WG III contribution to the Sixth Assessment Report

List of corrigenda to be implemented

The corrigenda listed below will be implemented in the Chapter during copy-editing.

CHAPTER 12

Document (Chapter, Annex, Supp. Material)	Page (Based on the final pdf FGD version)	Line	Detailed information on correction to make
Chapter 12	127-128	42-3	Li et al. (2020) calls for coordinated efforts to reduce emissions in trade flows in pairs of the economies with the highest leakage such as China and the United States, China and Germany, China and Japan, Russia and Germany. With: Li et al. (2020) calls for coordinated efforts to reduce emissions embodied in trade flows in pairs of economies with the highest leakage
Chapter 12	12-15	20	"Other technologies show mostly positive mitigation potentials ...." should be: "Other technologies show mostly positive mitigation costs ...."
Chapter 12	12-22	Industry, bottom	"The numbers for the industry sector typically have an uncertainty of ±25%" should be replaced by: "The numbers for the industry sector typically have an uncertainty of ±25%, unless indicated differently"
Chapter 12	39	1	In figure 2, the x-axis to show only years 2010 and 2100, everything in between removed
Chapter 12	40	12	OLD: "the annual net CO2 removal (i.e., gross removals, including A/R, minus gross emissions) on managed land reaches" - NEW: "the reported annual CO2 removal from AFOLU (mainly A/R) reaches"

Chapter 12	40	15	add footnote after the sentence ending with "percentile range)" - "Cumulative CDR from AFOLU cannot be quantified precisely because models use different reporting methodologies that in some cases combine gross emissions and removals, and use different baselines."
Chapter 12	40	18	OLD: "net CO2 removal on managed land" - NEW: "CO2 removal from AFOLU"
Chapter 12	41	2	Labels in Figure 12.3 OLD: Net removal on managed land - NEW: Removal from AFOLU
Chapter 12	41	3	OLD: "net CO2 removal on managed land (that is, gross removal through A/R minus emissions from deforestation)" - NEW: "CO2 removal from AFOLU (mainly A/R)
Chapter 12	41	3	OLD: Sequestration of three predominant CDR methods NEW: Sequestration through three predominant CDR methods
Chapter 12	41	1	OLD: "Net removal on managed land" - new: "AFOLU"
Chapter 12	41	1	OLD: "Net removal on managed land (n=344)" - new: "AFOLU (n = 344)"

Chapter 12	42	10	OLD: "AFOLU sector (through A/R)" - NEW "AFOLU (through A/R)"
Chapter 12	56	27	OLD: "Cumulative net CO2 removals on managed land (CDR through A/R minus land C losses due to deforestation)" - NEW: "cumulative CO2 removal from AFOLU (mainly through A/R), as reported from models, "
Chapter 12	4	16	OLD: "net CO2 removal on managed land (including A/R)" - NEW: "CO2 removal from AFOLU (mainly A/R)"
Chapter 12	4	16	NEW: If we can have a footnote in ES, then please add footnote to saying "Cumulative CDR from AFOLU cannot be quantified precisely because models use different reporting methodologies that in some cases combine gross emissions and removals, and use different baselines."
Chapter 12	4	19	OLD: "net CO2 removal on managed land (including A/R)" - NEW: "CO2 removal from AFOLU (mainly A/R)"
Chapter 12	4	16	OLD: "cumulative volumes of" - NEW: "reported cumulative volumes of"
Chapter 12	58	3	OLD: "CDR option" - New "CDR method"

Chapter 12: Cross-sectoral perspectives

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Executive summary

The total emission mitigation potential achievable by the year 2030, calculated based on sectoral assessments, is sufficient to reduce global greenhouse gas emissions to half of the current (2019) level or less (*robust evidence, high agreement*). This potential (32 to 44 $GtCO_2$-eq) requires implementation of a wide range of mitigation options. Options with mitigation costs lower than 20 USD tCO_2^{-1} make up more than half of this potential and are available for all sectors {12.2, Table 12.3}

Carbon Dioxide Removal (CDR) is a necessary element to achieve net zero CO_2 and GHG emissions both globally and nationally, counterbalancing residual emissions from hard-to-transition sectors. It is a key element in scenarios likely to limit warming to 2°C or lower by 2100 (*robust evidence, high agreement*). Implementation strategies need to reflect that CDR methods differ in terms of removal process, timescale of carbon storage, technological maturity, mitigation potential, cost, co-benefits, adverse side-effects, and governance requirements. All Illustrative Mitigation Pathways (IMPs) use land-based biological CDR (primarily Afforestation/Reforestation, A/R) and/or bioenergy with carbon capture and storage (BECCS) and some include direct air carbon capture and storage (DACCS). As a median value (5–95% range) across the scenarios likely limiting warming to 2°C or lower, cumulative volumes of BECCS, net CO_2 removal on managed land (including A/R), and DACCS reach 328 (168–763) $GtCO_2$, 252 (20–418) $GtCO_2$, and 29 (0–339) $GtCO_2$ for the 2020-2100 period, with annual volumes at 2.75 (0.52–9.45) $GtCO_2$ yr^{-1} for BECCS and 2.98 (0.23–6.38) $GtCO_2$ yr^{-1} for the net CO_2 removal on managed land (including A/R), and 0.02 (0–1.74) $GtCO_2$ yr^{-1} for DACCS, in 2050. {12.3, Cross-Chapter Box 8 in this chapter}

Despite limited current deployment, moderate to large future mitigation potentials are estimated for Direct Air Carbon Capture and Sequestration (DACCS), enhanced weathering (EW) and ocean-based CDR methods (including ocean alkalinity enhancement and ocean fertilisation) (*medium evidence, medium agreement*). The potential for DACCS (5–40 $GtCO_2$ yr^{-1}) is limited mainly by requirements for low-carbon energy and by cost (100-300 (full range: 84–386) USD tCO_2^{-1}). DACCS is currently at a medium technology readiness level. EW has the potential to remove 2–4 (full range: <1 to ~100) $GtCO_2$ yr^{-1}, at costs ranging from 50 to 200 (full range: 24–578) USD tCO_2^{-1}. Ocean-based methods have a combined potential to remove 1–100 $GtCO_2$ yr^{-1} at costs of 40–500 USD tCO_2^{-1}, but their feasibility is uncertain due to possible side-effects on the marine environment. EW and ocean-based methods are currently at a low technology readiness level. {12.3}

Realising the full mitigation potential from the food system requires change at all stages from producer to consumer and waste management, which can be facilitated through integrated policy packages *(robust evidence, high agreement*). Some 23-42% of global GHG emissions are associated with food systems, while there is still wide-spread food insecurity and malnutrition. Absolute GHG emissions from food systems increased from 14 to 17 $GtCO_2$-eq yr^{-1} in the period 1990-2018. Both supply and demand side measures are important to reduce the GHG intensity of food systems. Integrated food policy packages based on a combination of market-based, administrative, informative, and behavioural policies can reduce cost compared to uncoordinated interventions, address multiple sustainability goals, and increase acceptance across stakeholders and civil society *(limited evidence, medium agreement).* {7.2, 7.4, 12.4}

Diets high in plant protein and low in meat and dairy are associated with lower GHG emissions *(robust evidence, high agreement*). Ruminant meat shows the highest GHG intensity. Beef from dairy systems has lower emissions intensity than beef from beef herds (8-23 and 17-94 $kgCO_2$-eq (100g protein)$^{-1}$, respectively) when a share of emissions is allocated to dairy products. The wide variation in emissions reflects differences in production systems, which range from intensive feedlots with stock raised largely on grains through to rangeland and transhumance production systems. Where appropriate, a shift to diets with a higher share of plant protein, moderate intake of animal-source foods and reduced intake of added sugars, salt and saturated fats could lead to substantial decreases in GHG emissions. Benefits would also include reduced land occupation and nutrient losses to the surrounding environment, while at the same time providing health benefits and reducing mortality from diet-related non-communicable diseases. {7.4.5, 12.4}

Emerging food technologies such as cellular fermentation, cultured meat, plant-based alternatives to animal-based food products, and controlled environment agriculture, can bring substantial reduction in direct GHG emissions from food production *(limited evidence, high agreement)*. These technologies have lower land, water, and nutrient footprints, and address concerns over animal welfare. Access to low-carbon energy is needed to realize the full mitigation potential, as some emerging technologies are relatively more energy intensive. This also holds for deployment of cold chain and packaging technologies, which can help reduce food loss and waste, but increase energy and materials use in the food system. (*limited evidence, high agreement*). {11.4.1.3, 12.4}

Scenarios that likely to limit warming to 2°C or lower by 2100 commonly involve extensive mitigation in the AFOLU sector that at the same time provides biomass for mitigation in other sectors. Bioenergy is the most land intensive renewable energy option, but the total land occupation of other renewable energy options can become significant in high deployment scenarios *(robust evidence, high agreement)*. Growing demands for food, feed, biomaterials, and non-fossil fuels increase the competition for land and biomass while climate change creates additional stresses on land, exacerbating existing risks to livelihoods, biodiversity, human and ecosystem health, infrastructure, and food systems. Appropriate integration of bioenergy and other biobased systems, and of other mitigation options, with existing land and biomass uses can improve resource use efficiency, mitigate pressures on natural ecosystems and support adaptation through measures to combat land degradation, enhance food security, and improve resilience through maintenance of the productivity of the land resource base (*medium evidence, high agreement*). {3.2.5, 3.4.6, 12.5}

Bio-based products as part of a circular bioeconomy have potential to support adaptation and mitigation. Key to maximizing benefits and managing trade-offs are sectoral integration, transparent governance, and stakeholder involvement *(high confidence)*. A sustainable bioeconomy relying on biomass resources will need to be supported by technology innovation and international cooperation and governance of global trade to disincentivize environmental and social externalities (medium confidence). {12.5, Cross-Working Group Box 3}

Coordinated, cross-sectoral approaches to climate change mitigation should be adopted to target synergies and minimize trade-offs between sectors and with respect to sustainable development *(robust evidence, high agreement)*. This requires integrated planning using multiple-objective-multiple-impact policy frameworks. Strong inter-dependencies and cross-sectoral linkages create both opportunities for synergies and the need to address trade-offs related to mitigation options and technologies. This can only be done if coordinated sectoral approaches to climate change mitigations policies that mainstream these interactions are adopted. Integrated planning and cross-sectoral alignment of climate change policies are particularly evident in developing countries' NDCs pledged under the Paris Agreement, where key priority sectors such as agriculture and energy are closely aligned between the proposed mitigation and adaptation actions in the context of sustainable development and the SDGs. {12.6.2}

Carbon leakage is a critical cross-sectoral and cross-country consequence of differentiated climate policy *(robust evidence, medium agreement)*. Carbon leakage occurs when mitigation measures implemented in one country/sector lead to increased emissions in other countries/sectors. Global commodity value chains and associated international transport are important mechanisms of carbon leakage. Reducing emissions from the value chain and transportation can offer opportunities to mitigate three elements of cross-sectoral spill-overs and related leakage: 1) domestic cross-sectoral spill-overs within the same country; 2) international spill-overs within a single sector resulting from substitution of domestic production of carbon-intensive goods with their imports from abroad; and 3) international cross-sectoral spill-overs among sectors in different countries. {12.6.3}

Cross-sectoral considerations in mitigation finance are critical for the effectiveness of mitigation action as well as for balancing the often conflicting social, developmental, and environmental policy goals at the sectoral level *(medium evidence, medium agreement)*. True resource mobilisation plans that properly address mitigation costs and benefits at sectoral level cannot be developed in isolation of their cross-sectoral implications. There is an urgent need for multilateral financing institutions to align their frameworks and delivery mechanisms including the use of blended financing to facilitate cross-sectoral solutions as opposed to causing competition for resources among sectors. {12.6.4}

Understanding the co-benefits and trade-offs associated with mitigation is key to supporting societies to prioritize among the various sectoral policy options (*medium evidence, medium agreement*). For example, CDR options can have positive impacts on ecosystem services and the SDGs, but also potential adverse side-effects; transforming food systems has potential co-benefits for several SDGs, but also trade-offs; and land-based mitigation measures may have multiple co-benefits but may also be associated with trade-offs among environmental, social, and economic objectives. Therefore, the possible implementation of the different sectoral mitigation options would depend on how societies prioritise mitigation versus other products and services including food, material wellbeing, nature conservation and biodiversity protection, as well as on other considerations such as society's future dependence on CDR and on carbon-based energy and materials. {12.3, 12.4, 12.5, 12.6.1}

Governance of CDR, food systems and land-based mitigation can support effective and equitable policy implementation (*medium evidence, high agreement*). Effectively responding to climate change while advancing sustainable development will require coordinated efforts among a diverse set of state- and non-state-actors on global, multi-national, national, and sub-national levels. Governance arrangements in public policy domains that cut through traditional sectors are confronted with specific challenges, such as establishing reliable systems for monitoring, reporting and verification (MRV) that allow evaluation of mitigation outcomes and co-benefits. Effectively integrating CDR into mitigation portfolios can build on already existing rules, procedures and instruments for emissions abatement. Additionally, to accelerate research, development, and demonstration, and to incentivise CDR deployment, a political commitment to formal integration into existing climate policy frameworks is required, including reliable MRV of carbon flows. Food systems governance may be pioneered through local food policy initiatives complemented by national and international initiatives, but governance on the national level tends to be fragmented, and thus have limited capacity to address structural issues like inequities in access. The governance of land-based mitigation, including land-based CDR, can draw on lessons from previous experience with regulating biofuels and forest carbon; however, integrating these insights requires governance that goes beyond project-level approaches and emphasizes integrated land use planning and management within the frame of the SDGs. {7.4 Box 7.2, 7.6, 12.3.3, 12.4, 12.5}

12.1 Introduction

12.1.1 Chapter overview

The scope of this chapter was motivated by the need for a succinct bottom-up cross-sectoral view of greenhouse gas (GHG) emissions mitigation coupled with the desire to provide systemic perspectives of critical mitigation potentials and options that go beyond individual sectors and cover cross-sectoral topics such as food systems, land systems, and carbon dioxide removal (CDR) methods. Driven by this motivation, Chapter 12 provides a focused thematic assessment of CDR methods and food systems, followed by consideration of land-related impacts of mitigation options (land-based CDR and other mitigation options that occupy land) and other cross-sectoral impacts of mitigation, with emphasis on synergies and trade-offs between mitigation options, and between mitigation and other environmental and socio-economic objectives. The systems focus is unique to AR6 and is of critical policy relevance as it informs coordinated approaches to planning interventions that deliver multiple benefits and minimise trade-offs, and coordinated policy approaches to support such planning, to tap relatively under-explored areas for the strengthening and acceleration of mitigation efforts in the short to medium term, and for dealing with residual emissions in hard-to-transition sectors in the medium to long term.

Table 12.1 presents an overview of the cross-sectoral perspectives addressed in Chapter 12, mapping the chapter main themes to the sectoral and global chapters in this report. These mappings reflect the cross-sectoral aspects of mitigation options in the context of sustainable development, sectoral policy interactions, governance, implications in terms of international trade, spill-over effects, and competitiveness, and cross-sectoral financing options for mitigation. While some cross-sector technologies are covered in more detail in sectoral chapters, this chapter covers important cross-sectoral linkages and provides synthesis concerning costs and potentials of mitigation options, and co-benefits and trade-offs that can be associated with deployment of mitigation options. Additionally, Chapter 12 covers CDR methods and specific considerations related to land use and food systems, complementing Chapter 7. The literature assessed in the chapter includes both peer-reviewed and grey literature post IPCC AR5 including IPCC SR1.5, IPCC SRCCL and IPCC SROCC. Knowledge gaps are identified and reflected where encountered, as well as in a separate section. Finally, a strong link is maintained with sectoral chapters and the relevant global chapters of this report to ensure consistency.

12.1.2 Chapter content

Chapters 5 to 11 assess outcomes from mitigation measures that are applicable in individual sectors, and potential co-benefits and adverse side effects of these individual measures. Chapter 12 brings together the cross-sectoral aspects of these assessments including synergies and trade-offs as well as the implications of measures that have application in more than one sector and measures whose implementation in one sector impacts implementation in other sectors.

Taking stock of the sectoral mitigation assessments, Chapter 12 provides a summary synthesis of sectoral mitigation costs and potentials in the short and long term along with comparison to the top-down IAM assessment literature of Chapter 3 and the national/regional assessment literature of Chapter 4.

In the context of cross-sectoral synergies and trade-offs, the chapter identifies a number of mitigation measures that have application in more than one sector. Examples include measures involving product and material circularity, which contribute to mitigation of GHG emissions in a number of ways, such as treatment of organic waste to reduce methane emissions, avoid emissions through generation of renewable energy, and reduce emissions through substitution of synthetic fertilisers. Low carbon energy technologies such as solar and wind may be used for grid electricity supply, as embedded generation in the buildings sector (e.g., rooftop solar) and for energy supply in the agriculture sector. Nuclear and bio-based thermal electric generation can provide multiple synergies including base load to augment solar and wind, district heating, and seawater desalination. Grid-integrated hydrogen systems can buffer variability of solar and wind power and is being explored as a mitigation option in the transport and industry sectors. Carbon Capture and Storage (CCS) has potential application in a number of industrial processes (cement, iron and steel, petroleum refining and pulp and paper) and the fossil fuel electricity sector. When coupled with energy recovery from biomass (BECCS), CCS can help to provide CO_2 removal from the atmosphere. On the demand side, electric vehicles are also considered an option for balancing variable power, energy efficiency options find application across the sectors, as does reducing demand for goods and services, and improving material use efficiency. Focused inquiry into these areas of cross-sectoral perspectives is provided for CDR, food systems, and land-based mitigation options.

A range of examples of where mitigation measures result in cross-sectoral interactions and integration is identified. The mitigation potential of electric vehicles, including plug-in hybrids, is linked to the extent of decarbonisation of the electricity grid, as well as to the liquid fuel supply emissions profile. Making buildings energy positive, where excess energy is used to charge vehicles, can increase the potential of electric and hybrid vehicles. Advanced process control and process optimisation in industry can reduce energy demand and material inputs, which in turn can reduce emissions linked to resource extraction and manufacturing. Trees and green roofs planted to counter urban heat islands reduce the demand for energy for air conditioning and simultaneously sequester carbon. Material and product circularity contributes to mitigation, such as treatment of organic waste to reduce methane emissions, generate renewable energy, and to substitute for synthetic fertilisers.

The chapter also discusses cross-sectoral mitigation potential related to diffusion of General-Purpose Technologies (GPT), such as electrification, digitalisation, and hydrogen. Examples include the use of hydrogen as an energy carrier, which, when coupled with low carbon energy, has potential for driving mitigation in energy, industry, transport, and buildings (Box 12.5), and digitalisation has the potential for reducing GHG emissions through energy savings across multiple sectors.

The efficient realisation of the above examples of cross-sectoral mitigation would require careful design of government interventions across planning, policy, finance, governance, and capacity building fronts. In this respect, Chapter 12 assesses literature on cross-sectoral integrated policies, cross-sectoral financing solutions, cross-sectoral spill-overs and competitiveness effects, and on cross-sectoral governance for climate change mitigation.

Finally, in the context of cross-sectoral synergies and trade-offs, the chapter assesses the non-climate mitigation co-benefits and adverse effects in relation to SDGs, building on the fast-growing literature on the non-climate impacts of mitigation.

12.1.3 Chapter layout

The chapter is mapped into seven sections. Cost and potentials of mitigation technologies are discussed in Section 12.2, where a comparative assessment and a summary of sectoral mitigation cost and potentials is provided in coordination with the sectoral Chapters 5 to 11, along with a comparison to aggregate cost and potentials based on IAM outputs presented in Chapter 3.

Section 12.3 provides a synthesis of the state and potential contribution of CDR methods for addressing climate change. CDR options associated with the AFOLU and Energy sectors are dealt with in Chapters 6 and 7 and synthesised in Section 12.3. Other methods, not dealt with elsewhere, are covered in more detail. A comparative assessment is provided for the different CDR options in terms of costs, potentials, governance, impacts and risks, and synergies and trade-offs.

Section 12.4 assesses the literature on food systems and GHG emissions. The term 'food system' refers to a composite of elements (environment, people, inputs, processes, infrastructures, institutions, etc.) and activities that relate to the production, processing, distribution, preparation and consumption of food, and the outputs of these activities, including socio-economic and environmental outcomes. Climate change mitigation opportunities and related implications for sustainable development and adaptation are assessed, including those arising from food production, landscape impacts, supply chain and distribution, and diet shifts.

Section 12.5 provides a cross-sectoral perspective on land occupation and related impacts, risks and opportunities associated with land-based mitigation options as well as mitigation options that are not designated land-based, yet occupy land. It builds on SRCCL and Chapter 7 in this report, which covers mitigation in agriculture, forestry and other land use (AFOLU), including biomass production for mitigation in other sectors. In addition to an assessment of biophysical and socioeconomic risks, impacts and opportunities, this section includes a cross-chapter box (WGII and WGIII) on Mitigation and Adaptation via the Bioeconomy, and a box on Land Degradation Neutrality as a framework to manage trade-offs in land-based mitigation.

Section 12.6 provides a cross-sectoral perspective on mitigation, co-benefits, and trade-offs, including those related to sustainable development and adaptation. The synthesised sectoral mitigation synergies and trade-offs are mapped into options/technologies, policies, international trade, and finance domains. Cross-sectoral mitigation technologies fall into three categories in which the implementation of the technology: (i) occurs in parallel in more than one sector; (ii) could involve interaction between sectors, and/or (iii) could create resource competition among sectors. Policies that have direct sectoral effects include specific policies for reducing GHG emissions and non-climate policies that yield GHG emissions reductions as co-benefits. Policies may also have indirect cross-sectoral effects, including synergies and trade-offs that may, in addition, spill over to other countries.

Section 12.7 provides an overview of knowledge gaps, which could be used to inform further research.

Table 12.1 An overview of cross-sector perspectives addressed in Chapter 12

	Sectoral chapters							***Global chapters***				
Chapter 12 Themes	**Chapter 5**	**Chapter 6**	**Chapter 7**	**Chapter 8**	**Chapter 9**	**Chapter 10**	**Chapter 11**	**Chapter 13**	**Chapter 14**	**Chapter 15**	**Chapter 16**	**Chapter 17**
Costs & Potentials	Change in demand	Renewables CCU CCS Nuclear	Land-use Change	Urban planning Cities Demographics	Standards Electrification	Hybridisation Electric vehicles Fuel economy Decoupling	Technology Biomass CCU CCS	Enabling of mitigation		Finance of mitigation		Synergies and trade-offs to SDGs
CDR		BECCS	Land-based CDR		C storage in buildings				International Governance			
Food Systems	Food demand Wellbeing	Energy demand of some emerging mitigation options	Agricultural production Demand side measures	Urban food systems; controlled environment agriculture		Food transport	Food processing & packaging	Food system transformation	Governance			Food system and SDGs

Mitigation & land use		Land use/ occupation: bioenergy, hydro, solar, wind, nuclear	A/R, Biomass production, Bioenergy, Biochar		Land use and biomass supply	Land use and biomass supply	Land use and biomass supply		Governance			Co-benefits and adverse side effects
Cross-sectoral perspectives	Electrification, Hydrogen, Digitalisation , Circularity, Synergies, Trade-offs, Spill-overs							Policy interactions Policy packages Case studies Value chain & carbon leakage	Governance Leakage	Blended financing	General Purpose Technologies Electrification Hydrogen	SDGs Co-benefits Trade-offs Adaptation

12.2 Aggregation of sectoral costs and potentials

The aim of this section is to provide a consolidated overview of the net emissions reduction potentials and costs for mitigation options available in the various sectors dealt with in the sectoral chapters 6, 7, 9, 10 and 11 of this assessment report. This overview provides policy-makers with an understanding of which options are more or less important in terms of mitigating emissions in the short term (here interpreted as 2030), and which ones are more or less costly. The intention is not to provide a high level of accuracy for each technology cost or potential, but rather to indicate relative importance on a global scale and whether costs are low, intermediate or high. The section starts with an introduction (Section 12.2.1), providing definitions and the background. Next, ranges of net emission reduction potentials and the associated costs for the year 2030 are presented (Section 12.2.2) and compared to earlier estimates and with the outputs of integrated assessment models (IAMs) (Section 12.2.3). Finally, an outlook to the year 2050 is provided (Section 12.2.4).

12.2.1 Introduction

The term 'mitigation potential' is used here to report the quantity of net greenhouse gas emissions reductions that can be achieved by a given mitigation option relative to specified reference scenario. The net greenhouse gas emission reduction is the sum of reduced emissions and enhanced sinks. Several types of potential can be distinguished. The technical potential is the mitigation potential constrained by theoretical limits in addition to the availability of technology and practices. Quantification of technical potentials primarily takes into account technical considerations, but social, economic and/or environmental considerations are sometimes also considered, if these represent strong barriers for the deployment of an option. The economic potential, being the potential reported in this section, is the proportion of the technical potential for which the social benefits exceed the social costs, taking into account a social discount rate and the value of externalities (see glossary). In this section, only externalities related to greenhouse gas emissions are taken into account. They are represented by using different cost cut-off levels of options in terms of USD per tonne of avoided CO_2-eq emissions. Other potentials, such as market potentials, could also be considered, but they are not included in this section.

The analysis presented here is based, as far as possible, on information contained in Chapters 6, 7, 9, 10 and 11, where costs and potentials, referred to here as 'sectoral mitigation potentials' have been discussed for each individual sector. In the past, these were designated as bottom-up potentials, in contrast to the top-down potentials that are obtained from integrated energy-economic models and IAMs. However, IAMs increasingly include 'bottom-up' elements, which makes the distinction less clear. Still, sectoral studies often have more technical and economic detail than IAMs. They may also provide more up-to-date information on technology options and associated costs. However, aggregation of results from sectoral studies is more complex, and although interactions and overlap are corrected for as far as possible in this analysis, it is recognised that such systemic effects are much more rigorously taken into account in IAMs. A comparison is made between the sectoral results and the outcomes of the IAMs in Section 12.2.3.

Costs of mitigation options will change over time. For many technologies, costs will reduce as a result of technological learning. An attempt has been made to take into account the average, implementation-weighted costs until 2030. However, the underlying literature did not always allow such costs to be presented. For the year 2030, the results are presented similarly to AR4, with a breakdown of the potential in "cost bins". For the year 2050, a more qualitative approach is provided. The origins of the cost data in this section mostly are based on studies carried out in the period 2015-2020. Given the wide range of the cost bins that are used in this section it is not meaningful (and often not possible) to convert to USD-values for one specific year. This may lead to some extra uncertainty, but this is expected to be relatively small.

As indicated previously, net emission reduction potentials are presented based on comparison with a reference scenario. Unfortunately, not all costs and potentials found in the literature are determined against the same reference scenarios. In this assessment reference scenarios are based on what were assumed current-policy scenarios in the period 2015-2019. Typical reference scenarios are the SSP2 scenarios (Fricko et al. 2017) and the Current Policies scenario from the World Energy Outlook 2019 (IEA 2019). They can both be considered scenarios with middle-of-the-road expectations on population growth and economic development, but there are still some differences between the two (Table 12.2). The net emission reduction potentials reported here were generally based on analyses carried out before 2020, so the impact of the COVID-19 pandemic was not taken into account. For comparison, the Stated Policies scenario of the World Energy Outlook 2020 (IEA 2020a) is also shown, one of the scenarios in which the impact of COVID-19 was considered. Variations of up to 10% between the different reference scenarios exist with respect to macro-variables such as total primary energy use and total GHG emissions. The potential estimates presented below should be interpreted against this background. The total emissions under the reference scenarios in 2030 are expected to be in the range of 54 to 68 $GtCO_2$-eq yr^{-1} with a median of 60 $GtCO_2$-eq yr^{-1} (Chapter 4, Table 4.1).

For the energy sector the potentials are determined using the World Energy Outlook 2019 Current Policies Scenario as a reference (IEA 2019). However, for the economic assessment more recent LCOEs for different electricity generating technologies were used (IEA 2020a). For the agriculture, forestry and other land use (AFOLU) sector, the potentials were derived from a variety of studies. It may be expected that the best estimates, as averages, match with the reference in a middle-of-the-road scenario. For the buildings sector the Current Policies scenario of World Energy Outlook 2019 (IEA 2019) was used as a reference. For the transport sector, the references of the underlying sources were used. For the industry sector, the scenarios used have emissions that are slightly higher than in the Current Policies scenario from the World Energy Outlook 2019 (IEA 2019).

Table 12.2: Key characteristics of the scenarios that are used as a reference for determining costs and potentials. The values are for the year 2030.

	SSP2 reference (MESSAGE-GLOBIOM) (Fricko et al. 2017)	All reference scenarios median (25/75 per-centiles in parenthesis) (AR6 scenarios database)	WEO-2019 (Current Policies) (IEA 2019)	WEO-2020 (Stated Policies) (IEA 2020a)	AR6 Chapter 4 (Chapter 4, Table 4.1)
Real GDP (PPP) (10^{12} USD)	158 (USD_{2010})	159 (154–171)	3.6% p.a.↑ (2018 to 2030)	2.9% p.a.↑ (2019 to 2030)	
Population (billion)	8.30	8.30 (8.20–8.34)	8.60		
Total primary energy use (EJ)	627	670 (635–718)	710	660	
Total final energy use (EJ)	499	480 (457–508)	502	472	
Energy-related CO_2 emissions (Gt)	33.0	37.9 (34.7–41.4)	37.4	33.2*	37 (35–45)
CO_2 emissions energy and industry (Gt)	37.9	42.3 (39.0–45.8)		36.0	
Total CO_2 (emissions Gt)	40.6	45.7 (41.8–49.4)			43 (38–51)
Total greenhouse gas emissions ($GtCO_2$-eq)	52.7	59.7 (55.0–65.8)			60 (54-68)

*The difference between WEO-2020 and WEO-2019 is partly explained by the fact that WEO-2019 had two different reference scenarios: Current Policies and Stated Policies. WEO-2020 has only one reference: the Stated Policies Scenario (STEPS), which "is based on today's policy settings". The Stated Policy scenario in WEO-2019 had energy-related emissions of 34.9 $GtCO_2$.

12.2.2 Costs and potentials of options for 2030

In this section, we present an overview of mitigation options per sector. An overview of net emission reduction potentials for different mitigation options is presented in Table 12.3.

Firstly, a brief overview of the process of data collection is presented, with a more detailed overview being found in Supplementary Material SM 12.A.2. For the energy sector, the starting point for the determination of the emission reduction potentials was the Emissions Gap Report (UNEP 2017), but new literature was also assessed, and a few studies that provide updated estimates of the mitigation potentials were included. It was found that higher mitigation potentials than in the UNEP report are now reported for solar and wind energy, but at the same time electricity production by solar and wind energy in the reference scenario has increased, compared to earlier versions of the World Energy Outlook. The net effect is a modest increase of the average value of the potential, and a wider uncertainty range. Costs of electricity generating technologies are discussed in Chapter 6, (Section 6.4.7) with a summary of LCOEs from the literature being presented in Section 6.4.7. Mitigation costs of electricity production technology depend on local conditions and on the baseline technology being displaced, and it is difficult to determine the distribution over the cost ranges used in this assessment. However, it is possible to indicate a broad cost range for these technologies. These cost ranges are presented in Table 12.3. For onshore wind and utility scale solar energy, there is strong evidence that despite regional difference in resource potential and cost, a large part of the mitigation potential can be found in the negative cost category or at cost parity with fossil fuel based options. This is also the case for nuclear energy in some regions. Other technologies show mostly positive mitigation potentials, the highest mitigation costs are for CCS, bioelectricity with CCS, for details see Supplementary Material SM 12.A.2.

For the AFOLU sector, assessments of global net emission reduction studies were provided by Chapter 7 (Table 7.3). The number of studies depends on the type of mitigation action, but ranges from 5 to 9. Each of these studies relies on a much larger number of underlying data sources. From these studies, emission reduction ranges and best estimates were derived. The studies presented refer to different years in the period 2020 to 2050, and the mitigation potential presented for AFOLU primarily refers to the average over the period 2020 to 2050. However, because most of the activities involve storage of carbon in stocks that accumulate carbon, or conversely decay over time (e.g., forests, mangroves, peatland soils, agricultural soils, wood products), the 2020 to 2050 average provides a good approximation of the amount of permanent atmospheric CO_2 mitigation that could be available at a given price in 2030. The exception is BECCS which is in an early upscaling phase, so the potential estimated by Chapter 7 as an average for the 2020 to 2050 period is not included in Table 12.3. Note that for the energy sector a mitigation potential for BECCS is provided in Table 12.3.

The emission reduction potentials for the buildings sector were based on the analysis by Chapter 9 authors of a large number of sectoral studies for individual countries or regions. In total, the chapter analysed the results of 67 studies that assess the potential of technological energy efficiency and onsite renewable energy production and use, and the results of 11 studies that assess the potential of sufficiency measures helping avoid demand for energy and materials. The sufficiency measures were included in models by reorganization of human activities, efficient design, planning, and use of building space, higher density of building and settlement inhabitancy, redefining and downsizing goods and equipment, limiting their use to health, living, and working standards, and their sharing. Most of these studies targeted 2050 for the decarbonisation of buildings; the potentials in 2030 reported here rely on the estimates for 2030 provided by these studies or on the interpolated estimates targeting these 2050 figures. Based on these individual country studies, regional aggregate emission reduction percentages were found. The potential estimates were assembled in the order sufficiency, efficiency, renewable options, correcting the amount of the potential at each step for the interaction with preceding measures. Note that the option 'Enhanced use of wood products' was analysed by Chapter 7, but is listed under the buildings sector in Table 12.3, as such enhanced use of wood takes place predominantly in the construction sector.

For the transport sector, Chapter 10 provided data on the emission reduction potential for shipping. For the other transportation modes, additional sources were used to achieve a complete overview of emission reduction potentials (for further details, see Supplementary Material 12.A.2). A limited number of estimates for global emission reduction potential is available: the total number of sources is about 10, and some estimates rely on just one source. The data have been coordinated with Chapter 10 authors.

For the industrial sector, global emission reduction potentials per technology class per sector were derived by Chapter 11 authors, using primarily sectoral or technology-oriented literature. The analysis is based on about 75 studies, including sectoral assessments (Sections 11.4.1, 11.4.2, Figure 11.13).

For methane emission reduction from oil and gas operations, coal mining, waste treatment and wastewater, an analysis was done, based on three major data sources in this area (US EPA 2019; Harmsen et al. 2019; Höglund-Isaksson et al. 2020), and for oil and gas operations complemented by (IEA 2021a). A similar analysis for reductions of emissions of fluorinated gases was carried out based on analysis by the same institutes (Purohit and Höglund-Isaksson 2017; US EPA 2019; Harmsen et al. 2019). Data for CDR options not discussed previously (such as DACCS and enhanced weathering) were taken from Section 12.3. For more details about data sources and data processing, see Supplementary Material 12.A, Section SM 12.A.2.

In Table 12.4 mitigation potentials for all gases are presented in $GtCO_2$-eq. For most sectors the mitigation potentials (notably for methane emissions reductions from coal, oil and gas, waste and wastewater) have been converted to CO_2-eq using global warming potentials (GWP) values as presented in the 6th Assessment Report (Cross-Chapter Box 2 in Chapter 2). However, the underlying literature did not always accommodate this, in which cases older GWP values apply. Given the uncertainty ranges in the mitigation potentials in Table 12.3, the impact on the results of using different GWP values is considered to be very small.

Table 12.3: Detailed overview of global net GHG emission reduction potentials (GtCO$_2$-eq) in the various cost categories for the year 2030. Note that potentials within and across sectors cannot be summed, as the adoption of some options may affect the mitigation potentials of other options. Only monetary costs and benefits of options are taken into account. Negative costs occur when the benefits are higher than the costs. For wind energy, for example, this is the case if production costs are lower than those of the fossil alternatives. Ranges are indicated for each option separately, or indicated for the sector as a whole (see column "Notes"); they reflect full ranges. Cost ranges are not cumulative, e.g., to obtain the full potential below 50 USD tCO$_2$-eq^{-1}, the potentials in the cost bins <0, 0–20 and 20–50 USD tCO$_2$-eq^{-1} need to be summed together.

<table>
<tr><th rowspan="2">Emission reduction options (including carbon sequestration options)</th><th colspan="5">Cost categories (USD tCO$_2$-eq^{-1})</th><th rowspan="2">Notes</th></tr>
<tr><th><0</th><th>0–20</th><th>20–50</th><th>50–100</th><th>100–200</th></tr>
<tr><td><i>Energy sector</i></td><td></td><td></td><td></td><td></td><td></td><td><i>Cost ranges are derived as ranges of LCOEs for different electricity generating technologies and the potentials are updated from UNEP (2017).</i></td></tr>
<tr><td>Wind energy</td><td colspan="3">2.1–5.6
(majority in <0 range)</td><td></td><td></td><td>Costs for system integration of intermittent renewables are not included, but these are expected to have limited impact until 2030 and will depend on market design and cross-sectoral integration</td></tr>
<tr><td>Solar energy</td><td colspan="4">2.0–7.0
(majority in <0 range)</td><td></td><td>Ibid.</td></tr>
<tr><td>Nuclear energy</td><td colspan="5">0.88 ±50%</td><td></td></tr>
<tr><td>Bioelectricity</td><td></td><td colspan="2"></td><td colspan="2">0.86 ±50%</td><td>Biomass use for indoor heating and industrial heat is not included here. Currently, about 90% of renewable industrial heat consumption is biobased, mainly in industries that can use their own biomass waste and residues (IEA, 2020)</td></tr>
<tr><td>Hydropower</td><td></td><td colspan="3">0.32 ±50%</td><td></td><td>Mitigation costs show large variation and may end up beyond these ranges.</td></tr>
</table>

Geothermal energy		0.74 ±50%				Mitigation costs show large variation and may end up beyond these ranges.
Carbon capture and storage (CCS)				0.54 ±50%		
Bioelectricity with CCS				0.30 ±50%		
CH_4 emission reduction from coal mining	0.04 (0.01–0.06)	0.41 (0.15–0.64)	0.03 (0.02–0.05)	0.02 (0.01–0.03)		
CH_4 emission reduction from oil and gas operations	0.31 (0.12–0.56)	0.61 (0.23–1.30)	0.07 (0.03–0.20)	0.06 (0.00–0.29)	0.10 (0–0.29)	
Land-based mitigation options (including agriculture and forestry)						*Potentials for AFOLU are averages for the period 2020–2050, and represent a proxy for mitigation in 2030.* *Technical potentials listed below include the potentials already listed in the previous columns.* *Note that in Table 7.3 the same potentials are listed, but they are cumulative over the cost bins.*
Carbon sequestration in agriculture (soil carbon sequestration, agroforestry and biochar application)		0.50 (0.38–0.60)	0.73 (0.5–1.0)	2.21 (0.6–3.9)		Technical potential: 9.5 (range 1.1–25.3)
CH_4 and N_2O emission reduction in agriculture (reduced enteric fermentation, improved		0.35	-	0.28		Technical potential: 1.7 (range 0.5–3.2)

manure management, nutrient management, rice cultivation)		(0.11–0.84)		(0.19–0.46)		GWPs used from AR4 and AR5
Protection of natural ecosystems (avoid deforestation, loss and degradation of peatlands, coastal wetlands and grasslands)		2.28 (1.7–2.9)	0.12 (0.06–0.18)	1.63 (1.3–4.2)	0.22 (0.09–045	Technical potential 6.2 (range 2.8–14.4)
Restoration (afforestation, reforestation, peatland restoration, coastal wetland restoration)		0.15	0.57 (0.2–1.5)	1.46 (0.6–2.3)	0.66 (0.4–1.1)	Technical potential 5.0 (range 1.1–12.3)
Improved forest management, fire management		0.38 (0.32–0.44)	-	0.78 (0.32–1.44)		Technical potential 1.8 (range 1.1–2.8)
Reduce food loss and food waste						Feasible potential 0.5 (0.1–0.9) Technical potential 0.7 (0.1–1.6) Estimates reflect direct mitigation from diverted agricultural production only, not including land-use effects
Shift to sustainable healthy diets						Feasible potential 1.7 (1.0–2.7) Technical potential 3.5 (2.1–5.5) Estimates reflect direct mitigation from diverted agricultural production only, not including land-use effects

Buildings						*The numbers were corrected for the potential overlap between options in the order "sufficiency, efficiency, renewable measures" and they could be therefore added up. In 2050, much larger and cheaper potential is available (see 9.6 in Chapter 9); the potential in 2030 is lower and more expensive mostly due to various feasibility constrains.*
Sufficiency to avoid demand for energy services (e.g., efficient building use and increased inhabitancy and density)	0.56 (0.28–0.84)					
Efficient lighting, appliances and equipment, including ICT, water heating and cooking technologies	0.73 (0.54–0.91)					
New buildings with very high energy performance (change in construction methods, management and operation of buildings, efficient heating, ventilation and air conditioning)		0.35 (0.26–0.53)			0.83 (0.62–1.24)	
Onsite renewable production and use (often backed-up with demand-side flexibility and digitalization measures, typically installed in very new high energy performance buildings)		0.20 (0.15–0.30)			0.27 (0.20–0.40)	

Improvement of existing building stock (thermal efficiency of building envelopes, management and operation of buildings, and efficient heating, ventilation and air conditioning leading to “deep” energy savings)		0.27 (0.20–0.34)				Additionally, there is 0.50 (range 0.37–0.62) $GtCO_2$-eq of potential above a price of 200 USD tCO_2-eq^{-1}
Enhanced use of wood products						Technical potential 1.0 (range 0.04–3.7) Economic potential 0.38 (range 0.3–0.5) (varying carbon prices). Potential is mainly in the construction sector.
Transport						Options for the transportation sector have an uncertainty of ±50%.
Light duty vehicles – fuel efficiency	0.6					
Light duty vehicles – electric vehicles	0.5–0.7					Depending on the carbon intensity of the electricity supplied to the vehicles.
Light duty vehicles – shift to public transport	0.5					
Light duty vehicles – shift to bikes and e-bikes	0.2					
Heavy duty vehicles – fuel efficiency	0.4					
Heavy duty vehicles – electric vehicles	0.2					
Heavy duty vehicles – shift to rail						No data available.
Shipping – efficiency, optimisation, biofuels	0.5 (0.4–0.7)					

Aviation – energy efficiency	0.12–0.32					Limited evidence
Biofuels		0.6–0.8				
Industry						*The numbers for the industry sector typically have an uncertainty of ±25%.* *The numbers are corrected for overlap between the options, except for the 0.15 GtCO$_2$ potential in the highest cost bin. For the rest they can be aggregated to provide full potentials.*
Energy efficiency		1.14				This only applies to more efficient use of fuels. More efficient use of electricity is not included.
Material efficiency			0.93			
Circularity (enhanced recycling)			0.48			
Fuel switching			1.28	0.67	0.15	
Feedstock decarbonisation, process change				0.38		
Carbon capture, utilization and storage (CCU and CCS)					0.15	
Cementitious material substitution			0.28			
Reduction of non-CO$_2$ emissions		0.2				

Cross-sectorial						
Emission reduction of fluorinated gases	0.26 (0.01–0.50)	0.68 (0.55–0.90)	0.18 (0.01–0.42)	0.09 (0–0.20)	0.03 (0–0.05)	GWPs not updated
Reduction of CH_4 emissions from solid waste	0.33 (0.24–0.43)	0.11 (0.03–0.15)	0.06 (0.03–0.08)	0.04 (0.01–0.10)	0.08 (0.02–0.12)	
Reduction of CH_4 emissions from wastewater	0.02 (0–0.05)	0.03 (0.01–0.05)	0.04 (0.01–0.07)	0.03 (0.02–0.04)	0.07 (0.01–0.16)	
Direct air carbon capture and storage					very small	There is potential in these categories, but given the current technology readiness levels, for 2030 the potential is limited. Also, it is not certain whether the costs will already drop below 200 USD tCO_2^{-1} before 2030. In the longer term, much larger potentials are projected, see Section 12.3.1.
Enhanced weathering					very small	

For all options, uncertainty ranges of the mitigation potentials are given in Table 12.3. As far as possible, the ranges represent the variation in assessments found in the literature. This is the case for wind and solar energy, for the AFOLU options, for the methane mitigation options (coal, oil and gas, waste and wastewater) and for fluorinated gas mitigation. For the latter options, some variability exists for each cost bin, but aggregated over cost ranges the variation is much smaller, typically ±50%. For the buildings sector and the industrial sector options, the uncertainty in the mitigation potential is estimated by the lead authors of Chapters. For options for which only limited sources were available an uncertainty range of ±50% was used. Overall, the uncertainty range per option is typically in the range of ±20% to ±60%.

Despite these uncertainties, clearly a number of options with high potentials can be identified, including solar energy, wind energy, reducing conversion of forests and other natural ecosystems, and restoration of forests and other natural ecosystems. As mid-range values, they each represent 4 to 7% of total reference emissions for 2030. Soil carbon sequestration in agriculture and fuel switching in industry can also be considered as options with high potential, although it should be noted that these options consist of a number of discernible sub-options, see Table 12.3. It can be observed that for each sector, a variety of options is available. Many of the smaller options each make up 1 to 2% of the reference emissions for 2030. Within this group of smaller options there are some categories that, summed together, stand out as substantial: the energy efficiency options and the methane mitigations options.

Costs are highly variable across the options. All sectors have several options for which at least part of the potential has mitigation costs below 20 USD tCO_2^{-1}. The only exception is the industrial sector, in which only energy efficiency is available below this cost level. At the same time, a substantial part of the emission reduction potential comes at higher cost, much being in the 20 to 100 USD tCO_2^{-1} cost ranges. All sectors have substantial additional potential in these cost ranges; only for transportation is this limited. Aggregation of the potentials per cost bin shows that the potential in these cost bins is marginally smaller than in the two cheapest cost bins. For some options, potential was identified in the 100 to 200 tCO_2^{-1} cost bin. The mitigation potentials identified in this cost range make up only a small part of the total mitigation potential. It could be that there is limited potential in this range; however, a more plausible explanation, supported by several authors of sectoral chapters, is that this cost range is relatively unexplored.

In this assessment, the emphasis is on the specific mitigation costs of the various options, and these are often considered as an indicator to prioritise options. However, in such a prioritisation, other elements will also play a role, like the development of technology for the longer term (Section 12.2.4) and the need to optimise investments over longer time periods, see for example Vogt-Schilb et al. (2018) who argue that sometimes it makes sense to start with implementing the most expensive option.

In this section, an overview of emission mitigation options for the year 2030 was presented. The overview of the mitigation potential is based on a variety of approaches, relying on a large number of sources, and the number of sources varied strongly from sector to sector. The main conclusions from this section are: i) there is a variety of options per sector, ii) per sector the options combined show significant mitigation potential, iii) there are a few major options and a lot of smaller ones, and iv) more than half of the potential comes at costs below 20 USD tCO_2^{-1} (between sectors: *medium to robust evidence, high agreement*).

12.2.3 Aggregation of sectoral results and comparison with earlier analyses and Integrated Assessment Models

In this section, the mitigation potentials are aggregated per sector, and then to the global economy. These potentials, which are based on sectoral analysis, are then compared to the results from earlier assessments and the results from Integrated Assessment Models (IAMs). Given the incompleteness of data on the mitigation potential at mitigation costs larger than 100 USD tCO_2^{-1}, the focus will be on options with mitigation costs below 100 USD tCO_2^{-1}.

As suggested previously, the overview presented in Table 12.3 should be interpreted with care, as the implementation of one option may affect the mitigation potential of another option. Most sectoral chapters have supplied mitigation potentials that were already adjusted for overlap and mutual influences (industry, buildings, AFOLU). For the energy sector, interactions between the options will occur, but parallel implementation of all the options seems to be possible; if all options at costs levels below 100 USD tCO_2^{-1} would be implemented, this would lead to an additional power generation with no direct CO_2 emissions of 41% of the total projected generation in 2030. This seems to be possible, but as higher penetrations are relatively unexplored, we apply a smaller uncertainty range at the high end. For the calculation of the aggregate potentials in the energy sector, error propagation rules were applied. For the transport sector, there will be interaction between the technical measures on the one hand and the modal shift measures on the other hand. Given the small mitigation contribution of the modal shift options, these interactions will be negligible. The resulting aggregate mitigation potentials and their uncertainty ranges per (sub)sector are given in Table 12.4 (columns indicated as 'AR6'). This overview confirms the large potentials per sector, even when taking the uncertainty ranges into account.

Calculating aggregated mitigation potentials for the global economy requires that interactions between sectors also need to be taken into account (Section 12.6). First of all, there may be overlap between the electricity supply sector and the electricity demand sectors: if the electricity sector is extensively decarbonized, the avoided emissions due to electricity efficiency measures and local electricity production will be significantly reduced. Therefore, this demand-side mitigation potential is only taken into account for 25% (reflecting the degree of further decarbonisation of the power sector) in the cross-sectoral aggregation. For the other demand sectors, this problem does not arise. The industry sector did not provide estimates for electricity efficiency improvement and in the transport sector the utilization of electricity to date is very low. Electrification options may occur in all sectors, but this enhances the mitigation potential in combination with a decreased carbon intensity of the power sector. For other energy sector options, methane emission reduction from coal, oil and natural gas operations, the situation is more complex. The total emission reduction potential for fossil fuels in the other sectors is high. Should this potential be realised, this would lead to a reduction of the potential reported here. However, reducing fossil fuel use also leads to a reduction in the upstream CH_4 emissions, so in the case of reducing fossil fuel use, these upstream emissions will also be avoided, so no overestimate of the aggregate emission reduction potential occurs.

The total potential, given these corrections for overlap, leads to a mid-range value for the total mitigation potential at costs below 100 USD $tCO_2\text{-eq}^{-1}$ of 38 $GtCO_2$-eq. Given the fact that it is not to be expected that mitigation potentials of the various sectors are mutually correlated, i.e. it is not to be expected that mitigation potentials are all on the high side or all on the low side), the ranges are aggregated using error propagation rules, which leads to a range for the mitigation potential of 32 to 44 $GtCO_2$-eq.

Table 12.4 Overview of aggregate sectoral net GHG emission reduction potentials (GtCO$_2$-eq) for the year 2030 at costs below 100 USD tCO$_2$-eq^{-1}. Comparisons with earlier assessments are also provided. Note that sectors are not entirely comparable across the three different estimates.

Sector	**Mitigation potentials at costs less than 100 USD tCO$_2$-eq^{-1}**				
	AR6 best estimate	AR6 range	AR4 (Barker et al. 2007)	UNEP-2017 best estimate (UNEP 2017)	UNEP- 2017 range (UNEP 2017)
Electricity sector	11.0	7.9–12.5	6.2–9.3	10.3	9.5–11.0
Other energy sector (methane)	1.6	1.1–2.1		2.2	1.7–2.6
Agriculture	4.1	1.7–6.7	2.3–6.4	4.8	3.6–6.0
Forestry and other land-use related options	7.3	3.9–13.1	1.3–4.2	5.3	4.1–6.5
AFOLU demand-side options (estimates reflect direct mitigation from **diverted agricultural production only, not including land-use effects)**	2.2	1.1–3.6			1.3–3.4
Buildings (potentials up to 200 USD t**CO$_2$-eq^{-1} in parentheses)**	Dir 0.7 (1.1) Ind 1.3 (2.1) Tot 2.0 (3.2)	0.5–1.0 (0.7–1.5) 0.9–1.8 (1.5–3.1) 1.4–2.9 (2.3–4.6)	Dir 2.3–2.9 Ind 3.0–3.8 Tot 5.4–6.7	Dir 1.9 Ind 4.0 Tot 5.9	Dir 1.6–2.1

Transport	3.8	1.9–5.7	1.6–2.5	4.7	4.1–5.3
Industry	Dir 5.4	4.0–6.7	Dir 2.3–4.9 Ind 0.83 Tot 3.1–5.7	Dir 3.9 Ind 1.9 Tot 5.8	Dir 3.0–4.8
Fluorinated gases (all sectors)	1.2	0.7–1.5	NE	1.5	1.2–1.8
Waste and wastewater	0.7	0.6–0.8	0.4–1.0	0.4	0.3–0.5
Enhanced weathering	-	-	-	1.0	0.7–1.2
Total of all sectors	**38**	**32–44**	**15.8–31.1**	**38**	**35–41**

Dir = reduction of direct emissions, Ind = reduction of indirect emissions (related to electricity production), Tot = reduction of total emissions, NA = not applicable, NE = not estimated, AR4: Table 11.3, UNEP-2017: Chapter 4.

Mitigation costs and potentials for 2030 have been presented previously, notably in the AR4 Chapter 11 on Mitigation from a cross-sectoral perspective (Barker et al. 2007) and the Emissions Gap Report (UNEP 2017). Note that AR5 did not provide emission reduction potentials in this form. The aggregated potentials reported here are higher than those estimated in AR4. Note however, that AR4 suggested the potentials were underestimated by 10 to 15%, but a higher potential still remains in the current assessment. In a sector-by-sector comparison, higher potentials than in AR4 can be observed especially for the energy sector and the forestry sector, and to a more limited extent for the industry sector and the transport sector. For the energy sector, the change can largely be explained by the higher estimates for wind and solar energy and the improved understanding of how to integrate high shares of intermittent renewable energy sources into power systems. For industry and transport, the higher potentials can be partly explained by the inclusion of more options, like recycling and material efficiency (for industry) and electric transportation and modal shifts for transport. For buildings a lower potential can be observed compared to AR4, one reason is that the 2030 reference direct and indirect emissions were estimated as 45% and 11% higher in AR4 than they were in AR6 (signalling a much quicker actual switch to electricity than was thought 15 to 20 years ago, among other reasons). The other reason for a difference is that the scenarios considered in AR4 had 25 to 30 years between their start year until the target year of 2030 and the scenarios reviewed in AR6 has only 10 to 15 years before 2030. The current retrofitting rates of existing buildings and penetration rates of nearly zero energy buildings do not allow decarbonizing the sector over 10-15 years, but they do over a longer time period. A much larger potential than reported here for 2030 can still be realized in the timeframe up to 2050 (Section 9.6.2).

Another global analysis was done by McKinsey (2009) which presents a marginal abatement cost curve for 2030, suggesting a total potential of 38 $GtCO_2$-eq (note that the reference for this study is 70 $GtCO_2$-eq, which is at the high end of the reference range used in this assessment).

The potentials reported here are comparable with UNEP (2017). Note that material for the energy sector from the UNEP report was partly reused in this analysis. Furthermore, some options for the transport sector (aviation and biofuels) were identical to the estimates in the UNEP report. The remaining mitigation potentials are all based on new – and much more extended – assessment. There are some notable changes. The AR6 mitigation potential for forestry is substantially larger. For buildings the potential is smaller, mainly related to the smaller mitigation potential for electric appliances than in the UNEP report. But overall, the estimates of the total mitigation potential are well aligned, which confirms there is substantial consistency across various emissions estimates.

The results of the sectoral mitigation potentials are also compared with mitigation impacts as calculated by IAMs. To this end, cumulative sectoral potentials over cost ranges were determined, based on the information in Table 12.3. For options that are in various cost ranges, we assumed that they are evenly distributed over these cost ranges. The only exception is wind and solar energy, for which it is indicated that the majority of the mitigation potential is in the negative cost range. It was assumed that the fraction in the negative cost range was 60%; the remainder is evenly distributed over the other cost ranges. These cumulative potentials were compared with emission reductions realized in IAMs at certain price levels for CO_2. Note that these price levels selected in IAMs are average price levels – not all IAMs use globally uniform carbon prices, so underlying these cost levels, there may be regional differentiation. Data were taken from the AR6 scenarios database. Note that, strictly speaking, not all models in the database are IAMs; in this analysis all models in the database were used, but the term IAMs is used as shorthand in the text that follows. All scenarios likely to limit warming to 2˚C or lower are included for the comparison (i.e., the categories of scenarios C1-C3 in Chapter 3). A comparison per sector is provided in Figure 12.1. It is important to note that two different things are compared in this figure: on the one hand emission reduction potentials and on the other hand realisations of (part of) the potential within the context of a certain scenario. Having said that, a number of lessons can be learned from the comparison of both.

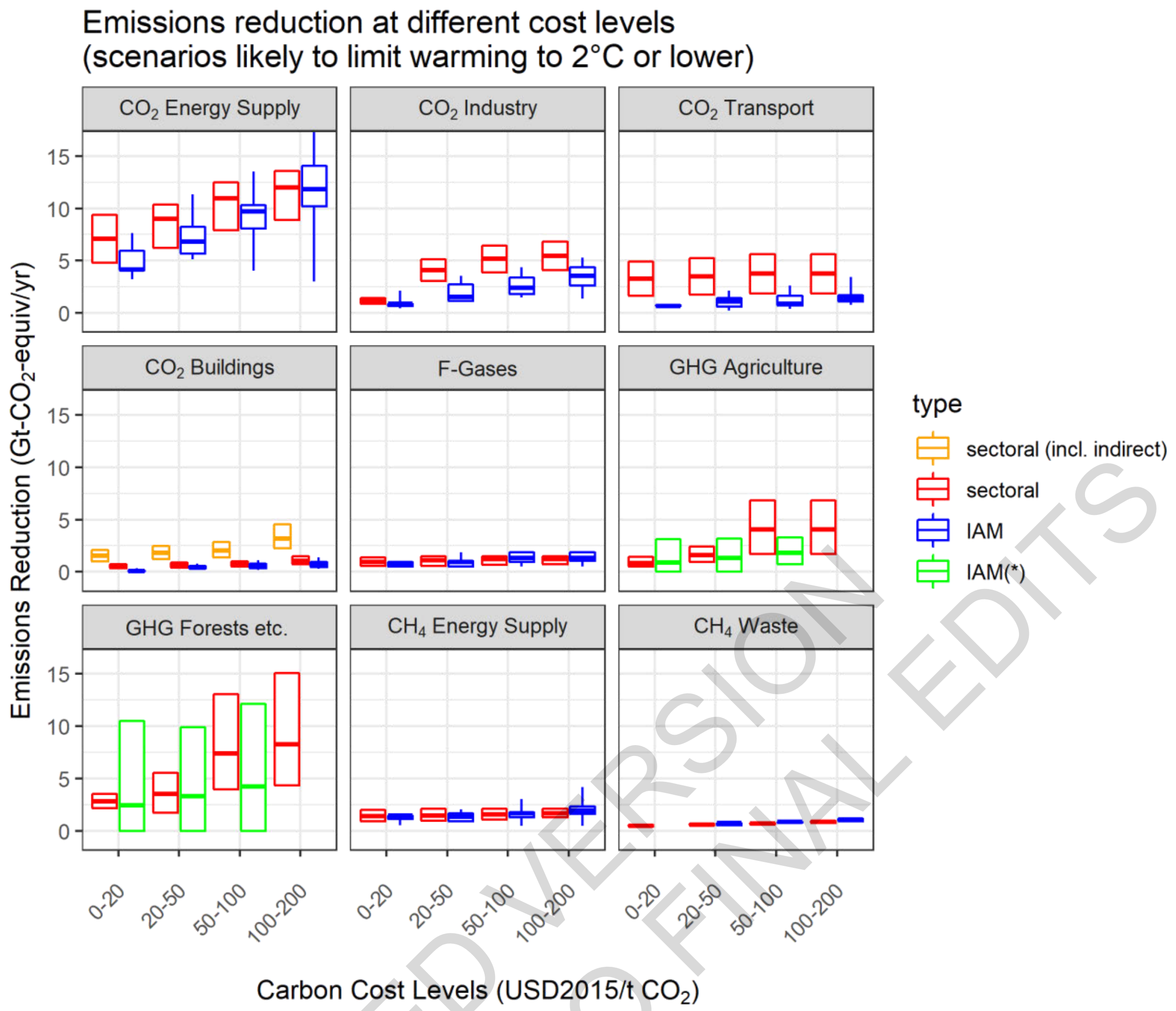


Figure 12.1 Comparison of sectoral estimates for the emission reduction potential with the emission reductions calculated using IAMs.

The latter are given as box plots of global emissions reduction for each sector (blue and green) at different global carbon cost levels (horizontal axis) for 2030, based on all scenarios likely limiting warming to 2°C or lower (see Chapter 3) in the AR6 scenarios database (IPCC 2021). **For IAMs, the cost levels correspond to the levels of the carbon price. Hinges in the blue box plots represent the interquartile ranges and whiskers extend to 5th and 95th percentiles while the hinges in the green box plots describe the full range, and the middle point indicates the mean, not the median. In red, the estimates from the sectoral analysis are given. In all cases, only direct emission reductions are presented, except for the orange boxes (for buildings), which include indirect emission reductions. The orange boxes are only given for reasons of completeness, also for buildings the blue boxes should be compared with the red boxes. Orange and red boxes represent the full ranges of estimates. For IAMs, global carbon prices are applied, which are subject to significant uncertainty.**

For the energy supply sector, the emission reductions projected by the IAMs are for the higher cost levels comparable with the potentials found in the sectoral analysis. But at lower cost levels, the emission reductions as projected by IAMs are smaller than for the sectoral analysis. This is likely due to the fact that high costs for solar energy and wind energy are assumed in IAM models (Krey et al. 2019; Shiraki and Sugiyama 2020). This is not surprising, as the scenario database comprises studies dating back to 2015. A more detailed comparison for the power sector is given in Figure 12.2. Both the sectoral analysis and the IAMs find that both solar and wind energy in particular show strong growth potential, although there is a continuing role for other low-carbon technologies, like nuclear energy and hydropower.

For the AFOLU sector, the sectoral studies provide net emission reduction potentials comparable with projections from the IAMs at costs levels up to 50 USD tCO_2-eq^{-1}. However, beyond that level the mitigation potential found in the sectoral analysis is larger than in the IAMs. For agriculture, it can be explained by the fact that carbon sequestration options, like soil carbon, biochar and agroforestry have little to no representation in IAMs. Similarly, for forestry and other land-use related options, the protection and restoration of other ecosystems than forests (peatland, coastal wetlands and savannas) are not represented in IAMs. Also note that some IAM baselines already have small carbon prices which induce land-based mitigation, while in others, mitigation, particularly from reduced deforestation is part of the storyline even without an implemented carbon price. Both of these effects dampen the mitigation potential available in the USD 100 tCO_2-eq^{-1} carbon price scenario from IAMs. Furthermore, estimates of mitigation through forestry and other land use related options from the AR6 IAM scenario database represent the net emissions from A/R and deforestation, thus are likely to be lower than the sectoral estimates of A/R potential expressed as gross removals.

For the buildings and transport sectors, the sectoral mitigation potentials are higher than those projected by the IAMs. The difference in the transport sector is particularly significant. One possible explanation is that options with negative costs are already included in the reference. In addition, some options, like avoiding demand for energy services in the building sector and model shift in transportation are less well represented in IAMs.

For the industry sector, the sectoral emission reduction potentials are somewhat higher than those reported on average by IAMs. The difference can well be explained by the fact that most IAMs do not include circularity options like material efficiency and recycling; these options together account for 1.5 $GtCO_2$-eq at costs levels from 20 USD tCO_2-eq^{-1} onwards.

For mitigation of emissions of methane and fluorinated gases, the comparability between the sectoral results and IAMs is good.

Overall, it is concluded that there are differences between the sectoral analysis and the IAM outcomes, but most of the differences can be explained by the exclusion of specific options in most IAMs. This comparability confirms the reliability of the sectoral analysis of emission reduction potential. It also demonstrates the added value of sectoral analyses of mitigation potentials: they can more rapidly adapt to changes in price levels of technologies and adopt new options for emission mitigation.

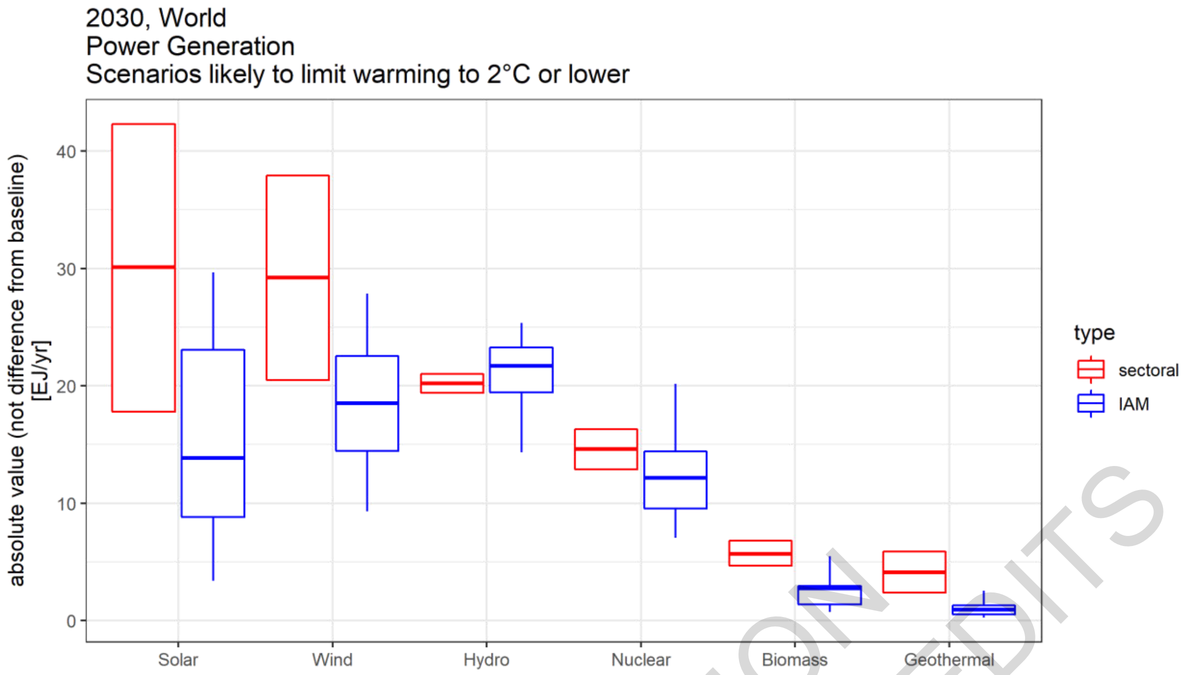


Figure 12.2 Electricity production in 2030 as calculated by Integrated Assessment Models (blue), compared with electricity production potentials found in the sectoral analysis (red).

In both cases cost cut-offs at 100 USD tCO_2^{-1} are applied. Hinges in the blue box plots represent the interquartile ranges and whiskers extend to 5 and 95 percentiles while the hinges in the red box plots describe the full range.

In this section, the information on individual options reported in Section 12.2.2 to sectoral and economy-wide totals has been aggregated. It is concluded that, based on the sectoral analysis, the global mitigation potential is in the range of 32 to 44 $GtCO_2$-eq. This mitigation potential is substantially higher than that reported in AR4, but it is comparable to the more recent estimate by UNEP (2017). Differences exist with the results of IAMs, but most of these can be well explained. The conclusion that the global potential is in this range can be drawn with *high agreement and robust evidence*.

Given the median projection of the reference emissions of 60 $GtCO_2$-eq in 2030, the range of mitigation potentials presented here is sufficient to bring down global emissions in the year 2030 to a level of 16 to 28 $GtCO_2$-eq. Taking into account that there is a range in reference projections for 2030 of 54 to 68 $GtCO_2$-eq, the resulting emissions level shows a wider range: 12 to 31 $GtCO_2$-eq. This is about at or below half of the most recent (2019) emission value of 59±6.6 $GtCO_2$-eq (*high confidence*).

12.2.4 Sectoral findings on emission pathways until 2050

As noted previously, a more qualitative approach is followed and less quantitative information is presented for 2050. The sectoral results are summarised in Table 12.5. In addition to the many technologies that already play a role by 2030 (Table 12.3) additional technologies may be needed for deep decarbonisation, for example for managing power systems with high shares of intermittent renewable sources and for providing new fuels and associated infrastructure for sectors that are hard to decarbonise. New processes also play an important role, notably for industrial processes. In general, stronger sector coupling is needed, particularly increased integration of energy end-use and supply sectors.

1 **Table 12.5: Mitigation options and their characteristics for 2050**

Sector	Major options	Degree to which net zero-GHG is possible
Energy sector	Range of supply side options possible (see 2030 overview). Increased share of electricity in final energy use Potentially important role for hydrogen, ammonia, etc.	Zero CO_2 energy system is possible
Agriculture, forestry and other land use	Options comparable to those in 2030. Permanence is important.	Some hard-to-abate activities will still have positive emissions, but for the sector as a whole, net negative emissions are possible through carbon sequestration in agriculture and forestry
Buildings	Sufficiency, high performance new and existing buildings with efficient HVAC esp. heat pumps, building management and operation, efficient appliances, onsite renewables backed up with demand flexibility and digitalisation measures	At least 8.2 $GtCO_2$ or 61% reduction, as compared to the baseline is possible with options on demand-side. This is a low estimate, because in some developing regions literature is not sufficient to derive a comprehensive estimate. Nearly net zero CO_2 emissions are possible if grid electricity will also be decarbonised. Carbon storage in buildings provides CDR.
Transport	Electrification can become a major option for many transport modes. For long-haul trucking, ships and aviation, in addition biofuels, hydrogen and potentially synthetic fuels can be applied.	To a large extent if the electricity sector is fully decarbonized and the deployment of alternative fuels for long-haul trucking, aviation and shipping is successful.
Industry	Stronger role for material efficiency and recycling. Full decarbonisation through new processes, CCS, CCU and hydrogen can become dominant	Approx. 85% reduction is possible. Net zero CO_2 emissions are possible with retrofitting and early retirement.

Cross-sectoral	Direct air carbon capture and storage Enhanced weathering Ocean-based methods	Contributes CDR to support net zero GHG by counterbalancing sectoral emissions

12.3 Carbon dioxide removal (CDR)

CDR refers to a cluster of technologies, practices, and approaches that remove and sequester carbon dioxide from the atmosphere and durably store the carbon in geological, terrestrial, ocean reservoirs, or in products. Despite the common feature of removing carbon dioxide, CDR methods can be very different (Smith et al. 2017). There are proposed methods for removal of non-CO_2 greenhouse gases such as methane (Jackson et al. 2019, 2021) but scarcity of literature on these methods prevents assessment here.

A number of CDR methods (e.g., Afforestation/Reforestation (A/R), Bioenergy with carbon capture and storage (BECCS), soil carbon sequestration (SCS), biochar, wetland/peatland restoration and coastal restoration) are dealt with elsewhere in this report (Chapters 6 and 7). These methods are synthesised in Section 12.3.2. Others, not dealt with elsewhere, i.e., Direct Air Carbon Capture and Storage (DACCS), enhanced weathering of minerals (EW) and ocean-based approaches including ocean fertilisation (OF) and alkalinity (OA) enhancement, are discussed in Sections 12.3.1.1 to 12.3.1.3 below (see also IPCC SROCC and WGI, Section 5.6). Some methods such as BECCS and DACCS involve carbon storage in geological formations, which is discussed in Chapter 6. The climate system and the carbon cycle responses to CDR deployment and each method's physical and biogeochemical characteristics such as storage form and duration are assessed in Chapters 4 and 5 of the WGI report.

START CROSS-CHAPTER BOX 8 HERE

Cross-Chapter Box 8 Carbon Dioxide Removal: Key characteristics and multiple roles in mitigation strategies

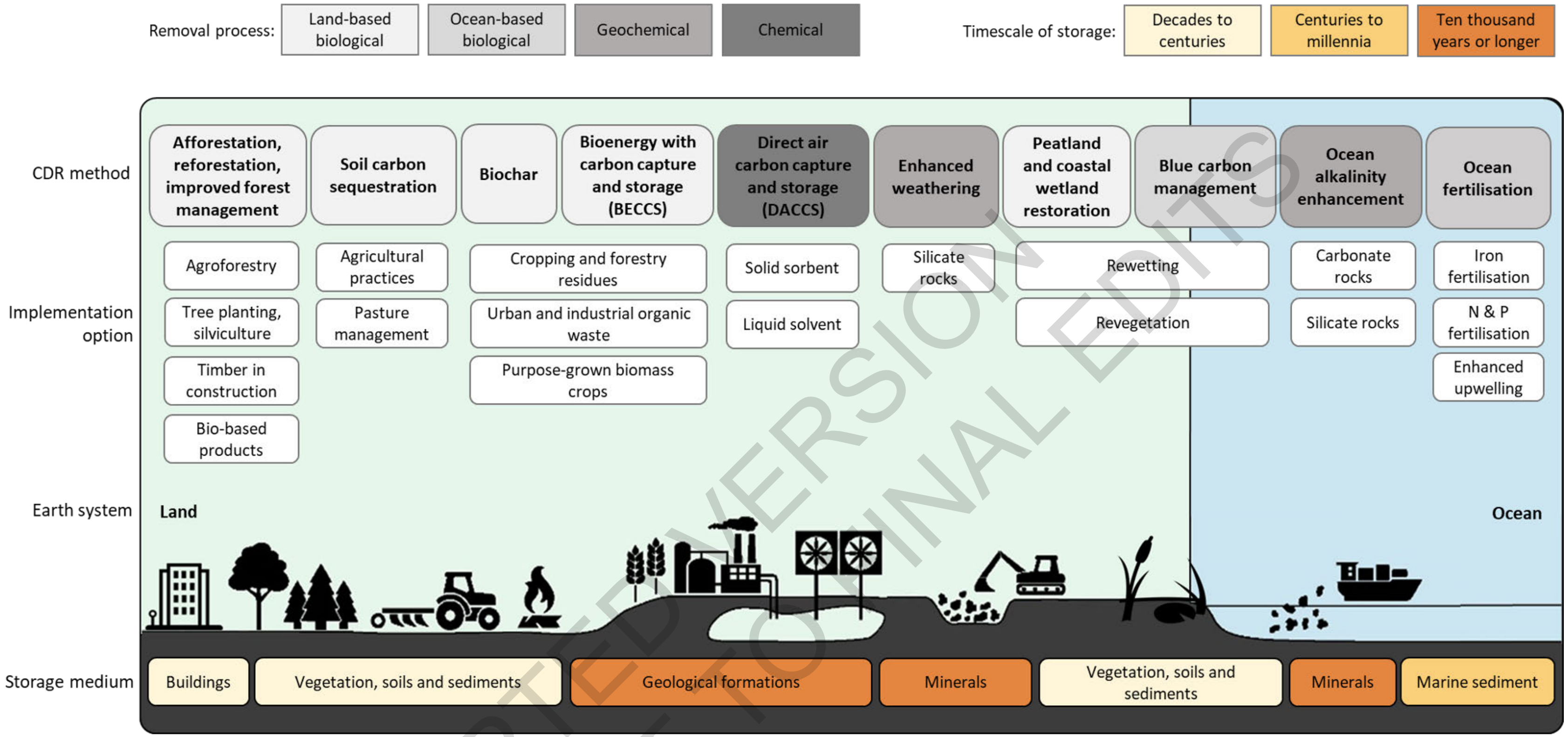
Oliver Geden (Germany), Alaa Al Khourdajie (United Kingdom/Syria), Chris Bataille (Canada), Göran Berndes (Sweden), Holly Jean Buck (the United States of America), Katherine Calvin (the United States of America), Annette Cowie (Australia), Kiane de Kleijne (the Netherlands), Jan Minx (Germany), Gert-Jan Nabuurs (the Netherlands), Glen Peters (Australia/Norway), Andy Reisinger (New Zealand), Peter Smith (United Kingdom), Masahiro Sugiyama (Japan)

Carbon Dioxide Removal (CDR) is a necessary element of mitigation portfolios to achieve net zero CO_2 and GHG emissions both globally and nationally, counterbalancing residual emissions from 'hard-to-transition' sectors such as industry, transport and agriculture. CDR is a key element in scenarios likely to limit warming to 2°C or lower, regardless of whether global emissions reach near-zero, net zero or net-negative levels (Sections 3.3, 3.4, 3.5 in Chapter 3 and Section 12.3 in this chapter). While national mitigation portfolios aiming at net zero or net-negative emissions will need to include some level of CDR, the choice of methods and the scale and timing of their deployment will depend on the ambition for gross emission reductions, how sustainability and feasibility constraints are managed, and how political preferences and social acceptability evolve (Section 12.3.3). This box gives an overview of CDR methods, presents a categorisation based on the key characteristics of removal processes and storage timescales, and clarifies the multiple roles of CDR in mitigation strategies. The term *negative emissions* is used in this report only when referring to the net emissions outcome at a systems level (e.g., *net negative emissions* at global, national, sectoral or supply chain levels).

Categorisation of the main CDR methods

CDR refers to anthropogenic activities that remove CO_2 from the atmosphere and store it durably in geological, terrestrial, or ocean reservoirs, or in products. It includes anthropogenic enhancement of biological, geochemical or chemical CO_2 sinks, but excludes natural CO_2 uptake not directly caused by human activities. Increases in land carbon sink strength due to CO_2 fertilisation or other indirect effects of human activities are not considered CDR (see Glossary). Carbon Capture and Storage (CCS) and Carbon Capture and Utilisation (CCU) applied to CO_2 from fossil fuel use are not CDR methods as they do not remove CO_2 from the atmosphere. CCS and CCU can, however, be part of CDR methods if the CO_2 has been captured from the atmosphere, either indirectly in the form of biomass or directly from ambient air, and stored durably in geological reservoirs or products (Sections 11.3.6 in Chapter 11 and Section 12.3 in this chapter).

There are many different CDR methods and associated implementation options (Cross-Chapter Box 8, Figure 1). Some of these methods (including afforestation and improved forest management, wetland restoration and SCS) have been practiced for decades to millennia, although not necessarily with the intention of removing carbon from the atmosphere. Conversely, methods such as Direct Air Carbon Capture and Storage (DACCS), Bioenergy with Carbon Capture and Storage (BECCS) and Enhanced Weathering are novel, and while experience is growing, their demonstration and deployment are limited in scale. CDR methods have been categorised in different ways in the literature, highlighting different characteristics. In this report, as in AR6 WGI, the categorisation is based on the role of CDR methods in the carbon cycle, i.e., on the removal process (*land-based biological*; *ocean-based biological*; *geochemical*; *chemical*) and on the timescale of storage (*decades to centuries*; *centuries to millennia*; *ten thousand years or longer*). The timescale of storage is closely linked to the storage medium: carbon stored in ocean reservoirs (through enhanced weathering, ocean alkalinity enhancement or ocean fertilisation) and in geological formations (through BECCS or DACCS) generally has longer storage times and is less vulnerable to reversal through human actions or disturbances such as drought and wildfire than carbon stored in terrestrial reservoirs (vegetation, soil). Furthermore, carbon stored in vegetation or through SCS has shorter storage times and is more vulnerable than carbon stored in buildings as wood products; as biochar in soils, cement and other materials; or in chemical products made from biomass or potentially through direct air capture (WGI, Section 5.6, Figure 5.36; WGIII, Section 11.3.6; Fuss et al. 2018; Minx et al. 2018; NAS 2019). Within the same category (e.g., land-based biological CDR) options often differ with respect to other dynamic or context-specific dimensions such as mitigation potential, cost, potential for co-benefits and adverse side-effects, and technology readiness level (Section 12.3, Table 12.6).



Cross-Chapter Box 8, Figure 1: Carbon Dioxide Removal taxonomy.

Methods are categorised based on removal process (grey shades) and storage medium (for which timescales of storage are given, yellow/brown shades). Main implementation options are included for each CDR method. Note that specific land-based implementation options can be associated with several CDR methods, e.g., agroforestry can support soil carbon sequestration and provide biomass for biochar or BECCS.

Source: This figure is an extended version of Figure 2 in (Minx et al. 2018).

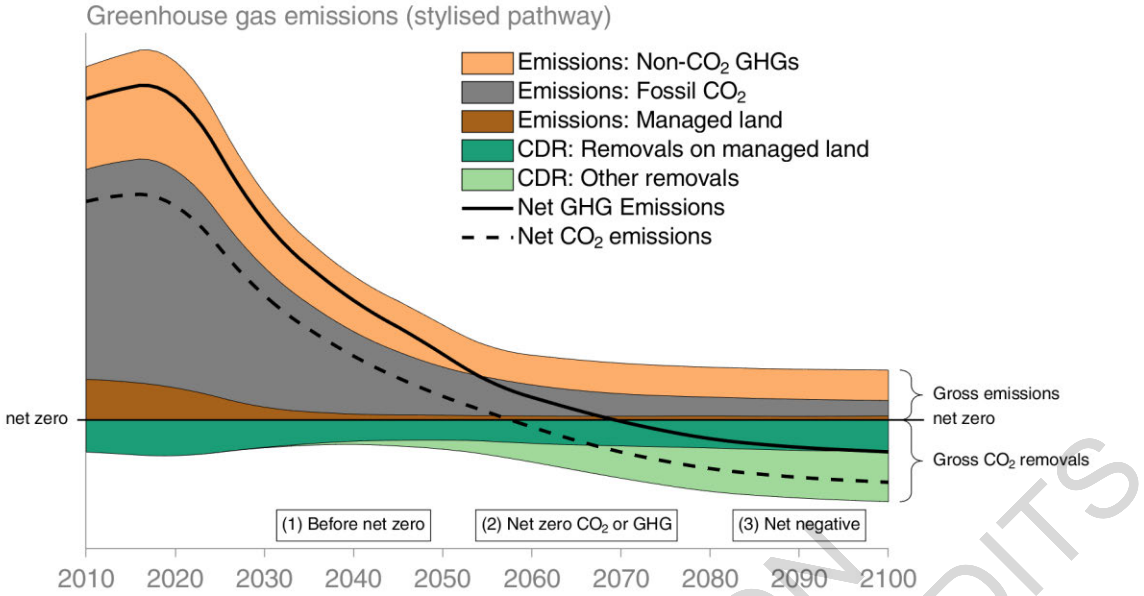
Roles of CDR in mitigation strategies

Within ambitious mitigation strategies at global or national levels, CDR cannot serve as a substitute for deep emissions reductions but can fulfil multiple complementary roles: (1) further reduce net CO_2 or GHG emission levels in the near-term; (2) counterbalance residual emissions from 'hard-to-transition' sectors , such as CO_2 from industrial activities and long-distance transport (e.g., aviation, shipping), or methane and nitrous oxide from agriculture, in order to help reach net zero CO_2 or GHG emissions in the mid-term; (3) achieve and sustain net-negative CO_2 or GHG emissions in the long-term, by deploying CDR at levels exceeding annual residual gross CO_2 or GHG emissions (Sections 2.7.3 in Chapter 2, 3.3 and 3.5 in Chapter 3).

In general, these roles of CDR are not mutually exclusive and can exist in parallel. For example, achieving net zero CO_2 or GHG emissions globally might involve some countries already reaching net-negative levels at the time of global net zero, allowing other countries more time to achieve this. Equally, achieving net-negative CO_2 emissions globally, which could address a potential temperature overshoot by lowering atmospheric CO_2 concentrations, does not necessarily involve all countries reaching net-negative levels (Cross-Chapter Box 3 in Chapter 3; Rajamani et al. 2021; Rogelj et al. 2021).

Cross-Chapter Box 8, Figure 2 shows these multiple roles of CDR in a stylised ambitious mitigation pathway that can be applied to global and national levels. While such mitigation pathways will differ in their shape and exact composition, they include the same basic components: CO_2 emissions from fossil sources, CO_2 emissions from managed land, non-CO_2 emissions, and various forms of CDR. Figure 2 also illustrates the importance of distinguishing between gross CO_2 removals from the atmosphere through deployment of CDR methods and the net emissions outcome (i.e., gross emissions minus gross removals).

CDR methods currently deployed on managed land, such as afforestation or reforestation and improved forest management, lead to CO_2 removals already today, even when net emissions from land use are still positive, e.g., when gross emissions from deforestation and draining peatlands exceed gross removals from afforestation or reforestation and ecosystem conservation (Sections 2.2 in Chapter 2, 7.2 in Chapter 7, Cross-Chapter Box 6 in Chapter 7). As there are currently no removal methods for non-CO_2 gases that have progressed beyond conceptual discussions (Jackson et al. 2021), achieving net zero GHG implies gross CO_2 removals to counterbalance residual emissions of both CO_2 and non-CO_2 gases, applying GWP100 as the metric for reporting CO_2-equivalent emissions, as required for emissions reporting under the Rulebook of the Paris Agreement (Cross-Chapter Box 2 in Chapter 2).



Cross-Chapter Box 8, Figure 2: Roles of CDR in global or national mitigation strategies. Stylised pathway showing multiple functions of CDR in different phases of ambitious mitigation: (1) further reducing net CO_2 or GHG emission levels in near-term; (2) counterbalancing residual emissions to help reach net zero CO_2 or GHG emissions in the mid-term; (3) achieve and sustain net-negative CO_2 or GHG emissions in the long-term.

Net zero CO_2 emissions will be achieved earlier than net zero GHG emissions. As volumes of residual non-CO_2 emissions are expected to be significant, this time-lag could reach one to several decades, depending on the respective size and composition of residual GHG emissions at the time of net zero. Furthermore, counterbalancing residual non-CO_2 emissions by CO_2 removals will lead to net-negative CO_2 emissions at the time of net zero GHG emissions (Cross-Chapter Box 3 in Chapter 3).

END CROSS-CHAPTER BOX 8 HERE

While many governments have included A/R and other forestry measures into their NDCs under the Paris Agreement (Moe and Røttereng 2018; Fyson and Jeffery 2019; Mace et al. 2021), and a few countries also mention BECCS, DACCS and enhanced weathering in their mid-century low emission development strategies (Buylova et al. 2021), very few are pursuing the integration of a broad range of CDR methods into national mitigation portfolios so far (Box 12.1 in Section 12.3.3) (Schenuit et al. 2021). There are concerns that the prospect of large-scale CDR could, depending on the design of mitigation strategies, obstruct near-term emission reduction efforts (Lenzi et al. 2018; Markusson et al. 2018), mask insufficient policy interventions (Geden 2016; Carton 2019), might lead to an overreliance on technologies that are still in their infancy (Anderson and Peters 2016; Larkin et al. 2018; Grant et al. 2021), could overburden future generations (Lenzi 2018; Shue 2018; Bednar et al. 2019) might evoke new conflicts over equitable burden-sharing (Pozo et al. 2020; Lee et al. 2021; Mohan et al. 2021), could impact food security, biodiversity or land rights (Buck 2016; Boysen et al. 2017; Dooley and Kartha 2018; Hurlbert et al. 2019; Dooley et al. 2021), or might be perceived negatively by stakeholders and broader public audiences (Royal Society and Royal Academy of Engineering 2018; Colvin et al. 2020). Conversely, without considering different timescales of carbon storage (Fuss et al. 2018; Hepburn et al. 2019) and implementation of reliable measurement, reporting and verification of carbon flows (Mace et al. 2021), CDR deployment might not deliver the intended benefit of removing CO_2 durably from the atmosphere. Furthermore, without appropriate incentive schemes and market designs

(Honegger et al. 2021b), CDR implementation options could see under-investment. The many challenges in research, development and demonstration of novel approaches, to advance innovation according to broader societal objectives and to bring down costs, could delay their scaling up and deployment (Nemet et al. 2018). Depending on the scale and deployment scenario, CDR methods could bring about various co-benefits and adverse side effects (see below). All this highlights the need for appropriate CDR governance and policies (Section 12.3.3).

The volumes of future global CDR deployment assumed in IAM-based mitigation scenarios are large compared to current volumes of deployment, which presents a challenge since rapid and sustained upscaling from a small base is particularly difficult (de Coninck et al. 2018; Nemet et al. 2018; Hanna et al. 2021). All Illustrative Mitigation Pathways (IMPs) likely to limit warming to 2°C or lower use some form of CDR. Across the full range of similarly ambitious IAM scenarios (scenario categories C1-C3; see Section 3.3.), the annual net CO_2 removal (i.e., gross removals, including A/R, minus gross emissions) on managed land reaches 0.86 [0.01–4.11] $GtCO_2$ yr^{-1} by 2030, 2.98 [0.23–6.38] $GtCO_2$ yr^{-1} by 2050, and 4.19 [0.1–6.91] $GtCO_2$ yr^{-1} by 2100 (values are the medians and bracketed values denote the 5-95 percentile range). The annual BECCS deployment is 0.08 [0–1.09] $GtCO_2$ yr^{-1}, 2.75 [0.52–9.45] $GtCO_2$ yr^{-1}, and 8.96 [2.63–16.15] $GtCO_2$ yr^{-1} for these years, respectively. The annual DACCS deployment reaches 0 [0–0.02] $GtCO_2$ yr^{-1} by 2030, 0.02 [0–1.74] $GtCO_2$ yr^{-1} by 2050, and 1.02 [0–12.6] $GtCO_2$ yr^{-1} by 2100 (Figure 12.3)[1]. Cumulative volumes of BECCS, net CO_2 removal on managed land, and DACCS reach 328 [168–763] $GtCO_2$, 252 [20–418] $GtCO_2$, and 29 [0–339] $GtCO_2$ for the 2020-2100 period, respectively. Reaching the higher end of CDR volumes is subject to issues regarding their feasibility (see below), especially if achieved with only a limited number of CDR methods. Recent studies have identified some drivers for large-scale CDR deployment in IAM scenarios, including insufficient representation of variable renewables, a high discount rate that tends to increase initial carbon budget overshoot and therefore inflates usage of CDR to achieve net-negative emissions at later times, omission of CDR methods aside from BECCS and A/R (Köberle 2019; Emmerling et al. 2019; Hilaire et al. 2019), and limited deployment of demand-side options (Grubler et al. 2018; Daioglou et al. 2019; van Vuuren et al. 2018). The levels of CDR in IAMs in modelled pathways would change depending on the allowable overshoot of policy targets such as temperature or radiative forcing and the costs of non-CDR mitigation options (Johansson et al. 2020; van der Wijst et al. 2021).(see also Section 3.2.2)

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[1] FOOTNOTE[1] We use representative options for labels of each variable reported in the AR6 scenarios database.

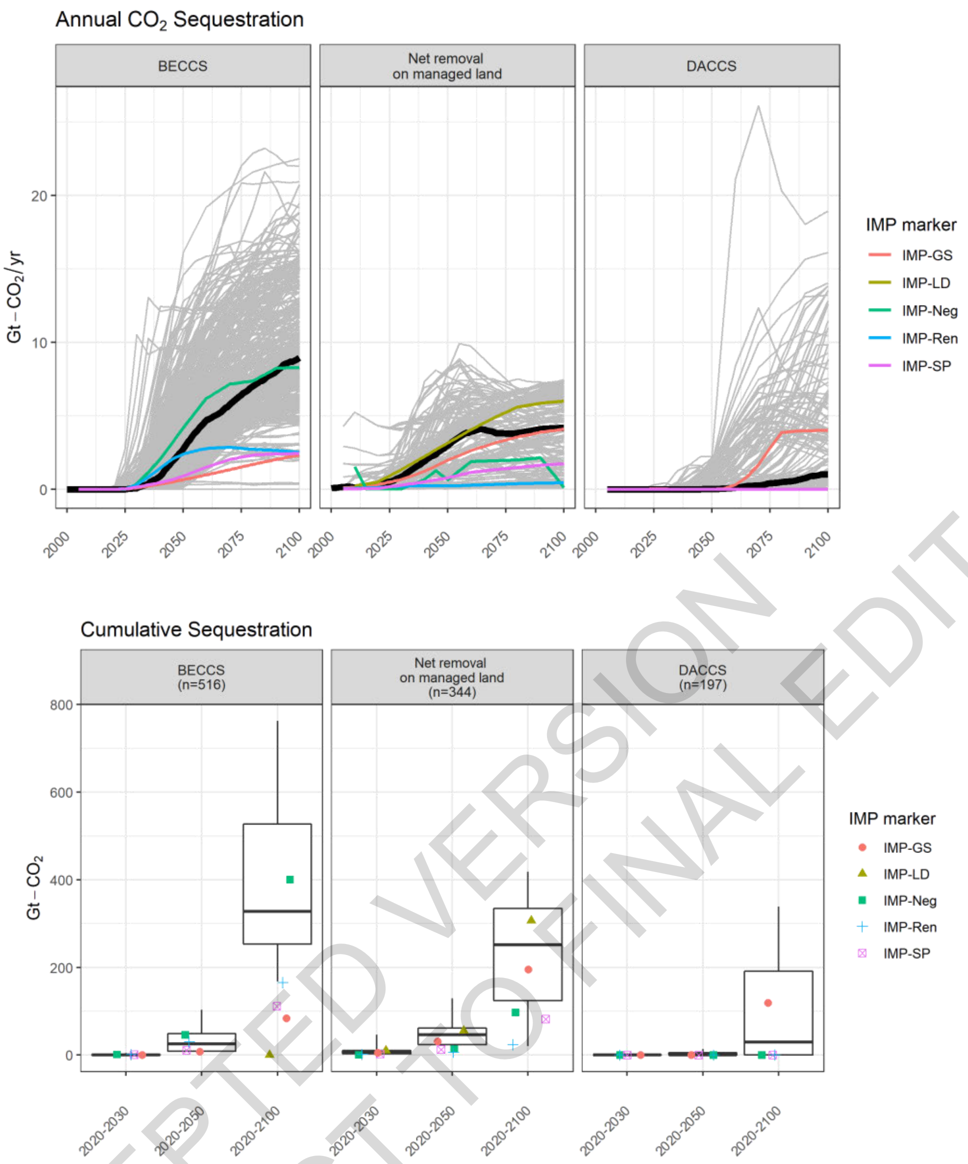


Figure 12.3 Sequestration of three predominant CDR methods: BECCS, net CO₂ removal on managed land (that is, gross removal through A/R minus emissions from deforestation) , and DACCS (upper panels) annual sequestration and (lower panels) cumulative sequestration.

The IAM scenarios correspond to those likely limiting warming to 2°C or lower. The black line in each of the upper panels indicates the median of all the scenarios in categories C1-C3. Hinges in the lower panels represent the interquartile ranges while whiskers extend to 5th and 95th percentiles. The IMPs are highlighted with colours, as shown in the key. The number of scenarios is indicated in the header of each panel. The number of scenarios with a non-zero DACCS value is 146.

While many CDR methods are gradually being explored, IAM scenarios have focused mostly on BECCS and A/R (Tavoni and Socolow 2013; Fuhrman et al. 2019; Rickels et al. 2019; Calvin et al. 2021; Diniz Oliveira et al. 2021) Although some IAM studies have also included other methods such as DACCS (Chen and Tavoni 2013; Marcucci et al. 2017; Realmonte et al. 2019; Akimoto et al. 2021; Fuhrman et al. 2020, 2021a), enhanced weathering (Strefler et al. 2021), SCS and biochar (Holz et al. 2018) there is much less literature compared to studies on BECCS (Hilaire et al. 2019). A large scale, coordinated IAM study on BECCS ("EMF-33") has been conducted (Muratori et al. 2020; Rose et al. 2020a) but none exists for other CDR methods. A recent review proposes a combination of various CDR methods (Fuss et al. 2018) but more in-depth literature on such a portfolio approach is limited

(Strefler et al. 2021). A multi-criteria analysis has identified pathways with CDR portfolios different from least-cost pathways often dominated by BECCS and A/R (Rueda et al. 2021).

At the national and regional level, the role of land-based biological CDR methods has long been analysed, but there is little detailed technoeconomic assessment of the role of other CDR. There is a small but emerging literature providing such assessments for developed countries (Baik et al. 2018; Sanchez et al. 2018; Patrizio et al. 2018; Larsen et al. 2019; Daggash et al. 2018; Kato and Kurosawa 2019; Kraxner et al. 2014; Breyer et al. 2019; McQueen et al. 2020; Bistline and Blanford 2021; Jackson et al. 2021; Kato and Kurosawa 2021; García-Freites et al. 2021; Negri et al. 2021) while the literature outside of developed countries is limited (Alatiq et al. 2021; Fuhrman et al. 2021b; Weng et al. 2021).

In IAMs, CDR is contributed mainly by the energy sector (through BECCS) and AFOLU sector (through A/R) (See Figure 12.3). IAMs are starting to include other CDR methods, such as DACCS and enhanced weathering (Section 12.3.1), which are yet to be attributed to specific sectors in IAMs. Following IPCC guidance for UNFCCC inventories, A/R and SCS are reported in LULUCF, while BECCS would be reported in the sector where the carbon capture occurs, that is, the energy sector in the case of electricity and heat production, and the industry sector for BECCS linked to manufacturing (e.g., steel or hydrogen) (Tanzer et al. 2020; Bui et al. 2021; Tanzer et al. 2021).

12.3.1 CDR methods not assessed elsewhere in this report: DACCS, enhanced weathering and ocean-based approaches

This section assesses the CDR methods that are not carried out solely within conventional sectors and so are not covered in other parts of the report: direct air carbon capture and storage, enhanced weathering, and ocean-based approaches. It provides an overview of each CDR method, costs, potentials, risks and impacts, co-benefits, and their role in mitigation pathways. Since these processes, approaches and technologies have medium to low technology readiness levels, they are subject to significant uncertainty.

12.3.1.1 Direct Air Carbon Capture and Storage (DACCS)

Direct air capture (DAC) is a chemical process to capture ambient CO_2 from the atmosphere. Captured CO_2 can be stored underground (direct air capture carbon and storage, DACCS) or utilised in products (direct air capture carbon and utilisation, DACCU). DACCS shares with conventional CCS the transport and storage components but is distinct in its capture part. Because CO_2 is a well-mixed GHG, DACCS can be sited relatively flexibly, though its locational flexibility is constrained by the availability of low-carbon energy and storage sites. Capturing the CO_2 involves three basic steps: a) contacting the air, b) capturing on a liquid or solid sorbent or a liquid solvent, c) regeneration of the solvent or the sorbent (with heat, moisture and/or pressure). After capture, the CO_2 stream can be stored underground or utilised. The duration of storage is an important consideration; geological reservoirs or mineralisation result in removal for > 1000 years. The duration of the removal through DACCU (Breyer et al. 2019) varies with the lifetime of respective products (Wilcox et al. 2017; Gunnarsson et al. 2018; Bui et al. 2018; Creutzig et al. 2019; Royal Society and Royal Academy of Engineering 2018; Fuss et al. 2018), ranging from weeks to months for synthetic fuels to centuries or more for building materials (e.g., concrete cured using mineral carbonation) (Hepburn et al. 2019). The efficiency and environmental impacts of DACCS and DACCU options depend on the carbon intensity of the energy input (electricity and heat) and other life-cycle assessment (LCA) considerations (Jacobson 2019; Global CO2 Initiative 2018). See Chapters 6 and 11 for further details regarding carbon capture and utilisation. Another key consideration is the net carbon CO_2 removal of DACCS over its life cycle (Madhu et al. 2021). Deutz and Bardow (2021) and Terlouw et al. (2021) demonstrated that the life-cycle net emissions of DACCS systems can be negative, even for existing supply chains and some current energy mixes. They found that the GHG-intensity of energy sources is a key factor.

DAC options can be differentiated by the specific chemical processes used to capture ambient CO_2 from the air and recover it from the sorbent (Fasihi et al. 2019). The main categories are a) liquid solvents with high-temperature regeneration, b) solid sorbents with low temperature regeneration and c) regenerating by moisturising of solid sorbents. Other approaches such as electro-swing (Voskian and Hatton 2019) have been proposed but are less developed. Compared to other CDR methods, the primary barrier to upscaling DAC is its high cost and large energy requirement (*high confidence*) (Nemet et al. 2018), which can be reduced through innovation. It has therefore attracted entrepreneurs and private investments (IEA 2020b).

Status: There are some demonstration projects by start-up companies and academic researchers, who are developing various types of DAC, including aqueous potassium solvent with calcium carbonation and solid sorbents with heat regeneration (NASEM 2019). These projects are supported mostly by private investments and grants or sometimes serve utilisation niche markets (e.g., CO_2 for beverages, greenhouses, enhanced oil recovery). As of 2021, there are more than ten plants worldwide, with a scale of $ktCO_2$ yr^{-1} or smaller (IEA 2020b; NASEM 2019; Larsen et al. 2019). Because of the fundamental difference in the CO_2 concentration in the capture stage, DACCS does not benefit directly from RD&D of conventional CCS. Public RD&D programs dedicated to DAC have therefore been proposed (NASEM 2019; Larsen et al. 2019). Possible research topics include development of new liquid solvents, novel solid sorbents, and novel equipment or system designs, and the need for third-party evaluation of techno-economic aspects has also been emphasized (NASEM 2019). However, since basic research does not appear to be a primary barrier, both NASEM (2019) and Larsen et al. (2019) argue for a stronger focus on demonstration in the US context. Though the US and UK governments have begun funding DACCS research (IEA 2020b), the scale of R&D activities is limited.

Costs: As the process captures dilute CO_2 (~0.04%) from the ambient air, it is less efficient and more costly than conventional carbon capture applied to power plants and industrial installations (*high confidence*) (with a CO_2 concentration of ~10%). The cost of a liquid solvent system is dominated by the energy cost (because of the much higher energy demand for CO_2 regeneration, which reduces the efficiency) while capital costs account for a significant share of the cost of solid sorbent systems (Fasihi et al. 2019). The range of the DAC cost estimates found in the literature is wide (60–1000 USD tCO_2^{-1}) (Fuss et al. 2018) partly because different studies assume different use cases, differing phases (first plant vs. *n*th plant) (Lackner et al. 2012), different configurations, and disparate system boundaries. Estimates of industrial origin are often on the lower side (Ishimoto et al. 2017). Fuss et al. (2018) suggest a cost range of 600–1000 USD tCO_2^{-1} for first-of-a-kind plants, and 100–300 USD tCO_2^{-1} as experience accumulates. An expert elicitation study found a similar cost level for 2050 with a median of around 200 USD tCO_2^{-1} (Shayegh et al. 2021) (*medium evidence, medium agreement*). NASEM (2019) systematically evaluated the costs of different designs and found a range of 84–386 USD_{2015} tCO_2^{-1} for the designs currently considered by active technology developers. This cost range excludes the site-specific costs of transportation or storage.

Potentials: There is no specific study on the potential of DACCS but the literature has assumed that the technical potential of DACCS is virtually unlimited provided that high energy requirements could be met (*medium evidence, high agreement*) (Lawrence et al. 2018; Marcucci et al. 2017; Fuss et al. 2018) since DACCS encounters less non-cost constraints than any other CDR method. Focusing only on the Maghreb region, Breyer et al. (2020) reported an optimistic potential 150 $GtCO_2$ at less than 61 USD tCO_2^{-1} for 2050. Fuss et al. (2018) suggest a potential of 0.5–5 $GtCO_2$ yr^{-1} by 2050 because of environmental side effects and limits to underground storage. In addition to the ultimate potentials, Realmonte et al. (2019) noted the rate of scale-up as a strong constraint on deployment. Meckling and Biber (2021) discuss a policy roadmap to address the political economy for upscaling. More systematic analysis on potentials is necessary; first and foremost on national and regional levels, including the requirements for low-carbon heat and power, water and material demand, availability of geological storage and the need for land in case of low-density energy sources such as solar or wind power.

Risks and impacts: DACCS requires a considerable amount of energy (*high confidence*), and depending on the type of technology, water, and make-up sorbents, while its land footprint is small compared to other CDR methods (Smith et al. 2016), but depending on the source of energy for DACCS (e.g., renewables vs. nuclear), it could require a significant land footprint (NASEM 2019; Sekera and Lichtenberger 2020). The theoretical minimum energy requirement for separating CO_2 gas from the air is ~0.5 GJ tCO_2^{-1} (Socolow et al. 2011). Fasihi et al. (2019) reviewed the published estimates of energy requirements and found that for the current technologies, the total energy requirement is ~4–10 GJ tCO_2^{-1} , with heat accounting for about 80% and electricity about 20% (McQueen et al. 2021). At a 10 $GtCO_2$ yr^{-1} sequestration scale, this would translate into 40–100 EJ yr^{-1} of energy consumption (32–80 EJ yr^{-1} for heat and 8–20 EJ yr^{-1} electricity), which can be contrasted with the current primary energy supply of ~600 EJ yr^{-1} and electricity generation of ~100 EJ yr^{-1}. For the solid sorbent technology, low-temperature heat could be sourced from heat pumps powered by low-carbon sources such as renewables (Breyer et al. 2020), waste heat (Beuttler et al. 2019), and nuclear energy (Sandalow et al. 2018). Unless sourced from a clean source, this amount of energy could cause environmental damage (Jacobson 2019). Because DACCS is an open system, water lost from evaporation must be replenished. Water loss varies, depending on technology (including adjustable factors such as the concentration of the liquid solvent) as well as environmental conditions (e.g., temperate vs. tropical climates). For a liquid solvent system, it can be 0–50 tH_2O tCO_2^{-1} (Fasihi et al. 2019). A water loss rate of ~1–10 tH_2O tCO_2^{-1} (Socolow et al. 2011) would translate into ~10–100 GtH_2O = 10–100 km^3 to capture 1ma0 $GtCO_2$ from the atmosphere. Some solid sorbent technologies actually produce water as a by-product, for example 0.8–2 tH_2O tCO_2^{-1} for a solid-sorbent technology with heat regeneration (Beuttler et al. 2019; Fasihi et al. 2019). Large-scale deployment of DACCS would also require a significant quantity of materials, and energy to produce them (Chatterjee and Huang 2020). Hydroxide solutions are currently being produced as a by-product of chlorine but replacement (make-up) requirement of such materials at scale exceeds the current market supply (Realmonte et al. 2019). The land requirements for DAC units are not large enough to be of concern (Madhu et al. 2021). Furthermore, these can be placed on unproductive lands, in contrast to biological CDR. Nevertheless, to ensure that CO_2-depleted air does not enter the air contactor of an adjacent DAC system, there must be enough space between DAC units, similar to wind power turbines. Considering this, Socolow et al. (2011) estimated a land footprint of 1.5 km^2 $MtCO_2^{-1}$. In contrast, large energy requirements can lead to significant footprints if low-density energy sources (e.g., solar PV) are used (Smith et al. 2016). For the issues associated with CO_2 utilisation and storage, see Chapter 6.

Co-benefits: While Wohland et al. (2018) proposed solid sorbent-based DAC plants as a Power-to-X technology that could use excess renewable power (at the time of low or even negative prices), such operation would add additional costs. Installations would need to be designed for intermittent operations (i.e., at low load factors) which would negatively affect capital and operation costs (Sandalow et al. 2018; Daggash et al. 2018) as a high time-resolution model suggests a high utilisation rate (Breyer et al. 2020). Solid sorbent DAC designs can potentially remove more water from the ambient air than needed for regeneration, thereby delivering surplus water that would contribute to Sustainable Development Goal (SDG) 6 (*Clean Water and Sanitation*) in arid regions (Sandalow et al. 2018; Fasihi et al. 2019).

Trade-offs and spill over effects: Liquid solvent DACCS systems need substantial amounts of water (Fasihi et al. 2019), although much less than BECCS systems (Smith et al. 2016), which could negatively affect SDG 6 (*Clean Water and Sanitation*). Although the high energy demand of DACCS could affect SDG 7 (*Affordable and Clean Energy*) negatively through potential competition or positively through learning effects (Beuttler et al. 2019), its impact has not been thoroughly assessed yet.

Role in mitigation pathways: There are a few IAM studies that have explicitly incorporated DACCS. Stringent emissions constraints in these studies lead to high carbon prices, allowing DACCS to play an important role in mitigation. Chen and Tavoni (2013) examined the role of DACCS in an IAM (WITCH) and found that incorporating DACCS in their IAM reduces the overall cost of mitigation and tends to postpone the timing of mitigation. The scale of capture goes up to 37 $GtCO_2$ yr^{-1} in 2100. (Akimoto et al. 2021) introduced DACCS in the integrated assessment model DNE21+, and also found the long-term marginal cost of abatement is significantly reduced by DACCS. Marcucci et al. (2017) ran MERGE-ETL, an integrated model with endogenous learning, and showed that DACCS allows for a model solution for the 1.5°C target, and that DACCS substitutes for BECCS under stringent targets. In their analysis, DACCS captures up to 38.3 $GtCO_2$ yr^{-1} in 2100. Realmonte et al. (2019) modelled two types of DACCS (based on liquid and solid sorbents) with two IAMs (TIAM-Grantham and WITCH), and showed that in deep mitigation scenarios, DACCS complements, rather than substitutes, other CDR methods such as BECCS, and that DACCS is effective at containing mitigation costs. At the national scale, Larsen et al. (2019) utilised the Regional Investment and Operations (RIO) Platform coupled with the Energy PATHWAYS model, and explicitly represented DAC in US energy systems scenarios. They found that in a scenario that reaches net zero emissions by 2045, about 0.6 $GtCO_2$ or 1.8 $GtCO_2$ of DACCS would be deployed, depending on the availability of biological carbon sinks and bioenergy. The modelling supporting the European Commission’s initial proposal for net zero GHG emissions by 2050 incorporated DAC, whose captured CO_2 is used for both synthetic fuel production (DACCU) and storage (DACCS) (Capros et al. 2019). Fuhrman et al. (2021a) evaluated the role of DACCS across 5 shared socioeconomic pathways with the GCAM modelling framework and identified a substantial role of DACCS in mitigation and a decreased pressure on land and water resources from BECCS, even under the assumption of limited energy efficiency improvement and conservative cost declines of DACCS technologies. The newest iteration of the World Economic Outlook by IEA (2021b) deploys CDR on a limited scale, and DACCS removes 0.6 $GtCO_2$ in 2050 for its Net Zero CO_2 Emissions scenario.

Status, costs, potentials, risk and impacts, co-benefits, trade-offs and spillover effects and the role in mitigation pathways of DACCS are summarised in Table 12.6.

12.3.1.2 Enhanced weathering

Enhanced weathering involves a) the mining of rocks containing minerals that naturally absorb CO_2 from the atmosphere over geological timescales (as they become exposed to the atmosphere through geological weathering), b) the comminution of these rocks to increase the surface area, and c) the spreading of these crushed rocks on soils (or in the ocean/coastal environments; Section 12.3.1.3) so that they react with atmospheric CO_2 (Schuiling and Krijgsman 2006; Hartmann et al. 2013; Beerling et al. 2018; Goll et al. 2021). Construction waste, and waste materials from mining can also be used as a source material for enhanced weathering. Silicate rocks such as basalt, containing minerals rich in calcium and magnesium and lacking metal ions such as nickel and chromium, are most suitable for enhanced weathering (Beerling et al. 2018); they reduce soil solution acidity during dissolution, and promote the chemical transformation of CO_2 to bicarbonate ions. The bicarbonate ions can precipitate in soils and drainage waters as a solid carbonate mineral (Manning 2008), or remain dissolved and increase alkalinity levels in the ocean when the water reaches the sea (Renforth and Henderson 2017).

The modelling study by Cipolla et al. (2021) found that rate of weathering is greater in high rainfall environments, and was increased by organic matter amendment.

Status: Enhanced weathering has been demonstrated in the laboratory and in small scale field trials (TRL 3–4) but has yet to be demonstrated at scale (Beerling et al. 2018; Amann et al. 2020). The chemical reactions are well understood (Gillman 1980; Gillman et al. 2001; Manning 2008), but the behaviour of the crushed rocks in the field and potential co-benefits and adverse-side effects of enhanced weathering require further research (Beerling et al. 2018). Small scale laboratory experiments have calculated weathering rates that are orders of magnitude slower than the theoretical limit for mass transfer-controlled forsterite (Renforth et al. 2015; Amann et al. 2020) and basalt dissolution (Kelland et al. 2020). Uncertainty surrounding silicate mineral dissolution rates in soils, the fate of the released products, the extent of legacy reserves of mining by-products that might be exploited, location and availability of rock extraction sites, and the impact on ecosystems remain poorly quantified and require further research to better understand feasibility (Renforth 2012; Moosdorf et al. 2014; Beerling et al. 2018). Closely monitored, large-scale demonstration projects would allow these aspects to be studied (Smith et al. 2019a; Beerling et al. 2020).

Costs: Fuss et al. (2018), in a systematic review of the costs and potentials of CDR methods including enhanced weathering, note that costs are closely related to the source of the rock, the technology used for rock grinding and material transport (Hartmann et al. 2013; Renforth 2012; Strefler et al. 2018). Due to differences in the methods and assumptions between studies, literature ranges are highly uncertain and range from 15–40 USD tCO_2^{-1} to 3460 USD tCO_2^{-1} (Köhler et al. 2010; Taylor et al. 2016). Renforth (2012) reported operational costs in the UK of applying mafic rocks (rocks with high magnesium and iron silicate mineral concentrations) of 70–578 USD tCO_2^{-1}, and for ultramafic rocks (rocks rich in magnesium and iron silicate minerals but with very low silica content - the low silica content enhances weathering rates) of 24–123 USD tCO_2^{-1}. Beerling et al. (2020) combined a spatially resolved weathering model with a technoeconomic assessment to suggest costs of between 54–220 USD tCO_2^{-1} (with a weighted mean of 118–128 USD tCO_2^{-1}). Fuss et al. (2018) suggested an author judgement cost range of 50–200 USD tCO_2^{-1} for a potential of 2–4 $GtCO_2$ yr^{-1} from 2050, excluding biological storage.

Potentials: In a systematic review of the costs and potentials of enhanced weathering, Fuss et al. (2018) report a wide range of potentials (*limited evidence, low agreement*). The highest reported regional sequestration potential, 88.1 $GtCO_2$ yr^{-1}, is reported for the spreading of pulverised rock over a very large land area in the tropics, a region considered promising given the higher temperatures and greater rainfall (Taylor et al. 2016). Considering cropland areas only, the potential carbon removal was estimated by Strefler et al. (2018) to be 95 $GtCO_2$ yr^{-1} for dunite and 4.9 $GtCO_2$ yr^{-1} for basalt. Slightly lower potentials were estimated by Lenton (2014) where the potential of carbon removal by enhanced weathering (including adding carbonate and olivine to both oceans and soils) was estimated to be 3.7 $GtCO_2$ yr^{-1} by 2100, but with mean annual removal an order of magnitude less at 0.2 GtC-eq yr^{-1} (Lenton 2014). The estimates reported in Smith et al. (2016) are based on the potential estimates of Lenton (2014). Beerling et al. (2020) estimate that up to 2 $GtCO_2$ yr^{-1} could be removed by 2050 by spreading basalt onto 35-59% (weighted mean 53%) of agricultural land of 12 countries. Fuss et al. (2018) provide an author judgement range for potential of 2–4 $GtCO_2$ yr^{-1} for 2050.

Risks and impacts: Mining of rocks for enhanced weathering will have local impacts and carries risks similar to that associated with the mining of mineral construction aggregates, with the possible additional risk of greater dust generation from fine comminution and land application. In addition to direct habitat destruction and increased traffic to access mining sites, there could be adverse impacts on local water quality (Younger and Wolkersdorfer 2004).

Co-benefits: Enhanced weathering can improve plant growth by pH modification and increased mineral supply (Kantola et al. 2017; Beerling et al. 2018), can enhance SCS in some soils (Beerling et al. 2018) thereby protecting against soil erosion (Wrigth and Upadhyaya 1998), and increasing the cation exchange capacity, resulting in increased nutrient retention and availability (Baldock and Skjemstad 2000; Yu et al. 2017; Guntzer et al. 2012; Tubana et al. 2016; Manning 2010; Haque et al. 2019; Smith et al. 2019a; Gillman 1980; Gillman et al. 2001). Through these actions, it can contribute to the UN SDGs 2 *Zero Hunger*, 15 *Life of Land* (by reducing land demand for croplands), 13 *Climate Action* (through CDR), 14 *Life Below Water* (by ameliorating ocean acidification) and 6 *Clean Water and Sanitation* (Smith et al. 2019a). To more directly ameliorate ocean acidification while increasing CDR and reducing impacts on land ecosystems, alkaline minerals could instead be directly added to the ocean (Section 12.3.1.3). There are potential benefits in poverty reduction through employment of local workers in mining (Pegg 2006).

Trade-offs and spill over effects: Air quality could be adversely affected by the spreading of rock dust (Edwards et al. 2017), though this can partly be ameliorated by water-spraying (Grundnig et al. 2006). As noted above, any significant expansion of the mining industry would require careful assessment to avoid possible detrimental effects on biodiversity (Amundson et al. 2015). The processing of an additional 10 billion tonnes of rock would require up to 3000 TWh, which could represent approximately 0.1-6 % of global electricity in 2100. The emissions associated with this additional energy generation may reduce the net carbon dioxide removal by up to 30% with present day grid average emissions, but this efficiency loss would decrease with low-carbon power (Beerling et al. 2020).

Role in mitigation pathways: Only one study to date has included enhanced weathering in an integrated assessment model to explore mitigation pathways (Strefler et al. 2021).

Status, costs, potentials, risk and impacts, co-benefits, trade-offs and spill over effects and the role in mitigation pathways of enhanced weathering are summarised in Table 12.6.

12.3.1.3 Ocean-based methods

The ocean, which covers over 70% of the Earth's surface, contains ~38,000 GtC, some 45 times more than the present atmosphere, and oceanic uptake has already consumed close to 30-40% of anthropogenic C emissions (Gruber et al. 2019, Sabine et al. 2004). The ocean is characterised by diverse biogeochemical cycles involving carbon, and ocean circulation has much longer timescales than the atmosphere, meaning that additional anthropogenic carbon could potentially be stored in the ocean for centuries to millennia for methods that increase deep ocean dissolved carbon concentrations or temporarily bury the carbon; or essentially permanently (over ten thousand years) for methods that store the carbon in mineral forms or as ions by increasing alkalinity (Siegel et al., 2021) (Cross-Chapter Box 8 Figure 1). A wide range of methods and implementation options for marine CDR have been proposed (Gattuso et al. 2018; Hoegh-Guldberg et al. 2018; GESAMP 2019). The most studied ocean-based CDR methods are ocean fertilisation, alkalinity enhancement (including electrochemical methods) and intensification of biologically driven carbon fluxes and storage in marine ecosystems, referred to as "blue carbon". The mitigation potentials, costs, co-benefits and trade-offs of these three options are discussed below. Less well studied are methods including artificial upwelling, terrestrial biomass dumping into oceans, direct CO_2 removal from seawater (with CCS), and sinking marine biomass into the deep ocean or harvesting it for bioenergy (with CCS) or biochar (GESAMP 2019). These methods are summarized briefly below. Potential climate response and influence on the carbon budget of ocean-based CDR methods are discussed in Chapter 5 in WGI AR6.

Ocean fertilisation (OF). One natural mechanism of carbon transfer from the atmosphere to the deep ocean is the ocean biological pump, which is driven by the sinking of organic particles from the upper ocean. These particles derive ultimately from primary production by phytoplankton and most of them are remineralised within the upper ocean with only a small fraction reaching the deep ocean where the carbon can be sequestered on centennial and longer timescales**.** Increasing nutrient availability would stimulate uptake of CO_2 through phytoplankton photosynthesis producing organic matter, some of which would be exported into the deep ocean, sequestering carbon. In areas of the ocean where macronutrients (nitrogen, phosphorus) are available in sufficient quantities (about 25% of the total area), the growth of phytoplankton is limited by the lack of trace elements such as iron. Thus, OF CDR can be based on two implementation options to increase the productivity of phytoplankton (Minx et al. 2018): macronutrient enrichment and micronutrient enrichment. A third option highlighted in GESAMP (2019) is based on fertilisation for fish stock enhancement, for instance, as naturally occurs in eastern boundary current systems. Iron fertilisation is the best studied OF option to date, but knowledge so far is still inadequate to predict global ecological and biogeochemical consequences.

Status: OF has a natural analogue: periods of glaciation in the geological past are associated with changes in deposition of dust containing iron into the ocean. Increased formation of phytoplankton has also been observed during seasonal deposition of dust from the Arabian Peninsula and ash deposition on the ocean surface after volcanic eruptions (Jaccard et al., 2013; Achterberg et al. 2013; Olgun et al. 2013; Martínez-García et al. 2014). OF options may appear technologically feasible, and enhancement of photosynthesis and CO_2 uptake from surface waters is confirmed by a number of field experiments conducted in different areas of the ocean, but there is scientific uncertainty about the proportion of newly formed organic carbon that is transferred to deep ocean, and the longevity of storage (Blain et al. 2008; Williamson et al. 2012; Trull et al. 2015). The efficiency of OF also depends on the region and experimental conditions, especially in relation to the availability of other nutrients, light and temperature (Aumont and Bopp 2006). In the case of macronutrients, very large quantities are needed and the proposed scaling of this technique has been viewed as unrealistic (Williamson and Bodle 2016).

Costs: Ocean fertilisation costs depend on nutrient production and its delivery to the application area (Jones 2014). The costs range from 2 USD tCO_2^{-1} for fertilisation with iron (Boyd 2008) to 457 USD tCO_2^{-1} for nitrate (Harrison 2013). Reported costs for macronutrient application at 20 USD tCO_2^{-1} (Jones 2014), contrast with higher estimates by (Harrison 2013) reporting that low costs are due to overestimation of sequestration capacity and underestimation of logistical costs. The median of OF cost estimates, 230 USD tCO_2^{-1} (Gattuso et al., 2021) indicates low cost-effectiveness, albeit uncertainties are large.

Potentials: Theoretical calculations indicate that organic carbon export increases 2–20 kg per gram of iron added, but experiments indicate much lower efficiency: a significant part of the CO_2 can be emitted back the atmosphere because much of the organic carbon produced is remineralised in the upper ocean. Efficiency also varies with location (Bopp et al. 2013). Between studies, there are substantial differences in the ratio of iron added to carbon fixed photosynthetically, and in the ratio of iron added to carbon eventually sequestered (Trull et al. 2015), which has implications both for the success of this strategy, and its cost. Estimates indicate potentially achievable net sequestration rates of 1–3 $GtCO_2$ yr^{-1} for iron fertilisation, translating into cumulative CDR of 100–300 $GtCO_2$ by 2100 (Ryaboshapko and Revokatova 2015; Minx et al. 2018), whereas OF with macronutrients has a higher theoretical potential of 5.5 $GtCO_2$ yr^{-1} (Harrison 2017; Gattuso et al. 2021). Modelling studies show a maximum effect on atmospheric CO_2 of 15–45 ppmv in 2100 (Zeebe and Archer 2005; Aumont and Bopp 2006; Keller et al. 2014; Gattuso et al. 2021).

Risks and impacts: Several of the mesoscale iron enrichment experiments have seen the emergence of potentially toxic species of diatoms (Silver et al. 2010; Trick et al. 2010). There is also (limited) evidence of increased concentrations of other GHGs such as methane and nitrous oxide during the subsurface decomposition of the sinking particles from iron-stimulated blooms (Law 2008). Impacts on marine biology and food web structure are not well known, however OF at large scale could cause changes in nutrient distributions or anoxia in subsurface water (Fuhrman and Capone 1991; DFO 2010). Other potential risks are perturbation to marine ecosystems via reorganisation of community structure, enhanced deep ocean acidification (Oschlies et al. 2010) and effects on human food supply.

Co-benefits: Co-benefits of OF include a potential increase in fish biomass through enhanced biological production (Minx et al. 2018) and reduced ocean acidification in the short term in the upper ocean (by CO_2 removal), though it could be enhanced in the long term in the ocean interior (by CO_2 release) (Oschlies et al., 2010; Gattuso et al. 2018).

Trade-offs and spill-over effects: Potential drawbacks include subsurface ocean acidification and deoxygenation (Oschlies et al., 2010; Cao and Caldeira 2010; Williamson et al. 2012); altered regional meridional nutrient supply and fundamental alteration of food webs (GESAMP 2019); and increased production of N_2O and CH_4 (Jin and Gruber 2003; Lampitt et al. 2008). Ocean fertilisation is considered to have negative consequences for eight SDGs, and a combination of both positive and negative consequences for seven SDGs (Honegger et al. 2020).

Ocean Alkalinity enhancement (OAE). CDR through 'ocean alkalinity enhancement' or 'artificial ocean alkalinisation' (Renforth and Henderson 2017) can be based on: 1) the dissolution of natural alkaline minerals that are added directly to the ocean or coastal environments; 2) the dissolution of such minerals upstream from the ocean (e.g., 'enhanced weathering', Section 12.3.1.2); 3) the addition of synthetic alkaline materials directly to the ocean or upstream; and 4) electrochemical processing of seawater. In the case of 2), minerals are dissolved on land and the dissolution products are conveyed to the ocean through runoff and river flow. These processes result in chemical transformation of CO_2 and sequestration as bicarbonate and carbonate ions (HCO_3^-, CO_3^{2-}) in the ocean. Imbalances between the input and removal fluxes of alkalinity can result in changes in global oceanic alkalinity and therefore the capacity of the ocean to store C. Such alkalinity-induced changes in partitioning of C between atmosphere and ocean are thought to play an important role in controlling climate change on timescales of 1000 years and longer (e.g., Zeebe 2012). The residence time of dissolved inorganic carbon in the deep ocean is around 100,000 years. However, residence time may decrease if alkalinity is reduced by a net increase in carbonate minerals by either increased formation (precipitation) or reduced dissolution of carbonate (Renforth and Henderson 2017). The alkalinity of seawater could potentially also be increased by electrochemical methods, either directly by reactions at the cathode that increase the alkalinity of the surrounding solution that can be discharged into the ocean, or by forcing the precipitation of solid alkaline materials (e.g., hydroxide minerals) that can then be added to the ocean (e.g., Rau et al. 2013; La Plante et al. 2021).

Status: OAE has been demonstrated by a small number of laboratory experiments (in addition to enhanced weathering, Section 12.3.1.2). The use of enhanced ocean alkalinity for C storage was first proposed by Kheshgi (1995) who considered the creation of highly reactive lime that would readily dissolve in the surface ocean and sequester CO_2. An alternative method proposed the dissolution of carbonate minerals (e.g., $CaCO_3$) in the presence of waste flue gas CO_2 and seawater as a means capturing CO_2 and converting it to bicarbonate ions (Rau and Caldeira 1999; Rau 2011). House et al. (2007) proposed the creation of alkalinity in the ocean through electrolysis. The fate of the stored carbon is the same for these proposals (i.e., HCO_3^- and CO_3^{2-} ions), but the reaction pathway is different. Enhanced weathering of silicate minerals such as olivine could add alkalinity to the ocean, for example, by placing olivine sand in coastal areas (Montserrat et al. 2017; Meysman and Montserrat 2017)**.** Some authors suggest use of maritime transport to discharge calcium hydroxide (slaked lime, SL) (Caserini et al. 2021).

Costs: Techno-economic assessments of OAE largely focus on quantifying overall energy and carbon balances. Cost ranges are 40–260 USD tCO_2^{-1} (Fuss et al. 2018). Considering life cycle carbon and energy balances for various OA options, adding lime (or other reactive calcium or magnesium oxide/hydroxides) to the ocean would cost 64–260 USD tCO_2^{-1} (Renforth et al. 2013; Renforth & Kruger 2013; Caserini et al. 2019). Rau (2008) and Rau et al. (2018) estimate that electrochemical processes for increasing ocean alkalinity may have a net cost of 3–160 USD tCO_2^{-1}, largely depending on energy cost and co-product (H_2) market value. In the case of direct addition of alkaline minerals to the ocean (i.e., without calcination), the cost is estimated to be 20–50 USD tCO_2^{-1} (Harvey 2008; Köhler et al. 2013; Renforth and Henderson 2017).

Potentials: For OAE, the ocean theoretically has the capacity to store thousands of $GtCO_2$ (cumulatively) without exceeding pre-industrial levels of carbonate saturation (Renforth and Henderson 2017) if the impacts were distributed evenly across the surface ocean. The potential of increasing ocean alkalinity may be constrained by the capability to extract, process, and react minerals (Section 12.3.1.2); the demand for co-benefits (see below), or to minimise impacts around points of addition. Important challenges with respect to the detailed quantification of the CO_2 sequestration efficiency include nonstoichiometric dissolution, reversed weathering and potential pore water saturation in the case of adding minerals to shallow coastal environments (Meysman and Montserrat 2017). Fuss et al. (2018) suggest storage potentials of 1–100 $GtCO_2$ yr^{-1}. (González and Ilyina 2016) suggested that addition of 114 Pmol of alkalinity to the surface ocean could remove 3400 $GtCO_2$ from the atmosphere.

Risks and impacts: For OAE, the local impact of increasing alkalinity on ocean chemistry can depend on the speed at which the impacted seawater is diluted/circulated and the exchange of CO_2 from the atmosphere (Bach et al. 2019)**.** Also, more extreme carbonate chemistry perturbations due to non-equilibrated alkalinity could affect local marine biota (Bach et al. 2019), although biological impacts are largely unknown . Air-equilibrated seawater has a much lower potential to perturb seawater carbonate chemistry. However, seawater with slow air-sea gas exchange, in which alkalinity increases, consumes CO_2 from the surrounding water without immediate replenishment from the atmosphere, which would increase seawater pH and saturation states and may impact marine biota (Meysman and Montserrat 2017; Montserrat et al. 2017). It may be possible to use this effect to ameliorate ocean acidification. Like enhanced weathering, some proposals may result in the dissolution products of silicate minerals (e.g., Si, Fe, K, Ni) being supplied to ocean ecosystems (Meysman and Montserrat 2017; Montserrat et al. 2017). Ecological and biogeochemical consequences of OA largely depend on the minerals used. When natural minerals such as olivine are used, the release of additional Si and Fe could have fertilising effects (Bach et al. 2019). In addition to perturbations to marine ecosystems via reorganisation of community structure, potentially adverse effects of OA that should be studied include the release of toxic trace metals from some deposited minerals (Hartmann et al. 2013).

Co-benefits: Intentional addition of alkalinity to the oceans through OAE would decrease the risk to ocean ecosystems caused by the CO_2-induced impact of ocean acidification on marine biota and the global carbon cycle (Doney et al. 2009; Köhler et al. 2010; Rau et al. 2012; Williamson and Turley 2012; Albright et al. 2016; Bach et al. 2019). OA could be jointly implemented with enhanced weathering (see section 12.3.1.2), spreading the finely crushed rock in the ocean rather than land. Regional alkalinisation could be effective in protecting coral reefs against acidification (Feng et al. 2016) (Mongin et al., 2021) and coastal OA could be part of a broader strategy for geochemical management of the coastal zone, safeguarding specific coastal ecosystems from the adverse impact of ocean acidification, such as important shellfisheries (Meysman and Montserrat 2017).

Trade-offs and spill-over effects: There is a paucity of research on biological effects of alkalinity addition. The very few studies that have explored the impact of elevated alkalinity on ocean ecosystems have largely been limited to single species experiments (Cripps et al. 2013; Gore et al. 2019) and a constrained field study quantifying the net calcification response of a coral reef flat to alkalinity enhancement (Albright et al. 2016). The addition rate would have to be great enough to overcome mixing of the local seawater with the ambient environment, but not sufficient to detrimentally impact ecosystems. More research is required to assess locations in which this may be feasible, and how such a scheme may operate (Renforth and Henderson 2017). The environmental impact of large-scale release of natural dissolution products into the coastal environment will strongly depend on the scale of olivine application, the characteristics of the coastal water body (e.g., residence time) and the particular biota present (e.g., coral reefs will react differently compared with seagrasses) (Meysman and Montserrat 2017). Model simulations (González et al. 2018) suggest that termination of OA implemented on a massive scale under a high CO_2 emission scenario (RCP8.5) might pose high risks to biological systems sensitive to rapid environmental changes because it would cause a sharp increase in ocean acidification. For example, OA termination would lead to a decrease in surface pH in warm shallow regions where vulnerable coral reefs are located, and a drop in the carbonate saturation state. However, other studies with lower levels of OA have shown no termination effect (Keller et al., 2014).

Blue carbon management. The term "blue carbon" was used originally to refer to biological carbon sequestration in all marine ecosystems, but it is increasingly applied to CDR associated with rooted vegetation in the coastal zone, such as tidal marshes, mangroves and seagrasses. Potential for carbon sequestration in other coastal and non-coastal ecosystems, such as macroalgae (e.g., kelp), is debated (Krause-Jensen et al., 2018; Krause-Jensen and Duarte, 2016). In this report, blue carbon refers to CDR through coastal blue carbon management.

Status: In recent years, there has been increasing research on the potential, effectiveness, risks, and possibility of enhancing CO_2 sequestration in shallow coastal ecosystems (Duarte, 2017). About 20% of the countries that are signatories to the Paris Agreement refer to blue carbon approaches for climate change mitigation in their NDCs and are moving toward measuring blue carbon in inventories. About 40% of those same countries have pledged to manage shallow coastal ecosystems for climate change adaptation (Kuwae and Hori 2019).

Costs: There are large differences in cost of CDR applying blue carbon management methods between different ecosystems (and at the local level). Median values are estimated as 240, 30,000, and 7,800 USD tCO_2^{-1}, respectively for mangroves, salt marsh and seagrass habitats (Gattuso et al. 2021). Currently estimated cost effectiveness (for climate change mitigation) is very low (Siikamäki et al. 2012; Bayraktarov et al. 2016; Narayan et al. 2016).

Potentials: Globally, the total potential carbon sequestration rate through blue carbon CDR is estimated in the range 0.02–0.08 $GtCO_2$ yr^{-1} (Wilcox et al. 2017; National Academies of Sciences 2019). Gattuso et al. (2021) estimate the theoretical cumulative potential of coastal blue carbon management by 2100 to be 95 $GtCO_2$, taking into account the maximum area that can be occupied by these habitats and historic losses of mangroves, seagrass and salt marsh ecosystems.

Risks and impacts: For blue carbon management, potential risks relate to the high sensitivity of coastal ecosystems to external impacts associated with both degradation and attempts to increase carbon sequestration. Under expected future warming, sea-level rise and changes in coastal management, blue carbon ecosystems are at risk, and their stored carbon is at risk of being lost (Bindoff et al. 2019).

Co-benefits: Blue carbon management provides many non-climatic benefits and can contribute to ecosystem-based adaptation, also reducing emissions associated with habitat degradation and loss (Howard et al. 2017; Hamilton and Friess 2018). Shallow coastal ecosystems have been severely affected by human activity; significant areas have already been deforested or degraded and continue to be denuded. These processes are accompanied by carbon emissions. The conservation and restoration of coastal ecosystems, which will lead to increased carbon sequestration, is also essential for the preservation of basic ecosystem services, and healthy ecosystems tend to be more resilient to the effects of climate change.

Trade-offs and spill-over effects: Blue carbon management schemes should consist of a mix of restoration, conservation and areal increase, including complex engineering interventions that enhance natural capital, safeguard their resilience and the ecosystem services they provide, and decrease the sensitivity of such ecosystems to further disturbances.

Overview of other ocean-based CDR approaches

Artificial Upwelling This concept uses pipes or other methods to pump nutrient-rich deep ocean water to the surface where it has a fertilizing effect (see OF section). To achieve CO_2 removal at a Gt magnitude, modelling studies have shown that artificial upwelling would have to be implemented on a massive scale (over 50% of the ocean to deliver maximum rate of $10GtCO_2$ yr^{-1} under RCP 8.5) (Oschlies et al., 2010, Keller et al. 2014). Because the deep water is much colder than surface water, at massive scale this could cool the Earth's surface by several degrees, but the cooling effect would cease as the deeper ocean warms, and would reverse, leading to rapid warming, if the pumping ceased (Oschlies et al., 2010, Keller et al. 2014).

Furthermore, the cooling would also severely alter atmospheric circulation and precipitation patterns (Kwiatkowski et al. 2015). Several upwelling approaches have been developed and tested (Pan et al., 2016) and more R&D is underway.

Terrestrial biomass dumping There are proposals to sink terrestrial biomass (crop residues or logs) into the deep ocean as a means of sequestering carbon (Strand and Benford 2009). Sinking biochar has also been proposed (Miller and Orton, 2021). Decomposition would be inhibited by the cold and sometimes hypoxic/anoxic environment on the ocean floor, and absence of bacteria that decompose terrestrial lignocellulosic biomass, so storage timescale is estimated at hundreds to thousands of years (Strand and Benford 2009)(Burdige 2005). Potential side-effects on marine ecosystems, chemistry, or circulation have not been thoroughly assessed. Neither have these concepts been evaluated with respect to the impacts on land from enhanced transfer of nutrients and organic matter to the ocean, nor the relative merits of alternative applications of residues and biochar as an energy source or soil amendment (Chapter 7).

Marine biomass CDR options Proposals have been made to grow macroalgae (Duarte et al., 2017) for BECCS (N'Yeurt et al. 2012; Duarte et al. 2013; Chen et al., 2015), to sink cultured macroalgae into the deep sea, or to use marine algae for biochar (Roberts et al., 2015). Naturally growing sargassum has also been considered for these purposes (Bach et al., 2021). Froehlich et al. (2019) found a substantial area of the ocean (ca. 48 million km^2) suitable for farming seaweed. N'Yeurt et al. (2012) suggested that converting 9% of the oceans to macroalgal aquaculture could take up 19 $GtCO_2$ in biomass, generate 12 Gt per annum of biogas, and the CO_2 produced by burning the biogas could be captured and sequestered. Productivity of farmed macroalgae in the open ocean could potentially be enhanced through fertilizing via artificial upwelling (Fan et al., 2020) or through cultivation platforms that dive at night to access nutrient-rich waters below the, often nutrient-limited, surface ocean. If the biomass were sunk, it is unknown how long the carbon would remain in the deep ocean and what the additional impacts would be. Research and development on macroalgae cultivation and use is currently underway in multiple parts of the world, though not necessarily directly focused on CDR.

Extraction of CO_2 from seawater (with storage) CO_2 can be extracted by applying a vacuum, or by purging with a gas low in CO_2 (Koweek et al., 2016). CO_2 stripping can also be accomplished by acidifying seawater with a mineral acid, or through electrodialysis and electrolysis, to convert bicarbonate ions (HCO_3^-) to CO_2 (Eisaman et al., 2018; Eisaman 2020; Willauer et al., 2017, Digdaya et al., 2020) Sharifian et al., 2021). The removal of CO_2 from the ocean surface leads to undersaturation in the water, thus forcing CO_2 to move from the atmosphere into the ocean to restore equilibrium. Electrochemical seawater CO_2 extraction has been modeled, prototyped, and analyzed from a techno-economic perspective (Eisaman et al., 2012; Willauer et al., 2017; de Lannoy et al., 2018; Eisaman et al., 2018a; Eisaman et al., 2018b).

Status, costs, potentials, risk and impacts, co-benefits, trade-offs and spill-over effects and the role in mitigation pathways of ocean-based approaches are summarised in Table 12.6.

12.3.1.4 Feasibility assessment

Following the framework presented in Section 6.4 and Annex II.11, a multi-dimensional feasibility assessment on the CDR methods covered here is provided in Figure 12.4, taking into account the assessment presented in this section. Both DACCS and EW perform positively on the geophysical and technological dimensions while for ocean-based approaches performance is mixed. There is limited evidence to assess social-cultural, environmental/ecological, and institutional dimensions as the literature is still nascent for DACCS and EW, while these aspects are positive for blue carbon and mixed or negative for ocean fertilization. On the economic dimension, the cost is assessed negatively for all CDR methods.

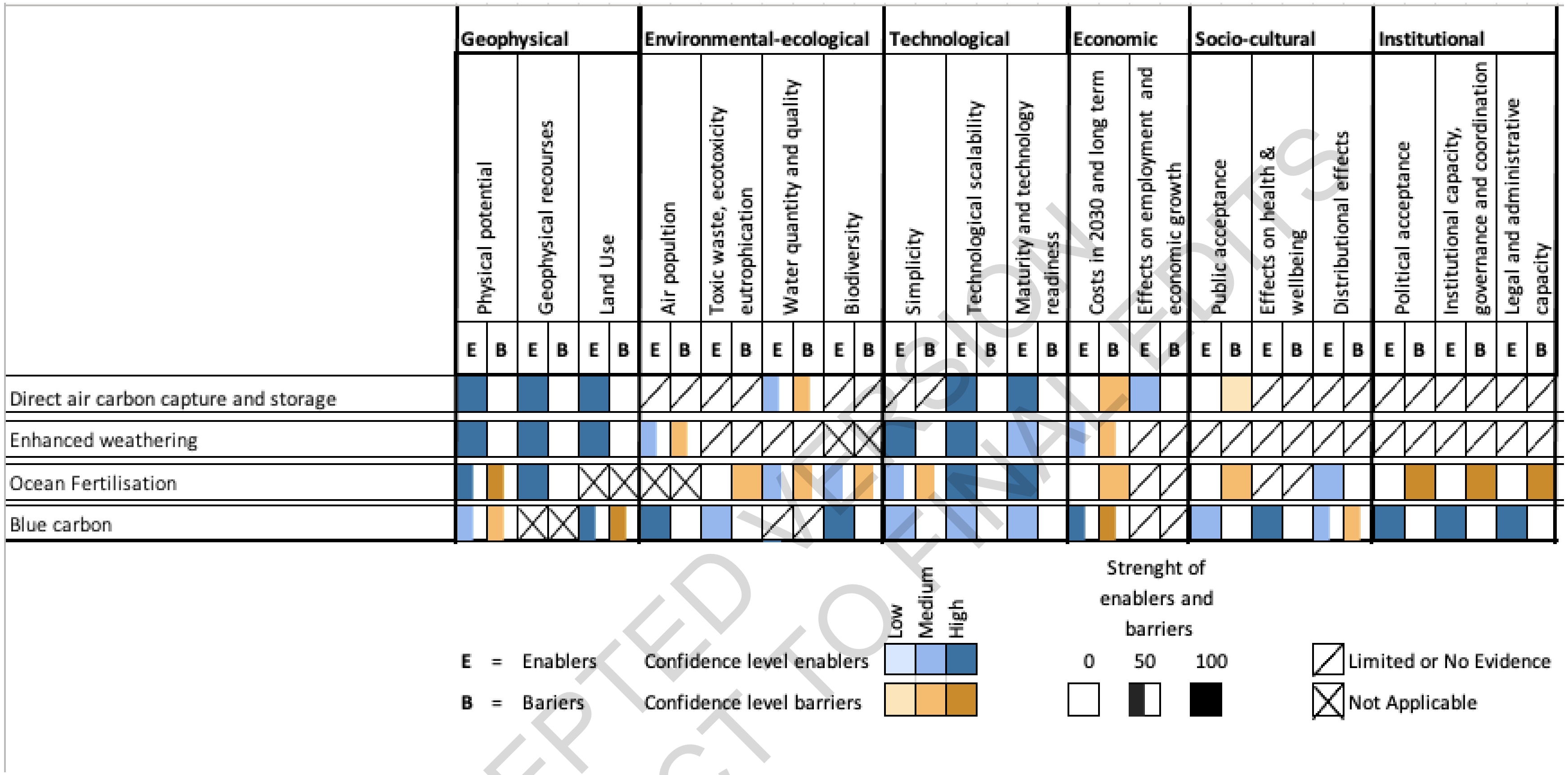


Figure 12.4 Summary of the extent to which different factors would enable or inhibit the deployment of the carbon dioxide methods DACCS, EW, ocean fertilisation and blue carbon management.

Blue bars indicate the extent to which the indicator enables the implementation of the CDR method (E) and orange bars indicate the extent to which an indicator is a barrier (B) to the deployment of the method, relative to the maximum possible barriers and enablers assessed. An 'X' signifies the indicator is not applicable or does not affect the feasibility of the method, while a forward slash indicates that there is no or limited evidence whether the indicator affects the feasibility of the method. The shading indicates the level of confidence, with darker shading signifying higher levels of confidence. Supplementary Material 12.B provides an overview of the factors affecting the feasibility of CDR methods and how they differ across context (e.g., region), time (e.g., 2030 versus 2050), and scale (e.g., small versus large), and includes a line of sight on which the assessment is based. The assessment methodology is explained in Annex II, Part II, Section 11.

12.3.2 Consideration of methods assessed in sectoral chapters; A/R, biochar, BECCS, soil carbon sequestration

Status: BECCS, A/R, soil carbon sequestration (SCS) and biochar are land-based biological CDR methods (Smith et al. 2016). BECCS combines biomass use for energy with CCS to capture and store the biogenic carbon geologically (Section 6.4.2.6); A/R and SCS involve fixing atmospheric carbon in biomass and soils, and biochar involves converting biomass to biochar and using it as a soil amendment. These CDR methods can be associated with both co-benefits and adverse side-effects, see Section 7.4, 12.5 and (Schleicher et al. 2019; Smith et al. 2019b; Hurlbert et al. 2019; Mbow et al. 2019; Olsson et al. 2019; Smith et al. 2016; Babin et al. 2021; Dooley et al. 2021).

Among CDR methods, BECCS and A/R are most commonly selected by IAMs to meet the requirements of likely limiting warming to 2°C or lower. This is partially because of the long lead time required to refine IAMs to include additional methods and update technoeconomic parameters. Currently, few IAMs represent SCS or biochar (Frank et al. 2017). Given the removal potential of SCS and biochar and some potential co-benefits, more efforts should be made to include these methods within IAMs, so that their mitigation potential can be compared to other CDR methods, along with possible co-benefits and adverse side effects (Smith et al. 2016; Rogelj et al. 2018) (Section 12.5).

Potential: The technical potential for BECCS by 2050 is estimated at 0.5–11.3 $GtCO_2$-eq yr^{-1} (Chapter 7, Table 7.3). These potentials do not include avoided emissions resulting from the use of heat, electricity and/or fuels provided by the BECCS system, which depends on substitution patterns, conversion efficiencies, and supply chain emissions for the BECCS and substituted energy systems (see Box 7.7 in Chapter 7). The mitigation effect of BECCS also depends on how deployment affects land carbon stocks and sink strength (see section 7.4.4).

As detailed in Chapter 7, the technical potential for gross removals realised through A/R in 2050 is 0.5–10.1 $GtCO_2$-eq yr^{-1} , and for improved forest management the potential is 1–2.1 $GtCO_2$-eq yr^{-1} (including both CDR and emissions reduction). Technical potential for SCS in 2050 is estimated to be 0.6–9.4 $GtCO_2$-eq yr^{-1}, for agroforestry it is 0.3–9.4 $GtCO_2$-eq yr^{-1}, and for biochar it is 0.2–6.6 $GtCO_2$-eq yr^{-1}. Peatland and coastal wetland restoration have a technical potential of 0.5–2.1 $GtCO_2$-eq yr^{-1} in 2050, with an estimated 80% of the potential being CDR. Note that these potentials reflect only biophysical and technological conditions and become reduced when factoring in economic, environmental, socio-cultural and institutional constraints (Table 12.6).

Costs: Costs across technologies vary substantially (Smith et al. 2016) and were estimated to be 15–400 USD tCO_2^{-1} for BECSS, 0–240 USD tCO_2^{-1} for A/R, -45–100 USD tCO_2^{-1} for SCS and10–345 USD tCO_2^{-1} for biochar. Fuss et al. (2018), estimated abatement cost ranges for BECCS, A/R, SCS and biochar to be 100–200, 5–50, 0–100, and 30–120 tCO_2-eq^{-1} respectively, corresponding to 2100 potentials. Ranges for economic potential (<100 USD tCO_2^{-1}) reported in Chapter 7 are 0.5–3.0 $GtCO_2$ yr^{-1} (A/R); 0.6–1.9 $GtCO_2$ yr^{-1} (improved forest management); 0.7–2.5 $GtCO_2$ yr^{-1} (SCS); 0.4–1.1 $GtCO_2$ yr^{-1} (agroforestry); 0.3–1.8 $GtCO_2$ yr^{-1} (biochar); 0.2–0.8 $GtCO_2$ yr^{-1} (peatland and coastal wetland restoration).

Risks, impacts, and co-benefits: a brief summary of risks, impacts and co-benefits is provided here and more detail is provided in chapter 7 and Section 12.5. A/R and biomass production for BECCS and biochar potentially compete for land, water and other resources, implying possible adverse outcomes for ecosystem health, biodiversity, livelihoods and food security (medium evidence, high agreement) Smith et al. 2016; Heck et al. 2018; Hurlbert et al. 2019; Mbow et al. 2019) (Chapter 7). SCS requires addition of nitrogen and phosphorus to maintain stoichiometry of soil organic matter, leading to a potential risk of eutrophication (Fuss et al. 2018). Apart from possible negative effects associated with biomass supply, adverse side-effects from biochar are relatively low if the biomass is uncontaminated (Tisserant and Cherubini 2019).

Possible climate risks relate to direct and/or indirect land carbon losses (A/R, BECCS, biochar), increased N_2O emissions (BECCS, SCS), saturation and non-permanence of carbon storage (A/R, SCS) (Jia et al. 2019; Smith et al. 2019b) (Chapter 7), and potential CO_2 leakage from deep geological reservoirs (BECCS) (Chapter 6). Land cover change associated with A/R and biomass supply for BECCS and biochar may cause albedo changes that reduce mitigation effectiveness (Jia et al. 2019; Fuss et al. 2018). Potentially unfavourable albedo change resulting from biochar use can be minimised by incorporating biochar into the soil (Fuss et al. 2018)(Chapter 7)

Concerning co-benefits, A/R and biomass production for BECCS or biochar could improve soil carbon, nutrient and water cycling (robust evidence, high agreement), and contribute to market opportunities, employment and local livelihoods, economic diversification, energy security, and technology development and transfer (medium evidence, high agreement) (Chapter 7)(Fuss et al. 2018). It may contribute to reduction of other air pollutants, health benefits, and reduced dependency on imported fossil fuels. A/R can improve biodiversity if native and diverse species are used, and (Fuss et al. 2018). For biochar, additional co-benefits include increased crop yields and reduced drought impacts, reduced CH_4 and N_2O emissions from soils (Section 7.4.5.2) (Joseph et al., 2021). SCS can improve soil quality and resilience and improve agricultural productivity and food security (Frank et al. 2017; Smith et al. 2019c).

Role in Mitigation Pathways: Biomass use for BECCS in 2050 is 61 EJ yr^{-1} (13–208 EJ yr^{-1}, 5-95 percentile range) in scenarios limiting warming to 1.5˚C with no or limited overshoot (C1, excluding traditional energy). This corresponds to 5.3 $GtCO_2$ yr^{-1} (1.1-18 $GtCO_2$ yr^{-1}) CDR, if assuming 28 kg C GJ^{-1} biomass carbon content and 85% capture rate in BECCS systems. In scenarios likely to limit warming to 2˚C (C3), biomass use for BECCS in 2050 is 28 EJ yr^{-1} (0–96 EJ yr-1, 5-95 percentile range), corresponding to 2.4 $GtCO_2$ yr^{-1} (0-8.3 $GtCO_2$ yr^{-1}) CDR. Cumulative net CO_2 removals on managed land (CDR through A/R minus land C losses due to deforestation) in the period 2020-2100 is 262 $GtCO_2$ (17–397 $GtCO_2$) and 209 $GtCO_2$ (20–415 $GtCO_2$) in C1 and C3 scenarios, respectively (5-95 percentile range).

Uncertainties remain in two main areas: the availability of land and biomass, which is affected by many factors (see Chapter 7) (Anandarajah et al. 2018), and the role of other mitigation measures including CDR methods other than A/R and BECCS. Strong near-term climate change mitigation to limit overshoot, and deployment of other CDR methods than A/R and BECCS, may significantly reduce the contribution of these CDR methods in scenarios limiting warming to 1.5˚C or 2˚C (Köberle 2019; Hasegawa et al. 2021).

Trade-offs and spill-overs: Some land-based biological CDR methods, such as BECCS and A/R, demand land. Combining mitigation strategies has the potential to increase overall carbon sequestration rates (Humpenöder et al. 2014). However, these CDR methods may also compete for resources (Frank et al. 2017). Land-based mitigation approaches currently propose the use of forests (i) as a source of woody biomass for bioenergy and various biomaterials, and (ii) for carbon sequestration in vegetation, soils, and forest products. Forests are therefore required to provide both provisioning (biomass feedstock) and regulating (carbon sequestration) ecosystem services. This multifaceted strategy has the potential to result in trade-offs (Makkonen et al. 2015). Some land-based mitigation options could conflict with biodiversity goals, e.g., A/R using monoculture plantations can reduce species richness when introduced into (semi-)natural grasslands (Smith et al. 2019a; Dooley et al. 2021). When trade-offs exist between biodiversity protection and mitigationobjectives, biodiversity is typically given a lower priority, especially if the mitigation option is considered risk-free and economically feasible (Pörtner et al. 2021). Approaches that promote synergies, such as sustainable forest management (SFM), reducing deforestation rates, cultivation of perennial crops for bioenergy in sustainable farming practices, and mixed-species forests in A/R, can mitigate biodiversity impacts and even improve ecosystem capacity to support biodiversity while mitigating climate change (Pörtner et al. 2021) (Section 12.5). Systematic land-use planning could help to deliver land-based mitigation options that also limit trade-offs with biodiversity (Longva et al. 2017) (Cross-Working Group Box 3: Mitigation and Adaptation via the Bioeconomy).

Status, costs, potentials, risk and impacts, co-benefits, trade-offs and spill-over effects and the role in mitigation pathways of A/R, biochar, SCS, peatland and coastal wetland restoration, agroforestry and forest management are summarised in Table 12.6. See also 12.5.

Table 12.6 Summary of status, costs, potentials, risk and impacts, co-benefits, trade-offs and spill over effects and the role in mitigation pathways for CDR methods. TRL = Technology Readiness Level. Author judgement ranges (assessed by authors in the literature) are shown, with full literature ranges shown in brackets

CDR option	Status (TRL)	Cost (USD tCO_2^{-1})	Mitigation Potential ($GtCO_2\ yr^{-1}$)	Risk & Impacts	Co-benefits	Trade-offs and spill over effects	Role in modelled mitigation pathways	Section
DACCS	6	100–300 (84–386)	5–40	Increased energy and water use.	Water produced (solid sorbent DAC designs only).	Potentially increased emissions from water supply and energy generation.	In a few IAMs; DACCS complements other CDR methods.	{12.3.1.1}
Enhanced weathering	3–4	50–200 (24–578)	2–4 (<1–95)	Mining impacts; air quality impacts of rock dust when spreading on soil.	Enhanced plant growth, reduced erosion, enhanced soil carbon, reduced pH, soil water retention.	Potentially increased emissions from water supply and energy generation.	In a few IAMs; EW complements other CDR methods.	{12.3.1.2}
Ocean alkalinity enhancement	1–2	40–260	1–100	Increased seawater pH and saturation states and may impact marine biota. Possible release of nutritive or toxic elements and compounds. Mining impacts.	Limiting ocean acidification.	Potentially increased emissions of CO_2 and dust from mining, transport and deployment operations.	No data.	{12.3.1.3}
Ocean fertilisation	1–2	50–500	1–3	Nutrient redistribution, restructuring of the ecosystem, enhanced oxygen consumption and acidification in deeper waters, potential for decadal-to-millennial-scale return to the atmosphere of nearly all the extra carbon	Increased productivity and fisheries, reduced upper ocean acidification.	Subsurface ocean acidification, deoxygenation; altered meridional supply of macro-nutrients as they are utilized in the iron-fertilized region and become unavailable for transport to, and	No data.	{12.3.1.3}

				removed, risks of unintended side effects.		utilization in other regions, fundamental alteration of food webs, biodiversity		
Blue carbon management in coastal wetlands	2–3	Insufficient data, estimates range from ~ 100 to ~ 10000	<1	If degraded or lost, coastal blue carbon ecosystems are likely to release most of their carbon back to the atmosphere; potential for sediment contaminants, toxicity, bioaccumulation and biomagnification in organisms; issues related to altering degradability of coastal plants; use of subtidal areas for tidal wetland carbon removal; effect of shoreline modifications on sediment redeposition and natural marsh accretion; abusive use of coastal blue carbon as means to reclaim land for purposes that degrade capacity for carbon removal.	Provide many non-climatic benefits and can contribute to ecosystem-based adaptation, coastal protection, increased biodiversity, reduced upper ocean acidification; could potentially benefit human nutrition or produce fertiliser for terrestrial agriculture, anti-methanogenic feed additive, or as an industrial or materials feedstock.	If degraded or lost, coastal blue carbon ecosystems are likely to release most of their carbon back to the atmosphere. The full delivery of the benefits at their maximum global capacity will require years to decades to be achieved	Not incorporated in IAMs, but in some bottom-up studies: small contribution.	{12.3.1.3}, Chapter 7, Section 7.4
BECCS	5–6	15–400	0.5–11	Competition for land and water resources, to grow biomass feedstock. Biodiversity and carbon stock loss if from unsustainable biomass harvest.	Reduction of air pollutants; fuel security, optimal use of residues, additional income, health benefits and if implemented well can enhance biodiversity, soil health and land carbon	Competition for land with biodiversity conservation and food production	Substantial contribution in IAMs and bottom -up sectoral studies	Chapter 7, Section 7.4

Afforestation/Reforestation	8–9	0–240	0.5–10	Reversal of carbon removal through wildfire, disease, pests may occur. Reduced catchment water yield and lower groundwater level if species and biome are inappropriate.	Enhanced employment and local livelihoods, improved biodiversity, improved renewable wood products provision, soil carbon and nutrient cycling. Possibly less pressure on primary forest.	Inappropriate deployment at large scale can lead to competition for land with biodiversity conservation and food production.	Substantial contribution in IAMs and also in bottom-up sectoral studies.	Chapter 7, Section 7.4
Biochar	6–7	10–345	0.3–6.6	Particulate and GHG emissions from production; biodiversity and carbon stock loss from unsustainable biomass harvest.	Increased crop yields and reduced non-CO_2 emissions from soil; and resilience to drought.	Environmental impacts associated particulate matter; competition for biomass resource.	In development - not yet in global mitigation pathways simulated by IAMs.	Chapter 7, Section 7.4
Soil Carbon Sequestration in croplands and grasslands	8–9	45–100	0.6–9.3	Risk of increased nitrous oxide emissions due to higher levels of organic nitrogen in the soil; risk of reversal of carbon sequestration.	Improved soil quality, resilience and agricultural productivity.	Attempts to increase carbon sequestration potential at the expense of production. Net addition per hectare is very small; hard to monitor.	In development - not yet in global mitigation pathways simulated by IAMs; in bottom-up studies: with medium contribution.	Chapter 7, Section 7.4
Peatland and coastal wetland restoration	8–9	Insufficient data	0.5–2.1	Reversal of carbon removal in drought or future disturbance. Risk of increased methane emissions.	Enhanced employment and local livelihoods, increased productivity of fisheries, improved biodiversity, soil carbon and nutrient cycling.	Competition for land for food production on some peatlands used for food production.	Not in IAMs but some bottom-up studies with medium contribution.	Chapter 7, Section 7.4

Agroforestry	8–9	Insufficient data	0.3–9.4	Risk that some land area lost from food production; requires high skills.	Enhanced employment and local livelihoods, variety of products improved soil quality, more resilient systems.	Some trade-off with agricultural crop production, but enhanced biodiversity, and resilience of system.	No data from IAMs, but in bottom-up sectoral studies. with medium contribution.	Chapter 7, Section 7.4
Improved Forest management	8–9	Insufficient data	0.1–2.1	If improved management is understood as merely intensification involving increased fertiliser use and introduced species, then it could reduce biodiversity and increase eutrophication.	In case of sustainable forest management, it leads to enhanced employment and local livelihoods, enhanced biodiversity, improved productivity.	If it involves increased fertiliser use and introduced species it could reduce biodiversity and increase eutrophication and upstream GHG emissions.	No data from IAMs, but in bottom-up sectoral studies with medium contribution.	Chapter 7, Section 7.4

12.3.3 CDR governance and policies

As shown in Cross-Chapter Box 8 in this Chapter, CDR fulfils multiple functions in different phases of ambitious mitigation: (1) further reducing net CO_2 or GHG emission levels in the near-term; (2) counterbalancing residual emissions (from hard-to-transition sectors like transport, industry, or agriculture) to help reach net zero CO_2 or GHG emissions in the mid-term; (3) achieving and sustaining net-negative CO_2 or GHG emissions in the long-term. While inclusion of emissions and removals on managed land (LULUCF) is mandatory for developed countries under UNFCCC inventory rules (Grassi et al. 2021), not all Annex I countries have included land-based biological removals when setting domestic mitigation targets in the past, but updated NDCs for 2030 indicate a shift, most notably in the European Union (Gheuens and Oberthür 2021; Schenuit et al. 2021). The early literature on CDR governance and policy has been mainly conceptual rather than empirical, focusing on high-level principles (see the concerns listed in introduction of Section 12.3) and the representation of CDR in global mitigation scenarios (see Section 3.2.2). However, with the widespread adoption of net zero targets and the recognition that CDR is a necessary element of mitigation portfolios to achieve net zero CO_2 or GHG emissions, countries with national net zero emissions targets have begun to integrate CDR into modelled national mitigation pathways, increase research, development & demonstration (RD&D) efforts on CDR methods, and consider CDR-specific incentives and policies (Honegger et al. 2021b; Schenuit et al. 2021), (Box 12.1). Nevertheless, this increasing consideration of CDR has not yet extended to net-negative targets and policies to achieve these. While the use of CDR at levels that would lead to net negative CO_2 or GHG emissions in the long-term has been assumed in most global mitigation scenarios that limit warming to 1.5°C, net-negative emissions trajectories and BECCS as the main CDR method modelled to achieve these have not been mirrored by corresponding UNFCCC decisions so far (Fridahl 2017; Mohan et al. 2021). Likewise, only a few national long-term mitigation plans or legal acts already entail a vision for net-negative GHG emissions (Buylova et al. 2021), for example Finland, Sweden, Germany and Fiji).

For countries with emissions targets aiming for net zero or lower, the core governance question is not whether CDR should be mobilised or not, but which CDR methods governments want to see deployed by whom, by when, at which volumes and in which ways (Bellamy and Geden 2019; Minx et al. 2018). The choice of CDR methods and the scale and timing of their deployment will depend on the respective ambitions for gross emission reductions, how sustainability and feasibility constraints are managed, and how political preferences and social acceptability evolve (Bellamy 2018; Forster et al. 2020; Fuss et al. 2020; Waller et al. 2020; Clery et al. 2021; Iyer et al. 2021; Rogelj et al. 2021). As examples of emerging CDR policymaking at (sub-)national levels show, policymakers are beginning to incorporate CDR methods beyond those currently dominating global mitigation scenarios, i.e. BECCS and afforestation/reforestation (Box 12.1) (Bellamy and Geden 2019; Buylova et al. 2021; Schenuit et al. 2021; Uden et al. 2021). CDR policymaking is faced with the need to consider method-specific timescales of CO_2 storage, as well as challenges in MRV and accounting, potential co-benefits, adverse side effects, interactions with adaptation and trade-offs with SDGs (Table 12.6) (Dooley and Kartha 2018; McLaren et al. 2019; Buck et al. 2020; Honegger et al. 2020; Brander et al. 2021; Dooley et al. 2021; Mace et al. 2021). Therefore, CDR governance and policymaking is expected to focus on responsibly incentivising RD&D and targeted deployment, building on both technical and governance experience with already widely practiced CDR methods like afforestation/reforestation (Lomax et al. 2015; Field and Mach 2017; Bellamy 2018; Carton et al. 2020; VonHedemann et al. 2020), as well as learning from two decades of slow-moving CCS deployment (Buck 2021; Martin-Roberts et al. 2021; Wang et al. 2021). For some less well understood methods and implementation options, such as ocean alkalinisation or enhanced weathering, investment in RD&D can help in understanding the risks, rewards, and uncertainties of deployment (Nemet et al. 2018; Fajardy et al. 2019; Burns and Corbett 2020; Goll et al. 2021).

START BOX 12.1 HERE

Box 12.1 Case Study: Emerging CDR policy, research and development in the United Kingdom

Climate change mitigation policies in the UK have been motivated since 2008 by a domestic, legally-binding framework. This framework includes a 2050 target for net greenhouse gas emissions, interim targets and an independent advisory body called the Climate Change Committee (Muinzer 2019). It has led successive UK governments to publish mitigation plans to 2050, causing policy to be more forward-looking (Averchenkova et al. 2021).

The UK's targets include emissions and removals from LULUCF. In 2008 the target for 2050 was an economy-wide net emissions reduction of at least 80% below 1990 levels. Even the first government plans to achieve this target proposed deployment of removal methods, specifically afforestation and wood in construction, increased soil carbon and BECCS (HM Government 2011).

Adoption of the Paris Agreement in 2015 caused the government to change the legislated 2050 target to a reduction of at least 100% (i.e. net zero). Since then, removal of CO_2 and other greenhouse gases has received greater prominence as a distinct topic. The most recent national plan (published October 2021) proposes deployment not only of the methods mentioned above, but also DACCS, biochar and enhanced weathering. The government has committed to amend accounting of UK targets to include a wider range of removal methods beyond LULUCF, and set a target of 5 $MtCO_2$ yr^{-1} from methods such as BECCS, DACCS and enhanced weathering by 2030. It is consulting on markets and incentives for deployment, and exploring new requirements for MRV (HM Government 2021).

In parallel to these policy developments, the UK funds research into technical, environmental and social aspects of removal (Lezaun et al. 2021). Research on some elements (e.g., forestry, CCS, soils, bioenergy) have been funded for well over a decade, but the first programme dedicated to greenhouse gas removal ran during 2017-2021. This has been followed by two new programmes with greater focus on demonstration, totalling £100m over four years (HM Government 2021). A wide variety of methods is supported in these programmes, covering approaches such as CO_2 capture from seawater and capture of methane from cattle, in addition to those included already in national mitigation scenarios.

Deployment of removal methods has lagged expectations, as national targets for tree planting are not being met and infrastructure for CO_2 transport and storage is not yet in place (Climate Change Committee 2021). While public awareness around carbon removal is low, studies indicate support in general, provided it is perceived as enhancing rather than impeding action to reduce emissions (Cox et al. 2020a).

END BOX 12.1 HERE

Since the enhancement of carbon sinks is a form of climate change mitigation (Honegger et al. 2021a), CDR governance challenges will in many respects be similar to those around emissions reduction measures, as will policy instruments like RD&D funding, carbon pricing, tax or investment credits, certification schemes, and public procurement (see Sections 13.4, 13.6, 14.4, 14.5). Effectively integrating CDR into mitigation portfolios can build on already existing rules, procedures and instruments for emissions abatement (Torvanger 2019; Fridahl et al. 2020; Zakkour et al. 2020; Honegger et al. 2021b; Mace et al. 2021; Rickels et al. 2021). Additionally, to accelerate RD&D and to incentivise CDR deployment a political commitment to formal integration into existing climate policy frameworks is required (*robust evidence, high agreement*) (Lomax et al. 2015; Geden et al. 2018; Honegger and Reiner 2018; VonHedemann et al. 2020; Schenuit et al. 2021) To avoid that CDR is misperceived as a substitute for deep emissions reductions, the prioritisation of emissions cuts can be signalled and achieved with differentiated target setting for reductions and removals (Geden et al. 2019; McLaren et al. 2019). Similarly, sub-targets are conceivable for different types of CDR, to prioritise preferred methods according to characteristics such as removal processes or timescales of storage (Smith 2021).

IPCC guidance on quantifying removals is available for land-based biological CDR methods (IPCC 2006, 2019), but has yet to be developed for other CDR methods (Royal Society and Royal Academy of Engineering 2018). Challenges with development of estimation algorithms, data collection, and attribution between sectors and countries will need to be overcome (Luisetti et al. 2020; Wedding et al. 2021). Trusted methodologies for MRV, required to enable private sector participation will need to address the permanence, leakage, and saturation challenges with land and ocean-based biological methods (Mace et al. 2021). Protocols that also capture social and ecological co-benefits, could encourage the adoption of biological CDR methods such as SCS, biochar, A/R and blue carbon management *(robust evidence, high agreement)* (VonHedemann et al. 2020; Macreadie et al. 2021).

Private capital and companies, impact investors, and philanthropy will play a role in technical demonstrations and bringing down costs, as well as creating demand for carbon removal products on voluntary markets, which companies may purchase to fulfil corporate social responsibility-driven targets (Friedmann 2019; Fuss et al. 2020; Joppa et al. 2021). Niche markets can provide entry points for limited deployment of novel CDR methods (Cox and Edwards 2019), but targeting currently existing revenue streams by using CO_2 captured from the atmosphere in Enhanced Oil Recovery and other utilisation routes (Mackler et al. 2021; Meckling and Biber 2021) is contested, and highlights the importance of choosing appropriate system boundaries when assessing supply chains (Tanzer and Ramírez 2019; Brander et al. 2021). While the private sector will play a distinct role in scaling CDR, governments will need to commit to developing infrastructure for the transport and storage of CO_2, including financing, permitting, and regulating liabilities (Sanchez et al. 2018; Mace et al. 2021; Mackler et al. 2021).

International governance considerations include global technology transfer around CDR implementation options (Batres et al. 2021); land use change that could affect food production and land condition, and cause conflict around land tenure and access (Dooley and Kartha 2018; Hurlbert et al.

2019; Milne et al. 2019); and efforts to create sustainable and just supply chains for CDR (Fajardy and Mac Dowell 2020; Tan et al. 2021), such as resources used for BECCS, enhanced weathering, or ocean alkalinisation. International governance would be particularly important for methods posing transboundary risks, especially for ocean-based methods. Specific regulations have so far only been developed in the context of the London Protocol, an international treaty that explicitly regulates ocean fertilisation and allows parties to govern other marine CDR methods like ocean alkalinity enhancement (GESAMP 2019; Burns and Corbett 2020; Boettcher et al. 2021)(see Section 14.4.5).

Engagement of civil society organisations and publics will be important for shaping CDR policy and deployment (*medium evidence, high agreement*). Public awareness of CDR and its role in national net zero emissions strategies is generally very low (Cox et al. 2020a), and perceptions differ across countries and between methods (Bertram and Merk 2020; Spence et al. 2021; Sweet et al. 2021; Wenger et al. 2021). When awareness increases, social processes will shape political attitudes on CDR (Shrum et al. 2020), as will efforts to frame particular CDR methods as 'natural' or 'technological' (Osaka et al. 2021), and the policy instruments chosen to support CDR (Bellamy et al. 2019). Lack of confidence in CDR implementation options from both publics and investors, and lack of trust in project developers (Cox et al. 2020b) have hampered support for CCS (Thomas et al. 2018) and is expected to affect deployment of CDR methods with geological storage (Gough and Mander 2019). On local and regional scales, CDR projects will need to consider air and water quality, impacts to human health, energy needs, land use and ecological integrity, and local community engagement and procedural justice. Bottom-up and community driven strategies are important for deploying equitable carbon removal projects (Hansson et al. 2021; Batres et al. 2021).

12.4 Food systems

12.4.1 Introduction

This section complements Chapter 7 by reviewing recent estimates of food system emissions and assessing options beyond the agriculture, forestry and land use sectors to mitigate food systems GHG emissions. A food system approach enables identification of cross-sectoral mitigation opportunities including both technological and behavioural options. Further, a system approach permits evaluation of policies that do not necessarily directly target primary producers or consumers, but other food system actors with possibly higher mitigation efficiency. A food system approach was introduced in the IPCC Special Report on Climate Change and Land (SRCCL) (Mbow et al. 2019). Besides major knowledge gaps in the quantification of food system GHG emissions (Section 12.4.2), the SRCCL authors identified as major knowledge gaps the understanding of the dynamics of dietary change (including behavioural patterns, the adoption of plant-based dietary patterns, and interaction with human health and nutrition of sustainable healthy diets and associated feedbacks); and instruments and mechanisms to accelerate transitions towards sustainable and healthy food systems.

Sufficient food and adequate nutrition are fundamental human needs (HLPE 2020; Ingram 2020). Food needs to be grown and processed, transported and distributed, and finally prepared and consumed. Food systems range from traditional, involving only few people and short supply chains, to modern food systems, comprising complex webs involving large numbers of stakeholders and processes that grow and transform food commodities into food products and distribute them globally (HLPE 2017; Gómez and Ricketts 2013). A 'food system' includes all food chain activities (production, processing, distribution, preparation, consumption of food) and the management of food loss and wastes. It also includes institutions and infrastructures influencing any of these activities, as well as people and systems impacted (HLPE 2017; FAO 2018a). Food choices are determined by the food environment consisting of the "physical, economic, political and socio-cultural context in which consumers engage with the food system to acquire, prepare and consume food" (HLPE 2017). Food system outcomes encompass food and nutrition, productivity, profit and livelihood of food producers and other actors in food value chains, but also social outcomes and the impact on the environment (Zurek et al. 2018). 'Sustainable healthy diets' have been defined by FAO and WHO (FAO and WHO 2019) as "dietary patterns that promote all dimensions of individuals' health and wellbeing; have low environmental pressure and impact; are accessible, affordable, safe and equitable; and are culturally acceptable."

The SRCCL estimated overall global anthropogenic emissions from food systems to range between 10.8 and 19.1 $GtCO_2$-eq yr^{-1}, equivalent to 21-37% of total anthropogenic emissions (Rosenzweig et al. 2020a; Mbow et al. 2019). The authors identified major knowledge gaps for the GHG emission inventories of food systems, particularly in providing disaggregated emissions from the food industry and transportation. The food system approach taken in the SRCCL (Mbow et al. 2019) evaluates the synergies and trade-offs of food system response options and its implications for food security, climate change adaptation and mitigation. This integrated framework allows the identification of fundamental attributes of responses to maximise co-benefits, while avoiding maladaptation measures and adverse side effects. A food system approach supports the design of interconnected climate policy responses to tackle climate change, incorporating perspectives of producers and consumers. The SRCCL (Mbow et al. 2019) found that the technical mitigation potential by 2050 of demand-side responses at 0.7–8.0 $GtCO_2$-eq yr^{-1} is comparable to supply-side options at 2.3–9.6 $GtCO_2$-eq yr^{-1}. This shows that mitigation actions need to go beyond food producers and suppliers to incorporate dietary changes and consumers' behavioural patterns and reveals that producers and consumers need to work together to reduce GHG emissions.

Though total production of calories is sufficient for the world population (Wood et al. 2018; Benton et al. 2019), availability and access to food is unequally distributed, and there is a lack of nutrient-dense foods, fruit and vegetables (Berners-Lee et al. 2018; Kc et al. 2018). In 2019, close to 750 million people were food insecure. An estimated 2 billion people lacked adequate access to safe and nutritious food in both quality and quantity (FAO et al. 2020). Two billion adults are overweight or obese through inadequate nutrition, with an upward trend globally (FAO et al. 2019). Low intake of fruit and vegetables is further aggravated by high intake rates of refined grains, sugar and sodium together leading to a high risk of non-communicable diseases such as cardiovascular disease and type 2 diabetes (Springmann et al. 2016; Clark et al. 2018, 2019; GBD 2017 Diet Collaborators et al. 2019; Willett et al. 2019) *(robust evidence, high agreement)*. At least 340 million children under 5 years of age experience lack of vitamins or other essential bio-available nutrients, including almost 200 million suffering from stunting, wasting or overweight (UNICEF 2019).

Bodirsky et al. (Bodirsky et al. 2020) find that global prevalence of overweight will increase to 39–52% of world population in 2050 (from 29% in 2010; range across the Socioeconomic

Pathways studied), and 13–20% obese people (9% in 2010). The prevalence of underweight people was predicted to approximately halve, with absolute numbers stagnating at 0.4–0.7 billion. Although many studies represent future pathways of diets and food systems, there are few holistic and consistent narratives and quantification of the future pathways of diets and food systems (Mora et al. 2020; Mitter et al. 2020). Alternative pathways for improved diets and food systems have been developed, emphasising climate, environmental and health co-benefits (Bajželj et al. 2014; Hedenus et al. 2014; Damerau et al. 2016; Weindl et al. 2017a,b; Springmann et al. 2018a; Bodirsky et al. 2020; Prudhomme et al. 2020; Hamilton et al. 2021), reduced food waste and closing yield gaps (Pradhan et al. 2014; Bajželj et al. 2014), nitrogen management (Bodirsky et al. 2014), urban and peri-urban agriculture (Kriewald et al. 2019) and different sustainability targets (Henry et al. 2018b). The FAO has examined three alternative food system scenarios: "business as usual", "towards sustainability", and "stratified societies" (FAO 2018b). Others have identified research priorities or changes in legislation needed to support adoption of improved food systems (Mylona et al. 2018).

Malnutrition aggravates susceptibility of children to various infectious diseases (Farhadi and Ovchinnikov 2018; França et al. 2009), and infectious diseases can also decrease nutrient uptake, thereby promoting malnutrition (Farhadi and Ovchinnikov 2018). Contamination of food with bacteria, viruses, parasites and microbial toxins can cause foodborne illnesses (Abebe et al. 2020; Ricci et al. 2017; Gallo et al. 2020), foodborne substances such as food additives and specific proteins can cause adverse reactions, and contamination with toxic chemical substances used in agriculture and food processing, can lead to poisoning or chronic diseases (Gallo et al. 2020). Further, health risks from food systems may originate from the use of antibiotics in livestock production and the occurrence of anti-microbial resistance in pathogens (ECDC et al. 2015; Bennani et al. 2020), or zoonotic diseases such as COVID-19 (Vågsholm et al. 2020; Gan et al. 2020; Patterson et al. 2020).

Modern food systems are highly consolidated, through vertical and horizontal integration (Swinnen and Maertens 2007; Folke et al. 2019). This consolidation has led to uneven distribution of power across the food value chain, with influence concentrated among a few actors in the post-farm gate food supply chain (e.g., large food processors and retailers), and has contributed to a loss of indigenous agriculture and food systems, for example on Pacific Islands (Vogliano et al. 2020). While agricultural producers contribute a higher proportion of GHG emissions compared with other actors in the supply chain, they have relatively little power to change the system (Leip et al. 2021; Clapp 2019; Group of Chief Scientific Advisors 2020).

In 2016, the agriculture, fisheries, and forestry sectors employed 29% of working people; employment within these sectors was 4% in developed countries, down from 9% in 1995, and 57% in least developed countries, down from 71% in 1995 (World Bank 2021). Employment in other (non-agriculture) food system sectors, such as the food processing industry and service sectors, differs between food systems. The share of total non-farm food system employment ranges from 10% in traditional food systems (e.g., Sub-Saharan Africa), to over 50% in food systems in transition (e.g., Brazil), to high shares (80%) in modern food systems (e.g., U.S.) (Townsend et al. 2017). The share of the food expenditures that farmers receive is decreasing; at the global level, this share has been estimated at 27% in 2015 (Yi et al. 2021).

12.4.2 GHG emissions from food systems

12.4.2.1 Sectoral contribution of GHG emissions from food systems

New calculations using EDGAR.v6 (Crippa et al. 2021a) and FAOSTAT (FAO 2021) databases provide territorial-based food system GHG emissions by country globally for the time period 1990 to 2018 (Crippa et al. 2021b). The data are calculated based on a combination of country-specific data and aggregated information as described by Crippa et al. (2021b) and Tubiello et al. (2021). The data show that, in 2018, 17 $GtCO_2$-eq yr^{-1} (95% confidence range 13–23 $GtCO_2$-eq yr^{-1}, calculated according to Solazzo et al. (2020) were associated with the production, processing, distribution, consumption of food and management of food system residues. This corresponded to 31% (range 23-42%) of total anthropogenic GHG emissions of 54 $GtCO_2$-eq yr^{-1}. Based on the IPCC sectoral classification (Table 12.7 and Figure 12.5), the largest contribution of food systems GHG emissions in 2018 was from agriculture, i.e. livestock and crop production systems (6.3 $GtCO_2$-eq yr^{-1}, range 2.6–11.9) and land use, land use change and forestry (LULUCF) (4.0 $GtCO_2$-eq yr^{-1}, range 2.1–5.9) (Figure 12.5). Emissions from energy use were 3.9 $GtCO_2$-eq yr^{-1} (3.6–4.4**)**, waste management 1.7 $GtCO_2$-eq yr^{-1} (0.9–2.6), and industrial processes and product use 0.9 $GtCO_2$-eq yr^{-1} (0.6–1.1). The share of GHG emissions from food systems generated outside the AFOLU (agriculture and LULUCF) sectors has increased over recent decades, from 28% in 1990 to 39% in 2018.

Energy. Emissions from energy use occur throughout the food supply chain. In 2018, the main contributions came from energy industries supplying electricity and heat (970 $MtCO_2$-eq yr^{-1}), manufacturing and construction (920 $MtCO_2$-eq yr^{-1}, of which 29% was attributable to the food, beverage, and tobacco industry), and transport (760 $MtCO_2$-eq yr^{-1}). These emissions were almost entirely as CO_2. Energy emissions from forestry and fisheries amounted to 480 $MtCO_2$-eq yr^{-1}, with 91% of emissions as CO_2. Emissions from residential and commercial fuel combustion contributed 250 $MtCO_2$-eq yr^{-1} (79% of emissions as CO_2, and with emissions of 1.7 $MtCH_4$ yr^{-1}) and 130 $MtCO_2$-eq yr^{-1} (with 98% of emissions as CO_2), respectively.

Refrigeration uses an estimated 43% of energy in the retail sector (Behfar et al. 2018) and significantly increases fuel consumption during distribution. Besides being energy intensive, supermarket refrigeration also contributes to GHG emissions through leakage of refrigerants (F-gases), although their contribution to food system GHG emissions is estimated to be minor (Crippa et al. 2021b). The cold chain accounts for approximately 1% of global GHG emissions, but as the number of refrigerators per capita in developing countries is reported to be one order of magnitude lower than the number in developed countries (19 m^3 versus 200 m^3 refrigerated storage capacity per 1000 inhabitants), the importance of refrigeration to total GHG emissions is expected to increase (James and James 2010). Although refrigeration gives rise to GHG emissions, both household refrigeration and effective cold chains could contribute to a substantial reduction in losses of perishable food and thus in emissions associated with food provision (University of Birmingham 2018; James and James 2010). A trade-off exists between reducing food waste and increased refrigeration emissions, with the benefits depending on type of produce, location and technologies used (Wu et al. 2019; Sustainable Cooling for All 2018).

Transport has overall a minor importance for food system GHG emissions, with a share of 5% to 6% (Poore and Nemecek 2018a; Crippa et al. 2021b). The largest contributor to food system transport GHG emissions was road transport (92%), followed by marine shipping (4%), rail (3%), and aviation (1%). Only looking at energy needs, air or road transport consumes one order of magnitude higher energy (road: 70–80 MJ t^{-1} km^{-1}; aviation: 100–200 MJ t^{-1} km^{-1}) than marine shipping (10–20 MJ t^{-1} km^{-1}) or rail (8–10 MJ t^{-1} km^{-1}) (FAO 2011). For specific food products with high water content, relatively low agricultural emissions and high average transport distances, the share of transport in total GHG emissions can be over 40% (e.g., bananas, with total global average GHG emissions of 0.7 $kgCO_2$-eq kg^{-1}) (Poore and Nemecek 2018a), but transport is a minor source of GHG emissions for most food products (Poore and Nemecek 2018a).

Industry. Direct industrial emissions associated with food systems are generated by the refrigerants industry (580 $MtCO_2$-eq yr^{-1} as F-gases) and the fertiliser industry for ammonia production (280 $MtCO_2$-eq yr^{-1} as CO_2) and nitric acid (60 $MtCO_2$-eq yr^{-1} as N_2O). The industry sector data account for CO_2 stored in urea (-50 $MtCO_2$-eq yr^{-1}). Packaging contributed about 6% of total food system emissions (0.98 $GtCO_2$-eq yr^{-1}, 91% as CO_2, with CH_4 emissions of 2.8 Mt CH_4 yr^{-1}). Major emissions sources are pulp and paper (60 $MtCO_2$-eq yr^{-1}) and aluminium (30 $MtCO_2$-eq yr^{-1}), with ferrous metals, glass, and plastics making a smaller contribution. High shares of emissions from packaging are found for beverages and some fruit and vegetables (Poore and Nemecek 2018a).

Waste. Management of waste generated in the food system (including food waste, wastewater, packaging waste etc.) leads to biogenic GHG emissions, and contributed 1.7 $GtCO_2$-eq yr^{-1} to food systems' GHG emissions in 2018. Of these emissions, 55% were from domestic and commercial wastewater (30 $MtCH_4$ yr^{-1} and 310 ktN_2O yr^{-1}), 36% from solid waste management (20 $MtCH_4$ yr^{-1} and 310 ktN_2O yr^{-1}), and 8% from industrial wastewater (4 $MtCH_4$ yr^{-1} and 80 ktN_2O yr^{-1}). Emissions from waste incineration and other waste management systems contributed 1%.

Table 12.7: **GHG emissions from food systems by sector according to IPCC classification in Mt gas yr^{-1} and food systems' share of total anthropogenic GHG emissions in 1990 and 2015.**

Sector	**CO_2**	**CH_4**	**N_2O**	**F-gases**	**GHG**	**CO_2**	**CH_4**	**N_2O**	**F-gases**	**GHG**
	Emissions (Mt gas yr^{-1})					**Share of total sectoral emissions (%)**				
					1990					
1 Energy	2212	10	0	-	2583	10.5	10.2	26.7	-	10.7
2 Industrial Processes	190	0	0	0	263	14.5	0	38	4.8	16.2
3 Solv + Product Use	0	-	-	-	0	0.2	-	-	-	0.2
4 Agriculture	102	142	5	-	5370	100	100	99.2	-	99.8
5 LULUCF	4946	-	0	-	5080	181	-	194	-	182
6 Waste	3	40	0	-	1155	29	72.4	99.1	-	73.2
Total	**7453**	**192**	**6**	**0**	**14452**	**29.3**	**65.2**	**84.5**	**4.8**	**40.3**
Total [$MtCO_2$-eq yr^{-1}]	**7453**	**5243**	**1755**	**0**	**14452**	**29.3**	**63.9**	**84.5**	**0.3**	**40.3**
					2018					
1 Energy	3449	13	0	-	3927	10.1	9.5	24.1	-	10.2
2 Industrial Processes	242	0	0	0	881	7.9	0	28.6	58	20.1

3 Solv + Product Use	7	-	-	-	7	4.1	-	-	-	3.6
4 Agriculture	140	161	7	-	6326	100	100	99.1	-	99.7
5 LULUCF	3823	-	1	-	3982	190	-	229	-	191
6 Waste	5	58	0	-	1699	30.6	71.8	99.1	-	72.9
Total	**7666**	**231**	**8**	**0**	**16821**	**19.3**	**61.6**	**83.7**	**58**	**31.1**
Total [MtCO2-eq yr^{-1}]	**7666**	**6317**	**2256**	**581**	**16821**	**19.3**	**60.2**	**83.7**	**53.6**	**31.1**

Notes: Agricultural emissions include the emissions from the whole sector; biomass production for non-food use currently not differentiated. Non-food system AFOLU emissions are a carbon sink, therefore the share of AFOLU food system emissions is > 100%. Source: EDGARv5 (Crippa et al. 2019, 2021b), and FAOSTAT (FAO 2021). Solv+Produc Use = Solvent and Other Product Use; LULUCF: Land Use, Land-Use Change & Forestry.

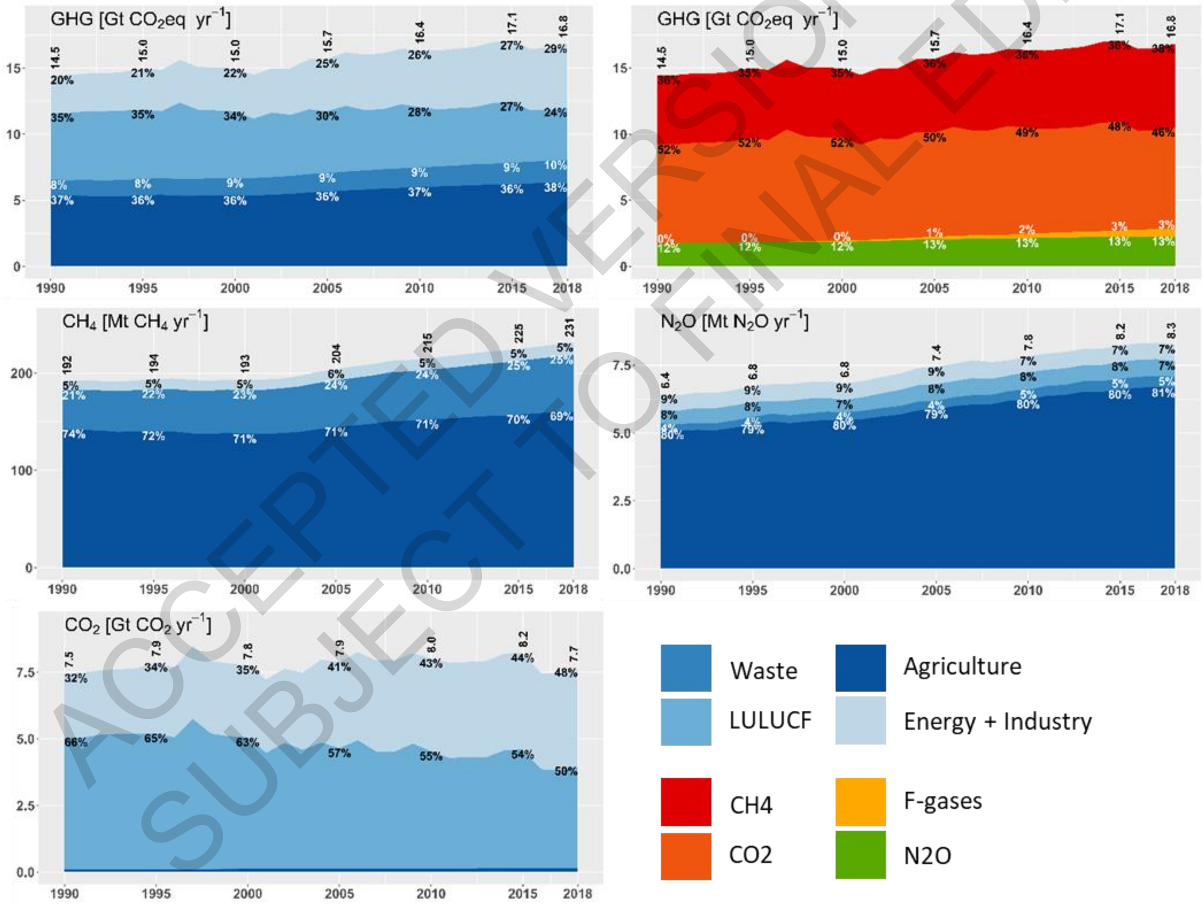


Figure 12.5: Food system GHG emissions from the agriculture, LULUCF, waste, and energy & industry sectors. Source: Crippa et al. (2021b).

12.4.2.2 GHG intensities of food commodities

There is high variability in the GHG emissions of different food products and production systems (Figure 12.6). GHG emissions intensities – measured using attributional Life Cycle Assessment, considering the full supply-chain, expressed as CO_2-eq per kg of product or per kg of protein – are generally highest for ruminant meat, cheese, and certain crustacean species (e.g., farmed shrimp and prawns, trawled lobster) (Nijdam et al. 2012; Clark and Tilman 2017; Clune et al. 2017; Hilborn et al. 2018; Poore and Nemecek 2018) *(robust evidence, high agreement)*. Generally, beef from dairy systems has a lower footprint (8–23 kgCO_2-eq (100g protein)$^{-1}$ than beef from beef herds (17–94 kgCO_2-eq (100g protein)$^{-1}$ (Figure 12.6, re-calculated from Poore and Nemecek (2018) using AR6 GWPs based on a 100 year horizon) *(medium evidence, high agreement)*. The wide variation in beef emissions reflects differences in production systems, which range from intensive feedlots with stock raised largely on grains through to rangeland and transhumance production systems. Dairy systems are generally more intensive production systems, with higher digestibility feed than beef systems. Further, emissions from dairy systems are shared between milk and meat, which brings GHG footprints of beef from dairy herds beef closer to those of meat from monogastric animals with emissions intensities of pork (4.4–13 kgCO_2-eq per 100g protein) and poultry meat (2.3–11 kgCO_2-eq per 100g protein) (Poore and Nemecek 2018a).

Emission intensities for farmed fish ranged from 2.4–11 kgCO_2-eq per 100g protein (Poore and Nemecek 2018a). For Norwegian seafood, large differences have been found ranging from 1.1 kgCO_2-eq per kg edible product for herring to more than 8 kgCO_2-eq per kg edible product for salmon shipped by road and ferry from Oslo to Paris (Winther et al. 2020). For capture fish, large differences in emissions have been found, ranging from 0.2–7.9 kgCO_2-eq per kg landed fish (Parker et al. 2018), although an environmental comparison of capture fish to farmed foods should include other indicators such as overfishing. Plant-based foods generally have lower GHG emissions (-2.2–4.5 kgCO_2-eq per 100g protein) than farmed animal based foods (Clune et al. 2017; Hilborn et al. 2018; Clark and Tilman 2017; Nijdam et al. 2012; Poore and Nemecek 2018a) *(robust evidence, high agreement)*. Several plant-based foods are associated with emissions from land use change, for example, palm oil, soy and coffee (Poore and Nemecek 2018a), although emissions intensities are context-specific (Meijaard et al. 2020) and for plant-based proteins, GHG footprints per serving remain lower than those of animal source proteins (Kim et al. 2019).

In traditional production systems, especially in developing countries, livestock serve multiple functions, providing draught power, fertiliser, investment and social status, besides constituting an important source of nutrients (Weiler et al. 2014). In landscapes dominated by forests or cropland, semi-natural pastures grazed by ruminants provide heterogeneity that supports biodiversity (Röös et al. 2016). Grazing on marginal land and the use of crop residues and food waste can provide human-edible food with lower demands for cropland (Röös et al. 2016; van Zanten et al. 2018; Van Hal et al. 2019). Animal protein requires more land than vegetable protein, so switching consumption from animal to vegetable proteins could reduce the pressure on land resources and potentially enable additional mitigation through expansion of natural ecosystems, storing carbon while supporting biodiversity, or reforestation to sequester carbon and enhance wood supply capacity for the production of biobased products substituting fossil fuels, plastics, cement, etc. (Searchinger et al. 2018; Schmidinger and Stehfest 2012; Hayek et al. 2021). At the same time, alternatives to animal-based meat and other livestock products are being developed (Figure 12.6). Their increasing visibility in supermarkets and catering services, as well as falling production prices, could make meat substitutes competitive in one to two decades (Gerhardt et al. 2019). However, uncertainty around their uptake creates uncertainty around their effect on future GHG emissions.

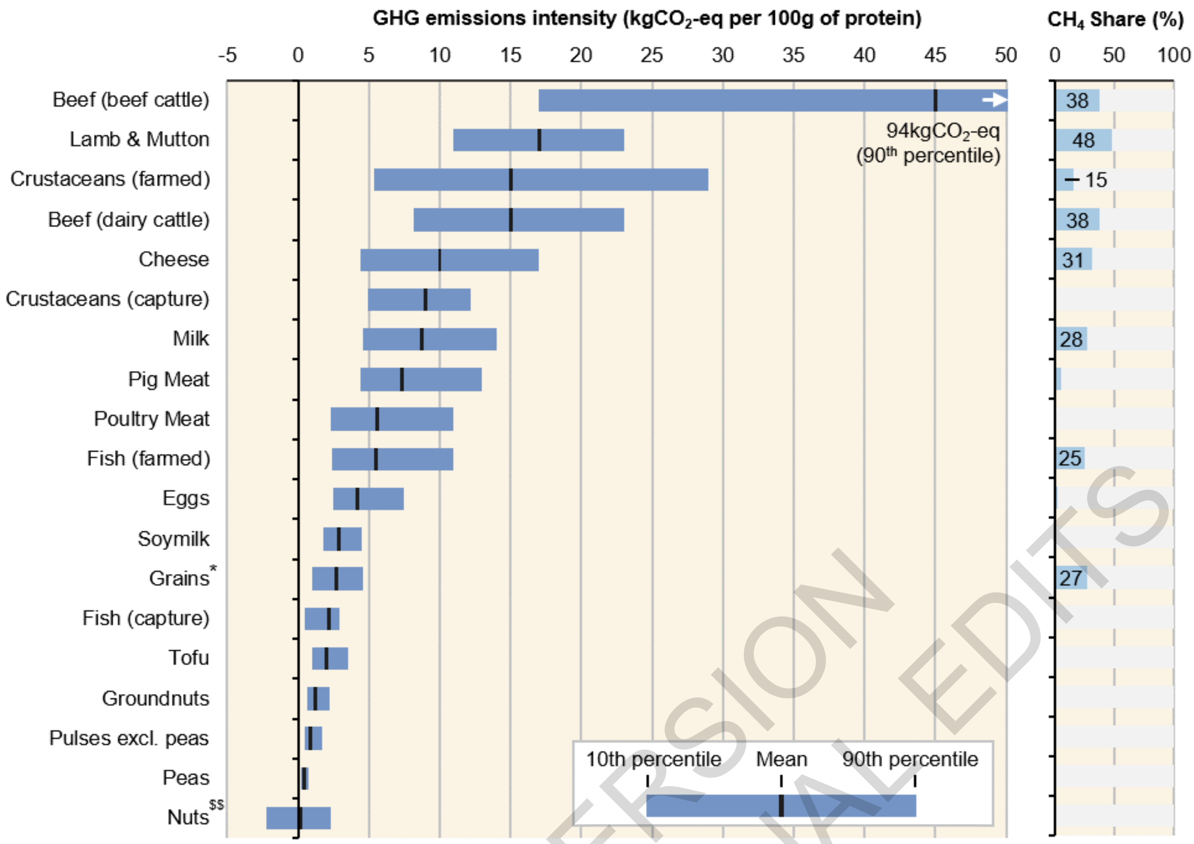


Figure 12.6: Ranges of GHG intensities [kgCO$_2$-eq per 100 g of protein, 10th-90th percentile] in protein-rich foods, quantified via a meta-analysis of attributional Life Cycle Assessment studies using economic allocation

Aggregation of CO$_2$, CH$_4$, and N$_2$O emissions in Poore and Nemecek, (2018) updated to use IPCC-AR6 100-year GWP. Data for capture fish, crustaceans, and cephalopods from Parker et al. (2018), with post-farm data from (Poore and Nemecek 2018a), where the ranges represent differences across species groups. CH$_4$ emissions include emissions from manure management, enteric fermentation, and flooded rice only.

***Grains are not generally classed as protein-rich, but they provide ~41% of global protein intake. Here grains are a weighted average of wheat, maize, oats, and rice by global protein intake (FAO Food Balance Sheets).**

$^{\$\$}$Conversion of annual to perennial crops can lead to carbon sequestration in woody biomass and soil, shown as negative emissions intensity.

Source: Poore and Nemecek, 2018; Parker et al., 2018

12.4.2.3 Territorial national per capita GHG emissions from food systems

Food systems are connected to other societal systems, such as the energy system, financial system, and transport system (Leip et al. 2021). Also, food systems are dynamic and continuously changing and adapting to existing and anticipated future conditions. Food production systems are very diverse and vary by farm size, intensity level, farm specialisation, technological level, production methods (e.g., organic, conventional, etc.), with environmental and social consequences (Herrero et al. 2017; Fanzo 2017; Václavík et al. 2013; Herrero et al. 2021).

Various frameworks have been proposed to assess sustainability of food systems, including metrics and indicators on environmental, health, economic and equity issues, pointing to the importance of recognizing the multi-dimensionality of food system outcomes (Béné et al. 2020; Chaudhary et al. 2018; Gustafson et al. 2016; Eme et al. 2019; Hallström et al. 2018; Hebinck et al. 2021; Zurek et al. 2018). Data platforms are being developed, but so far comprehensive data for evidence-based food system policy are lacking (Fanzo et al. 2020).

To visualise several food systems dimensions in a GHG context, Figure 12.7 shows GHG emissions per capita and year for regional country aggregates (Crippa et al. 2021a,b), indicated by the size of the bubbles. The GHG emissions presented here are based on territorial accounting similar to the UNFCCC GHG inventories: emissions are assigned to the country where they occur, not where food is consumed (Section 12.4.2.1 and Crippa et al., 2021a, b). The colours of the bubbles indicate the relative contribution of one of the following risk factors to deaths, according to the classification used in the Global Burden of Disease Study: Child and maternal malnutrition (red, deficiencies of iron, zinc or Vitamin A, or low birth weight or child growth failure), Dietary risks (yellow, for example diets low in vegetables, legumes, whole grains or diets high in red and processed meat and sugar-sweetened beverages) or High body-mass index (blue). The combined contribution of these three risk factors to total deaths varies strongly and is between 28% and 88% of total deaths. Figure 12.7 shows that dietary risk factors are prevalent throughout all regions. Though not a complete measure of the health impact of food, these were selected as a proxy for nutritional adequacy and balance of diets, avoidance of food insecurity, over- or mal-nutrition and associated non communicable diseases (GBD 2017 Diet Collaborators 2018; GBD 2017 Diet Collaborators et al. 2019).

The data are plotted in a matrix with share of GHG emissions from energy use (Crippa et al. 2021b) on the y-axis and the wholesale cost of food (Springmann et al. 2021) on the x-axis. The share of GHG emissions from energy use is taken as a proxy for the structure of food supply in a region (Section 12.4.1), and the cost for food as a proxy for the structure of the demand side and the access to (healthy) food (Chen et al. 2016; Hirvonen et al. 2019; Finaret and Masters 2019; HLPE 2020; Springmann et al. 2021), though acknowledging the limitations of such a simplification.

While total food system emissions in 2018 range between 0.9 and 8.5 tCO_2-eq cap^{-1} yr^{-1} between regions, the share of energy emissions relative to energy and land-based (agriculture and food system land use change) emissions ranges between 3% and 78%. Regional expenditures for food range from 3.0–8.8 USD cap^{-1} day^{-1} (Figure 12.7), though there is high variability within countries and the costs of nutrient-adequate diets often exceeds those of diets delivering adequate energy (Bai et al. 2020; Hirvonen et al. 2019; FAO et al. 2020). Thus, low-income households in industrialised countries can also be affected by food insecurity (Penne and Goedemé 2020).

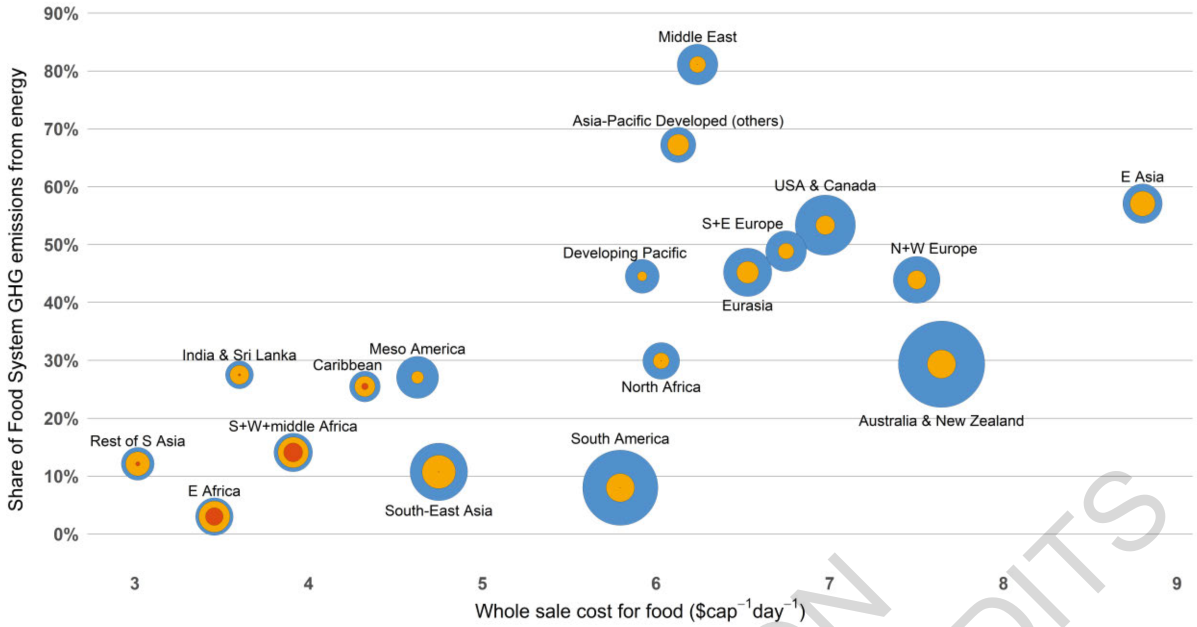


Figure 12.7: Regional differences in health outcome, territorial per capita GHG emissions from national food systems, and share of food system GHG emission from energy use.

GHG emissions are calculated according to the IPCC Tier 1 approach and are assigned to the country where they occur, not necessarily where the food is consumed. Health outcome is expressed as relative contribution of each of the following risk factors to their combined risk for deaths: Child and maternal malnutrition (red), Dietary risks (yellow) or High body-mass index (blue).

Source: cost for food (whole sale price) per capita (Springmann et al. 2021); Territorial food system GHG emissions: EDGAR v.6 (Crippa et al. 2021a), recalculated according to Crippa et al. (2021) using AR6-GWPs; Deaths attributed to dietary factors: (IHME 2018; GBD 2017 Diet Collaborators et al. 2019).

12.4.3 Mitigation opportunities

GHG emissions from food systems can be reduced by targeting direct or indirect GHG emissions in the supply chain including enhanced carbon sequestration, by introducing sustainable production methods such as agro-ecological approaches which can reduce system-level GHG emissions of conventional food production and also enhance resilience (HLPE 2019), substituting food products with high GHG intensities with others of lower GHG intensities, by reducing food over-consumption or by reducing food loss and waste. The substitution of food products with others that are more sustainable and/or healthier is often called ‘dietary shift’.

Clark et al. (2020) showed that even if fossil fuel emissions were eliminated immediately, food system emissions alone would jeopardize the achievement of the 1.5ºC target and threaten the 2ºC target. They concluded that both demand-side and supply-side strategies are needed, including a shift to a diet with lower GHG intensity and rich in plant-based ‘conventional’ foods (e.g., pulses, nuts), or new food products that could support dietary shift. Such dietary shift needs to overcome socio-cultural, knowledge, and economic barriers to significantly achieve GHG mitigation (Section 12.4.5).

Food losses occur at the farm, post-harvest and food processing/wholesale stages of a food supply chain, while in the final retail and consumption stages the term food waste is used (HLPE 2014). Typically, food losses are linked to technical issues such as lack of infrastructure and storage while food waste is often caused by socio-economic and behavioural factors. Mitigation opportunities through reducing food waste and loss exist in all food supply chain stages and are described in the sub-sections below.

Food system mitigation opportunities are divided into five categories as given in Table 12.8:

- Food production from agriculture, aquaculture, and fisheries (Chapter 7.4 and Section 12.4.3.1)
- Controlled environment agriculture (Section 12.4.3.2)
- Emerging food production technologies (Section 12.4.3.3)
- Food processing industries (Section 12.4.3.4)
- Storage and distribution (Section 12.4.3.5)

Food system mitigation opportunities can be either incremental or transformative (Kugelberg et al. 2021). Incremental options are based on mature technologies, for which processes and causalities are understood, and their implementation is generally accepted by society. They do not require a substantial change in the way food is produced, processed, or consumed and might lead to a (slight) shift in production systems or preferences. Transformative mitigation opportunities have wider food system implications and usually coincide with a significant change in food choices. They are based on technologies that are not yet mature and are expected to require further innovation (Klerkx and Rose 2020), and/or mature technologies that might already be part of some food systems but are not yet widely accepted and have transformative potential if applied at large scale, e.g. consumption of insects (Raheem et al. 2019a). Many emerging technologies might be seen as a further step in agronomic development where land-intensive production methods relying on the availability of naturally available nutrients and water are successively replaced with crop variants and cultivation practices reducing these dependencies at the cost of larger energy input (Winiwarter et al. 2014). Others suggest a shift to agro-ecological approaches combining new scientific insights with local knowledge and cultural values (HLPE 2019). Food system transformation can lead to regime shifts or (fast) disruptions (Pereira et al. 2020) if driven by events that are out of control of private or public measures and have a 'crisis' character (e.g., BSE, Skuce et al. 2013).

Table 12.8 summarises the main characteristics of food system mitigation opportunities, their effect on GHG emissions, and associated co-benefits and adverse effects.

Table 12.8: Food system mitigation opportunities

$ Direct and indirect GHG effects: D – Direct emissions except emissions from energy use, E – Energy demand, M – Material demand, FL – food losses, FW – food waste; direction of effect on GHG mitigation: (+) increased mitigation, (0) neutral, (-) decreased mitigation.
& Co-benefits/Adverse effects: H - health aspects, A - Animal welfare, R - resource use, L - Land demand, E – Ecosystem services; (+) co-benefits, (-) adverse effects.

Food system mitigation options (I: incremental; T: transformative)		**Direct and indirect effect on GHG mitigation (+/0/-) $**	**Co-benefits / Adverse effects &**	**Source**
Food from agricultural, aquaculture and fisheries	(I) Dietary shift, in particular increased share of plant-based protein sources	D+ ↓ GHG footprint	A+ Animal welfare L+ Land sparing H+ Good nutritional properties, potentially ↓ risk from zoonotic diseases, pesticides and antibiotics	1-5
	(I/T) Digital agriculture	D+ ↑ logistics	L+ Land sparing R+ ↑ resource use efficiencies	6-7
	(T) Gene technology	D+ ↑ productivity or efficiency	H+ ↑ nutritional quality E0 ↓ use of agrochemicals; ↑ probability of off-target impacts	7-11
	(I) Sustainable intensific. land use optimisation	D+ ↓ GHG footprint E0 Mixed effects	L+ Land sparing R- Might ↑ pollution/biodiversity loss	7, 12
	(I) Agroecology	D+ ↓ GHG/area, positive micro-climatic effects E+ ↓ energy, possibly ↓ transport FL+ Circular approaches	E+ Focus on co-benefits/ecosystem services R+ Circular, ↑ nutrient and water use efficiencies	13-17
Controlled environment agriculture	(T) Soilless agriculture	D+ ↑ productivity, weather independent FL+ Harvest on demand E- Currently ↑ energy demand, but ↓ transport, building spaces can be used for renewable energy	R+ Controlled loops ↑ nutrient and water use efficiency L+ Land sparing H+ Crop breeding can be optimised for taste and/or nutritional quality	18-24

Food system mitigation options (I: incremental; T: transformative)		**Direct and indirect effect on GHG mitigation (+/0/-) $^{\$}$**	**Co-benefits / Adverse effects $^{\&}$**	**Source**
Emerging Food Production technologies	(T) Insects	D0 Good feed conversion efficiency FW+ Can be fed on food waste	H0 Good nutritional qualities but attention to allergies and food safety issues required	25-28
	(I/T) Algae and bivalves	D+ ↓ GHG footprints	A+ Animal welfare L+ Land sparing H+ Good nutritional qualities; risk of heavy metal and pathogen contamination R+ Biofiltration of nutrient-polluted waters	29-32
	(I/T) Plant-based alternatives to animal-based food products	D+ No emissions from animals, ↓ inputs for feed	A+ Animal welfare L+ Land sparing H+ Potentially ↓ risk from zoonotic diseases, pesticides and antibiotics; but ↑ processing demand	31-33
	(T) Cellular agriculture (including cultured meat, microbial protein)	D+ No emissions from animals, high protein conversion efficiency E- ↑ energy need FLW+ ↓ food loss & waste	A+ Animal welfare R+ ↓ emissions of reactive nitrogen or other pollutants H0 Potentially ↓ risk from zoonotic diseases, pesticides and antibiotics; ↑ research on safety aspects needed	3, 24, 34-42
Food processing and packaging	(I) Valorisation of by-products, FLW logistics and management	M+ Substitution of bio-based materials FL+ ↓ of food losses		43-44
	(I) Food conservation	FW+ ↓ of food waste E0 ↑ energy demand but also energy savings possible (e.g., refrigeration, transport)		45-46

Food system mitigation options (I: incremental; T: transformative)		**Direct and indirect effect on GHG mitigation (+/0/-)** $^{\$}$	**Co-benefits / Adverse effects** $^{\&}$	**Source**
	(I) Smart packaging and other technologies	FW+ ↓ of food waste M0 ↑ material demand and ↑ material-efficiency E0 ↑ energy demand; energy savings possible	H+ Possibly ↑ freshness/reduced food safety risks	46-49
	(I) Energy efficiency	E+ ↓ energy		50
Storage and distribution	(I) Improved logistics	D+ ↓ transport emissions FL+ ↓ losses in transport FW- Easier access to food could ↑ food waste		46-47 51-53
	(I) Specific measures to reduce food waste in retail and food catering	FW+↓ of food waste E+ ↓ downstream energy demand M+ ↓ downstream material demand		54-56
	(I) Alternative fuels/transport modes	D+ ↓ emissions from transport		
	(I) Energy efficiency	E+ ↓ energy in refrigeration, lightening, climatisation		57-58
	(I) Replacing refrigerants	D+ ↓ emissions from the cold chain		50, 59-60

[1] (McDermott and Wyatt 2017); [2] (Foyer et al. 2016); [3] (Semba et al. 2021); [4] (Weindl et al. 2020); [5] (Hertzler et al. 2020); [6] (Finger et al. 2019); [7] (Herrero et al. 2020); [8] Steinwand and Ronald (2020); [9] Zhang et al. (2020a); [10] (Ansari et al. 2020); [11] (Eckerstorfer et al. 2021); [12] (Folberth et al. 2020); [13] (HLPE 2019); [14] (Wezel et al. 2009); [15] (Van Zanten et al. 2018); [16] (van Zanten et al. 2019); [17] (van Hal et al. 2019); [18] (Beacham et al. 2019); [19] (Benke and Tomkins 2017); [20] (Gómez and Gennaro Izzo 2018); [21] (Maucieri et al. 2018); [22] (Rufí-Salís et al. 2020); [23] (Shamshiri et al. 2018); [24] (Graamans et al. 2018); [25] (Fasolin et al. 2019); [26] (Garofalo et al. 2019); [27] (Parodi et al. 2018); [28] (Varelas 2019); [29] (Gentry et al. 2020); [30] (Peñalver et al. 2020); [31] (Torres-Tiji et al. 2020); [32] (Willer and Aldridge 2020); [33] (Fresán et al. 2019); [34] (Mejia et al. 2019); [35] (Tuomisto 2019); [36] (Thorrez and Vandenburgh 2019); [37] (Tuomisto and Teixeira de Mattos 2011);

[38] (Mattick et al. 2015); [39] (Mattick 2018); [40] (Souza Filho et al. 2019); [41] Chriki and Hocquette (Chriki and Hocquette 2020); [42] Hadi and Brightwell (Hadi and Brightwell 2021); [43] (Göbel et al. 2015); [44] (Caldeira et al. 2020); [45] (Silva and Sanjuán 2019); [46] (FAO 2019a); [47] (Molina-Besch et al. 2019); [48] (Poyatos-Racionero et al. 2018); [49] (Müller and Schmid 2019); [50] (Niles et al. 2018); [51] (Lindh et al. 2016); [52] (Wohner et al. 2019); [53] (Bajželj et al. 2020); [54]. (Buisman et al. 2019); [55] (Albizzati et al. 2019); [56] (Liu et al. 2016); [57] (Chaomuang et al. 2017); [58] (Lemma et al. 2014); [59] (McLinden et al. 2017); [60] (Gullo et al. 2017).

12.4.3.1 Food from agriculture, aquaculture, and fisheries

Agricultural food production systems range from smallholder subsistence farms to large animal production factories, in open spaces, greenhouses, rural areas or urban settings.

Dietary shift. Studies demonstrate that a shift to diets rich in plant-based foods, particularly pulses, nuts, fruits & vegetables, such as vegetarian, pescatarian or vegan diets, could lead to substantial reduction of greenhouse gas emissions as compared to current dietary patterns in most industrialized countries, while also providing health benefits and reducing mortality from diet-related non-communicable diseases (Ernstoff et al. 2020; Semba et al. 2020; Theurl et al. 2020; Costa Leite et al. 2020; Chen et al. 2019; Jarmul et al. 2020; Willett et al. 2019; Bodirsky et al. 2020; Hamilton et al. 2021; Springmann et al. 2018a).

Pulses such as beans, chickpeas, or lentils, have a protein composition complementary to cereals, providing together all essential amino acids (McDermott and Wyatt 2017; Foyer et al. 2016). Bio-availability of proteins in foods is influenced by several factors, including amino-acid composition, presence of anti-nutritional factors, and preparation method (Hertzler et al. 2020; Weindl et al. 2020; Semba et al. 2021). Soy beans, in particular, have a well-balanced amino acid profile with high bio-availability (Leinonen et al. 2019). Pulses are part of most traditional diets (Semba et al. 2021) and supply up to 10-35% of protein in low-income countries, but consumption decreases with increasing income and they are globally only a minor share of the diet (McDermott and Wyatt 2017). Pulses play a key role in crop rotations, fixing nitrogen and breaking disease cycles, but yields of pulses are relatively low and have seen small yield increases relative to those of cereals (Barbieri et al. 2021; McDermott and Wyatt 2017; Foyer et al. 2016; Semba et al. 2021).

Technological innovations have made food production more efficient since the onset of agriculture (Winiwarter et al. 2014; Herrero et al. 2020). Emerging technologies include digital agriculture (using advanced sensors, big data), gene technology (crop bio-fortification, genome editing, crop innovations), sustainable intensification (automation of processes, improved inputs, precision agriculture) (Herrero et al. 2020), or multi-trophic aquaculture approaches (Sanz-Lazaro and Sanchez-Jerez 2020; Knowler et al. 2020), though literature on aquaculture and fisheries in the context of GHG mitigation is limited.

Such technologies may contribute to a reduction of GHG emission at the food system level enhanced provision of food, better consideration of ecosystem services, or contribute to nutrition sensitive agriculture, for example, by increasing the nutritional quality of staple crops, increasing the palatability of leguminous crops such as lupines, or the agronomic efficiency or resilience of crops with good nutritional characteristics.

For details on agricultural mitigation opportunities refer to Chapter 7.4.

12.4.3.2 Controlled-environment agriculture

Controlled-environment agriculture is mainly based on hydroponic or aquaponic cultivation systems that do not require soil. Aquaponics combine hydroponics with a re-circulating aquaculture compartment for integrated production of plants and fish (Junge et al. 2017; Maucieri et al. 2018), while aeroponics is a further development of hydroponics that replaces water as a growing medium with a mist of nutrient solution (Al-Kodmany 2018). Aquaponics could potentially produce proteins in urban farms, but the technology is not yet mature and its economic and environmental performance is unclear (O'Sullivan et al. 2019; Love et al. 2015).

Controlled-environmental agriculture is often undertaken in urban environments to take advantage of short supply chains (O'Sullivan et al. 2019), and might use abandoned buildings or be integrated in supermarkets, producing for example herbs 'on demand'.

Optimising growing conditions, hydroponic systems achieve higher yields than un-conditioned agriculture (O'Sullivan et al. 2019); and yields can be further enhanced in CO_2-enriched atmospheres (Armanda et al. 2019; Shamshiri et al. 2018). By using existing spaces or modular systems that can be vertically stacked, this technology minimises land demand, however it is energy intensive and requires large financial investments. So far, only a few crops are commercially produced in vertical farms, including lettuce and other leafy greens, herbs and some vegetables due to their short growth period and high value (Benke and Tomkins 2017; Beacham et al. 2019; O'Sullivan et al. 2019; Armanda et al. 2019). Through breeding, other crops could reach commercial feasibility, or crops with improved taste or nutritional characteristics can be grown (O'Sullivan et al. 2019).

In controlled-environment agriculture, photosynthesis is fuelled by artificial light through LEDs or a combination of natural light with LEDs. Control of the wave band and light cycle of the LEDs and micro-climate can be used to optimise photosynthetic activity, yield and crop quality (Gómez and Gennaro Izzo 2018; Shamshiri et al. 2018).

Co-benefits of controlled-environment agriculture include minimising water and nutrient losses as well as agro-chemical use (Farfan et al. 2019; Shamshiri et al. 2018; O'Sullivan et al. 2019; Armanda et al. 2019; Al-Kodmany 2018; Rufí-Salís et al. 2020) *(robust evidence, high agreement)*. Water is recycled in a closed system and additionally some plants generate fresh water by evaporation from grey or black water, and high nutrient use efficiencies are possible. Food production from controlled-environment agriculture is independent of weather conditions and able to satisfy some consumer demand for locally-produced fresh and diverse produce throughout the year (O'Sullivan et al. 2019; Al-Kodmany 2018; Benke and Tomkins 2017).

Controlled-environment agriculture is a very energy intensive technology (mainly for cooling) and its GHG intensity depends therefore crucially on the source of the energy. Options for reducing GHG intensity include reducing energy use through improved lighting and cooling efficiency or by employing low-carbon energy sources, potentially integrated into the building structure (Benke and Tomkins 2017).

Comprehensive studies assessing the GHG balance of controlled-environment agriculture are lacking. The overall GHG emissions from controlled-environment agriculture is therefore uncertain and depends on the balance of reduced GHG emissions from production and distribution and reduced land requirements, versus increased external energy needs.

12.4.3.3 Emerging foods and production technologies

A diverse range of novel food products and production systems are emerging, that are proposed to reduce GHG emissions from food production, mainly by replacing conventional animal-source food with alternative protein sources. Assessments of the potential of dietary changes are given in Chapter 5.3 and Chapter 7.4. Here, we assess the GHG intensities of emerging food production technologies. This includes products such as insects, algae, mussels and products from bio-refineries, some of which have been consumed in certain societies and/or in smaller quantities (Pikaar et al. 2018; Jönsson et al. 2019; Govorushko 2019; Raheem et al. 2019a; Souza Filho et al. 2019). The novel aspect considered here is the scale at which they are proposed to replace conventional food with the aim to reduce both negative health and environmental impact. To fully realize the health benefits, dietary shifts should also encompass a reduction in consumption of added sugars, salt, and saturated fats, and potentially harmful additives (Curtain and Grafenauer 2019; Fardet and Rock 2019; Petersen et al. 2021).

Meat analogues have attracted substantial venture capital, and production costs have dropped considerably in the last decade, with some reaching market maturity (Mouat and Prince 2018; Santo et al. 2020), but there is uncertainty whether they will 'disrupt' the food market or remain niche products. According to Kumar et al. (2017), the demand for plant-based meat analogues is expected to increase as their production is relatively cheap and they satisfy consumer demands with regard to health and environmental concerns as well as ethical and religious requirements. Consumer acceptance is still low for some options, especially insects (Aiking and de Boer 2019) and cultured meat (Siegrist and Hartmann 2020; Chriki and Hocquette 2020).

Insects. Farmed edible insects have a higher feed conversion ratio than other animals farmed for food, and have short reproduction periods with high biomass production rates (Halloran et al. 2016). Insects have good nutritional qualities (Parodi et al. 2018). They are suited as a protein source for both humans and livestock, with high protein contents and favourable fatty acid composition (Raheem et al. 2019b; Fasolin et al. 2019). If used as feed, they can grow on food waste and manure; if used as food, food safety concerns/regulations can restrict the use of manure (Raheem et al. 2019b) or food waste (Varelas 2019) as growing substrates, and the dangers of pathogenic or toxigenic microorganisms and incidences of anti-microbial resistance need to be managed (Garofalo et al. 2019).

Algae and bivalves have a high protein content and a favourable nutrient profile and can play a role in providing sustainable food. Bivalves are high in omega-3 fatty acids and vitamin B12 and therefore well-suited as replacement of conventional meats, and have a lower GHG footprint (Willer and Aldridge 2020; Parodi et al. 2018). Micro- and macro algae are rich in omega-3 and omega-6 fatty acids, anti-oxidants and vitamins (Peñalver et al. 2020; Parodi et al. 2018; Torres-Tiji et al. 2020). Kim et al. (2019) show that diets with modest amounts of low-food chain animals such as forage fish, bivalves, or insects have similar GHG intensities to vegan diets. Algae and bi-valves can be used to filter nutrients from waters, though care is required to avoid accumulation of hazardous substances (Willer and Aldridge 2020; Gentry et al. 2020).

Plant-based meat, milk and egg analogues. Demand for plant-based proteins is increasing and incentivising the development of protein crop varieties with improved agronomic performance and/or nutritional quality (Santo et al. 2020). There is also an emerging market for meat replacements based on plant proteins, such as pulses, cereals, soya, algae and other ingredients mainly used to imitate the taste, texture and nutritional profiles of animal-source food (Boukid 2021; Kumar et al. 2017). Currently, the majority of plant-based meat analogues is based on soy (Semba et al. 2021). While other products still serve a 'niche' market, their share is growing rapidly and some studies project a sizeable share within a decade (Kumar et al. 2017; Jönsson et al. 2019). In particular, plant-based milk alternatives have seen large increases in the market share (Jönsson et al. 2019). A LCA of 56 plant-based meat analogues showed mean GHG intensities (farm to factory) of 0.21–0.23 kgCO_2-eq per 100 g of product or 20 g of protein for all assessed protein sources (Fresán et al. 2019). Higher footprints were found in the meta-review by Santo et al. (2020). Including preparation, Meija et al. (2019) found higher emissions for burgers and sausages as compared to minced products.

Cellular agriculture. The use of fungi, algae and bacteria is an old process (beer, bread, yoghurt) and serves, among others, for the preservation of products. The concept of cellular agriculture (Mattick 2018) covers bio-technological processes that use micro-organisms to produce acellular (fermentation based cellular agriculture) or cellular products. Yeasts, fungi or bacteria can synthesise acellular products such as haem, milk and egg proteins, or protein-rich animal feed, other food ingredients, and pharmaceutical and material products (Rischer et al. 2020; Mendly-Zambo et al. 2021). Cellular products include cell tissues such as muscle cells to grow cultured meat, fish or other cells (Post 2012; Rischer et al. 2020) and products where the micro-organisms will be eaten themselves (Pikaar et al. 2018; Sillman et al. 2019; Schade et al. 2020). Single cell proteins, combined with photovoltaic electricity generation and direct air capture of carbon dioxide are proposed as highly land- and energy-efficient alternatives to plant-based protein (Leger et al. 2021). Some microbial proteins are produced in a 'bioreactor' and use Haber-Bosch nitrogen and vegetable sugars or atmospheric CO_2 as source of N and C (Simsa et al. 2019; Pikaar et al. 2018). Cultured meat is currently in the research stage and some challenges remain, such as the need for animal-based ingredients to ensure fast/effective growth of muscle cells, tissue engineering to create different meat products, the production at scale and at competitive costs, and regulatory barriers (Rubio et al. 2019; Stephens et al. 2018; Post et al. 2020; Post 2012; Tuomisto 2019). Only a few studies to date have quantified the GHG emissions of microbial proteins or cultured meat, suggesting GHG emissions at the level of poultry meat (Tuomisto and Teixeira de Mattos 2011; Mattick et al. 2015; Souza Filho et al. 2019; Tuomisto 2019).

A review of LCA studies on different plant-based, animal source and nine 'future food' protein sources (Parodi et al. 2018) concluded that insects, macro-algae, mussels, myco-proteins and cultured meat show similar GHG intensities per unit of protein (mean values ranging 0.3–3.1 kgCO_2-eq per 100 g of protein), comparable to milk, eggs, and tuna (mean values ranging 1.2–5.4 kgCO_2-eq per 100 g of protein); while *chlorella* and *spirulina* consume more energy per unit of protein and were associated with higher GHG emissions (mean values ranging 11–13 kgCO_2-eq per 100 g of protein). As the main source of GHG emissions from insects and cellular agriculture foods is energy consumption, their GHG intensity improves with increased use of low-carbon energy (Smetana et al. 2015; Pikaar et al. 2018; Parodi et al. 2018).

Future foods offer other benefits such as lower land requirements, controlled systems with reduced losses of water and nutrients, increased resilience, and possibly reduced hazards from pesticide and antibiotics use and zoonotic diseases, although more research is needed including allergenic and other safety aspects, and possibly reduced protein bioavailability (Tzachor et al. 2021; Alexander et al. 2017; Stephens et al. 2018; Parodi et al. 2018; Santo et al. 2020; Fasolin et al. 2019; Chriki and Hocquette 2020; Hadi and Brightwell 2021) *(medium evidence, high agreement).* Research is needed also on the effect of processing (Wickramasinghe et al. 2021), though a randomized crossover trial comparing appetizing plant foods with meat alternatives found several beneficial and no adverse effects from the consumption of the plant-based meats (Crimarco et al. 2020).

12.4.3.4 Food processing and packaging

Food processing includes preparation and preservation of fresh commodities (fruit and vegetables, meat, seafood and dairy products), grain milling, production of baked goods, and manufacture of pre-prepared foods and meals. Food processors range from small local operations to large multi-national food producers, producing food for local to global markets. The importance of food processing and preservation is particularly evident in developing countries lacking cold chains for the preservation and distribution of fresh perishable products such as fresh fish (Adeyeye 2017; Adeyeye and Oyewole 2016).

Mitigation in food processing largely focuses on reducing food waste and fossil energy usage during the processing itself, as well as in the transport, packaging and storage of food products for distribution and sale (Silva and Sanjuán 2019). Reducing food waste provides emissions savings by reducing wastage of primary inputs required for food production. Another mitigation route, contributing to the circular economy (Sections 12.6.1.2 and Cross-Working Group Box 3 in this Chapter), is by valorisation of food processing by-products through recovery of nutrients and/or energy. No global analyses of the emissions savings potential from the processing step in the value chain could be found.

Reduced food waste during food processing can be achieved by seeking alternative processing routes (Atuonwu et al. 2018), improved communication along the food value chain (Göbel et al. 2015), optimisation of food processing facilities, reducing contamination, and limiting damages and spillage (HLPE 2014). Optimisation of food packaging also plays an important role in reducing food waste, in that it can extend product shelf life; protect against damage during transport and handling; prevent spoilage; facilitate easy opening and emptying; and communicate storage and preparation information to consumers (Molina-Besch et al. 2019).

Developments in smart packaging are increasingly contributing to reducing food waste along the food value chain. Strategies for reducing the environmental impact of packaging include using less, and more sustainable, materials and a shift to re-usable packaging (Coelho et al. 2020). Active packaging increases shelf life through regulating the environment inside the packaging, including levels of oxygen, moisture and chemicals released as the food ages (Emanuel and Sandhu 2019). Intelligent packaging communicates information on the freshness of the food through indicator labels (Poyatos-Racionero et al. 2018), and data carriers can store information on conditions such as temperature along the entire food chain (Müller and Schmid 2019).

LCA can be used to evaluate the benefits and trade-offs associated with different processing or packaging types (Silva and Sanjuán 2019). Some options, such as aluminium, steel and glass, require high energy investment in manufacture when produced from primary materials, with significant savings in energy through recycling being possible (Camaratta et al. 2020). However, these materials are inert in landfill. Other packaging options, such as paper and biodegradable packaging, may require a lower energy investment during manufacture, but may require larger land area and can release methane when consigned to anaerobic landfill where there is no methane recovery. Nevertheless, packaging accounts for only 1-12% (typically around 5%) of the GHG emissions in the life cycle of a food system (Wohner et al. 2019; Crippa et al. 2021b), suggesting that its benefits can often outweigh the emissions associated with the packaging itself.

The second component of mitigation in food processing relates to reduction in fossil energy use. Opportunities include energy efficiency in processes (also discussed in Chapter 11.3), the use of heat and electricity from low-carbon energy sources in processing (Chapter 6), through off-grid thermal processing (sun drying, food smoking) and improving logistics efficiencies. Energy intensive processes with energy saving potential include milling and refining (oil seeds, corn, sugar), drying, and food safety practices such as sterilisation and pasteurisation (Niles et al. 2018). Packaging also plays a role: reduced transport energy can be achieved through reducing the mass of goods transported and improving packing densities in transport vehicles (Lindh et al. 2016; Molina-Besch et al. 2019; Wohner et al. 2019). Choice of packaging also influences refrigeration energy requirements during transport and storage.

12.4.3.5 Storage and distribution

Transport mitigation options along the supply chain include improved logistics, the use of alternative fuels and transport modes, and reduced transport distances. Logistics and alternative fuels and transport modes are discussed in Chapter 10. Transport emissions might increase with increasing demand for a diversity of foods as developing countries become more affluent. New technologies that enable food on demand or online food shopping systems might further increase emissions from food transport; however, the consequences are uncertain and might also entail a shift from individual traffic to bulk transport. The impact on food waste is also uncertain as more targeted delivery options could reduce food waste, but easier access to a wider range of food could also foster over-supply and increase food waste. Mitigation opportunities in food transport are inherently linked to decarbonisation of the transport sector (Chapter 10).

Retail and the food service industry are the main factors shaping the external food environment or 'food entry points'; they are the "physical spaces where food is obtained; the built environment that allows consumers to access these spaces" (HLPE 2017). These industries have significant influence on consumers' choices and can play a role in reducing GHG emissions from food systems. Opportunities are available for optimisation of inventories in response to consumer demands through advanced IT systems (Niles et al. 2018), and for discounting foods close to sell-by dates, which can both serve to reduce food spoilage and wastage (Buisman et al. 2019).

As one of the highest contributors to energy demand at this stage in the food value chain, refrigeration has received a strong focus in mitigation. Efficient refrigeration options include advanced refrigeration temperature control systems, and installation of more efficient refrigerators, air curtains and closed display fridges (Chaomuang et al. 2017). Also related to reducing emissions from cooling and refrigeration is the replacement of hydrofluorocarbons which have very high GWPs with lower GWP alternatives (Niles et al. 2018). The use of propane, isobutane, ammonia, hydrofluoroolefins and CO_2 (refrigerant R744) are among those that are being explored, with varying success (McLinden et al. 2017). In recent years, due to restrictions on high GWP-refrigerants, a considerable growth in the market availability of appliances and systems with non-fluorinated refrigerants has been seen (Eckert et al. 2021)

Energy efficiency alternatives generic to buildings more broadly are also relevant here, including efficient lighting, HVAC systems and building management, with ventilation being a particularly high energy user in retail, that warrants attention (Kolokotroni et al. 2015).

In developing countries particularly, better infrastructure for transportation and expansion of processing and manufacturing industries can significantly reduce food losses, particularly of highly perishable food (Niles et al. 2018; FAO 2019a).

12.4.4 Enabling food system transformation

Food system mitigation potentials in AFOLU are assessed in Chapter 7.4, and food system mitigation potentials linked to demand side measures are assessed in Chapter 5. Studies suggest that using supply and demand-side policies are implemented in combination makes ambitious mitigation targets easier to achieve (Latka et al. 2021a; Temme et al. 2020; Global Panel on Agriculture and Food Systems for Nutrition 2020; Clark et al. 2020) *(high agreement; limited evidence).*

The trends in the global and national food systems towards a globalisation of food supply chains and increasing dominance of supermarkets and large corporate food processors (Dries et al. 2004; Neven and Reardon 2004; Baker and Friel 2016; Andam et al. 2018; Popkin and Reardon 2018; Reardon et al. 2019; Pereira et al. 2020) has led to environmental, food insecurity and malnutrition problems. Studies therefore call for a transformation of current global and national food systems to solve these problems (Schösler and Boer 2018; McBey et al. 2019; Kugelberg et al. 2021). This has not yet been successful, including due to insufficient coordination between relevant food system policies (Weber et al. 2020) *(medium evidence, high agreement)*.

Different elements of food systems are currently governed by separate policy areas that in most countries scarcely interact or cooperate (iPES Food 2019; Termeer et al. 2018). This compartmentalisation makes the identification of synergetic and antagonistic effects difficult and faces the possibility of failure due to unintended and unanticipated negative impacts on other policy areas and consequently lack of agreement and social acceptance (Mylona et al. 2018; Brouwer et al. 2020; Mausch et al. 2020; Hebinck et al. 2021) (Section 12.4.5). This could be overcome through cooperation across several policy areas (Sections 12.6.2; 13.7), in particular agriculture, nutrition, health, trade, climate, environment policies, and an inclusive and transparent governance structure (Bhunnoo 2019; Diercks et al. 2019; Herrero et al. 2021; iPES Food 2019; Termeer et al. 2018; Mausch et al. 2020; Kugelberg et al. 2021), making use of potential spill-over effects (Kanter et al. 2020; OECD 2021).

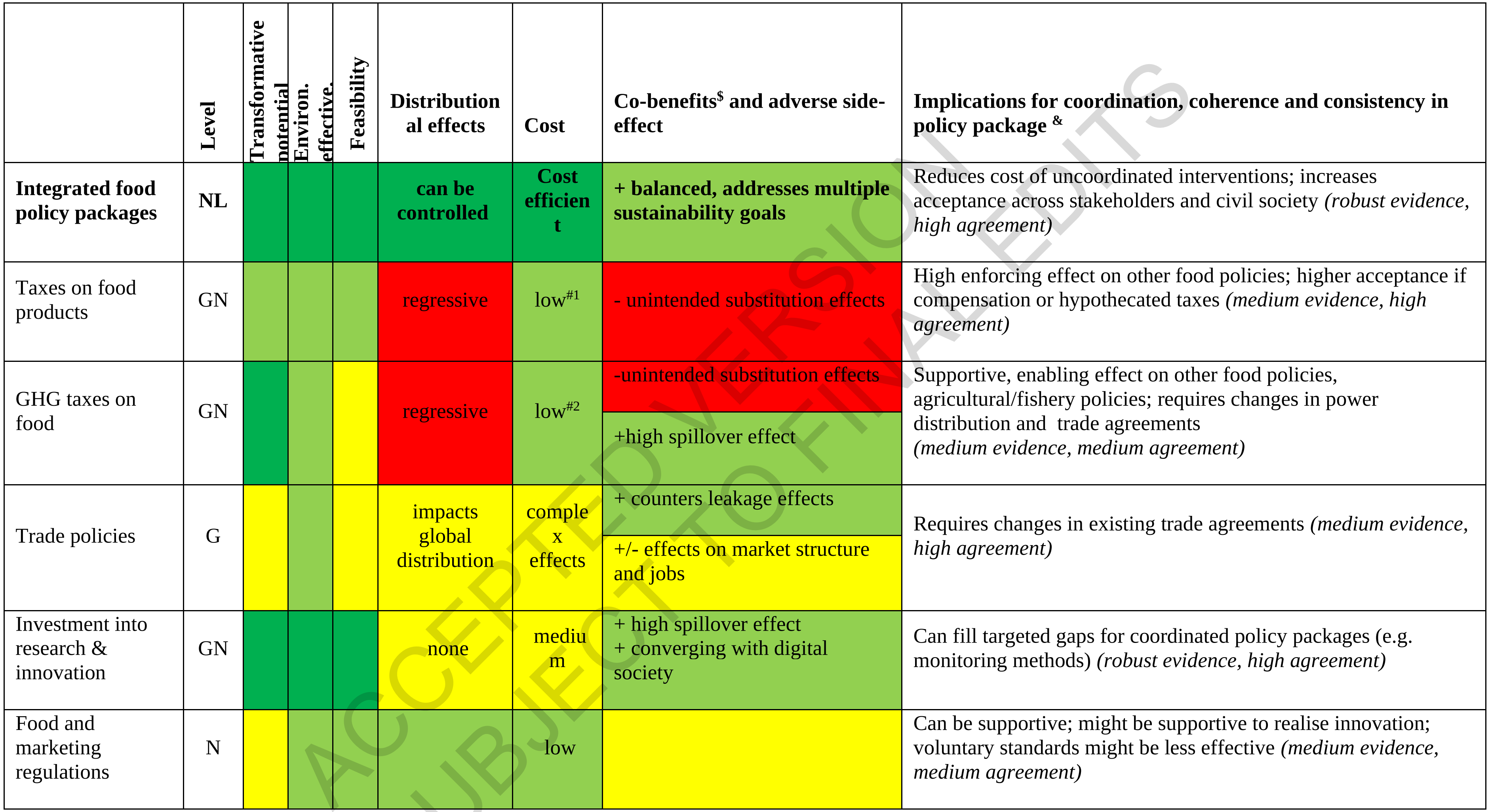
Transformation of food systems may come from technological, social or institutional innovations that start as niches but can potentially lead to rapid changes, including changes in social conventions (Centola et al. 2018; Benton et al. 2019).

Where calories and ruminant animal-source food are consumed in excess of health guidelines, reduction of excess meat (and dairy) consumption is amongst the most effective measures to mitigate GHG emissions, with a high potential for environment, health, food security, biodiversity, and animal welfare co-benefits (Stylianou et al. 2021; Chai et al. 2019; Semba et al. 2020; Willett et al. 2019; Chen et al. 2019; Hamilton et al. 2021; Hedenus et al. 2014; Kim et al. 2019; Theurl et al. 2020; Springmann et al. 2018a) *(robust evidence, high agreement)*. Dietary changes are relevant for several SDGs, apart from SDG 13 (climate action), including SDG 2 (zero hunger), SDG 3 (good health and well-being), SDG 6 (clean water and sanitation), SDG 12 (responsible consumption and production), SDG 14 (life below water) and SDG 15 (life on land) (Bruce M et al. 2018; Vanham et al. 2019; Mbow et al. 2019; Herrero et al. 2021) (Section 12.6.1). However, behavioural change towards diets of lower environmental impact and higher nutritional qualities faces barriers both from agricultural producers and consumers (Apostolidis and McLeay 2016; Aiking and de Boer 2018; de Boer et al. 2018; Milford et al. 2019), and requires policy packages that combine informative instruments with behavioural, administrative and/or market-based instruments, and are attentive to the needs of, and engage, all food system stakeholders including civil society networks, and change the food environment (see Section 12.4.1) (Stoll-Kleemann and Schmidt 2017; Kraak et al. 2017; El Bilali 2019; Cornelsen et al. 2015; iPES Food 2019; Milford et al. 2019; Temme et al. 2020) *(robust evidence, high agreement)*.

Table 12.9 summarizes the implications of a range of policy instruments discussed in more detail in the following sub-sections and highlights the benefits of integrated policy packages. Furthermore, Table 12.9 assesses transformative potential, environmental effectiveness, feasibility, distributional effect, cost, and cost-benefits and trade-offs of individual policy instruments, as well as their potential role as part of coherent policy packages. Table 12.9 shows that information and behavioural policy instruments can have significant but small effects in changing diets *(robust evidence, medium agreement)*, but are mutually enforcing and might be essential to lower barriers and increase acceptance of market-based and administrative instruments *(medium evidence, high agreement)*.

The policy instruments are assessed in relation to shifting food consumption and production towards increased sustainability and health. This includes lowering GHG emissions, although not in all cases is this the primary focus of the instrument, and in some cases lowering GHG emission may not even be explicitly mentioned.

Table 12.9: Assessment of food system policies targeting (post-farm gate) food chain actors and consumers



	Level	Transformative potential	Environ. effective.	Feasibility	Distributional effects	Cost	Co-benefits[$] and adverse side-effect	Implications for coordination, coherence and consistency in policy package [&]
Integrated food policy packages	**NL**	🟩	🟩	🟩	**can be controlled**	**Cost efficient**	**+ balanced, addresses multiple sustainability goals**	Reduces cost of uncoordinated interventions; increases acceptance across stakeholders and civil society *(robust evidence, high agreement)*
Taxes on food products	GN	🟢	🟢	🟢	regressive	low[#1]	- unintended substitution effects	High enforcing effect on other food policies; higher acceptance if compensation or hypothecated taxes *(medium evidence, high agreement)*
GHG taxes on food	GN	🟩	🟢	🟨	regressive	low[#2]	-unintended substitution effects +high spillover effect	Supportive, enabling effect on other food policies, agricultural/fishery policies; requires changes in power distribution and trade agreements *(medium evidence, medium agreement)*
Trade policies	G	🟨	🟢	🟨	impacts global distribution	complex effects	+ counters leakage effects +/- effects on market structure and jobs	Requires changes in existing trade agreements *(medium evidence, high agreement)*
Investment into research & innovation	GN	🟩	🟩	🟩	none	medium	+ high spillover effect + converging with digital society	Can fill targeted gaps for coordinated policy packages (e.g. monitoring methods) *(robust evidence, high agreement)*
Food and marketing regulations	N	🟨	🟢	🟢	🟢	low	🟨	Can be supportive; might be supportive to realise innovation; voluntary standards might be less effective *(medium evidence, medium agreement)*

Organisational level procurement policies	NL	🟩	🟩	🟢	🟢	low	+ can address multiple sustainability goals	Enabling effect on other food policies; reaches large share of population *(medium evidence, high agreement)*
Sustainable food-based dietary guidelines	GNL	🟩	🟩	🟢	none	low	+ can address multiple sustainability goals	Little attention so far on environmental aspects; can serve as benchmark for other policies (labels, food formulation standards, etc.) *(medium evidence, medium agreement).*
Food labels/ information	GNL	🟨	🟩	🟢	education level relevant	low	+ empowers citizens + increases awareness + multiple objectives	Effective mainly as part of a policy package; incorporation of other objectives (e.g. animal welfare, fair trade...); higher effect if mandatory *(medium evidence, medium agreement).*
Nudges	NL	🟩	🟩	🟢	none	low	+ possibly counteracting information deficits in population subgroups	High enabling effect on other food policies, *(medium evidence, high agreement)*

Colour code: **Effect of measures:** negative 🟥, none/unclear 🟨, slightly positive 🟩, positive 🟢; **Level:** G: global/multinational, N: national, L: local; **#1** Minimum level to be effective 20% price increase; **#2** Minimum level to be effective 50-80USD per tCO2eq. **$** In addition, all interventions are assumed to address health and climate change mitigation. **&** Requires coordination between policy areas, participation of stakeholders, transparent methods and indicators to manage trade-offs and prioritisation between possibly conflicting objectives; and suitable indicators for monitoring and evaluation against objectives.

12.4.4.1 Market based instruments

Taxes and subsidies: Food-based taxes have largely been implemented to reduce non-communicable diseases and sugar intake, particularly those targeting sugar-sweetened beverages (WHO 2019). Many health-related organisations recommend the introduction of such taxes to improve the nutritional quality of marketed products and consumers' diets (Park and Yu 2019; Wright et al. 2017; WHO 2019), even though the impacts of food taxes are complex due to cross-price and substitution effects and supplier reactions (Blakely et al. 2020; Gren et al. 2019; Cornelsen et al. 2015) and their regressive effect (WHO 2019). Subsidies and taxes are found to be effective in changing dietary behaviour at levels above 20% price increase (Niebylski et al. 2015; Mozaffarian et al. 2018; Nakhimovsky et al. 2016; Hagenaars et al. 2017; Cornelsen et al. 2015), even though longer term effects are scarcely studied (Cornelsen et al. 2015) and effects of sugar tax with tax rates lower than 20% have been observed for low-income groups (Temme et al. 2020).

Modelling results show only small consumption shifts with moderate meat price increases; and high price increases are required to reach mitigation targets, even though model predictions become highly uncertain due to lack of observational data (Zech and Schneider 2019; Fellmann et al. 2018; Bonnet et al. 2018; Mazzocchi 2017; Latka et al. 2021b). Taxes applied at the consumer level are found to be more effective than levying the taxes at the production side (Springmann et al. 2017).

Unilateral taxes on food with high GHG intensities have been shown to induce increases in net export flows, which could reduce global prices and increase global demand. Indirect effects on GHG mitigation therefore could be reduced by up to 70–90% of national results (Fellmann et al. 2018; Zech and Schneider 2019) *(limited evidence, high agreement)*. The global mitigation potential for GHG taxation of food products at 52 USD $kgCO_2\text{-eq}^{-1}$ has been estimated at 1 $GtCO_2\text{-eq yr}^{-1}$ (Springmann et al. 2017).

Studies have shown that taxes can improve the nutritional quality of diets and reduce GHG emissions from the food system, particularly if accompanied by other policies that increase acceptance and elasticity, and reduce regressive and distributional problems (Niebylski et al. 2015; Hagenaars et al. 2017; Mazzocchi 2017; Springmann et al. 2017; Wright et al. 2017; Henderson et al. 2018; Säll 2018; FAO et al. 2020; Penne and Goedemé 2020) *(robust evidence, high agreement)*.

Trade: Since the middle of the last century, global trade of agricultural products has contributed to boosting productivity and reducing commodity prices, while also incentivising national subsidies for farmers to remain competitive in the global market (Benton et al. 2019). Trade liberalisation has been coined as an essential element of sustainable food systems, and as one element required to achieve sustainable development, that can shift pressure to regions where the resources are less scarce (Traverso and Schiavo 2020; Wood et al. 2018). However, Clapp (2017) argues that the main economic benefit of trade liberalisation flows to large transnational firms. Benton and Bailey (2019) argue that low food prices in the second half of last century contributed to both yield and food waste increases, and to a focus on staple crops to the disadvantage of nutrient dense foods. However, global trade can also contribute to economic benefits such as jobs and income, reduce food insecurity and facilitate access to nutrients (Wood et al. 2018; Hoff et al. 2019; Traverso and Schiavo 2020; Geyik et al. 2021) and has contributed to increased food supply diversity (Kummu et al. 2020). The relevance of trade for food security, and adaptation and mitigation of agricultural production, has also been discussed in Mbow et al. (2019).

Trade policies can be used to protect national food system measures, by requiring front-of-package labels, or to impose border taxes on unhealthy products (Thow and Nisbett 2019). For example, in the frame of the Pacific Obesity Prevention in Communities (OPIC), the Fijian government implemented three measures (out of seven proposed) that eliminated import duties on fruits and vegetables, and imposed 15% import duties on unhealthy oils (Latu et al. 2018). Trade agreements, however, have the potential to undermine national efforts to improve public health (Unar-Munguía et al. 2019). GHG mitigation efforts in food supply chains can be counteracted by GHG leakage, with a general increase of environmental and social impact in developing countries exporting food products, and a decrease in the developed countries importing food products (Wiedmann and Lenzen 2018; Sandström et al. 2018; Fellmann et al. 2018). The demand for agricultural commodities has also been associated with tropical deforestation, though a robust estimate on the extent of embodied deforestation in food commodities is not available (Pendrill et al. 2019).

Investment into research & innovation: El Bilali (2019) assessed research gaps in the food system transition literature and found a need to develop comparative studies that enable the assessment of spatial variability and scalability of food system transitions. The author found also that the role of private industry and corporate business is scarcely researched, although they could play a major role in food system transitions.

The InterAcademy Partnership assessed how research can contribute to providing the required evidence and opportunities for food system transitions, with a focus on climate change impacts and mitigation (IAP 2018). The project builds on four regional assessments of opportunities and challenges on food and nutrition security in Africa (NASAC 2018), the Americas (IANAS 2018), Asia (AASSA 2018), and Europe (EASAC 2017). The Partnership concludes with a set of research questions around food systems, that need to be better understood: (i) how are sustainable food systems constituted in different contexts and at different scales, (ii) how can transition towards sustainable food systems be achieved, and (iii) how can success and failure be measured along sustainability dimensions including climate mitigation?

12.4.4.2 Regulatory and administrative instruments

Marketing regulations: Currently, 16 countries regulate marketing of unhealthy food to children, mainly on television and in schools (Taillie et al. 2019), and many other efforts are ongoing across the globe (European Commission 2019). The aim to counter the increase in obesity in children and target products high in saturated fats, trans-fatty acids, free sugars and/or salt (WHO 2010) was endorsed by 192 countries (Kovic et al. 2018). Nutrition and health claims for products are used by industry to increase sales, for example in the sport sector or for breakfast cereals. They can be informative, but can also be misleading if misused for promoting unhealthy food (Ghosh and Sen 2019; Sussman et al. 2019; Whalen et al. 2018).

Strong statutory marketing regulations can significantly reduce the exposure of children to, and sales of, unhealthy food compared with voluntary restrictions (Kovic et al. 2018; Temme et al. 2020). Data on effectiveness of marketing regulations with a broader food sustainability scope are not available. On the other hand, regulations that mobilise private investment into emerging food production technologies can be instrumental in curbing the cost and making them competitive (Bianchi et al. 2018a).

Voluntary sustainability standards: Voluntary sustainability standards are developed either by a public entity or by private organisations to respond to consumers' demands for social and environmental standards (Fiorini et al. 2019). For example, the Dutch "Green Protein Alliance", an alliance of government, industry, NGOs and academia, formulated a goal to shift the ratio of protein consumption from 60% animal source proteins currently to 40% by 2050 (Aiking and de Boer 2020), and Cool Food Pledge signatories (organisations that serve food, such as restaurants, hospitals and universities) committed to a 25% reduction in GHG emissions by 2030, compared with 2015 (Cool Food 2020). For firms, obtaining certification under such schemes can be costly, and costs are generally borne by the producers and/or supply chain stakeholders (Fiorini et al. 2019). The effectiveness of private voluntary sustainability standards is uncertain. Cazzolla Gatti et al. (2019) have investigated the effectiveness of the Roundtable on Sustainable Palm Oil on halting forest loss and habitat degradation in Southeast Asia and concluded that production of certified palm oil continued to lead to deforestation.

Organisational procurement: Green public procurement is a policy that aims to create additional demand for sustainable products (Bergmann Madsen 2018; Mazzocchi and Marino 2019) or decrease demand for less sustainable products (e.g., the introduction of "Meatless Monday" by the Norwegian Armed Forces (Milford and Kildal 2019; Cheng et al. 2018; Wilts et al. 2019; Gava et al. 2018)). To improve dietary choices, organisations can increase the price of unsustainable options while decreasing the price of sustainable ones, or employ information or choice architecture measures (Goggins and Rau 2016; Goggins 2018). Procurement guidelines exist at global, national, organisational or local levels (Neto and Gama Caldas 2018; Noonan et al. 2013). Procurement rules in schools or public canteens increase the accessibility of healthy food and can improve dietary behaviour and decrease purchases of unhealthy food (Cheng et al. 2018; Temme et al. 2020).

Food regulations: Novel foods based on insects, microbial proteins or cellular agriculture must go through authorisation processes to ensure compliance with food safety standards before they can be sold to consumers. Several countries have 'novel food' regulations governing the approval of foods for human consumption. For example, the European Commission, in its update of the Novel Food Regulation in 2015, expanded its definition of novel food to include food from cell cultures, or that produced from animals by non-traditional breeding techniques (EU 2015).

For animal product analogues, regulatory pathways and procedures (Stephens et al. 2018) and terminology issues (defining equivalence questions) (Carrenõ and Dolle 2018; Pisanello and Ferraris 2018) need clarification, as does their relation to religious rules (Chriki and Hocquette 2020).

Examples of legislation targeting food waste include the French ban on wasting food approaching best-before dates, requiring its donation to charity organisations (Global Alliance for the Future of Food 2020). In Japan, the Food Waste Recycling Law set targets for food waste recycling for industries in the food sector for 2020, ranging between 50% for restaurants and 95% for food manufacturers (Liu et al. 2016).

12.4.4.3 Informative instruments.

Sustainable Food-Based Dietary Guidelines: National food based dietary guidelines (FBDGs) provide science-based recommendations on food group consumption quantities. They are available for 94, mostly upper- and middle-income countries globally (Wijesinha-Bettoni et al. 2021), adapted to national cultural and socio-economic context, and can be used as a benchmark for food formulation standards or public and private food procurement, or to inform citizens (Bechthold et al. 2018; Temme et al. 2020). Most FBDGs are based on health considerations and only a few mention environmental sustainability aspects (Bechthold et al. 2018; Ritchie et al. 2018; Ahmed et al. 2019; Springmann et al. 2020). Implementation of FBDGs so far focuses largely in the education and health sectors, with few countries also using their potential for guiding food system policies in other sectors (Wijesinha-Bettoni et al. 2021).

Despite the fact that 1.5 billion people follow a vegetarian diet from choice or necessity and the position statements of various nutrition societies point out that vegetarian diets are adequate if well planned, few FBDGs give recommendations for vegetarian diets (Costa Leite et al. 2020). An increase in consumption of plant-based food is a recurring recommendation in FBDGs, though an explicit reduction or limit of animal source proteins is not often included, with the exception of red or processed meat (Temme et al. 2020). To account for changing dietary trends, however, FBDGs need to incorporate sustainability aspects (Herforth et al. 2019). A healthy diet respecting planetary boundaries has been proposed by Willett et al. (2019), though some authors have questioned the validity of the nutritional (Zagmutt et al. 2019) or environmental implications, such as water use (Vanham et al. 2020). In October 2019, 14 global cities pledged to adhere to this ‘planetary health diet’ (C40 Cities 2019).

Education on food/nutrition and environment: Some consumers are reluctant to adopt sustainable healthy dietary patterns because of a lack of awareness of the environmental and health consequences of what they eat, but also out of suspicion towards alternatives that are perceived as not ‘natural’ and that seem to be difficult to integrate into their daily dietary habits (Hartmann and Siegrist 2017; Stephens et al. 2018; McBey et al. 2019; Siegrist and Hartmann 2020) or simply lack of knowledge on how to prepare or eat unfamiliar foods (Aiking and de Boer 2020; El Bilali 2019; Temme et al. 2020). Misconceptions may contribute, for example, the belief that packaging or ‘food miles’ dominate the climate impact of food (Macdiarmid et al. 2016). However, spill-over effects can induce sustainable behaviour from ‘entry points’ such as concerns about food waste (El Bilali 2019). Early-life experiences are crucial determinants for adopting healthy and sustainable lifestyles (Bascopé et al. 2019; McBey et al. 2019), so improved understanding of sustainability aspects in the education of public health practitioners and in university education is proposed (Wegener et al. 2018). Investment in education, particularly of women (Vermeulen et al. 2020), might lower the barrier for stronger policies to be accepted and effective (McBey et al. 2019; Temme et al. 2020) *(medium evidence, high agreement).*

Food labels: Instruments to improve transparency and information on food sustainability aspects are based on the assumption of the ’rational’ consumer. Information gives the necessary freedom of choice, but also the responsibility to make the ‘right choice’ (Kersh 2015; Bucher et al. 2016). Studies find a lack of consumer awareness about the link between own food choices and environmental effect (Grebitus et al. 2016; Leach et al. 2016; de Boer et al. 2018; Hartmann and Siegrist 2017) and so effective messaging is required to raise awareness and acceptance of potentially stricter food system policies.

Back-of-package labels usually provide detailed nutritional information (Temple 2019). Front-of-package labels simplify and interpret the information: for example, the traffic light system or the Nutri-Score label used in France (Kanter et al. 2018b) and the health star rating used in Australia and New Zealand (Shahid et al. 2020) provide an aggregate rating based on product attributes such as energy, sugar, saturated fat and fibre content; other labels warn against frequent consumption (e.g., in the 1990s Finland introduced a mandatory warning for products high in salt; the keyhole label was introduced in Sweden in 1989 (Storcksdieck genannt Bonsmann et al. 2020); and ‘high in’ (energy/ saturated fat/ sugar) labels were introduced in Chile in 2016 to reduce obesity (Corvalán et al. 2019)). Front-of-package labels serve also as an incentive to industry to produce healthier or more sustainable products, or can serve as a marketing strategy (Van Loo et al. 2014; Kanter et al. 2018b; Apostolidis and McLeay 2016). Carbon footprint labels can be difficult for consumers to understand (Hyland et al. 2017), and simple, interpretative summary indicators used in front-of-package labels (e.g., traffic lights) are more effective than more complex ones (Tørris and Mobekk 2019; Ikonen et al. 2019; Bauer and Reisch 2019; Temple 2019) (*robust evidence, high agreement)*. Reviews find mixed results but overall a positive effect of food labels in improving direct purchasing decisions (Sarink et al. 2016; Anastasiou et al. 2019; Shangguan et al. 2019; Hieke and Harris 2016; Temple 2019), and in raising level of awareness, thus possibly increasing success of other policy instruments (Al-Khudairy et al. 2019; Samant and Seo 2016; Miller et al. 2019; Temple 2019; Apostolidis and McLeay 2016) *(medium evidence, high agreement)*.

12.4.4.4 Behavioural instruments.

Choice architecture: Information is more effective if accompanied by reinforcement through structural changes or by changing the food environment, such as through product placement in supermarkets, to overcome the intention-behaviour gap (Bucher et al. 2016; Broers et al. 2017; Tørris and Mobekk 2019). Behavioural change strategies have also been shown to improve efficiencies of school food programs (Marcano-Olivier et al. 2020).

Environmental considerations rank behind financial, health, or sensory factors for determining citizens' food choices (Leach et al. 2016; Hartmann and Siegrist 2017; Rose 2018; Neff et al. 2018; Gustafson et al. 2019). There is evidence that choice architecture ("nudging") can be effective in influencing purchase decisions, but regulators do not normally explore this option (Broers et al. 2017). Examples of green nudging include making the sustainable option the default option, enhancing visibility, accessibility of, or exposure to, sustainable products and reducing visibility and accessibility of un-sustainable products, or increasing the salience of healthy sustainable choices through social norms or food labels (Bucher et al. 2016; Wilson et al. 2016; Broers et al. 2017; Al-Khudairy et al. 2019; Bauer and Reisch 2019; Ferrari et al. 2019; Weinrich and Elshiewy 2019; Cialdini and Jacobson 2021). Available evidence suggests that choice architecture measures are relatively inexpensive and easy to implement (Ferrari et al. 2019; Tørris and Mobekk 2019), they are a preferred solution if a restriction of choices is to be avoided (Wilson et al. 2016; Kraak et al. 2017; Vecchio and Cavallo 2019), and can be effective (Arno and Thomas 2016; Bianchi et al. 2018b; Cadario and Chandon 2018; Bucher et al. 2016) if embedded in policy packages (Wilson et al. 2016; Tørris and Mobekk 2019) *(medium evidence, high agreement)*.

Choice architecture measures are also facilitated by growing market shares of animal-free protein sources taken up by discount chains and fast food companies, that enhance visibility of new products and ease integration into daily life for consumers, particularly if sustainable products are similar to the products they substitute (Slade 2018). This effect can be further increased by media and role models (Elgaaied-Gambier et al. 2018).

12.4.5 Food Systems Governance

To support the policies outlined in Section 12.4.4, food system governance depends on the cooperation of actors across traditional sectors in several policy areas, in particular agriculture, nutrition, health, trade, climate, and environment (Termeer et al. 2018; Bhunnoo 2019; Diercks et al. 2019; iPES Food 2019; Rosenzweig et al. 2020b). Top-down integration, mandatory mainstreaming, or boundary-spanning structures like public-private partnerships may be introduced to promote coordination (Termeer et al. 2018). "Flow-centric" rather than territory-centric governance combined with private governance mechanisms has enabled codes of conduct and certification schemes (Eakin et al. 2017), for example the *Roundtable for Sustainable Palm Oil* (RSPO), as well as commodity chain transparency initiatives and platforms like *Trase* (Pirard et al. 2020; Meijaard et al. 2020). Trade agreements are an emerging arena of governance in which improving GHG performance may be an objective, and trade agreements can involve sustainability assessments.

Research on food system governance is mostly non-empirical or case study based, which means that there is limited understanding of which governance arrangements work in specific social and ecological contexts to produce particular food system outcomes (Delaney et al. 2018). Research has identified a number of desirable attributes in food systems governance, including adaptive governance (Termeer et al. 2018), a systems perspective (Whitfield et al. 2018), governance that considers food system resilience (Moragues-Faus et al. 2017; Ericksen 2008; Meyer 2020), transparency, participation of civil society (Duncan 2015; Candel 2014), and cross-scale governance (Moragues-Faus et al. 2017).

Food systems governance has multiple targets and objectives, not least contributing the achievement of the SDGs. GHG emissions from food systems can be impacted by both interventions targeted at different parts of the food system and interventions in other systems, such as reducing deforestation or promoting reforestation (Lee et al. 2019). For example, policies targeting health can contribute to diet shifts away from red meat, while also influencing GHG emissions (Springmann et al. 2018b; Semba et al. 2020); national and local food self-sufficiency policies may also have GHG impacts (Loon et al. 2019; Kriewald et al. 2019). Cross-sectoral governance could enhance synergies between reduced GHG emissions from food systems and other goals; however, integrative paradigms for cross-sectoral governance between food and other sectors have faced implementation challenges (Delaney et al. 2018). For example, in the late 2000s, the water-energy-food nexus emerged as a framework for cross-sectoral governance, but has not been well-integrated into policy (Urbinatti et al. 2020), perhaps because of perceptions that it is an academic concept, or that it takes a technical-administrative view of governance; simply adopting the paradigm is not sufficient to develop effective nexus governance (Cairns and Krzywoszynska 2016; Weitz et al. 2017; Pahl-Wostl et al. 2018). Other policy paradigms and theoretical frameworks that aim to integrate food systems governance include system transition, agroecology, multifunctionality in agriculture (Andrée et al. 2018), climate-smart agriculture (Taylor 2018) and the circular economy (Box 12.2). Cross-sectoral coordination on food systems and climate governance could be aided by internal recognition and ownership by agencies, dedicated budgets for cross-sectoral projects, and consistency in budgets (Pardoe et al. 2018); see also Box 12.1 and Box 12.2.

Food systems governance is still fragmented at national levels, which means that there may be a proliferation of efforts that cannot be scaled and are ineffective (Candel 2014). National policies can be complemented or possibly pioneered by initiatives at the local level (de Boer et al. 2018; Rose 2018). The city-region has been proposed as a useful focus for food system governance (Vermeulen et al. 2020); for example, the Milan Urban Food Policy Pact involves 180 global cities committed to integrative food system strategies (Candel 2019; Moragues-Faus 2021). Local food policy groups and councils that assemble stakeholders from government, civil society, and the private sector have formed trans-local networks of place-based local food policy groups, with over two hundred food policy councils worldwide (Andrée et al. 2018). However, the fluidity and lack of clear agendas and membership structures may hinder their ability to confront fundamental structural issues like unsustainable diets or inequities in food access (Santo and Moragues-Faus 2019).

Early characterisations of food systems governance featured a binary distinction between global and local scales, but this has been replaced by a relational approach where the local governance is seen a process that relies on the interconnections between scales (Lever et al. 2019). Cross-scalar governance is not simply an aggregation of local groups, but involves the telecoupling of distant systems; for example, transnational NGO networks have been able to link coffee retailers in the global North with producers in the global South via international NGOs concerned about deforestation and social justice (Eakin et al. 2017). Global governance institutions like the *Committee on World Food Security* can promote policy coherence globally and reinforce accountability at all levels (McKeon 2015), as can norm-setting efforts like the 'Voluntary Guidelines for the Responsible Governance of Tenure of Land, Fisheries and Forests' (FAO 2012). Global multi-stakeholder processes like the *UN Food Systems Summit* can foster the development of principles for guiding further actions based on sound scientific evidence. The European Commission's *Farm to Fork* strategy aims to promote policy coherence in food policy at EU and national levels, and could be the exemplar of a genuinely integrated food policy (Schebesta and Candel 2020).

START BOX 12.2 HERE

Box 12.2: Case Study: The Finnish Food2030 Strategy

Until 2016, the strategic goals of Finnish food policy were split between different programs and Ministries, resulting in fragmented national oversight of the Finnish food system. To enable policy coordination, a national food strategy was adopted in 2017 called Food2030 (Government of Finland 2017). Food2030 embodies a holistic food system approach and addresses multiple outcomes of the food system, including the competitiveness of the food supply chain and the development of local, organic and climate-friendly food production, as well as responsible and sustainable consumption.

The specific policy mix covers a range of policy instruments to enable changes in agro-food supply, processing and societal norms (Kugelberg et al. 2021). The government provides targeted funding and knowledge support to drive technological innovations on climate solutions to reduce emissions from food and in the agriculture, forestry and land-use sectors. In addition, the Finnish government applies administrative means, such as legislation, advice, guidance on public procurement and support schemes to diversify and increase organic food production to 20% of arable land, which in turn improve the opportunities of small-scale food production and steer public bodies to purchase local and organic food. The Finnish government applies educational and informative instruments to enable a shift to healthy and sustainable dietary behaviours. The policy objective is to reduce consumption of meat and replace it with other sources of protein, aligned with nutrition recommendations and avoiding food waste. The Ministry of Agriculture and Forestry, in collaboration with the Finnish Farmer's unions (MTK) and the Union of Swedish-speaking Farmers and Forest Owners in Finland (SLC), ran a two-year multi-media campaign in 2018 with key messages on sustainability, traceability and safety of the locally-produced food (Ministry of Agriculture and Forestry 2021). A "Food Facts website project" (Luke 2021), funded by the Ministry of Agriculture and Forestry in collaboration with the Natural Resources Institute Finland and the Finnish Food Safety Authority, helps to raise knowledge about food, which could shape responsible individual food behaviour, for example choosing local and sustainable foods and reducing food waste.

A critical enabler for developing a shared food system strategy across sectors and political party boundaries was the implementation of a one-year inclusive, deliberative and consensual stakeholder engagement process. A wide range of stakeholders could exert real influence during the vision-building process, resulting in strong agreement on key policy objectives, and subsequently an important leverage point to policy change (Kugelberg et al. 2021). Moreover, cross-sectoral coordination of Food2030 and the government's wider climate action programs are enabled by a number of institutional mechanisms and collaborative structures, for example the Advisory board for the food chain, formally established during the agenda-setting stage of Food2030, inter-ministerial committees to guide and assess policy implementation, and Our common dining table, a multi-stakeholder partnership that assembles 18 food system actors to engage in reflexive discussions about the Finnish food system.

Critical barriers to strategy and policy formulation include a lack of attention to integrated impact assessments (Kugelberg et al. 2021), which blurs a transparent overview of potential trade-offs and hidden conflicts. There were few policy evaluations from independent organisations to inform policymaking, reducing the opportunities for more progressive policy approaches. Monitoring and food policy evaluation is very close to the ministry in charge, which hampers critical thinking about policy measures (Hildén et al. 2014). In addition, there is a lack of standardised indicators covering the whole food system, which hinders comprehensive oversight of government's progress towards a sustainable food system (Kanter et al. 2018a). Some of the problems related to monitoring, reporting and verification (MRV) are typical for countries in the EU. To improve MRV will probably require structural changes, such as efforts to build up institutional capacity and application of new technology, development of standardised indicators covering the whole food system, regulations on transparency and verification, and mechanisms to enable reflexive discussions between business, farmers, public, NGOs and the government (Meadowcroft and Steurer 2018; Kanter et al. 2020).

END BOX 12.2 HERE

12.5 Land-related impacts, risks and opportunities associated with mitigation options

12.5.1 Introduction

This Section provides a cross-sectoral perspective on land occupation and related impacts, risks and opportunities associated with land-based mitigation options as well as mitigation options that are not designated land-based, yet occupy land. It builds on Chapter 7, that covers mitigation in agriculture, forestry and other land use (AFOLU), including future availability of biomass resources for mitigation in other sectors. It complements Section 12.4, which covers mitigation inherent in the food system, as well as Chapters 6, 9, 10 and 11 that cover mitigation in the energy, transport, building and industry sectors, and Chapters 3 and 4 that cover land and biomass use, primarily in energy applications, in mitigation and development pathways in the near- to mid-term (Chapter 4) and in pathways compatible with long-term goals (Chapter 3).

The deployment of climate change mitigation options often affects land and water conditions, and ecosystem capacity to support biodiversity and a range of ecosystem services (IPCC 2019; IPBES 2019) (r*obust evidence*, *high agreement*). It can increase or decrease terrestrial carbon stocks and sink strength, hence impacting the mitigation effect positively or negatively. As for any other land uses, impacts, risks and opportunities associated with mitigation options that occupy land depend on deployment strategy and on contextual factors that vary geographically and over time (Doelman et al. 2018; Hurlbert et al. 2019; Smith et al. 2019a; Wu et al. 2020) *(robust evidence, high agreement).*

The SR1.5 found that large areas may be utilised for A/R and energy crops in modelled pathways limiting warming to 1.5 C (Rogelj et al. 2018). The SRCCL investigated the implications of land-based mitigation measures for land degradation, food security and climate change adaptation. It focussed on identification of synergies and trade-offs associated with individual land-based mitigation measures (Smith et al. 2019b). In this section we expand beyond the scope of the SRCCL assessment to include also mitigation measures that occupy land while not considered land-based measures, we discuss ways to minimise potential adverse effects, and we consider the potential for synergies through integrating mitigation measures with other land uses, by applying a systems perspective that seeks to meet multiple objectives from multi-functional landscapes. Mitigation measures with zero land occupation, e.g., offshore wind and kelp farming, are not considered,

12.5.2 Land occupation associated with different mitigation options

As reported in Chapter 3, in scenarios limiting warming to 1.5ºC with no or limited overshoot, median area dedicated for energy crops in 2050 is 1.99 (0.56 to 4.82) Mkm^2 and median forest area increased 3.22 (-0.67 to 8.90) Mkm^2 in the period 2019-2050 (5-95 percentile range, scenario category C1). For comparison, the total global areas of forests, cropland and pasture (year 2015) are in the SRCCL estimated at about 40 Mkm^2, 15.6 Mkm^2, and 27.3 Mkm^2, respectively (additionally, 21 Mkm^2 of savannahs and shrublands are also used for grazing) (IPCC 2019). The SRCCL concluded that conversion of land for A/R and bioenergy crops at the scale commonly found in pathways limiting warming to 1.5°C or 2°C is associated with multiple feasibility and sustainability constraints, including land carbon losses *(high confidence)*. Pathways in which warming exceeds 1.5°C require less land-based mitigation, but the impacts of higher temperatures on regional climate and land, including land degradation, desertification, and food insecurity, become more severe (Smith et al. 2019b).

Depending on emission-reduction target, the portfolio of mitigation options chosen, and the policies developed to support their implementation, different land-use pathways can arise with large differences in resulting agricultural and forest area. Some response options can be more effective when applied together (Smith et al. 2019c); for example, dietary change, efficiency increases, and reduced wastage can reduce emissions as well as the pressure on land resources, potentially enabling additional land-based mitigation such as A/R and cultivation of biomass crops for biochar, bioenergy and other bio-based products. The SRCCL (Smith et al. 2019c) report that dietary change combined with reduction in food loss and waste can reduce the land requirement for food production by up to 5.8 Mkm^2 (0.8–2.4 Mkm^2 for dietary change; about 2 Mkm^2 for reduced post-harvest losses, and 1.4 Mkm^2 for reduced food waste (see also Sections 7.4 and 12.4 and Parodi et al. 2018; Springmann et al. 2018; Clark et al. 2020; Rosenzweig et al. 2020b). Stronger mitigation action in the near term targeting non-CO_2 emissions reduction and deployment of other CDR options (DACCS, enhanced weathering, ocean-based approaches, see 12.3) can reduce the land requirement for land-based mitigation (Obersteiner et al. 2018; van Vuuren et al. 2018).

Global Integrated Assessment Models (IAMs) provide insights about the roles of land-based mitigation in pathways limiting warming to 1.5°C or 2°C; interaction between land-based and other mitigation options such as wind and solar power; influence of land-based mitigation on food markets, land use and land carbon; and the role of BECCS vis- à-vis other CDR options (See Chapter 3). However, IAMs do not capture more subtle changes in land management and in the associated industrial/energy systems due to relatively coarse temporal and spatial resolution, and limited representation of land quality and feedstocks/management practices, interactions between biomass production and conversion systems, and local context, e.g., governance of land use (Daioglou et al. 2019; Rose et al. 2020; Welfle et al. 2020; Calvin et al. 2021). A/R have generally been modelled as forests managed for carbon sequestration alone, rather than forestry providing both carbon sequestration and biomass supply (Calvin et al. 2021). Because IAMs do not include options to integrate new biomass production with existing agricultural and forestry systems (Paré et al. 2016; Mansuy et al. 2018; Cossel et al. 2019; Braghiroli and Passarini 2020; Moreira et al. 2020; Djomo et al. 2020; Strapasson et al. 2020; Rinke Dias de Souza et al. 2021), they may over-estimate the total additional land area required for biomass production. On the other hand, some integrated biomass production systems may prove less attractive to landholders than growing biomass crops in large blocks, from logistic, economic, or other points of view (Ssegane et al. 2016; Busch 2017; Ferrarini et al. 2017).

Land occupation associated with mitigation options other than A/R and bioenergy is rarely quantified in global scenarios. Stressing large uncertainties (e.g., type of biomass used and share of solar PV integrated in buildings), (Luderer et al. 2019) modelled land occupation and land transformation associated with a range of alternative power system decarbonisation pathways in the context of a global 2°C climate stabilisation effort. On a per-MWh basis, bioelectricity with CCS was most land-intensive, followed by hydropower, coal with CCS, and concentrated solar power (CSP), which in turn were around five times as land-intensive as wind and solar photovoltaics (PV). A review of studies of power densities (electricity generation per unit land area) confirmed the relatively larger land occupation associated with biopower, although hydropower overlaps with biopower (van Zalk and Behrens 2018). This study also quantifies the low land occupation of nuclear energy, similar to fossil energy sources.

The land occupation of PV depends on the share of ground-mounted vs. buildings-integrated PV, the latter assumed to reach 75% share by 2050 in (Luderer et al. 2019). van de Ven et al. (2021) assumed a 3% share of urbanized land in 2050 available for rooftop PV, referring to (Capellán-Pérez et al. 2017; Dupont et al. 2020) reporting 2-3% availability of urbanized surface area, when considering factors such as roof slopes and shadows between buildings, and threshold relating to energy return on investment. Referring to (De Castro et al. 2013; MacKay 2013; Ong et al. 2013; Smil; Capellán-Pérez et al. 2017) state that land occupation of solar technologies is underestimated in studies assuming ideal conditions, with real occupation being five to ten times higher.

Production of hydrogen and synthetic hydrocarbon fuels via electrolysis and hydrocarbon synthesis is subject to conversion losses that vary depending on technology, system integration and source of carbon (Wulf et al. 2020; Ince et al. 2021)(cross-ref 6.4.4.1 and 6.4.5.1). Indicative electricity-to-hydrocarbon fuel efficiency loss is estimated at about 60% (Ueckerdt et al. 2021). The advantage of smaller land occupation for solar, wind, hydro and nuclear, compared with biomass-based options, is therefore smaller for hydrocarbon fuels than for electricity. Furthermore, biofuels are often co-produced with other bio-based products, which further reduces their land occupation, although comparisons are complicated by inconsistent approaches to allocating land occupation between co-products (Ahlgren et al. 2015; Czyrnek-Delêtre et al. 2017).

Note that comparisons on a per-MWh basis do not reflect the GHG emissions associated with the power options, or that the different options serve different functions in power systems. Reservoir hydropower and biomass-based dispatchable power can complement other balancing options (e.g., battery storage, grid extensions and demand-side management (Göransson and Johnsson 2018; Chapter 6) to provide power stability and quality needed in power systems with large amounts of variable electricity generation from wind and solar power plants. Furthermore, the requirements of transport in grids, pipelines etc. differ. For example, electricity from buildings-integrated PV can be used in the same location as it is generated.

The character of land occupation, and, consequently, the associated impacts (see 12.5.3), vary considerably among mitigation options and also for the same option depending on geographic location, scale, system design and deployment strategy (Olsson et al. 2019; Ioannidis and Koutsoyiannis 2020; van de Ven et al. 2021). Land occupation associated with different mitigation options can be large uniform areas (e.g., large solar farms, reservoir hydropower dams, or tree plantations), or more distributed occupation, such as wind turbines, solar PV, and patches of biomass cultivation integrated with other land uses in heterogeneous landscapes (Cacho et al. 2018; Jager and Kreig 2018; Correa et al. 2019; Englund et al. 2020a). Studies with broader scope, covering total land use requirement induced by plant infrastructure, provide a more complete picture of land footprints. For example, Wu et al. (2021) quantified a land footprint by the infrastructure of a pilot solar plant being three times the onsite land area. Sonter et al. (2020b) found significant overlap of mining areas (82% targeting materials needed for renewable energy production) and biodiversity conservation sites and priorities, suggesting that strategic planning is critical to address mining threats to biodiversity (See section 12.5.4) along with recycling and exploration of alternative technologies that use that use abundant minerals (See Chapter 11, Box Critical Minerals and The Future of Electro-Mobility and Renewables)

There are also situations where expanding mitigation is more or less decoupled from additional land use. The use of organic consumer waste, harvest residues and processing side-streams in the agriculture and forestry sectors can support significant volumes of bio-based products with relatively lower land-use change risks than dedicated biomass production systems (Hanssen et al. 2019; Spinelli et al. 2019; Mouratiadou et al. 2020). Such uses can provide waste management solutions while increasing the mitigation achieved from the land that is already used for agricultural and forest production. Bioenergy accounts for about 90% of renewable heat used in industrial applications, mainly in industries that can use their own biomass waste and residues, such as the pulp and paper industry, food industry, and ethanol production plants (see Chapters 6 and 11) (IEA 2020c). Heat and electricity produced on-site from side-streams but not needed for the industrial processes can be sold to other users, e.g., district heating systems. Surplus waste and residues can also be used to produce solid and liquid biofuels, or be used as feedstock in other industries such as the petrochemical industry (IRENA 2018; Lock and Whittle 2018; Thunman et al. 2018; IRENA 2019; Haus et al. 2020; Chapters 6 and 11). Electrification and improved process efficiencies can reduce GHG emissions and increase the share of harvested biomass that is used for production of bio-based products (Johnsson et al. 2019; Madeddu et al. 2020; Lipiäinen and Vakkilainen 2021; Rahnama Mobarakeh et al. 2021; Silva et al. 2021; Chapter 11). Besides integrating solar thermal panels and solar PV into buildings and other infrastructure, floating solar PV panels in, e.g., hydropower dams (Ranjbaran et al. 2019; Cagle et al. 2020; Haas et al. 2020; Lee et al. 2020; Gonzalez Sanchez et al. 2021), and over canals (Lee et al. 2020; McKuin et al. 2021) could decouple renewable energy generation from land use while simultaneously reducing evaporation losses and potentially mitigating aquatic weed growth and climate change impacts on water body temperature and stratification (Cagle et al. 2020; Exley et al. 2021; Gadzanku et al. 2021; Solomin et al. 2021).

12.5.3 Consequences of land occupation: biophysical and socioeconomic risks, impacts and opportunities

Land occupation associated with mitigation options can present challenges related to impacts and trade-offs, but can also provide opportunities and in different ways support the achievement of additional societal objectives, including adaptation to climate change. This section focuses on mitigation options that have significant risks, impacts and/or co-benefits with respect to land resources, food security and the environment. Bioenergy (with or without CCS), biochar and bio-based products require biomass feedstocks that can be obtained from purpose-grown crops, residues from conventional agriculture and forestry systems, or from biomass wastes, each with different implications for the land. Here we consider separately (i) “biomass-based systems”, including dedicated biomass crops (e.g., perennial grasses, short rotation woody crops) and biomass produced as a co-product of conventional agricultural production (e.g. maize stover), and (ii) “afforestation/reforestation”, including forests established for ecological restoration, plantations grown for forest products and agroforestry, where biomass may also be a co-product. We then discuss impacts and opportunities common to both systems, before considering impacts and opportunities associated with non-land-based mitigation options that nevertheless occupy land.

Biomass-based systems

Mitigation options that are based on the use of biomass, that is, bioenergy/BECCS, biochar, wood buildings, and other bio-based products, can have different positive and negative effects depending on the character of the mitigation option, the land use, the biomass conversion process, how the bio-based products are used and what other product they substitute (Leskinen et al. 2018; Howard et al. 2021; Myllyviita et al. 2021). The impacts of the same mitigation option can therefore vary significantly and the outcome in addition depends on previous land/biomass use (Cowie et al. 2021). As biomass-based systems commonly produce multiple food, material and energy products, it is difficult to disentangle impacts associated with individual bio-based products (Ahlgren et al. 2015; Djomo et al. 2017; Obydenkova et al. 2021). As for other mitigation options, governance has a critical influence on outcome, but larger scale and higher expansion rate generally translates into higher risk for negative outcomes such as competition for scarce land, freshwater and phosphorous resources, displacement of natural ecosystems, and diminishing capacity of agro-ecosystems to support biodiversity and essential ecosystem services, especially if produced without sustainable land management and in inappropriate contexts (Popp et al. 2017; Dooley and Kartha 2018; Hasegawa et al. 2018; Heck et al. 2018; Humpenöder et al. 2018; Fujimori et al. 2019; Hurlbert et al. 2019; IPBES 2019; Smith et al. 2019b; Drews et al. 2020; Hasegawa et al. 2020; Schulze et al. 2020; Stenzel et al. 2021) (*medium evidence, high agreement*).

Removal of crop and forestry residues can cause land degradation through soil erosion and decline in nutrients and soil organic matter (Cherubin et al. 2018) (*robust evidence, high agreement*). These risks can be reduced by retaining a proportion of the residues to protect the soil surface from erosion and moisture loss and maintain or increase soil organic matter (See Section 7.4.3.6); incorporating a perennial groundcover into annual cropping systems (Moore et al. 2019); and by replacing nutrients removed, such as by applying ash from bioenergy combustion plants (Kludze et al. 2013; Harris et al. 2015; Warren Raffa et al. 2015; de Jong et al. 2017) while safeguarding against contamination risks (Pettersson et al. 2020) (*medium evidence, high agreement*). Besides topography, soil, and climate conditions, sustainable residue removal rates also depend on the fate of extracted biomass. For example, to maintain the same level of soil organic carbon, the harvest of straw, if used for combustion (which would return no carbon to fields), was estimated to be only 26% of the rate that could be extracted if used for anaerobic digestion involving return of recalcitrant carbon to fields (Hansen et al. 2020). Similarly, biomass pyrolysis produces biochar which can be returned to soils to counteract C losses associated with biomass extraction (Joseph et al. 2021; Lehmann et al. 2021).

Expansion of biomass crops, especially monocultures of exotic species, can pose risks to natural ecosystems and biodiversity through introduction of invasive species and land use change, also impacting the mitigation value (*robust evidence, high agreement*) ((Liu et al. 2014; El Akkari et al. 2018). Cultivation of conventional oil, sugar, and starch crops tends to have larger negative impact than lignocellulosic crops (Núñez-Regueiro et al. 2020). Social and environmental outcomes can be enhanced through integration of suitable plants (such as perennial grasses and short rotation woody crops) into agricultural landscapes (within crop rotations or through strategic localization, e.g., as contour belts, along fencelines and riparian buffers). Such integrated systems can provide shelter for livestock, retention of nutrients and sediment, erosion control, pollination, pest and disease control, and flood regulation (*robust evidence, high agreement*) (See Figure 12.8 below; Box 12.3 and Cross-Working Group Box 3) (Berndes et al. 2008; Christen and Dalgaard 2013; Asbjornsen et al. 2014; Holland et al. 2015; Ssegane et al. 2015; Dauber and Miyake 2016; Milner et al. 2016; Ssegane and Negri 2016; Styles et al. 2016; Crews et al. 2018; Zalesny et al. 2019; Englund et al. 2020b, 2021). (Zheng et al. 2016; Osorio et al. 2019). (Ferrarini et al. 2017; Henry et al. 2018a). Many of the land use practices described above align with agroecology principles [cross ref WGII CCB Nature-Based Solutions, WGII 5.14 box 5.11] and can simultaneously contribute to climate change mitigation, climate change adaptation and reduced risk of land degradation (IPCC 2019) (*robust evidence, high agreement*).

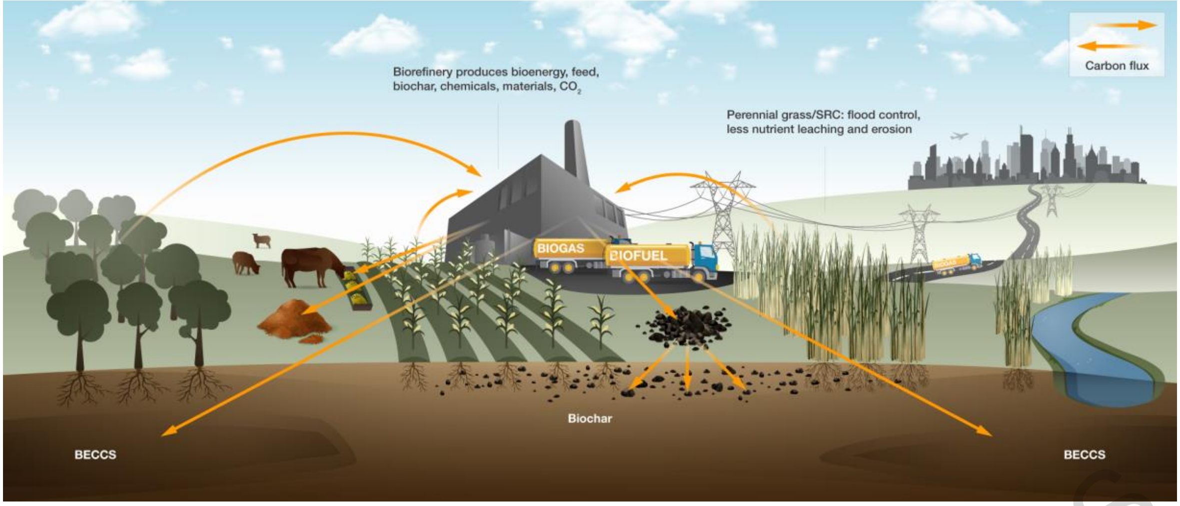


Figure 12.8 Overview of opportunities related to selected land-based climate change mitigation options

Afforestation/Reforestation (A/R)

When A/R activities comprise the establishment of natural forests, the risk to land is primarily associated with potential displacement of previous land use to new locations, which could indirectly cause land use change including deforestation (see Sections 7.4.2 and 7.6.2.4). A/R (including agroforestry) aimed at providing timber, fibre, biomass, non-timber resources and other ecosystem services can provide renewable resources to society and long-term livelihoods for communities. Forest management and harvesting regimes around the world will adjust in different ways as society seeks to meet climate goals. The outcome depends on forest type, climate, forest ownership and the character and product portfolio of the associated forest industry (Lauri et al. 2019; Favero et al. 2020). How forest carbon stocks, biodiversity, hydrology, etc. are affected by changes in forest management and harvesting in turn depends on both management practices and the characteristics of the forest ecosystems (Eales et al. 2018; Griscom et al. 2018; Kondo et al. 2018; Nieminen et al. 2018; Thom et al. 2018; Runting et al. 2019; Tharammal et al. 2019) (*robust evidence*, *medium agreement*). As described above, the GHG savings achieved from producing and using bio-based products will in addition depend on the character of existing societal systems, including technical infrastructure and markets, as this determines the product substitution patterns.

Environmental and socio-economic co-benefits are enhanced when ecological restoration principles are applied (Gann et al. 2019) along with effective planning at landscape level and strong governance (Morgan et al., 2020). For example, restoration of natural vegetation and establishing plantations on degraded land enable organic matter to accumulate in the soil and have potential to deliver significant co-benefits for biodiversity, land resource condition and livelihoods (See Box 12.3 and Cross-Working Group Box 3). Tree planting and agroforestry on cleared land can deliver biodiversity benefits (Seddon et al. 2009; Kavanagh and Stanton 2012; Law et al. 2014), with biodiversity outcomes influenced by block size, configuration and species mix (Cunningham et al. 2015; Paul et al. 2016)(*robust evidence, high agreement)*.

Risks and opportunities common to biomass production and A/R mitigation options

Biomass-based systems and A/R can contribute to addressing land degradation through land rehabilitation or restoration (Box 12.3). Land-based mitigation options that produce biomass for bioenergy/BECCS or biochar through land *rehabilitation* rather than land *restoration* imply a trade-off between production / carbon sequestration and biodiversity outcomes (Hua et al. 2016; Cowie et al.

2018). Restoration, seeking to establish native vegetation with the aim to maximise ecosystem integrity, landscape connectivity, and conservation of on-ground C stock, will have higher biodiversity benefits than rehabilitation measures (Lin et al. 2013). However, sequestration rate declines as forests mature, and the sequestered C is vulnerable to loss through disturbance such as wildfire, so there is a higher risk of reversal of the mitigation benefit compared with use of biomass for substitution of fossil fuels and GHG-intensive building materials (Russell and Kumar 2017; Dugan et al. 2018; Anderegg et al. 2020). Trade-offs between different ecosystem services, and between societal objectives including climate change mitigation and adaptation, can be managed through integrated landscape approaches that aim to create a mosaic of land uses, including conservation, agriculture, forestry and settlements (Freeman et al. 2015; Nielsen 2016; Reed et al. 2016; Sayer et al. 2017) where each is sited with consideration of land potential and socioeconomic objectives and context (Cowie et al. 2018) *(limited evidence, high agreement).*

Impacts of biomass production and A/R on the hydrological cycle and water availability and quality, depend on scale, location, previous land use/cover and type of biomass production system. For example, extraction of logging residues in forests managed for timber production has little effect on hydrological flows, while land use change to establish dedicated biomass production can have a significant effect (Teter et al. 2018; Drews et al. 2020). Deployment of A/R can affect temperature, albedo and precipitation locally and regionally, and can mitigate or enhance the effects of climate change in the affected areas (Stenzel et al. 2021b) and Section 7.2.4). A/R activities can increase evapotranspiration impacting groundwater and downstream water availability, but can also result in increased infiltration to groundwater and improved water quality (Farley et al. 2005; Zhang et al. 2016, 2017; Lu et al. 2018) and can be beneficial where historical clearing has caused soil salinisation and stream salinity (Farrington and Salama 1996; Marcar 2016). There is *limited evidence* that very large-scale land use or vegetation cover changes can alter regional climate and precipitation patterns, e.g., downwind precipitation depends on upwind evapotranspiration from forests and other vegetation (Keys et al. 2016; Ellison et al. 2017; van der Ent and Tuinenburg 2017).

Another example of beneficial effects includes perennial grasses and woody crops planted to intercept runoff and subsurface lateral flow, reducing nitrate entering groundwater and surface waterbodies (e.g Woodbury et al. 2018; Femeena et al. 2018; Griffiths et al. 2019). In India, (Garg et al. 2011) found desirable effects as a result of planting Jatropha on wastelands previously used for grazing (which could continue in the Jatropha plantations): soil evaporation was reduced, as a larger share of the rainfall was channelled to plant transpiration and groundwater recharge, and less runoff resulted in reduced soil erosion and improved downstream water conditions. Thus, adverse effects can be reduced and synergies achieved when plantings are sited carefully, with consideration of potential hydrological impacts (Davis et al. 2013).

Several biomass conversion technologies can generate co-benefits for land and water. Anaerobic digestion of organic wastes (e.g., food waste, manure) produces a nutrient-rich digestate and biogas that can be utilised for heating and cooking or upgraded for use in electricity generation, industrial processes, or as transportation fuel (See Chapter 6) (Parsaee et al. 2019; Hamelin et al. 2021). The digestate is a rich source of nitrogen, phosphorus and other plant nutrients, and its application to farmland returns exported nutrients as well as carbon (Cowie 2020b). Studies have identified potential risks, including Mn toxicity, Cu and Zn contamination, and ammonia emission, compared with application of undigested animal manure (Nkoa 2014). Although the anaerobic digestion process reduces pathogen risk compared with undigested manure feedstocks, it does not destroy all pathogens (Nag et al. 2019). Leakage of methane is a significant risk that needs to be managed, to ensure mitigation potential is achieved (Bruun et al. 2014). Anaerobic digestion of wastewater, such as sugarcane vinasse, reduces methane emissions and pollution loading as well as producing biogas (Parsaee et al. 2019).

Biorefineries can convert biomass to food, feed and biomaterials along with bioenergy (Aristizábal-Marulanda and Cardona Alzate 2019; Schmidt et al. 2019). Biorefinery plants are commonly characterised by high process integration to achieve high resource use efficiency, minimise waste production and energy requirements, and maintain flexibility towards changing markets for raw materials and products (Schmidt et al. 2019). Emerging technologies can convert biomass that is indigestible for monogastric animals or humans (e.g., algae, grass, clover or alfalfa) into food and feed products. For example, Lactic acid bacteria can facilitate the use of green plant biomass such as grasses and clover to produce a protein concentrate suitable for animal feed and other products for material or energy use (Lübeck and Lübeck 2019). Selection of crops suitable for co-production of protein feed along with biofuels and other bio-based products can significantly reduce the land conversion pressure by reducing the need to cultivate other crops (e.g., soybean) for animal feeding (Bentsen and Møller 2017; Solati et al. 2018). Thus, such solutions, using alternatives to high-input, high-emission grain-based feed, can enable sustainable intensification of agricultural systems with reduced environmental impacts (Jørgensen and Lærke 2016). The use of seaweed and algae as biorefinery feedstock can facilitate recirculation of nutrients from waters to agricultural land, thus reducing eutrophication while substituting purpose-grown feed (Thomas et al. 2021).

Pyrolysis can convert organic wastes, including agricultural and forestry residues, food waste, manure, poultry litter and sewage sludge, into combustible gas and biochar, which can be used as a soil amendment (Joseph et al. 2021; Schmidt et al. 2021; Chapter 7). Pyrolysis facilitates nutrient recovery from biomass residues, enabling return to farmland as biochar, noting, however, that a large fraction of nitrogen is lost during pyrolysis (Joseph et al. 2021). Conversion to biochar aids the logistics of transport and land application of materials such as sewage sludge, by reducing mass and volume, improving flow properties, stability and uniformity, and decreasing odour. Pyrolysis is well-suited for materials that may be contaminated with pathogens, microplastics, per- and polyfluoroalkyl substances, such as abattoir and sewage wastes, removing these risks, and reduces availability of heavy metals in feedstock (Joseph et al. 2021). Applying biochar to soil sequesters biochar-carbon for hundreds to thousands of years and can further increase soil carbon by reducing mineralisation of soil organic matter and newly added plant carbon (Singh et al. 2012; Wang et al. 2016a; Weng et al. 2017; Lehmann et al. 2021). Biochars can improve a range of soil properties, but effects vary depending on biochar properties, which are determined by feedstock and production conditions (Singh et al. 2012; Wang et al. 2016a), and on the soil properties where biochar is applied (e.g. Razzaghi et al. 2020). Biochars can increase nutrient availability, reduce leaching losses (Singh et al. 2010; Haider et al. 2017) and enhance crop yields particularly in infertile acidic soils (Jeffery et al. 2017), thus supporting food security under changing climate. Biochars can enhance infiltration and soil water-holding capacity, reducing runoff and leaching, increasing water retention in the landscape and improving drought tolerance and resilience to climate change (Quin et al. 2014; Omondi et al. 2016). (See Chapter 7 for review of biochar's potential contribution to climate change mitigation).

Both A/R and dedicated biomass production could have adverse impacts on food security and cause indirect land use change if deployed in locations used for food production (IPCC 2019). But the degree of impact associated with a certain mitigation option also depends on how deployment takes place and also the rate and total scale of deployment. The highest increases in food insecurity due to deployment of land-based mitigation are expected to occur in Sub-Saharan Africa and Asia (Hasegawa et al. 2018). The land area that could be used for bioenergy or other land-based mitigation options with low to moderate risks to food security depends on patterns of socioeconomic development, reaching limits between 1 and 4 million km^2 (IPCC 2019; Hurlbert et al. 2019; Smith et al. 2019c).

The use of less-productive, degraded/marginal lands has received attention as an option for biomass production and other land-based mitigation that can improve the productive and adaptive capacity of the lands (Liu et al. 2017; Qin et al. 2018; Dias et al. 2021; Kreig et al. 2021) (Section 7.4.4 and Cross-Working Group Box 3). The potential is however uncertain as biomass growth rates may be low, a variety of assessment approaches have been used, and the identification of degraded/marginal land as "available" has been contested, as much low productivity land is used informally by impoverished communities, particularly for grazing, or may be economically infeasible or environmentally undesirable for development of energy crops (Baka 2013, 2014; Haberl et al. 2013; Fritz et al. 2013) (*medium evidence, low agreement*).

As many of the SDGs are closely linked to land use, the identification and promotion of mitigation options that rely on land uses described above can support a growing use of bio-based products while advancing several SDGs, e.g., SDG2 "Zero hunger", SDG6 "Clean water and sanitation", SDG7 "Affordable and Clean Energy" and SDG15 "Life on Land" (Fritsche et al. 2017; IRP 2019; Blair et al. 2021). Policies supporting the target of Land Degradation Neutrality (LDN; SDG 15.3) encourage planning of measures to counteract loss of productive land due to unsustainable agricultural practices and land conversion, through sustainable land management, and strategic restoration and rehabilitation of degraded land (Cowie et al. 2018). LDN can thus be an incentive for land-based mitigation measures that build carbon in vegetation and soil, and can provide impetus for land use planning to achieve multifunctional landscapes that integrate land-based mitigation with other land uses (see Box 12.3). The application of sustainable land management practices that build soil carbon will enhance productivity and resilience of crop and forestry systems, thereby enhancing biomass production (Henry et al. 2018a). Non-bio-based mitigation options can enhance land-based mitigation: enhanced weathering, that is, adding ground silicate rock to soil to take up atmospheric CO_2 through chemical weathering (Section 12.3), could supply nutrients and alleviate soil acidity, thereby boosting productivity of biomass crops and A/R, particularly when combined with biochar application (Haque et al. 2019; De Oliveira Garcia et al. 2020; Buss et al. 2021) Land rehabilitation and enhanced landscape diversity through production of biomass crops could simultaneously contribute to climate change mitigation, climate change adaptation, addressing land degradation, increasing biodiversity and improving food security in the longer term (Mackey et al. 2020; Chapter 7).

Wind power

The land requirement and impacts (including visual and noise impacts) of on-shore wind turbines depend on the size and type of installation, and location (Ioannidis and Koutsoyiannis 2020). Wind power and agriculture can coexist in beneficial ways and wind power production on agriculture land is well established (Fritsche et al. 2017; Miller and Keith 2018a). Spatial planning and local stakeholder engagement can reduce opposition due to visual landscape impacts and noise (Frolova et al. 2019; Hevia-Koch and Ladenburg 2019). Repowering, i.e., replacing with higher capacity wind turbines, can mitigate additional land requirement associated with deployment towards higher share of wind in power systems (Pryor et al. 2020).

Mortality and disturbance risks to birds, bats and insects are major ecological concerns associated with wind farms (Thaxter et al. 2017; Cook et al. 2018; Heuck et al. 2019; Coppes et al. 2020; Choi et al. 2020; Fernández-Bellon 2020; Marques et al. 2020; Voigt 2021). Careful siting is critical (May et al. 2021), while painting blades to increase the visibility can also reduce mortality due to collision (May et al. 2020). Theoretical studies have suggested that wind turbines could lead to warmer night temperatures due to atmospheric mixing (Keith et al. 2004), later confirmed through observation (Zhou et al. 2013), although Vautard et al. (2014) found limited impact at scales consistent with climate policies. More recent studies report mixed results; indications that the warming effect could be substantial with widespread deployment Miller and Keith 2018b and conversely limited impacts on regional climate at 20% of US electricity from wind. (Pryor et al. 2020).

Solar power

As for wind power, land impacts of solar power depend on the location, size and type of installation (Ioannidis and Koutsoyiannis 2020). Establishment of large-scale solar farms could have positive or negative environmental effects at the site of deployment, depending on the location. Solar PV and CSP power installations can lock away land areas, displacing other uses (Mohan 2017). Solar PV can be deployed in ways that enhance agriculture: for example, Hassanpour Adeh et al. (2018) found that biomass production and water use efficiency of pasture increased under elevated solar panels. PV systems under development may achieve significant power generation without diminishing agricultural output (Miskin et al. 2019). Global mapping of solar panel efficiency showed that croplands, grasslands and wetlands are located in regions with the greatest solar PV potential (Adeh et al. 2019). Dual-use agrivoltaic systems are being developed that overcome previously recognised negative impact on crop growth, mainly due to shadows (Armstrong et al. 2016; Marrou et al. 2013b,a), thus facilitating synergistic co-location of solar photovoltaic power and cropping (Miskin et al. 2019; Adeh et al. 2019). Assessment of the potential for optimising deployment of solar PV and energy crops on abandoned cropland areas produced an estimate of the technical potential for optimal combination at 125 EJ per year (Leirpoll et al. 2021).

Deserts can be well-suited for solar PV and CSP farms, especially at low latitudes where global horizontal irradiance is high, as there is lower competition for land and land carbon loss is minimal, although remote locations may pose challenges for power distribution (Xu et al. 2016). Solar arrays can reduce the albedo, particularly in desert landscapes, which can lead to local temperature increases and regional impacts on wind patterns (Millstein and Menon 2011). Modelling studies suggest that large-scale wind and solar farms, for example in the Sahara (Li et al. 2018), could increase rainfall through reduced albedo and increased surface roughness, stimulating vegetation growth and further increasing regional rainfall (Li et al. 2018) (*limited evidence*). Besides impacts at the site of deployment, wind and solar power affect land through mining of critical minerals required by these technologies (Viebahn et al. 2015; McLellan et al. 2016; Carrara et al. 2020).

Nuclear power

Nuclear power has land impacts and risks associated with mining operations (Falck 2015; Winde et al. 2017; Srivastava et al. 2020) and disposal of spent fuel (IAEA 2006a; Ewing et al. 2016; Bruno et al. 2020), but the land occupation is small compared to many other mitigation options. Substantial volumes of water are required for cooling (Liao et al. 2016), as for all thermal power plants, but most of this water is returned to rivers and other water bodies after use (Sesma Martín and Rubio-Varas 2017). Negative impacts on aquatic systems can occur due to chemical and thermal pollution loading (Fricko et al. 2016; Raptis et al. 2016; Bonansea et al. 2020). The major risk to land from nuclear power is that a nuclear accident leads to radioactive contamination. An extreme example, the 1986 Chernobyl accident in Ukraine, resulted in radioactive contamination across Europe. Most of the fallout concentrated near Belarus, Ukraine and Russia, where some 125,000 km^2 of land (more than a third of which was in agricultural use) was contaminated. About 350,000 people were relocated away from these areas (Sovacool 2008; IAEA 2006b). About 116,000 people were permanently evacuated from the 4,200 km Chernobyl exclusion zone (IAEA 2006a). New reactor designs with passive and enhanced safety systems reduce the risk of such accidents significantly (Section 6.4.2.4). An example of alternatives to land reclamation for productive purposes, a national biosphere reserve has been established around Chernobyl to conserve, enhance and manage carbon stocks and biodiversity (Deryabina et al. 2015; Ewing et al. 2016), although invertebrate and plant populations area affected (Mousseau and Møller 2014, 2020).

Hydropower

Reservoir hydropower projects submerge areas as dams are established for water storage. Hydropower can be associated with significant and highly varying land occupation and carbon footprint (Poff and Schmidt 2016; Scherer and Pfister 2016a; Ocko and Hamburg 2019; dos Santos et al. 2017). The flooding of land causes CH_4 emissions due to the anaerobic decomposition of submerged vegetation and there is also a loss of C sequestration due to mortality of submerged vegetation. The size of GHG emissions depends on the amount of vegetation submerged. The carbon in accumulated sediments in reservoirs may be released to the atmosphere as CO_2 and CH_4 upon decommissioning of dams, and while uncertain, estimates indicate that these emissions can make up a significant part of the cumulative GHG emissions of hydroelectric power plants (Almeida et al. 2019; Moran et al. 2018; Ocko and Hamburg 2019). Positive radiative forcing due to lower albedo of hydropower reservoirs compared to surrounding landscapes can reduce mitigation contribution significantly (Wohlfahrt et al. 2021).

Hydropower can have high water usage due to evaporation from dams (Scherer and Pfister 2016b). Hydropower projects may impact aquatic ecology and biodiversity, necessitate the relocation of local communities living within or near the reservoir or construction sites and affect downstream communities (in positive or negative ways) (Barbarossa et al. 2020; Moran et al. 2018). Displacement as well as resettlement schemes can have both socio-economic and environmental consequences including those associated with establishment of new agricultural land (Nguyen et al. 2017; Ahsan and Ahmad 2016). Dam construction may also stimulate migration into the affected region, which can lead to deforestation and other negative impacts (Chen et al. 2015). Impacts can be mitigated through basin-scale dam planning that considers GHG emissions along with social and ecological effects (Almeida et al. 2019). Land occupation is minimal for run-of-river hydropower installations, but without storage they have no resilience to drought and installations inhibit dispersal and migration of organisms (Lange et al. 2018). Reservoir hydropower schemes can regulate water flows and reduce flood damage to agricultural production (Amjath-Babu et al. 2019). On the other hand, severe flooding due to failure of hydropower dams has caused fatalities, damage to infrastructure and loss of productive land (Lu et al. 2018; Zhang et al. 2016, 2017; Farley et al. 2005)(Farrington and Salama 1996; Marcar 2016; Kalinina et al. 2018).

12.5.4 Governance of land-related impacts of mitigation options

The land sector (Chapter 7) contributes to mitigation via emissions reduction and enhancement of land carbon sinks, and by providing biomass for mitigation in other sectors. Key challenges for governance of land-based mitigation include social and environmental safeguards (Larson et al. 2018; Sills et al. 2017; Duchelle et al. 2017); insufficient financing (Turnhout et al. 2017); capturing co-benefits; ensuring additionality, addressing non-permanence of carbon sequestration; monitoring, reporting, and verification (MRV) of emissions reduction and carbon dioxide removals; and avoiding leakage or spill-over effects. Governance approaches to addressing these challenges are discussed in section 7.6, and include MRV systems and integrity criteria for project-level emissions trading; payments for ecosystem services; land use planning and land zoning; certification schemes, standards and codes of practice.

With respect to renewable energy options that occupy land, the focus of governance has been directed to technological adoption and, public acceptance (Sequeira and Santos 2018), rather than land use. Recent work has found that spatial processes shape the emerging energy transition, creating zones of friction between global investors, national and local governments, and civil society (McEwan 2017;

Jepson and Caldas 2017). For example, hydropower and ground-based solar parks in India have involved enclosure of lands designated as degraded, displacing pastoral use by vulnerable communities, constituting forms of spatial injustice (Yenneti et al. 2016). Hydropower leads to dam-induced displacement, and though this can be addressed through compensation mechanisms, governance is complicated by a lack of transparency in resettlement data (Kirchherr et al. 2016, 2019). Renewable energy production is resulting in new land conflict frontiers where degraded land is framed as having mitigation value such as for palm oil production and wind power in Mexico (Backhouse and Lehmann 2020); land use conflict as well as impacts on wildlife from large-scale solar installations have also emerged in the southwestern United States (Mulvaney 2017). The renewable energy transition also involves the extraction of critical minerals used in renewable energy technologies, such as lithium and cobalt. Governance challenges include the lack of transparent greenhouse gas accounting for mining activities (Lee et al. 2020a), and threats to biodiversity from land disturbance, which require strategic planning to address (Sonter et al. 2020a). Strategic spatial planning is needed more generally to address trade-offs between using land for renewable energy and food: for example, agriculture can be co-located with solar photovoltaics (Barron-Gafford et al. 2019) or wind power (Miller and Keith 2018a). Integrative spatial planning can integrate renewable energy with not just agriculture, but mobility and housing (Hurlbert et al. 2019). Integrated planning is needed to avoid scalar pitfalls, and local and regional contextualised governance solutions need to be sited within a planetary frame of reference (Biermann et al. 2016). Greater planning and coordination are also needed to ensure co-benefits from land-based mitigation (see Box 12.3) as well as from CDR and efforts to reduce food systems emissions.

In emerging domains for governance such as land-based mitigation, global institutions, private sector networks and civil society organisations are also playing key roles in terms of norm-setting. The shared languages and theoretical frameworks, or cognitive linkages (Pattberg et al. 2018) that arise with polycentric governance can not only be helpful in creating expectations and establishing benchmarks for (in)appropriate practices where enforceable ‘hard law’ is missing (Karlsson-Vinkhuyzen et al. 2018; Gajevic Sayegh 2020), but can also form the basis of voluntary guidelines or niche markets (See also the case study in Box 12.3) However, the ability to apply participatory processes for developing voluntary guidelines and other participatory norm-setting endeavours varies from place to place. Social and cultural norms shape the ability of women, youth, and different ethnic groups to participate in governance fora, such as those around agroecological transformation (Anderson et al. 2019). Furthermore, establishing new norms alone does not solve structural challenges such as lack of access to food, confront power imbalances, or provide mechanisms to deal with uncooperative actors (Morrison et al. 2019).

START BOX 12.3 HERE

Box 12.3 Land Degradation Neutrality as a framework to manage trade-offs in land-based mitigation

The UNCCD introduced the concept of Land Degradation Neutrality (LDN), defined as “a state whereby the amount and quality of land resources necessary to support ecosystem functions and services and enhance food security remain stable or increase within specified temporal and spatial scales and ecosystems” (UNCCD 2015), and it has been adopted as a target of Goal 15 of the SDGs, Life on Land. At December 2020, 124 (mostly developing) countries have committed to pursue voluntary LDN targets.

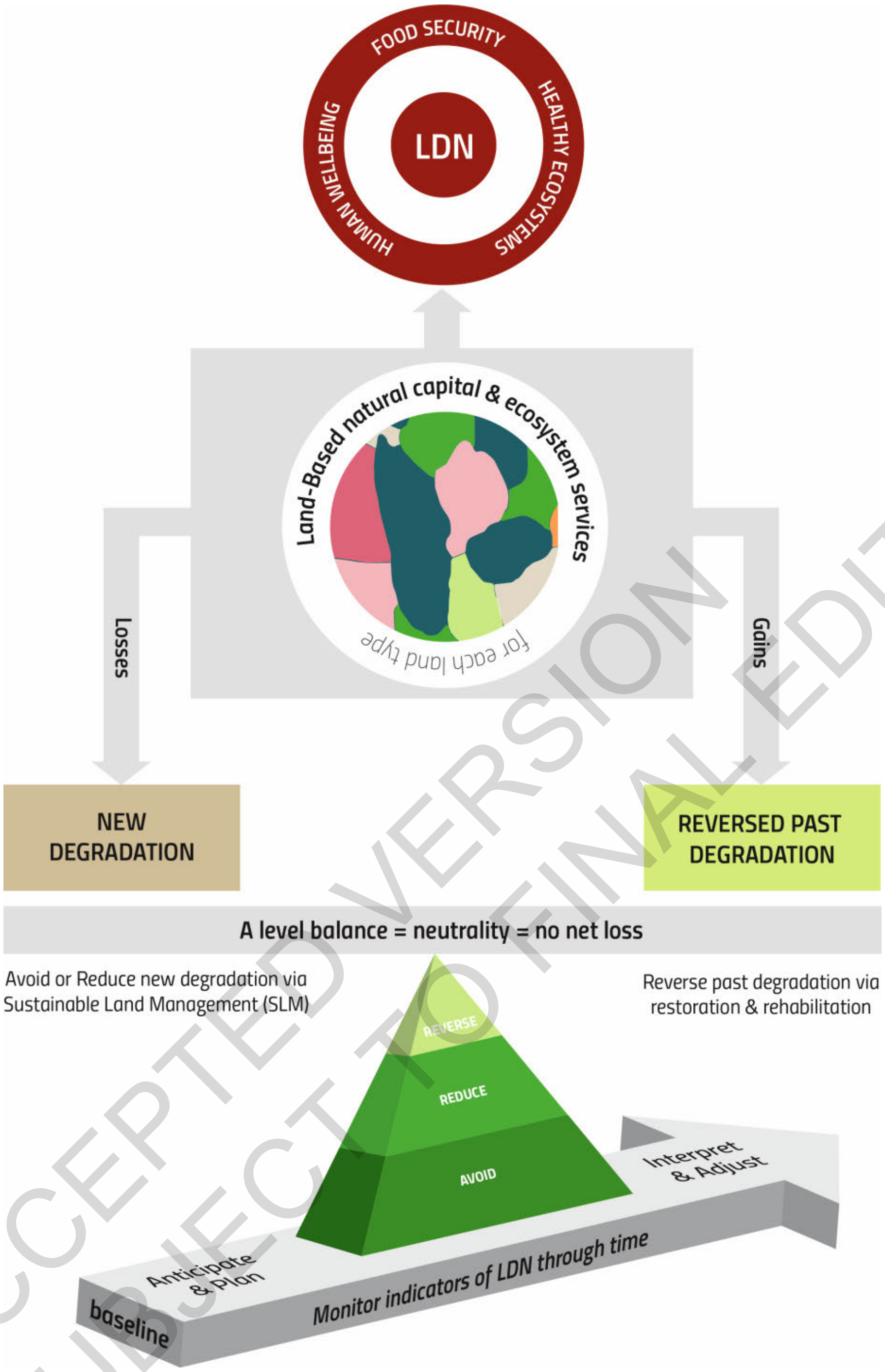
The goal of LDN is to maintain or enhance land-based natural capital, and its associated ecosystem services such as provision of food and regulation of water and climate, while enhancing the resilience of the communities that depend on the land. LDN encourages a dual-pronged approach promoting sustainable land management (SLM) to avoid or reduce land degradation, combined with strategic effort in land restoration and rehabilitation to reverse degradation on degraded lands and thereby deliver the target of "no net loss" of productive land (Orr et al. 2017).

In the context of LDN, land restoration refers to actions undertaken with the aim of reinstating ecosystem functionality, whereas land rehabilitation refers to actions undertaken with a goal of provision of goods and services (Cowie et al. 2018). Restoration interventions can include destocking to encourage regeneration of native vegetation; shelter belts of local species established from seed or seedlings, strategically located to provide wildlife corridors and link habitat; and rewetting drained peatland. "Farmer-managed natural regeneration" is a low-cost restoration approach in which regeneration of tree stumps and roots is encouraged, stabilising soil and enhancing soil nutrients and organic matter levels (Chomba et al. 2020; Lohbeck et al. 2020). Rehabilitation actions include establishment of energy crops, or afforestation with fast-growing exotic trees to sequester carbon or produce timber. Application of biochar can facilitate rehabilitation by enhancing nutrient retention and water holding capacity, and stimulating microbial activity (Cowie 2020a).

SLM, rehabilitation and restoration activities undertaken towards national LDN targets have potential to deliver substantial CDR through carbon sequestration in vegetation and soil. In addition, biomass production, for bioenergy or biochar, could be an economically viable land use option for reversing degradation, through rehabilitation. Alternatively, a focus on ecological restoration (Gann et al. 2019) as the strategy for reversing degradation will deliver greater biodiversity benefits.

Achieving neutrality requires estimating the likely impacts of land-use and land management decisions, to determine the area of land, of each land type, that is likely to be degraded (Orr et al. 2017). This information is used to plan interventions to reverse degradation on an equal area of the same land type. Therefore, pursuit of LDN requires concerted and coordinated efforts to integrate LDN objectives into land-use planning and land management, underpinned by sound understanding of the human-environment system and effective governance mechanisms.

Countries are advised to apply a landscape-scale approach for planning LDN interventions, in which land uses are matched to land potential, and resilience of current and proposed land uses is considered, to ensure that improvement in land condition is likely to be maintained (Cowie 2020a). A participatory approach that enables effective representation of all stakeholders is encouraged, to facilitate equitable outcomes from planning decisions, recognising that decisions on LDN interventions are likely to involve trade-offs between various environmental and socio-economic objectives (Schulze et al. 2021). Planning and implementation of LDN programmes provides a framework in which locally-adapted land-based mitigation options can be integrated with use of land for production, conservation and settlements, in multifunctional landscapes where trade-offs are recognised and managed, and synergistic opportunities are sought. LDN is thus a vehicle to focus collaboration in pursuit of the multiple land-based objectives of the multilateral environmental agreements and the SDGs.



Box 12.3, Figure 1 Schematic illustrating the elements of the Land Degradation Neutrality conceptual framework.

Source: (Cowie et al. 2018)

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Table 12.10 collates risks, impacts and opportunities associated with different mitigation options that occupy land.

Table 12.10 Summary of impacts, risks and co-benefits associated with land occupation by mitigation options considered in section 12.5.

Mitigation option	Impacts and risks	Opportunities for co-benefits
Non-bio-based options that may displace food production		
Solar farms	Land use competition; Loss of soil carbon; heat island effect (scale dependent) {12.5.3, 12.5.4}	Target areas unsuitable for agriculture such as deserts {12.5.3}
Hydropower (dams)	Land use competition, displacement of natural ecosystems, CO_2 and CH_4 emissions {12.5.3, 12.5.4}	Water storage (including for irrigation) and regulation of water flows; Pumped storage can store excess energy from other renewable generation sources. {12.5.3}
Non-bio-based options that can (to a varying degree) be integrated with food production		
Wind turbines	May affect local/regional weather and climate (scale dependent) Impact on wildlife and visual impacts {12.5.3}	Design and siting informed by t visual landscape impacts, relevant habitats, and flight trajectories of migratory birds. {12.5.3}
Solar panels	Land use competition {12.5.3}	Integration with buildings and other infrastructure. integration with food production is being explored {12.5.2}
Enhanced weathering	Disturbance at sites of extraction; Ineffective in low rainfall regions {12.3.1.2}	Increase crop yields and biomass production through nutrient supply and increasing pH of acid soils; synergies with biochar {12.5.3}
Bio-based options that may displace existing food production		
A/R	Land use competition, potentially leading to indirect land use change; reduced water availability; loss of biodiversity {12.5.3}	Strategic siting to minimise adverse impacts on hydrology, land use, biodiversity {12.5.3}
Biomass crops	Land use competition, potentially leading to indirect land use change; reduced water availability; reduced soil fertility; loss of biodiversity {12.5.3}	Strategic siting to minimise adverse impacts / enhance beneficial effects on land use, landscape variability, biodiversity, soil organic matter, hydrology and water quality {12.5.3}
Bio-based options that can (to a varying degree) be combined with food production		

Agroforestry	Competition with adjacent crops and pastures reduces yields {7.4.3.3}	Shelter for stock and crops, diversification, biomass production, increases soil organic matter and soil fertility. Increased biodiversity and perennial vegetation enhance beneficial organisms; can reduce need for pesticides {7.4.3.3, 12.5.3}
Soil carbon management in croplands and grasslands	Increase in nitrous oxide emissions if fertiliser used to enhance crop production; Reduced cereal production through increased crop legumes and pasture phases could lead to indirect land use change {7.4.3.1, 7.4.3.6}	Increasing soil organic matter improves soil health, increases crop and pasture yields, and resilience to drought, can reduce fertiliser requirement, nutrient leaching and need for land use change. {7.4.3.1}
Biochar addition to soil	Land use competition if biochar is produced from purpose-grown biomass. Loss of forest carbon stock and impacts on biodiversity if biomass is harvested unsustainably. {12.5.3}	Facilitate beneficial use of organic residues, to return nutrients to farmland. Increase land productivity to increase C sequestration in vegetation and soil. Increase nutrient-use efficiency, and reduce requirement for chemical fertiliser. {7.4.3.2, 12.5.3}
Harvest residue extraction and use for bioenergy, biochar and other bio-products	Decline in soil organic matter and soil fertility {12.5.3}	Retain portion of stubble; return nutrients e.g. as ash Utilising forest residues for bioenergy reduces fuel load and wildfire risk {7.4.3.2, 12.5.3}
Manure management (i.e., for biogas)	Risk of fugitive emissions Can contain pathogens {7.4.3.7, 12.5.3}	Biogas as renewable energy source. Apply digestate as soil amendment {12.5.3}
Options that don't occupy land used for food production		
Management of organic waste (food waste, bio-solids, organic component of MSW)	Can contain contaminants (heavy metals, persistent organic pollutants, pathogens) {12.5.3}	Processing using anaerobic digestion or pyrolysis produces renewable gas and soil amendment, enabling return of nutrients to farmland. (note that some feedstock nitrogen is lost in pyrolysis) {12.5.3}
A/R and biomass production on degraded non-forested land (e.g., abandoned agricultural land)	High labour and material inputs can be needed to restore productivity on degraded land. Abandoned land can support informal grazing and have significant biodiversity value. Reduced water availability. {12.5.3}	Application of biochar can re-establish nutrient cycling; bioenergy crops can add organic matter, restoring soil fertility, and can remove heavy metals, enabling food production. {7.4.3.2, 12.5.3}

START CROSS-WORKING GROUP BOX 3 HERE

Cross-Working Group Box in Working Group II, Chapter 5

Cross-Working Group Box 3: Mitigation and Adaptation via the Bioeconomy

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Summary statement

The growing demand for biomass offers both opportunities and challenges to mitigate and adapt to climate change and natural resource constraints (high confidence). Increased technology innovation, stakeholder integration and transparent governance structures and procedures at local to global scales are key to successful bioeconomy deployment maximizing benefits and managing trade-offs (high confidence).

Limited global land and biomass resources accompanied by growing demands for food, feed, fibre, and fuels, together with prospects for a paradigm shift towards phasing out fossil fuels, set the frame for potentially fierce competition for land[2] and biomass to meet burgeoning demands even as climate change increasingly limits natural resource potentials (*high confidence*).

Sustainable agriculture and forestry, technology innovation in bio-based production within a circular economy and international cooperation and governance of global trade in products to reflect and disincentivize their environmental and social externalities, can provide mitigation and adaptation via bioeconomy development that responds to the needs and perspectives of multiple stakeholders to achieve outcomes that maximize synergies while limiting trade-offs *(high confidence)*.

Background

There is *high confidence* that climate change, population growth and changes in per capita consumption will increase pressures on managed as well as natural and semi-natural ecosystems, exacerbating existing risks to livelihoods, biodiversity, human and ecosystem health, infrastructure, and food systems (Conijn et al. 2018; IPCC 2018, 2019; Lade et al. 2020). At the same time, many global mitigation scenarios presented in IPCC assessment reports rely on large GHG emissions reduction in the AFOLU sector and concurrent deployment of reforestation/afforestation and biomass use in a multitude of applications (Rogelj et al. 2018; Hanssen et al. 2020; AR6 WG1 Ch.4 and Ch.5; AR6 WG3 Ch.3 and Ch.7).

Given the finite availability of natural resources, there are invariably trade-offs that complicate land-based mitigation unless land productivity can be enhanced without undermining ecosystem services (e.g., Obersteiner et al. 2016; Campbell et al. 2017; Conijn et al. 2018; Caron et al. 2018; WRI 2018; Heck et al. 2018; Smith et al. 2019). Management intensities can often be adapted to local conditions

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FOOTNOTE[2] For lack of space the focus is on land only although the bioeconomy also includes sea-related bioresources.

with consideration of other functions and ecosystem services, but at a global scale the challenge remains to avoid further deforestation and degradation of intact ecosystems, in particular biodiversity-rich systems (AR6 WGII Cross-Chapter Box on NATURAL), while meeting the growing demands. Further, increased land-use competition can affect food prices and impact food security and livelihoods (To and Grafton 2015; Chakravorty et al. 2017), with possible knock-on effects related to civil unrest (Abbott et al. 2017; D'Odorico et al. 2018).

Developing new bio-based solutions while mitigating overall biomass demand growth

Many existing bio-based products have significant mitigation potential. Increased use of wood in buildings can reduce GHG emissions from cement and steel production while providing carbon storage (Churkina et al. 2020). Substitution of fossil fuels with biomass in manufacture of cement and steel can reduce GHG emissions where these materials are difficult to replace. Dispatchable power based on biomass can provide power stability and quality as the contribution from solar and wind power increases (See AR6 WG3 Ch.6), and biofuels can contribute to reducing fossil fuel emissions in the transport and industry sectors (See AR6 WG3 Ch.10 and Ch.11). The use of bio-based plastics, chemicals and packaging could be increased, and biorefineries can achieve high resource-use efficiency in converting biomass into food, feed, fuels and other bio-based products (Aristizábal-Marulanda and Cardona Alzate 2019; Schmidt et al. 2019). There is also scope for substituting existing bio-based products with more benign products. For example, cellulose-based textiles can replace cotton, which requires large amounts of water, chemical fertilizers and pesticides to ensure high yields.

While increasing and diversified use of biomass can reduce the need for fossil fuels and other GHG-intensive products, unfavourable GHG balances may limit the mitigation value. Growth in biomass use may in the longer term also be constrained by the need to protect biodiversity and ecosystems' capacity to support essential ecosystem services. Biomass use may also be constrained by water scarcity and other resource scarcities, and/or challenges related to public perception and acceptance due to impacts caused by biomass production and use. Energy conservation and efficiency measures and deployment of technologies and systems that do not rely on carbon, e.g., carbon-free electricity supporting, inter alia, electrification of transport as well as industry processes and residential heating (IPCC 2018; UNEP 2019), can constrain the growth in biomass demand when countries seek to phase out fossil fuels and other GHG-intensive products while providing an acceptable standard of living. Nevertheless, demand for bio-based products may become high where full decoupling from carbon is difficult to achieve (e.g., aviation, bio-based plastics and chemicals) or where carbon storage is an associated benefit (e.g., wood buildings, BECCS, biochar for soil amendments), leading to challenging trade-offs (e.g., food security, biodiversity) that need to be managed in environmentally sustainable and socially just ways.

Changes on the demand side as well as improvements in resource-use efficiencies within the global food and other bio-based systems can also reduce pressures on the remaining land resources. For example, dietary changes toward more plant-based food (where appropriate) and reduced food waste can provide climate change mitigation along with health benefits (Willett et al. 2019; WG3 Ch 7.4 and Ch 12.4) and other co-benefits with regard to food security, adaptation and land use (Smith et al. 2019a; Mbow et al. 2019; WG2 Ch.5). Advancements in the provision of novel food and feed sources (e.g., cultured meat, insects, grass-based protein feed and cellular agriculture) can also limit the pressures on finite natural resources (Parodi et al. 2018; Zabaniotou 2018; WG3 Ch 12.4).

Circular bioeconomy

Circular economy approaches (AR6 WG3 Ch 12.6) are commonly depicted by two cycles, where the biological cycle focuses on regeneration in the biosphere and the technical cycle focuses on reuse, refurbishment and recycling to maintain value and maximize material recovery (Mayer et al. 2019a). Biogenic carbon flows and resources are part of the biological carbon cycle, but carbon-based products can be included in, and affect, both the biological and the technical carbon cycles (Velenturf et al. 2019; Winans et al. 2017; Kirchherr et al. 2017). The integration of circular economy and bioeconomy principles has been discussed in relation to organic waste management (Teigiserova et al. 2020), societal transition and policy development (Bugge et al. 2019; European Commission 2018) as well as COVID-19 recovery strategies (Palahí et al. 2020). To maintain the natural resource base, circular bioeconomy emphasizes sustainable land use and the return of biomass and nutrients to the biosphere when it leaves the technical cycle.

Biomass scarcity is an argument for adopting circular economy principles for the management of biomass as for non-renewable resources. This includes waste avoidance, product reuse and material recycling, which keep down resource use while maintaining product and material value. However, reuse and recycling is not always feasible, e.g., when biofuels are used for transport and bio-based biodegradable chemicals are used to reduce ecological impacts where losses to the environment are unavoidable. A balanced approach to management of biomass resources could take departure in the carbon cycle from a value-preservation perspective and the possible routes that can be taken for biomass and carbon, considering a carbon budget defined by the Paris Agreement, principles for sustainable land use and natural ecosystem protection.

Land use opportunities and challenges in the bioeconomy

Analyses of synergies and trade-offs between adaptation and mitigation in the agriculture and forestry sectors show that outcomes depend on context, design and implementation, so actions have to be tailored to the specific conditions to minimize adverse effects (Kongsager 2018). This is supported in literature analyzing the nexus between land, water, energy and food in the context of climate change which consistently concludes that addressing these different domains together rather than in isolation would enhance synergies and reduce trade-offs (Obersteiner et al. 2016; D’Odorico et al. 2018; Soto Golcher and Visseren-Hamakers 2018; Momblanch et al. 2019; Froese et al. 2019).

Nature-based solutions addressing climate change can provide opportunities for sustainable livelihoods as well as multiple ecosystem services, such as flood risk management through floodplain restoration, saltmarshes, mangroves or peat renaturation (UNEP 2021; AR6 WGII Cross-Chapter Box on NATURAL). Climate-smart agriculture can increase productivity while enhancing resilience and reducing GHG emissions inherent to production (Lipper et al. 2014; Singh and Chudasama 2021). (Bell et al. 2018; FAO 2019b; Singh and Chudasama 2021) Similarly, climate-smart forestry considers the whole value chain and integrates climate objectives into forest sector management through multiple measures (from strict reserves to more intensively managed forests) providing mitigation and adaptation benefits (Nabuurs et al. 2018; Verkerk et al. 2020) (AR6 WG3 Ch 7.3).

Agroecological approaches can be integrated into a wide range of land management practices to support a sustainable bioeconomy and address equity considerations (HLPE 2019). Relevant land-use practices, such as agroforestry, intercropping, organic amendments, cover crops and rotational grazing, can provide mitigation and support adaption to climate change via food security, livelihoods, biodiversity and health co-benefits (Bezner Kerr et al. 2019; Bharucha et al. 2020; Clark et al. 2019b; D’Annolfo et al. 2017; Garibaldi et al. 2016; Ponisio et al. 2015; Renard and Tilman 2019; HLPE 2019; Sinclair et al. 2019; Córdova et al. 2019; Mbow et al. 2019; Bezner Kerr et al. 2021; and AR6 WGII Chapter 2 Cross-Chapter Box NATURAL). Strategic integration of appropriate biomass production systems into agricultural landscapes can provide biomass for bioenergy and other bio-based products while providing co-benefits such as enhanced landscape diversity, habitat quality, retention of nutrients and sediment, erosion control, climate regulation, flood regulation, pollination and biological pest and disease control (Christen and Dalgaard 2013; Asbjornsen et al. 2014; Englund et al. 2020; Cacho et al. 2018; Dauber and Miyake 2016; Holland et al. 2015; Milner et al. 2016; Ssegane et al. 2015; Ssegane and Negri 2016; Styles et al. 2016; Zalesny et al. 2019; Zumpf et al. 2017; HLPE 2019; Cubins et al. 2019; Alam and Dwivedi 2019; Olsson et al. 2019) (AR6 WGIII Chapter 12 Box 12.3 on UNCCD-LDN). Such approaches can help limit environmental impacts from intensive agriculture while maintaining or increasing land productivity and biomass output.

Cross-Working Group Box 3, Figure 1 Left: High-input intensive agriculture, aiming for high yields of a few crop species, with large fields and no semi-natural habitats. Right: Agroecological agriculture, supplying a range of ecosystem services, relying on biodiversity and crop and animal diversity instead of external inputs, and integrating plant and animal production, with smaller fields and presence of semi-natural habitats.

Credit: Jacques Baudry (left); Valérie Viaud (right), published in (van der Werf et al. 2020)

Transitions from conventional to new biomass production and conversion systems include challenges related to cross-sector integration and limited experience with new crops and land use practices, including needs for specialized equipment (Thornton and Herrero 2015; HLPE 2019; AR6 WG2 Section 5.10). Introduction of agroecological approaches and integrated biomass/food crop production can result in lower food crop yields per hectare, particularly during transition phases, potentially causing indirect land use change, but can also support higher and more stable yields, reduce costs, and increase profitability under climate change (Muller et al. 2017; Seufert and Ramakutty 2017; Barbieri et al. 2019; HLPE 2019; Sinclair et al. 2019; Smith et al. 2019a, 2020). Crop diversification, organic amendments, and biological pest control (HLPE 2019) can reduce input costs and risks of occupational pesticide exposure and food and water contamination (González-Alzaga et al. 2014; EFSA 2017; Mie et al. 2017), reduce farmers' vulnerability to climate change (e.g., droughts and spread of pests and diseases affecting plant and animal health (Delcour et al. 2015; FAO 2020) and enhance provisioning and sustaining ecosystem services, such as pollination (D'Annolfo et al. 2017; Sinclair et al. 2019).

Barriers toward wider implementation include absence of policies that compensate land owners for providing enhanced ecosystem services and other environmental benefits, which can help overcome short term losses during the transition from conventional practices before longer term benefits can accrue. Other barriers include limited access to markets, knowledge gaps, financial, technological or labour constraints, lack of extension support and insecure land tenure (Jacobi et al. 2017; Kongsager 2017; Hernández-Morcillo et al. 2018; Iiyama et al. 2018; HLPE 2019). Regional-level agroecology transitions may be facilitated by co-learning platforms, farmer networks, private sector, civil society groups, regional and local administration and other incentive structures (e.g. price premiums, access to credit, regulation) (Coe et al. 2014; Pérez-Marin et al. 2017; Mier y Terán Giménez Cacho et al. 2018;

HLPE 2019; Valencia et al. 2019; SAEPEA 2020). With the right incentives, improvements can be made with regard to profitability, making alternatives more attractive to land owners.

Governing the solution space

Literature analysing the synergies and trade-offs between competing demands for land suggest that solutions are highly contextualized in terms of their environmental, socioeconomic and governance-related characteristics, making it difficult to devise generic solutions (Haasnoot et al. 2020). Aspects of spatial and temporal scale can further enhance the complexity, for instance where transboundary effects across jurisdictions or upstream-downstream characteristics need to be considered, or where climate change trajectories might alter relevant biogeophysical dynamics (Postigo and Young 2021). Nonetheless, there is broad agreement that taking the needs and perspectives of multiple stakeholders into account in a transparent process during negotiations improves the chances of achieving outcomes that maximize synergies while limiting trade-offs (Ariti et al. 2018; Metternicht 2018; Favretto et al. 2020; Kopáček 2021; Muscat et al. 2021). Yet differences in agency and power between stakeholders or anticipated changes in access to or control of resources can undermine negotiation results even if there is a common understanding of the overarching benefits of more integrated environmental agreements and the need for greater coordination and cooperation to avoid longer-term losses to all (Aarts and Leeuwis 2010; Weitz et al. 2017). There is also the risk that strong local participatory processes can become disconnected from broader national plans, and thus fail to support the achievement of national targets. Thus, connection between levels is needed to ensure that ambition for transformative change is not derailed at local level (Aarts and Leeuwis 2010; Postigo and Young 2021).

Decisions on land uses between biomass production for food, feed, fibre or fuel, as well as nature conservation or restoration and other uses (e.g., mining, urban infrastructure), depend on differences in perspectives and values. Because the availability of land for diverse biomass uses is invariably limited, setting priorities for land-use allocations therefore first depends on making the perspectives underlying what is considered as ‘high-value’ explicit (Fischer et al. 2007; Garnett et al. 2015; De Boer and Van Ittersum 2018; Muscat et al. 2020). Decisions can then be made transparently based on societal norms, needs and the available resource base. Prioritization of land-use for the common good therefore requires societal consensus-building embedded in the socioeconomic and cultural fabric of regions, societies and communities. Integration of local decision-making with national planning ensures local actions complement national development objectives.

International trade in the global economy today provides important opportunities to connect producers and consumers, effectively buffering price volatilities and potentially offering producers countries access to global markets, which can be seen as an effective adaptation measure (Baldos and Hertel 2015; Costinot et al. 2016; Hertel and Baldos 2016; Gouel and Laborde 2021; AR6 WG2 Ch 5.11). But there is also clear evidence that international trade and the global economy can enhance price volatility, lead to food price spikes and affect food security due to climate and other shocks, as seen recently due to the COVID-19 pandemic (Cottrell et al. 2019; WFP-FSIN 2020; Verschuur et al. 2021; AR6 WG2 Ch 5.12). The continued strong demand for food and other bio-based products, mainly from high- and middle-income countries, therefore requires better cooperation between nations and global governance of trade to more accurately reflect and disincentivize their environmental and social externalities. Trade in agricultural and extractive products driving land-use change in tropical forest and savanna biomes is of major concern because of the biodiversity impacts and GHG emissions incurred in their provision (Hosonuma et al. 2012; Forest Trends 2014; Smith et al. 2014; Henders et al. 2015; Curtis et al. 2018; Pendrill et al. 2019; Seymour and Harris 2019; Kissinger et al. 2021; AR6 WG2 CCP Tropical Forests).

In summary, there is significant scope for optimizing use of land resources to produce more biomass while reducing adverse effects (*high confidence*). Context-specific prioritisation, technology innovation in bio-based production, integrative policies, coordinated institutions and improved governance mechanisms to enhance synergies and minimize trade-offs can mitigate the pressure on managed as well as natural and semi-natural ecosystems (*medium confidence*). Yet, energy conservation and efficiency measures, and deployment of technologies and systems that do not rely on carbon-based energy and materials, are essential for mitigating biomass demand growth as countries pursue ambitious climate goals (*high confidence*).

END CROSS-WORKING BOX 3 HERE

12.6 Other cross-sectoral implications of mitigation

This section presents further cross-sectoral considerations related to GHG mitigation. Firstly, various cross-sectoral perspectives on mitigation actions are presented. Then, sectoral policy interactions are presented. Finally, implications in terms of international trade spill-over effects and competitiveness, and finance flows and related spill-over effects at the sectoral level are addressed.

12.6.1 Cross-sectoral perspectives on mitigation action

Chapters 5 to 11 present mitigation measures applicable in individual sectors, and potential co-benefits and adverse side effects[3] of these individual measures. This section builds on the sectoral analysis of mitigation action from a cross-sectoral perspective. Firstly, Section 12.6.1.1 brings together some of the observations presented in the sectoral chapters to show how different mitigation actions in different sectors can contribute to the same co-benefits and result in the same adverse side effects, thereby demonstrating the potential synergistic effects. The links between these co-benefits and adverse side effects and the SDGs is also demonstrated. In Section 12.6.1.2, the focus turns from sector-specific mitigation measures to mitigation measures which have cross-sectoral implications, including measures that have application in more than one sector and measures where implementation in one sector impacts on implementation in another. Finally, Section 12.6.1.3 notes the cross-sectoral relevance of a selection of General-Purpose Technologies, a topic that is covered further in Chapter 16.

12.6.1.1 A cross-sectoral perspective on co-benefits and adverse side effects of mitigation measures, and links the SDGs

A body of literature has been developed which addresses the *co-benefits* of climate mitigation action, (Karlsson et al. 2020). *Adverse side effects* of mitigation are also well documented. Co-benefits and adverse side-effects in individual sectors and associated with individual mitigation measures are discussed in the individual sector chapters (Sections 5.2, 6.7.7, 7.4, 7.6, 8.2, 8.4, 9.8, 10.1.1, 11.5.3), as well as in previous IPCC General and Special Assessment reports. The term *co-impacts* has been proposed to capture both the co-benefits and adverse side-effects of mitigation. An alternative framing is one of multiple objectives, where climate mitigation is placed alongside other objectives when

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[3] FOOTNOTE[3] Here, the term co-benefits is used to refer to the additional benefits to society and the environment that are realised in parallel with emissions reductions, while an understanding of adverse side effects highlights where policy and decision makers are required to make trade-offs between mitigation benefits and other impacts. The choice of language differs to some degree in other chapters.

assessing policy decisions (Ürge-Vorsatz et al. 2014; Mayrhofer and Gupta 2016; Cohen et al. 2017; Bhardwaj et al. 2019).

The identification and assessment of co-benefits has been argued to serve a number of functions (Section 1.4) including using them as a leverage for securing financial support for implementation, providing justification of actions which provide a balance of both short and long-term benefits and obtaining stakeholder buy-in *(robust evidence, low agreement)* (Karlsson et al. 2020). Assessment of adverse side-effects has been suggested to be useful in avoiding unforeseen negative impacts of mitigation and providing policy and decision makers with the information required to make informed trade-offs between climate and other benefits of actions (Ürge-Vorsatz et al. 2014; Bhardwaj et al. 2019; Cohen et al. 2019) *(high evidence, low agreement)*.

Various approaches to identifying and organising co-impacts in specific contexts and across sectors have been proposed towards providing more comparable and standardised analyses. However, consistent quantification of co-impacts, including cost-benefit analysis, and the utilisation of the resulting information, remains a challenge (Ürge-Vorsatz et al. 2014; Floater et al. 2016; Mayrhofer and Gupta 2016; Cohen et al. 2019; Karlsson et al. 2020). This challenge is further exacerbated when considering that co-impacts of a mitigation measure in one sector can either enhance or reduce the co-impacts associated with mitigation in another, or the achievement of co-benefits in one geographic location can lead to adverse side effects in another. For example, the production of lithium for batteries for energy storage has the potential to contribute to protecting water resources and reducing wastes associated with coal fired power in many parts of the world, but mining of lithium has the potential for creating water and waste challenges if not managed properly (Agusdinata et al. 2018; Kaunda 2020).

While earlier literature has suggested that co-impacts assessments can support adoption of climate mitigation action, a more recent body of literature has suggested limitations in such framing (Ryan 2015; Bernauer and McGrath 2016; Walker et al. 2018). Presenting general information on co-impacts as a component of a mitigation analysis does not always lead to increased support for climate mitigation action. Rather, the most effective framing is determined by factors relating to local context, type of mitigation action under consideration and target stakeholder group. More work has been identified to be required to bring context into planning co-impacts assessments and communication thereof (Ryan 2015; Bernauer and McGrath 2016; Walker et al. 2018) *(low evidence, low agreement)*.

An area where the strong link between the cross-sectoral co-impacts of mitigation action and global government policies is being clearly considered is in the achievement of the SDGs (Chapters 1 and 17, individual sectoral chapters) (Obergassel et al. 2017; Doukas et al. 2018; Markkanen and Anger-Kraavi 2019; Smith et al. 2019; van Soest et al. 2019). Figure 12.9 demonstrates these relationships from a cross-sectoral perspective. It shows the links between sectors which give rise to emissions, the mitigation measures that can find application in the sector, co-benefits and adverse side effects of mitigation measures and the SDGs (noting that the figure is not intended to be comprehensive). Such a framing of co-impacts from a cross-sectoral perspective in the context of the SDGs could help to further support climate mitigation action, particularly within the context of the Paris Agreement (Gomez-Echeverri 2018) (*medium evidence, medium agreement*). Literature sources utilised in the compilation of this diagram are presented in Supplementary Material 12.C.

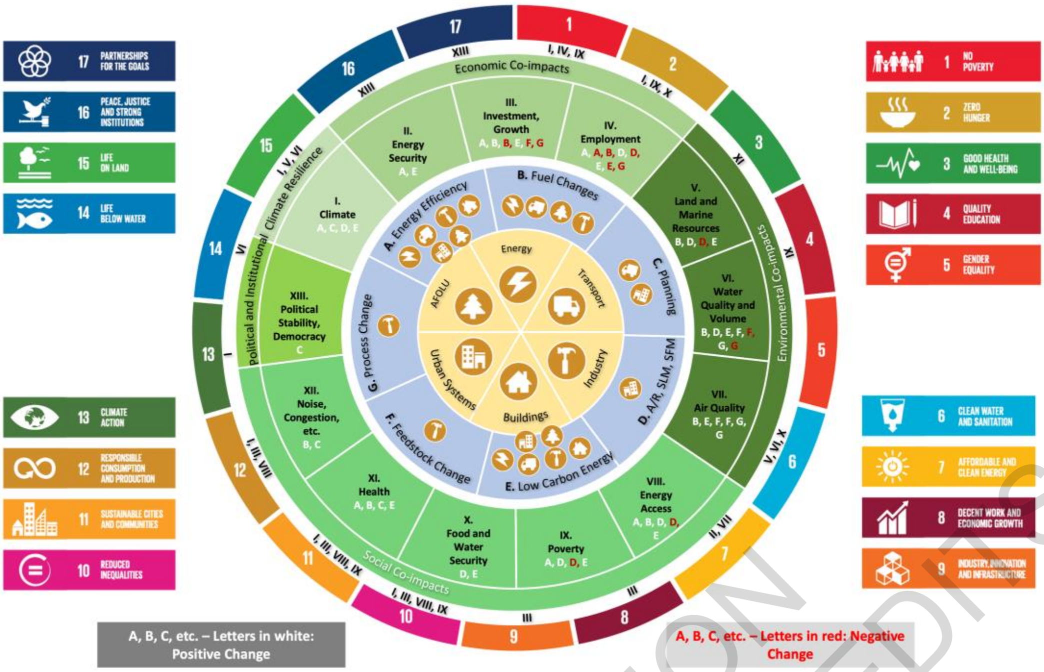

Figure 12.9 Co-benefits and adverse side effects of mitigation actions with links to the SDGs.

The inner circle represents the sectors in which mitigation occurs. The second circle shows different generic types of mitigation actions (A to G), with the symbols showing which sectors they are applicable to. The third circle indicates different types of climate related co-benefits (green letters) and adverse side effects (red letters) that may be observed as a result of implementing each of the mitigation actions (as indicated by the letters A-G). Here I relates to climate resilience, II-IV economic co-impacts, V-VII environmental, VIII-XII social, and XIII political and institutional. The final circle maps co-benefits and adverse side-effects to the SDGs (Cohen et al. 2021).

12.6.1.2 Mitigation measures from a cross-sectoral perspective

Three aspects of mitigation from a cross-sectoral perspective are considered, following (Barker et al. 2007):

- mitigation measures used in more than one sector;
- implications of mitigation measures for interaction and integration between sectors; and
- competition among sectors for scarce resources.

A number of mitigation measures find application in more than one sector. Renewable energy technologies such as solar and wind may be used for grid electricity supply, as embedded generation in the buildings sector and for energy supply in the agriculture sector (Chapters 6, 7 and 8) (Shahsavari and Akbari 2018). Hydrogen and fuel cells, coupled with low carbon energy technologies for producing the hydrogen, is being explored in transport, urban heat, industry and for balancing electricity supply (Chapters 6, 8, 11) (Dodds et al. 2015; Staffell et al. 2019). Electric vehicles are considered an option for balancing variable power (Kempton and Tomić 2005; Liu and Zhong 2019). Carbon Capture and Storage (CCS) and Carbon Capture and Utilisation (CCU) has potential application in a number of industrial processes (cement, iron and steel, petroleum refining and pulp and paper) (Chapters 6 and 11) (Leeson et al. 2017; Garcia and Berghout 2019) and the fossil fuel electricity sector (Chapter 6). When coupled with energy recovery from biomass , CCS can provide a carbon sink (BECCS) (Section 12.5). On the demand side, energy efficiency options find application across the sectors (Chapters 6, 8, 9, 10, and 11), as does reducing demand for goods and services (Chapter 5), and improving material efficiency (Section 11.3.2).

A range of examples of where mitigation measures result in cross-sectoral interactions and integration is identified. The mitigation potential of electric vehicles, including plug-in hybrid hybrids, is linked to the extent of decarbonisation of the electricity grid, as well as to the liquid fuel supply emissions profile (Lutsey 2015). Making buildings energy positive, where excess energy is used to charge vehicles, can increase the potential of electric and hybrid vehicles (Zhou et al. 2019). Advanced process control and process optimisation in industry can reduce energy demand and material inputs (Section 11.3), which in turn can reduce emissions linked to resource extraction and manufacturing. Reductions in coal-fired power generation through replacement with renewables or nuclear power result in a reduction in coal mining and its associated emissions. Increased recycling results in a reduction in emissions from primary resource extraction. CCU can contribute to the transition to more renewable energy systems via power-to-X technologies, which enables the production of CO_2-based fuels/e-fuels and chemicals using carbon dioxide and hydrogen (Breyer et al. 2015; Anwar et al. 2020). Certain reductions in the AFOLU sector are contingent on energy sector decarbonisation. Trees and green roofs planted to counter urban heat islands reduce the demand for energy for air conditioning and simultaneously sequester GHGs (Kim and Coseo 2018; Kuronuma et al. 2018). Recycling of organic waste avoids methane generation if the waste would have been disposed of in landfill sites, can generate renewable energy if treated through anaerobic digestion and can reduce requirements for synthetic fertiliser production if the nutrient value is recovered (Creutzig et al. 2015). Liquid transport biofuels links to the land, energy and transport sectors (Section 12.5.2.2).

Demand-side mitigation measures, discussed in Chapter 5, also have cross-sectoral implications which need to be taken into account when calculating mitigation potentials. Residential electrification has the potential to reduce emissions associated with lighting and heating particularly in developing countries where this is currently met by fossil fuels and using inefficient technologies, but will increase demand for electricity (Chapters 5, 8 and Sections 6.6.2.3, 8.4.3.1). Many industrial processes can also be electrified in the move away from fossil reductants and direct energy carriers (Chapter 11). The impact of electrification on electricity sector emissions will depend on whether electricity generation is based on fossil fuels in the absence of CCS or low carbon energy sources (Chapter 5).

At the same time, saving electricity in all sectors reduces the demand for electricity, thereby reducing mitigation potential of renewables and CCS. Demand side flexibility measures and electrification of vehicle fleets are supportive of more intermittent renewable energy supply options (Sections 6.3.7, 6.4.3.1 and 10.3.4). Production of maize, wheat, rice and fresh produce requires lower energy inputs on a life cycle basis than poultry, pork and ruminant based meats (Section 12.4) (Clark and Tilman 2017). They also require less land and area per kilocalorie or protein output (Clark and Tilman 2017; Poore and Nemecek 2018), and so replacing meat with these products makes land available for sequestration, biodiversity or other societal needs. However, production of co-products of the meat industry, such as leather and wool, is reduced, resulting in a need for substitutes. Further discussion and examples of cross-sectoral implications of mitigation, with respect to cost and potentials, are presented in Section 12.2. One final example on this topic included here is that of Circular Economy (Box 12.4).

Finally, in terms of competition among sectors for scarce resources, this issue is often considered in the assessments of mitigation potentials linked to bioenergy and diets (vegetable vs. animal food products), land use and water (Section 12.5, Cross-Working Group Box 3 in this Chapter) (*robust evidence, high agreement*). It is, however, also relevant elsewhere. Constraints have been identified in the supply of indium, tellurium, silver, lithium, nickel and platinum that are required for implementation of some specific renewable energy technologies (Watari et al. 2018; Moreau et al. 2019). Other studies have shown constraints in supply of cobalt, one of the key elements used in production of lithium-ion batteries, which has been assessed for mitigation potential in energy, transport and buildings sectors (Jaffe 2017; Olivetti et al. 2017) (*medium evidence, high agreement*), although alternatives to cobalt are being developed (Olivettie et al. 2017; Watari et al. 2018).

START BOX 12.4 HERE

Box 12.4: Circular Economy from a Cross-Sectoral Perspective

Circular economy approaches consider the entire life cycle of goods and services, and seek to design out waste and pollution, keep products and materials in use, and regenerate natural systems (The Ellen MacArthur Foundation 2013; CIRAIG 2015). The use of Circular Economy for rethinking how society's needs for goods and services is delivered in such a way as to minimise resource use and environmental impact and maximise societal benefit has been discussed elsewhere in this assessment report (Chapter 5 and Section 5.3.4). A wide range of potential application areas is identified, from food systems to bio-based products to plastics to metals and minerals to manufactured goods. Circular economy approaches are implicitly cross-sectoral, impacting the energy, industrial, AFOLU, waste and other sectors. They will have climate and non-climate co-benefits and trade-offs. The scientific literature mainly investigates incremental measures claiming but not demonstrating mitigation; highest mitigation potential is found in the industry, energy, and transport sector; mid-range potential in the waste and building sector; and lowest mitigation gains in agriculture (Cantzler et al. 2020). Circular economy thinking has been identified to support increased resilience to the physical effects of climate change and contribute to meeting other UN SDGs, notably SDG12 (responsible consumption and production) (The Ellen MacArthur Foundation 2019).

Circular economy approaches to deployment of low-carbon infrastructure have been suggested to be important to optimise resource use and mitigate environmental and societal impacts caused by extraction and manufacturing of composite and critical materials as well as infrastructure decommissioning (Jensen and Skelton 2018; Sica et al. 2018; Salim et al. 2019; Watari et al. 2019; Jensen et al. 2020; Mignacca et al. 2020). The circular carbon economy is an approach inspired by the circular economy principles that rely on a combination of technologies, including CCU, CCS and CDR, to enable transition pathways especially relevant in economies dependent on fossil fuel exports (Lee et al. 2017; Alshammari 2020; Morrow and Thompson 2020; Zakkour et al. 2020). The integration of circular economy and bioeconomy principles (See Cross-Working Group Box 3 in this Chapter on mitigation and adaption via the bioeconomy) is conceptualised in relation to policy development (European Commission 2018) as well as COVID-19 recovery strategies (Palahí et al. 2020)￼ emphasising the use of renewable energy sources and sustainable management of ecosystems with transformation of biological resources into food, feed, energy and biomaterials.

At this stage, however, there is no single globally agreement of how circular economy principles are best to be implemented, and differential government support for circular economy interventions is observed in different jurisdictions.

END BOX 12.4 HERE

12.6.1.3 Cross-sectoral considerations relating to emerging general purpose technologies

General Purpose Technologies (GPTs) include, but are not limited to, additive manufacturing, artificial intelligence, biotechnology, hydrogen, digitalisation, electrification, nanotechnology and robots (de Coninck et al. 2018). Many of the individual sectoral chapters have identified the roles that such technologies can have in supporting mitigation of GHG emissions. Section 16.2.2.3 presents an overview of the individual technologies and specific applications thereof.

In this chapter, which focuses on cross-sectoral implications of mitigation, it is highlighted that certain of these GPTs will find application across the sectors, and there will be synergies and trade-offs when utilising these technologies in more than sector. One example here is the use of hydrogen as an energy carrier, which when coupled with low carbon energy, has potential for driving mitigation in energy, industry, transport, and buildings. The increased uptake of hydrogen across the economy requires establishment of hydrogen production, transport and storage infrastructure which could simultaneously support multiple sectors, although there is the potential to utilise existing infrastructure in some parts of the world (Alanne and Cao 2017).

Box **12.5** provides for further details on hydrogen in the context of cross-sectoral mitigation specifically, while further details on the role of hydrogen in individual sectors are provided in Chapters 6, 8. 9, 10 and 11. In contrast, the benefits of digitalisation, which could potentially give rise to substantial energy savings across multiple sectors, need to be traded off against demand for electricity to operate consumer devices, data centres, and data networks. Measures are required to increase energy efficiency of these technologies (IEA 2017). Section 5.3.4.1 of this report provides further information on energy and emissions benefits and costs of digitalisation.

With respect to co-impacts of GPTs, the other focus of this chapter, it is highlighted that assessment of the environmental, social and economic implications of such technologies is challenging and context specific with multiple potential cross-sectoral linkages (de Coninck et al. 2018). Each GPT would need to be explored in context of what it is being used for, and potentially in the geographical context, in order to understand the co-impacts of its use.

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Box 12.5: Hydrogen in the context of cross-sectoral mitigation options

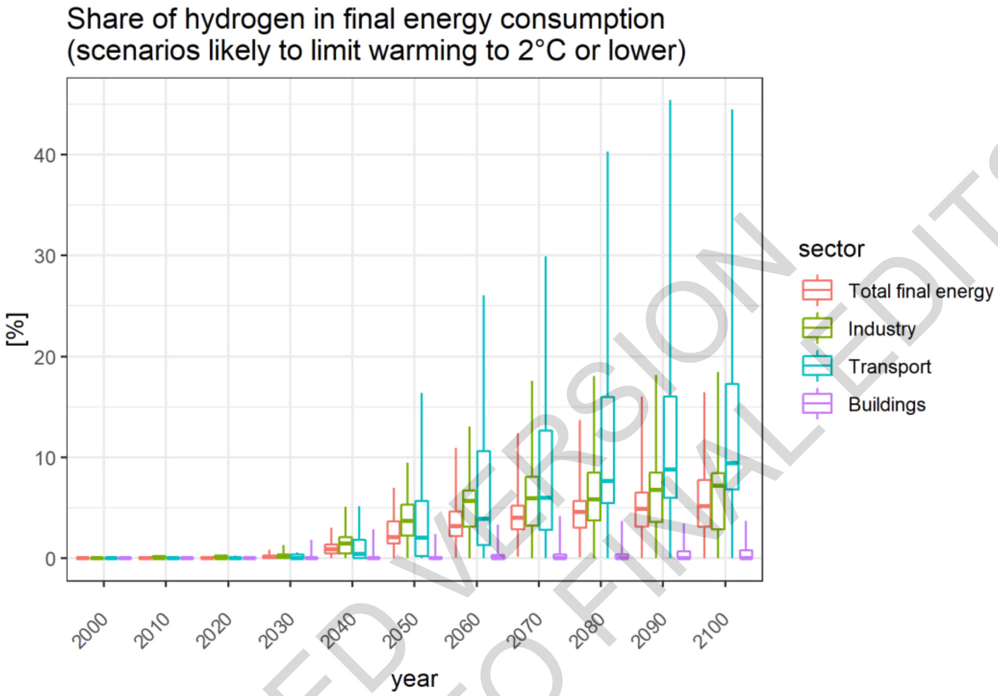
The interest in hydrogen as an intermediary energy carrier has grown rapidly in the years since the 5th Assessment Report of WGIII (AR5) was published. This is reflected in this WGIII assessment report, where the term ‘hydrogen’ is used more than five times more often than in AR5. In Chapter 6 of this report, it is shown that hydrogen can be produced with low carbon impact from fossil fuels (Section 6.4.2.6), renewable electricity and nuclear energy (Section 6.4.5.1), or biomass (Section 6.4.2.5). In the energy sector, hydrogen is one of the options for storage of energy in low-carbon electricity systems (Sections 6.4.4.1 and 6.6.2.2). But, also importantly, hydrogen can be produced to be used as a fuel for sectors that are hard-to-decarbonise; this is possible directly in the form of hydrogen, but also in the form of ammonia or other energy carriers (Section 6.4.5.1). In the transport sector, fuel cell engines (Section 10.3.3) running on hydrogen can become important, especially for heavy duty vehicles (Section 10.4.3). In the industry sector hydrogen already plays an important role in the chemical sector (for ammonia and methanol production (Box 11.1 in Chapter 11) and in the fuel sector (in oil refinery processes and for biofuel production (IEA 2019b). Beyond the production of ammonia and methanol for both established and novel applications, the largest potential industrial application for low-carbon hydrogen is seen in steelmaking (Section 11.4.1.1). Hydrogen and hydrogen-derivatives can play a further role as substitute energy carriers (Section 11.3.5) and for the production of intermediate chemical products such as methanol, ethanol and ethylene when combined with CCU (Section 11.3.6). For the building sector, the exploration of the usefulness of hydrogen is at an early stage (Box 9.4 in Chapter 9).

An overview report (IEA 2019b) already sees opportunities in 2030 for buildings, road freight and passenger vehicles. This report also suggests a high potential application in iron and steel production, aviation and maritime transport, and for electricity storage. Several industry roadmaps have been published that map out a possible role for hydrogen until 2050. The most well-known and ambitious is the roadmap by the Hydrogen Council (2017), which sketches a global scenario leading to 78 EJ hydrogen use in 2050, mainly for transport, industrial feedstock, industrial energy and to a lesser extent for buildings and power generation. Hydrogen makes up 18% of total final energy use in this vision. An analysis by IRENA on hydrogen from renewable sources comes to a substantially lower number: 8 EJ (excluding hydrogen use in power production and feedstock uses). On a regional level, most roadmaps and scenarios have been published for the European Union, e.g. by the Fuel Cell and Hydrogen Joint Undertaking (Blanco et al. 2018; EC 2018; FCH 2019; Navigant 2019). All these reports have scenario variants with hydrogen share in final energy use of 10% to over 20% by 2050.
When it comes to the production of low-carbon hydrogen, the focus of the attention is on production from electricity from renewable sources via electrolysis, so-called ‘green hydrogen’. However, ‘blue hydrogen’, produced out of natural gas with CCS is also often considered. Since a significantly increasing role for hydrogen would require considerable infrastructure investments and would affect existing trade flows in raw materials, governments have started to set up national hydrogen strategies, both potential exporting (e.g. Australia) and importing (e.g. Japan) countries (METI 2017; COAG Energy Council 2019).

As already reported in Chapter 6 (Section 6.2.4.1) production costs of green hydrogen are expected to come down from the current levels of above 100 USD MWh-1. Price expectations are: 40–60 € MWh-1 for both green and blue hydrogen production in the EU by 2050 (Navigant 2019) with production costs already being lower in North Africa; 42–87 USD MWh-1 for green hydrogen in 2030 and 20 – 41 USD MWh-1 in 2050 (BNEF 2020); 75 € MWh-1 in 2030 (Glenk and Reichelstein 2019). For fossil-based technologies combined with CCS, prices may range from 33 – 80 USD MWh-1 (Table 6.8 in Chapter 6,). Such prices can make hydrogen competitive for industrial feedstock applications, and probably for several transportation modes in combination with fuel cells, but without further incentives, not necessarily for stationary applications in the coming decades: wholesale natural gas prices are expected to range from 7–31 USD MWh-1 across regions and scenarios, according to the World Energy Outlook (IEA 2020a); coal prices mostly are even lower than natural gas prices (all fossil fuel prices refer to unabated technology and untaxed fuels). The evaluation of macro-economic impacts is relatively rare. A study by (Mayer et al. 2019b) indicated that a shift to hydrogen in iron and steel production would lead to regional GDP losses in the range of 0.4–2.7% in 2050 across EU+3 with some regions making gains under a low-cost electricity scenario.

The IAM scenarios imply a modest role played by hydrogen, with some scenarios featuring higher levels of penetration. The consumption of hydrogen is projected to increase by 2050 and onwards in scenarios likely limiting global warming to 2°C or below, and the median share of hydrogen in total final energy consumption is 2.1% in 2050 and 5.1% in 2100 (Box 12.4, Figure 1) (Numbers are based on the AR6 scenarios database.) There is large variety in hydrogen shares, but the values of 10% and more of final energy use that occur in many roadmaps are only rarely reached in the scenarios. Hydrogen is predominantly used in the industry and transportation sectors. In the scenarios, hydrogen is produced mostly by electrolysis and by biomass energy conversion with CCS (Box 12.5, Figure 1). Natural gas with CCS is expected to only play a modest role; here a distinct difference between the roadmaps quoted before and the IAM results is observed.

It is concluded that there is increasing confidence that hydrogen can play a significant role, especially in the transport sector and the industrial sector. However, there is much less agreement on timing and volumes, and there is also a range of perspectives on role of the various production methods of hydrogen.



Box 12.5, Figure 1 Fraction of hydrogen (H2, red) in total final energy consumption, and those for each sector. Hinges represent the interquartile ranges and whiskers extend to 5 and 95 percentiles.

END BOX 12.5 HERE

12.6.2 Sectoral policy interactions (synergies and trade-offs)

A taxonomy of policy types and attributes is provided by Section 13.6. In addition, the sectoral chapters provide an in-depth discussion of important mitigation policy issues such as policy overlaps, policy mixes, and policy interaction as well as policy design considerations and governance. The point of departure for the assessment in this chapter is a focus on cross-sectoral perspectives aiming at maximising policy synergies and minimising policy trade-offs.

Synergies and trade-offs resulting from mitigation policies are not clearly discernible from either sector-level studies or global and regional top-down studies. Rather, they would require a cross-sectoral integrated policy framework (von Stechow et al. 2015; Singh et al. 2019; Monier et al. 2018; Pardoe et al. 2018) or multiple-objective-multiple-impact policy assessment framework identifying key co-impacts and avoiding trade-offs (Ürge-Vorsatz et al. 2014) *(robust evidence, high agreement).*

Sectoral studies typically cover differentiated response measures while the IAM literature mostly uses uniform efficient market-based measures. This has important implications for understanding the differences in magnitude and distribution of mitigation costs and potentials of Section 12.2 (Rausch and Karplus 2014; Karplus et al. 2013). There is a comprehensive literature on the efficiency of uniform carbon pricing compared to sector-specific mitigation approaches, but relatively less literature on the distributional impacts of carbon taxes and measures to mitigate potential adverse distributional impacts (Åhman et al. 2017; Rausch and Reilly 2015; Mu et al. 2018; Wang et al. 2016b; Rausch and Karplus 2014). For example, in terms of cross-sectoral distributional implications, studies find negative competitiveness impacts for the energy intensive industries (Wang et al. 2016b; Åhman et al. 2017; Rausch and Karplus 2014). *(robust evidence, medium agreement)*

Strong inter-dependencies and cross-sectoral linkages create both opportunities for synergies and the need to address trade-offs. This calls for coordinated sectoral approaches to climate change mitigation policies that mainstream these interactions (Pardoe et al. 2018). Such an approach is also called for in the context of cross-sectoral interactions of adaptation and mitigation measures, examples are in the agriculture, biodiversity, forests, urban, and water sectors (Di Gregorio et al. 2017; Arent et al. 2014; Berry et al. 2015). Integrated planning and cross-sectoral alignment of climate change policies are particularly evident in developing countries' NDCs pledged under the Paris Agreement, where key priority sectors such as agriculture and energy are closely aligned between the proposed mitigation and adaptation actions in the context of sustainable development and the SDGs. An example is the integration between smart agriculture and low carbon energy (Antwi-Agyei et al. 2018; England et al. 2018). Yet, there appear to be significant challenges relating to institutional capacity and resources to coordinate and implement such cross-sectoral policy alignment, particularly in developing country contexts (Antwi-Agyei et al. 2018) *(robust evidence, high agreement).*

Another dimension of climate change policy interactions in the literature is related to trade-offs and synergies between climate change mitigation and other societal objectives. For example, in mitigation policies related to energy, trade-offs and synergies between universal electricity access and climate change mitigation would call for complementary policies such as pro-poor tariffs, fuel subsidies, and broadly integrated policy packages (Dagnachew et al. 2018). In agriculture and forestry, research suggests that integrated policy programs enhance mitigation potentials across the land-use-agriculture-forestry nexus and lead to synergies and positive spill-overs (Galik et al. 2019). To maximise synergies and deal with trade-offs in such a cross-sectoral context, evidence-based/informed and holistic policy analysis approaches like nexus approaches and multi-target back-casting approaches that take into account unanticipated outcomes and indirect consequences would be needed (Klausbruckner et al. 2016; van der Voorn et al. 2020; Hoff et al. 2019; see Box 12.6) *(robust evidence, high agreement).*

The consequences of large-scale land-based mitigation for food security, biodiversity, (Dasgupta 2021) the state of soil, water resources, etc. can be significant depending on many factors, such as economic development (including distributional aspects), international trade patterns, agronomic development, diets, land use governance and policy design, and not least climate change itself (Fujimori et al. 2018; Hasegawa et al. 2018; Van Meijl et al. 2018; Winchester and Reilly 2015). Policies and regulations that address other aspects apart from climate change can indirectly influence the attractiveness of land-based mitigation options. For example, farmers may find it attractive to shift from annual food/feed crops to perennial grasses and short rotation woody crops (suitable for bioenergy) if the previous land uses become increasingly restricted due to impacts on groundwater quality and eutrophication of water bodies (Sections 12.4 and 12.5) (*robust evidence, medium agreement).*

Finally, there are knowledge gaps in the literature particularly in relation to policy scalability and in relation to the extent and magnitude of policy interactions when scaling the policy to a level consistent with low GHG emissions pathways such as 2ºC and 1.5ºC.

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Box 12.6: Case Study, Sahara Forest Project in Aqaba, Jordan

Nexus Framing

Shifting to renewable (in particular solar) energy reduces dependency on fossil fuel imports and greenhouse gas emissions, which is crucial for mitigating climate change. Employing renewable energy for desalination of seawater and for cooling of greenhouses in integrated production systems can enhance water availability, increase crop productivity and generate co-products and co-benefits (e.g., algae, fish, dryland restoration, greening of the desert).

Nexus Opportunities

The Sahara Forest project integrated production system uses amply available natural resources, namely solar energy and seawater, for improving water availability and agricultural/biomass production, while simultaneously providing new employment opportunities. Using hydroponic systems and humidity in the air, water needs for food production are 50% lower compared to other greenhouses.

Technical and Economic Nexus Solutions

Several major technologies are combined in the Sahara Forest Project, namely electricity production through the use of solar power (PV or CSP), freshwater production through seawater desalination using renewable energy, seawater-cooled greenhouses for food production, and outdoor revegetation using run-off from the greenhouses.

Stakeholders Involved

The key stakeholders which benefit from such an integrated production system are from the water sector which urgently requires an augmentation of irrigation (and other) water, as well as from the agricultural sector, which relies on the additional desalinated water to maintain and increase agricultural production. The project also involves public and private sector partners from Jordan and abroad, with little engagement of civil society so far.

Framework Conditions

The Sahara Forest Project has been implemented at pilot scale so far, including the first pilot with one hectare and one greenhouse pilot in Qatar and a larger “launch station” with three hectares and two greenhouses in Jordan). These pilots have been funded by international organisations such as the Norwegian Ministry of Climate and Environment, Norwegian Ministry of Foreign Affairs and the European Union. Alignment with national policies, institutions and funding as well as upscaling of the project is underway or planned.

Monitoring and Evaluation and Next Steps

The multi-sectoral planning and investments that are needed to up-scale the project require cooperation among the water, agriculture, and energy sectors and an active involvement of local actors, private companies, and investors. These cooperation and involvement mechanisms are currently being established in Jordan. Given the emphasis on the economic value of the project, public-private partnerships are considered as the appropriate business and governance model, when the project is up-scaled. Scenarios for upscaling (seawater use primarily in low lying areas close to the sea, to avoid energy-intensive pumping) include 50MW of CSP, 50 hectares of greenhouses, which would produce 34,000 tons of vegetables annually, provide employment for over 800 people, and sequester more than 8,000 tons of CO2 annually.

Source: SFP Foundation; Hoff et al. 2019

END BOX 12.6 HERE

12.6.3 International trade spill-over effects and competitiveness

International spill-overs of mitigation policies are effects that carbon-abatement measures implemented in one country have on sectors in other countries. These effects include 1) carbon leakage in manufacture, 2) the effects on energy trade flows and incomes related to fossil fuel exports from major exporters, 3) technology and knowledge spill-overs; 4) transfer of norms and preferences via various approaches to establish sustainability requirements on traded goods, e.g., EU-RED and environmental labelling systems to guide consumer choices *(robust evidence, medium agreement)*. This section focuses on cross-sectoral aspects of international spill-overs related to the first two effects.

12.6.3.1 Cross-sectoral aspects of carbon leakage

Carbon leakage occurs when mitigation measures implemented in one country/sector lead to the rise in emissions in other countries/sectors. Three types of spill-overs are possible: 1) domestic cross-sectoral spill-overs when mitigation policy in one sector leads to the re-allocation of labour and capital towards the other sectors of the same country; 2) international spill-overs within a single sector when mitigation policy leads to substitution of domestic production of carbon-intensive goods with their imports from abroad; 3) international cross-sectoral spill-overs when mitigation policy in one sector in one country leads to the rise in emissions in other sectors in other countries. While the first two are described in Section 13.6, this section focuses on the third. Though some papers address this type of leakage, there is still significant lack of knowledge on this topic.

One possible channel of cross-sectoral international carbon leakage is through global value chains. Mitigation policy in one country not only leads to shifts in competitiveness across industries producing final goods but also across those producing raw materials and intermediary goods all over the world.

This type of leakage is especially important because the countries that provide basic materials are usually emerging or developing economies, many of which have no or limited regulation of GHG emissions. For this reason, foreign direct investment in developing economies usually leads to an increase in emissions (Bakhsh et al. 2017; Shahbaz et al. 2015; Kivyiro and Arminen 2014): in case of basic materials the effect of expansion of economic activity on emissions exceeds the effect of technological spill-overs, while for developed countries the effect is opposite (Pazienza 2019; Shahbaz et al. 2015). Meng et al. (2018) calculated that environmental costs for generating one unit of GDP through international trade was 1.4 times higher than that through domestic production in 1995. By 2009, this difference increased to 1.8 times. Carbon leakage due to the differences in environmental regulation was the main driver of this increase.

In order to address emissions leakage through global value chains, Liu and Fan (2017) propose the value-added-based emissions accounting principle, that makes possible to account for GHG emissions within the context of the economic benefit principle. Davis et al. (2011) notice that the analysis of value chains gives an opportunity to find the point where regulation would be the most efficient and the least vulnerable to leakage. For instance, transaction costs of global climate policy and the risks of leakage may be reduced if emissions are regulated at the extraction stage as there are far fewer agents involved in this process than in burning of fossil fuels or consumption of energy-intensive goods. Li et al. (2020)

calls for coordinated efforts to reduce emissions in trade flows in pairs of the economies with the highest leakage such as China and the United States, China and Germany, China and Japan, Russia and Germany.

Unfortunately, these proposals either face difficulties in collection and verification of data on emissions along value chains or require a high level of international cooperation which is hardly achievable at the moment. (Neuhoff et al. 2016; Pollitt et al. 2020) focus on the regulation of emissions embodied in global value chains through national policy instruments. They propose implementation of a charge on consumption of imported basic materials into the European emissions trading system. Such a charge, equivalent to around €80 tCO_2^{-1}, could reduce the EU's total CO_2 emissions by up to 10% by 2050 (Pollitt et al. 2020) without significant effects on competitiveness. This proposal is very close to border carbon adjustment introduced in the EU and described in more detail in Sections 13.2 and 13.6.

Cross-sectoral effects of carbon leakage also occur through the multiplier effect, when the mitigation policy in any sector in country A leads to the increase of relative competitiveness and therefore production of the same sector in country B that automatically leads to the expansion of economic activity in other sectors of country B. This expansion may in turn lead to the rise of production and emissions in country A as a result of feedback effects. These spill-overs should be taken into consideration while designing climate policy, along with potential synergies that may appear due to joint efforts. However, the scale of these effects with regards to leakage shouldn't be overestimated. Even for intrasectoral leakage, many *ex-ante* modelling studies generally suggest limited carbon leakage rates (Chapter 13). Intersectoral leakage should be even less significant. Interregional spill-over and feedback effects are well-studied in China (Zhang 2017; Ning et al. 2019). Even within a single country, interregional spill-over effects are much lower than intraregional effects, and feedback effects are even less intense. Cross-sectoral spill-overs across national borders as a result of mitigation policy should be even smaller, although these are less well-studied. In future, if the differences in carbon price between regions increase, leakage through cross-sectoral multipliers may play a more important role.

Another important cross-sectoral aspect of carbon leakage concerns the transport sector. If mitigation policy leads to the substitution of domestic carbon-intensive production with imports, one of the side effects of this substitution is the rise of emissions from transportation of imported goods. International transport is responsible for about a third of worldwide trade-related emissions, and over 75 percent of emissions for major manufacturing categories (Cristea et al. 2013). Carbon leakage would potentially increase the emissions from transportation significantly as the trade of major consuming economies of the EU and US would shift towards distant trading partners in East and South Asia. Meng et al. (2018) consider more distant transportation as one of the major contributors to the rise in emissions embodied in international trade from 1995 to 2009.

Emissions leakage due to international trade, investment and value chains is a significant obstacle to more ambitious climate policies in many regions. However, it doesn't mean that disruption of trade would reduce global emissions. Zhang et al. (2020) show that deglobalisation and the drop in international trade may result in emissions reductions in the short term, but in the longer term it will make each country build more complete industrial systems to satisfy their final demand, although they have comparative disadvantages in some production stages. As a result, emissions would increase. According to Zhang et al. (2020) for China, the decrease of the degree of global value chain participation (which ranges from 0 to 1) by 0.1 would lead to an increase in gross carbon intensity of China's exports of 11.7%. On distributional implications, Parrado and De Cian (2014) report that trade-driven spill-overs effects transmitted through imports of materials and equipment result in significant inter-sectoral distributional effects with some sectors witnessing substantial expansion in activity and emissions and others witnessing a decline in activities and emissions.

It should also be mentioned that international trade leads to important knowledge and technology spill-overs (Sections 16.3 and 16.5) and is critically important for achieving other Sustainable Development Goals (Section 12.6.1). Any policies imposing additional barriers to international trade should be therefore implemented with great caution and require comprehensive evaluation of various economic, social and environmental effects.

12.6.3.2 The spill-over effects on the energy sector

Cross-sectoral trade-related spill-overs of mitigation policies include their effect on energy prices. Other things being equal, regulation of emissions of industrial producers decreases the demand for fossil fuels that would reduce prices and encourage the rise of fossil fuel consumption in regions with no or weaker climate policies *(robust evidence, medium agreement).*

Arroyo-Currás et al. (2015) study the energy channel of carbon leakage with the REMIND IAM of the global economy. They come to the conclusion that the leakage rate through the energy channel is less than 16% of the emission reductions of regions who introduce climate policies first. This result doesn't differ much for different sizes and compositions of the early mover coalition.

Bauer et al. (2015) built a multi-model scenario ensemble for the analysis of energy-related spill-overs of mitigation policies and reveal huge uncertainty: energy-related carbon leakage rates vary from negative values to 50%, primarily depending on the trends in inter-fuel substitution.

Another kind of spill-over in energy sector concerns the "green paradox"; announcement of future climate policies causes an increase in production and trade in fossil-fuels in the short term (Jensen et al. 2015; Kotlikoff et al. 2016). The delayed carbon tax should therefore be higher than an immediately implemented carbon tax in order to achieve the same temperature target (van der Ploeg 2016). Studies also make a distinction between a "weak" and "strong" green paradox (Gerlagh 2011). The former refers to a short-term rise in emissions in response to climate policy, while the latter refers to rising cumulative damage.

The green paradox may work in different ways for different kinds of fossil fuels. For instance, Coulomb and Henriet (2018) show that climate policies in the transport and power-generation sectors increase the discounted profits of the owners of conventional oil and gas, compared to the no-regulation baseline, but will decrease these profits for coal and unconventional oil and gas producers.

Many studies also distinguish different policy measures by the scale of green paradox they provide. The immediate carbon tax is the first-best instrument from the perspective of the global welfare. Delayed carbon tax leads to some green paradox but it is less than in the case of the support of renewables (Michielsen 2014; van der Ploeg and Rezai 2019). With respect to the latter, support of renewable electricity has a lower green paradox than the support of biofuels (Gronwald et al. 2017; Michielsen 2014). The existence of the green paradox is an additional argument in favour of more decisive climate policy now: any postponements will lead to additional consumption of fossil fuels and consequently the need for more ambitious and costly efforts in future.

The effect of fossil fuel production expansion as a result of anticipated climate policy may be compensated by the effect of divestment. Delayed climate policy creates incentives for investors to divest from fossil fuels. Bauer et al. (2018) show that this divestment effect is stronger and thus announcing of climate policies leads to the reduction of energy-related emissions.

The implication of the effects of mitigation policies through the energy related spill-overs channel is of particular significance to oil-exporting countries *(medium evidence, medium agreement)*. Emissions reduction-measures lead to the decreasing demand for fossil fuels and consequently to the decrease in its exports from major oil- and gas- exporting countries. The case of Russia is one of the most illustrative. Makarov et al. (2020) show that the fulfilment of Paris Agreement parties of their NDCs would lead to 25% reduction of Russia's energy exports by 2030 with significant reduction of its economic growth rates. At the same time, the domestic consumption of fossil fuels is anticipated to increase in response to the drop of external demand that would provoke carbon leakage (Orlov and Aaheim 2017). Such spill-overs demonstrate the need for the dialogue between exporters and importers of fossil fuels while implementing the mitigation policies.

12.6.4 Implications of finance for cross-sectoral mitigation synergies and trade-offs

Finance is a principal enabler of GHG mitigation and an essential component of countries' NDC packages submitted under the Paris climate agreement (UNFCCC 2016). The assessment of investment requirements for mitigation along with their financing at sectoral levels are addressed in detail by sectoral chapters while the assessment of financial sources, instruments, and the overall mitigation financing gap is addressed by Chapter 15 (Sections 15.3, 15.4, and 15.5). The focus in this chapter with respect to finance is on the scope and potential for financing integrated solutions that create synergies between and among sectors.

Cross-sectoral considerations in mitigation finance are critical for the effectiveness of mitigation action as well as for balancing the often conflicting social, developmental and environmental policy goals at the sectoral level. True measures of mitigation policy impacts and hence plans for resource mobilisation that properly address costs and benefits cannot be developed in isolation of their cross-sectoral implications. Unaddressed cross-sectoral coordination and interdependency issues are identified as major constraints in raising the necessary financial resources for mitigation in a number of countries (Bazilian et al. 2011; Welsch et al. 2014; Hoff et al. 2019a).

Integrated financial solutions to leverage synergies between sectors, as opposed to purely sector-based financing at international, national, and local levels are needed to scale up GHG mitigation potentials. At the international level, Finance from Multilateral Development Banks (MDBs) is a major source of GHG mitigation finance in developing countries (World Bank Group 2015; Ha et al. 2016; Bhattacharya et al. 2016, 2018) *(medium evidence, medium agreement).* In 2018, MDBs reported a total of USD 30,165 million in financial commitments to climate change mitigation, with 71% of total mitigation finance being committed through investment loans and the rest in the form of equity, guarantees, and other instruments. GHG reduction activities eligible for MDB finance are limited to those compatible with low-emission pathways recognising the importance of long-term structural changes, such as the shift in energy production to low-carbon energy technologies and the modal shift to low-carbon modes of transport leveraging both greenfield and energy efficiency projects. Sector-wise, the MDBs mitigation finance for 2018 is allocated to renewable energy (29%), transport (18%), energy efficiency (18%), lower-carbon and efficient energy generation (7%), agriculture, forestry and land use (8%), waste and waste-water (8%), and other sectors (12%) (MDB 2019). Unfortunately, due to institutional and incentives issues MDBs finance has mostly focused on sectoral solutions and has not been able to properly leverage cross-sectoral synergies. At the national level, applied research has shown that integrated modelling of land, energy and water resources not only has the potential to identify superior solutions, but also reveals important differences in terms of investment requirements and required financing arrangements compared to the traditional sectoral financing toolkits (Welsch et al. 2014). Agriculture, forestry, nature-based-solutions (NBS) and other forms of land use are promising sectors for leveraging financing solutions to scale up GHG mitigation efforts (Section 15.4). Moving to more productive and resilient forms of land use is a complex task given the crosscutting nature of land-use that necessarily results in apparent trade-offs between mitigation, adaptation, and development objectives. Finance is one area to manage these trade-offs where there may be opportunities to redirect the hundreds of billions spent annually on land use around the world towards green activities, without sacrificing either productivity or economic development (Falconer et al. 2015). Nonetheless, that would require active public support in design of land use mitigation and adaptation strategies, coordination between public and private instruments across land-use sectors, and leveraging of policy and financial instruments to redirect finance toward greener land-use practices (*limited evidence, medium agreement*). For example, the Welsch et al. (2014) study on Mauritius shows that the promotion of a local biofuel industry from sugar cane could be economically favourable in the absence of water constraints, leading to a reduction in petroleum imports and GHG emissions while enhancing energy security. Yet, under a water-constrained scenario as a result of climate change, the need for additional energy to expand irrigation to previously rain-fed sugar plantations and to power desalination plants yields the opposite result in terms of GHG emissions and energy costs, making biofuels a sub-optimal option, and negatively affects their economics and the prospects for financing.

At the local level, integrated planning and financing are needed to achieve more sustainable outcomes. For example, at a city level integration is needed across sectors such as transport, energy systems, buildings, sewage and solid waste to optimise emissions footprints. How a city is designed will affect transportation demands, which makes it either more or less difficult to implement efficient public transportation, leading in turn to more or less emissions. Under such cases, solutions in terms of public and private investment paths and financing policies based on purely internal sector considerations are bound to cause adverse impacts on other sectors and poor overall outcomes (Gouldson et al. 2016).

Availability and access to finance are among the major barriers to GHG emissions mitigation across various sectors and technology options *(robust evidence, high agreement).* Resource maturity mismatches and risk exposure are two main factors limiting ability of commercial banks and other private lenders to contribute to green finance (Mazzucato and Semieniuk 2018). At all levels, mobilising the necessary resources to leverage cross-sectoral mitigation synergies would require the combination of public and private financial sources (Jensen and Dowlatabadi 2018). Traditional public financing would be required to synergise mitigation across sectors where the risk-return and time profiles of investment are not sufficiently attractive for the business sector. Over the years, private development financing through public-private partnerships (PPP) and other related variants has been a growing source of finance to leverage cross-sectoral synergies and manage trade-offs (Ishiwatari et al. 2019; Attridge and Engen 2019; Anbumozhi and Timilsina 2018). Promoting such blended approaches to finance along with result-based financing architectures to strengthen delivery institutions are advocated as effective means to mainstream cross-sectoral mitigation finance (Ishiwatari et al. 2019; Attridge and Engen 2019) *(limited evidence, high agreement)*. The World Bank group and the International Financial Corporation (IFC) have used the blended finance results-based approach to climate financing that addresses institutional, infrastructure, and service needs across sectors targeting developing countries and marginalised communities (GPRBA 2019; IDA 2019).

12.7 Knowledge Gaps

Finally, the literature review and analysis in Chapter 12 has taken account of the post-AR5 literature available and accessible to the chapter authors. Nonetheless, the assessment of the chapter is incomplete without mentioning knowledge gaps encountered during the assessment. These knowledge gaps include:

1) Interactions (synergies and trade-offs) between different CDR methods when deployed together are under-researched.

- Co-benefits and trade-offs with biodiversity and ecosystem services associated with the implementation of CDR methods.
- Constraining technical costs and potentials for CDR methods to define realistically achievable costs and potentials. Such research is useful for improving the representation of CDR methods in IAMs and country-level mitigation pathway modelling.

2) More work is required on how framing and communication of mitigation actions in terms of mitigation versus co-benefits potential affects public support in different contexts.

3) Additional research work is required to determine the cross-sectoral mitigation potential of emerging General Purpose Technologies.

4) Lack of literature on mitigation finance frameworks promoting cross-sectoral mitigation linkages.

5) Additional research is needed to better quantify the net GHG emissions and co-benefits and adverse effects of emerging food technologies.

- Research in social and behavioural sciences should invest in assessing effectiveness of instrument aiming at shifting food choices in different national contexts.
- A better evidence basis is required to understand synergistic effects of policies in food system policy packages.

6) Literature on regional/global mitigation potential of biomass production systems that are strategically deployed in agriculture/forestry landscapes, to achieve specific co-benefits.

7) Knowledge on land occupation and associated co-benefits and adverse side-effects from large-scale deployment of non-AFOLU mitigation options, and how such options can be integrated with agriculture and forestry to maximise synergies and minimise trade-offs.

Frequently Asked Questions (FAQs)

FAQ 12.1 How could new technologies to remove carbon dioxide from the atmosphere contribute to climate change mitigation?

Limiting the increase in warming to well below 2°C, and achieving net zero CO_2 or GHG emissions, will require anthropogenic CO_2 removal (CDR) from the atmosphere.

The CDR methods studied so far have different removal potentials, costs, co-benefits and side effects. Some biological methods for achieving CDR, like A/R or wetland restoration, have long been practiced. If implemented well, these practices can provide a range of co-benefits, but they can also have adverse side effects such as biodiversity loss or food price increases. Other chemical and geochemical approaches to CDR include Direct Air Carbon Capture and Storage (DACCS), Enhanced Weathering or Ocean Alkalinity Enhancement. They are generally less vulnerable to reversal than biological methods.

DACCS uses chemicals that bind to CO_2 directly from the air; the CO_2 is then removed from the sorbent and stored underground or mineralised. Enhanced Weathering involves the mining of rocks containing minerals that naturally absorb CO_2 from the atmosphere over geological timescales, which are crushed to increase the surface area and spread on soils (or elsewhere) where they absorb atmospheric CO_2. Ocean Alkalinity Enhancement involves the extraction, processing, and dissolution of minerals and addition to the ocean where they enhance sequestration of CO_2 as bicarbonate and carbonate ions in the ocean.

FAQ 12.2 Why is it important to assess mitigation measures from a systemic perspective, rather than only looking at their potential to reduce Greenhouse Gas (GHG) emissions?

Mitigation measures do not only reduce GHGs, but have wider impacts. They can result in decreases or increases in GHG emissions in another sector or part of the value chain to where they are applied. They can have wider environmental (e.g., air and water pollution, biodiversity), social (e.g., employment creation, health) and economic (e.g., growth, investment) co-benefits or adverse side effects. Mitigation and adaptation can also be linked. Taking these considerations into account can help to enhance the benefits of mitigation action, and avoid unintended consequences, as well as provide a stronger case for achieving political and societal support and raising the finances required for implementation.

FAQ 12.3 Why do we need a food systems approach for assessing GHG emissions and mitigation opportunities from food systems?

Activities associated with the food system caused about one-third of total anthropogenic GHG emissions in 2015, distributed across all sectors. Agriculture and fisheries produce crops and animal-source food, which are partly processed in the food industry, packed, distributed, retailed, cooked, and finally eaten. Each step is associated with resource use, waste generation, and GHG emissions.

A food systems approach helps identify critical areas as well as novel and alternative approaches to mitigation on both supply side and demand side of the food system. But complex co-impacts need to be considered and mitigation measures tailored to the specific context. International cooperation and governance of global food trade can support both mitigation and adaptation.

There is large scope for emissions reduction in both cropland and grazing production, and also in food processing, storage and distribution. Emerging options such as plant-based alternatives to animal food products and food from cellular agriculture are receiving increasing attention, but their mitigation potential is still uncertain and depends on the GHG intensity of associated energy systems due to relatively high energy needs. Diet changes can reduce GHG emissions and also improve health in groups with excess consumption of calories and animal food products, which is mainly prevalent in developed countries. Reductions in food loss and waste can help reduce GHG emissions further.

Recommendations of buying local food and avoiding packaging can contribute to reducing GHG emissions but should not be generalised as trade-offs exist with food waste, GHG footprint at farm gate, and accessibility to diverse healthy diets.

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