann et al. 2020) and material extraction is expected to rise to 140-200 Gt·yr^{-1} by 2060 (IRP 2020; OECD 2019a) providing unsustainable pressure on climate and environment and calling for fundamental improvements in material productivity.

Since 2014, global **industrial energy use** average annual growth rates (AAGR) was limited to 0.7% compared to 2.8% in 2000-2014, following new policies and trends demonstrated by China[10] (IEA 2020b,d). Whatever metric is applied, industry (mining, quarrying, manufacturing and construction) driven mostly by material production, dominates global energy consumption. About two fifths of energy produced globally goes to industry, directly or indirectly. Direct energy use (including energy used in coal transformation) accounts for 30% of final energy consumption. When supplemented by non-energy use, the share for the post AR5 period (2015-2018) stands on average at nearly 40% of final energy consumption, and at 28.4% of primary energy use.[11] With an account of indirect energy use for the generation of power and centralised heat to be consumed in industry, the latter scales up to 37%. Industrial energy use may be split by: material production and extraction (including coal transformation) –51% on average for 2015-2018, non-energy use (mostly chemical feedstock) –22%, and other energy use (equipment, machinery, food and tobacco, textile, leather etc.) –27%. Energy use for material production and feedstock[12] makes about three quarters (73%) of industrial energy consumption and is responsible for 96% of its increment in 2015-2018 (based on (IEA 2020d)).

For over a century **industrial energy efficiency** sustainably mitigates GHG emissions (Figure 11.2). In 2010-2018, global aggregated industrial energy efficiency indicators were progressing slower, than in 1971-2000. They accelerated since 2013 driven by high energy prices. IEA (2020a,b) estimates, that industrial energy intensity (per value added using market exchange rates) globally dropped by 16% in 2010-2018 after its 6% growth in 2000-2010, resulting in 2010-2018 decline by only 10.5%. COVID crisis slowed down energy intensity improvements by shifting industrial output towards more energy intensive basic materials (IEA 2020e). A physical indicator –industrial energy use per ton of extracted materials (ores and building materials as a proxy for materials going through the whole production chain to final products)—helps avoid the exchange rate selection problem and shows that energy intensity scaled down faster –by 20% in 2000-2018[13] and by 16% in 2010-2018, accelerated to 2.7%·yr^{-1} in 2014-2018 matching the values observed back in 1990-2000.

Specific energy consumption (SEC) per ton of iron and steel, chemicals and cement production in 2019 were about 20% below the 2000 level (IEA 2020a,b). This progress is driven by moving towards best available technologies (BATs) for each technology through new and highly efficient production facilities in China, India and elsewhere, and by the contribution from recycled scrap metals, paper and cardboard. SECs for the production of materials decline, as technologies improve, but the decline slows down, as potential thermodynamic (theoretical) limits for existing technologies are approached (Gutowski et al. 2013; Crijns-Graus et al. 2020). Thus, the energy saving effect of switching to secondary used material comes to the forefront, as SECs for many basic primary materials approach BATs. This highlights the need to push towards circular economy, materials efficiency, reduced demand, and fundamental process changes (e.g. towards electricity and hydrogen based steel making). Improved recycling rates allow for a substantial reduction in energy use along the whole production chain –material extraction, production, and assembling –which is in great excess of energy used for

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[10] FOOTNOTE China contributed three forth of global industrial energy use increment in 2000-2014. Since 2014 China's share in global industrial energy use slowly declines reaching about a third in 2018 (IEA 2020d).

[11] FOOTNOTE This fits well 28.8% average for 1900-2018 with a slow trend to decline by 0.01% yr^{-1}in response to the growing share of services in global GDP, around which about 60-years-long cycles can be observed.

[12] FOOTNOTE Mapping global flows of fuel feedstock allows for better tailoring downstream mitigation options for chemical products (Levi and Cullen 2018).

[13] FOOTNOTE In the EU, energy efficiency index improved by 20% in 2000-2018 (Odyssee-Mure 2020).

collection, separation, treatment, and scrap recycling minus energy used for scrap landfilling. IEA (2019b) estimates, that by increasing the recycling content of fabricated metals average SECs for steel and aluminium may by halved by 2060. Material efficiency coupled with energy efficiency can deliver much greater savings than energy efficiency alone. Gonzalez Hernandez et al. (2018c) stress that presently about half of steel or aluminium are scrapped in production or oversized for targeted services. They show that resource efficiency expressed in exergy as a single metric for both material and energy efficiency for global iron and steel sector is only 33%, while secondary steelmaking is about twice more efficient (66%), than ore-based production (29%). While shifting globally in ore-based production from average to BATs can save 6.4 $EJ \cdot yr^{-1}$, the saving potential of shifting to secondary steelmaking is 8 $EJ \cdot yr^{-1}$, and limited mostly by scrap availability and steel quality requirements.

11.2.2 New trends in emissions

GHG emission attributed to the industrial sector (see Chapter 2) in 2018 originates from industrial combustion (6.8 $GtCO_2$-eq direct and about 6 Gt indirect from electricity and heat generation[14]); industrial processes (4.9 $GtCO_2$-eq) and products use (0.2 Gt), as well as from waste (1.5 Gt) (Figure 11.4a-b). Overall industrial GHG emission amounts to 13.4 Gt for direct emissions (with 10 $GtCO_2$ contribution, Figure 11.4c) and 19.3 Gt if indirect emissions are added,[15] putting industry (23%, direct emissions) second after the energy sector in total GHG emissions and lifting it to the leading position since 2004 after indirect emissions are allocated (33% in 2018)[16]. The corresponding shares for 1990-2000 were 18-19% and 27-28% (Crippa et al. 2019). Therefore, in the 21st century there is a clear trend to the growing role and dominance of industry in overall GHG emissions.

In 2000-2010, direct industrial emissions have been growing faster (4.1% yr^{-1}), than in any other sector (see chapter 2), mostly due to the dynamics shown by basic materials extraction and production. After 2010, industry was third after AFOLU and transport in AAGR –1.7%, and after 2014 AAGR went below 1% or back to the 1970-2000 levels. In 1970–1990, industrial direct combustion-related emissions were growing modestly, and in 1990–2000 even switched to a slowly declining trend, steadily losing their share in overall industrial emissions. Electrification was the major driver behind both indirect and total industrial emissions in those years. This quiet evolution was interrupted in the beginning of the 21st century, when emissions increased by 52–72% depending on the metric applied (the fastest growth ever seen). Then all direct emissions froze temporarily in 2014–2016, partly in the wake of the financial crisis, and returned back to the growing trajectory in 2017–2018 (Figure 11.4a).

The industrial emission structure by sources becomes more diversified. Direct fuel combustion related emission dominates (51% in 2018), but its role is diminishing, as it comes only second (28%) after industrial processes (63%) in the contribution to 2010-2018 incremental direct emissions, followed by waste (9%, Figure 11.4 and Table 11.1)

). Therefore, to stop emission growth and to switch to zero carbon pathway more mitigation efforts are to be focused on industrial processes, products use and waste decarbonisation along with the transition to low carbon electrification (IRP 2020).

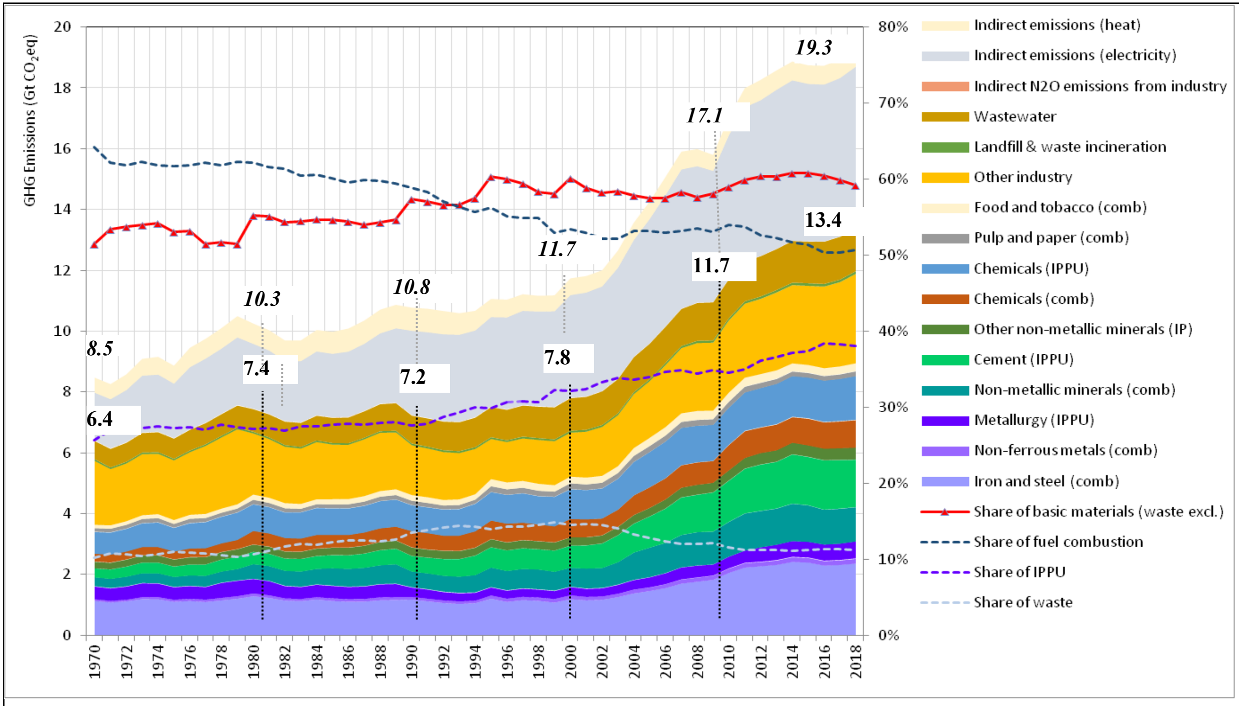
FOOTNOTE [14] Indirect emissions are assessed based on EDGAR database (Crippa et al. 2019) at about 6 Gt. IEA database also reports 6 Gt of CO_2 for 2018 (IEA 2020f).

FOOTNOTE [15] Based on Crippa et al. (2019). In 2018, CO_2 only emissions were 10 $GtCO_2$, or 9.8 Gt if waste and products use were excluded, which exceeds the IEA (2020d) estimate for 2018 of 8.5 Gt. Transportation of basic materials further contributes around 1 billion tCO_2-eq. to GHG emissions.

FOOTNOTE [16] According to IEA (2020f) industry fuel combustion CO_2 only emissions contributed 24% to total combustion emissions, but combined with indirect emission it accounted for 43% in 2018.

Basic materials production dominates direct industrial GHG emissions (about 60%, waste excluded)[17]. In a zero-carbon power world, with industry lagging behind in the decarbonisation of high-temperature processes and feedstock, it may replace energy sector as generator of indirect emissions embodied in capital stock[18]. According to Circle Economy (2020) and IRP (2020), GHG emissions embodied in buildings and infrastructure, machinery and transport equipment exceed 50% of their present carbon footprint.

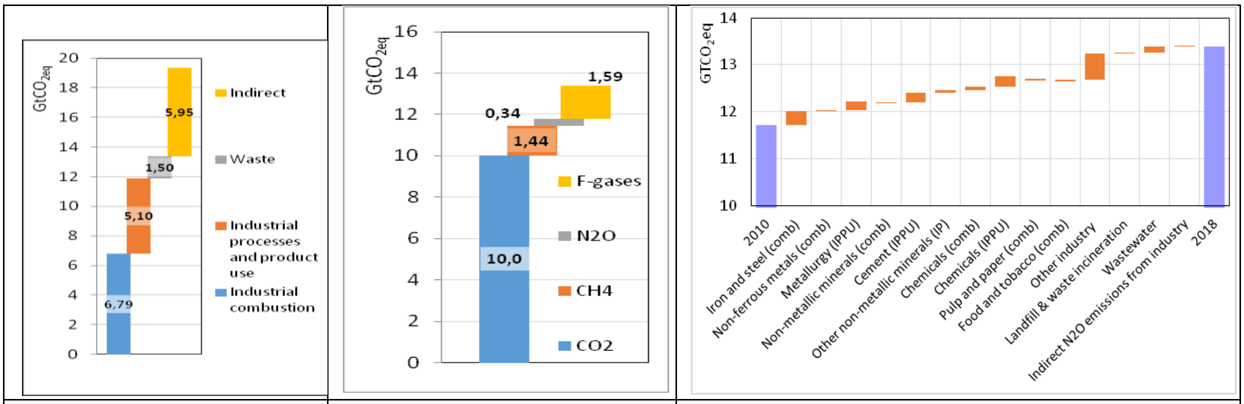
In 1970-2000, direct GHG emissions per unit of energy showed steady decline interrupted by noticeable growth in 2001–2018 driven by fast expansion of steel and cement production in China (Figure 11.2). Non-energy related GHG emissions per unit of extracted materials decline continuously, as the share of not carbon intensive building materials (aggregates and sand) grows.



(a) industrial emissions by sources (left scale) and emissions structure (right scale). Comb – indicates direct emissions from fuels combustion. IPPU – indicates emissions from industrial processes and product use. Indirect emissions from electricity and heat generation are shown on the top. Shares on the right are shown for direct emissions.

FOOTNOTE [17] IRP (2020) estimates the global carbon footprints of basic materials at 11.5 G CO_2-eq (4.8 Gt for metals, 4.4 for non-metallic minerals, 1.5 for plastics and rubber, and 0.9 for wood products), which is about 65% of direct and indirect industrial emissions (waste excluded). About 9 Gt are embedded in such stock building elements as buildings and infrastructure, vehicles, machinery and electronics. Hertwich et al. (2019) assess 2015 cradle-to-gate GHG emissions from the production of materials at 11.4 Gt CO_2-eq. Crippa et al. (2019) and IEA (2020a) assess scope 2 (direct and indirect industrial emissions) in 2018 at 10.7 Gt.

FOOTNOTE [18] According to IRP (2020), of 11.5 Gt CO_2-eq 2015 global materials GHG footprint about 5 Gt were embodied in buildings and infrastructure, and nearly 3 Gt in machinery, vehicles, and electronics.



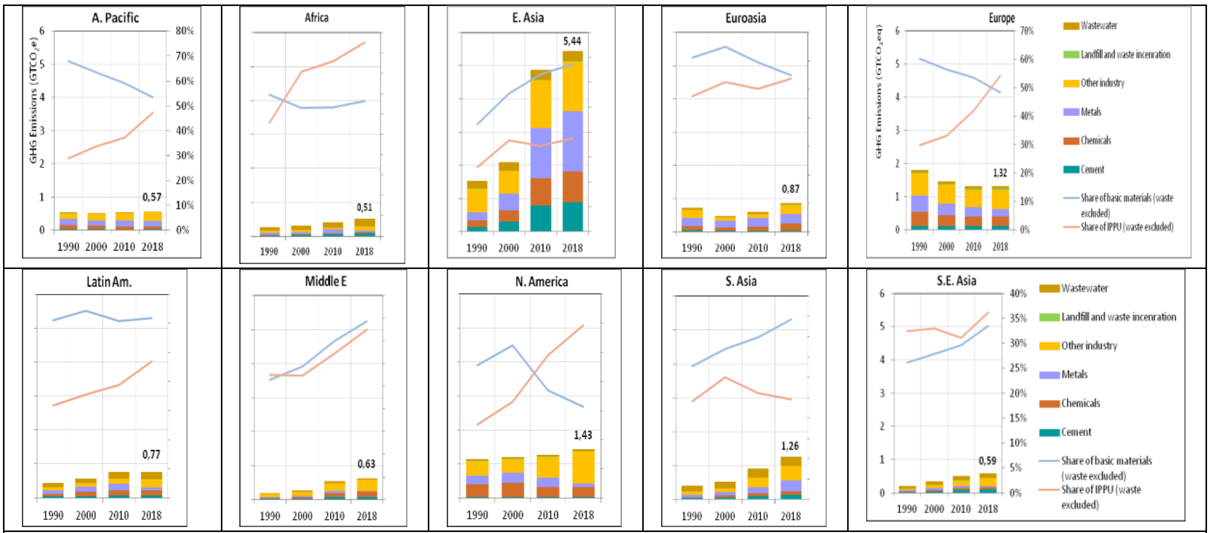
(b) 2018 emissions split by major sources

(c) 2018 direct emissions split by GHGs

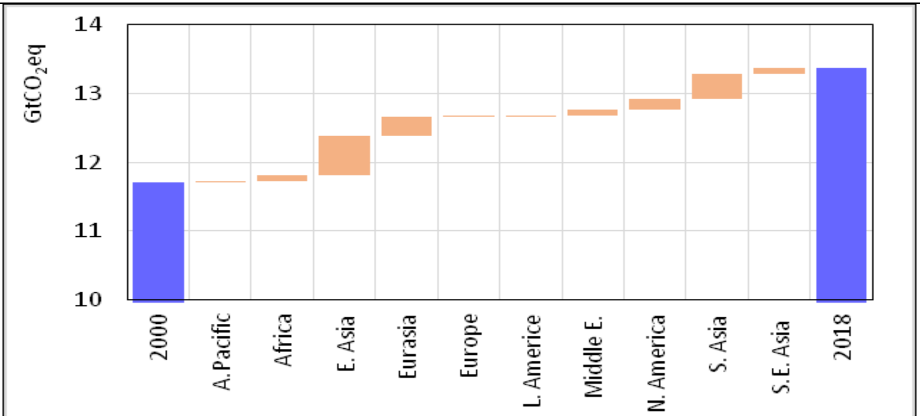
(d) 2010-2018 increments of GHG emissions by sources (direct emissions only)

Figure 11.4 Industrial sector direct global GHG emissions. 1970-2018

Source: Calculated based on emissions data from (Crippa et al. 2019). Indirect emissions were assessed using (IEA 2020b)



(a) industrial emissions by sources (right axes) and share of materials and emissions from industrial processes and product use in overall industrial emissions



(b) 2010-2018 increments of industrial GHG emissions in 10 world regions (direct emissions only)

Figure 11.5 Industrial sector GHG emissions (direct only) in 10 world regions. 1990-2018

Source: Calculated based on emissions data from (Crippa et al. 2019). Indirect emissions were assessed using (IEA 2020b).

Table 11.1 Dynamics and structure of industrial GHG emissions

		Average annual growth rates				Share in total industrial sector emissions					2018 emissions $GtCO_2$-eq
		1971-1990	1991-2000	2000-2010	2011-2018	1970	1990	2000	2010	2018	
Direct CO_2 emissions form fuel combustion	***Mining (excl. fuels), manufacturing industries and construction***	***0.17%***	***-0.23%***	***4.25%***	***0.89%***	***48.0%***	***39.6%***	***35.4%***	***36.5%***	***34.7%***	***6,711***
	Iron and steel	0.19%	0.15%	5.64%	1.75%	13.1%	10.8%	10.1%	11.8%	12.0%	2,328
	Chemical and petrochemical	3.51%	1.20%	3.18%	0.98%	3.2%	5.1%	5.2%	4.9%	4.7%	903
	Non-ferrous metals	2.07%	3.09%	1.10%	1.19%	0.7%	0.9%	1.1%	0.8%	0.8%	157
	Non-metallic minerals	2.92%	1.87%	6.24%	-0.25%	3.5%	4.9%	5.4%	6.7%	5.8%	1,128
	Paper, pulp and printing	0.71%	2.79%	0.21%	-2.98%	1.5%	1.3%	1.6%	1.1%	0.8%	151
	Food and tobacco	2.64%	1.60%	3.04%	-1.41%	1.3%	1.8%	1.9%	1.7%	1.4%	267
	Other	-1.43%	-2.88%	3.17%	1.29%	24.7%	14.8%	10.1%	9.4%	9.2%	1,778
Indirect emissions - electricity		***2.77%***	***1.95%***	***3.60%***	***1.25%***	***19.2%***	***26.4%***	***29.3%***	***28.4%***	***27.7%***	***5,369***
Indirect emissions – heat		***1.31%***	***-3.41%***	***2.18%***	***1.34%***	***5.5%***	***5.7%***	***3.7%***	***3.1%***	***3.0%***	***589***
Industrial processes CO_2	Total	***1.47%***	***2.11%***	***4.89%***	***2.29%***	***11.5%***	***12.3%***	***13.9%***	***15.2%***	***16.1%***	***3,121***
	Non-metallic minerals	2.22%	2.36%	5.44%	1.84%	6.0%	7.4%	8.6%	9.9%	10.1%	1,959
	Chemical and petrochemical	4.55%	2.36%	3.62%	1.69%	1.6%	3.1%	3.6%	3.5%	3.5%	682
	Metallurgy	-3.04%	0.27%	5.18%	5.71%	3.8%	1.6%	1.5%	1.7%	2.4%	460
	Other	1.56%	2.24%	-1.06%	-0.12%	0.2%	0.2%	0.2%	0.1%	0.1%	20
Industrial product use GHG		***-0.23%***	***-0.50%***	***-1.03%***	***0.22%***	***2.8%***	***2.1%***	***1.9%***	***1.1%***	***1.0%***	***199***
Other non-CO_2 GHG		***0.51%***	***3.65%***	***6.18%***	***4.47%***	***5.3%***	***4.7%***	***6.2%***	***7.7%***	***9.6%***	***1,857***
Waste GHG		***2.03%***	***1.51%***	***1.82%***	***1.32%***	***7.7%***	***9.1%***	***9.7%***	***7.9%***	***7.8%***	***1,504***
Total GHG		**1.14%**	**0.90%**	**3.92%**	**1.55%**	**100.0%**	**100.0%**	**100.0%**	**100.0%**	**100.0%**	**19,351**

Source: Calculated based on Crippa et al. (2019) and IEA (2020f).

11.2.3 Industrial development patterns and supply chains (regional)

A new rising wave in industrial emissions after 2000 is clearly associated with China's and other non-OECD Asian countries' economic growth, which dominated both absolute and incremental emissions (Figure 11.5a). Recent (2010–2018) trends show that regional contributions to additional emissions are distributed more evenly, while a large part still comes from Asian countries, where both rates of economic growth and the share of industrial emissions much exceed the global average. All other regions also contributed to total industrial GHG emissions. Structural shifts towards emissions from industrial processes and products use are common for all regions except S. Asia (Figure 11.5a).

Economic development. Regional differences in emission trends are determined by the differences observed in economic development, trade and supply chain patterns. The major source of industrial emissions is production of energy intensive materials, such as iron and steel, chemical and petrochemicals, non-ferrous metals and non-metallic products. Steel and cement are key inputs to urbanisation and infrastructure development (buildings and infrastructure are responsible for about three fourths of the steel stock). Cao et al. (2017) mapped countries by four progressive stages in cement stock per capita S-shape evolution as a function of income and urbanisation: initial stage for developing countries with a low level and slow linear growth; take-off stage with accelerated growth; slowdown stage; and finally shrinking stage (represented by just a few countries with very high incomes exceeding 40 thousand 2010USDint per capita) and urbanisation levels above 80%. This logic may be generalised to other materials from which in-use stock is built.[19] While globally cement in-use stock is about 12 tons per capita, in developed countries it is 15-30 tons per capita, but order of magnitude lower in developing states with high per capita escalation rates (Cao et al. 2017). When stocks for some materials saturate – per capita stock peaks – the 'scrap age' is coming (Pauliuk et al. 2013a). Steel in-use stock has already saturated in advanced economies at 14±2 ton per capita due to completed urbanisation and infrastructure developments and switch towards services-dominated economy. This saturation level is 3–4 times that of the present global average, which is below 4 tons per capita (Pauliuk et al. 2013a; UNEP 2011; Wiedenhofer et al. 2019). China is entering the maturing stage of steel and cement consumption resulting in a moderate projection of additional demand followed by expected industrial emissions peaking in the next 10-15 years (Bleischwitz et al. 2018; Zhou et al. 2013; Wu et al. 2019; OECD 2019a). But many developing countries are still urbanising, and the growing need for infrastructure services results in additional demand for steel and cement. Materials intensity of the global economy is projected by OECD (2019a) to decline at 1.3% yr^{-1} till 2060, driven by improving resource efficiency and the switch to circular economy, but with a projected tripling of global GDP it means a doubling of projected materials use (OECD 2019a). Under the business-as-usual scenario, India's demand for steel may more than quadruple over the next 30 years (de la Rue du Can et al. 2019; Dhar et al. 2020) – and that still only represents two-thirds of China's current steel production.

International trade and supply chain. In Equation 11.1 the share of allocated emissions (*Dm*) equals to unity when territorial emission is considered, and to the ratio of domestically used materials to total material production for the consumption-based emission accounting. Tracking consumption-based emissions allows it to detect "carbon leakage" and provides additional insights in the global effectiveness of national climate policies. Carbon emissions embodied in international trade are estimated to account for 20-30% of global carbon emissions (Meng et al. 2018; OECD.Stat 2019) and are the reason for different emissions patterns of OECD versus non-OECD countries (Chapter 2).

Based on OECD.Stat (2019) datasets, 2015 CO_2 emissions embodied in internationally traded industrial products (manufacturing and mining excluding fuels) by all countries are assessed at 3 $GtCO_2$, or 30% of direct CO_2 emission in the industrial sector as reported by Crippa et al. (2019). OECD countries

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[19] FOOTNOTE (Bleischwitz et al. 2018) use a similar approach to study material saturation effects for apparent consumption and stocks per capita for steel, cement, aluminum, and copper.

collectively have reduced territorial emissions (shares of basic materials in direct emissions in those regions decline, Figure 11.5b), but demonstrated no progress in reducing outsourced emissions embedded in imported industrial products (Arto and Dietzenbacher 2014; OECD.Stat 2019). Accounting for net carbon emissions embodied in international trade of only industrial products (1283 $MtCO_2$ in 2015) escalates direct OECD industrial CO_2 emissions (1333 $MtCO_2$ of energy-related and 502 $MtCO_2$ of industrial processes) 1.7 fold, 2.3-fold for the US, 1.5-fold for the EU, and more than triples it for the UK, while cut *Dm* by a third for China and Russia (OECD.Stat 2019; IEA 2020f). In most OECD economies, the amount of CO_2 embodied in net import from non-OECD countries is equal to, or even greater than, the size of their Paris 2030 emissions reduction commitments. In the UK, parliament Committee on Energy and Climate Change requested that a consumption-based inventory be complementarily used to assess the effectiveness of domestic climate policy in delivering absolute global emissions reductions (Barrett et al. 2013; UKCCC 2019a). It should be noted that the other side of the coin is that exports from countries with lower product emissions lead to overall less emissions than if production took place in high emission countries, and this is an effect less recognised and accounted for. In the coming years availability of large low-cost renewable electricity potential may become a new driver for relocation of such carbon intensive industries as steel production (Gielen et al. 2020).

11.3 Technological developments and options

The following overview of technical developments and mitigation options is organised in six partly 2 overlapping, interdependent and equally important strategies: (i) demand management, (ii) materials efficiency, (iii) circular economy and industrial waste, (iv) energy efficiency, (v) electrification and 4 fuel switching, and (vi) CCUS, feedstock and biogenic carbon.

11.3.1 Demand for materials

Demand for materials is a key driver of energy consumption and CO_2 emissions in the industrial sector. Rapid growth in material demand over the last quarter century has seen demand for key energy-intensive materials increase 2.5– to 3.5–fold (see Figure 11.6), with growth linked to, and often exceeding, population growth and economic development. The International Energy Agency explains, “as economies develop, urbanise, consume more goods and build up their infrastructure, material demand per capita tends to increase considerably. Once industrialised, an economy’s material demand may level off and perhaps even begin to decline.” (IEA 2019b).

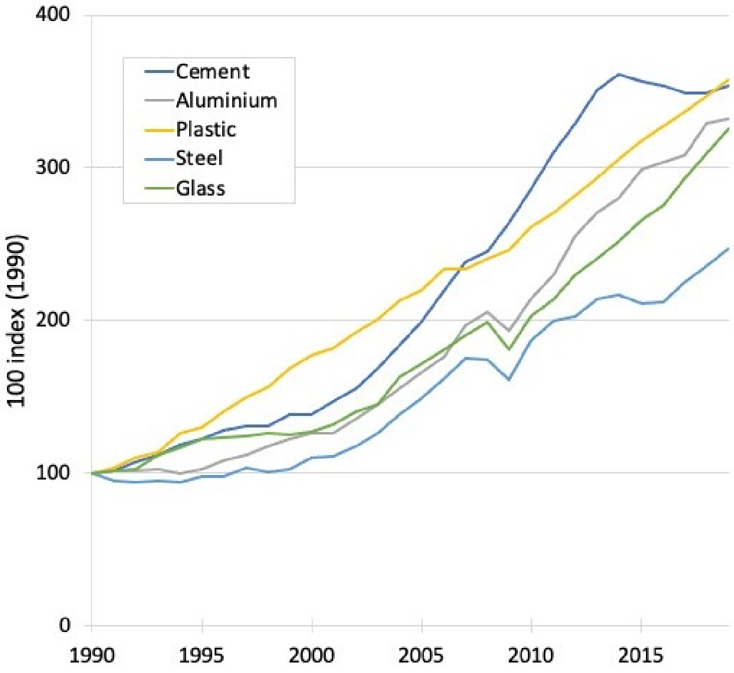


Figure 11.6 Growth in global demand for key materials, 1990 – 2017 Notes: Based on global production of key materials, shown indexed to 1990 levels (=100). Steel refers to crude steel production. Aluminium refers to primary aluminium production. Plastics refers to a subset of thermoplastic resin production. Sources: Geyer et al. (2017a); Levi and Cullen (2018); Wiedenhofer et al. (2019); World Steel Association (2020a); USGS (2020).

The Kaya-like identity presented earlier in the chapter (Equation 11.1) suggests that material demand can be decoupled from population and economic development by two means: (i) reducing the accumulated material stock (MStock) used to deliver material services; and, (ii) reducing the material (MPR +MSE) required to maintain material stocks (MStock). Such material demand reduction strategies are linked upstream to material efficiency strategies (the delivery of goods and services with less material demand, and thus energy and emissions) and to demand reduction behaviours, through concepts such as sufficiency, sustainable consumption and social practice theory (Spangenberg and Lorek 2019).

Recent modelling suggests that per capita material stocks saturate (level off) in developed countries and decouple from GDP. Pauliuk et al. (2013b) demonstrated this saturation effect in an analysis of in-use steel stocks in 200 countries, showing that per capita steel in stocks in countries with a long industrial history (e.g. US, UK, Germany) had saturated levels between 11 and 16 tons. More recently, Bleischwitz et al. (2018) confirmed the occurrence of a saturation effect for four materials (steel, cement, aluminium and copper) in four industrialised countries (Germany, Japan, UK, USA) together with China. These findings have led to the revision of some material demand forecasts, which previously had been based solely on population and economic trends.

The saturation effect for material stocks is critical for managing material demand in **developed countries**. Materials are required to meet demand for the creation of new stocks and the maintenance of existing stocks (Gutowski et al. 2017). Once saturation is attained, stocks can be maintained with much reduced demand for materials. Saturation allows material efficiency strategies (such as lightweight design, longer lifetimes, and more intense use) to reduce the required per capita level of material stocks, and material circularity strategies (closing material loops through remanufacture, reuse, recycling) to lessen the energy and carbon impacts required to maintain the material stock. However, it should be noted that some materials still show little evidence of saturation (i.e. plastics (Box 11.2), aluminium) and the changes in developed country infrastructures to meet carbon neutral aspirations will create new demand for materials.

For **developing countries**, who are still far from saturation levels, strong growth for new products and building infrastructure capacity may still drive global material demand. However, there is an expectation that economic development can be achieved at lower per capita material stock levels, based on the careful deployment of material efficiency and circularity by design principles.

Materials demand can also be influenced through urban planning, building codes and related socio-cultural norms that shape the overall demand for square meters per capita of floor space, mobility and transport infrastructures (Chapter 5).

11.3.2 Material efficiency

ME—the delivery of goods and services with less material—is increasingly seen as an important strategy for reducing GHG emissions in industry (IEA 2017a). Options to improve ME exist at every stage in the life-cycle of materials and products, as shown in Figure 11.7. This includes: designing products which are lighter, more optimised, last longer and with circular principles built-in; pushing manufacturing and fabrication process to use materials and energy more efficiently and recover material wastes; increasing the capacity, intensity of use, and lifetimes of product in use; improving the recovery of materials at end-of-life, through improved remanufacturing, reuse and recycling processes (for more specific example see (Allwood et al. 2012; Hertwich et al. 2019; Rissman et al. 2020; Scott et al. 2019)).

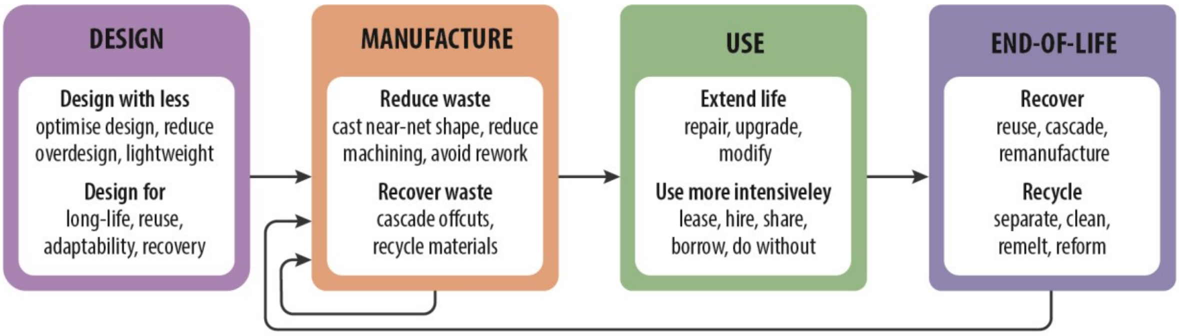


Figure 11.7 Material efficiency strategies across the value chain
Source: derived from strategies in Allwood et al. (2012).

ME provides plentiful options to reduce emissions, yet because interventions are dispersed across supply chains and span many different stakeholders, this makes assessing mitigation potentials and costs more challenging. For this reason, ME interventions have traditionally been under-represented in climate change scenario modelling and integrated assessment models (IAMs) (Grubler et al. 2018; Allwood 2018). However, two advances in the modelling of materials flows have underpinned the recent emergence of ME options being included in climate scenario modelling.

Firstly, over many years, the academic community has built up detailed global material flow maps of the processing steps involved in making energy-intensive materials. Some prominent recent examples include: steel (Gonzalez Hernandez et al. 2018c), pulp and paper (Van Ewijk et al. 2018), petrochemicals (Levi and Cullen 2018). In addition, material flow maps at the regional and sectoral levels have flourished: for example: steel (Serrenho et al. 2016) and cement (Shanks et al. 2019) in the UK; automotive sheet-metal (Horton et al. 2019); steel powder applications (Azevedo et al. 2018). The detailed and transparent physical mapping of material supply chains, in this manner, enables ME interventions to be traced back to where emissions are released, and allows these options to be compared against decarbonisation and traditional energy efficiency measures (Levi and Cullen 2018). For example, a recent analysis by Hertwich et al. (2019) makes the link between ME strategies and reducing greenhouse gas emissions in buildings, vehicles and electronics, while Gonzalez Hernandez et al. (2018a) examines leveraging ME as a climate strategy in EU policy. Research to explore the combined analysis of materials and energy, using exergy analysis (for steel, (Gonzalez Hernandez et al. 2018c)) allows promising comparisons across industrial sectors.

Secondly, many ME interventions result in immediate emissions savings (short-term), for example, light-weighting products, re-using today's product components, and improving manufacturing yields. Yet, for other ME actions emissions savings are delayed temporally (long-term). For example, designing a product for future reuse, or with a longer-life, only reaps emissions savings at the end of the product life, when emissions for a replacement product are avoided. Many durable products have long life-times (cars >10 years, buildings >40 years) which requires dynamic modelling of material stocks, over time, to enable these actions to be included in scenario modelling activities. Consequently, much effort has been invested recently to model material stocks in use, to estimate their lifetimes, and anticipate the future waste and replenishment materials to maintain existing stocks and grow the material stock base. Dynamic material models have been applied to material and product sectors, at the country and global level. These include, for example: vehicles stocks in the UK (Serrenho et al. 2017; Craglia and Cullen 2020) and in China (Liu et al. 2020); buildings stocks in the UK (Serrenho et al. 2019), China (Hong et al. 2016; Cao et al. 2018, 2019) and the European Union (Sandberg et al. 2016); electronic equipment in Switzerland (Thiébaud et al. 2017); specific material stocks, such as cement (Cao et al. 2017, 2020), construction materials (Sverdrup et al. 2017), plastics (Geyer et al. 2017a), copper (Daehn et al. 2017), and all metals (Elshkaki et al. 2018); all materials in China (Jiang et al. 2019), Switzerland (Heeren and Hellweg 2019) and the world (Krausmann et al. 2017).

These two advances in the knowledge base have allowed the initial inclusion of some ME strategies in energy and climate change scenario models. The International Energy Agency (IEA) first created a ME scenario (MES) in 2015, with an estimated 17% reduction in industrial energy demand in 2040 (IEA 2015). WEO (IEA 2019c) report includes a dedicated sub-chapter with calculations explicitly on industrial material efficiency. They also include ME options in their modelling frameworks and reporting, for example for petrochemicals (IEA 2018a), and in the Material Efficiency in Clean Energy Transitions report (IEA 2019b). In Grubler et al. (2018) 1.5 degree Low Energy Demand (LED) scenario, global material output decreases by 20% from today, by 2050, with one-third due to dematerialisation, and two-thirds due to ME, resulting in significant emissions savings. Material Economics analysis of Industrial Transformation 2050 (Material Economics 2019), found that resource efficiency and circular economy measures (i.e. ME) could almost halve the 530 $MtCO_2$ yr^{-1}emitted by the basic materials sectors in the EU by 2050. Finally, the Emissions gap report, UNEP (2019) includes an assessment of potential material efficiency savings in residential buildings and cars.

Clearly, more work is required to fully integrate ME strategies into mainstream climate change models and future scenarios. Efforts are focused on endogenising ME strategies within climate change modelling, assessing the synergies and trade-offs which exist between energy efficiency and ME interventions, and building up data for the assessment emissions saved and the cost of mitigation from real ME actions. This requires analysts to work in cross-disciplinary teams and to engage with stakeholders from across the full breadth of material supply chains.

11.3.3 Circular economy and industrial waste

Circular economy (CE) is one effective approach to mitigate industrial GHG emissions and has been widely promoted worldwide since IPCC fourth assessment report (AR4). From an industrial point of view, CE focuses on closing the loop for materials and energy flows by incorporating policies and strategies for more efficient energy, materials and water consumption, while emitting minimal waste to the environment (Geng et al. 2013). Moving away from the current linear mode of production (synthetically referred to as an "extract-produce-use-discard" model), the CE promotes the design of durable goods that can be easily repaired, with components that can be reused, remanufactured, and recycled (Wiebe et al. 2019). For example, in case of aluminium, approaching BAT-level SECs in total primary production can reduce energy consumption by only 4%, while continuous switching to production through using recycled materials can deliver 7 times as much energy savings. Improved recycling rate allows for a substantial reduction in energy use along the whole production chain — material extraction, production, and assembling, — which is in great excess of energy spent for collection, separation, treatment, and scrap recycling minus energy needed for scrap landfilling. IEA (2019b) estimates that by increasing the recycling content of fabricated metals average SECs for steel and aluminium may be by halved by 2060. Material efficiency can deliver greater savings, than energy efficiency alone. Gonzalez Hernandez et al. (2018b) stress that presently about half of steel or aluminium are scrapped or oversized for targeted services. They show that resource efficiency expressed in exergy as a single metric for both material and energy efficiency for global iron and steel sector is only 33%, while secondary steelmaking is about twice more efficient (66%), than ore-based production (29%). While shifting globally in ore-based production from average to BATs can save 6.4 $EJ\cdot yr^{-1}$, the saving potential of shifting to secondary steelmaking is 8 $EJ\cdot yr^{-1}$, and limited mostly by steel quality requirements (Gonzalez Hernandez et al. 2018c).

This systemic approach has been conducted at different levels, namely, at micro-level (within a single company, such as process integration and cleaner production), meso-level (between three or more companies, such as industrial symbiosis or eco-industrial parks) and macro-level (cross-sectoral cooperation, such as urban symbiosis or regional eco-industrial network). Each level requires different tools and policies, such as CE-oriented incentive and tax policies (macro-level), and eco-designing regulations (micro-level). The following paragraphs detail such efforts.

Micro-level: Increasing firms began to implement the concept of CE, particularly those multi-national companies, since they believe that multiple benefits can be obtained from CE efforts. Typical CE tools and policies at this level include cleaner production, eco-design, environmental labelling, process synthesis, and green procurement. For instance, as one of the leading chemical giants in the world, the Dow Chemical company has incorporated CE into their industrial practices. Their eco-innovations include the design of more recyclable and degradable plastics, a differentiated and market-driven portfolio of resins, films and adhesives that deliver a total package that is more sustainable, cost-efficient and capable of meeting new packaging and plastics preferences, paving roads by recycled plastics, etc (Hoffman 2016). Similarly, Renault, the French automaker, ensures that 85% of a new vehicle is recyclable when it reaches end of life and that 36% of that new vehicle's mass is made from recyclable materials (Wiebe et al. 2019). In addition, with the increasing public awareness, companies should observe consumer concerns for the environment and encourage the application of secondary raw materials while limiting the use of certain hazardous substances in the production of new materials.

Meso-level: Industrial park has been long implemented since it first appeared in Manchester, UK, at the end of 19th century, and it has been a kind of popular style in industrialised countries for maximizing energy and material efficiency, which has also merit for carbon reduction, as stated well in AR5. It reduces costs of infrastructure and utilities, by concentrating industrial activities in planned areas and complementary industries and services provided by industrial parks can entail diversified effects on the surrounding region and finally stimulate regional development (Geng and Zhao 2009). This is crucial for those small and medium enterprises (SMEs) because they often lack access to information and lack funds to sophisticated technologies. Typical CE tools and policies at this level include green supply chains and industrial symbiosis. Anchor companies within one industrial park can always green its supply chain to their upstream suppliers and downstream users and seek potential byproducts exchanges with their neighbouring firms. A common platform for sharing information and enhancing communication among industrial stakeholders through the application of information and telecommunication technologies would be helpful to facilitate the creation of industrial symbiosis. The main benefit of industrial symbiosis is the overall reduction of both virgin materials and final wastes. In particular, it can facilitate those small and medium sized enterprises to improve their growth and competitiveness. From climate perspective, this indicates significant industrial emission mitigation since the extraction, processing of virgin materials and the final disposal of industrial wastes are more energy-intensive. Due to these advantages, eco-industrial parks have been actively promoted, especially in East Asian countries, such as China, Japan and South Korea, where national indicators and governance exist (Geng et al. 2019). For instance, the successful implementation of industrial symbiosis at Dalian Economic and Technological Development Zone has achieved significant co-benefits, including GHG emission reduction, economic and social benefits and improved ecosystem functions (Liu et al. 2018). Another case at Ulsan industrial park, South Korea, uncovers that 60,522 tonne CO_2 equivalent were avoided annually through industrial symbiosis between two companies (Kim et al. 2018a). At the national level, other CE measures, such as increase the share of natural gas consumption, reduce the GHG emission factor of electricity grid, and improve the average efficiency of industrial coal-fired boilers, can significantly contribute to GHG emission reduction at industrial park level. The case of China shows a great potential of implementing these measures, estimating 111 million tonne CO_2 equivalent will be reduced in 213 national-level industrial parks in 2030 (Guo et al. 2018). As such, South Korea's national eco-industrial park project has reduced over 4.7 million tonne CO_2 equivalent through their industrial symbiosis efforts (Park et al. 2019). Unfortunately, such efforts are still few in many other countries and should be further promoted. Moreover, waste prevention as the top of the so-called "waste hierarchy" can be promoted on the meso-level for specific materials or product systems. For instance, the European Environment Agency published a report on plastic waste prevention approaches in all EU 28 member states (Wilts and Bakas 2019).

Macro-level: The macro- level uses both micro- and meso-level tools within a broader policy strategy, addressing the specific challenge of CE as a cross-cutting policy (Wilts et al. 2016). More synergy opportunities exist beyond the boundary of one industrial park. This indicates the necessity of scaling up industrial symbiosis to urban symbiosis. Urban symbiosis is defined as the use of byproducts (waste) from cities as alternative raw materials for energy sources for industrial operations (Sun et al. 2017). It is based on the synergistic opportunity arising from the geographic proximity through the transfer of physical sources (waste materials) for environmental and economic benefits. Japan is the first country to promote urban symbiosis. For instance, the Kawasaki urban symbiosis efforts can save over 114,000 tons of CO_2 emission annually (Geng et al. 2010). Another simulation study indicates that Shanghai (the largest Chinese city) has a potential of saving CO_2 emission up to 16.8 Mt through recycling all the available wastes (Dong et al. 2018). As such, the simulation of urban energy symbiosis networks in Ulsan, South Korea indicates that 243,396 ton yr^{-1} CO_2^{-1} emission and 48 million US Dollar yr^{-1} fuel cost can be saved (Kim et al. 2018b). Moreover, Wiebe et al. (2019) estimates that the adoption of the CE can lead to a significantly lower global material extraction compared to a baseline. Their global results range from a decrease of about 27% in metal extraction to 8% in fossil fuel extraction and use, 8% in forestry products, and about 7% in non-metallic minerals, indicating significant climate change benefits. Macro-perspective calculation on the circulation of iron in Japan's future society shows that CO_2 emissions from steel sector will be reduced by 56% assuming the CE, in which includes the following assumptions: the amount recovered from social stock is the same as the amount of inflow, and all scrap was used domestically, and the export of steel products is halved (LCS 2018a). The medals for the 2020 Tokyo Olympic game were made 100% from recycled metal resources produced using wastes from urban life (Tokyo 2020 2019). The key challenge is to go beyond ensuring proper waste management but to set metrics, targets and incentives to preserve the incorporated value in specific waste streams. Estimations for Germany have shown that despite recycling rates of 64% for all solid waste streams, these activities only lead to a resource reduction of only 18% (Steger et al. 2019). Realising these potentials will require the development of comprehensive strategies that include changed consumer behaviour, such as food waste prevention. In general, the identification of the most appropriate CE method for different countries requires understanding and information exchange on background conditions, local policies and a myriad of other factors influencing material flows from the local up to the global level (Tapia Carlos et al. 2019).

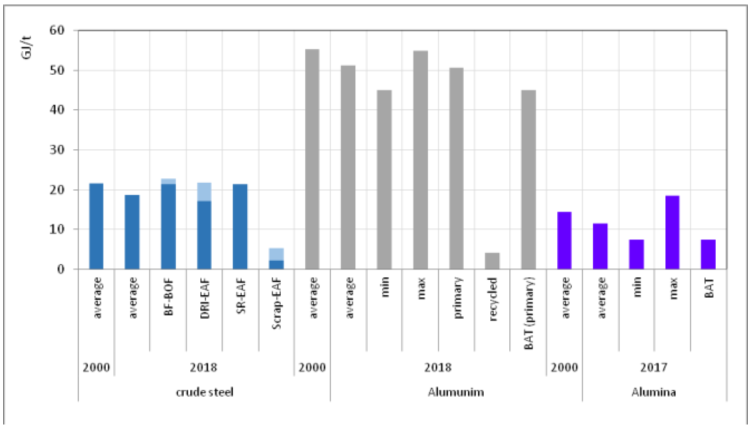
11.3.4 Energy Efficiency

Industrial energy efficiency is a number one mitigating option in the shorter term (Figure 11.1 and Figure 11.8). There are multiple directions to improve industrial energy efficiency (Tanaka 2011) including:

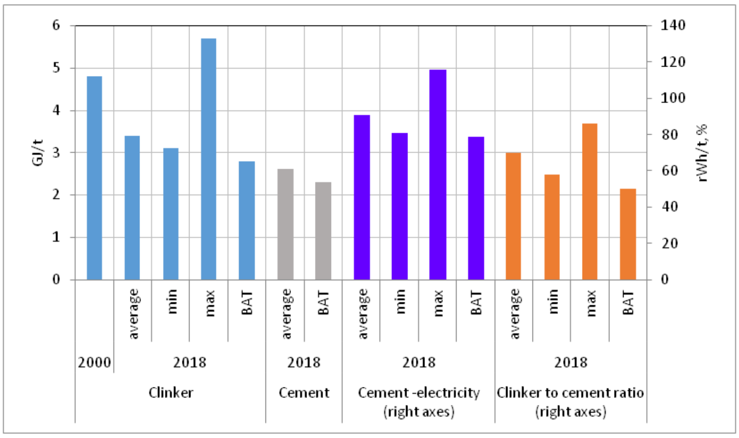
- empowering energy management with fourth industrial revolution (Industry 4.0) tools - artificial intelligence, learning tools and data-driven analytics, IoT and cloud computing, cyber physical systems and others - to identify and effectively reduce energy and materials use while increasing process productivity, reducing product reject rates and increasing materials yields;
- upgrading existing equipment close to BATs parameters to compensate for natural and moral efficiency degradation;
- substituting obsolete equipment, process lines and facilities by new state of art technologies;
- streamlining new processes allowing for eliminating some processing steps via using innovative production concepts, re-using and recycling products and materials;

There are two parallel processes: technological improvement efforts leading to relatively slow energy efficiency BATs progress and faster one - SECs decline towards BATs. Both slows as potential thermodynamic (theoretical) minimums are approached (Gutowski et al. 2013). The energy saving effect of switching to secondary materials comes to the forefront, as SECs for many basic primary materials approach BATs (Figure 11.8). This highlights the need to push towards circular economy (see Section 11.3.3). In recent years, the focus has been on effective spreading of the BAT technologies applying policies for worldwide diffusion of energy-saving technologies (see Section 11.6).

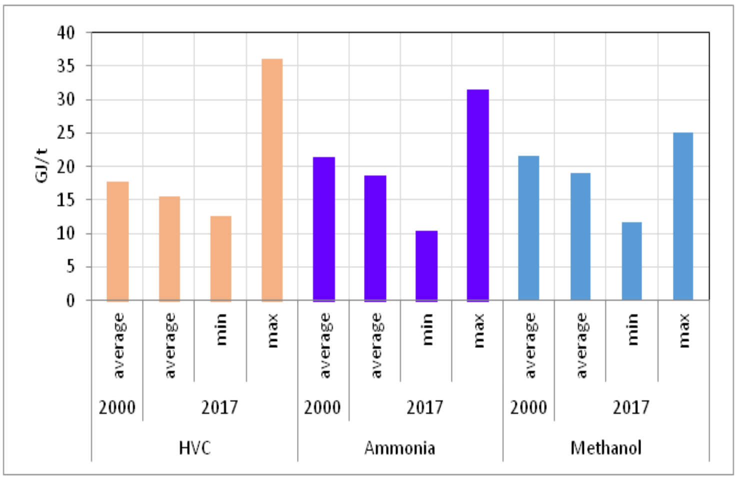
Below, we focused mainly on the technical progress and on new options that are reflected in the literature since AR5.



metals



cement and clinker



Chemicals

Figure 11.8 Energy efficiency potential for approaching BATs Energy accounting is based on final energy use. For steel both final (IEA) and primary (Worldsteel) accounting are shown. Sectoral boundaries for steel are defined as defined in (IEA 2020c).

Sources: Calculated based on UNIDO (2010); IEA (2020c, 2018b,c, 2019b, 2020b)); Hasanbeigi et al. (2012); IEA (2017a); Saygin et al. (2011); WBCSD (2016); Crijns-Graus et al. (2020); Napp et al. (2014); Moya and Pardo (2013); World Aluminium Institute (2020b).

11.3.4.1 Energy efficiency improvement at heat use

While about 10% of global GHG emissions originate from fuels combustion to produce high temperature heat for basic material production processes (ICEF 2019) limited efforts were given to decarbonise heat production. Appearing options include hydrogen, which can substitute fossil fuels in many industrial heating systems with moderate need for processes change, particularly in chemical synthesis, but with costs escalating while moving from grey to blue and then to green hydrogen. Biomass option to deliver net low-carbon high temperature heat is limited by low energy density, competing demands from other sectors and resulting biomass scarcity. Electrification can be used to supply low carbon high-temperature heat using resistance heating, microwaves, induction and electric arc furnaces with challenges to redesign technological process .and needs to sizable escalation of power infrastructure development even if demand-side management is possible (ICEF 2019). Limitations on low carbon high temperature heat leaves to CCU and to CCS an important role in decarbonisation of industry (Figure 11.8), but also pushes efficiency of high temperature heat use at the forefront.

Large potential is still left in using wasted heat. NEDO (2019) applies 'Reduce, Reuse, and Recycle' concept for its mobilisation. For *Reduce*, stands thermal insulation, cost effective technological innovations allowing reducing BATs for industrial heat using processes. Porous type insulators have been developed (Fukushima and Yoshizawa 2016), with thermal conductivity only halt that of is traditionally appalled heat-resistant bricks under conditions of high compressive strength capable withstanding process applications. For *Reuse,* as it was identified for EU industry, potential for utilisation of waste heat at temperatures below 200°C is 50% of the applicable demand (about 300 TWh yr^{-1}). Wasted heat at a temperatures 200-500°C has potential to cover up to 25% of demand, and that of 500°C and more – over 25% (Papapetrou et al. 2018). Survey conducted in Japan shows that 9% of the input energy is lost with wasted heat, of which heat below 199°C is responsible for 68% and that below 149°C - 29% (NEDO 2019). New development is high temperature heat pump: an installation has been developed raising temperature up to 165°C at coefficient of performance (COP) 3.5 by recovering heat from unused hot water (35-65°C) (Arpagaus et al. 2018). Study for the cement, glass and iron industries in China shows, that current technology enables only 7-13% of waste heat use for power generation. Potentially 40-57% of waste heat with temperatures above 150°C can be used for power generation via heat recovery (Lu et al. 2016).

In the dyeing industry, about 10% of the energy used in tenters (dryers) has been saved via applying the preheating system which use waste heat (Rakib et al. 2017). Exhaust heat utilisation may become unstable due to process intensity variations, but latent heat storage technology and intermediate storage units may be applied for the solution (Jiménez-Arreola et al. 2018). The development of thermoelectric conversion materials that produce power from unused heat and energy harvested from higher temperature environment is also progressing aiming for possible application in industrial processes (Ohta et al. 2018; Jood et al. 2018; Lv et al. 2018; Gayner and Kar 2016). It is expected to be applied to sensors power supply, a niche that uses large amounts of low voltage energy (Champier 2017).

11.3.4.2 Smart energy management

Industry 4.0 concept reflects the computerisation of industrial process and robotised manufacturing, real time sensors data collection and communication, the diffusion of network technologies and the Internet of Things (IoT), digital manufacturing and cyber-physical systems, the predictive control techniques – all integrated into smart energy management systems and into smart factories capable to rapidly adapt and respond to external changes. Smart energy systems allow for optimisation of innovative technologies, energy demand respond balancing of energy supply and demand including that on real time pricing, and product quality management, predicting and reducing the idle time for both men and robots (Legorburu and Smith 2018; Pusnik et al. 2016; Ferrero et al. 2020). Smart manufacturing (SM) systems that integrate manufacturing intelligence in real time through the entire production operation have not been yet widely spread in the industry. Examples have been demonstrated and integrated in real operation in the electrical appliance assembly industry (Yoshimoto 2016). Using the process controls and automation system together allows at low costs optimise processes, and improve productivity (Edgar and Pistikopoulos 2018). Big data analysis of factors affecting boiler efficiency, operation optimisation and load forecasting have shown that it leads to energy saving of 9% (Wang et al. 2017). In recent years smart energy systems have been integrated in real time operation in the electrical appliance assembly industry (Yoshimoto 2016).

11.3.4.3 Technology progress

The pneumatic strain energy accumulator is a so-called cascaded use of pressure when using compressed air, and experiments have shown an increase in system energy efficiency of 32-78% (Cummins et al. 2017). Due to the integration of the system for recovering latent heat and sensible heat from the exhaust of the dryer for milk powder production and the adsorbent system for dehumidification, energy consumption can be reduced from 8.4 MJ to 4.9 MJ heat per kg milk powder (Moejes et al. 2018).

Although the importance of applying energy efficient technologies is widely recognised, there are multiple barriers preventing companies from to using them including considerations of reliability, initial investment costs, and detailed specifications. To meet those optimisation tools have been developed (for example for choice of compressor in petrochemical industry (Taylan et al. 2016). The development of thermoelectric conversion materials that generate power from wasted heat for application in industrial processes is also progressing (Ohta et al. 2018; Jood et al. 2018; Lv et al. 2018; Gayner and Kar 2016). It is expected to serve as power supply to multiple sensors, which use low voltage, but large amounts of energy (Champier 2017).

11.3.5 Electrification and fuel switching

The principle of electrification and fuel switching as a GHG mitigation strategy is that industries, to the extent possible, transition their end uses of energy from a high GHG intensity energy carrier to lower or zero intensity carrier, including both its direct and indirect production and end-use GHG emissions. In general and non-exclusively, this implies a transition from coal (~0.09 $tCO_2 \cdot GJ^{-1}$ on combustion), refined petroleum products (~0.07 $tCO_2 \cdot GJ^{-1}$), and natural gas (~0.05 $tCO_2 \cdot GJ^{-1}$) to biofuels, electricity, hydrogen, ammonia, or net-zero synthetic hydrocarbon fuels. Switching to these energy carriers is not necessarily lower emitting, however; how they are made matters.

Switching from higher GHG intensity fossil fuel forms to lower intensity forms (e.g. from coal to natural gas for electricity production) has already been observed to reduce direct combustion CO_2 emissions in several jurisdictions. There are significant debates about the net effect of upstream production and fugitive emissions, but observers have noted that in the case of US power generation it would take a leakage rate of ~2.7% from natural gas production to undo the direct fuel switch effects, and the value is likely higher in most cases (Alvarez et al. 2012; Hausfather 2015). Coal mine methane emissions are also estimated to be substantially higher than previously estimated (Kholod et al. 2020). However, with the goal of net-zero emissions in mind to the meet the Paris Agreement targets and the potential for infrastructure lock-in effects (Tong et al. 2019), purely fossil fuel switching is a limited and potentially dangerous strategy, but can avoid significant cumulative emissions if utilised carefully until alternatives are available in sufficient volumes.

Biofuels come in many forms, including ones that are nearly identical to fossil fuels but sourced from biogenic sources. Biomethane, biomethanol, bioethanol are all commercially made today using fermentation and anaerobic digestion techniques, and are mostly “drop-in” compatible with fossil fuel equivalents. In principle they cycle carbon in and out of the atmosphere, but their life cycle GHG intensities are typically not GHG neutral due to land use changes, soil carbon depletion, fertiliser use, and other dynamics (Hepburn et al. 2019), and are highly case specific. Most commercial biofuel feedstocks come from agricultural (e.g. corn) and food waste sources, and the feedstock is limited; to meet higher levels of biomass use a transition to using higher cellulose feedstocks like straw, switchgrass and wood waste must be fully commercialised and deployed. Significant efforts have been made to make ethanol from cellulosic biomass, which promises much higher quantities, lower costs, and lower intensities, but commercialisation efforts, with a few exceptions, have largely not succeeded (Padella et al. 2019). The IEA estimates, however, that up to 20% of today's fossil methane use, including by industry, could be met with biomethane (IEA 2020g) by 2040, using a mixture of feedstocks and production techniques. Biofuels combustion will also be critical for producing negative emissions when combined with carbon capture and storage, i.e. BECCS. Finally, it should be noted that biofuel combustion can potentially have substantial negative local air quality effects, with implications for SDG 3 and SDG 7.

Switching to electricity for end-uses, or "direct electrification", is a highly discussed strategy for net-zero industrial decarbonisation (Palm et al. 2015; Lechtenböhmer et al. 2016; Davis et al. 2018; Bataille et al. 2018a; Material Economics 2019; UKCCC 2019b; Åhman et al. 2016; Axelson et al. 2018). Electricity is a flexible energy carrier that can be made from many forms of primary energy, with high potential process improvements in terms of quality and process controllability, digitisability, and no direct local air pollutants (Deason et al. 2018; McMillan et al. 2016; Jadun et al. 2017; Mai et al. 2018). The net GHG effect of electrification is contingent on how the electricity is made, and because total output increases can be expected for full effect it should be made with a very low or zero primary energy source (i.e. <50 grams $CO_2 \cdot kWh^{-1}$: e.g. hydroelectricity, nuclear energy, wind, solar photovoltaics, or fossil fuels with 95+% carbon capture and storage (Bruckner et al. 2014)). This has strong implications for the electricity sector and its generation mix; despite their low and falling costs, above 60-70% progressively higher mixes of variable wind and solar on a given grid will require a firm low or zero emissions generation source (e.g. nuclear energy, fossil fuels with CCS, hydrogen fuel cells or turbines) to moderate costs (Jenkins et al. 2018; Sepulveda et al. 2018). Regions that may be slower to reduce the GHG intensity of their electricity production will likely need to consider more aggressive use of other measures.

Steam boilers, curing, drying and small-scale process heating are readily electrifiable with appropriate fossil fuel to electricity price ratios (accounting for capital costs and efficiencies), and direct induction and infrared heating are available for higher temperature needs. These practices are uncommon outside regions with ample hydroelectric power due to the currently relatively low cost of coal, natural gas and heating oil, and especially when there is no carbon combustion cost. Industrial electrification is most viable in the near term with: minimal retrofitting and rebuild in processes; with relatively low energy costs; where the degree of process complexity and process integration is more limited and extensive process re-engineering would not be required; where combined heat and power is not used; where induction heating technologies are viable; and where process heating temperatures are lower (Deason et al. 2018).

The long-term potential for full process electrification is a very sector by sector and process by process phenomenon, with differing energy and capacity needs, load profiles, stock turnover, capacity for demand response, and characteristics of decision makers. It has been explored in: raw and recycled steel (Fischedick et al. 2014; Vogl et al. 2018a); ammonia (Philibert and IEA 2017; Bazzanella and Ausfelder 2017); and chemicals (Palm et al. 2015; Bazzanella and Ausfelder 2017). While most chemical production of feedstock chemicals (e.g. H_2, NH_3, CO, CH_3OH, C_2H_4, C_2H_6, C_2H_5OH) is done thermo-catalytically today, it is feasible to use direct electrocatalytic production, by itself or in combination with utilisation of previously captured carbon sources if a fossil fuel feedstock is used, or well-known bio-catalytic (e.g. fermentation) and thermo-catalytic processes (De Luna et al. 2019; Kätelhön et al. 2019; Bazzanella and Ausfelder 2017). It may even be commercially possible to electrify cement sintering and calcination through plasma or microwave options (Material Economics 2019).

Electrification requires more supply. For example, 75 TWh of electricity was used by steel in the EU in 2015 (1000 TWh total was used by industry) (Material Economics 2019), varying between their new process, circularity and CCUS scenarios, projects increased demand to 355 (+373%), 214 (+185%) and 238 (+217%) TWh. These values are consistent with (Vogl et al. 2018a), which projects a tripling of electricity demand in the German or Swedish steel industries if hydrogen direct reduced iron in the electric arc furnace (DRI EAFs) replace BF-BOFs. Material Economics (2019) was conservative with its use of electricity in chemical production, making preferential use of biofeedstocks and some CCUS, and electricity demand rose from 118 TWh to 510, 395 and 413 TWh. Bazzanella and Ausfelder (2017) exploring deeper reductions from the chemical sector using more electrochemistry, projected scenarios with higher electricity demands of 960–4900 TWh (140% of projected available clean electricity at the time) with maximal electricity use. In counterpoint, however, with revised wind capabilities and costs, the IEA (2019d) Offshore Wind Outlook indicates that ten times current EU electricity use could be produced if necessary. Greater use of electro-catalytic versus thermo-catalytic chemistry, as projected by De Luna et al. (2019), could greatly reduce these electricity needs, but the technology readiness levels are currently low. Finally, the UKCCC (2019b), which focussed primarily on CCS for industry in its "Further Ambition" scenario (the UK currently consumes about 300 TWh), in its supplementary "Further electrification" scenario projects an additional about 300 TWh for general electrolysis needs and another about 200 TWh for synthetic fuel production.

While it has been demonstrated that almost any end use can be directly electrified, instantaneous thermal loads for blast furnace-basic oxygen furnace (BF-BOF) steel production, limestone calcination for cement & lime production, and other end-uses where flame front (1000°C–1700°C) temperatures are currently needed, indicate the need for another energy carrier to minimise instantaneous generation and transmission needs. These needs can be met at varying current and potential future costs using: already discussed bioliquids or gases (already discussed), hydrogen, ammonia, or net-zero synthetic hydrocarbons.

Around 70 Mt yr^{-1} of pure hydrogen is produced today on purpose, 76% from natural gas and 23% from coal, resulting in roughly 830 $MtCO_2$ yr^{-1} in 2016-17, or 2.4% of global combustion and process CO_2 missions. Fuels refining (~410$MtCO_2$ yr^{-1}) and production of ammonia (420 $MtCO_2 \cdot yr^{-1}$) largely dominate its uses. Another 45 Mt hydrogen is being produced in mix with other gases, on purpose or as by-products, and used as fuel or to produce methanol or to reduce iron ores in steel making (IEA 2019e). Very low and potentially zero GHG (depending on the energy source) **hydrogen** can be made via: electrolysis of water (Glenk and Reichelstein 2019), also known as "green H_2"; partial oxidation of coal or naphtha or steam/auto methane reforming (SMR/ATR) combined with CCS (Leeson et al. 2017); methane pyrolysis, where the hydrogen and carbon are separated thermally (Ashik et al. 2015; Abbas and Wan Daud 2010), or via biomass gasification. All these processes would in turn need to be run using very low or zero GHG energy carriers for the resulting hydrogen to also be low GHG emissions.

Broadly speaking, **hydrogen** can contribute to a cleaner energy system in two ways. 1) Existing applications of hydrogen (e.g. nitrogen fertiliser production, refinery upgrading) can use hydrogen produced using alternative, cleaner production methods, and from a more diverse set of energy sources. 2) New applications can use low GHG hydrogen as an alternative to current fuels and inputs, or as a complement to the greater use of electricity in these applications. In these cases – for example in transport, heating, industry (e.g. hydrogen direct reduced iron steel production) and electricity – hydrogen can be used in its pure form, or be converted to hydrogen-based fuels, including ammonia, or synthetic net-zero hydrocarbons like methane or methanol (IEA 2019e). The IEA states that **hydrogen** could be used to help integrate more renewables, including by enhancing storage options and "exporting sunshine & wind" from places with abundant resources; decarbonise "hard to abate" sectors, namely steel, chemicals, trucks, ships and planes; and boost energy security by diversifying the fuel mix & providing flexibility to balance grids (Gül 2019).

Ammonia production, made from hydrogen and nitrogen using the Haber-Bosch process, is the largest chemical produced from fossil fuels, being used as feedstock for nitrogen fertilisers and explosives, as well as a cleanser, refrigerant and for other uses. Most ammonia is made today using methane as the hydrogen feedstock and heat source. Ammonia has been historically made using from hydrogen made using electrolysis, and could be again using renewable electricity (Philibert and IEA 2017).

Hydrogen can also be made into **methane, methanol** and other potential net-zero **synthetic hydrocarbons** using methanation, steam reforming and Fischer-Tropsch processes, all of which can provide higher degrees of storable and shippable high temperature energy as necessary using known industrial processes in novel combinations (Bataille et al. 2018a; Davis et al. 2018). Given their carbon content, if they are used as fuels, this carbon will eventually be oxidised and emitted as CO_2 to the atmosphere, making their net carbon emissions, and usability in a net-zero economy, dependent on the carbon source (Hepburn et al. 2019). It could be sourced from biomass sources, i.e. "biogenic" (Hannula 2016), captured from fossil fuels and utilised, or directly captured from the air (Keith et al. 2018), all with widely varying net emissions effects if combusted. See the next section on CCS and CCU, with elaboration on the potential utility of synthetic hydrocarbons.

Box 11.1 Hydrogen in industry

The "hydrogen economy" is a long-touted vision for the energy and transport sectors, and one that has gone through hype-cycles since the energy crises in the 1970s (Melton et al. 2016). The many and different visions of hydrogen futures have mainly been associated with fuel cells in vehicles, small-scale decentralised cogeneration of heat and electricity, and to a certain extent energy storage for electricity (Eames et al. 2006). However, nearly all hydrogen currently produced is used in industry, mainly for hydrotreating in oil refineries, to produce ammonia, and in other chemical processes.

In the context of net zero emissions new visions are beginning to form, ones in which hydrogen has a central role to play in decarbonising industry. Near-term industrial applications for hydrogen includes feeding it into ammonia production for fertilisers, while a more novel application would be in steel-making, as in the HYBRIT demonstration plant in Sweden (projected in 2026). As shown in sections 11.3.5 and 11.3.6 there are many other potential applications of hydrogen, some of which are still relatively unexplored. Hydrogen can be used to produce various fossil-free hydrocarbons for chemical feedstock based on carbon from biogenic sources or direct air capture of CO_2. Such applications could be anaerobic digestion and fermentation where more of the carbon in feedstock may be utilised if boosted with additional hydrogen, thus increasing the yield of biomethane and ethanol (Ericsson 2017; Huang et al. 2020).

The geographical distribution of the potential for "green" hydrogen from electrolysis powered by solar and wind, and "blue" hydrogen from fossil gas with CCS may reshape where heavy industry is located, how value chains are organised, and what gets transported in international shipping. Regions with bountiful solar and wind resources, or methane co-located with CCS geology, may become exporters of hydrogen or hydrogen carriers such as methanol and ammonia, or home to the production of iron and steel, organic platform chemicals, and other energy intensive basic materials. This in turn may generate new trade patterns and needs for bulk transport.

11.3.6 CCS, CCU, carbon sources, feedstocks and fuels

Carbon is an important and highly flexible building block for a wide range of fuels, organic chemicals and materials including methanol, ethanol, olefins, plastics, textiles, and wood and paper products. In this chapter we define CCS as requiring return of CO_2 from combustion or process gases or ambient air to the geosphere for geological time periods, i.e. thousands of years (IEA 2009; IPCC 2005; IEA 2019f; Bruhn et al. 2016). CCU is defined as being where carbon (as CO or CO_2) is captured from one process and reused for another, reducing emissions from the initial process, but is then potentially but not necessarily released to atmosphere in following processes (Bruhn et al. 2016; Tanzer and Ramírez 2019; Detz and van der Zwaan 2019). In both cases the net effect on atmospheric emissions depends on the initial source of the carbon, be it from a fossil fuel, from biomass, or from direct air capture (Hepburn et al. 2019; Cuéllar-Franca and Azapagic 2015) and the duration of storage or use, which can vary from days to millennia.

While CCS and CCU share common capture technologies, what happens to the CO_2 and therefor the strategies that will employ them can be very different. CCS can be described as a linear economy strategy maintaining CO_2 neutrality for fossil CO_2 that passes through the process, with highly varying partially negative emissions if the source is biogenic (Hepburn et al. 2019), and fully negative emissions if the source is air capture, all not considering the energy used to drive the above processes. CCS has been covered in other IPCC publications at length, for example, IPCC (2005), and in most mitigation oriented assessments since. The potentials and costs for CCS in industry vary considerably due to the diversity of industrial processes (Leeson et al. 2017), as well as the volume and purity of different flows of carbon dioxide (Naims 2016). As a general rule it is not possible to capture all the carbon dioxide emissions from an industrial plant, which becomes problematic and potentially expensive in a net-zero environment when these remnant emissions must be offset with land use, biomass or direct air capture followed by CCS, or other negative emissions strategies.

For concentrated CO_2 sources (e.g. from cleaning of wellhead formation gas to make it suitable for the pipeline network, from hydrogen production using steam methane reforming, ethanol fermentation, or from combustion of fossil fuels with oxygen in a nitrogen free environment, i.e. "oxycombustion") CCS is already amenable to commercial oil and gas techniques for acid gas injection at prices of USD10–40 $tCO_2\text{-eq}^{-1}$ sequestered (Wilson et al. 2003; Leeson et al. 2017). Most currently operating CCS facilities take advantage of concentrated CO_2 flows, for example, from formation gas cleaning on the Snoevit and Sleipner platforms in Norway, from syngas production for the Al Reyadah DRI steel plant in Abu Dhabi, and from SMR hydrogen production on the Quest upgrader in Alberta. Unfortunately, concentrated process CO_2 emissions are often exempted from existing cap and trade systems, and these opportunities for CCS have largely gone unexploited. Many existing projects partially owe their existence to the utilisation of the captured CO_2 for enhanced oil recovery, which in many cases counts as both CCS and CCU because of the relatively permanent nature of the CO_2 disposal upon injection (Mac Dowell et al. 2017). There are several industrial CCS strategies and pilot projects working to take advantage of the relative ease of concentrated CO_2 disposal (e.g. LEILAC for limestone calcination process emissions from cement production, HISARNA direct oxycombustion smelting for steel) (Bataille 2019a). An emerging option for storing carbon is methane pyrolysis by which methane is split into hydrogen and solid carbon that may subsequently be stored (Schneider et al. 2020).

There are several post-combustion CCS projects underway globally (IEA 2019f), generally focussed on energy production and processing rather than industry. Their costs are higher but evolving downward - Giannaris et al. (2020) suggest USD47 tCO_2^{-1} for a follow up 90% capture power generation plant based on learnings from the Saskpower Boundary Dam pilot - but crucially these costs are higher than implicit and explicit carbon prices almost everywhere, resulting in limited investment and learning in these technologies. A key challenge with all CCS strategies, however, is building a gathering and transport network for CO_2, especially from dispersed existing sites; hence most pilot are built near

EOR/geological storage sites, and the movement towards industrial clustering in the EU (ACT 2016) and UK (UKCCC 2019b), and as suggested in (IEA 2019e).

In the case of CCU, CO and CO_2 are captured and subsequently converted into valuable products (e.g. building materials, chemicals, synthetic fuels) (Daggash et al. 2018; Kätelhön et al. 2019; Styring et al. 2011; Von Der Assen et al. 2013; Bruhn et al. 2016; Vreys et al. 2019; Artz et al. 2018; Breyer et al. 2019; Brynolf et al. 2018). CCU has been envisioned as part of the "circular economy" but conflicting expectations on CCU and its association or not with CCS leads to different and contested framings (Palm and Nikoleris 2020). The duration of the CO_2 storage in these products varies from days to millennia according to the application, potentially but not necessarily replacing new fossil, biomass or direct air capture feedstocks, before meeting one of several possible fates: permanent burial, decomposition, recycling or combustion, all with differing GHG implications. While the environmental assessment of CCS projects is relatively straight forward, however, this is not the case for CCU technologies. The net GHG mitigation impact of CCU depends on several factors (e.g. the capture rate, the energy requirements, the lifetime of utilisation products, the production route that is substituted, and associated room for improvement along the traditional route) and has to be determined by life cycle cost analysis (e.g. Bruhn et al. (2016); Nocito and Dibenedetto (2020)). For example, steel mill gases containing carbon monoxide and carbon dioxide can be used as feedstock together with hydrogen for producing chemicals. In this way, the carbon originally contained in the coke used in the blast furnace is used again, or cascaded, and emissions reduced but not brought to zero. If only recycled once and then emitted, the maximum reduction is 50% (Tanzer and Ramírez 2019). The logic of using steel mill CO and CO_2 could equally be applied to gasified biomass, however, with a far lower net GHG footprint, likely negative, which CCU fed by fossil fuels cannot be if end-use combustion is involved.

Partly because of the complexity of the life cycle analysis accounting, the literature on CCU is not always consistent in terms of the net GHG impacts of strategies. For example, Artz et al. (2018), focussed not just on GHG mitigation but multi-attribute improvements to chemical processes from reutilisation of CO_2, suggests the largest reduction in the absolute amount of GHGs from CO_2 reutilisation could be achieved by coupling of highly concentrated CO_2 sources with carbon-free hydrogen or electrons from renewable power in so called "Power-to-fuel" scenarios. From the point view of maximising GHG mitigation suing surplus "curtailed" renewable power, however, Daggash et al. (2018) instead indicates the best use would be for direct air capture and CCS. These results depend on what system is being measured, and what the objective is.

There are several potential crucial transitional roles for synthetic hydrocarbons (e.g. methane, methanol, ethanol, diesel, jet fuel) constructed using fossil, biomass or direct carbon capture (DAC) CCU (Bataille 2019a; Fasihi et al. 2017; Bataille et al. 2018a; Sternberg and Bardow 2015; Breyer et al. 2015; Dimitriou et al. 2015). They can allow reductions in the GHG intensity of high value legacy transport, industry and real estate that currently runs on methane but cannot be easily or readily retrofitted. They can be used by existing long lived energy and feedstock infrastructure, transport and storage, which can compensate for seasonal supply fluctuations and contribute to enhancing energy security (Ampelli et al. 2015). Finally, they can reduce the GHG intensity of end-uses that are very difficult to run on electricity, hydrogen or ammonia (e.g. long-haul aviation).

A very large and important uncertainty is the long term demand for hydrocarbon (whether fossil, biomass or DAC based) fuels, feedstocks, and materials, and competition for biomass feedstock with other priorities, including agriculture, biodiversity and other proximate land use needs, as well as need for negative emissions through BECCS. The current global plastics production of around 350 $Mt \cdot yr^{-1}$ is almost entirely based on petroleum feedstock and recycling rates are very low. If this or future demand were to be 100 % biomass based it would require tens of exajoules of biomass feedstock. If demand can be lowered and recycling increased (mechanical as well as chemical) the demand for biomass feedstock can be much lower (Material Economics 2019). Promising routes in the short term would be to utilise CO_2 from anaerobic digestion for biogas and fermentation for ethanol in the production of methane or methanol (Ericsson 2017); methanol can be converted into ethylene and propylene in a methanol-to-olefins process and used in the production of plastics (Box 11.2). New process configurations where hydrogen is integrated into biomass conversion routes to increase yields and utilise all carbon in the feedstock are relatively unexplored (Ericsson and Lunds universitet. 2017; De Luna et al. 2019).

There are widely varying estimates of the capacity of CCU to reduce GHG emissions and meet the net-zero objective. According to Hepburn et al. (2019), the estimated potential for the scale of CO_2 utilisation in fuels varies widely, from 1 to 4.2 $GtCO_2 \cdot yr^{-1}$, reflecting uncertainties in potential market penetration, requiring carbon prices of around USD40 to USD80 tCO_2^{-1}, increasing over time. The high end represents a future in which synthetic fuels have sizeable market shares, due to cost reductions and policy drivers. The low end—which is itself considerable—represents very modest penetration into the methane and fuels markets, but it could also be an overestimate if CO_2-derived products do not become cost competitive with alternative clean energy vectors such as hydrogen or ammonia, or with direct sequestration. Brynolf et al. (2018) indicates that a key cost variable will be the cost of electrolysers for producing hydrogen. Kätelhön et al. (2019) estimate that up to 3.5 $GtC \cdot yr^{-1}$ could be displaced from chemical production by 2030 using CCU, but this would require clean electricity equivalent to 55% of estimated global power production, at the same time other sectors' demand would also be rising. Mac Dowell et al. (2017) suggest that while CCU, and specifically CO_2 based enhanced oil recovery, may be an important economic incentive for early CCS projects (up to 4-8% of required mitigation by 2050), it is unlikely the chemical conversion of CO_2 for CCU will account for more than 1% of overall mitigation.

The above CCU literature has identified that there may be a highly unpredictable competition between fossil, biogenic and direct air capture carbon to provide highly uncertain chemical feedstock, material and fuel needs. Fossil waste carbon will likely initially be plentiful but will add to net atmospheric CO_2 when released. Biogenic carbon is variably, partially net-negative, but the available stock will be finite and compete with biodiversity and agriculture needs for land. Direct air capture carbon will require significant amounts of low GHG electricity or methane with high capture rate CCS (Keith et al. 2018). There are clearly strong interactive effects between low carbon electrification, switching to biomass, hydrogen, ammonia, synthetic hydrocarbons via CCU, and CCS.

11.3.7 Strategy interactions and integration

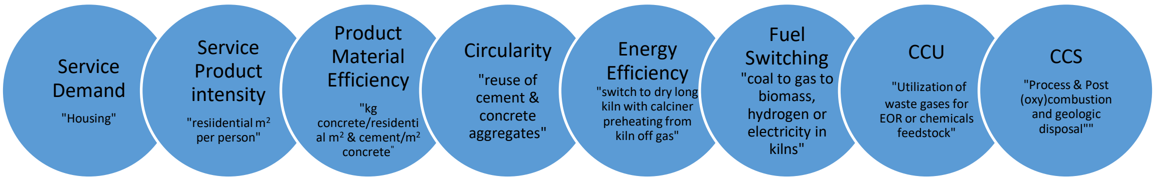


Figure 11.9 Fully interactive, non-sequential strategies for decarbonising industry

In this section we conceptually address how service demand, service product intensity, product material efficiency, energy efficiency, electrification and fuel switching, CCU and CCS may interact, and what conflicts and synergies may exist between them. To inform this we draw from a literature that has emerged largely since AR5 was completed (Åhman et al. 2016; Denis-Ryan et al. 2016; Bataille et al. 2018a; Wesseling et al. 2017b; Davis et al. 2018; Bataille 2019a; Axelson et al. 2018), that addresses integrative and interactive technical deep decarbonisation pathways for GHG intense industrial sectors, and how they interact with the rest of the economy. It is a common finding across this literature and a related scenario literature (Material Economics 2019; IEA 2019b; Energy Transitions Commission 2018; IEA 2020a) that deep decarbonisation of industry requires integrating multiple available options in the desired pathways. The combination of demand adjustment, circularity, material and energy efficiency, fossil use with CCS, electrification and fuel switching to bioliquids and gases, hydrogen, ammonia, or to net-zero synthetic hydrocarbons from CCU, biomass or direct air capture, varies from scenario to scenario with no option ignored; there is no 'silver bullet' and so all behavioural and technological options have to be mobilised (Figure 11.9). The key conclusion is that net zero emissions from the largest 'hard-to-abate' sources (e.g. steel, cement, ammonia, plastics) could be achieved by later this century by deploying already available and emerging multiple options packaged in different pathways. Material Economics (2019); IEA (2019b); Energy Transitions Commission (2018); UKCCC (2019a,b) and CAT (2020) identify pathways integrating energy efficiency, material efficiency, circular economy and innovative technologies options to cut GHG emissions across basic materials and value chains. Hence, from a technology options point of view, 'hard-to-abate' is a false designation of these sectors; "hard to transition" might be more appropriate, with more emphasis required on the policy mechanisms necessary to engage a challenging transition in highly competitive, currently GHG intense, price sensitive sectors with long lived capital stock (Bataille et al. 2018a; Bataille 2019a; Wesseling et al. 2017b).

While the strategies are not sequential and interact strongly, we discuss them in the order given. Reduced demand through reduced service demand and product intensity per service unit (Grubler et al. 2018; Van Vuuren et al. 2018) reduces the need for the next five strategies. Greater material efficiency (see earlier sections) reduces the need for the next four, and so on. By extension, CCS, and again by extension BE-CCS or DAC-CCS, is what remains if all other strategies are not sufficient to reach net zero emissions. But this reflects the growing demand for new energy and material intensive service and so loop is closed.

Circular economy introduces itself throughout, but mainly at the front end when designing materials and processes to be more materially efficient, efficient in use, and easy to recycle, and at the back end, when a material or product's services life has come to end, and it is time for recycling or sustainable disposal (Korhonen et al. 2018; Murray et al. 2017). The entire chain's potential will be maximised when these strategies are designed in ahead of time instead of considered on assembly, or as a retrofit (Allwood et al. 2012; Material Economics 2019; Gonzalez Hernandez et al. 2018a; IEA 2019b; Bataille 2019a). For example, when designing a building: 1) Is the building shell, interior mass and ducting orientated for passive heating and cooling, and can the shell and roof have building integrated solar PV or added easily, with hard-to-retrofit wiring already incorporated? 2) Are steel and well-made concrete only used where really needed (i.e. for shear, tension and compression strength), can sections be prefabricated off-sight, can other materials be substituted (e.g. wood)? 3) Can the interior fittings be built with easy to recycle plastics or other sustainably disposable materials (e.g. wood)? 4) Can this building potentially serve multiple purposes through its anticipated lifetime, are service conduits oversized and easy to access for retrofitting? 5) When it is time to be taken apart, can pieces be reused, and all components recycled at high purity levels, for example, can all the copper wiring be easily be found and removed, are the steel beams clearly tagged with their content? The answers to these questions will be very regionally and site specific, and require revision of educational curricula for the entire supply chain, as well as revision of building codes.

Energy efficiency in use and production is a key means to reduce upstream supply material and energy use. Energy saving potentials in industry are not exhausted and new decarbonised processes, for example, through electrification, must also be energy efficient. Reductions in specific energy use can be delivered through retrofits to best available technology (e.g. to dry lime kilns with preheating of the calciners using kiln heat in cement and general lime production).

Electrification and energy efficiency are deeply entangled, because fuel switching to electricity from fossil fuels in most cases improves GJ for GJ energy efficiency: resistance heaters are almost 100% efficient, heat pumps can be 300-400% efficient because they use electricity to concentrate or disperse ambient heat in a gas or liquid, induction melting can improve mixing and temperature control, and electric vehicle motors typically translate 90-95% of input electricity to motor drive in contrast to 40-45% for a modern stationary “truck size” internal combustion engine. The relevant question is instead how efficiently was the electricity made from primary energy? On a longer term horizon, direct electrocatalysis could allow the manipulation of key fuel and feedstock molecules (e.g. H_2, CO, CH_3OH, C_2H_5OH, C_2H_4, C_2H_6, CH_2O_2) in a much more direct and efficient way, bypassing inherent inefficiencies in thermalcatalysis (De Luna et al. 2019).

Even in a net-zero emission future carbon will remain a key and important building block in organic chemicals, fuels and materials. Thus, reaching net zero emissions, implies closing the loops on carbon and carbon dioxide through mechanical and chemical recycling, use of biomass feedstock or direct air capture, for the carbon needed in society (e.g., for plastics, chemicals, and aviation fuels). Addition of hydrogen can become important for maximising the potential of low GHG carbon sources (e.g. biomass) for fuels and feedstocks. One example is boosting the yield of biogas by adding hydrogen to the process and thereby utilising more of the carbon in the substrate.

As mentioned in the previous sections, there are potentially complicated physical and market fuel switching relationships between low GHG electricity, bioliquids and gases, hydrogen, ammonia, and synthetic hydrocarbons constructed using **CCU**, with remaining CO_2 potentially being disposed of using **CCS**. Whether or not they compete for a wide range of end uses and primary demand needs will be regional and whether or not infrastructure is available to supply them. Regions with less than optimal renewable energy resources, or not sufficient to meet needs, could also potentially indirectly import them as liquid or compressed hydrogen, ammonia or synthetic hydrocarbon feedstocks made in regions with relatively higher resources and lower needs (Armijo and Philibert 2019; Bataille 2019a).

There are different roles for different actors in relation to the different mitigation strategies (exemplified in Table 11.2). Energy efficiency is a relatively mature strategy, but technology development leads to new potentials for savings. Some aspects of circular economy are relatively well established (e.g., existing recycling of steel and paper) whereas others can be developed (e.g., design for reuse, new value chains and symbiosis). Materials efficiency exemplifies is a strategy where stronger efforts among all actors is needed to realise its potential.

Table 11.2 Indication and examples of the potential roles of different actors in relation to different mitigation strategies (Reader note: needs refinement, including an "all" column", e.g. for broad market instruments engaged by state/provincial/national actors)

Actors	Demand measures	Materials Efficiency	Circular Economy	Energy Efficiency	Electrification, hydrogen and fuel switching	CCU	CCS
Industrial sectors and associations	Support development of business models and approaches that make demand measures attractive	Training of designers, architects, engineers, etc. Develop design tools Map material flows	Facilitate new sectoral couplings and value chains Improve transparency on volumes and flows	Maintain high expertise and knowledge sharing. Transparency and benchmarking	Support innovation Share best practice.	Allocation rules, monitoring and transparency. Coordination and collaboration across sectors.	Transparency, monitoring and labelling. Coordination and collaboration for infrastructure
Corporations and companies	Digital solutions to reduce office space and travel Service oriented business models for lower product demand	Design for durability Design for light weight. Minimise industry scrap	Design for reuse and recycling Use recycled feedstock and develop industrial symbiosis	Maintain energy management systems	Develop and deploy new technologies in production	Develop new technologies. Engage in new value chains and collaborations.	Plan for CCS (e.g. plant compatibility and access to storage) and prepare for phase-out of plants with no prospects
International bodies	Best practice sharing. Knowledge building on demand options	Progressivity in international standards (e.g. ISO)	Transparency and regulation around products, waste handling and recycling	Maintain efforts for sharing good practice and knowledge	Coordinate innovation efforts and technology transfer. Access to lower cost finance. Establish leadership groups	Coordinate and develop accounting and standards. Ensure transparency	Align regulation to facilitate export, transport and storage.
Regional and national government, and cities	Reconsider spatial planning and regulation that has demand implications	Procurement guidelines and better indicators Build knowledge Collect material flow data	Regulation on product design (e.g., Ecodesign directive) Collect material flow data	Continue and develop energy efficiency policies	RD&D funding. Policy strategies for making investment viable (including carbon pricing instruments)	Align regulation to facilitate implementation and ensure accountability for emissions	Develop regulation and make investment viable Resolve long term accountability

Civil society	Information and advocacy related to norms	Strengthen lobby efforts	Engage in standards and transparency	Monitor progress	Assess renewable electricity and grid expansion	Develop standards and accounting rules	Ensure transparency and accountability

[Reader note: Chapter 11 team still discussing if the following related table & figure belong here or the end of 11.4.x]

Table 11.3 and Figure 11.10 summarise the estimated contributions from these strategies from the available literature.

Table 11.3 Assessment of apportionment of mitigation by strategy

	Interactive strategies							
	Service demand	Service product intensity	Product material efficiency	Circularity	Energy efficiency	Fuel switching	CCU	CCS
Range of contributions in the literature?								
IEA 2019								
ME 2019								
UKCCC (2019)								
IEA ETP 2020								

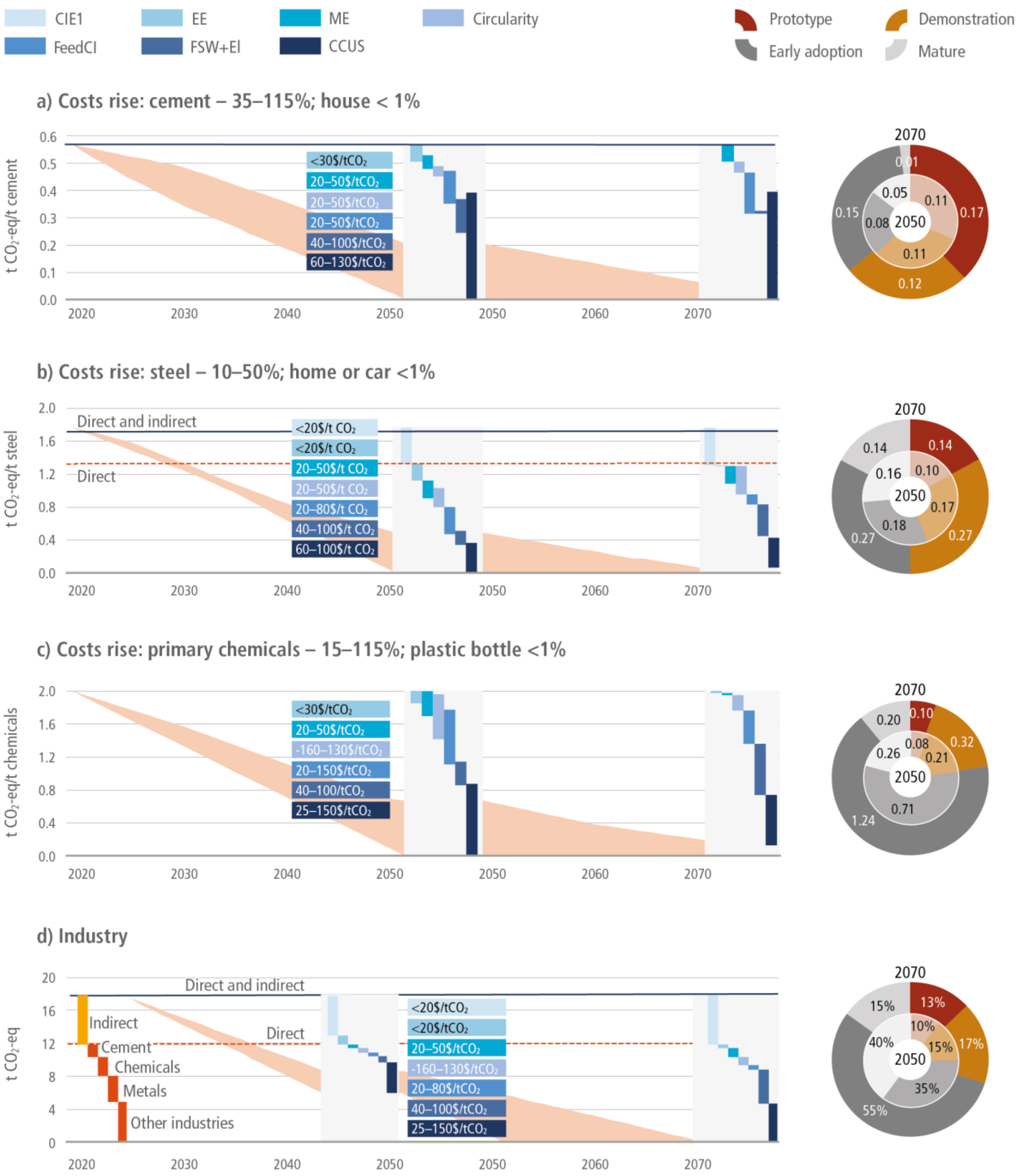


Figure 11.10 Potentials and costs for zero-carbon mitigation options for industry and basic materials CIEL –carbon intensity of electricity for indirect emissions; circular material flows (clinker substituted by coal fly ash, blast furnace slag or other by-products and waste, steel scrap, plastic recycling, etc.); FeedCI – feedstock carbon intensity (hydrogen, biomass, novel cement, natural clinker substitutes); FSW+El – fuel switch and processes electrification with low carbon electricity. Ranges for mitigation options are shown based on bottom-up studies for grouped technologies packages, not for single technologies. In circles contribution to mitigation from technologies based on their readiness are shown for 2050 (2040) and 2070. Direct emissions include fuel combustion and process emissions. Indirect emissions include emissions attributed to consumed electricity and purchased heat. For basic chemicals only methanol, ammonia and high-value chemicals are considered. Total for industry does not include emissions from waste. Base values for 2020 for direct and indirect emissions were calculated using 2018 GHG emission data (Crippa et al. (2019) and data for materials production from IEA (2020a) and World Steel Association (2020a). For steel base values for specific direct and indirect emissions are estimated at

1.33 tCO_2-eq·t^{-1} and 0.44 tCO_2-eq·t^{-1} totalling to 1.77 tCO_2-eq·t^{-1}. Using similar approach IEA (2020c) provides direct emissions at 1.4 and total at 2 tCO_2·t^{-1}, while World Steel Association (2020b) provides total at 1.85 tCO_2-eq·t^{-1}, and Gielen et al. (2020)) at 1.83 tCO_2·t^{-1}.

Sources: IEA (2020a); CAT (2020); IEA (2019b); Pauliuk et al. (2013a); IEA (2020c, 2019f,g); Material Economics (2019); Fawkes et al. (2016); WBCSD (2016); MPA (2020); CAMBUREAU (2020); ICEF (2019); Friedmann et al. (2019)

The following section, 11.4, will address measures interactive impacts in scenarios that have included more than one of these strategies in more detail.

11.4 Sector mitigation pathways and cross sector implications

The sub-section continues the discussion of the various strategy elements introduced in section 11.3 (spanning the range from service demand, service product intensity, product material efficiency, energy efficiency, electrification and fuel switching to CCU and CCS) and makes them explicit for the most relevant industry sectors. For the various sectors, Section 11.4.1 concludes with an overview on key technologies and processes, their technology readiness level (TRL), potential year of market penetration, mitigation potential and assessment of associated mitigation costs (Figure 11.10).

As some of the given options can be implemented on a short-term basis while others are less mature and can become important rather in the long-term it's obvious that the right sequencing of options is crucial to avoid lock-in effects. Temporal implementation and the discussion of the general (quantitative) role of the different options to achieve a net-zero industry is core to the second part of the subsection. Here mitigation pathways for the overall industry sector will be analysed. This comprises the collection and discussion of mitigation scenarios available in the literature with a high technological resolution for the industry sector in addition to a set of illustrative global and national GHG mitigation scenarios selected from chapter 3 and 4 representing different GHG mitigation ambitions respectively different pathways to achieve certain mitigation targets. Comparing very often technology focussed sector based scenarios with more top down oriented scenario approaches allows a reciprocal assessment of both worlds and helps to identify robust elements for the transformation of the sector. Eventually, scenarios allow at least crude estimates about the necessary investments as well as about relevant investment cycles and potential risks of stranded or depreciated assets. Comparison of real-world conditions within the sector (e.g. industry structure and logics, investment cycles, market behaviour, power and institutional capacity) and transformative pathways described in the scenarios helps to understand the need not only for technological change, but also for structural change and change in mind-set.

When undergoing a transformative process, it is obvious that interactions occur, within the sector but also on a cross-sectoral basis. Relevant interactions will be identified and discussed in the third and fourth part of the sub-section. Changes are induced along the whole value chains, i.e. switching to an alternative (climate-friendly, e.g. hydrogen based) steel making process has substantial impacts on the value chain and the associated sub-suppliers. In addition, cross-sectoral interactions will be discussed. This includes feedback loops with other end-use chapters, for example, higher material demand through market penetration of GHG mitigation technologies or measures (e.g. insulation materials for buildings, steel for windmills) or substantial demand on critical materials (e.g. rare earth demand for producing batteries for electric vehicles or photovoltaic modules). Generally, if consumption (or behaviour) driven additional material demand rises to a critical level material substitution, recycling/re-use and sustainable consumption patterns will become inevitable to limit GHG emissions and to enable a net-zero carbon strategy in the industry sector.

11.4.1 Sector specific mitigation potential and costs

Based on the general discussion of strategies across industry in Section 11.3 this sub-section focuses on the sector perspective and provides insights into the sector specific mitigation technologies and potential. As industry comprises many different areas discussion here is limited to the most important sources of GHG emissions, i.e. steel, cement and concrete as well as chemicals.

11.4.1.1 Steel

For the period leading up to 2020, in terms of end-use allocation globally, 40% of steel is used for structures, 20% for industrial equipment, 18% for consumer products, 13% for infrastructure, and 10% for vehicles (Bataille 2019b). The global production of crude steel increased by 29 % between 2008 and 2017 (World Steel Association 2019) and its CO_2 emissions represented about 7-10% of total global anthropogenic CO_2 emissions in 2019 depending on whether coke oven and blast furnace gases, sometimes used for electricity generation, are counted to steel or electricity (World Steel Association 2019; Olivier and Peters 2018; IEA 2020c). Steel production can be divided into primary production based on iron ore and secondary production based on steel scrap. The blast furnace converter route is the most important primary steel route globally, while the electric arc furnace (EAF) is the preferred process for the less energy and emissions-intensive melting and alloying of recycled steel scrap. In 2017, 72 % of global crude steel production was produced in blast furnace converters, while 28 % was produced in electric arc furnaces (World Steel Association 2019).

An estimated 15% energy efficiency improvement is possible within the basic oxygen furnace (BOF) process. Several options exist to significantly reduce GHG emissions related to steel production processes (Leeson et al. 2017; Axelson et al. 2018; BIrat 2011; Fischedick et al. 2014; Vogl et al. 2018a). Each could reduce specific CO_2 emissions of primary steel production by 80% or more relative to today's dominant blast /basic oxygen furnace route if input streams are based on carbon-free energy sources:

- *Increasing the share of the secondary route* can bring down emissions quickly and potential emissions savings are significant (Pauliuk et al. 2013a). However, realising this potential requires careful sorting and scrap management (Daehn et al. 2017), especially to eliminate copper contamination. Also, 85% of steel is recycled already; the gains are mainly to be made in quality. Furthermore, some amount of carbon is currently needed in the EAF to build CO_2 for the formation of foam slag. The source of this carbon as well as the carbon of the electrodes used in the EAF could be based from waste, biomass or air captured carbon in the future. If the electricity used in the EAF is also based on carbon-neutral sources and all heating is done electrically (some pre-heating is currently typically done with fossil fuels), this route can be nearly CO_2 neutral.
- *Hydrogen-based direct reduced iron (H-DRI)* is based on the already commercialised DRI technology. Most DRI facilities currently use a methane based syngas of H_2 and CO as both reductant and fuel. The reduction process of iron ore is typically followed by an EAF for melting and alloying. During a transitional period, DRI could start with methane or a mixture of methane and hydrogen as some of the methane (<=30% hydrogen can be substituted with green or blue hydrogen without the need to change the process). The Swedish HYBRIT project aims to transform the process almost entirely to hydrogen, with just enough carbon added to the steel for guaranteeing the required strength. If the hydrogen is produced based on carbon-free energy sources, this steel production process can be nearly CO_2 neutral (Vogl et al. 2018a).

- In the *aqueous electrolysis* route (small scale piloted as Siderwin during the EU ULCOS program, with Arcelor Mittal considering a commercial scale pilot), the iron ore is bathed in an electrolyte solution and an electric current is used to remove the oxygen, followed by an electric arc furnace for melting and alloying. In the *molten oxide electrolysis* route (being piloted by Boston Metals in the US), an electric current is used to directly reduce and melt the iron ore using electrolysis in one step, followed by alloying. These processes both promise a significant increase in energy efficiency compared with the direct reduced iron and blast furnace routes (Cavaliere 2019). If the electricity used is based on carbon-free energy sources, this steel production process can be nearly CO_2 neutral. Both processes would require supplemental carbon, but this is typically only up to 0.05% per tonne steel, with a maximum 2.1%. Aqueous electrolysis is possible with today's electrode technologies, while molten oxide electrolysis would require advances in high temperature electrodes.
- The *HIsarna® process* is a new type of coal-based smelting reduction process, which allows certain agglomeration stages (coking plant, sintering/pelletising) to be dispensed with. The iron ore, which can contain a certain amount of steel scrap, is directly reduced to pig iron in a single reactor. This process is suitable to be combined with CCS technology because of its relatively pure CO_2 exhaust gas flow. Specific CO_2 emission reductions of 80% are believed to be realisable relative to the conventional blast furnace route (Quader et al. 2016). The total GHG balance also depends on the further processing in a basic oxygen furnace or in an EAF. The HIsarna process was small scale piloted under the EU ULCOS program, and Tata Steel is considering a commercial scale pilot.

Reflecting the different conditions at existing and potential future plant sites, when choosing one of the above options a combination of different measures and structural changes (including electricity, hydrogen and CCUS infrastructure needs) will likely be necessary in the future to achieve deep reductions in CO_2 emissions of steel production.

In addition, increases in material efficiency (e.g. less, more targeted steel use per vehicle, building or piece of infrastructure) and increases in the intensity of material use (e.g. sharing cars instead of owning them) can contribute significantly to reduce emissions by reducing the need for steel production. The IEA (2019b) suggested that up to 24% of cement and 40% of steel demand could be plausibly reduced through strong material efficiency efforts. Potential material efficiency contribution for the EU is estimated to be much higher – 48% (Material Economics 2019). Recycling would cut the average CO_2 emissions per tonne of steel produced by 60% (Material Economics 2019), but globally secondary steel production is limited to 40–50% in various scenarios (IEA 2019b). CCU and more directly CCS are other options to reduce GHG emissions; mitigation potential of the usage of waste gases from the steel making process (that comprise inter alia CO_2, CO and H_2) depends amongst others on the full "ground to feedstock & air to ground" life cycle net GHGs that can be allocated to the process, i.e. if avoided emissions of alternative chemical production are fully incorporated (See section 11.3.6). Bio-based fuels can also substitute for some of the coal input, with emissions reductions of around 50%, but due to other demand for biomass this strategy is likely to be limited. Bio-based fuels can also substitute for some of the coal input, with emissions reductions of around 50%, but due to other demand for biomass this strategy is likely to be limited.

Abatement costs for these strategies vary considerably from case to case and for each a plausible cost range is difficult to establish (Leeson et al. 2017; Axelson et al. 2018; Fischedick et al. 2014; Vogl et al. 2018a; BIrat 2011).

11.4.1.2 Cement and concrete

The cement sector is regarded as a sector where emissions are especially “hard-to-abate” (Energy Transitions Commission 2018). Cement is used as the glue to hold together sand, gravel and stone aggregates to make concrete, the most consumed manufactured substance globally. The production of cement has been increasing faster than the global population since the middle of the last century (Scrivener, John, & Gartner, 2018). Despite significant improvements in energy efficiency over the last couple of decades (e.g. a systematic move from wet to dry kilns with calciner preheaters feeding off the kilns) the direct emissions of cement production (sum of energy and process emissions) are estimated to be 6.4–8.0‰ of total global anthropogenic CO_2 emissions in 2017 (Andrew 2019; Lehne and Preston 2018; Bataille 2020a). Typically, about 40% of these direct emissions are from process heating (e.g. for calcium carbonate (limestone) decomposition into calcium oxide at 850°, directly followed by combination with cementious materials at about 1,450°C to make clinker), while 60 % are process CO_2 emissions from the calcium carbonate decomposition (IEA 2018c; Andrew 2019; Kajaste and Hurme 2016). Some of the CO_2 is reabsorbed into concrete products and can be seen as avoided during the decades long life of the products; estimates of this flux vary between 15 and 27% of the direct emissions (Schneider 2019). Some companies (e.g. Carboncure) are utilising this CO_2 absorbing characteristic of cement to mix waste CO_2 into concrete as a hardening agent, both to dispose of the CO_2 and improve the final concrete.

One of the simplest and most effective ways to reduce cement and concrete emissions is to make stronger cement through better concrete mixing and aggregate sizing and dispersal; poorly and well-made concrete can varying in strength by a factor of 4 for a given volume (Scrivener et al. 2018; Habert et al. 2020). This argues for a refocus of the market away from “one size fits all”, often bagged, cements to professionally mixed clinker, cementious material and filler mixtures appropriate to the needs of the end use.

Architects, engineers and contractors also tend to overbuild with cement because it is cheap, corrosion and water resistant. Buildings and infrastructure can be purposefully designed to minimise cement use to its essential uses (e.g. compression strength and corrosion resistance), and replace its use with other materials (e.g. wood, stone, other fibres) for non-essential uses. This could reduce cement use by 20–30% (D’Alessandro et al. 2016; Imbabi et al. 2012; Brinkerhoff and GLDNV 2015; IEA 2019b; Lehne and Preston 2018; Habert et al. 2020).

Because so much of the emissions from concrete come from the limestone calcination to make clinker, anything that reduces use of clinker for a given amount of concrete reduces it GHG intensity. While 95% Portland cement is common in some markets, it is typically not necessary for all end-use application, and many markets will add blast furnace slag, coal fly ash, or natural pozzolanic materials to replace cement as cementious materials; 71% was the global average clinker content of cement in 2019 (IEA 2020a). All these materials are limited in volume, but combination of roughly 2–3 parts ground limestone and one part heated clays can also be used to replace clinker (Scrivener et al. 2018; Lehne and Preston 2018; Habert et al. 2020). Local building codes determine what mixes of cementious materials are allowed for given uses, and would need to be modified to allow these alternative mixtures.

Ordinary Portland cement process CO_2 emissions cannot be avoided or reduced through the use of non-fossil energy sources. For this reason, CCS technology, which could capture just the process emissions (e.g. the EU LEILAC project, which concentrates the process emissions from the limestone calciner, see following paragraph) or both the energy and process-related CO_2 emissions, is often mentioned as a potentially important element of an ambitious mitigation strategy in the cement sector. Different types of CCS processes can be deployed, including post-combustion technologies such as amine scrubbing and membrane-assisted CO_2-liquefation, oxycombustion in a low to zero nitrogen environment (full or partial) to produce a concentrated CO_2 stream for capture and disposal, or calcium-looping (Dean et al. 2011). These approaches have different strength and weaknesses concerning emission abatement potential, primary energy consumption, costs and retrofitability (Voldsund et al. 2019; Gardarsdottir et al. 2019; Hills et al. 2016).

The energy-related emissions of cement production can also be reduced by using biogenic energy sources, hydrogen or electricity for generating the high-temperature heat at the calciner. In an approach pursued by the LEILAC research project, the calcination process step is carried out in a steel vessel that is heated indirectly (Hills et al. 2017). In principle, this approach allows the potential *electrification of the calciner* by electrically heating the steel tank instead of using fossil burners. The LEILAC approach makes it possible to capture the process-related emissions in a comparatively pure CO_2 stream, which reduces the energy required for CO_2 capture and purification. This technology (LEILAC in combination with CCS) could reduce total furnace emissions by up to 85% compared with an unabated, fossil fuelled cement plant, depending on the type of energy sources used for heating (Hills et al. 2017).

In the long run, it may not be possible to completely decarbonise cement based on Portland cement chemistry, and alternatives will be needed. Several lower carbon chemistries are already on the market (e.g. carbonatable calcium silicate clinkers), and there is the potential for negative carbon cements based on carbonation of abundant magnesium silicates or ultramafic rocks (Scrivener et al. 2018; Lehne and Preston 2018).

All the above, however, require comprehensive education and continuing re-education for cement producers, architects, engineers, contractors and small, non-professional users of cements. Abatement costs for these strategies vary considerably (Moore 2017; Leeson et al. 2017; IEA 2019e; Wilson et al. 2003; Lehne and Preston 2018; Scrivener et al. 2018; Habert et al. 2020).

11.4.1.3 Chemicals

The chemical industry produces a broad range of products that are used in a wide variety of applications. The products range from plastics and rubbers to fertilisers, solvents, and specialty chemicals such as food additives and pharmaceuticals. The industry is the largest industrial energy user and its direct emissions were about 3% of total global anthropogenic CO_2 emissions in 2017 (IEA 2019e; Olivier and Peters 2018) and 9% of industrial sector emissions (Table 11.4). With regard to energy requirements and CO_2 emissions, ammonia, methanol, olefins, and chlorine production are of great importance (BOULAMANTI and MOYA RIVERA 2017). Ammonia is primarily used for nitrogen fertilisers, methanol for adhesives, resins, and fuels, whereas olefins and chlorine are mainly used for the production of polymers, which are the main components of plastics.

Technologies and process changes that enable the decarbonisation of chemicals production are specific to individual processes which may be changed to reduce, if not eliminate process emissions. However, a significant share of the sector's emissions is caused by the need for heat and steam in the production of primary chemicals (Box 11.2) (Bazzanella and Ausfelder 2017). The energy for this heat and steam is currently supplied almost exclusively through fossil fuels which could be substituted through low or zero carbon electricity (e.g. using electric boilers or high-temperature heating pumps), hydrogen and/or through the use of biomass (Bazzanella and Ausfelder 2017; Thunman et al. 2019).

The production of ammonia causes most CO_2 emissions in the chemical industry, about 30% (IEA 2018a). Ammonia is produced in a catalytic reaction between nitrogen and hydrogen – the latter most often produced through natural gas reforming (Material Economics 2019; Stork et al. 2018) and in some regions through coal gasification, which has several times higher associated CO_2 emissions. Future low-carbon options include hydrogen from electrolysis using on low or zero-carbon energy sources (Philibert and IEA 2017), natural reforming with CCS, or methane pyrolysis, a process in which methane is transformed into hydrogen and solid carbon (Material Economics 2019; Bazzanella and Ausfelder 2017) (see also Section 11.3.5 and Box 11.1). Steam cracking of naphtha and natural gas liquids for the production of olefins, i.e. ethylene, propylene and butenes, and other high value chemicals is the second most CO_2 emitting process in the chemical industry, accounting for another almost 20% of the emissions from the subsector. Future lower-carbon options include electrifying the heat supply in the steam cracker as described above, although this will not remove the associated process emissions from the cracking reaction itself or from the combustion of the by-products. Further in the future, electrocatalysis of carbon monoxide, methanol, ethanol, ethylene and formic acid could allow direct electric recombination of waste chemical products into new intermediate products (De Luna et al. 2019).

In a net-zero world carbon will still be needed for many chemical products, but the sector must also address the life-cycle emissions of its products which arise in the use phase, for example, CO_2 released from urea fertilisers, or at the end-of-life, for example, incineration of waste plastics. Reducing life-cycle emissions can partly be achieved by closing the material cycles (Figure 11.11) starting with material and product design planning for re-use, re-manufacturing, and recycling of products – ending up with chemical recycling which yields recycled feedstock that substitute virgin feedstocks for various chemical processes (Smet and Linder 2019; Rahimi and Garciá 2017). However, chemical recycling processes have significant carbon losses, which result in CO_2 emissions. Achieving carbon neutrality would thus require this CO_2 either to be recirculated through energy-consuming synthesis routes or to be captured and stored (Thunman et al. 2019; Geyer et al. 2017b; Lopez et al. 2018; Material Economics 2019). As all chemical products are unlikely to fit into chemical recycling systems, CCS can be used to capture and store a large share of their end-of-life emissions when combined with waste combustion plants or heat-demanding facilities like cement kilns (Leeson et al. 2017; Tang and You 2018).

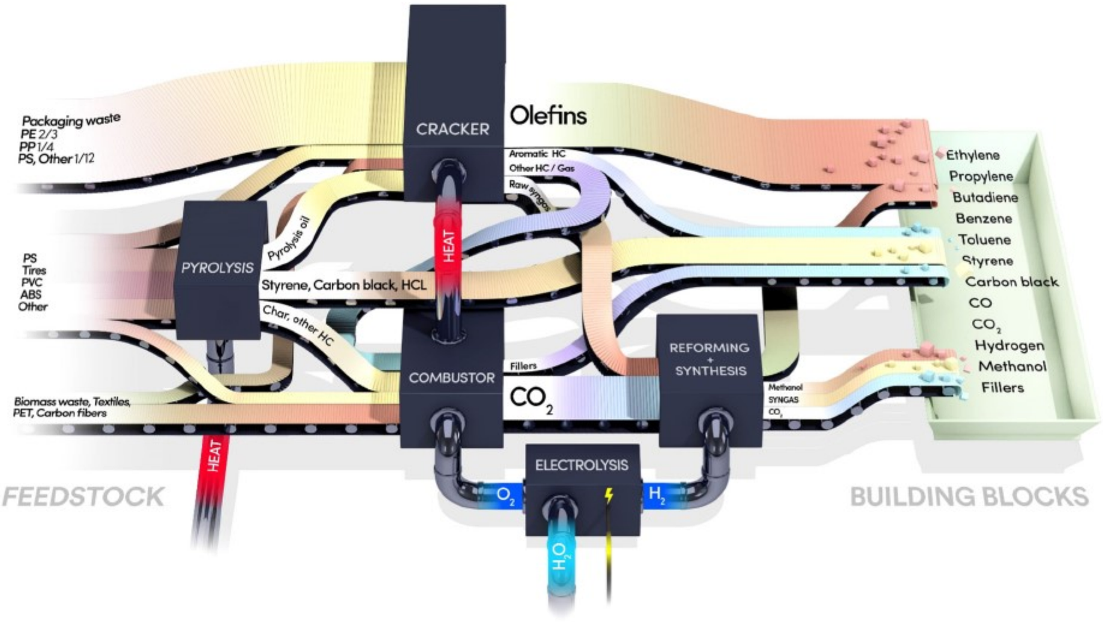


Figure 11.11 Schematic of a technical solution for closing the material cycle of plastics using thermochemical recycling, which can handle any type of plastic waste (sorted or mixed) with close to 100% carbon recovery

(Thunman et al. 2019), illustration by BOID

Looking more specifically at the different processes the following technologies and options could be key in avoiding the end-of life emissions of chemical products:

- *Chemical recycling* of plastics unsuitable for mechanical recycling was already mentioned (Smet and Linder 2019; Thunman et al. 2019). Through *pyrolysis* of old plastics, both gas and a naphtha-like pyrolysis oil can be generated, a share of which could replace fossil naphtha as a feedstock in the steam cracker (Honus et al. 2018a,b). Alternatively, waste plastics could be *gasified* and combined with low-carbon hydrogen to a syngas suitable for methanol production (e.g. for the MTO/MTA route) (Stork et al. 2018; Lopez et al. 2018).
- In the *methanol-to-olefin/aromatics route (MTO/MTA)*, low carbon methanol can be used as a feedstock for the production of olefins and aromatics production, substituting the steam cracker (Gogate 2019). The methanol can be produced from climate-neutral hydrogen and carbon from a non-fossil source (biomass or direct air capture), from chemical recycling of plastics or – during a transition period – from fossil CO_2 from industrial processes.
- Substituting fossil carbon at the inception of a product life-cycle for carbon from renewable sources processed in designated biotechnological processes or through direct air capture (Hatti-Kaul et al. 2020).

Reflecting the diversity of the sector the listed options can only be illustrative. However, while most of the chemical processes for doing all the above are well-known and have been used commercially at least partly, they have not been used at large scale and in an integrated way. In the past external conditions (e.g. availability and price of fossil feedstocks) have not set the necessary incentives to implement alternative routes and to avoid emitting combustion and process related CO_2 emissions to atmosphere. Most of these processes will very likely be more costly than using fossil fuels and full scale commercialisation would require significant policy support and the implementation of dedicated lead markets (Bataille et al. 2018a; Material Economics 2019; Wesseling et al. 2017b; Wyns et al. 2019). As in other sub-sectors abatement costs for the various strategies vary considerably across regions and products making it difficult to establish a plausible cost range for each (Bazzanella and Ausfelder 2017; De Luna et al. 2019; IEA 2018a; Philibert and IEA 2017; IEA 2017b; Axelson et al. 2018).

11.4.1.4 Other industry sectors

The other big sources of global industrial combustion and process CO_2 emissions are fossil fuel use by light manufacturing and industry (9.7% in 2016), non-ferrous metals like aluminium (0.8%), and pulp and paper (0.8%) (Bataille 2020a).

Light manufacturing and industry

Most of the fossil fuels used for light manufacturing & industry are used for steam or small scale process heat that can be replaced with low GHG electricity, through direct resistance, heat pumps, induction or thermomechanical processes (Lechtenböhmer et al. 2016). Madeddu et al. (2020) argue up to 78% of Europe's industrial energy requirements are electrifiable through existing commercial technologies. Some direct solar heating is possible as well for low temperature needs (<100°C), especially when combined with heat pumps, which have been commercially available rated to 150°C. Where flame-front temperatures (>1000°C) are required, hydrogen, biogenic or synthetic combustible hydrocarbons (methane, methanol, ethanol, LPG, etc.) can also be used (Bataille et al. 2018a).

There is also a large potential for energy cascading in industrial clusters similar to the one at Kalundborg, Denmark. Waste heat can be passed at lower and lower temperatures from facility to facility or circulated as low grade steam or hot water, and boosted as necessary using heat pumps and direct heating. Such geographic clusters would also enable lower cost infrastructure for hydrogen production and storage as well as CO_2 gathering, transport and disposal (IEA 2019e).

Aluminium and other non-ferrous metals

Demand for aluminium comes from a variety of end-uses where a reasonable cost, light-weight metal is desirable. It has historically been used in aircraft, window frames, strollers, beverage containers and more recently in cars. As fuel economy has become more desirable and design improvements have allowed crush bodies made of aluminium instead of steel, aluminium has become progressively more attractive for cars. Aluminium demand is net of manufacturing wastage reuse (about 15% of virgin and recycled input; see Figure 11.12) and end-of-life recycling (about 20–25% of what reaches market). Aluminium consumption rose from under 20 Mt·yr^{-1} in 1995 to almost 60 Mt in 2018 (International Aluminum Institute 2018), with the OECD forecasting increases in demand by 2060 for primary aluminium to 139 Mt·yr^{-1} and for secondary aluminium to 71 Mt (OECD 2019a).

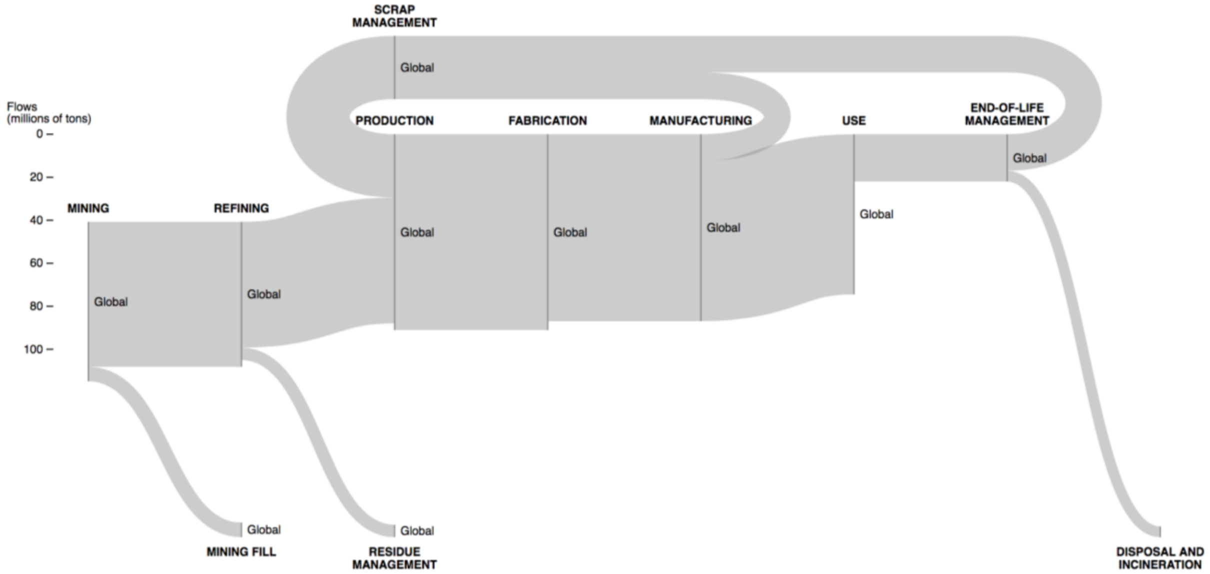


Figure 11.12 2016 Global aluminium life cycle flows
(World Aluminium 2018)

Raw (as opposed to recycled) aluminium is generally made in a two stage process, often geographically separated. In the first stage aluminium oxide is extracted from bauxite ore (often with other trace elements) using the Bayer hydrometallurgical process, which requires up to 200°C heat when sodium hydroxide is used to leach the aluminium oxide, and up to 1000°C for kilning. This is followed by electrolytic separation of the oxygen from the elemental aluminium using the Hall Heroult process, by far the most emissions intense part of making aluminium. This process has large potential emissions from the electricity used (12.5 MWh per tonne aluminium BAT, 14–15 MWh per tonne average). 1.5 tonnes of direct CO_2 are currently emitted for each tonne aluminium as the graphite electrodes are depleted and combine with oxygen, and if less than optimal conditions are maintained perfluorocarbons can be emitted with widely varying GHG intensity, up to the equivalent of 2 tonnes CO_2-eq.per tonne aluminium. PFC emissions, however, have been greatly reduced globally and almost eliminated in well-run facilities. Aluminium, if is not contaminated, is highly recyclable and requires 1/20 of the energy required to produce virgin aluminium; increasing aluminium recycling rates from the 20–25% global average is a key emissions strategy.

The use of low and zero GHG electricity (e.g. historically from hydropower) can greatly reduce the large indirect emissions associated with making aluminium. A partnership (aka Elysis Technologies Ltd.) between Apple, Rio Tinto and Alcoa with financial support from the province of Québec and the Canadian federal government has recently announced a fundamental modification to the Hall Heroult process by which the graphite electrode process emissions can be eliminated by substitution of inert electrodes. This technology is slated to be available in 2024 and is potentially retrofittable to existing facilities.

Smelting and otherwise processing of other non-ferrous metals like nickel, zinc, copper, magnesium and titanium with less overall emissions have relatively similar emissions reduction strategies (Bataille and Stiebert 2018): 1) Increase material efficiency. 2) Increase recycling of existing stock. 3) Pursue ore extraction processes (e.g. hydro- and electro-metallurgy) that allow more use of low carbon electricity as opposed to pyrometallurgy, which uses heat to melt and separate the ore after it has been crushed. These processes have been used occasionally in the past, but have generally not been used due to the relatively inexpensive nature of fossil fuels.

Pulp and Paper

The pulp and paper industry has pulp mills, integrated pulp and paper mills and paper mills using virgin pulpwood and recycled paper as feedstock. Pulp mills and integrated mills typically have access to bioenergy in the chemical pulping processes. Mechanical pulping mainly uses electricity for energy; decarbonisation thus depends on grid emission factors. With the exception of the lime kiln in kraft pulp mills, process temperature needs are typically less than or equal to 150ºC to 200ºC, mainly steam for heating and drying. This means that this sector can be relatively easily decarbonised through continued energy efficiency, fuel switching and electrification, including use of high temperature industrial heat pumps (Ericsson and Nilsson 2018); recent commercial versions of which have been capable of 150°C given a sufficient heat source. The PPI also has the capabilities, for example, resources and knowledge, to implement these changes. Inertia is mainly caused by equipment turn-over rates, relative fuel and electricity prices, and the profitability of investments.

A larger and more challenging issue is how the forestry industry can contribute to the decarbonisation of other sectors and how biogenic carbon will be used in a fossil-free society, for example, through developing the forest based bioeconomy (Bauer 2018; Pülzl et al. 2014). In recent years the concept of biorefineries has gained increasing traction. Most examples involve innovations for taking by-products or diverting small streams to produce fuels, chemicals and bio-composites that can replace fossil-based products, but there is little unity on what really constitutes a biorefinery (Bauer et al. 2017). Some of these options have limited scalability and the cellulose fibre remains the core product even in the relatively large shift from paper production to textiles fibre production.

Pulp mills have been identified as promising candidates for post-combustion capture and CCS (Onarheim et al. 2017), which could allow some degree of negative emissions. For deep decarbonisation across all sectors, notably switching to biomass feedstock for fuels, organic chemicals and plastics, the availability of biogenic carbon (in biomass or as biogenic CO_2, Chapter 7) becomes an issue. A scenario where biogenic carbon is CCU as feedstock implies large demands for hydrogen, completely new value chains and more closed carbon loops, an area which is as yet largely unexplored (Ericsson 2017).

11.4.1.5 Overview of estimates of specific mitigation potential and abatement costs of key technologies and processes for main industry sectors

Climate policy related literature focusing on deep industrial emission reductions has expanded rapidly since AR 5. An increasing body of research proposes deep decarbonisation pathways for energy intensive industries (Figure 11.10). Bataille et al. (2018a) address the question of whether it is possible to reduce GHG emissions to very low, zero, or negative levels, and identifies preliminary technological and policy elements that may allow the transition, including the use of policy to drive technological innovation and uptake. Material Economics (2019); IEA (2019b), Energy Transitions Commission (2018) and Climate Action Tracker CAT (2020) take steps to identify pathways integrating energy efficiency, material efficiency, circular economy and innovative technologies options to cut GHG emissions across basic materials and value chains. The key conclusion is that net zero emissions from the largest sources (steel, plastics, ammonia, and cement) could be achieved by 2050 by deploying already available multiple options packaged in different ways (Material Economics 2019; UKCCC 2019b; Davis et al. 2018). The studies assume that for those technologies that have a kind of breakthrough technology status further technological development and significant cost reduction can be expected.

Table 11.4, modified from Bataille (2020a) and built from Bazzanella and Ausfelder (2017); Energy Transitions Commission (2018); IEA (2019e); Axelson et al. (2018); McMillan et al. (2016); Wesseling et al. (2017b); UKCCC (2019b); Material Economics (2019); Philibert and IEA (2017); Davis et al. (2018); Bataille et al. (2018a) and IEA (2020c) , presents carbon intensities that could be achieved by implementing mitigation options in major basic material industries, mitigation potential, estimates for mitigation costs, TRL and potential year of market introduction (see also Figure 11.10).

Table 11.4 acknowledges that for many carbon intensive products a large variety of novel processes, inputs and practices capable of providing very deep emission reductions are already available and emerging. However, their application is subject to different economic and structural limitations, therefore in the scenarios assuming deep decarbonisation by 2050–2060 different technological mixes can be observed (Section11.4.2).

Table 11.4 Technological potentials and costs for deep decarbonisation of basic industries

Sector	Current (tCO_2-eq t^{-1})	GHG reduction	TRL	Breakeven Est. (USD per tCO_2-eq, %)	Year
Iron and Steel					
Current intensity – all steel (Worldsteel)	1.83				
Current intensity – ~BF-BOF/ Best BF-BOF & NG-DRI (with near zero GHG electricity)	2.3/1.8 & 0.7				
Current intensity – EAF (depends on electricity intensity)	>=0	Up to 99%			
Material efficiency (IEA 2019 “Material Efficiency…”)		Up to 40%	9	Subject to supply chain building codes and education	
More recycling; depends on available stock, recycling network and quality of scrap		?	9	Subject to logistical costs	Today
BF-BOF with top gas recirculation & CCUS[i]		60%	6–7	USD70–130/t	2025
Hisarna with concentrated CO_2 capture[ii]		80–90%	7	USD40–70/t	2025
Hydrogen DRI EAF[iii] - Fossil hydrogen with CCS is in operation, electrolysis based hydrogen scheduled for 2026		Up to 99%	7	34–68 €/t & 40 €/MWh	2030
Aqueous (e.g. SIDERWIN) or Molten Oxide (e.g. Boston Metals) Electrolysis[iv]		Up to 99%	3–5	?	2035–2040
Cement & Concrete					
Current intensity, about 60% is limestone calcination	0.55				
Building design to minimise concrete (IEA 2019b, 2020a)		Up to 24%	9	Low, education, design and logistics related	2025
Alternative lower GHG fuels, e.g. waste (biofuels and hydrogen see above)		40%	9	Cost of alt. fuels	Today
CCUS for process heating & $CaCO_3$ calcination CO_2 (e.g. LEILAC, possible retrofit)[v]		99% calc., <=90% heat	5–7	<=USD40/t calc. <=USD120/t heat	2025
Clinker substitution (e.g. limestone + calcined clays)[vi]		40–50%	9	Near zero, education, logistics, building code revisions	Today
Use of multi sized and well dispersed aggregates[vi]		Up to 75%	9	Near zero	Today
Magnesium or ultramafic cements[vi]		Negative?	1–4	?	2040
Aluminium & other non-ferrous					
Current Al intensity, from hydro to coal based electricity production. 1.5 tonne CO_2 are produced by graphite electrode decay	1.5 t/t + electricity req., i.e. 10/t (NG) to 18 t/t (Coal)				

	Inert electrodes + green electricity[vii]		100%	6–7	Relatively low	2024
	Hydro/Electrolytic smelting (with CO_2 CCUS if necessary)		Up to 99%	3–9	Ore specific	<2030
Chemicals (see also crosscutting feedstocks above)[viii]						
	Catalysis of ammonia from low/zero GHG hydrogen H_2	1.6 (NG), 2.5 (naphtha) 3.8 (coal)	<=99%	9	Cost of H_2	Today
	Electrocatalysis: CH_4, CH_3OH, C_2H_5OH, CO, olefins[ix]		Up to 99%	3	Cost: E, H_2, CO_x	2030
	Catalysis of olefins from: (m)ethanol; H_2 & CO_x directly		9%	9,3	Cost: H_2 & CO_x	<2030
	End-use plastics, mainly CCUS and recycling	1.3–4.2, about 2.4	94%	5–6	USD150–240/t	2030?
Pulp & Paper						
	Full biomass firing, inc. lime kilns		90–99%	9	About USD50/t	Today
Other manufacturing						
	Electrification using current tech (boilers, 90°C–140°C heat pumps		99%	9	Cost: E vs. NG	Today
	" using new tech (induction, plasma heating)		99%	3–6		2025
Cross-cutting (CCUS, H_2, net-zero $C_oO_xH_y$ fuels/feedstocks)						
	CCUS of post-combustion CO_2 diluted in nitrogen[V]		Up to 90%	6–7	<=USD120/t	2025
	CCUS of concentrated CO_2[V]		99%	9	<=USD40/t	Today
	H_2 prod: Steam or auto-thermal CH_4 reforming w/ CCS[V]		SMR<=90% ATR >90%	6*, 9**	56% @<=USD40/t chem**, <=USD120 heat*,+20%/kg	<=2025
	H_2 prod: coal with CCUS[V]		<=90%	6	+25-50%/kg	<=2025
	H_2 prod: Alkaline or PEM Electrolysis[x]		99%	9	about USD50/t or <USD20-30/MWh	Today
	H_2 prod: Reversible solid oxide fuel electrolysis[x]		99%	6–8	about USD40/t or <USD40/MWh	2025
	H_2 prod: CH_4 pyrolysis or catalytic cracking[xi]		99%	5	?	2030?
	Hydrogen as CH_4 replacement		<=10%	9	See above	Today
	Biogas or liquid replacement hydrocarbons		60–90%	9	Biomass USD/GJ; >=USD50/t,uncertain	Today
	Anaerobic digestion/fermentation: CH_4, CH_3OH, C_2H_5OH[xii]		Up to -99%	9	Biomass cost	Today
	Methane or methanol from H_2 & CO_x (CCUS for excess). Maximum -50% reduction if C source is FF		50–99%	6–9	Cost: H_2 & CO_x	Today

	850°C woody biomass gasification w/ CCS for excess carbon: CO, CO_2, H_2, H_2O, CH_4, C_2H_4 & C_6H_6[xiii]		Could be negative	7–8	about USD50–75/t, uncertain	2025
	Direct air capture for short and long chain $C_oO_xH_y$[xiv]		Up to 99%	3	Cost: E, H_2, CO_x about USD94–232/t	<=2030

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[i] Data for CCS costs for steel making: Leeson et al. (2017); Axelson et al. (2018); BIrat (2011);

[ii] Data for Hisarna: Axelson et al. (2018);

[iii] Data for hydrogen DRI electric arc furnaces: Fischedick et al. (2014); Vogl et al. (2018a);

[iv] Data for molten oxide electrolysis (also known as SIDERWIN): Axelson et al. (2018); Fischedick et al. (2014). The TRLs differ by source, the value provided is from Axelson et al. (2018) based on UCLOS SIDERWIN;

[v] Data for making hydrogen from SMR and ATR with CCUS:Moore (2017); Leeson et al. (2017); IEA (2019e). The cost of CCS disposal of concentrated sources of CO_2 at USD15–40 tCO_2-eq^{-1} is well established as commercial for direct or EOR purposes and is based on the long standing practise of disposing of hydrogen sulfide and oil brines underground:Wilson et al. (2003); Leeson et al. (2017). There is a wide variance, however, in estimated tCO_2-eq^{-1} breakeven prices for industrial post-combustion capture of CO_2 from sources highly diluted in nitrogen (e.g. Leeson et al. (2017) at USD60–170 tCO_2-eq^{-1}), but most fall under USD120 tCO_2-eq^{-1};

[vi] Data for clinker substitution and use of well mixed and multi sized aggregates: Lehne and Preston (2018); Scrivener et al. (2018); Habert et al. (2020);

[vii] Rio Tinto, Alcoa and Apple have partnered with the governments of Québec and Canada to formed a coalition to commercialise inert as opposed to sacrificial graphite electrodes by 2024, thereby making the standard Hall Heroult process very low emissions if low carbon electricity is used;

[viii] Data and other information: Bazzanella and Ausfelder (2017); De Luna et al. (2019); IEA (2018a); Philibert and IEA (2017); IEA (2017b); Axelson et al. (2018);

[ix] See De Luna et al. (2019) for a state of the art review of electrocatalysis, or direct recombination of organic molecules using electricity and catalysts;

[x] Data for hydrogen production from electrolysis: Bazzanella and Ausfelder (2017); Philibert and IEA (2017); Armijo and Philibert (2019); IEA (2019e, 2017b);

[xi] Data for methane pyrolysis to make hydrogen: Abbas and Wan Daud (2010). Data for hydrogen production from methane catalytic cracking: Amin et al. (2011); Ashik et al. (2015);

[xii] Data for anaerobic digestion or fermentation for the production of methane, methanol and ethanol:De Luna et al. (2019);

[xiii] Data for woody biomass gasification: Li et al. (2019); Meijden et al. (2011);

[xiv] Data on direct air capture of CO_2: Keith et al. (2018); Fasihi et al. (2019).

While deep GHG emissions reduction potential is assessed for various regions, assessment of associated costs is limited to only a few regions; nevertheless those analyses may be illustrative also at the global scale. UKCCC (2019b) provides costs assessments for different industrial subsectors (see Table 11.4) for the UK which range between 2–85 GBP per tCO_2-eq for the core option (reduction in GHG emissions by about 50% by 2050 applying energy efficiency (EE), ME, CCS, biomass and electrification) and in between 32–119 GBP per tCO_2-eq for a more ambitious option (90% emissions reduction via wide spread deployment of hydrogen, electrification or bioenergy for stationary industrial heat/combustion and more ambitious resource and energy efficiency improvements) and scale up to 33–299 GBP per tCO_2-eq when full deployment of low-carbon options (EE and ME) is limited.

In Material Economics (2019) costs are provided for separate technologies and subsectors, and also by pathways, each including new industrial processes, circular economy and CCS components in different proportions allowing for the transition to net-zero industrial emission in the EU by 2050. That means that the study provides information about the three main mid- to long-term options which could enable a widely abatement of GHG emissions. Given different electricity price scenarios, average abatement costs associated with the circular economy-dominated pathway are 12–75 euro tCO_2-eq^{-1}, for the carbon capture-dominated pathway 79 euro tCO_2-eq^{-1}, and for the new processes dominated scenario 91 euro tCO_2-eq^{-1}. Consequently, net-zero emission pathways are about 3–25% costlier compared to the baseline (Material Economics 2019). According to Energy Transitions Commission (2018), cement decarbonisation would cost on average 110–130 USD tCO_2^{-1} depending on the cost scenario. Rootzén and Johnsson (2016) state that CO_2 avoidance costs for the cement industry vary from 25 to 110 € tCO_2^{-1}, depending on the capture option considered and on the assumptions made with respect to the different cost items involved. According to Energy Transitions Commission (2018), steel can be decarbonised on average at 60 USD tCO_2^{-1}, with highly varying costs depending on low carbon electricity prices..

For customers of final products, information about CO_2 abatement costs are not very useful. A different approach has been developed to assess the costs of mitigation by estimating the potential impacts of supply-side decarbonisation on final products prices. Material Economics (2019) shows that with deep decarbonisation, depending on the pathway, steel costs grow by 20–30%; plastics by 20–45%; ammonia by 15–60% and cement by 70–115%. Looking at the price for the final product shows for instance, that a car becomes only 0.5% more expensive, supported by Rootzén and Johnsson (2016). Similarly, a bottled soft drink becomes 1% more expensive (see Figure 11.10). The impact of decarbonisation on prices faced by end consumers varies by sector, but overall it would be small in comparison to material cost increase. Rootzén and Johnsson (2017) indicate that decarbonising cement making, while doubling the cost of cement, would add to the costs of a residential building <1%. IEA (2020a) indicates that the impact on the end-user products are rather small, even in a net-zero scenario. Price increase by only about 0.2% are calculated for a car and 0.6% for a house based on higher costs for steel and cement respectively.

Zero-carbon steel may cost 20% more per tonne than conventional steel. For companies competing globally with low margins this is substantive and first movers could be pushed out of business. On the other hand it is unlikely to add more than 180 USD (around 1%) to the cost of a car (Energy Transitions Commission 2018). According to Rootzén and Johnsson (2016), at a carbon price of 113 USD tCO_2^{-1}, the retail car price would have to be increased by approximately 113–141 USD to cover the increases in automotive steel costs. With the reference retail price set at USD31,500, this represents an increase in the retail price of less than 0.5%. Zero-carbon cement cost may double comparing with conventional one, but concrete will be only 10–30% more expensive and this is unlikely to add more than USD15,000 or 3% to the price of a house (Energy Transitions Commission 2018). Rootzén and Johnsson (2017) evaluates the impact of imposing a price tag on the CO_2 embedded in construction materials, and concludes that even with a high price for CO_2, the impact is likely to be limited to 1% of overall construction project costs.

Thus, the price impact scales down going across the value chain and might be acceptable for a significant share of customers. However, it has to be reflected that the cumulative price increase could be more significant if several different zero-carbon materials (e.g. steel, plastics, aluminium) in the production process of a certain product have to be combined, indicating the importance of material efficiency being applied along with production decarbonisation.

Box 11.2 Plastics and climate change

The global production of plastics has increased rapidly over the past 70 years, with a compound annual growth rate (CAGR) of 8.4 %, about 2.5 times the growth rate for global GDP (Geyer et al. 2017b) and higher than other materials since 1970 (IEA 2019b). Global production of plastics is now more than 400 million tonnes, including synthetic fibres (ibid.) The use of plastics is still up to 20 times higher in developed countries than in developing countries with low signs of saturation and the potential for an increased use is thus still very large (IEA 2018a). Plastics is the largest output category from the petrochemical industry, which as a whole currently uses about 14 % of petroleum and 8 % of natural gas (IEA 2018a). Forecasts for plastic production assuming continued growth at recent rates of about 3.5% point towards a doubled production by 2035, following record-breaking investments in new and increased production capacity in recent years (CIEL 2017). The IEA calculated that even in a world where transport demand for oil falls considerably by 2050 from the current ~100 mbpd, feedstock demand for chemicals will rise from ~12 mbpd to 15–18 mbpd (IEA 2019b). Projections for increasing plastic production as well as petroleum use together with the lack of investments in break-through low-emission technologies do not align with necessary emission reductions.

About half of the petroleum that goes into the chemical industry is used for plastics, and around half of this is combusted in these energy intensive production processes. GHG emissions from plastic production depend on the feedstock used (ethane based production is associated with lower emissions than naphtha based), the type of plastic produced (production of simple polyolefins is associated with lower emissions than more complex plastics such as polystyrene), and the contextual energy system (e.g. the GHG intensity of the electricity used) but weighted averages have been estimated to be 1.8 $tCO_2\text{-eq}\cdot t^{-1}$ for North American production (Daniel Posen et al. 2017) and 2.3 $tCO_2\text{-eq}\cdot t^{-1}$ for European production (Material Economics 2019). In regions more dependent on coal power the numbers are likely to be even higher. The production of plastics can thus conservatively be estimated to emit between 750 and 950 $MtCO_2\text{-eq}\cdot yr^{-1}$.

11.4.2 Transformation pathways

This sub-section continues the discussion of the different strategy elements introduced in sections 11.3 and 11.4.1 and gives an impression about their respective (quantitative) roles within the broad range of mitigation pathways reflecting different future perspectives and needs.

For the discussion of the general role and temporal implementation of the different options for achieving a net-zero industry, similar to other sector chapters a set of illustrative mitigation pathways will be analysed. The findings from theses analysis will be set into the context of results from a broader range of IAM based scenarios discussed in Chapter 3 (Section 11.4.2.1). In addition, specific studies and associated scenarios will be discussed which provide a particularly high technological resolution of the industry sector and allow a much deeper look into the interplay of different mitigation strategies (Section 11.4.2.2). The comparison of the typically more technology-focused sector-based scenarios with more top down oriented scenario approaches provides the opportunity for a reciprocal assessment and helps to identify robust elements for the transformation of the sector. Some of the scenarios allow for at least rough estimates of the necessary investments associated with the described transformation pathways as well as about relevant investment cycles and potential risks of stranded or depreciated assets. In some specific cases cost comparison can be translated into expected difference costs not only for the overall sector, but also for relevant materials or even consumer products.

11.4.2.1 Central results from (top down) scenarios analysis and illustrative pathways discussion

Chapter 3 conducted a comprehensive analysis of scenarios based on integrated assessment models (IAM). The scenarios span a broad range from baseline scenarios to the description of pathways that are compatible with the 1.5°C target. For selected indicators chapter 3 already discusses relevant aspects for the industry sector and shows the differences between the scenarios categories (Figure 11.13).

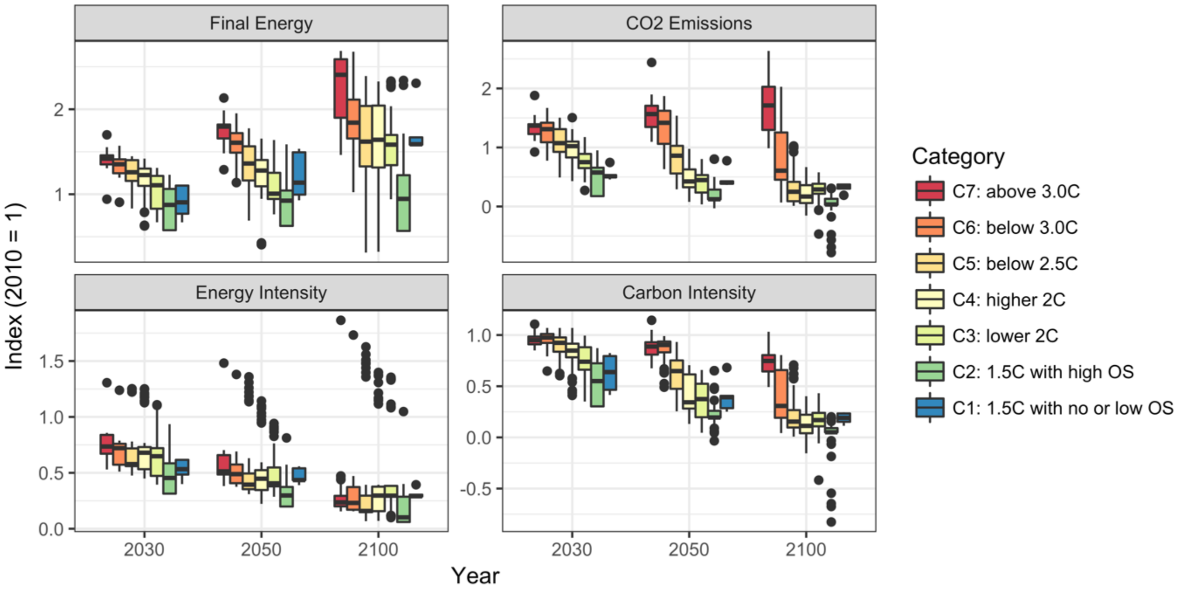


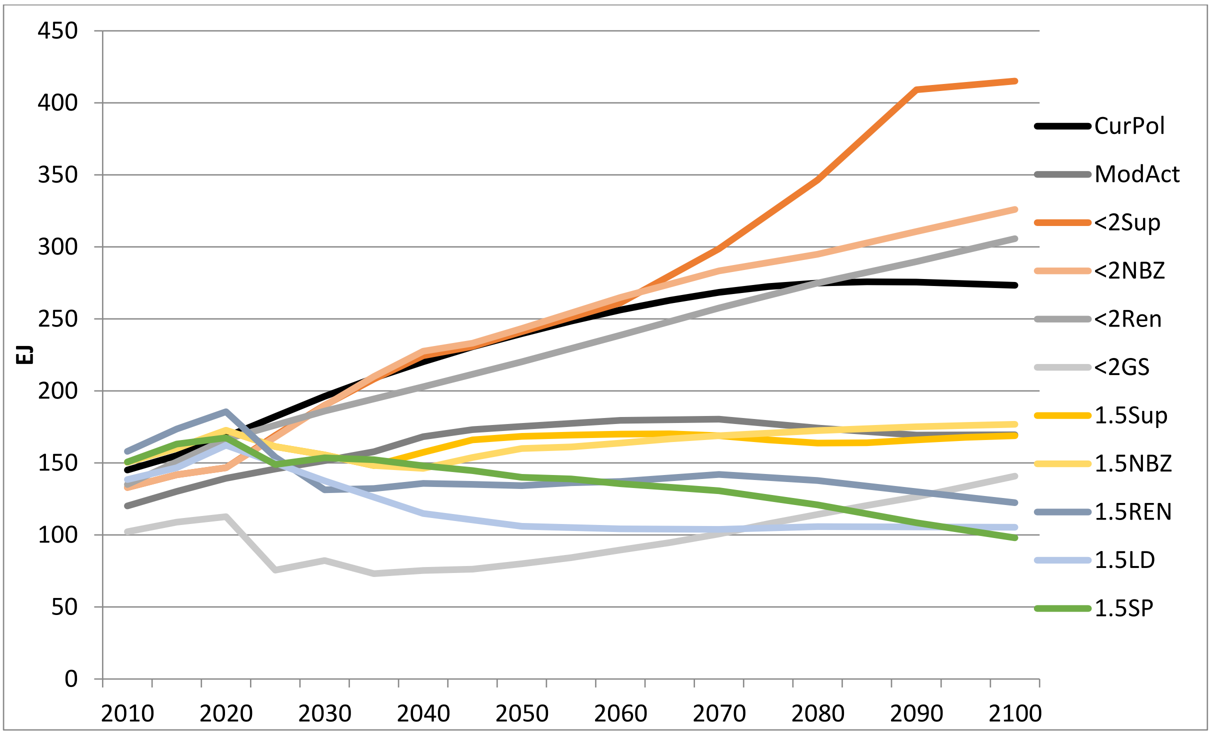
Figure 11.13 Industrial final energy (top left), CO_2 emissions (top right), energy intensity (bottom left), and carbon intensity (bottom right). Energy intensity is final energy per unit of GDP. Carbon intensity is CO_2 emissions per EJ of final energy. All indicators are indexed to 2010, where values less than 1 indicate a reduction. Scenarios are grouped based on temperature categories 3.3.2.

The main results from the analysis are (for more detailed numbers Chapter 3):

- While baseline scenarios show decline in energy intensity total final energy demand and associated industry related CO_2 emissions increase. Ambitious mitigation scenarios show up to 40% reduction in final energy demand in 2050 and 2100 with respect to 2010, but even some stringent mitigation scenarios show continued growth of final energy demand throughout the 21st century.
- Reduction of CO_2-emissions in the sector are achieved through a combination of technologies which includes nearly all options that have been discussed in this chapter (Section 11.3 and 11.4.1). However, they are big differences with regard to the intensity the scenarios implement the various options. This seems to be particularly true for CCS for industrial applications and material efficiency and material demand management (i.e. service demand, service product intensity). The latter options are still underrepresented in many global IAM.
- There are only a few scenarios (7% of assessed scenarios in chapter 3) which come out with negative CO_2-emissions throughout the century for the industry sector starting as early as the 2040s while most scenarios assessed (including the majority of 1.5°C scenarios) end up with positive CO_2-emissions. In comparison to the whole system most scenarios expect slower decrease of industry related emissions.

Reflecting the huge amount of scenarios assessed in chapter 3 it is not possible to have a more detailed discussion about the differences and the rationale behind the differences. This is one reason for the illustrative pathways approach which is used in this report and for discussion of sector specific scenarios in the following sub-section 11.4.2.2. For the illustrative pathways approach sets of scenarios have been selected which represent different levels of GHG mitigation ambitions, scenarios which rely on different key strategies or even exclude some mitigation options, delayed actions and SDG oriented pathways. For more detailed information about the selection see Section 3.3.2.

In addition to the more aggregated discussion of IAMs illustrative pathways (IPs) allow a deeper look into the scenarios. Figure 11.14 shows for the range of IPs the resulting development of final energy and the share of electricity and hydrogen in the final energy mix.



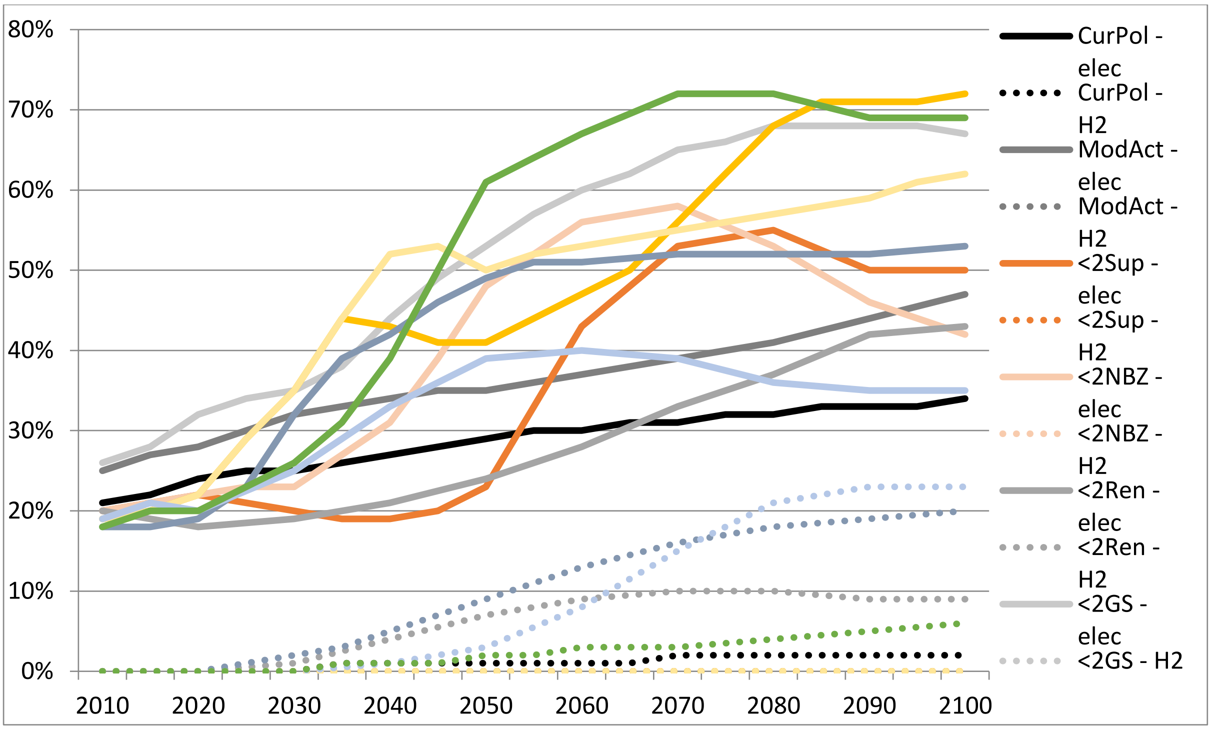


Figure 11.14 Comparison of industry sector final energy demand (upper figure) and share of electricity and hydrogen in the final energy mix (lower figure) for different mitigation scenarios representing illustrative pathways (figure based on IIASA scenario data base).

The Figure 11.14 show some general trends. First of all they indicate that with growing mitigation ambition with the exemption of scenarios where CDR is allowed (cf. < 2Sup and > 2 NBZ) final energy demand shrinks. Based on the underlying assumptions consequently scenarios 1.SSP and 1.SLD are characterised by the lowest final energy demand. With regard to electricity share in the final energy mix all scenarios show substantially increasing shares. Again the scenario < 2Sup is somehow an exemption as electricity share stays at current level by around 20% in 2050. For the ambitious mitigation scenarios, the analysis of IPs shows a range between 30% and more than 60% in 2050 with an steep increase between 2030 and 2040. On the other hand share of hydrogen shows a flatter curve with some scenarios reaching 10% in 2050. This is true for the scenarios that have a strong focus on renewable energy electrification (e.g. 1.5 Ren and <2 Ren) and where hydrogen is necessary as complementary element.

11.4.2.2 In-depth discussion and "reality" check of pathways from specific sector scenarios

Since AR5 a number of studies providing a high technological level of detail for the industry sector have been released which describe how the industry sector can significantly reduce its GHG emissions until the middle of the century. Many of these studies at the same time try to specifically reflect the particular industry sector characteristics and barriers that hinder industry to follow an optimal transformation pathway. When analysing the technologically detailed studies it has to be acknowledged that they vary in respect to different characteristics. In respect to their geographical scope, some studies analyse the prospects for industry sector decarbonisation on a global level (IEA 2017a; Grubler et al. 2018; Energy Transitions Commission 2018; Tchung-Ming et al. 2018; IEA 2020a, 2019b, 2020c; CAT 2020); while other studies e.g. European Commission (2018); Material Economics (2019) focus on individual world regions such as Europe. Numerous studies also exist for China, where most industry related emissions come, that develop consistent mid-century mitigation pathways (e.g. Zhou et al.

(2019)).[20] In regard to their sectoral scope, some studies include the entire industry sector, while others focus on selected GHG emission intensive sectors, such as steel, chemicals and/or concrete. Industry sector decarbonisation studies also differ in regard to whether they develop coherent scenarios or whether they focus on discussing and analysing selected key mitigation strategies, without deriving full energy and emission scenarios. Coherent scenarios are developed in IEA (2017a); Grubler et al. (2018); Energy Transitions Commission (2018); Tchung-Ming et al. (2018); IEA (2020a, 2019b, 2020c) on the global level and in Climact (2018); European Commission (2018); Material Economics (2019) on the European level. Recent literature analysing selected key mitigation strategies, for example IEA (2019b) and Material Economics (2019) has focused either exclusively or to a large extent on analysing the potential of materials efficiency and circular economy measures to reduce the need for primary raw materials relative to a business-as-usual development. IEA (2020a) also provides deep insides in single mitigation strategies for the industry sector, particularly the role of CCS and material efficiency.

Available scenarios from different studies reveal notable differences in their respective choices of CO_2 mitigation strategies in the industry sector.[21] To focus the discussion mainly scenarios from IEA are used here. On the one hand it has to be acknowledged that they only represent a small segment of the huge scenario family, but on the other hand this approach enables us to show the chronological development of scenarios coming from the same institution, using the same modelling approach (which allows a technology-rich analytical backcasting approach), but reflect additional requests that emerge over time. The following overview (Table 11.5) includes as its oldest scenario the 2DS scenario from the "Energy Technology Perspectives (ETP)" study (IEA 2017a), which intends to describe in great technological detail how the global energy system could transform by 2060 so as to be in line with limiting global warming to below 2°C relative to pre-industrial temperatures. In this scenario, total CO_2 emission are 74 % lower in 2060 than in 2014 (39% lower in the industry sector). The B2DS scenario of the same study intends to show how far known clean energy technologies could go if pushed to their practical limits, allowing the future temperature increase to be limited to "well below" 2°C and lowering total CO_2 emissions by 100% by 2060, making use of negative emissions (and by 75 % relative to 2014 in the industry sector).

Table 11.5 broadens the scope and includes with IEA (2019b) and IEA (2020c) additional scenarios that assess potential mitigation pathways, in the case of IEA (2020c) even to 2070. Technologies penetration assumed in the CTS scenario by 2060 allows for an industrial emission cut of 45% compared with the 2017 level and a 50% cut against projected 2060 emissions in the Reference Technology Scenario RTS from the same study (IEA 2019b). This is similar to the level of IEA's 2DS scenario. Energy efficiency improvements and deployment of BATs contribute 46% to cumulative emission reduction in 2018–2060, while fuel switch (15%), material efficiency (19%) and deployment of innovative processes (20%) provide the other part. IEA (2020a,c) which continuous the Energy

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FOOTNOTE [20] In addition, there are many other studies available which have developed country-specific, technologically detailed scenarios for industry decarbonisation e.g. (Gerbert et al. 2018) and a few which have investigated the decarbonisation prospects of individual industrial clusters (Schneider 2019), but these types of studies are not discussed here.

FOOTNOTE [21] The global mitigation scenario studies that provide detailed representations of the industry sector and are discussed in the following (IEA 2017a; Grubler et al. 2018; Energy Transitions Commission 2018; IEA 2020a, 2019b, 2020c) solely focus on CO_2 emissions, that is non-CO_2 emissions of the industrial sector are neglected. The study by (IEA 2017a) uses projections of non-CO_2 emissions as documented in the scenario database of the IPCC AR5 to determine the residual CO_2-only budgets for its scenarios' temperature targets. Non-CO_2 emissions make up only a small share of the industry sector's current CO_2-eq. emissions and include N_2O emissions (e.g. from nitric and adipic acid production), CH_4 emissions (e.g. from chemical production and iron and steel production) and various F-gases (such as perfluorocarbons from primary aluminium production and semiconductor manufacturing) (Gambhir et al. 2017; USEPA and ICF 2012). Mitigation options for these non-CO_2 emissions are discussed in (Gambhir et al. 2017).

Technology Perspectives series of the IEA include the new Sustainable Development Scenario (SDS). It describes a trajectory for emissions which is consistent with reaching global “net-zero” CO_2 emissions by around 2070 for the energy sector.[22] In 2070 following the SDS pathway the net-zero balance is reached through a compensation of the remaining CO_2-emissions (fossil fuel combustion and industrial processes still lead to around 3 $GtCO_2$) by a combination of BECCS and to a lesser degree Direct Air Capture and storage. In IEA (2020c) with the Faster Innovation Case (FIC) a second scenario is provided that shows a possibility to reach net-zero emissions level globally already in 2050, assuming that technology development and market penetration can be significantly accelerated. Innovation plays a major role in this scenario as almost half of all the additional emissions reductions in 2050 relative to the reference case in which current policies are supposed to be successfully implemented would be from technologies that are in an early stage of development and have not yet reached the market today (IEA 2020c).

FOOTNOTE [22] Following the description of IEA SDS 2020 would limit the global temperature rise to below 1.8°C with a 66% probability if CO_2 emissions remain at net zero after 2070. If CO_2 emissions were to fall below net zero after 2070, then this would increase the possibility of reaching 1.5°C by the end of the century (IEA 2020c).

Table 11.5 Perspectives on industrial sector mitigation potential (comparison of different IEA scenarios)

Reduction of direct CO_2 emissions	Scenario assumptions[i]	IEA (2017a, 2020a)		IEA (2019b)	IEA (2020a,c)	
		2030	2050	2060	2050	2070
Baseline direct emissions from industrial sector						
Reference Technology Scenario (RTS)	Industry sector improvements in energy consumption and CO_2 emissions are incremental, in line with currently implemented and announced policies and targets.	9.8 $GtCO_2$	10.4 $GtCO_2$	9.7 $GtCO_2$		
Emission reduction potential						
2°C Scenario (2DS)	Assumes the decoupling of production in industry from CO_2 emissions growth across the sector that would be compatible with limiting the rise in global mean temperature to 2°C by 2100.	-7% vs 2014[ii] -20% vs RTS	-39% vs 2014ii -50% vs RTS			
Beyond 2°C Scenario (B2DS)	Pushes the available CO_2 abatement options in industry to their feasible limits in order to aim for the "well below 2°C" target.	-28% vs 2014ii -38% vs RTS	-75% vs 2014ii -80% vs RTS			
Clean Technology Scenario (CTS)	Energy efficiency and deployment of BATs contribute 46% to cumulative emission reduction in 2018–2060; fuel switch -15%; material efficiency -19%; deployment of innovative processes -20%.			5 Gt CO_2 or -45% vs 2017 level and -50% from 2060 RTS level		
Sustainable Development Scenario 2020 (SDS 2020)	Leads to net-zero emissions globally by 2070. Remaining emissions in some sectors (including industry) in 2070 will be compensated by negative emissions in other areas (e.g. through BECCS and DAC)				~ 4.0	~ 0.6
Net Zero Emission 2050 (NZE)	Net-zero emissions are reached already by 2050 in comparison to SDS	~6.5 $GtCO_2$				

Reduction of direct CO_2 emissions	Scenario assumptions[i]	IEA (2017a, 2020a)		IEA (2019b)	IEA (2020a,c)	
		2030	2050	2060	2050	2070
Faster Innovation Case (FIC)	Achieves net-zero emissions status already by 2050 based on accelerated development and market penetration of technologies which have currently not yet reached the market.				0.8 Gt CO_2 (mainly steel and chemical industry)	

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[i] Based on bottom-up technology modelling of five energy-intensive industry subsectors (cement, iron and steel, chemicals and petrochemicals, aluminium and pulp and paper).

[ii] Industrial direct CO_2 emissions reached 8.3 GtCO_2 in 2014, 24% of global CO_2 emissions.

Source: (IEA 2017a, 2019b).

Two studies should be addressed here which complement the discussion of the IEA scenarios but are related to IEA data base at the same time.[23] The ETC Supply Side scenario builds on the ETP 2017 study, investigating additional emission reduction potentials in the emissions intensive sectors such as heavy industry and heavy-duty transport so as to be able to reach net-zero emission by the middle of the century. The LED scenario (Grubler et al. 2018) also builds on the ETP 2017 study, but focuses on the possible potential of very far-reaching efforts to reduce future energy demand.

A comparison of the different mitigation scenarios shows that they depend on how individual mitigation strategies in the industry sector are assessed, namely e.g. energy efficiency improvement, material demand reductions, the use of CCS and the final energy use of the (potentially) low or zero-carbon energy sources biomass, hydrogen (and hydrogen based products like methanol) and electricity (see Figure 11.10). The use of CCS for example is assessed very differently in the analysed scenarios, playing a major role in the B2DS scenario (3.2 $GtCO_2$ in 2050), the ETC Supply Side scenario (5.4 $GtCO_2$ in 2050) and the IEA (2020a) scenarios, while it is explicitly excluded in the LED scenario. In the latter scenario, on the other hand, considerable emission reductions are assumed to be achieved by far-reaching reductions in material demand relative to a baseline development. In other words, the analysed scenarios also suggest that to reach very strong emission reductions from the industry sector either CCS needs to be deployed to a great extent or considerable material demand reductions will need to be realised. Such demand reductions only play a minor role in the 2DS scenario and no role in the ETC Supply Side scenario. The SDS described in IEA (2020a) provides a pathway where both CCS and material efficiency play a major role (Figure 11.15). In SDS material efficiency is a relevant factor in several parts of industry, explicitly steel, cement, and chemicals. Combining the different material efficiency options including to a substantial part lifetime extension (particularly of buildings) leads to 29% less steel production by 2070, 26% less cement production, and 25% less chemicals production respectively in comparison to the reference line used in the study (Stated Policy Scenario: STEPS) (Figure 11.16). As Figure 11.15 indicates only a small part of the technologies needed for achieving the necessary reduction of emissions is already mature today. Technology development and market penetration are therefore absolutely key.

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[23] FOOTNOTE 23 Several other two global mitigation scenarios (e.g. from Tchung-Ming et al. (2018), Shell Sky Scenario from Shell (2018)) are not included in the following scenario comparison as the study's energy and emission base year data on the industry sector deviates considerably from the other three studies included in the comparison, which all use IEA data. Furthermore, unlike the other studies, Tchung-Ming et al. (2018) do not provide detailed information on the steel, chemicals and concrete sub-sectors. Not included here but worth to be mentioned are many other sector specific studies, for example Napp et al. (2019, 2014), which consider more technologically advanced decarbonisation routes for the sector.

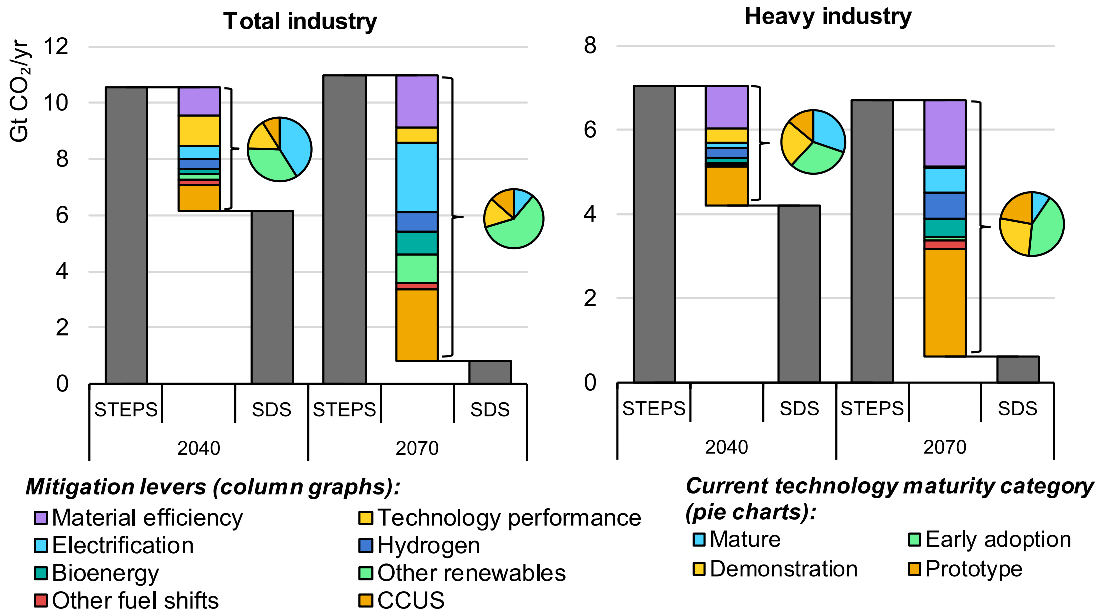


Figure 11.15 Mix of mitigation options for reducing industrial CO$_2$-emissions in the industry sector for the SDS (distinguishing between the total industry sector and heavy industry) for 2040 and 2070
(IEA 2020a)

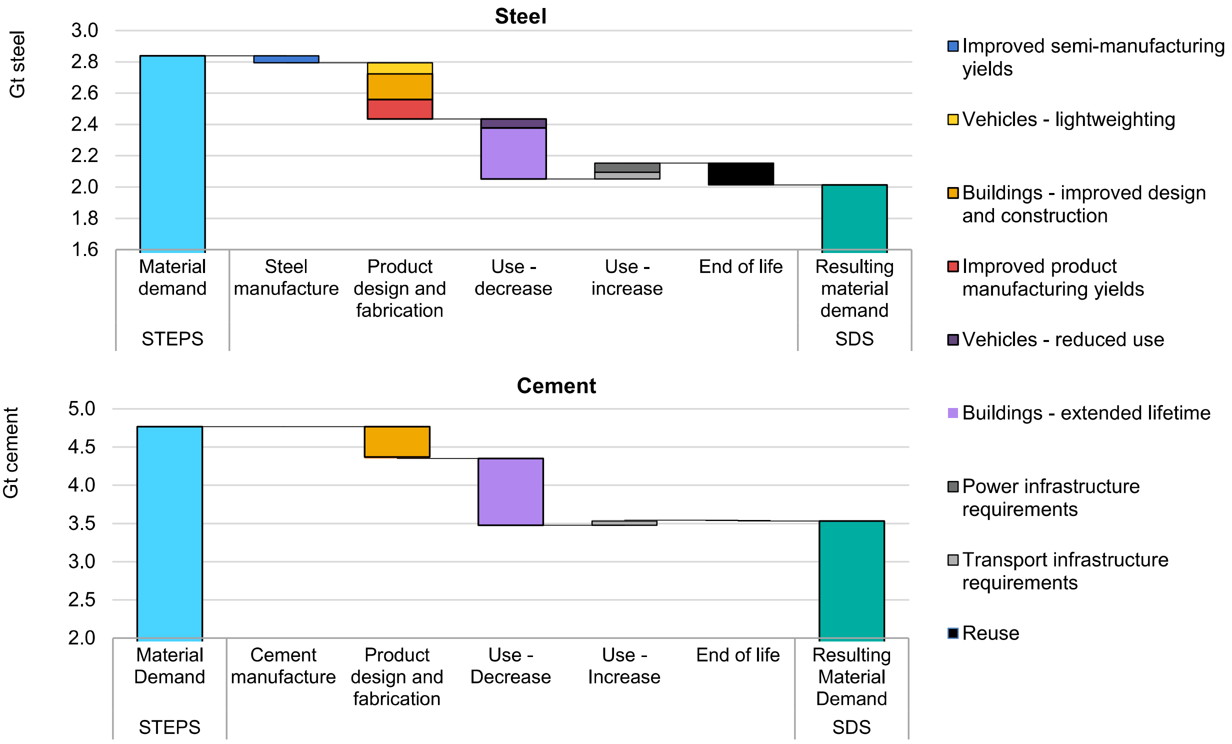


Figure 11.16 Different drivers for demand reduction in SDS compared to STEPS for the steel and cement sector
(IEA 2020a)

In all scenarios, the relevance of biomass and electricity in industrial final energy demand increases, especially in the more ambitious scenarios SDS, ETC Supply Side and LED. The sum of the shares of the low-carbon energy sources electricity, biomass and hydrogen/e-fuels in final energy demand of the industry sector increases in the analysed scenarios to some 50% or more by 2050 and above 60% in 2070, starting from roughly one third ((IEA 2017a), (IEA 2020a)). While in all scenarios electrification becomes more and more important hydrogen or hydrogen-derived fuels, on the other hand, do not contribute to industrial final energy demand by the middle of the century in 2DS and B2DS, while LED (1% final energy share in 2050) and particularly ETC supply side (25% final energy share in 2050) consider hydrogen or hydrogen-derived fuels as a significant option. This fits very well with the results for the transport sector where the two older IEA scenarios show only very little use of hydrogen. In the updated IEA scenarios hydrogen and hydrogen based fuels already play a more important role. In the SDS share in industry final energy is around 10% (IEA 2020a) and in the Faster Innovation Case around 12% (IEA 2020c) in 2050. In the latter case this is based on the assumption that by 2050 on average each year 22 hydrogen based steel plants come into operation (IEA 2020c). In SDS around 60% of the hydrogen is produced onsite via water electrolysis while the remaining 40% is generated in fossil fuel plants (methane reforming) coupled with CCS facilities.

B2DS was based on the assumption that many available abatement options in the industry sector are pushed to their feasible limits. As a consequence of the 217 Gt cumulative direct CO_2 emissions reductions in the B2DS, energy efficiency and BAT deployment contribute the largest share (42%), followed by innovative processes and CCS (37%). Switching to lower carbon fuels and feedstocks accounts for 13% of the reductions, with the remaining 8% from material efficiency strategies in manufacturing processes.

In IEA (2020h), the latest World Energy Outlook published during the AR6 period, energy efficiency improvement plays a major role in the scenario differentiation going from STEPS (stated policies) to the SDS (net-zero emissions by 2070) scenario in the 2020-2030 period, with energy efficiency gains for steel making increasing from 1.1% to 1.7% yr^{-1}, cement production from 0.6% to 1.0%, aluminium provision from 0.3% to 1.4%, and paper production from 0.75% to 1.5%. Going one step further which is described in the NZE2050 scenario (net zero emissions by 2050) annual energy efficiency improvement for steel production does not change, the number for cement increases from 1.0% to 1.75%, for aluminium from 1.3 to 2.6%, and for paper from 1.6% to 2.1%. In 2030 in many areas processes are implemented that work closely to the technological potential. Share of low carbon hydrogen in ammonia and methanol production goes from 0% in STEPS to 3.5% in SDS to 15% in NZE2050. Additionally, in NZE2050 approx. 25% of total heat used in the sector is electrified directly with heat pumps or indirectly with synthetic fuels already by 2030.

The analysis leads to the point that the relevance of individual mitigation strategies in different scenarios depends not only on a scenario's level of ambition in regard to emission mitigation. Instead, implicit or explicit assumptions about: a) the costs associated with each strategy, b) future technological progress and availability of individual technologies and c) the future public or political acceptance of individual strategies are likely to be main reasons for the observed differences between the analysed scenarios. For many energy intensive products technologies capable of deep emission cuts are already available. Their application is subject to different economic and resources constrains (incremental investments needs, products prices escalation, requirements for escalation of new low carbon power generation). Limited by those factors resulting technological mix assumed in different scenarios allows for only limited level of decarbonisation by the middle of the century.

At the same time various scenarios suggest that if constraints can be solved different pathways can be imagined that lead to significantly stronger CO_2 emission reductions in the industry sector by 2050-2070 and even net-zero emission pathways are possible, for example, the SDS and NZE2050 scenarios in the 2020 IEA publications. However, so far there is no consensus on the most plausible or most desirable mix of key mitigation strategies to be pursued. In addition it has to be stressed that suitable pathways are very country-specific and depend on the economic structure, resource potentials, available sectoral technology pathways, and political preferences and processes of the country or region in question (Bataille 2019a).

The key conclusion of the scenario analysis is that significant cuts in GHG emissions and even close to net zero emissions from the largest sources (steel, plastics, ammonia, and cement) could be achieved by 2050 by deploying already well known options. However, it becomes clear that a significant shift is needed from a transition process in the past mainly based on marginal (incremental) changes (with a strong focus on energy efficiency efforts) to a transformational change. To limit the barriers that are associated with transformational change, besides overcoming the valley of death for technologies or processes with breakthrough character, it is required to carefully identify structural change processes which are connected with substantial changes of the existing system (including the whole process chain). This has to be done at an early stage and has to be linked with considerations about preparatory measures which are able to flank the changes and to foster the establishment of new structures (Section 11.6). Following transformational change could also lead to relocation of production processes, if for instance steel production moves partly to countries like Australia or South Africa substantial iron ore reserves and a large potential for low-cost renewable electricity potential (Gielen et al. 2020; Bataille 2019a). Those countries could become exporters of hydrogen based direct reduced iron (DRI), with the consequence that iron reduction value added in the source country increases while it decreases in the DRI importing countries, even if the final steel EAF production process remains there, while noting most value added in steel manufacturing is not in the primary but intermediate and final firm & consumer product production.

In addition, the right sequencing of the various mitigation options and building appropriate bridges between the different strategies are important factors. Rissman et al. (2020) proposes three phases of technology deployment for the industry sector: (1) energy/material efficiency improvement (mainly incremental improvements) and electrification in combination with demonstration projects for technologies that will be important in subsequent phases (2020-2035), (2) structural shifts based on technologies which reach maturity in phase (1) as CCS and alternative materials (2035-2050) (3) widespread deployment for processes and energy technologies that are nascent today like hydrogen applications. There are no strong boundaries between the different phases and all phases have to be accompanied by suitable policies like R&D programs and market incentives.

Taking the steel sector as an illustrative example, sector specific scenarios examining the possibility to reach GHG reduction beyond 80% indicate that robust measures comprise direct reduction of iron (DRI) with hydrogen in combination with efforts to further close the loops and increase availability of scrap metal (reducing the demand for primary steel). As hydrogen based DRI might not be a fully mature technology before 2035 (depending on further developments of the policy framework and technological progress), risk of path dependencies has to be taken into consideration when reinvestments in existing production capacities will be required in the coming years. For existing plants, implementation of energy efficiency measures (e.g. utilisation of waste heat, improvement of pumps) could build a bridge for further mitigation measures, but have only limited unexhausted potential. As many GHG mitigation measures are associated with high investment costs and missing operating experience, a step by step implementing process might be an appropriate strategy to avoid investment leakage (given the mostly long operation times, investment cycles have to be used so as not to miss opportunities) and to gain experience. In the case of steel, companies can start with the integration of a natural gas based reduction plant feeding the reduced iron to an existing blast furnace, blending and later replacing the natural gas by with fossil with CCS or electrolysis based hydrogen in a second stage, and later transitioning to a full hydrogen DRI EAF or molten oxide electrolysis EAF (https://salcos.salzgitter-ag.com), all without disturbing the local upstream and downstream supply chains.

It is worth mentioning the flexibility of implementing transformational changes not the least depends on the age profile and projected longevity of existing capital stock (Tong et al. 2019), especially the willingness to accept high GHG intensity investments intentionally stranded, or stranded by markets evolving in a low carbon direction. This is a relevant aspect in all producing countries, but particularly in those countries with a rather young industry structure (i.e. comparative low age of existing facilities on average). Figure 11.17 describes this aspects for China using the survival rate as a proxy (Tong et al. 2019).

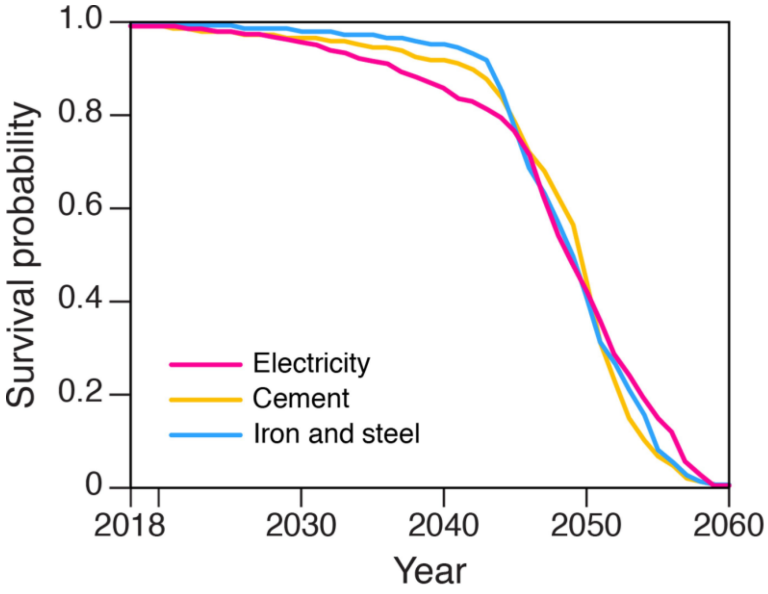
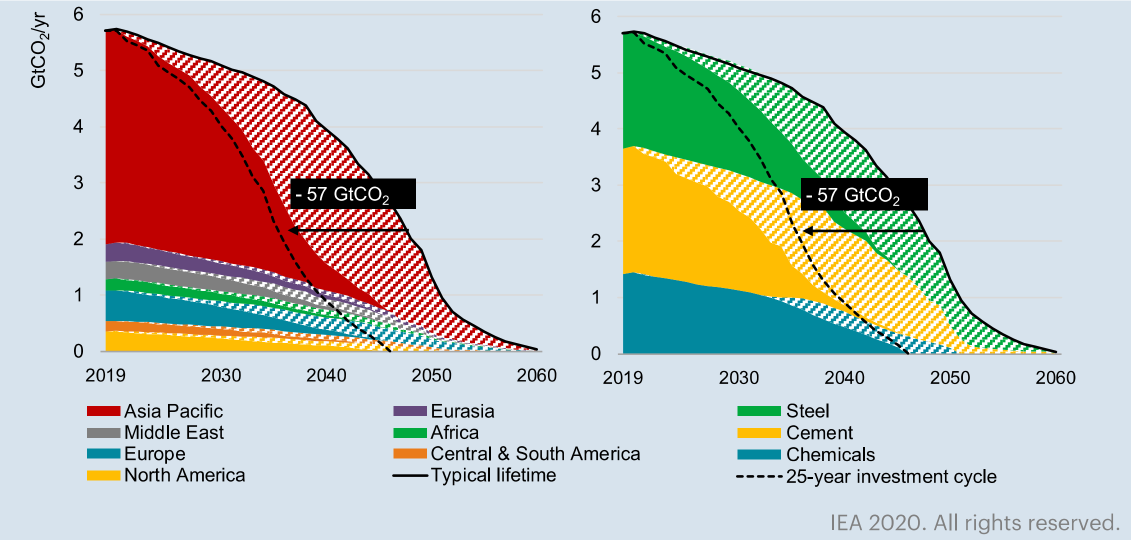


Figure 11.17 Survival curves for power and major industries in China (survival probability assesses the probability that a specific plant is still in operation at a specific point in time)

In general, early adoption of new technologies plays a major role. Considering the long operation time (lifetime) of industrial facilities (e.g. steel mills, cement kilns) early adoption of new technologies is needed to avoid lock-in. For the SDS 2020 scenario, IEA (2020i) calculated the potential cumulative reduction of CO_2-emissions from the steel, cement and chemicals sector to be around 57 $GtCO_2$ if production technology is changed at its first mandatory retrofit, typically 25 years, rather than at 40 years, which would be the typical retrofitted lifetime of a steel, cement or chemicals plant (Figure 11.18).



Notes: Typical lifetimes for steel, cement and chemicals assets are 40 years and 30 years respectively. In the "25-year investment cycle" case considered here, all assets are replaced by or converted to low-carbon alternatives at the first 25-year refurbishment point after the new technologies are assumed to be commercially available.

Figure 11.18 Avoiding "lock in" of CO_2 emission at the next investment point in heavy industrial sectors in the Sustainable Development Scenario

(IEA 2020i)

Another important finding of the scenario analysis is that available strategies can be packaged in different ways (Material Economics 2019; UKCCC 2019a). There is no single option that can do the whole job, so all technological options will likely have to be mobilised. It should be noted that although seeing progress, material efficiency respectively demand management is still not very well represented in the scenario literature. Besides IEA (2020a) two of the few exceptions are Material Economics (2019) for the EU and Zhou et al. (2019) for China. Zhou et al. (2019) describes consistent mitigation pathways for China which go beyond the official Chinese target for 2030 and 2050. For 2050 CO_2-emissions in the Reinventing Fire scenario are at a level that is 42% below the countries emissions in 2010. For achieving that goal reduction of emissions in the industry sector play an important role. As expected the major part of CO_2-emissions reduction relative to the Reference Scenario is associated to energy-efficiency improvement (53%). Around 8% are related to decarbonisation of the energy mix, 24% is based on an accelerated structural shift and almost 13% related to production demand reduction (mainly based on extension of building and infrastructure lifetime as well as reduction of material losses in the production process and application of higher quality materials particularly high-quality cement) (Zhou et al. 2019).

For the three most important sub-sectors in industry Table 11.6 shows results from Material Economics (2019) for the EU. The combination of circularity, material and energy efficiency, fossil and waste fuels mix, electrification, hydrogen and biomass use varies from scenario to scenario with no of these options ignored. On the contrary, for CCS the authors set a strong default, in all scenarios CCS is not included as a mitigation option. Scenario studies for single European countries like Germany support the Material Economics (2019) analysis und show that high mitigation standards and even net-zero emissions can be reached without application of CCS and with limited contribution of synthetic carbon neutral fuels. In those scenarios described in Samadi and Barthel (2020) a huge part of the mitigation is based on material efficiency, circular economy, material substitution but even partly life-style changes.

Table 11.6 Contribution to emission reduction for industrial deep decarbonisation from different pathways[24]

	Steel	Plastics	Ammonia	Cement
	Contribution to emission reduction (%) (range represents the three different pathways of the study)			
Circularity	5–27	7–27		10–44
Energy efficiency	5–23	2–9		1–5
Fossil fuels and waste fuels	9–41	0–27		0–51
Decarbonised electricity	36–59	16–22		29–71
Biomass for fuel or feedstock	5–9	18–22		0–9
End-of-life plastic		16–35		
	Required electrification level			
Growth of electricity demand (times compared with 2015)	3–5	3–4		2–5
	Investments and production costs escalation			
Investment needs growth (% versus BAU)	25–65	122–199	6–26	22–49
Cost of production (% versus BAU)	2–20	20–43	15–111	70–115

Source: Material Economics (2019).

According to the analysis net-zero emission pathways in the reflected sectors require significantly higher investments compared to BAU, 25% to 65% for steel, 122-199% for plastics, 6-26% for Ammonia and 22-49% for cement (Material Economics 2019), although the differences between the sectors are quite significant.

Other sector specific scenario analyses indicate that pathways to net-zero CO_2 emissions in the emissions intensive sectors are not only technically possible by mid-century, but can be realised with limited additional costs. According Energy Transitions Commission (2018) deep decarbonisation from four major industry subsectors (plastics, steel, aluminium and cement) is achievable: CO_2 emissions could be reduced significantly on a global level with 2015–2050 incremental capital investments amounting 5.5–8.4 USD trillion, representing about 0.1% of aggregate GDP over that period. The UKCCC (2019a) study assesses that total incremental costs (compared to a theoretical scenario with no climate change policy action at all) for cutting industrial emissions by 90% by 2050 is 0.2% of expected 2050 UK GDP (UKCCC 2019a). The additional costs of a transition to a net-zero emission industry in the EU are estimated to be an annual 0.2% of EU GDP by 2050 and additional investment is 0.2% of gross fixed capital formation (Material Economics 2019). The IEA (2020a) indicates the required annual incremental global investment in heavy industry is approximately USD 40 billion 2019 USD yr^{-1} moving from the stated policies (STEPS) to the SDS scenario 2020-2040, rising to USD 55 billion yr^{-1} 2040-2070, effectively 0.05-0.07% of global annual GDP today. Global gross fixed capital formation averages, with wide variance, 2-4% of GDP, meaning global GFCF would need to be increased by just over 2% before allowing for growth in GDP.[25]

FOOTNOTE [24] Note: In the described scenarios CCS was not taken into consideration as a mitigation option by the authors.

FOOTNOTE [25] (The World Bank 2020b): https://data.worldbank.org/indicator/NE.GDI.FTOT.KD.ZG

11.4.3 Cross sectorial interactions

Industry is a very complex web of sectors, sub-sectors and intersectoral interaction and dependence, with associated mitigation opportunities and co-benefits and costs (OECD 2019b) getting and meeting demand for materials, equipment and other products from other sectors. Embedded GHGs emissions in stocks accumulated as well as operational emissions to run this stock in other sectors much depends on emission intensity, weight and other features of provided industrial products. In a zero-carbon power world, with industry lagging behind in the decarbonisation of high-temperature processes and feedstock, it may replace energy sector as generator of indirect emissions embodied in capital stock (Lamb et al. 2021). This issue of material efficiency improvements and additional material demand originating from decarbonisation in other sectors attracts much attention since AR5 (IEA 2019b; IRP 2020).

High entropy ('exhausted' heat) is generated when heat consumption conditions (temperature and volume) do not match industry demand allowing extra heat to be exported from the facility. In Denmark 1.36 TWh of district heating demand (5.1% of total demand) could be met with industrial waste heat (Bühler et al. 2017). In Austria up to 35% of total heat demand can be met with industrial waste heat (Karner et al. 2016). European case studies argue that urban CO_2 emissions could be halved by urban clustering, allowing up to 100% excess heat utilisation from industry (Karner et al. 2018). According to estimates for the Swedish chemical industry, there are many cases where 30-50% of waste heat generated on-site is recovered and reused or sold, with paybacks below 3 years (Eriksson et al. 2018). The UKCCC (2019b) indicates that industrial clustering may be essential for achieving the necessary efficiencies of scale and to build the infrastructure needed for: industrial electrification; carbon capture, transport and disposal; hydrogen production and storage; heat cascading between industries and to other potential heat users (e.g. residential and commercial buildings).

Clinker substitution in the cement industry is well established way of reducing CO_2 emissions, with potential for up to 95% substitution with blast furnace slag in limited applications and up to 45% with other substitutes applications (Scrivener et al. 2018). Comparing with present practice there is still room for improvement. In Iran, CO_2 emissions and energy costs could be reduced by 13% and 11%, respectively by such substitution (Jokar and Mokhtar 2018). Better building design using combination of different materials (e.g. steel; carbon, wood, and hemp fibres) can reduce cement and concrete use by 20–30% safeguarding building performance (Bataille et al. 2018a; Scrivener et al. 2018). Using wood as substitute for steel and concrete could save 14-31% of global CO_2 embodied in new buildings emissions by using 34-100% of the world's sustainable timber resources (Oliver et al. 2014), although the quantity should be traced because of the wood scarcity for transition pathways. High-rise buildings are increasingly being built using Cross Laminated Timber (CLT) instead of concrete and steel, with the additional benefit of shortening construction periods (examples: 18-storey UBA building in Canada; 18-storey mixed use building in Norway; 10-storey apartment in Melbourne, Australia.) Although CLT is superior to ordinary timber in terms of fire resistance, but when widely used, risk should be recognised as an issue.

As application of variable renewable energy (e.g. wind at night) requires industries provide growing demand or storage flexibility comparing with today's practice (Schriever and Halstrup 2018). The combination of CHPs, thermal storage, and solar power can reduce the annual energy cost for industry on average by 8.3% (Scheubel et al. 2017). To become more valuable resource demand response industry has to integrate deeper with the grid (Shoreh et al. 2016). Vogl et al. (2018a) suggest that hydrogen DRI EAF steel plants could adjust electrolytic hydrogen production schedule for variable electricity production loads while storage hydrogen can be smoothly used for steel making. The IEA (2019e) Future of Hydrogen reports suggests that hydrogen production and storage networks could from locations with already existing hydrogen production & storage, like refineries on seacoasts. These locations, if collocated with other industries, could be ideal for system load balancing and demand response, and in case of district heating systems - for heat cascading.

11.4.3.1 Environmental pressure on industry

The environmental awareness that investors, shareholders and customers demand from companies has been increasing steadily. It is reflected in growing number of environmental management accounting and reporting schemes (e.g., the Carbon Disclosure Project, Environmental Product Declarations and others, Qian et al. (2018)) requiring companies to disclose both direct and indirect CO_2 emissions, and creating implicit GHG liabilities. This requires harmonised and widely accepted methods for environmental and carbon footprint accounting. From an investor perspective there are both physical risks (e.g., potential damages from climate change to business) and transition risks (e.g., premature devaluation of assets driven by new policies and technologies deployment and changes in public and private consumer preferences (NGFS 2019a). Accompanied by reputational risks this leads to increased attention to Sustainable and Responsible Investment (SRI) principles (NGFS 2019b).

11.4.3.2 Net GHG intensity of industrial production supply chains and the economic structure at large

By estimating the energy and process emissions profile of a given manufacturing process, it is possible to estimate the energy consumption, CO_2 and other GHG emissions required for each part/machine/equipment production, thereby enabling the net GHG impact to be assessed (Uluer et al. 2016; Tanaka et al. 2018; LCS 2018b). For example, the global shift in production due to globalisation is causing delocalisation to less efficient production areas, resulting in an increase in energy use of 1 to 6% depending on the sector (Forin et al. 2018). Japan's Keidanren has started to promote a scheme by each industry to start efforts to reduce GHG in the global value chain, including material procurement, product use stages, and disposal, regardless of geographical restrictions, and the quantitative visualisation (Keidanren 2018).

11.4.4 Links to climate change, mitigation, adaptation

Sectors that are particularly vulnerable to climate change include agriculture, forestry, fisheries and aquaculture, and their downstream processing industries (IPCC 2021a). Many of the energy intensive industries are located to have access to fresh water (e.g., pulp and paper) or sea transport (e.g., petrochemicals). Risks of major concern for industry include disrupted supply chains and energy supplies due to extreme weather events, as well as risks associated with droughts, floods with dirty water, sea level rise and storm surges (IPCC 2021b). Adaptation measures may in turn affect the demand for basic materials (e.g., steel and cement), for example, to build sea walls and protect infrastructure but we have not found any estimates of the potential demand. Increased heat stress is unsafe for outdoor labourers and can reduce worker productivity, for example, in outdoor construction, resource extraction and waste handling (IPCC 2021c).

11.5 Industrial infrastructure, policy, and SDG contexts

11.5.1 Existing industry infrastructures

Countries are on different economic development paths. Some are already industrialised while emerging economies have yet to build basic infrastructure to allow for better mobility, housing, sanitation, and other services. The stock of material already in use and available in each country therefore differs, and transition pathways will require a different mix of strategies, depending on each country's material demand and stock of long-lived assets already in place. Industrialised economies have greater opportunities for reusing and recycling materials, and emerging economies have greater opportunities to avoid carbon lock-in. The IEA projected that more than 90% of the additional 2050 production of key materials will be in non-OECD countries (IEA 2017a). As incomes rise in emerging economies, the industry sector will grow in tandem to meet the increased demand for the manufactured goods and raw materials essential for infrastructure development. The energy needed to support this growth is likely to constitute a large portion of the increase in the emerging economies' GHG emissions in the future unless new low carbon pathways are identified and promoted.

Emissions are typically categorised by the territory, sub-sector or group of technologies from which they emanate. An alternative sub-division is that between existing sources that will continue to generate emissions in the future, and those that are yet to be built (Erickson et al. 2015). The rate of emissions from existing assets will eventually tend to zero, but in a timeframe that is relevant to existing climate and energy goals, the cumulative contribution to emissions from existing infrastructure and equipment is likely to be substantial. Aside from the magnitude of the contribution, the distinction between emissions from existing and forthcoming assets is instructive because of the difference in approach to mitigation that may be necessary or desirable in each instance to avoid getting locked into decades of highly carbon-intensive operations (Lecocq and Shalizi 2014).

Details of the methodologies to assess 'carbon lock-in' or 'committed emissions' differ across studies but the core components of the approaches adopted are common to each: an account of the existing level of emissions for the scope being assessed is established; this level is projected forward with a decay function that is informed by assessments of the current age and typical lifetimes of the underlying assets. From this, a cumulative emissions figure is calculated. The future emissions intensity of the assets is usually assumed to remain constant, implying that nothing is done to retrofit mitigating technologies (e.g. carbon capture) or alter the way in which the plant is operated (e.g. switching to an alternative fuel). While the quantities of emissions derived are often referred to as 'committed' or 'locked-in', their occurrence is of course dependent on a suite of economic, technology and policy developments that are highly uncertain.

Data on the current age profile and typical lifetimes of emissions-intensive industrial equipment are difficult to procure and verify and most of the studies conducted in this area contain little detail on the global industrial sector. Two recent studies are exceptions, both of which cover the global energy system, but contain detailed and novel analysis on the industrial sector (Tong et al. 2019; IEA 2020a). Tong et al. (2019) use unpublished unit-level data from China's Ministry of Ecology and Environment to obtain a more robust estimate of the age profile of existing capacity in the cement and iron and steel sectors in the country. The IEA (2020a) uses proprietary global capacity datasets for the iron and steel, cement and chemicals sectors, and historic energy consumption data for the remaining industry sectors as a proxy for the rate of historic capacity build-up.

Both studies come to similar estimates on the average age of cement plants and blast furnaces in China of around 10-12 years old, which are the figures for which they have overlapping coverage. Both studies also use the same assumption of the typical lifetime of assets in these sectors of 40 years, whereas the IEA (2020a) study uses 30 years for chemical sector assets and 25 years for other industrial sectors. The studies come to differing estimates of cumulative emissions by 2050 from the industry sector; 196 $GtCO_2$ in the IEA (2020a) study, and 162 $GtCO_2$ in the Tong et al. (2019) study. This difference is attributable to a differing scope of emissions, with the IEA (2020a) study including industrial process emissions (which for the cement sector in particular are substantial) in addition to the energy-related emissions quantities accounted for in the Tong et al. (2019) study. After correcting for this difference in scope, the emissions estimates compare favourably.

The IEA (2020a) study provides supplementary analysis for the industry sector, examining the impact of considering investment cycles alongside the typical lifetimes assumed in its core analysis of emissions from existing industrial assets. For three heavy industry sectors – iron and steel, cement and chemicals – the decay function applied to emissions from existing assets is re-simulated using a 25-year investment cycle assumption. This is 15 years shorter than the typical lifetimes assumed for assets in the iron and steel and cement sectors, and 5 years shorter than that considered for the chemical sector. The shorter timeframe for the investment cycle is a simplified way of representing the intermediate investments that are made to extend the life of a plant, such as the re-lining of a blast furnace, which can occur multiple times during the lifetime of an installation. These investments can often be similar in magnitude to that of replacing the installation, and they represent key points for intervention to reduce emissions. The findings of this supplementary analysis are that around 40%, or 60 $GtCO_2$, could be avoided by 2050 if near-zero emissions options are available to replace this capacity, or units are retired, retro-fitted or refurbished in a way that significantly mitigates emissions (e.g. retro-fitting carbon capture, or fuel or process switching to utilise bioenergy or low-carbon hydrogen).

The cost of retrofitting or retiring a plant before the end of its lifetime depends on plant specific conditions as well as a range of economic, technology and policy developments. For industrial decarbonisation it may be a greater challenge to accelerate the development and deployment of zero emission technologies and systems than to handle the economic costs of retiring existing assets before end of life. The 'lock-in' also goes beyond the lifetime of key process units, such as blast furnaces and crackers, since they are typically part of large integrated plants or clusters with industrial symbiosis, as well as infrastructures with feedstock storage, ports, and pipelines. Individual industrial plants are often just a small part of a complex network of many facilities in an industrial supply chain. In that sense, current assessments of 'carbon lock-in' rely on simplifications due to the high the complexity of industry.

Conditions are also sub-sector and context specific in terms of, for example, mitigation options, industry structures, markets, value chains and geographical location. For example, the hydrogen steelmaking joint venture in Sweden involves three different companies headquartered in Sweden (in mining, electricity and steelmaking, respectively), two of which are state owned, with a shared vision and access to iron ore, fossil-free electricity and high-end steel markets (Kushnir et al. 2020). In contrast, chemical clusters may consist of several organisations that are subsidiaries to large multinational corporations with headquarters across the world, that also compete in different markets. Even in the presence of a local vision for sustainability this makes it difficult to engage in formalised collaboration or get support from headquarters (Bauer and Fuenfschilling 2019).

Furthermore, it is relevant to consider also institutional and behavioural lock-in (Seto et al. 2016). This can be seen from two sides. On one side, existing high-emitting practices may be favoured through formal and informal institutions (e.g., regulations and social norms or expectations, respectively), for example around building construction and food packaging. On the other side, mitigation options may face corresponding institutional barriers. Examples include how cars are conventionally scrapped (i.e., crushed, leading to copper contamination of steel) rather than being dismantled, or slow permitting procedures for new infrastructure and industrial installations for reducing emissions.

11.5.2 Current industrial and broader policy context

The basic motivation for industrial policy historically has been economic development and wealth creation. Industrial policy can be progressive and promote new developments or be protective to help infant or declining industries. It may also involve the phase-out of industries, including efforts to retrain workers and create new jobs. Industrial policy is not one policy intervention but rather the combined effects of many policy instruments that are coordinated towards an industrial goal. Industrial policies can be classified as being either vertical or horizontal depending on whether singular sectors or technologies are targeted (e.g. through R&D, tariffs and subsidies) or the whole economy (e.g., education, infrastructure, and general tax policies). The horizontal policies are not always thought of as industrial policy although taking a broad view, including policy coordination and institution building, is important for industrial policy to be effective (see e.g., Andreoni and Chang (2019)).

In the past ten years there has been increasing interest and attention to industrial policy. One driver is the desire to retain industry or re-industrialise in regions within Europe and North America where industry has declining shares of GDP. The need for economic growth and poverty eradication is a key driver in developing countries. An important aspect is the need to meet the "dual challenge of creating wealth for a growing population while staying within planetary boundaries" (Altenburg and Assman 2017). The need for industrial policy that supports environmental goals and green growth has been analysed by, for example, Rodrik (2014); Aiginger (2014); Warwick (2013) and Busch et al. 2018). Similar ideas are taken up in OECD reports on green growth (OECD 2011) and system innovation (OECD 2015). However, these approaches to green industrial policy and innovation tend to focus on opportunities for manufacturing industries to develop through new markets for cleaner technologies. They rarely include explicit attention to the necessity of zero emissions and the profound changes in production, use and recycling of basic materials that this entails. This may also involve the phase-out or repurposing of industries that currently rely on fossil fuels and feedstock.

The policy implications of zero emissions for heavy industries are relatively unexplored although some analyses in this direction are available (e.g., Åhman et al. (2016; Philibert and IEA (2017); Wesseling et al. (2017a); Bataille et al. (2018b); Wyns et al. (2019)). For industry, there has been a long time focus on energy efficiency policies through voluntary and negotiated agreements, energy audit schemes and various programs targeting industry (IPCC 2014). Since AR5, interest in circular economy policies has increased and they have become more prevalent across regions and countries, for example, in EU, China, U.S., Japan and Brazil (e.g. Geng et al. (2019); McDowall et al. (2017); Ranta et al. (2018)). For electrification and CCUS, efforts are nascent and mainly focused on technology development and demonstrations. Policies for demand reduction and materials efficiency are still relatively unexplored (e.g., IEA (2019b); Pollitt et al. (2019)). Since zero emissions in industry is a new governance challenge it will be important to build awareness and institutional capacity in industrialised as well as developing countries.

In the context of climate change policy, it is fair to say that industry has been sheltered from the increasing costs that decarbonisation may entail. This is particularly true for the energy and emissions intensive industries where cost increases and lost competitiveness may lead to carbon leakage, i.e., that industry relocates to regions with less stringent climate policies. These industries typically pay no or very low energy taxes and where carbon pricing exists (e.g., in the European Trading Scheme) they are sheltered through free allocation of emission permits and potentially compensated for resulting electricity price increases. For example, Okereke and McDaniels (2012) shows how the European steel industry was successful in avoiding cost increases and how information asymmetry in the policy process was important for that purpose (Okereke and McDaniels 2012).

11.5.3 Co-benefits of Mitigation Strategies and SDGs

The deployment of climate change mitigation strategies is primarily influenced by its costs and potential, but also by other broader sustainable development factors such as the SDGs. Climate change mitigation actions therefore impacts economic, social and environmental goals and are classified as co-benefits when the impacts are positive or risk when the mitigation strategies results negative impacts. Potential co-benefits can therefore serve as additional drivers, whiles risks can inhibit the deployment of available strategies. Actions taken to mitigate climate change can also have direct interactions with SDGs, involving both positive synergies and negative trade-off (Fuso Nerini et al. 2019).

Given the range of stakeholders involved in climate actions and their interest and priorities, the nature of co-benefits and risk can affect decision-making processes and behaviour of stakeholders. As such, these co-benefits form an important variable that inform the adoption of climate change mitigation strategies. Indeed, while reporting that co-benefits, in addition to avoided climate change costs, is commonly overlooked in policymaking, Karlsson et al. (2020) further reported after a review of 239 peer-reviewed articles that diverse co-benefits categories covering air quality, diet, physical activity, soil and water quality, biodiversity, economic performance, and energy security dominate the literature. Co-benefits which may be derived from specific climate mitigation actions outlined in this chapter are further elaborated upon in the subsequent sub-sections.

Figure 11.19 maps the five technological developments and options assessed in this chapter to positive co-benefits to the 17 SDGs to highlight potential co-benefits.

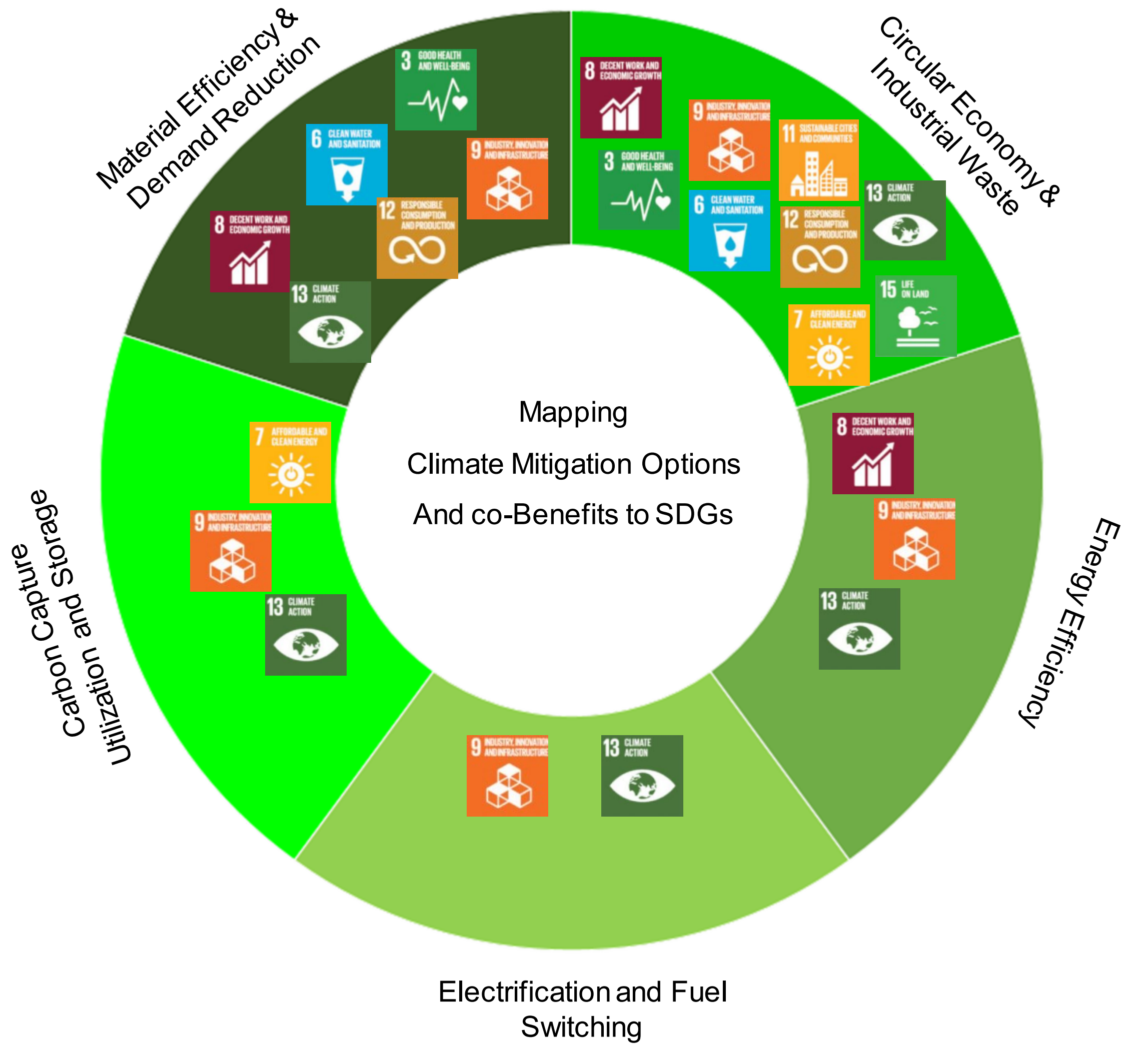


Figure 11.19 Mapping climate mitigation options and co-benefits to SDGs

11.5.3.1 SDGs co-Benefits through Material Efficiency and Demand Reduction

While the full potential of material efficiency as an energy and climate instrument for heavy industries is yet to be fully acknowledged and leveraged (Gonzalez Hernandez et al. 2018a), its co-benefit potential remains significant. For instance, materials efficiency provides opportunities to reduce the pressures and impacts on environmental systems (SDG 6) (Olivetti and Cullen 2018). In addition, material efficiency strategies usually require new business models. As such this provides potential co-benefits of increased employment and economic opportunities (SDG 8) through new business models such as through the circular economy (Genovese et al. 2017; Schroeder et al. 2019) and industrial symbiosis (Guo et al. 2016). Material efficiency also provides co-benefit opportunities through infrastructural development (SDG 9) (Mathews et al. 2018) to support the range of potential material efficiency strategies including light-weighting, re-using, re-manufacturing, recycling, diverting scrap, extending product lives, using products more intensely, improving process yields, and substituting materials (Allwood et al. 2011). Worrell et al. (2016) also emphasises how material efficiency improvements, in addition to limiting the impacts of climate change help deliver sustainable production and consumption (SDG 12) co-benefits through environmental stewardship.

In terms of demand reduction, whiles sustainable consumption delivered through reduction in the demand for new products has direct impact as a climate change mitigation strategy, it is also positively associated with happiness and life satisfaction (Guillen-Royo 2019). In addition, both Binder and Blankenberg (2017) and Dhandra (2019) showed that sustainable consumption is positively related to life satisfaction and subjective well-being (SDG 3). As co-benefits, the reduction in consumption and demand for products and services also generates a reduction in post-consumption waste (SDG 6) (Govindan 2018; Minelgaitė and Liobikienė 2019). Despite these co-benefits, there is the risk of negative impact on economic growth through reduced sales taxes (Thomas 2003). As reported by Sudmant et al. (2018) and Dawkins et al. (2019), reduction in consumption also contributes to enhancing SDG 13 through climate action.

11.5.3.2 SDGs co-Benefits from Circular Economy and Industrial Waste

While the circular economy first emerged due to aspirations for waste avoidance, resource depletion, closed-loop recycling, etc., it has now evolved as a tool for a broader systemic national policy due to its potential wider benefits (Geng et al. 2013). It represents new business models that encourage design for re-use and improve materials recovery and so represents a departure from the traditional linear production and consumption systems, with a wide range of potential co-benefits to wider SDG goals.

Genovese et al. (2017) for instance demonstrated the clear advantages from an environmental and responsible consumption and production point of view (SDG 12). Many studies have outlined new business models based on the circular economy model and so have highlighted novel economic model that fosters sustainable economic growth and the generation of new jobs (SDG 8) (Antikainen and Valkokari 2016), global competitiveness and innovation in business and the industrial sector (Pieroni et al. 2019), such as its potential synergies with industry 4.0 (Garcia-Muiña et al. 2018). Following a review of the literature, Schroeder et al. (2019) also identified linkages between circular economy practices and SDGs based on a relationship scoring system and highlighted SDG 6 (Clean Water and Sanitation), SDG 7 (Affordable and Clean Energy), SDG 8 (Decent Work and Economic Growth), SDG 12 (Responsible Consumption and Production), and SDG 15 (Life on Land) as SDGs which strongly benefit from circular economy practices. With the potential to impact on all levels of the value chain (micro, meso- and macro-level of the economy), circular economy has also been identified as key industrial strategy to managing waste across sectors.

Through the management of industrial waste using circular economy practices, studies such as Geng et al. (2012) and Bonato and Orsini (2017) have pointed out co-benefits to SDGs beyond clear environmental and economic benefits, highlighting how such it also benefits SDG 3 and 11 through improved social relations between industrial sectors and local societies, and improved public environmental awareness and public health level.

11.5.3.3 SDG co-Benefits from Energy Efficiency

Improvements in the energy efficiencies can be delivered through a number of technical options and policies, which delivers in addition to energy and resource saving, other socio-economic and environmental co-benefits. At the macro-level, this can include clean energy security (SDG 7) delivered through low carbon energy systems (Fankhauser and Jotzo 2018). Whiles, a vast majority of the extant literature including Sari and Akkaya (2016); Garrett-Peltier (2017) and Allan et al. (2017) amongst others points out that energy efficiency improvements can deliver superior employment opportunities (SDG 8) in a green economy, some limited number of studies have also reported that it can negatively impact on employment (Costantini et al. 2018) and economic growth (Mahmood and Ahmad 2018). Energy efficiency has also been reported to deliver positive changes in productivity (SDG 8) through industrial innovation (SDG 9) (Kang and Lee 2016; Bashmakov 2019).

11.5.3.4 SDGs co-Benefits from Electrification and Fuel Switching

Electricity as an energy carrier is intermediate steps in the energy-supply chain between primary sources and end-use applications and so provides convenience and technical means to pursue electrification and fuel switching with renewables. Indeed, new technological innovations together with sustainable development policy initiatives such as the SDGs are driving energy transition to cleaner electricity through fuel switching. With energy being such an important cross-cutting issue to sustainable development, some SDGs, such as SDG 1, 3, 4, 5 (UNDP 2018) are co-beneficiaries to using electrification and fuel switching as a climate action mitigation option.

11.5.3.5 SDGs co-Benefits from CCU, CCS, feedstock and biogenic carbon

Beyond the very direct links between energy and climate change, energy also represents a cross-cutting issue, which is central to all of the SDGs and so fundamental to development. Against this background, given the efforts to transition to cleaner energy systems and sustainable sources of energy, the role to be played by such systems in terms of its co-benefit contributions to SDGs also remains critical. CCU and CCS have been identified as playing key roles in a sustainable energy future. For CCS, despite the fact that the high cost of the capture and storage process have limited the technology from entering mainstream use, it also offers co-benefit advantages besides its carbon mitigation potential. Such co-benefits for CCS include control of non-CO_2 pollutants (SDG 3), direct foreign investment and know-how (SDG 9), enhanced oil recovery from existing resources, diversified employment prospects and skills (SDG 8) (Bonner 2017). For CCU, the main co-benefit related contributions are expected within the context of energy transition processes, and in societal advancements that are linked to technological progress (Olfe-Kräutlein 2020). Therefore, the expectations are that the deployment of CCU technologies would have least potential for meeting the SDG targets relating to society/people, compared with the anticipated contributions to the pillars of ecology and economy. Indeed, SDGs 7 to 11 have considerable significance for the sustainable implementation of CCU technologies.

Advancements in the development and deployment of both CCS and CCU fosters climate action (SDG 13) through the harvesting of CO_2 and its use as a resource in industrial production systems. Whiles there are economic costs involved with the deployment of CCS and CCU (Bataille et al. 2018a), there are also significant economic and developmental costs associated with taking no action because just like the way energy is a cross-cutting issue to SDGs, because of the potential negative impact of climate change, CCS and CCU has been argued as providing public good (Bergstrom and Ty 2017) and co-benefits to key SDGs (Schipper et al. 2011). On the other hand, Fan et al. (2018) among others have noted the potential lock-in of existing energy structures due to CCS.

11.6 Policy approaches and strategies

Industrial decarbonisation is possible on the mid-century horizon, but requires regionally and sectorally specific term policy strategies that consider whole innovation and supply chains (Grubb et al. 2017, 2015; Åhman et al. 2016; Wesseling et al. 2017b; Bataille et al. 2018a). Transformations of industrial systems entails changes across and within existing value-chains, new sectoral couplings, and requires enabling electricity, hydrogen and other infrastructures. Low carbon transitions are likely to be contested, non-linear and require a multi-level perspective policy approach that addresses a large spectrum of social, political, cultural and technical changes as well as accompanying phase-out policies and involve a wide range of actors, including civil society groups, local authorities, labour unions, industry associations etc. (Geels et al. 2017; Rogge and Johnstone 2017; Yamada and Tanaka 2019). It is generally found that policy strategies needed will likely combine a mix of instruments including: carbon pricing, material efficiency and high quality recycling policies, sectoral technology roadmaps for new production processes as well as technology push and market pull policies, and planning and support for new infrastructure (See Figure 11.20) (Creutzig 2019; Flanagan et al. 2011; Mehling and Tvinnereim 2018; Rogge et al. 2017; Bataille et al. 2018a). The combination of the above will depend on specific sectoral market barriers, technology maturity, and local political and social acceptance (Rogge and Reichardt 2016; Hoppmann et al. 2013; De la Rue du Can et al. 2014). Industrial decarbonisation policies need to be innovative and definitive about zero emissions to trigger the level of investment needed for the profound changes in production, use and recycling of basic materials needed (Nilsson et al. 2017). Inclusive and transparent governance that assesses industry decarbonisation progress, monitors innovation and accountability, and provides regular recommendations for policy adjustments is also important for progressing, with the UK Climate Change Commission providing one model (Mathy et al. 2016; Bataille 2020b; Bataille et al. 2018a).

The level of policy experience and institutional capacity varies widely across the mitigation options presented in this chapter (see 11.3). Energy efficiency is a well-established policy field with decades of experience from voluntary and negotiated agreements, regulations, standards, energy audits and DSM programs (see AR5). DSM programs have also included demand response and load management. The potential for demand flexibility in industry may increase with electrification and can be facilitated through progressive alignment of power market regulation.

Materials demand management is not well understood from a policy perspective and it is an option which is mostly neglected in for example low-carbon industry roadmaps although it may represent significant potential (IEA 2019b, 2020a). It is also mostly neglected in architectural and civil engineering education, infrastructure and building codes, general engineering education, and urban planning (see Chapter 5, Section 5.6). Likewise, materials efficiency, is an undeveloped policy field. For example, the overuse of steel and concrete in construction is well documented but policies or strategies (e.g., education, design guidelines, or regulation) for improving the situation appear weak or lacking. Various circular economy solutions are gaining increasing interest from policy makers with examples such as economic incentives for repair and reuse, initiatives to reduce planned obsolescence, and setting targets for recycling. Metals and plastics recycling is far from the potentials in terms of quantity and quality. Barriers that policy could address are often specific to the different material loops (e.g., copper contamination for steel and lack of technologies or poor economics for plastics).

There is also growing interest from policymakers in electrification and fuel switching but the focus so far has been mainly on innovation and developing technical supply-side solutions rather than creating market demand, although the concept of green public procurement is gaining traction. The situation is similar for CCU and CCS. CCS simply represents an additional cost to producers, and this must be handled through fiscal incentives like tax benefits, carbon pricing, green subsidies, or regulation and permit procedures. Direct electrification, hydrogen production and storage, and CCS also requires significant infrastructure investments.

Since industrial decarbonisation is a relatively new policy field there is little international collaboration on facilitation. The potential of global governance to contribute to the decarbonisation of energy intensive industry through intergovernmental and transnational institutions has remained very much underexploited (Oberthür et al. 2020).

As carbon pricing, through carbon taxes or cap and trade schemes, has remained a central avenue for climate policy this section begins with a review of how the industrial sector has responded to these instruments. The rest of the section is then structured into five key topics, following insights on key failures that policy must address to enable and support large-scale transformations as well as the need for complementary mixes of policies to achieve this goal (Grillitsch et al. 2019a; Weber and Rohracher 2012; Rogge and Reichardt 2016). The section describes how the need to focus on long-term transitions rather than incremental changes can be managed through the planning and strategising of transition pathways; discusses the role of research, development, and innovation policy; highlights the need for market creation and re-creation; reflects on the necessity of establishing and maintaining a level of knowledge and capacity in the policy domain about the industrial transition challenge; points to the critical importance of coherence across geographical and policy contexts. The section concludes with a reflection on how different groups of actors could act to take up different parts of the responsibility for mitigating climate change in the industrial sector.

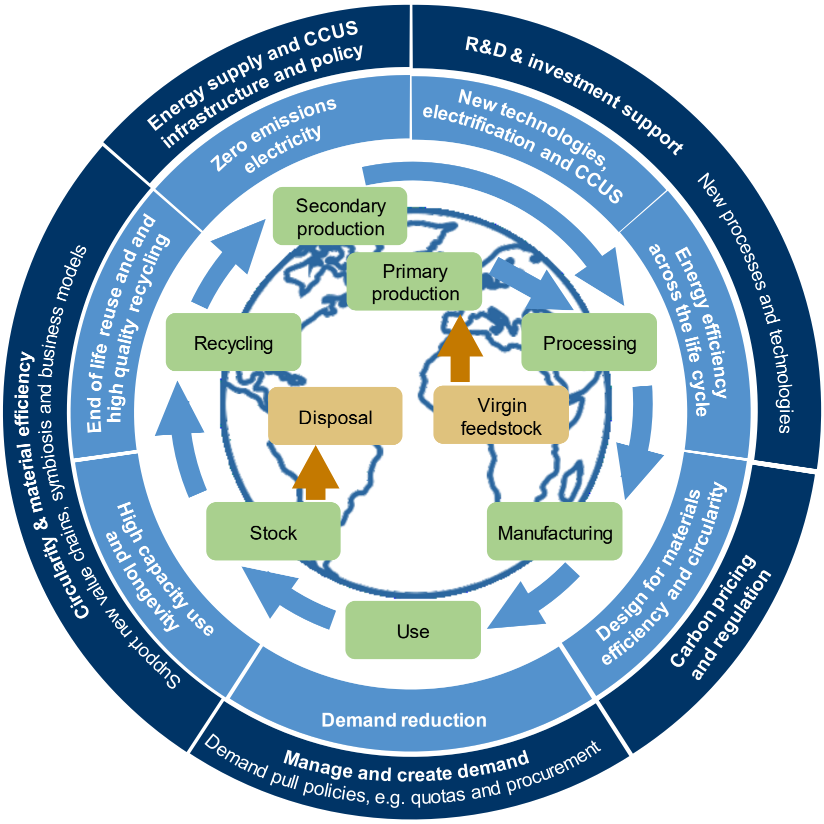


Figure 11.20 Schematic Figure showing the life cycle of materials (green), mitigation options (light blue) and policy approaches (dark blue).

11.6.1 Carbon Prices and Carbon Markets

Internalising the cost of GHG emissions in consumer and producer investment decisions has been a major strategy promoted by economists and considered by policy makers to mitigate emissions cost-effectively and to incentivise low carbon innovations in a technology neutral way (Stiglitz et al. 2017; Boyce 2018). As of May 1st, 2020, 61 carbon schemes have been implemented or are scheduled by law for implementation, covering 22.5% of global GHG emissions (The World Bank 2020a). The distribution across schemes is almost evenly spread with 31 emissions trading schemes, spread across national and subnational jurisdictions and 30 carbon taxes, primarily implemented on a national level.

Assessments of pricing mechanisms show generally that they lead to reduced emissions, even in sectors that receive free allocation such as industry (Bayer and Aklin 2020; Narassimhan et al. 2018; Martin et al. 2016; Haites et al. 2018; Metcalf 2019). However, questions remain as to whether these schemes can bring emissions down fast enough to reach the temperature goals of the Paris Agreement (World Bank 2019b; Tvinnereim and Mehling 2018; Boyce 2018) and how energy intensive trade-exposed industries are being covered as emission caps are tightened. The High-Level Commission on Carbon Prices estimated that a carbon-price level of at least USD 40–80 $tCO_2\text{-eq}^{-1}$ in 2020 and USD 50–100 $tCO_2\text{-eq}^{-1}$ by 2030 would be consistent with achieving the Paris agreement goals. Nevertheless, in the 58 carbon price schemes implemented worldwide today, only five countries have carbon taxes prices above USD 40 (Stiglitz et al. 2017). Furthermore, industries are allowed exemptions and receive provisions that shelter them from any significant price increase in virtually all pricing schemes (Haites 2018). These provisions are justified by concerns about loss of competitiveness and carbon leakage that may result from relocation and increased imports from jurisdictions with no, or less, GHG emission regulations (Branger and Quirion 2014). The most common ways of addressing the risk of carbon leakage have been to provide free allocation of emissions allowances to industry or tax rebates based on a firm's output, its historical production or its production capacity (Rissman et al. 2020). However, such approaches to lowering the carbon price for industry reduces the incentive to mitigate emissions. One approach to still incentivise emissions mitigation has been to negotiate voluntary agreements with industries to achieve specific energy efficiency or emission reduction targets in exchange for carbon price rebates and exemptions (Cornelis 2019; Martin et al. 2014; Wakabayashi and Arimura 2016). Overall, current carbon prices are well below the levels needed to motivate investments in high-cost options that are needed to reach net zero emissions (see Section 11.4.1.5).

Two alternative mechanisms to address leakage have been suggested in Bataille et al. (2018a), production pricing with border carbon adjustments (BCAs) for imports and consumption pricing where points of compliance are downstream, for example, at the final consumers levels (Munnings et al. 2019; Neuhoff et al. 2015). The latter approach provides a carbon price signal to end-use products and thus, for example, incentivises material efficiency but provides weak incentives for mitigating production emissions (Skelton and Allwood 2017). Implementation of consumption carbon pricing is challenged by the need of product GHG traceability and enforcement transaction costs (Munnings et al. 2019). Alternatively, BCAs represent another approach to address leakage applied to imports to create a level playing field in a competitive market, but implementation poses significant technical and legal challenges (Rocchi et al. 2018; Sakai and Barrett 2016; Jakob et al. 2014). Technical challenges arise because estimating the tax adjustment requires reliable data on the carbon content of products imported and legal challenges because BCAs can be perceived as a protectionist measure violating the principle of equal treatment under the regulations of World Trade Organization (WTO). However the absence of carbon prices can also been perceived as a subsidy for dirty production (Al Khourdajie and Finus 2020;

Stiglitz 2006). Another argument supporting BCA implementation is the possibility to induce low carbon investment in nonregulated regions (Cosbey et al. 2019). Thus far, California is the only jurisdiction that has implemented BCA applied on electricity imports from neighbouring states (Mehling et al. 2019; Caron et al. 2015) and is now considering BCAs for the cement sector.

The efficacy of carbon prices to achieve transformative emissions reductions has been challenged by additional real world implementation problems, such as highly regionally fragmented carbon markets (Keohane and Victor 2011; Grubb et al. 2015) and the difficult social acceptance of price increases (Raymond 2019; Bailey et al. 2012). The higher carbon prices likely needed to incentivise industry to adopt low carbon solutions pose social equity issues and resistance (Huang et al. 2019; Wang et al. 2019; Grainger and Kolstad 2010; Bataille et al. 2018c; Hourcade et al. 2018). Carbon pricing is also criticised for promoting mainly incremental low-cost options and not investments in radical technical change or the transformation of sociotechnical systems (Rosenbloom et al. 2020; Stiglitz 2019; Vogt-Schilb et al. 2018).

Clear and stable emission reduction trajectories towards 2050 goals are essential for sending strong signals to businesses. Many researchers recognise that complementary policies must be developed to set current production and consumption patterns toward a path consistent with achieving the Paris agreement goals (Schmalensee and Stavins 2017; Kirchner et al. 2019; Vogt-Schilb and Hallegatte 2017; Bataille et al. 2018c). In this broader policy context, proceeds from pricing schemes can be used to ease the transition and support high abatement cost options.

11.6.2 Transition pathways planning and strategies

Decarbonising the industry sector requires transitioning how material and products are produced and used today to development pathways that include the strategies outlined in Section 11.3 and Figure 11.20. Such broad approaches require the development of transition planning that assesses the impacts of the different strategies and consider local conditions and social challenges that may result from conflicts with established practices and interests, with planning and strategies directly linked to these challenges.

Sector visions, roadmaps and pathways have been useful tools for investigating sectoral futures (e.g. the IEA steel and cement technology roadmaps), but for various reasons are usually not set within a larger context of socio-economic goals, with clear objectives and policy direction for the sectors. Grillitsch et al. (2019b), working from the socio-technical transitions literature, focuses on the need for maintaining “directionality” for innovation (e.g. towards net-zero transformation), the capacity for iterative technological and policy “experimentation” and learning, “demand articulation” (e.g. engagement of material efficiency and high value circularity), and “policy coordination” as four main framing challenges. Wesseling et al. (2017b) bridges from the socio-technical transitions literature to a world more recognisable by executives and engineers, composed of structural components that include actors (e.g. firms, trade associations, government, research organisations, consumers, etc.), institutions (e.g. legal structures, norms, values and formal policies or regulations), technologies (e.g. facilities, infrastructure) and system interactions.

Several studies (Material Economics 2019; Wyns et al. 2019; Åhman et al. 2017; Bataille et al. 2018a) offer detailed transition plans using roughly the same five overarching strategies: 1) policies to encourage material efficiency and high quality circularity; 2) “supply push” R&D and early commercialisation as well as “demand pull” lead/niche markets to help emerging technologies cross “the valley of death”; 3) carbon pricing or regulations with competitiveness provisions to trigger innovation and systemic carbon efficiency; 4) long run, low cost finance mechanisms to enable investment and reduce risk; 5) infrastructure planning and construction (e.g. CO_2 transport and disposal, electricity and hydrogen transmission and storage), and institutional support (e.g. labour market training and transition support; electricity market reform).. Wesseling et al. (2017b) and Bataille et al. (2018a) further add a key initial enabling step would be to conduct an ongoing stakeholder pathways process, including all stakeholders with effective "veto" power in the process (i.e. firms, unions, government, communities, indigenous groups), to gather information, educate stakeholders, debate options, and build a working consensus to allow for robust, politically resilient policy package formation, as well as to focus on supply chain collaborations to bridge the cost pass-through challenge (e.g. the Swedish HYBRIT steel project; the ELYSIS consortium to commercialise inert electrodes in aluminium making). Åhman et al. (2017) additionally focusses on *common but differentiated responsibilities* (UNFCCC article 3) to resolve the potential conflicts between current free trade regimes and industrial policies needed for deep decarbonisation; they argue a new approach must seek agreement on what constitutes fair and acceptable industrial, competition and trade policies in a world evolving to net-zero GHG emissions. All three of these papers articulate the need for a shared global R&D effort for key low to zero carbon technologies. Finally, UKCCC (2019b) and Bataille et al. (2018a) reiterate that the transition plan must be designed such that the special supports for industry are designed to end once the price of climate damages is fully internalised in all levels of the supply chain (e.g. through material and energy GHG intensity pricing (Skelton and Allwood 2017)); if support policies are not ramped back technology may get trapped at a suboptimal level (Grillitsch et al. 2019b).

Detailed sectoral roadmaps that assess the technical, economic, social and political opportunities and provide a clear path to low-carbon development are needed. For example, the German state of North Rhine Westphalia passed a Climate Process Law that resulted in the adoption of a Climate Protection Plan that set subsector targets through a transparent stakeholder engagement process based on scenario development and identification of low carbon options (Lechtenböhmer et al. 2015), see Box 11.3. Another example is the UK set of Industrial Decarbonisation and Energy Efficiency Roadmaps to 2050 as well as the UK Strategic Growth Plan, which are accompanied by Action Plans for each energy intensive subsector. These include voluntary commitments by government, industry and other parties to help the sector decarbonises and improve energy efficiency.

Strategies for the circular economy provide a case in point. According to a report looking at the global mass of material flowing the economy in 2015, the world economy is only nine percent circular (PACE 2019) and there is a great need for new and innovative policies to induce the production of products with higher material value retention potential as well as to stimulate the growth of circular business models (OECD 2019c). Countries or regions have adopted high-level framework policies to plan for transitions along circular economy pathways; these include Germany's "Closed Substance Cycle and Waste Management Act" in 1996; Japan's Basic Law for Establishing the Recycling-based Society in 2000; China's Circular Economy Promotion Law of 2009; and the EU's "Closing the Loop - an EU Action Plan for the Circular Economy" adopted in 2015 (Wang et al. 2018). In China, national and local governments have developed regulations, technical standards, guidelines and evaluation indicators to monitor progress, organised at the national, industry park and city levels (Geng et al. 2012; Guo et al. 2017; Su et al. 2013; Wang et al. 2018) and a recent material flow analysis assessed the progress of the circular economy in China over time and found that circularity increased from 2.7% to 5.8% between 1995 and 2015. Wang et al. (2020) although large uncertainties remain regarding proper quantitative indicators. Cities and countries also play a major role in revalorising waste and set ambitious targets to reduce waste flowing in their jurisdiction. Strategising for such pathways includes planning for a mix of supply-push and demand-pull measures based on standards, certifications, extended producer responsibility (EPR) schemes, public procurement, tax reduction, mandates and more as further described in the next sections.

Box 11.3 IN4Climate NRW – Initiative for a climate-friendly industry in North Rhine-Westphalia (NRW)

IN4Climate NRW (www.in4climate.nrw) was launched in September 2019 by the state government of North Rhine-Westphalia (IN4climate.NRW 2019). As a platform for collaboration between representatives from industry, science and politics, IN4climate.NRW is unique in Germany in offering a common space to develop innovative strategies for a climate-neutral industrial sector bringing together different perspectives and competencies.

North Rhine-Westphalia is Germany's industrial heartland. Around 19% of North Rhine-Westphalia's greenhouse gas emissions have their origin in the industry sector, consequently the sector bears a particular responsibility when it comes to climate protection. On the other hand the industry sector, more than in other states stands for high quality jobs and export power. It is the clear understanding and belief of the NRW government that the current position can only be maintained if the regional industry will be successful in positioning itself as a frontrunner for becoming GHG-neutral while at the same time securing competitiveness.

In working together across different branches (currently stakeholder from more than 30 companies representing amongst others steel, cement, chemical, aluminium industry, refineries, energy utilities, TSOs) and enabling a direct interaction between industry and government officials IN4Climate provides an added value to the participating companies. People from the different areas are working together in so called innovation teams and underlying working groups with a self-organised process of setting their milestones and working schedule while reflecting long-term needs as well as short-term requirements based on political or societal discussions.

The innovation teams aim to identify and set concrete impulses for development and implementation of breakthrough technologies, specify necessary infrastructures (e.g. hydrogen infrastructure) and appropriate policy settings (i.e. integrated state, national, European policy mix). They also include an attempt to create a discourse between the public and the industry sectors as a kind of sounding board for the early detection of barriers and obstacles.

The initiative has been successful so far, developed for example a clear vision for a hydrogen strategy and an associated policy framework as well as a broader decarbonisation strategy for the whole sector. It is present at the national level as well as at the European level. As being successful and unique IN4Climate meanwhile works as a blueprint for other regions and is often visited by companies and administration staff from other German states.

It is particularly the so far missing intensive and dedicated cross-branches cooperation that can be seen as success factor. Facing substantial transformation needs associated with structural changes and infrastructure challenges very often solutions can't be provided and realised by single branches, but need cooperation and coordination. Even more, chicken and egg problems like the construction of new infrastructures (e.g. hydrogen, CO_2) require cooperation and new modes of collaboration. For this purpose IN4Climate provides the appropriate basis and brings in the necessary link.

11.6.3 Technological research, development, and innovation

Policies for research, development, and innovation (RDI) for industry are present in most countries but it is only recently, and mainly in developed countries, that decarbonisation of emissions intensive industries has become a prioritised topic. These industries are characterised by large dominant actors and mature process technologies on the primary production side of the value chain. Investments in RDI are commonly low and aimed at incremental improvements to processes and products (Wesseling et al. 2017b). Although RDI is required across all mitigation options, this section is focused on innovations to eliminate primary process emissions through electrification, fuel switching and CCU.

11.6.3.1 *Basic Research & Small scale physical piloting (Demonstration)*

Investing in RDI for low-CO_2 process emissions is risky and unfocussed without properly assessing options, technology readiness levels, and roadmaps towards technology demonstration and commercialisation. For industry, important areas for RDI can be identified since there is generally a limited set of possible process options for producing basic materials using fossil-free energy and feedstock, or CCU and CCS (11.3.5; 11.3.6).

In the absence of strong climate policy, many innovations that are necessary for industrial decarbonisation are uncompetitive and risky compared to established process technologies. In addition to undertaking technology RDI it is therefore useful to explore support policy options. Public RDI policies may include public funding of basic research, technological and business model experimentation, pilot and demonstration plants, as well as support for education and training – which further have the positive side-effect of leading to spill-overs and network effects through labour market mobility and collaboration (Nemet et al. 2018). Innovative business models will not emerge if the transition is not considered along the full value chain with a focus on materials efficiency, circularity, and new roles for industry in a transitioning energy system, including possibly providing demand response for electricity (Vogl et al. 2018a).

Collaborative innovation across sectors with complementing knowledge bases and capabilities is important as several of the options for reducing process emissions involve new or stronger sectoral couplings (Tönjes et al. 2020). One example is linking chemicals to forestry in the upscaling of forest biorefineries although it has proven to be difficult to engage a diverse group of actors in such collaborations (Karltorp and Sandén 2012; Bauer et al. 2018). Heterogeneous collaboration and knowledge exchange can be encouraged through conscious design of RDI programs and by supporting network initiatives involving diverse actor groups (Van Rijnsoever et al. 2015; Söderholm et al. 2019).

11.6.3.2 *Policy support from demonstration to market*

Basic research is relatively inexpensive compared to piloting, demonstrations, and early commercialisation, and arguably much of it has already been achieved for the key needed technologies for very low and zero emissions basic materials (see table in 11.4.1). This includes electricity and hydrogen-based processes, electrothermal technologies, high temperature heat pumps, catalysis, etc. The research and development that has occurred however, has generally not focussed on complete supply chains for new primary material production pathways. This was the purpose of the UK Offshore Wind Accelerator Project https://www.carbontrust.com/offshore-wind/owa/, a collaborative research, development and deployment programme to reduce costs and overcome market barriers. Coordinated by the UK Carbon Trust and working with wind turbine manufacturers the project looked across the supply chain for floating offshore and identified what components manufacturers could innovate and produce by themselves, and where there were gaps beyond the capability of any one firm. This process led to several key areas of work where the government and firms could work together; once the concepts were piloted and proven, the firms went back into a competitive mode. The project illustrates the potential importance of third parties, including government, in creating platforms and opportunities for cross-industry exchange and collaboration (Tönjes et al. 2020).

Most countries already maintain government labs for doing defence and non-defence basic research and small-scale piloting (e.g. the US Defense Advanced Research Project Authority and civilian ARPA-Energy programs). The European Union operated the Ultra Low Carbon Steel (ULCOS) project (Quader et al. 2016), which lead to several small scale pilot that are now going to larger scale firm pilots (e.g., HISARNA, HYBRIT and SIDERWIN). Supported by the EU, several cement firms are working together on the cement LEILAC project, where a new form of limestone calciner is being developed to concentrate the process CO_2 emerging from quicklime production (about 60% of cement emissions) for eventual utilisation or geological storage (as one of many options for cement, see for example, Plaza et al. (2020)). If LEILAC works, it is conceivable that existing cement plants globally that are located near CCS opportunities could have their emission reduced by 60% with one major retrofit of the kiln. Arguably, the basic research and small-scale piloting component of technology push can be well served.

Once technologies have proved they work at the small scale the time comes to test them at a quasi-commercial scale in a real-world environment. This is a very expensive stage, where costs jump by a magnitude with no increased revenue (i.e. no imminent market pull is available), and because of this it is very risky and where many technologies and projects are deemed as not economically viable (see, e.g., Åhman et al. (2018) and Nemet et al. (2018)). The HISARNA, SIDERWIN and LEILAC prototypes are all at the stage of scaling up.

Given the resource requirement, a diversified portfolio of investors and support is required to share the risk. LEILAC includes several firms, as did the UK Offshore Wind Accelerator. Government funds will also be required, which could be refunded in the future through an equity position, royalty or taxes. Direct subsidies are likely to be needed beyond the demonstration of new technological concepts to the commercialisation of first-of-a-kind industrial-scale plants (Hellsmark and Jacobsson 2012). When possible, important continuous learning and innovation can also happen in special market segments. For example, while PV enjoyed a lot of government support in early periods (Kavlak et al. 2018), there were also existing “natural” markets for solar PV generation. PV was pioneered for satellites, where the value of electricity without on-board fuel was very large, and then progressively captured other markets poorly served by other generating technologies, usually remote locations markets for which fuel delivery was prohibitively costly.

Finally, large capital flows towards deployment of low carbon solutions will not materialise without risk minimisation through all the above. These crucial connections point to the need for comprehensive regional and sectoral industrial strategies (UKCCC 2019b; Wesseling et al. 2017b; Bataille et al. 2018a; Wyns et al. 2019).

11.6.4 Market Creation

Markets for decarbonised industrial processes and products must be created, taking into account that contemporary markets have been shaped in a context of low-cost fossil fuels – which in many contexts even remain subsidised for the energy-intensive industries. Policy will thus be needed to establish the first niche markets which are essential for refining new decarbonised technologies, trouble shooting, and for building manufacturing economies of scale. Market creation does however go beyond the nurturing, shielding, and empowerment of early niches (Smith and Raven 2012; Raven et al. 2016) and must also consider how to significantly re-shape existing markets to create space for decarbonised solutions and crowd out fossil-based ones (Mazzucato 2016). The perceptive of an increasing durable demand for low carbon products induce manufacturers to invest in decarbonisation strategies (Olatunji et al. 2019). Market creation policy instruments unlock demand for low carbon products by addressing market barriers and stimulating new markets. Experience in the development of renewables demonstrates the importance of creating market demand pull and investor certainty to induce innovation (Newell et al. 1998; Hoppmann et al. 2013). The successful experience of feed-in tariffs, renewable portfolio standards and market auctions can provide examples to design market pull instruments to impulse industry decarbonisation investment (Vogl and Åhman 2019; Girod 2016; Nilsson et al. 2017).

11.6.4.1 Carbon content certifications

The development of carbon labels corresponds to a growing demand from consumers desiring information about the climate impacts of their consumption (Darnall et al. 2012; Feucht and Zander 2018; Tan et al. 2014). Carbon labels fill this information gap by empowering consumers purchasing decisions and creating higher value for low carbon products and materials (Cohen and Vandenbergh 2012; Vanclay et al. 2011). The willingness to pay for lower carbon products has been found to be positive but to depend on socioeconomic consumer characteristics, cultural preferences and the product considered (Li et al. 2017; Shuai et al. 2014; Tait et al. 2016; de-Magistris and Gracia 2016; Feucht and Zander 2018). Companies and governments that favour low carbon products and who are seeking to achieve environmental, social, and governance (ESG) goals also need readily available and reliable information about the carbon content of products and materials they purchase and produce (Munasinghe et al. 2016; Long and Young 2016).

Numerous methodologies have been developed by public and private organisations to meet the needs for credible and comparable environmental metrics at the product and organisation levels. Most follow life cycle assessment standards as described in ISO 14040 and ISO 14044, ISO 14067 for climate change footprint only and ISO 14025 (2006) for environmental product declaration (EPD) but the way system boundaries are applied in practice varies (Liu et al. 2016; Wu et al. 2014). Adoption has been challenged by the complexity and the profusion of applications which contribute to confuse stakeholders (Guenther et al. 2012; Gadema and Oglethorpe 2011; Brécard 2014). The options of applying different system boundaries and allocation principles involve value judgements that in turn influence the results (McManus et al. 2015; Tanaka 2008; Overland 2019; Finnveden et al. 2009). A more systematic and international coordinated approach based on transparent and reliable data is needed to induce global low carbon market development (Pandey et al. 2011; Tan et al. 2014; Darnall et al. 2012).

11.6.4.2 Public Procurement

Governments spend a large portion of their budget on the provision of products and material through infrastructure development, general equipment and miscellaneous goods. The OECD estimates that an average of 29% of general government expenditure go to public procurements in OECD countries, representing 12% of GDP, which makes government a powerful market actor. Public procurement can therefore create a significant market pull and be used to pursue strategic environmental goals (Ghisetti 2017). The EU commission has developed environmental criteria that can be directly inserted in tender documents (Igarashi et al. 2015; European Commission 2016). These criteria are voluntary, and the extent of their application varies across public authorities (Bratt et al. 2013; Testa et al. 2016; Michelsen and de Boer 2009). In the Netherlands, companies achieving a desirable certification level under the national CO_2 Performance Ladder obtain a competitive advantage in public procurement (Rietbergen and Blok 2013; Rietbergen et al. 2015). Implementation can be challenged by the complexity of criteria, the lack of credible information to check and the added time needed for selection (Testa et al. 2012; Cheng et al. 2018; Geng and Doberstein 2008; Zhu et al. 2013; Liu et al. 2019b; Bratt et al. 2013; Lundberg et al. 2015). Local, regional and national authorities can use their purchasing power to guarantee markets for low carbon products and material and therefore contribute to creating lead markets for low-carbon practices and design. Using public procurement for innovative products, such as low-carbon materials, sets specific requirements on the scale, organisation, and formulation of procurement processes (Wesseling and Edquist 2018). The state of California in the United States passed the Buy Clean California Act (AB 262) that establishes maximum acceptable global warming potential for eligible steel and glass construction materials for public procurement (USGBC-LA 2018). See Box 11.4.

Box 11.4 Buy Clean California Act

In October 2017, California passed Assembly Bill (AB) 262, the Buy Clean California Act, a new law requiring state-funded building projects to consider the global warming potential (GWP) of certain construction materials during procurement. The bill requirements are two-pronged: manufacturers of eligible materials must submit facility-specific environmental product declaration (EPD), and the eligible materials must demonstrate (through submitted EPDs) GWP below the product-specific compliance limits defined by the state Department of General Services (DGS), which will regulate policy implementation. The eligible materials include structural steel, carbon steel rebar, flat glass, and mineral wool insulation. An amendment (Assembly Bill 1817) to the original Buy Clean California Act passed in June 2018, extending the timeline for compliance (USGBC-LA 2018). In January 2019, state agencies were to request, from bidders, voluntary submission of facility-specific EPDs. As of January 2020, successful bidders must submit facility-specific EPDs. By January 2021, DGS will establish and publish the "maximum acceptable" GWP for each product category, which bidders must meet for eligible materials to be used for state-funded projects. By July 1, 2021 DGS will gauge GWP compliance of eligible materials with EPDs (DGS 2020; USGBC-LA 2018).

DGS allows the exclusion of emissions that occur during the fabrication stages (bending, tempering, cutting, and additional processing of base materials) of the products. DGS is considering two options to establish the GWP limit: 1) Use an industry wide EPD for an eligible material. 2) Calculate an average using producer facility specific EPDs. A tolerance is still expected to be added accounting for uncertainty in the life cycle assessment process. EPDs will be obtained from awarding agencies as well as those found on publicly available databases.

Prior to adoption of the Buy Clean California Act, the California Department of Transportation (Caltrans) had been evaluating the use of life-cycle assessment and EPDs in evaluating materials. Caltrans established the Caltrans EPD Implementation Project to begin collecting EPDs for construction materials. In addition to the materials specified in Buy Clean California Act (noted above), the Caltrans project includes materials used extensively in transportation (concrete, asphalt, and aggregate). Also, the California High-Speed Rail project had begun using EPDs as part of its procurement process. The High-Speed Rail Sustainability Report states that the construction projects will: 1) require EPDs for construction materials including steel products and concrete mix designs, and 2) require "optimised life-cycle scores for major materials" and include additional strategies to reduce impacts across the life cycle of the project (Simonen et al. 2019).

Several other states such as Washington, Oregon, and Minnesota are developing similar type of Buy Clean regulation (BGA 2020; Simonen et al. 2019).

11.6.4.3 Private Procurement

The number of companies producing sustainability reports has increased rapidly over the last decade (Higgins and Coffey 2016; Jackson and Belkhir 2018). This trend has mainly been driven by consumer concerns, investor requests, and as a business strategy to gain a competitive advantage (Koberg and Longoni 2019; Sullivan and Gouldson 2012; Zhou and Wen 2019; Higgins and Coffey 2016; Ibáñez-Forés et al. 2016). For example, Apple and the governments of Québec and Canada are the financier and lead market maker in the Elysis consortium to bring inert electrodes to market for bauxite smelting to make zero carbon aluminium. Aluminium is a very small part of the value in a laptop or smartphone, so even very expensive low emissions aluminium will add to Apple's brand at very little cost per unit sold.

A key operational change has been the standardisation of definitions of Scope 1 (direct emissions), Scope 2 (indirect emissions from purchased electricity) and Scope 3 (emissions from supply chain's impacts and consumer's usage and end-of-life recycling practices) (Pankaj Bhatia et al. 2011). For many corporations, the emissions impact within their supply chain far exceeds their operations direct emissions (CDP 2019). Therefore, the opportunities to reduce emissions through purchasing goods and services from the supply chain have much greater potentials than from direct emissions. Therefore, the most determined companies have developed internal carbon abatement strategies that incorporate their supply chains' emissions (Martí et al. 2015; Blome et al. 2014; Gimenez and Sierra 2013) and designed low carbon incentive policies or procurement contracts that encourage or require their suppliers into improving their product carbon footprint (Liu et al. 2019a). However, there is a lack of consistency and comparability in the way firms are reporting emissions, which limits the assessment of companies' actual ambition and progress (Blanco et al. 2016; Liu et al. 2015; Burritt and Schaltegger 2014; Sullivan and Gouldson 2012; Rietbergen et al. 2015). In average only 22% of scope 3 emissions were reported on a sample of 265 companies' voluntary reporting scope 3. More research is needed to assess the current impacts of corporate voluntary climate actions and how these efforts contribute to meet the Paris agreement's goals (Rietbergen et al. 2015; Wang and Sueyoshi 2018). There is currently no obligation for firms to report scope 3 emissions.

11.6.4.4 Mandatory Performance Standards

Policy makers can set minimum performance standards or maximum emission content specifications through legislation to increase the use of low carbon products, the construction of low carbon developments and foster reuse and recyclability.

Construction of buildings is the largest end-use of carbon-intensive materials and the share of embodied emissions in total building's carbon footprint is increasing as building energy efficiency is improving and energy supply is decarbonised (Chastas et al. 2016). The IEA estimates that buildings are currently responsible for 39% of global energy related carbon emissions, of which 25% is from materials and construction (GABC and IEA 2017). As a result, jurisdictions are increasingly considering new requirements in building codes to reduce embodied emissions. This is the case of France's new building code which is shifting from a thermal regulation (RT 2012) to an environmental regulation (RE 2020) to include embodied carbon LCA metrics for encouraging use of low-carbon building materials (Schwarz et al. 2019; Ministère de la Transition écologique et solidaire 2018). The 2018 International Green Construction Code (IGCC) provides technical requirements that can be adopted by jurisdictions for encouraging low carbon building construction, including minimum construction waste in term of kg per m^2 of new construction, development of waste management plan, specific share of recycled, biobased and regional content materials and environmental declaration of a minimum number of products and material used. IGCC also covers minimum longevity and durability of structural, building envelope, and hardscape materials (Art. 1001.3.2.3) and permit future reconfiguration, dismounting and disassembly of partitions (Celadyn 2014). Conventional building codes should be revised as they often prescribe over use of energy intensive material stemming from misconceptions such as that more cement in a concrete mixture has better performing results as shown in Taylor, P., Bektas, F., Yurdakul, E. and Ceylan (2012). Low carbon Building Rating Systems, such as LEEDs, are voluntary standards which include specific requirements on material resources in their rating scale. The county of Marin in California, has adopted an ordinance adding a performance compliance specification with embodied carbon limits, depending on minimum specified compressive strength, provided as maximum embodied carbon in $kgCO_2$-eq m^{-3}, to be specified applicable to all private and public construction projects and per EPD. Trade-offs between energy performance achievement and material used in building construction needs to be further assessed and considered as low carbon building code requirements develop and a global LCA metric should be used (Röck et al. 2020; Shadram et al. 2020).

11.6.4.5 Financial Incentives

Fossil-free basic materials production will often lead to higher costs of production, for example, 20–40% more for steel, 70–115% more for cement, and potentially 15–60% for chemicals (Material Economics 2019). There is a nascent literature on what are effectively material "feed-in-tariffs" to bridge the commercialisation "valley of death" (Wilson and Grubler 2011) of early development of low carbon materials (Neuhoff et al. 2018; Sartor and Bataille 2019; Bataille et al. 2018a; Wyns et al. 2019). Renewable electricity support schemes have typically been price-based (e.g., production subsidies and feed-in-tariffs) or volume-based (e.g., quota obligations and certificate schemes) and both principles can be applied when thinking about low carbon materials. Auction schemes are typically used for larger scale projects, for example, off-shore wind parks.

Based on how feed-in-tariffs worked, a contract for difference (CfD) could guarantee a minimum and higher than market price for a given volume of early low GHG materials. CfDs could be based on a minimum effective carbon price reflecting parity with the costs of current higher emitting technologies, or directly on the higher base capital and operating costs for a lower GHG material (Richstein 2017; Chiappinelli et al. 2019; Sartor and Bataille 2019). CfDs can also be offered through low carbon material procurement where an agreed price offsets the incremental cost of buying low carbon content product or material. Private firms, by themselves or collectively, can also guarantee a higher than market price for low carbon materials from their supplier for marketing purposes (Bataille et al. 2018a; Bataille 2020b). Quota obligations for low GHG materials is another option but the complexity of markets and diversity of applications for most materials, and the uncertainty around resulting certificate prices probably makes it less viable than other options (Vogl et al. 2020). Reverse auctions (by which the lowest bidder gets the production subsidy) for low carbon materials is also an option but it remains to be analysed and explored. While these financial incentive schemes have been implemented for renewable energy, their application to incentivise and support low GHG material production have yet to be developed and implemented. The German government is currently developing a draft law which will allow companies to bid for CfD for the steel, chemical and cement sectors (Agora Energiewende and Wuppertal Institut 2019). To cover the costs, the German government plans to introduce a climate surcharge on end products on selected emission-intensive materials (steel, cement, aluminium and plastics).

11.6.4.6 Extended producer responsibility

EPR systems are increasingly used by policy maker to require producers to take responsibility for the end life of their outputs and to cover the cost of recycling of materials or otherwise responsibly managing problematic waste types (Kaza et al. 2018). According to the OECD, there are about 400 EPR systems in operation worldwide, three-quarters of which have been established over the last two decades. One-third of EPR systems cover small consumer electronic equipment, followed by packaging and tyres (each 17%), vehicles, lead-acid batteries and a range of other products (OECD 2016).

While the economic value of some discarded materials such as steel, paper and aluminium is generally high enough to justify the cost and efforts of recycling, at current rates of 85% above 60% and 43% respectively (Cullen and Allwood 2013) (UNEP 2011), others like plastic or concrete have a much lower re-circularity value (UNEP 2011) leading to low recycling of these materials even when covered by EPR schemes. A case in point is plastics – largely used for packaging and vehicles which are both covered by EPR schemes for example in Europe. Most plastic waste does however end up in landfills or dumped in the environment, only 9% is recycled and 12% incinerated globally (Geyer, Jambeck, and Law 2017; UNEP 2018). Collected waste plastics from OECD countries were largely exported to China until a ban in 2018 constraining OECD countries to review their practice (Qu et al. 2019). EPR schemes may thus need to be strengthened to actually achieve a reduced use of virgin, carbon intensive materials.

11.6.5 Knowledge and capacity

It is important that government bodies, academia and other actors strengthen their knowledge and capacities for the broad transformational changes envisioned for industry. In Japan, the industry has been voluntarily working on carbon reduction, under the Framework of Keidanren's Commitment to a Low-carbon Society since 2009. Government and scientific experts regularly review their commitments and discuss results, monitoring methods, and reconsidering goals. Industry federations/associations can obtain advice in the follow-up meeting from other industries and academics. The energy and transport sectors have decades of building institutions and expertise, whereas industrial decarbonisation is largely a new policy domain. Most countries have experience in, for example, energy efficiency policies, some areas of research and innovation, waste management, regulations for operational permits and pollution control, worker safety and perhaps fuel switching. There is less experience with market demand pull policies although low carbon public procurement is increasingly being tested. Circular economy policies are evolving but potential policies for managing material demand or demand growth are less understood. Material efficiency policies through, for example, product standards or regulation against planned obsolescence are nascent but relatively unexplored (Gonzalez Hernandez et al. 2018a).

All this argues for active co-oversight, management and assessment by government, firms, sector associations and other actors, in effect the formation of an active industrial policy that includes decarbonisation in its broader mandate of economic and social development (OECD 2019b; Bataille 2020b). Important aspects of governance include mechanisms for monitoring, transparency and accountability. It may involve the development of new evaluation approaches, including a greater focus on ex-ante evaluations and assessment of, for example, readiness and capacities, rather than ex-post evaluations of outcomes. Such organisational routines for learning have been identified as a key aspect of policy capacity to govern evolutionary processes (Karo and Kattel 2018; Kattel and Mazzucato 2018). Although many governments have adopted ideas of focusing resources on the mission or challenge of climate change mitigation, comparisons between Western and East-Asian contexts show significant differences in the implementation of governance structures (Wanzenböck et al. 2020; Mazzucato et al. 2020; Karo 2018). Overall, improved knowledge and stronger expertise is important also to handle information asymmetries and the risk of regulatory capture.

11.6.6 Policy coherence and integration

An industrial net-zero transition, while technically feasible with small advancements in our current technological capability, involves not just a shift in production technology but major shifts in demand, material efficiency, circularity, supply chain structure and geographic location, labour training and adaptation, finance, and industrial policy. This transition must also link decarbonisation to larger social goals (e.g. air and water quality, low carbon growth, poverty alleviation, sustainable development goals) (OECD 2019b).

A net-zero industrial transformation implies substantial changes to industrial supply chains and labour force preparation. All or portions of production may move where clean resources are most abundant and relatively inexpensive. For example, steel making has historically located itself near iron ore and coal resources for making steel using BF-BOFs, whereas in the future it may be located near iron ore and zero GHG electricity for making electrified steel via hydrogen DRI EAF or molten oxide electrolysis, or close to carbon storage site (Fischedick et al. 2014; Vogl et al. 2018b; Bataille 2020b). This indicates large changes in industrial and supply chain structure, with directly associated needs for employment and skills. Some sectors will grow and some will shrink, with differing skill needs. Each new workforce cohort needs the general specific skill to provide the employment that is needed at each stage in the transition, implicating a need for co-ordination with policies for education and retraining.

Depending on what mixes of deep decarbonisation strategies are followed in a given region (e.g. CCU, CCS, electrification, hydrogen, and biomass), infrastructure will need to be planned, financed and constructed; for example, see the industry portion of the UKCCC Net-zero Technical Report (UKCCC 2019b). CCU and CCS will require transport pipelines to allow transfer of captured CO_2 for utilisation and disposal. Electrification will require market design and transmission to support increased generation, transmission, and instantaneous capacity requirements. Hydrogen will require CCU/CCS utilisation or disposal if created from methane, and electrolysis/fuel-cell facilities and transmission if created using electricity. Biomass will require field collection and transport to point of use. CO_2 pipelines and expanded electricity transmission both have natural monopoly characteristics (i.e. least cost is provided by one supplier due to economies of scale) which are normally governed (i.e. prices are set) and planned by national and regional grid operators and their regulators. Industrial clustering (a.k.a. eco-parks), such as those planned in Rotterdam and Teeside, UK, would allow more physical and cost-effective sharing of electricity, CCU, CCS, and hydrogen infrastructure but is dependent on physical planning, permitting, and infrastructure policies.

What costing analysis that exists (e.g. the European Union Long Term Strategy, informed by (Material Economics 2019; Wyns et al. 2019)), indicates an increased upfront need for financial capital (e.g. from 2.0 to 2.8 % of GDP yr^{-1}). This indicates a need for policies to encourage long term, patient capital that reflects society's preferences for investment in industrial decarbonisation and the minimum 10 or more years horizon before there are significant new commercially available processes.

All the above indicate the need for general industrial policy as part of a coherent general economic, taxation, investment, employment and social policy for climate change mitigation (Wesseling et al. 2017a; Bataille et al. 2018a; Wyns et al. 2019).

Box 11.5 Circular economy policy

The implementation of a circular economy relies on the operationalisation of the R-imperatives or strategies which extend from the original 3Rs: Reduce, Reuse and Recycle to the consideration of additional Rs (Reike et al. 2018). The Rs implementation strategies are diverse across countries (Kalmykova et al. 2018; Ghisellini et al. 2016) but, in practice, the lower forms of value retention of materials, such as recycling and recover (energy), often dominate policies due to their easier implementable and measurable capacity. The lack of policies for more efficient material use with a higher value retention has been explained with institutional failures, lack of coordination as well as lack of strong advocates (Gonzalez Hernandez et al. 2018a). Policy levels span from micro (such as consumer or company) to meso (eco-industrial parks) and macro (provinces, regions and cities) (Geng et al. 2019). The creation of eco-industry parks ("industrial clusters") has been encouraged by governments to facilitate waste exchanges between facilities, where by-products from one industry is used as a feedstock to another (Tian et al. 2014; Winans et al. 2017; Jiao and Boons 2014; Shi and Yu 2014; Ding and Hua 2012). Systematic assessment of wastes and resources is carried out to assess possible exchange between different supply chains and identify synergies of waste streams that include metal scraps, waste plastics, water heat, bagasse, paper, wood scraps, ash, sludge and others (Ding and Hua 2012; Shi and Yu 2014).

Policies addressing market barriers to circular business development need to ensure that the full environmental costs of production, consumption and waste management are reflected in market prices, demonstrate that circular products meet quality performance standards and increase market opportunities for circular products exchange, notably through industrial symbiosis clusters and trading platforms (Hartley et al. 2020; Hertwich 2020; Kirchherr et al. 2018; OECD 2019a). While the concept of a circular economy has grown in policy relevance, the development of data collection and indicators are nascent and need to ramp up to quantify the impacts and provide evidence to improve circular economy as well as materials efficiency policies.

There has been also a growing interest from policy makers to better understand the socio-economic impacts of transitioning to circular economies (Llorente-González and Vence 2020). Results are multiples with some assessment showing positive GDP growth and job creation by shifting to more labour-intensive recycling plants and repair services than resource extraction activities (Cambridge Econometrics et al. 2018; WRAP and Alliance Green 2015). For example, the International Labour Organization estimates that worldwide employment would grow by 0.1% by 2030 under a circular economy scenario (ILO, 2018). However questions remain if the type of job created are concentrated in low-wage labour-intensive circular activities which may need targeted policy instruments to improve working conditions (Llorente-González and Vence 2020).

11.6.7 Roles and responsibilities

While all climate policy requires topic specific adaptive governance for long term effectiveness (Mathy et al. 2016), deep decarbonisation of heavy industry has special governance challenges, different from those for the electricity, transport or buildings sectors (Åhman et al. 2017; Wesseling et al. 2017a; Bataille et al. 2018a). Investments are rare, capital intensive and very "lumpy"; in an atmosphere where transformative innovation is required the process is very capital focussed with non-diversifiable risks unless several companies are involved; there are significant infrastructure needs for electricity, hydrogen and CCUS; given there is no "natural " market for low emissions materials, there is a need to manage both the supply and demand sides of the market, especially in early phase through lead supplier and markets; and there is a very high probability of surprises and substantial learning, which could affect policy choice, direction and stringency.

Different types of actors thus have to take different roles and responsibilities in developing, supporting, and implementing policies for an industrial transition. Figure 11.21 below shows how the different core parts of integrated policy making for a industrial transition may depend on efforts from different actors groups, and highlights the responsibility of these actor groups in developing a progressive and enabling policy context for the transition. This includes policy makers at local, national, and international arenas as well as civil society organisations, industry firms, and interest organisations.

Actors	**Direction:** Planning and strategising pathways to net zero	**Innovation:** RDI for new technologies and other solutions	**Market creation:** Create and shape demand-pull for various solutions	**Knowledge and capacity:** Build institutional capacity across various actors	**Coherence:** Establish international and national policy coherence
International bodies and multilateral collaboration	More attention to industry in NDCs Monitor progress and identify gaps	Include heavy industry decarbonisation in technology cooperation (e.g. Mission Innovation)	International standards and GHG labels	Support knowledge building on industrial decarbonisation	Align other Conventions and arenas (e.g. WTO) with climate targets
Regional and national government, and cities	Require and facilitate roadmaps at various levels Set targets Sun-set clauses and phase-out agreements for polluting plants	Support experimentation and urban living-labs for recycling and demand management (e.g. sharing economy)	Public procurement Repurposing of buildings Green infrastructure investments	Support and facilitate materials efficient and circular solutions through building codes, recycling and waste management policy	Support vertical policy coherence (i.e., international, national, city level)
Civil society	Monitor and evaluate leaders and laggards. Support transparency	Engage in responsible innovation programs, experimentation and social innovation	Progressive standards and criteria for materials and products (e.g. LCA-based), including updating	Engage in policy processes Consumer information and knowledge	Coordination across policy domains (trade, climate, etc.) Monitor policy coherence
Industrial sectors and associations	Net zero emissions targets Roadmaps Policy strategies	Share best practice Coordination and collaboration Efficient markets for new technology (e.g. licensing)	Progressive standards and criteria for materials and products (e.g. LCA-based), including updating	Education and retraining for designers, engineers, architects, etc. Information sharing and transparency to reduce information asymmetry.	Coordination across policy domains (trade, climate, etc.) Explore sectoral approaches
Corporations and companies	Net zero emissions targets Corporate and plant level roadmaps for reaching targets	Lead and participate in R&D, pilots, and demonstrations Increase and direct R&D efforts	Marketing and procurement of low emissions materials and products. Phase-out of high emission products	Engage in value chains for increased recycling and materials efficiency	MNCs avoid race to the bottom, and strategically account for high carbon price as part of transition strategy

Figure 11.21 Illustration of the potential roles of different actors in key areas (to highlight agency) for a low carbon transition

11.7 Knowledge gaps

An increasing body of research proposes deep decarbonisation pathways for energy intensive industries including such mitigation options as materials efficiency, circular economy and new primary processes. These options are underrepresented in climate change scenario modelling and integrated assessment models, some of which do not even reflect evolution of demand for basic materials, which is a key driver behind energy consumption and GHGs emissions in the industrial sector. As a result, no agreement is reached so far between bottom-up and top-down studies on the effectiveness and costs for many promising mitigation options, their respective roles, sequencing and packaging within various mitigation pathways.

A significant shift is needed from the transition process of the past mainly based on marginal (incremental) changes (with a strong focus on energy efficiency efforts) to a transformational change, while there is limited knowledge of the ability to implement such change effectively with a sufficient flexibility in switching pathways, as for some options technology readiness levels are currently low.

There is a knowledge gap on comparable, comprehensive, and detailed quantitative information on costs and potentials associated with the mitigation options for deep decarbonisation in industry, as cost estimates are not often comparable due to the regional or country focus, differences in costs metrics, currencies, discount rates, and energy prices across studies and regions.

A very large and important uncertainty is the availability of biomass for deep decarbonisation pathways due to competition for biomass feedstock with other priorities and the extent to which electrification can reduce the demand for bioenergy in the industry, transport and energy sectors.

CCUS is an important mitigation option in industry, for which the potentials and costs vary considerably depending on the diversity of industrial processes, the volume and purity of carbon dioxide flows, the energy requirements, the lifetime of utilisation products, the production route.

Effectiveness of packaged mitigation policies in industry is poorly known, as so far it has largely been sheltered from the impacts of climate policy due to the concerns for competitiveness and carbon leakage. There is lack of integration of material efficiency and circularity with energy and climate policies which partly results from the inadequacy of monitored indicators to inform policy debates and set targets, lack of high-level political focus and industrial lobbing; uncoordinated policy across institutions and sequential nature of decision-making along supply chains.

Industry as a whole is a very complex web of sectors, sub-sectors and inter-sectoral interaction and dependence, with associated mitigation opportunities and co-benefits and costs. Additional knowledge is needed to understand these inter-sectoral interactions in the transformation processes.

Industrial climate mitigation policy is supplemental to many other policy instruments developed to reach multiple industrial goals, for the range of stakeholders with their interest and priorities reflecting the assessment of co-benefits and risk and affecting decision-making processes and behaviour of stakeholders. Better knowledge is needed to identify the co-benefits for the adoption of climate change mitigation strategies.

11.8 Frequently Asked Questions

FAQ 11.1 What are the key options to reduce industrial emissions?

To decarbonise industry requires that we pursue several options simultaneously. These include energy efficiency, materials demand management, improving materials efficiency, more circular material flows, electrification, as well as Carbon Capture and Utilization (CCU) and Carbon Capture and Storage

(CCS). Renewable energy and feedstock resources are sufficient to meet future demand for fossil free industrial production, especially if we can reduce the need for resource extraction and energy intensive primary processing. Recycling is an option to achieve this. It includes chemical recycling of plastics, i.e., breaking them down to produce new monomer building blocks, potentially based on biogenic carbon and hydrogen instead of fossil feedstock. Hydrogen can also be used as a reduction agent instead of coke in ironmaking. Process emissions from cement production can be captured and stored or used as feedstock for chemicals and materials. Electricity and hydrogen needs can be very large but the potential for renewable electricity is not a limiting factor.

FAQ 11.2 Isn´t industrial decarbonisation very costly? Will it not conflict with sustainable development?

The answer is both yes and no to both questions. In most cases and in early stages of deployment, it will make the production of primary basic materials such as cement, steel, or polyethylene more expensive. However, demand management, materials efficiency and more circular material flows can dampen the effect of such cost increases. In addition, the cost of energy intensive materials is typically a very small part of the total price of products, such as appliances, a bottle of soda or a building, so the effect on consumers is very small. Getting actors to pay more for zero emission materials is a challenge in supply chains with a strong focus on cutting costs, but it is not a significant problem for the broader economy. Reduced demand for services such as square meters of living space or kilometres of car travel is an option where material living standards are already high. If material living standards are very low, increased material use is often needed for more sustainable development. The options of materials and energy efficiency, and more circular material flows, generally have synergies with sustainable development. Increased use of electricity, hydrogen, CCU and CCS may have both positive and negative implications for sustainable development.

FAQ 11.3 What needs to happen for a low carbon industry transition?

Broad and sequential policy strategies for industrial development that pursue several mitigation options at the same time are more likely to reduce emissions cost effectively. Options such as materials and energy efficiency, albeit important, are not in themselves enough to reach zero emissions. Less electricity will be needed if electrification is pursued in parallel with options that reduce the need for the energy intensive virgin materials production. Based on shared visions or pathways for a zero-emission industry, industrial policy needs to support development of new technologies and solutions as well as market creation for low and zero emission materials and products. This implies coordination across policy domains including research and innovation, waste and recycling, product standards, digitisation, taxes, regional development, infrastructure, public procurement, permit procedures and more to make sustainable transition to carbon neutral industry worldwide.

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