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Abstract

This paper investigates the potential role and contribution of carbon pricing in transforming emission pathways towards net zero GHG emissions. It reviews carbon pricing's impacts, overall and in the electricity sector in selected jurisdictions to date. The paper also analyses the current and potential application of emissions pricing (e.g. emissions trading schemes or carbon taxes) in food systems. The analysis finds that carbon pricing could contribute to net zero pathways alongside other policies, yet price levels and coverage to date have been too low to reduce emissions in line with the Paris Agreement's goals. Carbon pricing's contribution to net zero pathways could be further strengthened, including by incentivising demand-side shifts, sequencing policies and enhancing international carbon pricing collaboration. Applying emissions pricing in food systems faces significant short-term technical, methodological, and political barriers and could have just transition implications but reducing emissions from food systems could also lead to many co-benefits.

Keywords : carbon pricing, carbon tax, emissions trading system, ETS, net zero, climate change, climate mitigation, greenhouse gas emissions, transformative change, revenue recycling, just transition, policy packages, food systems, agriculture, supply-side, demand-side.

JEL Codes : H23, Q52, Q54, Q56, Q58.

Résumé

Ce document analyse le rôle et la contribution potentiels de la tarification du carbone dans la transformation des profils d'évolution des émissions de gaz à effet de serre en vue d'atteindre zéro émission nette. Il passe en revue les effets de cette tarification de manière globale et dans le secteur de l'électricité au sein de juridictions sélectionnées, sur la base des données disponibles jusqu'à présent. Le document analyse également l'application effective et potentielle de la tarification des émissions (sous forme de systèmes d'échange de quotas d'émission ou de taxes carbone, par exemple) dans les systèmes alimentaires. Il ressort de l'analyse que, conjuguée à d'autres mesures, la tarification du carbone pourrait contribuer à des profils d'évolution orientés vers zéro émission nette, mais que ses niveaux de prix et son champ d'application sont pour l'instant insuffisants pour susciter une réduction des émissions compatible avec la réalisation des objectifs de l'Accord de Paris. La contribution de la tarification du carbone à des profils d'évolution permettant d'atteindre zéro émission nette pourrait être renforcée, à travers la mise en place d'incitations pour agir sur la demande, l'échelonnement des politiques publiques et l'amélioration de la collaboration internationale en matière de tarification du carbone. L'application de la tarification du carbone dans les systèmes alimentaires se heurte dans l'immédiat à d'importants obstacles techniques, méthodologiques et politiques et pourrait avoir des conséquences dans la perspective d'une transition juste, mais la réduction des émissions émanant des systèmes alimentaires pourrait aussi avoir d'importantes retombées bénéfiques.

Mots-clés : tarification du carbone, taxe carbone, système d'échange de quotas d'émission, SEQE, zéro émission nette, changement climatique, atténuation du changement climatique, émissions de gaz à effet de serre, changements transformateurs, réaffectation des recettes, transition juste, panoplies de politiques publiques, systèmes alimentaires, agriculture, action sur l'offre, action sur la demande.

Codes JEL : H23, Q52, Q54, Q56, Q58

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Executive summary

A transformative change in greenhouse gas (GHG) emission pathways is needed to reach net zero by 2050 and meet the temperature goals of the Paris Agreement. Despite the strengthening of many national climate change policies and targets (e.g. towards net zero emissions), global GHG emissions are still projected to lead to an average global temperature rise above 1.5°C. A transformative change, encompassing both demand- and supply-side shifts, will be needed to deliver the pace and scale of emission reductions required to limit global warming to 1.5°C.

Carbon pricing could potentially play an important role in pathways to net zero GHG emissions, but, overall, price levels and coverage have been too low to date to do so. In 2022, 68 carbon pricing schemes[1] existed globally, covering around 23% of global GHG emissions, with most prices below EUR 50/tCO_2. In 2021-2022, ETS and carbon tax price levels hit record highs in many jurisdictions. Despite this, GHG emissions covered by carbon pricing schemes have, on average, fallen by 0-2% per year since the 1990s. To limit global warming to the Paris Agreement goal of 1.5°C, annual average GHG emissions reductions of almost 8% globally are needed between 2020-30. This paper aims to improve understanding of carbon pricing's potential role and contribution to transforming emission pathways to net zero GHG emissions, by drawing on current experiences. This paper also explores the potential application of emissions pricing in food systems and identifies data gaps and questions to guide future research.

Disentangling the impact of carbon pricing on sectoral emission pathways is challenging, but available evidence indicates that carbon pricing alone has so far been insufficient to reach net zero GHG emissions. Evidence from the electricity sector in the EU, New Zealand, and California suggest that carbon pricing, related revenues and other policies have together contributed to reducing GHG emissions and increasing the share of renewables in the electricity mix.

Pathways to net zero GHG emissions need to be carefully designed and take just transition aspects into account. Rapid and broad transformative change and associated emission reductions could have some short-term negative social impacts. While estimates indicate an overall net gain in employment from a transition to net zero, new jobs may not necessarily occur in the places where jobs are lost, and new jobs may also require new skills. Targeted and proactive measures, based on social dialogue, investments, and social protection, will be needed to help affected workers, communities, and regions.

Comprehensive policy packages that simultaneously address the supply- and demand-side could create enabling conditions for a just transition to net zero. Although challenging, significant GHG emissions reduction potential exists in addressing demand-side changes alongside supply-side shifts. Carbon pricing could play an important role in incentivising demand-side shifts as part of a wider policy package that carefully uses carbon pricing revenues. For example, carbon pricing could help reduce consumption levels of carbon-intensive goods, encourage a shift to less carbon-intensive goods, or improve the carbon intensity of existing goods.

[1] Carbon taxes and Emissions Trading Schemes (ETS)

Sequencing the deployment of polices could help to reduce or remove potential barriers and increase the effectiveness of carbon pricing. For example, deploying green industrial policies (e.g. research and development support) before carbon pricing could encourage greater use of low-carbon alternatives. Once these alternatives are commercially available, carbon pricing could discourage the use of conventional, high-emitting goods or services. In addition, enhanced international co-operation and collaboration could incentivise more ambitious carbon pricing by helping to overcome obstacles and facilitate exchanges of good practices. For example, enhanced international collaboration such as, linking carbon pricing schemes, could help improve cost efficiency and address competitiveness concerns.

Reducing food systems' emissions is vital in pathways to achieve net zero GHG emissions, yet decarbonising food systems is far from straightforward, despite significant potential. This is because emissions from an individual food product can be spread across different countries, sectors, and gases. There are GHG emissions differences between the production of different food products and within production of the same food product. There are also differences in the feasibility and cost of reducing emissions in different food system parts. Reducing some food systems emissions, such as biogenic CH_4 from enteric fermentation, is difficult given the lack of commercially available means of doing so, whereas reducing other types of emissions, such as from synthetic fertiliser or manure may be more feasible. Moreover, significant support to high-emitting food products continues to be provided by many countries. Targeting emissions reductions from the most GHG-intensive food items could help to substantially decrease the overall emissions from food systems.

This report explores the potential for emissions pricing policies to support emissions reductions in food systems. To help reduce GHG emissions from food systems, an emissions pricing scheme covering GHG emissions pertinent to food systems (i.e. CO_2, CH_4, N_2O) could be applied via the polluter-pays (ETS or carbon taxes) or the beneficiary-pays principle (abatement payments and offsets). Such a scheme, depending on its design, could help to encourage production of individual food products to become more efficient and/or encourage shifts to foods with a lower GHG intensity.[2] Yet to date, emissions pricing is only marginally used in global food systems, covering a small part of GHG emissions.

Implementing emissions pricing in food systems has significant short-term methodological, technical, and political barriers. High variability in the emissions intensity between and within different food products, and across geographies, makes it methodologically challenging to provide an accurate price signal. In particular, given the many steps from farm to fork, and the many actors involved, it can be technically challenging to monitor, report and verify emissions for a food product across its lifecycle. There are also significant political challenges associated with policies that could increase food prices, as illustrated by the political difficulties triggered by the high food inflation rate in 2021 and 2022. There could also be negative just transition implications of emissions pricing in food systems, e.g. reductions in agricultural employment in specific sub-sectors (e.g. livestock production), for farmer income, and for food security. Implementing emissions pricing in food systems would require carefully designed policy packages to meet multiple policy objectives while addressing the triple challenge facing food systems of providing food security and nutrition for a growing global population, maintaining livelihoods in the food chain and contributing to environmental sustainability. Given the challenges of applying emissions pricing in food systems, some countries are likely to prioritise the use of non-pricing policies in this area.

Nevertheless, reducing GHG emissions from food systems could have many co-benefits and potential policy avenues on the supply- and the demand-side could be explored. These co-benefits could include positive outcomes for biodiversity, water demand, and local environmental pollution. Further exploration is needed of the potential applicability, feasibility, and political acceptability of different means to reduce GHG emissions from food systems, and the role of GHG emissions pricing approaches in this.

[2] Other non-pricing policies could also be used to address emissions from food systems, but an exploration of these is outside the scope of this analysis.

1. Introduction

To meet the temperature goals of the Paris Agreement, greenhouse gas (GHG) emission levels need to drop significantly over the next decade. However, global GHG emissions have continued to increase, with some brief exceptions (e.g. at the start of the COVID-19 pandemic). Although countries continue to improve their climate change policies, current efforts remain insufficient to reach the long-term goals of the Paris Agreement (UNFCCC, 2022[1]). Rapid and deep emission reductions across all sectors and systems, including energy, transport, industry and food, will be required to limit global warming to 1.5°C. Both demand- and supply-side shifts will be needed. Such rapid and broad transformative changes are likely to have distributional impacts and just transition considerations need to be kept in mind when designing pathways to net zero GHG emissions.

Carbon pricing, covering both carbon taxes and emissions trading schemes (ETS), could play an important role in policy packages towards net zero GHG emissions. Carbon pricing provides an effective and cost-efficient approach to reducing GHG emissions with the potential to incentivise changes on both the supply- and demand-side. The use and coverage of carbon pricing has continued to expand over the past three decades, and carbon prices have reached record highs in recent years. Despite these developments, overall carbon price levels and coverage are not aligned with what is needed to reach the temperature goals of the Paris Agreement. There is thus potential to further strengthen the role of carbon pricing in pathways to net zero.

The aim of this paper is to improve understanding of the potential role and contribution of carbon pricing in transforming emission pathways towards net zero GHG emissions. This paper reviews the impacts of carbon pricing to date at an overall level and in the electricity sector in three selected jurisdictions – the EU, New Zealand, and California. Furthermore, the paper provides an in-depth analysis of the current and the potential application of emissions pricing in food systems, as well as identifying data gaps and questions that can help guide future research.

This paper is structured as follows. Section 2. outlines why rapid and deep emissions reductions are needed and how to ensure that a transition to a low-carbon society can be also a just transition. Section 3. explores carbon pricing's contribution to transforming emissions pathways to date by investigating effects of carbon pricing overall. Section 4. outlines potential ways of improving carbon pricing's effectiveness in transforming emission pathways to reach net zero, by exploring demand-side policies, policy sequencing and international co-operation and collaboration. Section 5. takes a deep dive in food systems and explores the current use and potential role of emissions pricing in food systems.

2. The imperative of reaching net zero emissions in a just manner

Key messages

To limit global warming to 1.5°C, average global GHG emissions need to fall by more than 7.6% per year between 2020-30. Such rates of emission reductions would be unprecedented given that average global GHG emissions increased by an average of 1.3% per year between 2010-19.

Despite the strengthening of many national climate policies and the adoption of various net zero targets, estimates indicate the full and timely implementation of these various commitments could limit global warming only to 1.7°C. However, many countries are not on track to implement current pledges and there is limited alignment between near-term policies and mid- and long-term strategies.

A transformative change, encompassing both demand- and supply-side shifts, will be required to deliver GHG emissions reductions at the pace and scale needed to reach net zero by 2050.

Rapid and broad transformative change can lead to some short-term negative social impacts. An overall net gain in employment is estimated from the transition to net zero, however new jobs may not necessarily occur in the same place as where jobs are lost. Moreover, new jobs may also require new skills. Just transition considerations thus need to be kept in mind when designing pathways to net zero GHG emissions.

Targeted and proactive measures based on social dialogue that include investments in job creation and re-skilling programmes, as well as social protection policies and packages will be needed to help affected workers, communities, and regions in the transition.

2.1. Greenhouse gas emissions trends and climate action

The latest science provides a stark message on the need to urgently transform GHG emissions trends (IPCC, 2021[2]). However, global GHG emissions have increased between 2010-19 (averaging 1.3% per annum (p.a.)) – albeit at a lower rate compared to the previous decade's increase of 2.1% annually. Furthermore, average GHG emissions in 2010-19 were 9 $GtCO_2$-eq higher than average annual emissions in 2000-10[3] (Friedlingstein et al., 2020[3]) (Louise Jeffery et al., 2018[4]) (Dhakal et al., 2022[5]).

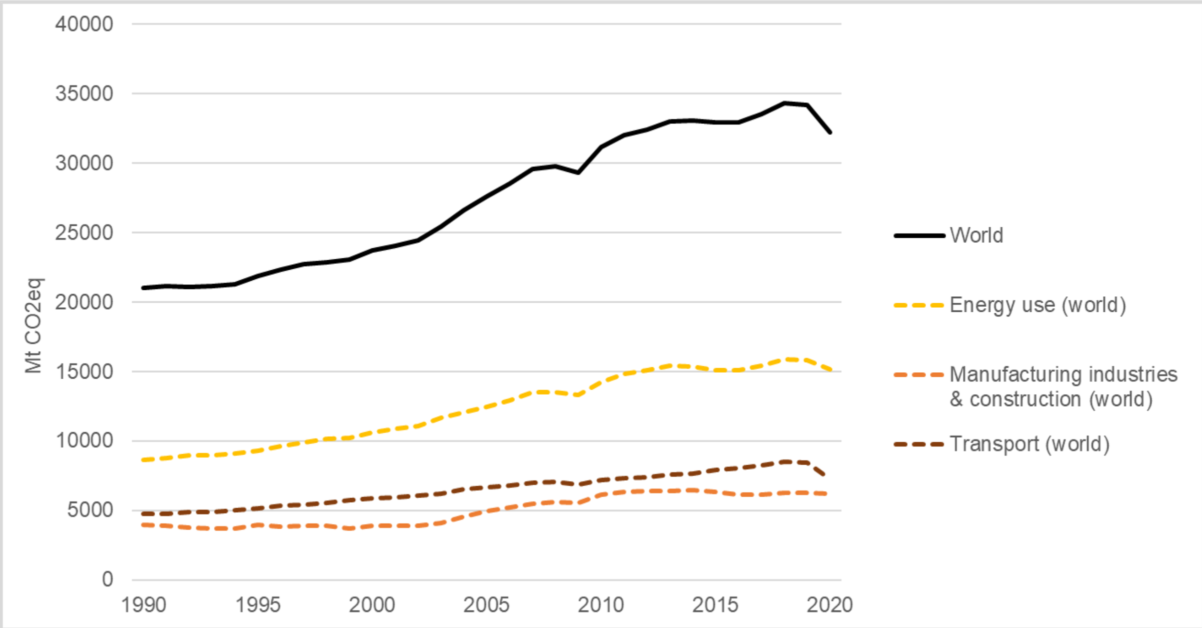
Globally, CO_2 equivalent (CO_2eq) emissions[4] from all sectors (excluding land use, land-use change and forestry (LULUCF)) grew overall between 1990-2020 by an average of 1.5% per year. There are significant

[3] Average annual GHG emissions from 2010-19 were 56 $GtCO_2$-eq while they were 47 $GtCO_2$-eq for 2000-10.

[4] Given missing global emissions data from various sectors over the specified period (1990-2020), fuel combustion emissions from energy have been used. Although this does not account for all emissions it is estimated to account for over 80% of Annex I countries' emissions while the global share is around three quarters (IEA, 2022[417]).

variations between different sectors - see Figure 2.1. For example, in the energy,[5] industry,[6] and transport sector, CO_2eq emissions from fuel combustion between 1990-2020 had average annual increases of 2.4%, 1.8% and 1.7%, respectively (OECD, 2022[6]). There have also been significant variations between and within different country groupings - see Figure 2.2. For example, CO_2eq emissions from fuel combustion in OECD countries decreased 0.2% p.a. from 1990-2020 on aggregate, while CO_2eq emissions in non-OECD countries increased by 2.8% p.a. (OECD, 2022[6]). At the same time, the emissions intensity[7] of economies decreased between 1990-2020 in OECD countries (average annual decrease of 2%) and non-OECD countries (average annual decrease of 2.2%), with significant differences between and within groups, see Figure 2.2 (IEA, 2022[7]).

Figure 2.1. CO_2eq emissions from fuel combustion globally and in selected sectors from 1990-2020



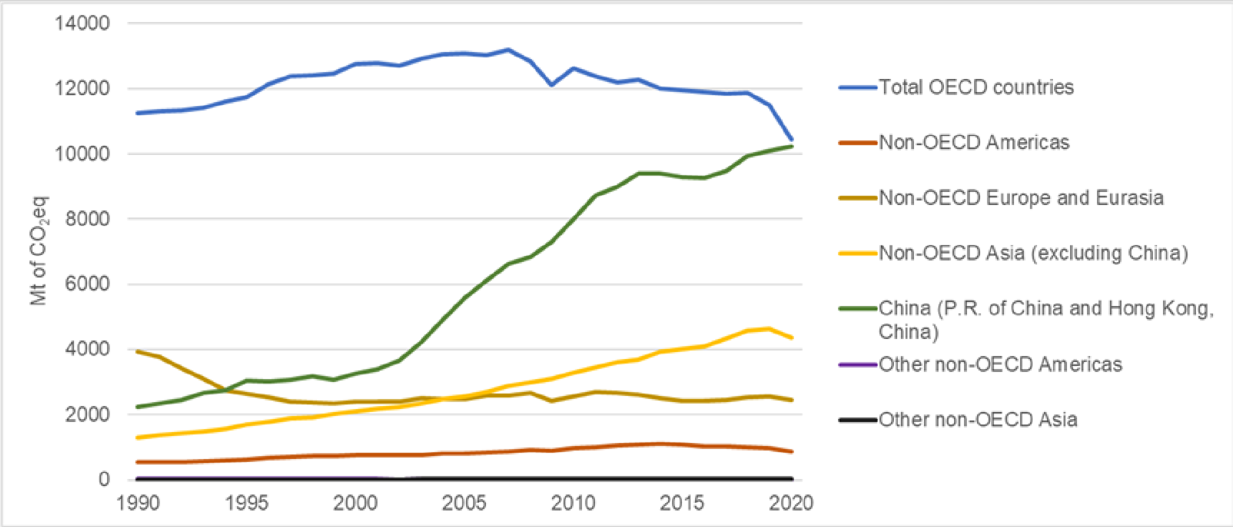
Note: World GHG emissions is the total amount of GHG emissions, while the dotted lines represent global fuel combustion emissions in specific sectors.
Source: Authors based on numbers from (OECD, 2022[6]).

[5] The term “Energy sector” refers to combined GHG emissions from “electricity and heat production” and “other energy industry use” (OECD, 2022[6])

[6] The term “Industry sector” refers to GHG emissions from “manufacturing industries and construction” (OECD, 2022[6])

[7] The emissions intensity of economies is measured as kg of CO_2 emission per GDP measured in USD (IEA, 2022[7]).

Figure 2.2. CO_2eq emissions from fuel combustion across OECD and non-OECD countries from 1990-2020



Note: OECD and non-OECD country group emissions.
Source: Authors based on numbers from (OECD, 2022[6]).

Despite some progress, full implementation of Parties' Nationally Determined Contributions (NDCs) will be insufficient to limit global warming to 1.5°C. Implementing the NDCs submitted by 193 Parties as of September 2022[8] could result in global warming of 2.6°C by 2100 and 2.4°C global warming for conditional NDCs by 2100 (UNEP, 2022[8]). This represents some limited progress compared to previous NDCs (UNEP, 2021[9]) and a higher chance of global emissions peaking before 2030 (UNFCCC, 2022[1]). Prospects improve when taking net zero commitments into account. For example, according to analysis by UNEP, implementing unconditional NDCs together with net zero targets could further limit global warming to between 1.8-2.1°C (UNEP, 2022[8]). Estimates by the IEA indicate full and timely implementation of conditional NDCs, net zero targets, sectoral pledges[9] and other climate-related commitments could limit global warming to 1.7°C (IEA, 2022[10]). However, reaching these targets requires full implementation of current pledges and some countries, including G20 countries, are not on track to achieve their initial or updated NDCs for 2030 (UNEP, 2022[8]).

Limiting global warming to 1.5°C also requires alignment of short-, mid- and long-term strategies. Many NDC pledges are not yet backed up by near-term policies (Hausfather and Moore, 2022[11]). Furthermore, of the 62 Parties that submitted Long-Term Low Emissions Development Strategies (LT-LEDS)10, only 8% are aligned with their NDCs, while 49% did not indicate a link between their NDC and LT-LEDS (UNFCCC, 2022[12]). Consistency between LT-LEDS and net zero targets is also essential to ensure policies deployed enable Parties to meet their net zero commitments (Falduto and Rocha, 2020[13]) (Jeudy-Hugo, Lo Re and Falduto, 2021[14]), yet few Parties have aligned their LT-LEDS with their net zero

[8] The 166 NDCs represent 193 Parties and include 142 new or updated NDCs communicated by 169 Parties as recorded in the interim NDC registry until 23 September 2022. These cover 94.9% of total global emissions in 2019 (excluding LULUCF), equivalent to 52.6 Gt CO_2 eq (UNFCCC, 2022[1]).

[9] Sectoral pledges from the steel, cement, aviation and shipping industries (IEA, 2022[10])

[10] The latest LT-LEDS synthesis report includes data from the 53 latest available LT-LEDS, which represents 62 Parties as of 23 September 2022 (UNFCCC, 2022[12]).

commitments (e.g. Costa Rica (Government of Costa Rica, 2019[15]) (Jeudy-Hugo, Lo Re and Falduto, 2021[14])).

Continuing along current emission pathways will not achieve the rapid and deep emission reductions needed to limit global warming to the temperature goal of the Paris Agreement. Annual average global emissions reductions of more than 7.6% per year from 2020-30 (UNEP, 2019[16]) are needed to be consistent with limiting global warming to 1.5°C. Such emission reductions would be unprecedented given that the global GHG emissions have increased by an average of 1.3% p.a. between 2019-19 (IPCC, 2021[2]).

A transformative approach that disrupts existing development patterns and delivers emission reductions at a greater pace and scale than at present is needed for global emissions to peak no later than 2025[11] and to reach net zero around mid-century (IPCC, 2022[17]). There is increasing recognition that reaching net zero emissions requires a transformative change within and across sectors and systems[12]. Transformative change is defined by the IPCC as "a system-wide change that requires more than technological change through consideration of social and economic factors that, with technology, can bring about rapid change at scale" (IPCC, 2019[18]).

Both demand- and supply-side policies will be needed to deliver emission reductions at the pace and scale required to limit global warming to 1.5°C. Supply-side measures, such as phasing out fossil fuels, developing low-carbon infrastructure and rolling out efficiency measures, are important to reduce supply-side emissions, and send signals to the demand-side regarding the need to decarbonise. There is also a need for demand-side measures that can lower overall demand and thereby help reduce the need for materials and energy upstream on the supply-side (IPCC, 2022[19]) (Green and Denniss, 2018[20]).[13]

Some demand-side measures are dependent on supply-side changes. For instance, encouraging increased recycling of waste (e.g. plastic, metal, glass and paper) requires appropriate recycling infrastructure. Similarly, encouraging people to avoid short-haul flights may require an extension of train networks to provide feasible alternatives, alongside measures promoting train travel. Using policy packages to deploy both supply- and demand-side policies simultaneously, or in sequence, could more effectively reduce emissions compared to pursing one policy at the time (Prest, 2022[21]).[14] However, support for climate change policies vary widely depending on the area and instrument.[15] Yet, (Dechezleprêtre et al., 2022[22]) reports that the source of funding for climate policies, as well as the use of any revenues generated, could alter the level of support. Similarly improving information about the co-

[11] Global GHG emissions need to peak between 2020 and at the latest before 2025 in global modelled pathways that limit warming to 1.5°C (>50%) with no or limited overshoot and in those that limit warming to 2°C (>67%) and assume immediate action (IPCC, 2022[17])

[12] A sector refers to individual sectors such as the transport and building sectors, whereas systems (e.g. food systems) are cross-sectoral and cut across multiple sectors (Babiker et al., 2022[192]).

[13] For instance, moving from fossil fuel-based electricity towards renewable-based electricity can be accelerated with policy instruments to stimulate supply (e.g. grants for research, development and deployment) and demand (e.g. price signals) (IEA, 2022[392]) (IEA, 2022[393]). In the agricultural sector, where abatement challenges involve strategic, political and technical aspects, supply- and demand-side levers are needed to trigger a shift away from emissions-intensive production and consumption (IPCC, 2022[338]) (WRI, 2016[339]).

[14] For instance, the deployment of sector-level regulation and early mitigation action, combined with lifestyle changes can contribute to climate mitigation and well-being outcomes (Bertram et al., 2018[340]).

[15] For instance, energy policies such as low-carbon technology subsidies are broadly supported by the public, while food policies, especially a tax on beef, receives less support (Dechezleprêtre et al., 2022[22]). Although not reported by (Dechezleprêtre et al., 2022[22]), the existence of significant emissions intensity differences for the same food product could also lead to application difficulties as well as to lower support (see section on Technical and methodological challenges related to applying emissions pricing in food systems).

benefits of climate policies, including carbon pricing could help to increase support (Dabla-Norris et al., 2023[23]).

2.2. Just transition considerations for reaching net zero

Targeted, proactive measures are needed to take just transition considerations into account in pathways towards net zero emissions. A rapid and broad transformative change across sectors and systems will likely be "inherently disruptive" (Pathak et al., 2022, p. 37[24]). It is, therefore, expected to have some negative impacts on, e.g. communities dependent on employment in high-emissions sectors. Transformative change is not only about accelerated efforts to mitigate climate change, but also about ensuring a fair transition for affected workers, communities and sectors (Lecocq et al., 2022, p. 100[25]) (Newell and Mulvaney, 2013[26]).

In a transition towards low-emissions societies, new industries, such as the clean energy industry, will expand while others, such as fossil fuel industries, will need to be phased out. Just transition considerations thus need to be kept in mind. While there is currently no commonly agreed definition of a just transition, the International Labour Organisation's (ILO) definition[16] is commonly cited and used (UN DESA, 2022[27]) (EurWORK, 2022[28]) (EC, 2022[29]) (UN Global Compact, 2022[30]).

Just transition considerations are recognised and incorporated in several important declarations and agreements. In the Paris Agreement, Parties agreed to "Taking into account the imperatives of a just transition of the workforce and the creation of decent work and quality jobs in accordance with nationally defined development priorities" (UNFCCC, 2015[31]). The Silesia Declaration on Solidarity and Just Transition adopted at COP24 in 2018 highlights the importance of social dialogue for gaining public approval of the changes needed to achieve the goals of the Paris Agreement (UNFCCC, 2018[32]). Just transition considerations are also related to several Sustainable Development Goals (SDGs) including SDG 13 on climate action, SDG 8 on decent work, SDG 7 on affordable and clean energy, SDG 9 on industry, innovation and infrastructure (Pollin, 2021[33]).

The transition to a low-emissions society is projected to generate an overall net gain in energy sector employment under different scenarios see Table 2.1. Scenarios estimating employment impacts from a transition to a low-emissions society are only available for few sectors such as the energy sector. The IEA's Announced Pledges Scenario (APS)[17] projects an overall net gain of 15 million energy sector jobs by 2030. The IEA's Net Zero Emissions (NZE) scenario[18] estimates an overall estimated net gain of 25 million energy sector jobs by 2030 (IEA, 2022[10]). The ILO's Sustainable Energy scenario[19] estimates an

[16] According to the ILO: "a Just Transition means greening the economy in a way that is as fair and inclusive as possible to everyone concerned, creating decent work opportunities and leaving no one behind" (ILO, 2022[413]).

[17] The Announced Pledges Scenario (APS) bases its projections on climate commitments made by governments across the globe as of mid-2021. These include NDCs and longer-term net zero targets and assume these will be met in full and on time (IEA, 2021[39]). The APS scenario estimates overall energy sector employment will increase from 65 million jobs in 2021 to 80 million jobs by 2030, equivalent to an estimated overall net gain of 15 million jobs.

[18] The Net Zero Emissions (NZE) scenario outlines a narrow yet achievable pathway for net zero CO_2 emissions in the global energy sector. This pathway is aligned with limiting global warming to 1.5°C by 2100 (IEA, 2021[39]). The NZE scenario estimates energy sector employment will reach 90 million by 2030.

[19] The Sustainable Energy scenario outlines the employment impact between IEA's energy pathways limiting global warming to 2°C and the business-as-usual (BAU) scenario where global warming increases 6°C by 2030 (ILO, 2018[35]). In this scenario, it is estimated that around 18 million jobs will be created while 1.5 million jobs will be lost globally by 2030. Of the jobs lost, ILO estimates around 5 million jobs could be reallocated to other industries within the same country.

overall gain of 21.5 million energy sector jobs by 2030, while the ILO's Circular Economy scenario[20] estimates a net gain of 17 million jobs in the energy sector by 2030 (ILO, 2019[34]) (ILO, 2018[35]).

At the country level, available scenarios also project a net gain in energy sector employment. For example, in Australia, a scenario where 50% of electricity originates from renewable energy sources by 2030, is estimated to lead to the creation of 28,000 new jobs (Climate Council of Australia, 2016[36]). In India, the implementation of its 2019 NDC could lead to a 30% increase in jobs (1.6 million jobs in total) by 2030, while a high ambition renewable energy roadmap could lead to a 46% job increase (2.3 million jobs in total) (IASS et al., 2019[37]). Furthermore, few OECD countries have regions where over 5% of employment in a given sector risks employment losses from a net zero transition (OECD, 2021[38]).

Table 2.1. Estimated employment net gain in the energy sector by 2030 in different scenarios

	Announced Pledges Scenario (APS)	Net Zero Emissions (NZE)	Sustainable Energy Scenario	Circular Economy Scenario
Globally	15 million	25 million	21.5 million*	17 million*

Note: * including jobs reallocated to other sectors or industries.
Source: (IEA, 2022[10]) (IEA, 2021[39]), (ILO, 2019[34]) and (ILO, 2018[35]).

Most jobs lost in the energy sector in a transition to a low-emissions society are estimated to be in the fossil fuel industry. In IEA scenarios, the largest decline in jobs from 2019-30 is estimated to occur within coal supply and in oil and gas[21] (IEA, 2022[10]) (IEA, 2022[40]). In the ILO's Energy Sustainability scenario, job losses are projected to specifically occur in petroleum extraction and refining; coal mining and coal-based electricity generation (ILO, 2019[34]). In the ILO's Circular Economy scenario, job losses are estimated to occur in the manufacturing of basic iron and steel, mining of copper, manufacturing and production of wood and cork (ILO, 2018[35]). In the US and Japan, employment in coal powered electricity and coal extraction is expected to decrease by more than 73% by 2040 (Bibas, Chateau and Lanzi, 2021[41]).

Fossil fuel and extractive industries are often concentrated in specific regions or communities where they support many jobs. The loss of jobs in certain regions or communities may not necessarily coincide with the location where new jobs will be created and could lead to significant economic impacts in affected areas (Banerjee and Schuitema, 2022[42]). New jobs may also require new skills and could require reskilling programmes in some countries. Currently, 45% of global energy sector workers are high-skilled, while less than 10% are low-skilled. In emerging markets and developing economies (EMDEs), the majority of workers are involved in low-skilled manual labour tasks while in advanced economies, such tasks are often automated (IEA, 2022[40]).

The creation of new jobs in the transition to a low-emissions society is likely to require reskilling and retraining. In the ILO's Energy Sustainability scenario, around 5 million jobs are estimated to be eligible for reallocation, i.e. workers could transfer their skills to a similar job in another industry. However, 18 million jobs are estimated to be created in other industries. As these jobs would require workers to shift from one industry to another, it would require retraining and reskilling of workers (ILO, 2019[34]) (ILO, 2018[35]).

[20] The Circular Economy scenario outlines the employment impact between a BAU scenario and a scenario in which there is a sustained 5% annual increase in recycling rates (plastics, glass, pulp, metals and minerals) across countries and related services (ILO, 2018[35]). In this scenario, it is estimated that around 7 million jobs will be created and around 30 million jobs will be lost globally by 2030, with around 50 million jobs reallocated.

[21] Global employment in coal supply is projected to decrease by approximately 2 million in the APS and 2.5 million in the NZE scenarios between 2019-30 (IEA, 2022[10]), while global employment in oil and gas in the same period is projected to decrease by around 0.5 million in the APS and 2 million in the NZE scenarios (IEA, 2022[40]).

Targeted and proactive government measures are needed to help specific groups in the transition and reduce adverse impacts. These measures could encompass different elements including:

- Social dialogues between workers, companies, trade unions and governments to understand the context, impacts and needs of different groups as well as to map out opportunities (ILO, 2015[43]).
- Public and private investments in low-carbon industries to create job opportunities, as well as investments in training and re-skilling programs to ensure skilled workers are available to take up emerging jobs (Smith, 2017[44]) (World Bank, 2021[45]).
- Social protection policies and packages providing income and other types of support to workers in between jobs, as well as other measures which could help workers during a transition, such as temporary transport opportunities to new jobs or social housing in areas close to new jobs (World Bank, 2021[45]) (OECD, 2019[46]).

These elements could together help enhance understanding of the context and needs, indicate the best-suited policy sequence, and develop context-specific policies to support a just transition in affected regions and communities. As can be seen in Box 2.1, some countries have already started to deploy policies to support a just transition. Wider use of such initiatives could help to ease the transition in different regions and countries.

Box 2.1. Targeted just transition policies in selected countries

Various examples of targeted just transition policies and programmes are already being seen globally. In the **EU**, the Just Transition Mechanism provides financial support, transition plans and technical assistance to help carbon-intensive EU regions transition away from fossil fuels (European Commission, 2022[47]). The impact of the mechanism is yet to be assessed as it started in 2021 (EC, 2022[48]). Nevertheless, at the end of 2022, **Spain** was approved to receive EUR 869 million from the Just Transition Fund, to aid its Just Transition Strategy for the phase-out of coal mines and coal power plants. The funds will be used to provide financial assistance, initiate new businesses and job opportunities, as well as for the re-training and reskilling of affected workers (EC, 2022[49]). In **Poland**, following a workforce reduction of 75% in the mining community, the Polish Government worked with labour unions to develop social packages and special privileges for the mining community (Szpor and Ziółkowska, 2018[50]) (Zinecker et al., 2018[51]). In 2018, **Canada** announced plans to replace 90% of its coal powered electricity with non-emitting sources by 2030. To mitigate the expected negative effect, a task force was established to initiate social dialogues with citizens across the country. These dialogues emphasised the need for research on just transition impacts, funding for affected communities, skills development, and the need for local infrastructure projects (Task Force on Just Transition for Canadian Coal Power Workers and Communities, 2018[52]). In response, the government committed finance from the 2019 budget to worker transition centres, economic coal worker support programmes, and an infrastructure fund to improve diversification in affected local communities (Government of Canada, 2019[53]) (WRI, 2021[54]). In **Colombia**, the government is developing a national just transition strategy for the labour force which is due to be finalised in 2023. The strategy includes a social dialogue between employers, workers, citizens and the government to contribute to the design and implementation of the strategy (Government of Colombia, 2020[55]).

A just transition is not only about phasing out jobs in high emitting industries, but also about phasing out the demand for high emitting products and services with supporting policy packages. In Germany, as part of the aim to phase-out coal power nationally by 2038, the government granted payments to incentivise the closure of uncompetitive coal mines (WRI, 2021[56]) (Wettengel, 2020[57]). In addition, the government is developing energy alternatives by investing in clean energy, as well as in the development of jobs in affected regions (Raitbaur, 2021[58]). Yet, given the German energy-reliance on Russian gas, some coal-fired power plants have recently been taken into use again to help increase Germany's energy independence (Connolly, 2022[59]). In Egypt, the government conducted fossil fuel reforms in 2014. To lessen the impact of these, the government promoted the reform together with other measures such as a boost of minimum wages in the public sector (AfDB/OECD/UNDP, 2014[60]), increased food subsidies (Clarke, 2014[61]) and implemented progressive taxation (Clarke, 2014[61]) (Zinecker et al., 2018[51]). In Indonesia, the removal of fossil fuel subsidies was replaced with infrastructure investments, economic transfers to villages and poverty reduction programs (Zinecker et al., 2018[51]).

Some countries are explicitly linking climate action and just transitions in their NDCs[22]. Canada's NDC notes the government's intention to invest in initiatives to develop skills, job opportunities, and legislation to support the livelihoods of workers and communities in the transition to reaching their net zero target (Government of Canada, 2021[62]). Chile's NDC update includes a just transition and sustainable development social pillar which is seen an enabling condition for implementing the NDC. Chile plans to develop a strategy[23] that protects the most vulnerable in the process of decarbonising the energy mix by 2050 (Chile, 2020[63]). Other countries highlighted plans to include just transition aspects in implementing their NDC to tackle the impact of climate policies on different societal groups. Some countries also considered social and economic consequences of climate measures by including economic diversification plans, just transition or social aspects in their design of climate policies. Although not all countries specifically included the term "just transition" in their NDCs, many allude to it by including social and economic workforce impacts and climate justice elements in their NDCs (UNFCCC, 2022[1]).

[22] As per 23 September 2022 (UNFCCC, 2022[1]).

[23] The strategy includes retraining measures as well as a plan to monitor the improvements in areas where coal-fired plants will close (Energy, 2021[415]) (Guzmán, 2022[352]).

3. Understanding the contribution of carbon pricing in transforming emission pathways to net zero

Key messages

While the use and coverage of carbon pricing has expanded globally since 1990, a large part of global GHG emissions are currently not covered by any form of carbon pricing and most schemes have carbon prices below EUR 50/tCO_2.

Although carbon price levels have increased in certain jurisdictions in recent years, only 19% of emissions in OECD and G20 countries were priced at levels estimated to be in line with limiting global warming to 1.5°C this century.

Available analysis indicates that GHG emissions covered by carbon pricing schemes have on average fallen by 0-2% per year since the 1990s.

Disentangling the impact of carbon pricing on sectoral emission pathways is challenging.

Evidence from the electricity sector in the EU, New Zealand, and California suggest that carbon pricing, related revenues and other policies have together contributed to reducing GHG emissions and increasing the share of renewables in the electricity mix in these jurisdictions.

Further research and analysis exploring the effectiveness of carbon pricing alongside different policies in key sectors could be helpful in determining carbon pricing's contribution to transforming emission pathways and the policy mix which generates the greatest emissions reductions without exacerbating negative distributional impacts.

3.1. Carbon pricing's overall impact and contribution to reaching net zero emissions

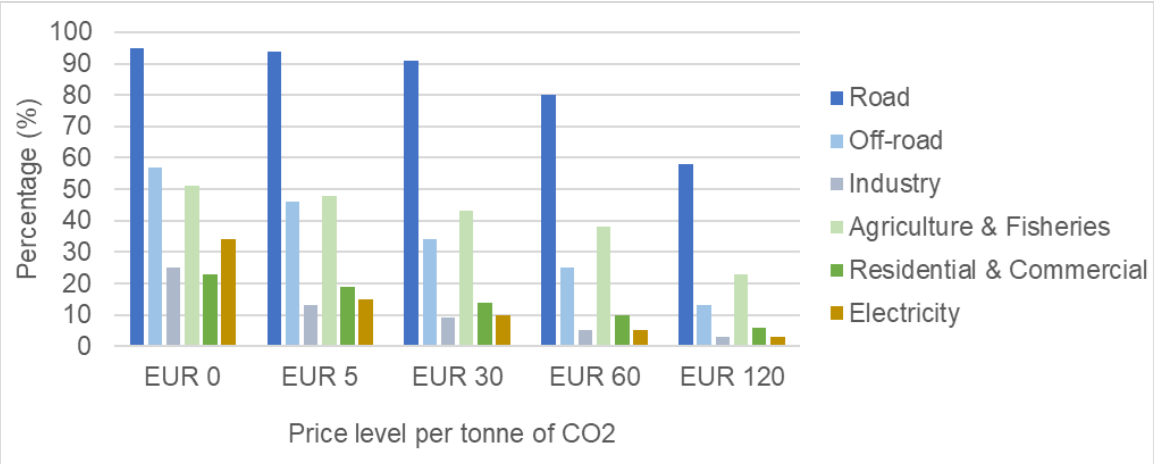
3.1.1. Brief overview of carbon pricing developments over time

Although the use and coverage of carbon pricing has expanded since 1990, a large part of carbon emissions are currently not covered by any form of carbon pricing. Finland was the first country to implement a carbon pricing scheme in 1990. In 2015, 38 carbon pricing schemes were in place globally covering almost 12% of GHG emissions in 2015 (World Bank, 2022[64]). As of April 2022, 68 carbon pricing schemes were in place covering 23% of global GHG emissions in 2022 (World Bank, 2022[65]). [24] Recent

[24] Of the carbon pricing schemes in place in 2022, 37 were carbon taxes while the remaining 34 were ETS (World Bank, 2022[65])

increases in the coverage of emissions by carbon pricing schemes have been due to the Chinese ETS - covering 8.8% of global GHG emissions in 2022 (World Bank, 2022[64]). In 44 OECD and G20 countries in 2018, 31% of carbon emissions were priced at EUR 5/tCO_2 or above[25] while 69% of carbon emissions were unpriced. The highest percentage of emissions priced at EUR 5/tCO_2 or above in 2018 in 44 OECD and G20 countries was the road transport sector (94%), while the industry sector had the lowest percentage (13%) of emissions priced at EUR 5/tCO_2 or above (see Figure 3.1) (OECD, 2022[66]).

Figure 3.1. Share of CO_2 emissions priced at various levels in 44 OECD & G20 countries in 2018



Note: The numbers are extracted from the OECD's database on Effective Carbon Rates (ECR), which measured carbon pricing of CO_2 emissions from energy use in 44 OECD and G20 countries in 2018. The dataset included emissions under carbon taxes, ETS, and fuel excise taxes. As this graph includes only CO_2 emissions, most agricultural emissions are missing as these are mainly non-CO_2.
Source: based on numbers from (OECD, 2022[66]).

There are also several proposals for new or revised carbon pricing schemes currently underway. In the US, the Transportation and Climate Initiative between 13 states is proposing to implement an ETS to help reduce transport emissions (World Bank, 2022[65]). In the EU, the 'Fit for 55' package proposes to increase the EU ETS' GHG emissions reduction target from 43% to 61%, and potentially to 63%[26], by 2030 compared to 2005 levels.[27] Ukraine, which already has a carbon tax, has announced plans to implement a national ETS by 2025. Furthermore, Colombia and Thailand have announced the potential launch of pilot ETS. In Viet Nam, a 2020 law outlined the organisation and development of a carbon market and Pakistan is exploring the role of carbon markets in helping to achieve its NDC (World Bank, 2022[65]).

Over the last few years, several important carbon pricing-related developments have taken place. For instance, the Article 6 rulebook that allows countries to voluntarily co-operate to achieve NDC emissions targets, was agreed at COP26 in Glasgow in 2021 (UNFCCC, 2022[67]). Carbon trading under Article 6

[25] The carbon price level set at EUR 5/tCO_2 or above is one of several thresholds used to calculate the percentage of emissions priced at various levels in different countries and sectors. The other price level thresholds used in the report include: EUR 0/tCO_2 or above; EUR 30/tCO_2 or above, EUR 60/tCO_2 or above, EUR 90/tCO_2 or above, and EUR 120/tCO_2 or above (OECD, 2022[66]).

[26] According to a proposal for faster EU action and energy dependence in June 2022, the 2030 GHG emissions reduction target for the EU ETS is proposed to be increased from 61% to 63% (European Parliament, 2022[351]).

[27] This would extend the current carbon price signal (which applies to around 22% of EU GHG emissions), to over 67.6% of GHG emissions in the EU by 2030 (IEEP, 2021[395]). The proposal also suggests the inclusion of the maritime transport sector in the existing ETS, and a separate ETS for buildings and road transport (European Council, 2022[394]) (European Parliament, 2022[351])

could help both to cut the cost of reducing emissions, and contribute to ensuring that countries fulfil their NDCs (World Bank, 2022[68]). At COP26, countries also agreed to phase down unabated coal power and put an end to inefficient fossil fuel subsidies, which disrupt carbon price signals (UNFCCC, 2021[69]). This pledge was reiterated at COP27, yet no further actions were agreed (UNFCCC, 2022[70]). In the EU, a Carbon Border Adjustment Mechanism (CBAM)[28] has been adopted with a transitional phase for five emission-intensive sectors with high carbon leakage risks[29] due to start in 2026. A non-compliant pilot will take place between 2023-26.

In recent years, carbon price levels have hit record-highs in ETS in many jurisdictions. Since 2021, several jurisdictions, mostly in advanced economies, have experienced increases in observed ETS price levels (see Figure 3.2) (World Bank, 2022[65]). Record high ETS prices were seen at the start of 2022 in the EU ETS (EUR 96/tCO_2), the UK ETS (EUR 104/tCO_2),[30] and the New Zealand ETS (NZ ETS) (EUR 51/tCO_2).[31] Record high ETS price levels were seen in mid-2022 in the Swiss ETS (EUR 80/tCO_2),[32] the Québec and California price-linked Cap & Trade schemes (EUR 31/tCO_2)[33] and in the Regional Greenhouse Gas Initiative (RGGI) (EUR 15/tCO_2) (ICAP, 2022[71]).[34] Price increases were also seen in the ETS of the Republic of Korea (K-ETS), where the price increased from EUR 8/tCO_2, in July 2021, to EUR 26/tCO_2 in January 2022.

Most of the increases seen in ETS prices were driven by changes and proposed revisions to climate policies. In the EU ETS, prices increased due to the temporary removal of 900 million allowances, the decision to increase the 2030 mitigation target, and a reform proposal advocating a reduction of the EU ETS cap (World Bank, 2022[65]). In New Zealand, prices increased following the removal of the NZD 35/tCO2 fixed-price option on emission allowances as well as expectations of tightened ETS rules. In the Republic of Korea, the proposal to tighten the 2030 emissions target led to significant price increases in June 2021. In the linked Québec-California schemes, the opening of the ETS market to investment firms partially led to price increases (World Bank, 2022[65]).

[28] CBAM aims to prevent carbon leakage by adding a carbon price, equivalent to the EU ETS price, to goods imported into the EU from countries without carbon pricing schemes in place (European Union, 2022[416]). In the past, the allocation of free allowances under the EU ETS aimed to address concerns of carbon leakage, however as the allocation of free allowances is to be phased-out by 2035, CBAM is expected to increasingly take its place (European Union, 2022[416]).

[29] The five transitional CBAM sectors are cement, iron and steel, aluminium, fertilisers, and electricity (European Union, 2022[416]).

[30] Price in original currency GBP 90/tCO_2, converted via exchange rate: GBP 1 = EUR 1.16 (assessed on 27 October 2022) (Google finance, 2022[353])

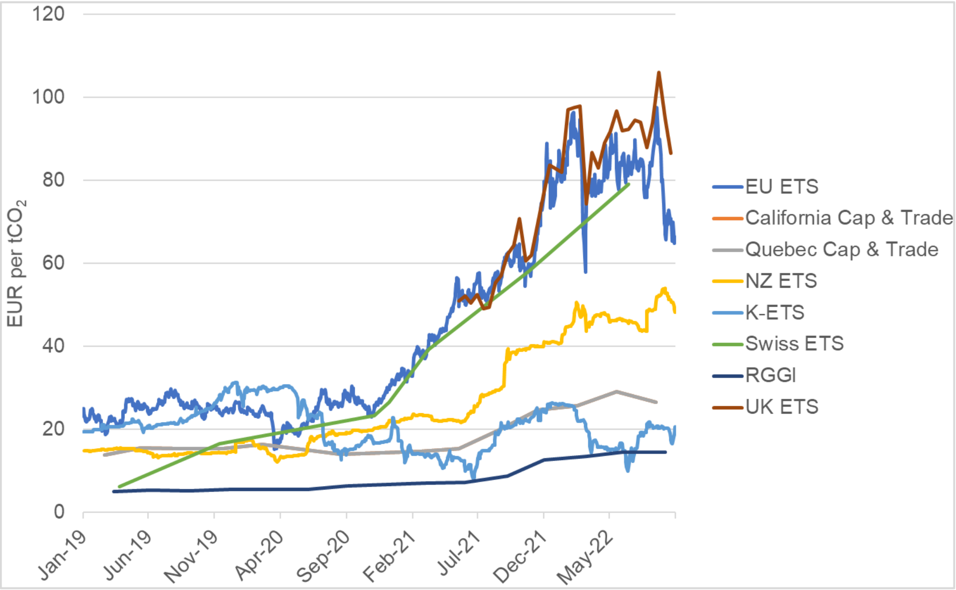
[31] Price in original currency NZD 87/tCO_2, converted via exchange rate NZD 1 = EUR 0.58 (accessed on 29 October 2022) (Google finance, 2022[353])

[32] Price in original currency CHF 79.1/tCO_2, converted via exchange rate CHF 1 = EUR 1.01 (accessed on 27 October 2022) (Google finance, 2022[353])

[33] Price in original currency USD 31/tCO_2, converted via Exchange rate: USD 1 = EUR 1 (assessed on 27 October 2022) (Google finance, 2022[353]). This currency conversion applies throughout the document.

[34] ICAP's allowance price explorer was accessed on 27 October 2022.

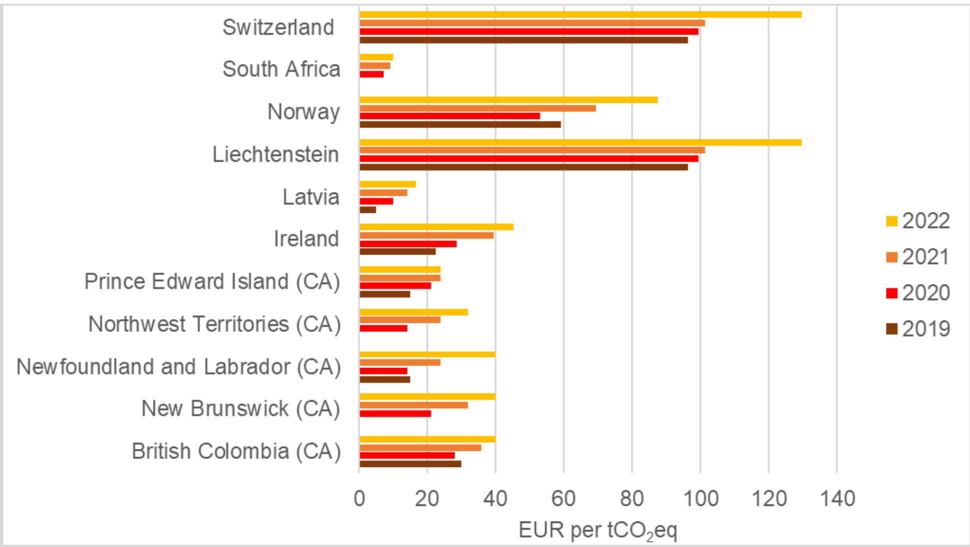
Figure 3.2. ETS price developments in selected schemes from 2019-22



Source: Based on data from (ICAP, 2022[71]).

Carbon tax rates also increased in 2021 and in early 2022 (see Figure 3.3), although relatively less than ETS price levels. In 2021, the average increase of all carbon taxes was around EUR 5/tCO_2eq, and by April 2022, average carbon taxes increased by an additional EUR 6/tCO_2eq. This led to record-high carbon taxes in several jurisdictions such as Latvia and South Africa. The carbon tax developments stemmed from planned increases (e.g. Canadian provinces and Ireland); wider fiscal reforms (e.g. Norway and Ukraine); or were triggered as jurisdictions did not reach their intermediate GHG targets (e.g. Switzerland and Liechtenstein) (see Box 3.1) (World Bank, 2022[65]).

Figure 3.3. Carbon tax developments



Note: Prices were originally in USD, yet a conversion rate of USD 1 = EUR 1 have been used and prices are shown in EUR.
Source: Based on data from (World Bank, 2022[64]).

Box 3.1. Linking carbon tax rates to GHG emissions reductions: experiences in Switzerland and Liechtenstein

Switzerland

The carbon tax is a central instrument in Switzerland's climate policies under its CO_2 act. The carbon tax has been in place since 2008, and it is applied to fossil heating and process fuels (i.e. oil, natural gas, coal) (The Federal Assembly of the Swiss Confederation, 2013[72]). The tax is a complementary instrument to the Swiss ETS and covers approximately 33% of GHG emissions in Switzerland (World Bank, 2022[64]). The tax rate is linked to intermediate emission reductions targets, inscribed in Switzerland's CO_2 law. If such targets are not met, the tax rate increases. In 2020, Switzerland planned to reduce its CO_2 emissions from fuel combustions by 33%, compared to 1990 levels. Yet, emissions were reduced by 31%, thereby triggering an automatic tax rate increase from EUR 89/tCO_2 (CHF 96/tCO_2) to EUR 111/tCO_2 (CHF 120/tCO_2) by January 2022 (Federal Office for the Environment, 2021[73]). Future emission reductions targets and associated price triggers have not yet been defined, as the CO_2 tax is under revision (World Bank, 2022[64]).

Liechtenstein

The carbon tax in Liechtenstein is implemented via a bilateral treaty with Switzerland, requiring Liechtenstein to apply the Swiss federal legislation on a carbon tax domestically (World Bank, 2022[64]). Thus, any carbon tax rate changes in Switzerland also apply to Liechtenstein's carbon tax. Thus, if Switzerland does not reach their interim targets, the carbon tax rate increases in both countries. In Liechtenstein, the carbon tax applies to CO_2 emissions from production, extraction and import of coal and other fuels subject to the Swiss mineral oil tax, equivalent to 80.6% of domestic GHG emissions (Landesgesetzblatt, 2021[74]) (World Bank, 2022[64]). The outcome of the revision of the Swiss carbon tax will affect the development of the future carbon tax rate in Liechtenstein.

In jurisdictions where carbon pricing is applied, there are significant variations, and often a trade-off, between price levels and emissions coverage (see Table 3.1). As of April 2022, the highest carbon tax was applied in Sweden (EUR 129.9/tCO_2e), covering 40% of Sweden's emissions. In terms of the coverage of emissions by carbon taxes in 2022, carbon taxes applied in South Africa and Singapore covered 80% of their respective GHG emissions, at prices of EUR 9.8/tCO_2eq and EUR 3.7/tCO_2eq in 2022 respectively. The carbon tax in Lichtenstein has both a high coverage and high price level. Yet, Lichtenstein is a high-income country with relatively small overall emissions compared to other jurisdictions. For ETS, a similar trade-off between price levels and coverage emerges. The highest emissions trading prices were found in the EU, where ETS prices amounted to EUR 86.5/tCO_2eq in 2022 with a coverage of 40.7% of total EU GHG emissions. The largest coverage of emissions by ETS was found in California, US (74%) and Nova Scotia, Canada (85%), with price levels of EUR 30.8/tCO_2eq and EUR 23.1/tCO_2eq in 2022, respectively (World Bank, 2022[64]) (ICAP, 2022[75]). If GHG emissions are to be reduced rapidly across economies, there is potential for extending the coverage and price level in existing carbon pricing schemes, as well as extending the use of carbon pricing schemes in jurisdictions where such schemes are not yet applied.

Table 3.1. Carbon price levels and GHG emissions coverage across jurisdictions in 2022

	Price (EUR/tCO_2eq)*	Coverage (% of jurisdictions GHG emissions)	GHG emissions coverage ($MtCO_2eq$)
Carbon tax			
Sweden	129.9	40%	65
South Africa	9.8	80%	574
Singapore	3.7	80%	71
Lichtenstein	129.8	80.6%	-
ETS			
EU	86.5	40.7%	4001
California, US	30.8	74%	418
Nova Scotia, Canada	23.1	85%	17

Note: *Original price in the specified source is given in USD, however, given that the USD 1 = EUR 1, the currency have been changed to EUR. The GHG emissions coverage have been calculated using numbers from the specified GHG emissions in the jurisdiction from 2018. However, for Liechtenstein, this number is not provided.
Source: (World Bank, 2022[64]).

3.1.2. Carbon pricing and GHG emissions

Although overall carbon price levels have increased in recent years, only 19% of emissions in OECD and G20 countries were priced at levels that are estimated to be aligned with meeting the goals of the Paris Agreement, which is to limit global warming to well below 2°C, and preferably to 1.5°C. To achieve this goal, carbon pricing levels needed to be at least EUR 40-80/tCO_2 in 2020 and reach EUR 50-100/tCO_2 by 2030 (Stiglitz et al., 2017[76]). However, as of April 2022, explicit carbon price levels in nine jurisdictions[35], equivalent to 4% of global emissions, were at or above the estimated price range required by 2030 (World Bank, 2022[65]). In 2018, 44 OECD and G20 countries were responsible for about 80% of energy-related global CO_2 emissions (explicit[36] and implicit[37] carbon pricing), yet, only 19% of these emissions were priced at EUR 60/tCO_2 or more. Moreover, out of the 44 countries, only nine[38] had more than 50% of their emissions priced at EUR 60/tCO_2 or above in the same year (OECD, 2021[77]). The average explicit carbon price across 71 countries was, in 2018, EUR 1.78/tCO_2eq. While this had more than doubled by 2021, to EUR 4.29/tCO_2eq, this is still significantly below what is needed to meet the goals of the Paris Agreement (OECD, 2022[78]).

Carbon prices in place in several countries or sectors are considerably lower than the level that is estimated to be needed to reduce emissions at the required pace and scale (Sulistiawati, 2022[79]) (Gokhale, 2021[80]) (Sen, 2021[81]). An overview analysis of different carbon pricing schemes (Best, Burke and Jotzo, 2020[82]) indicated that an increase of one Euro per tonne CO_2/eq was associated with a 0.3% reduction in growth of GHG emissions from fossil fuels. The impact of carbon pricing depends on several factors, including on the design of the carbon pricing scheme itself (including on how revenue from carbon pricing is used).

35 New Zealand, Switzerland, Finland, EU, Norway, UK, Liechtenstein, Sweden, Uruguay (World Bank, 2022[65]).

36 Explicit carbon pricing refers to the pricing of products directly equivalent to the carbon content of the product, such as carbon taxes or ETS (World Bank, 2022[65]).

37 Implicit carbon pricing refers to the pricing of products associated with GHG emissions, yet not directly equivalent to the carbon content of the product (World Bank, 2022[65]).

38 Switzerland, Luxembourg, Norway, Slovenia, Iceland, France, Ireland, Italy, and the Netherlands (OECD, 2021[77]).

There are several projections outlining expected market behaviour changes from higher carbon prices. For instance, increasing a carbon price in the electricity sector from 22.1% to 68% is projected to increase the shift from coal to renewable energy (Wong and Zhang, 2022[83]).

Assessing behavioural changes in response to high carbon prices is challenging for multiple reasons. This includes that price levels have historically been low, and participants have benefited from exemptions which dilutes the price signal (Venmans, Ellis and Nachtigall, 2020[84]), and that the impact of prices on demand varies by sector. In theory, carbon pricing internalises the external cost of carbon emissions, sending a price signal encouraging low-carbon investment, innovation, and consumption (Stiglitz et al., 2017[76]). Yet, in practice the carbon price signal is often diluted by e.g. exemptions, low price levels and fossil fuel support. Indeed, almost all existing carbon pricing schemes include some forms of exemptions.[39] Exemptions are not intentionally applied to disrupt the carbon price signal, but often to address competitiveness concerns and to prevent carbon leakage.

Pricing emissions based on the social cost of carbon is one way to ensure that carbon price levels are high enough to potential trigger behavioural change and help mitigate climate change. The social cost of carbon is calculated by measuring the damage incurred by the release of an additional ton of CO_2 in monetary terms (OECD, 2018[85]). As such, it puts a monetary value on CO_2 emitting activities and also enables an economic assessment of the potential benefits from emission reductions. There are different ways of calculating the social cost of carbon. For instance, in the US, the current estimate for the social cost of carbon is USD 51/tCO_2 (Chemnick, 2021[86]), yet there are some calls for this estimate to be increased, see for example (Rennert et al., 2022[87]).

"Positive" carbon pricing that incentivises GHG emission reductions is often counteracted by "negative pricing" in the form of fossil fuel support,[40] which incentivises increases in GHG emissions. Fossil fuel support in OECD and G20 countries[41] has fluctuated since 2012, and support fell significantly in 2020 due to low fossil fuel demand and low prices from the early stages of the COVID-19 pandemic (OECD, 2019[88]) (OECD, 2022[89]). Despite governments allocating one-third of funds to environmentally positive measures during COVID-19,[42] fossil fuel support in 2021 (USD 697 billion) almost doubled compared to 2020 (USD 362 billion) (OECD, 2022[89]). This sharp increase is likely due to the increased energy prices as the global economy rebounded in 2021. Many countries have not yet fulfilled long-stranding pledges to phase out

[39] In the EU ETS, exemptions are in the form of free allowances for certain emissions-intensive and trade-exposed facilities. In the UK, the climate change levy applies to energy used by businesses, with exemptions for electricity, gas and solid fuels used in certain forms of transport or not used in the UK (UK Government, 2022[396]). In Singapore, exemptions do not apply for entities covered under the carbon tax to maintain a strong price signal, yet, the tax rate is currently relatively low – although it will be increasing over time (Ministry of Sustainability and the Environment, n.d.[397]).

[40] Fossil fuel support is defined by the OECD to include direct budgetary transfers, tax expenditures and induced transfers (i.e. price support) that provide a benefit or preference to fossil fuel production or consumption (OECD, 2015[398]).

[41] Australia, Brazil, Canada, the People's Republic of China, Germany, France, United Kingdom, Indonesia, India, Italy, Japan, Republic of Korea, Mexico, Russian Federation, Republic of Türkiye, United States, South Africa, Algeria, Angola, Argentina, Azerbaijan, Bahrain, Bangladesh, Bolivia, Brunei Darussalam, Colombia, Ecuador, Egypt, Gabon, Ghana, Iraq, Iran, Kazakhstan, Kuwait, Libya, Malaysia, Nigeria, Oman, Pakistan, Qatar, Saudi Arabia, Sri Lanka, Chinese Taipei, Thailand, Trinidad And Tobago, Turkmenistan, Ukraine, United Arab Emirates, Uzbekistan, Venezuela, Viet Nam (OECD, 2022[89]).

[42] During the early stages of the pandemic, many countries announced their commitment to a green recovery using stimulus packages. An assessment shows that around 33% (USD 1090 billion / EUR 1090 billion) of total recovery spending since the start of the pandemic was allocated to environmentally positive measures. 14% (USD 290 billion / EUR 290 billion) of total recovery spending was allocated to environmentally mixed or negative measures, while 52% was found to not have had a direct environmental impact (OECD, 2022[354]).

inefficient fossil fuel subsidies in part because of efforts to protect vulnerable households from surging energy price impacts (OECD, 2022[89]). During COVID-19 many countries also implemented carbon pricing policies encouraging increased emissions, however, most of these policies were time-limited (Nachtigall, Ellis and Errendal, 2022[90]). Furthermore, across 71 countries, fossil fuel support applied to approximately 22% of emissions (OECD, 2022[78]). The vast sums of support allocated to fossil fuels are likely to disrupt the carbon price signal. Thus, while current carbon pricing can contribute to reducing societies' fossil fuel reliance, it is unlikely alone to initiate a shift towards net zero (Rosenbloom et al., 2020[91]) (OECD, 2021[92]).

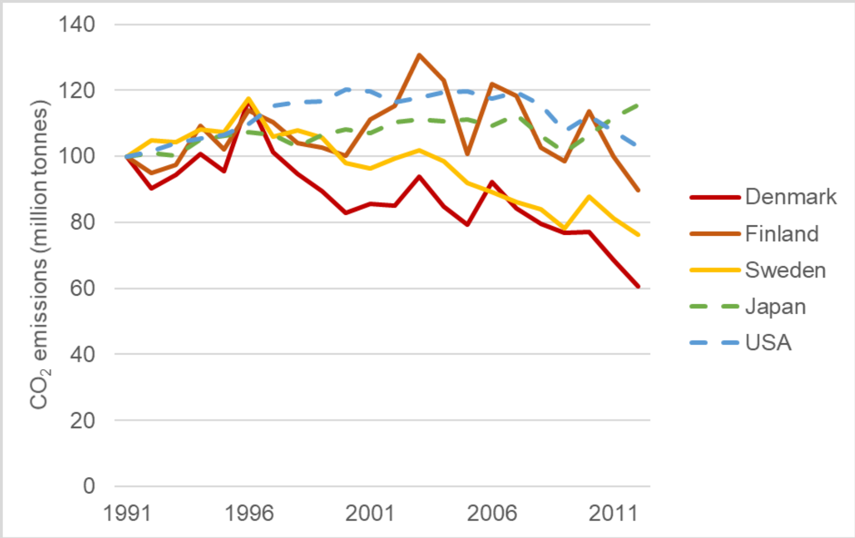
Despite historically low average explicit carbon prices and coverage, explicit carbon pricing is estimated to have contributed to reducing emissions to some extent. However, there is little ex-post evidence of the impact of carbon pricing on emissions reductions (World Bank, 2022[65]). A meta-review of 37 ex-post studies of carbon pricing since 1990 (mainly in Europe) found that carbon pricing schemes contributed to reducing GHG emissions in covered schemes by an average 0-2% per year (Green, 2021[93]). However, to keep the 1.5°C temperature target within reach, annual average emission reductions of 7.6% globally between 2020-30 are needed (UNEP, 2019[16]). Yet, (Green, 2021[93]) points out that there is considerable variation between the results of different studies, due to the use of different methods, different time periods and assessments of different sectors. Certain sectors show large emission reductions from carbon pricing.[43]

Carbon pricing may also have contributed to lower average annual growth in CO_2 emissions in countries with established carbon pricing schemes applied at the national level. In a cross-country study, Best, Burke and Jotzo (2020[82]) found that the average annual growth of CO_2 emissions overall, and not just in sectors covered by carbon pricing, was 2 percentage points (pp) lower in countries with established carbon pricing schemes applied at the national level than in countries without. The study also found that a carbon price increase of EUR 1/tCO_2 was associated with emissions reductions of approximately 0.3 pp in the following year (Best, Burke and Jotzo, 2020[82]). Figure 3.4 indicates how CO_2 emission trends of similar countries that implemented carbon pricing in the early 1990s[44] have declined more than countries without carbon pricing at the national level (e.g. USA) or those which implemented carbon pricing later (e.g. Japan in 2012). However, this analysis does not establish a causal relationship between carbon pricing and GHG emission reductions. Furthermore, policies other than carbon pricing could also impact emission reductions.

[43] For instance, German manufacturing firms reduced emissions by 25-28% (5-5.6% p.a.) compared to non-regulated entities during Phase II of the EU ETS. This was mostly due to energy efficiency improvements and a decline in the consumption of natural gas and petroleum products (Wagner, Rehdanz and Petrick, 2014[358]). In France, manufacturing firms covered under Phase II of the EU ETS reduced emissions by 13.5-19.8% (2.7-4% p.a.), mostly driven by fuel switching (Wagner et al., 2014[357]).

[44] Finland implemented carbon pricing in 1990, Sweden in 1991, and Denmark in 1992.

Figure 3.4. CO_2 emissions trends of selected countries with and without carbon pricing



Note: The y-axis uses the year 1991 as the Index year (equal 100). The x-axis presents the time range from 1991 to 2016. Finland put carbon pricing in place in 1990, Sweden did it in 1991, and for Denmark this was 1992. Japan put carbon pricing in place in 2012, while the USA has not put nation-wide carbon pricing in place.
Source: (OECD, 2022[94]).

3.1.3. Carbon pricing's impact on technological innovation and air pollution

Experience to date suggests that carbon pricing has contributed to technological innovations in some sectors, but this has mainly focused on short-term, low-cost changes rather than long-term, systemic changes. In theory, pricing GHG emissions creates incentives which encourage cost-effective decarbonisation. Furthermore, carbon pricing is also expected to incentivise innovation and the development of low-carbon technology. There is no consensus regarding the overall impact of carbon pricing on technology and innovation, yet available studies seem mostly to agree on a small carbon pricing impact.[45] Nevertheless, available evidence suggests that current carbon pricing levels will not alone be able to drive the long-term technology investments needed to achieve annual emissions reductions of 7.6% between 2020-30 (Lilliestam, Patt and Bersalli, 2020[95]) (Lilliestam, Patt and Bersalli, 2022[96]).

By helping to reduce emissions, carbon pricing could also contribute to co-benefits such as reduced air pollution,[46] yet ex-post studies confirming such an impact are currently limited and effects are difficult to disentangle. Since carbon pricing schemes help reduce fossil fuel combustion, they indirectly contribute to reducing local air pollutants and risks to human health, together with other policies such as stringent

[45] An ex-post analysis of several existing carbon pricing schemes found no to very little empirical evidence for increased technological change measured as increased innovation or zero-carbon investments (Lilliestam, Patt and Bersalli, 2020[95]). The same study (Lilliestam, Patt and Bersalli, 2020[95]) along with a more recent study (Lilliestam, Patt and Bersalli, 2022[96]) found that carbon pricing schemes had not triggered technological change, but instead had contributed to short-term effects, especially fuel switching. Another study utilising ex-ante and ex-post studies, found that carbon pricing schemes have had small but significant and positive impacts on low-carbon innovation (van den Bergh and Savin, 2021[344]). Carbon pricing's impact on technology in France (Gloriant, 2019[355]) and Switzerland (Ott and Weber, 2018[356]) were also insignificant. However, Calel and Dechezleprêtre (2016[345]) found that the EU ETS from 2005-10 increased low-carbon innovation by up to 10%, and that nearly 1% of the European low-carbon patenting increase in that five-year period could be attributed to the EU ETS.

[46] Air pollutants, such as as sulphur dioxide (SO_2), nitrogen oxides (NO_x), and fine particulate matter ($PM_{2.5}$), increases the risk of respiratory diseases, posing significant risks to human health (WHO, 2021[364]) (EEA, 2022[363]).

emissions standards. However, given that many policies ran in parallel with carbon pricing and contributed to emission reductions, it is difficult to disentangle the effects of individual policies.[47] In Europe, increases in air pollution are also found to cause substantial reductions in economic activity, due to reduced output per capita (Dechezleprêtre, Rivers and Stadler, 2019[97]). Well-designed policy packages including carbon pricing could help to address environmental challenges as well as improve selected issues related to health, reducing both GHG emissions and the economic burden of chronic health problems (OECD, 2012[98]).

3.1.4. Distributional impacts of carbon pricing and revenue recycling

Choices on how to recycle revenues from carbon pricing are important, and can help to address potential distributional effects of carbon pricing. Depending on how the carbon pricing scheme is designed (e.g. whether increased carbon prices for a specific good or service are offset by reduced energy or other prices for the same good or service), carbon prices can lead to increased prices of various carbon-intensive products. Such increased prices will lead to economic impacts for certain groups, especially if there is no low-carbon alternative. The distributional impact of carbon pricing differs between income levels, as well as between developed and developing countries.[48] Furthermore, there are also different impacts between households located in rural or urban areas.[49] These distributional impacts often lead to decreased public acceptability and opposition.[50] However, public support can be increased for instance by recycling carbon pricing revenues to help decrease distributional impacts (Borghesi and Ferrari, 2022[99]) (Maestre-Andrés, Drews and van den Bergh, 2019[100]).

The design of carbon pricing systems, and the use of revenues can influence public support and its environmental effectiveness. Carbon pricing exemptions can be used for instance to reduce the impact on vulnerable groups or businesses, yet this would also diminish carbon pricing's ability to reduce emissions.

[47] In the EU, a study found that the level of particulate matter (PM2.5) damage for EU ETS participating countries increased in Phase I, had no significant impact in Phase II and significantly decreased in Phase III (Wan et al., 2021[359]). Other measures implemented in parallel with the EU ETS also contributed to reducing pollutant levels, e.g. the 1979 Convention on Long-Range Transboundary Air-Pollution (LRTAP) (EEA, 2021[400]), and the 2010 European National Emission Reduction Commitment (NEC) Directive[47] (EC, 2016[365]). From the deployment of the EU ETS (and other policies) in 2005 until 2019, EU Member States jointly cut emission of sulphur dioxide (SO_2), nitrogen oxides (NO_x), and $PM_{2.5}$ by 77%, 42%, and 29%, respectively (EC, 2016[365]) (EEA, 2022[363]). In China, a carbon pricing scenario resulting in 4% CO_2 reductions annually between 2015-30, found that emission reductions from carbon pricing could be correlated with national pollution level reductions in SO_2 (25%), NO_X (19%) and $PM_{2.5}$ (4.7%) levels (Li et al., 2018[360]). Another ex-ante study (Zhang et al., 2021[361]) found that differentiated carbon pricing policies for various Chinese regions, could be associated with effectively alleviating PM2.5 pollution levels nationally. Furthermore, a carbon price of EUR 35/tCO2 by 2030 in China and India, is associated with an estimated annual reduction of 300,000 and 170,000 premature deaths (Perry, 2019[362]).

[48] In developing countries, high-income groups have a higher share of energy expenditure, compared to low-income groups. Thus, carbon pricing has mainly proportional or progressive impacts in developing countries (Dorband et al., 2019[342]) (Ohlendorf et al., 2020[341]). In developed countries, carbon pricing has regressive impacts as low-income households have higher shares of energy expenditure and could be more affected by carbon pricing than high-income households (Dorband et al., 2019[342]).

[49] Rural households often have higher levels of car-ownership, due to lower access to public transport. An explicit carbon price applied to fuel (or an implicit carbon price via fuel excise taxes) is thus likely to have greater impacts on rural households (Mattioli et al., 2019[366]) (Zhao and Bai, 2019[367]) (Gwilliam, 2013[368]).

[50] Examples of opposition have been seen in France with the yellow vest movement in 2019 following an increase in carbon prices on fuel (Douenne and Fabre, 2020[369]), in Mexico with citizens protesting after fuel prices were increased by 20% (Agren, 2017[370]), and in Alberta, Canada, where voters elected a conservative government in 2019 that promised to repeal the state-wide carbon tax (Rieger, 2019[403]).

Targeted and time-limited exemptions could therefore be used (Nachtigall, Ellis and Errendal, 2022[90]). Revenue recycling is another important design feature. Revenue recycling can help increase public acceptance of carbon pricing and help improve economic, social, and environmental outcomes. Many existing carbon pricing schemes earmark specific amounts of revenue for specific projects, while others allocate it for the general budget (World Bank, 2022[65]) (Santikarn, Kardish and Ackva, 2019[101]).[51] As ETS prices fluctuate, revenues vary from year to year, thus it can be challenging to consistently fund specific programmes or purposes solely from these revenues (Santikarn, Kardish and Ackva, 2019[101]) (Vaidyula and Alberola, 2015[102]). In contrast, carbon tax revenues are more stable and can be earmarked for specific purposes. However, a high proportion of carbon tax revenues tends to be allocated to the general budget.[52]

Carbon pricing revenues could be used to support the development of training and skills related to new job opportunities and could thereby support a just transition towards net zero. As discussed in section 2. , new activities and investments required to move to a net zero economy can create new job opportunities, which may require new skills (IEA, 2021[103]). Carbon pricing revenues could potentially be used to support the development of such new skills.[53]

3.2. Carbon pricing in the electricity sector and its contribution to changing emissions pathways towards net zero to date

Understanding how carbon pricing has affected emissions pathways in key sectors to date could provide insights on its potential contribution in the transition to net zero. Across the world, carbon pricing schemes have mainly been applied to the electricity and industry sector. The electricity sector produces a single output whereas the industry sector produces various products through a variety of different processes. Emissions from the electricity sector often originate from large point sources, making it easier to monitor and verify emissions. Furthermore, zero- and low-carbon electricity-generation alternatives are commercially available. It is therefore easier to apply carbon pricing to the electricity sector. In addition, the electricity sector represents an important component in the decarbonisation of other sectors via electrification of demand (e.g. increased penetration of electric vehicles can help in the decarbonisation of the transport sector) (Pathak et al., 2022[24]) (IEA, 2021[103]). It could be useful to better understand if, and how, carbon pricing has affected key developments in the electricity sector. Experiences with the EU ETS, California's Cap & Trade scheme and NZ ETS (see Box 3.2) including on coverage, price trends, GHG impacts, provide useful insights and are outlined below. The following section generally assesses impacts

[51] For example, in the EU, at least 50% of the ETS auction revenue must be used for climate and energy-related purposes, with 72% of revenues used for such purposes in 2020 (ERCST, 2022[402]). In Québec, all revenue is directed to the Green Fund that helps initiate climate change projects. Revenues from the RGGI are to benefit the whole economy, and to date have been invested in energy efficiency, renewable energy and direct bill assistance among others.

[52] For instance, in France and Sweden, the 2020 carbon tax revenue went to the general budget, while Japan's carbon tax revenue was earmarked for green projects or infrastructure (Postic and Fetet, 2021[401]).

[53] For example, in California's Cap and Trade scheme, 48% of proceeds are directed to California Climate Investment programmes to benefit priority groups, mainly disadvantaged and low-income communities (California Climate Investments, 2022[379]). Some of this revenue supports training for jobs in environmental sectors and industry transitions through worker-focused training that build skills and secure economic opportunities for priority populations (California Climate Investments, 2021[346]). Within the EU ETS, 2% of total allowances for the 2021-30 period is to go towards a Modernisation fund, which supports just transitions aspects, such as reskilling of workers in carbon-dependent EU regions (Modernisation Fund, 2022[375]). Some EU Member States may attribute further funds to just transition aspects including from the ETS revenues returned to them.

up until 2020, it does not take recent events in 2022, such as the European energy crisis onset by Russia's war against Ukraine, into account.

Box 3.2. Overview of Emission Trading Schemes in the EU, California, and New Zealand

- **EU ETS**: The EU ETS has been in operation since 2005 and is currently in its fourth Phase (2021-30). The system covers the power and industry sectors, as well as aviation within the European Economic Area and to/from Switzerland and the UK. The scheme covered approximately 39% of the EU's CO_2eq emissions in 2019. The price level remained at USD 50/tCO_2 for most of 2021 and peaked at around EUR 96/tCO_2 in early 2022. The price level for 2022 has generally been around EUR 80/tCO_2. Revenue generated from the auctioning of emission allowances is returned to Member States and at least 50% is earmarked for climate- and energy-associated purposes.
- **California's Cap & Trade scheme**: California's ETS, also called the Cap & Trade scheme was implemented in 2013. It applies to the power, industry, buildings, and transport sectors, and covered 74% of California's CO_2eq emissions in 2019. In most of 2021, the price remained above EUR 15/tCO_2, and in 2022 the price remained above EUR 25/tCO_2. Revenue generated goes to California's Greenhouse Gas Reduction Fund, where at least 35% of revenue is attributed to disadvantaged and low-income communities, as well as investments in climate mitigation projects.
- **New Zealand's ETS**: The NZ ETS was introduced in 2008. It covers the power, industry, building, transport, domestic aviation, waste and forestry sectors, equivalent to 49% of national CO_2eq emissions in 2019. In the first half of 2021, the carbon price was between EUR 20-30/tCO_2. From November 2021 to August 2022, the price level increased to be between EUR 40-50/tCO_2. Revenues from the NZ ETS is not earmarked for specific purpose, and it thus goes to the general budget.

Source: (ICAP, 2022[75]) and (ICAP, 2022[71])(accessed 2 November, 2022).

3.2.1. GHG emissions reductions in the electricity sector and the impact of carbon pricing

Electrification is an important element in decarbonisation pathways (Pathak et al., 2022[24]) (IEA, 2021[103]). If cleaner energy sources are available, carbon pricing has been found to contribute to increasing fuel switching, and, therefore, to reducing GHG emissions (Gugler, Haxhimusa and Liebensteiner, 2021[104]). In 2020, the global average carbon price required to switch from existing coal power to clean power was EUR 59/tCO_2. This section investigates if and how carbon pricing has contributed to reducing GHG emissions in the electricity sector.

Within the EU ETS, power sector emissions have overall been reduced by 43% in 2020 compared to 2005 levels, equivalent to a 2.9% annual decrease, with the greatest decline seen in Phase III (2013-20). Since the launch of the EU ETS in 2005, the power sector[54] has accounted for the largest share of both entities (at least 59%) and emissions (at least 66%) covered by the scheme (EEA, 2022[105]). In Phase III, the

[54] The term "power sector" here refers to combustion related GHG emissions, which covers emissions mainly from electricity and heat generation but also some manufacturing industries. Due to the lack of other data on solely electricity emissions, this aggregated dataset is used as a proxy indicator to assess the EU ETS' reduction of GHG emissions in the electricity sector (ETC/CME, 2021[343]).

power sector's total GHG emissions declined by 38% when comparing emissions from the first and the last year. These emissions reductions were mainly due to a combination of fuel switches encouraged by increased EU ETS prices in 2019 and 2020, increased competitiveness and penetration of renewable electricity systems, as well as lower demand onset by COVID-19. In Phase II (2008-12), power sector emissions declined by 9%, which overall was due to changes in the fuel mix (a shift to gas and increased renewable energy sources) and the Global Financial Crisis (GFC) (EEA, 2013[106]). The 2008 GFC led to low emissions in the starting year of Phase II, thus making it difficult to obtain large emissions reductions by 2012, the final year of Phase II. In Phase I (2005-07), the EU ETS pilot, total emissions increased by 2%, likely due to the oversupply of allowances leading to low prices and low mitigation incentives (EEA, 2011[107]). From 2005-21, the share of renewables in the electricity mix increased from 15.4% to 37.4%, and decreased for fossil fuels from 52.8% to 37% (European Commission, 2021[108]) (Ritchie, Roser and Rosado, 2020[109]). Between 2013-20, nearly 75% of revenue generated were used by Member States for climate and energy related purposes, primarily renewable energy deployment (EEA, 2021[110]). Policies other than the EU ETS, such as the Energy Efficiency Directive[55], the Renewable Energy Directive[56] and declining costs of renewables are also estimated to have contributed to this trend.

California's Cap and Trade scheme is estimated to having had the largest impact on emissions reductions in the electricity sector amongst several state-level climate policies. Between 2013, when California's Cap and Trade programme was implemented and 2019, GHG emissions from the electricity sector declined by 35.7%. In the same period, in-state and imported gas-generated electricity fell by 28.3% and 16.4%, while in-state and imported electricity generated from renewables increased by 80.6% and 51.6%, respectively (California Air Resources Board, 2021[111]). In the overall electricity supply, electricity generated from renewables increased from 21.9% to 33% between 2015-20. Amongst 17 state-level policies (8 climate and 9 energy policies) addressing CO_2 emissions in the electricity sector, the Cap and Trade scheme was found to have the largest impact on emissions reductions (Martin and Saikawa, 2017[112]). Other policies and developments are expected to also have delivered emissions reductions (IEA, 2020[113]) such as the Renewable Portfolio Standard, the Emissions Performance Standard (Martin and Saikawa, 2017[112]), federal tax credits for wind and solar and declining renewable energy prices (California Air Resources Board, 2021[114]) (California Energy Commissions, 2019[115]) (LAO, 2020[116]).

GHG emissions from the electricity sector decreased in New Zealand from 2010-19, yet the impact of the NZ ETS compared to other policy instruments cannot be assessed due to limited available data. The NZ ETS was launched in 2008 and the electricity sector[57] was first included in the scheme in 2010 (Environmental Protection Authority, 2022[117]). From March 2010 to March 2021, CO_2eq emissions in the electricity sector decreased by 36% (Ministry of Business, 2022[118]). The emissions reductions mainly stem from fuel switching, with coal and gas being replaced by renewables, mainly hydropower. From 2010-20, the share of electricity generated by renewables increased from 74.3% to 81.1%, fell for natural gas from 21.1% to 13.8%, and increased for coal from 4.4% to 5% (Ministry of Business, Innovation and Employment, 2021[119]) (Ministry of Business, Innovation and Employment, 2022[120]). The increase in the share of renewables has been driven by several policies including the 2011-21 National Energy Strategy and the 2011-16 Energy Efficiency and Conservation Strategy, complementing the NZ ETS, as well as decreased cost of renewables (IEA, 2017[121]). Given limited data assessing the impact of the NZ ETS in

[55] The Energy Efficiency Directive has helped promote energy efficiency by for instance requiring large operators to conduct energy audits every four years or requiring MS to take policy measures to generate energy savings by for instance promoting a change to renewable energy carriers (EC, 2021[373]).

[56] The Renewable Energy Directive, introduced in 2009, requires that by 2020, 20% of all energy in the EU's gross final energy consumption had to originate from renewable sources (European Parliament, 2021[372]).

[57] The electricity sector is in this paper referring to the stationary energy sector which includes all fossil fuels (gas and coal) used in electricity generation and in the direct production of industrial heat, as well as geothermal energy (Environmental Protection Authority, 2022[117]).

the electricity sector, it is currently difficult to estimate the emission reductions that can be attributed to the NZ ETS or other climate policies. However, low price levels in the NZ ETS between 2010 to the start of 2020 (under EUR 20/tCO_2) may not have created sufficient decarbonisation incentives (IEA, 2017[121]).

3.2.2. Carbon pricing's impact on electricity costs

Carbon pricing applied to a high-emitting electricity sector can increase the electricity costs for consumers, which can help curb demand but also lead to distributional impacts and discourage end-use electrification. Electricity is a relatively inelastic good in the short-term and more elastic in the long-term in developed regions such as the EU (Csereklyei, 2020[122]) and the US (Burke and Abayasekara, 2018[123]).[58] Pricing carbon in the electricity sector increases wholesale electricity prices, which producers can pass through to consumers leading to higher retail prices (Nazifi, Trück and Zhu, 2021[124]) (Fabra and Reguant, 2014[125]). However, in some countries (e.g. France) electricity prices can be highly regulated by governments, which may bias the overall impact and pass-through to consumers. In countries without such regulations, increased retail prices could help curb electricity demand to some degree. Yet, increased prices could also lead to distributional effects as electricity taxes tend to be regressive in developed countries. Furthermore, high electricity prices could discourage end-use electrification unless low-carbon electricity is available. Using carbon pricing revenues in a targeted manner could help address some of these impacts. The following paragraphs investigate if carbon pricing in the electricity sector has influenced electricity costs and if any distributional effects have been identified in the three jurisdictions explored in this paper.

Electricity prices in the EU have increased alongside EU ETS prices from 2012-20, and some studies identify a link between the EU ETS and price increases. From 2012-20, EU ETS prices increased by 28% from an average of EUR 7/tCO_2 in 2012 to an average of EUR 25/tCO_2 in 2020, equivalent to a 3.5% annual increase. However, from 2012-18, the EU ETS price level remained low (around EUR 7/tCO_2), while significant price increases were first seen from the beginning of 2018 (ICAP, 2022[71]). In the same period (2012-18) household electricity prices increased slightly from EUR 0.19/kWh to EUR 0.21/kWh (adjusted for inflation and including taxes) (Eurostat, 2022[126]). There are only few ex-post studies in this area, with one (Ahamada and Kirat, 2015[127]) linking the effect of increasing carbon pricing and increasing electricity prices in France and Germany.[59] As EU ETS prices reached record high levels at the start of 2022, and energy prices soared following the Russian invasion of Ukraine in early 2022, electricity prices and the distributional effects thereof are likely to have increased (Adolfsen et al., 2022[128]).

Electricity prices in California have increased significantly from 2013-20 due to a combination of factors, including but not limited to, the Cap and Trade Scheme. From 2013-20, the Californian carbon price increased from EUR[60] 10/tCO_2 to EUR 16.5/tCO_2 (ICAP, 2022[71]). In the same period, residential electricity prices slightly increased from EUR 0.16/kWh to EUR 0.20/kWh (EIA, 2022[129]). Nevertheless, in 2020, the average residential price for electricity in California was the sixth highest in the US, with an electricity price 60% above the average price of all other states (EUR 0.13/kWh) (California Center for Job & the Economy, 2020[130]). From 2013-20, residential electricity consumption in the state grew by 6.7%, from 0.089 Gigawatt hour (GWh) to 0.095 GWh (EIA, 2022[129]). Compared to 2010, the average residential electricity price in California in 2020 had grown by 32.6%, whereas the electricity bill for other states only grew by 3.2% over

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[58] The long-term elasticity is due to the ability of consumers to change behaviour and invest in energy efficiency measures (Cleary and Palmer, 2020[374]).

[59] This study found that from 2008-10, a 1% price increase in the EU ETS resulted in 0.19% - 0.21% price increase in French electricity prices and 0.13% - 0.14% higher electricity prices in Germany (Ahamada and Kirat, 2015[127]). This could be due to the fact that the EU ETS is not the only price applied to electricity, and other taxes and levies (excises, VAT, renewable energy levies, capacity levies, environmental taxes, etc.), which differ between EU Member States, are also likely to affect electricity prices (EC, 2020[371]).

[60] Exchange rate: USD 1 = EUR 1 (assessed on 27 October 2022) (Google finance, 2022[353])

the same period (California Center for Job & the Economy, 2020[130]). This significant increase in prices of Californian electricity is due to increased electricity consumption and state programs such as the Cap and Trade scheme, the Renewable Portfolio Standard and other policies. Moreover, the coverage of electricity generators' fixed costs occurs through volumetric rates in California. In recent years, electricity retail sales have declined, due the use of increased solar power, especially in wealthier households. This has made it harder for electricity generators to cover fixed costs and has resulted in increased electricity prices (LAO, 2020[116]). The high electricity prices in California are likely to have distributional effects caused by multiple factors including the recovery of fixed costs, which has been shifted to non-solar power users who are often less wealthy households (Borenstein, Fowlie and Sallee, 2021[131]).

In New Zealand, electricity and NZ ETS prices increased from 2009-20. By July 2009, the NZ ETS price was approximately EUR 9/tCO_2, by July 2015 it had dropped to around EUR 4/tCO_2 and then increased to about EUR 17/tCO_2 by 2020 (ICAP, 2022[71]). In the same period, average residential electricity rates (adjusted for inflation) increased by approximately 10%, from EUR 0.16/kWh (NZD[61] 0.28/kWh) in 2009 to EUR 0.18/kWh (NZD 0.31/kWh) in 2020 (Ministry of Business, Innovation & Employment, 2022[132]). The increase in electricity prices could be linked with the increasing price level of the NZ ETS, however, no ex-post assessment indicating this could be found. A 2010 report found that the NZ ETS had not caused any noticeable changes in electricity prices and that retail electricity prices in New Zealand increased regularly before the ETS was implemented (Ministry for the Environment and the Ministry for Primary Industries, 2012[133]). Furthermore, average annual residential expenditure on electricity per household only slightly increased (1.6%) from EUR 1,260 (NZD 2,172) in 2009 to EUR 1,280 (NZD 2,208) in 2020 while the electricity consumption per household decreased by 7.4% (Ministry of Business, Innovation & Employment, 2022[132]). This suggests that increasing prices contributed to curbing electricity demand – however, no causal relationship has been established.

Available data analysed indicates that carbon pricing has, together with other policies, contributed to various impacts to date, yet, further research disentangling the direct carbon pricing impact is needed. Data disentangling the impact of carbon pricing is limited and it remains difficult to fully assess the impact of carbon pricing in the electricity sector. However, across the three schemes analysed, available data suggests that carbon pricing contributes to reducing emissions, while the use of revenues from carbon pricing can help increase the share of renewables. Yet, carbon pricing does not generate these effects on its own, but rather in combination with other policies. This suggests that carbon pricing on its own is not sufficient to bring about the emissions reductions needed to transform emission pathways, to reach net zero by 2050 and to limit global warming to 1.5°C this century (Rosenbloom et al., 2020[91]) (OECD, 2021[92]). Moreover, if carbon price levels are higher, for example set according to the social cost of carbon, it could further increase the potential of carbon pricing to trigger behavioural change and emission reductions (see section 3.1). However, it does suggest that carbon pricing's effectiveness increases together with other policies. Further research and analysis exploring the effectiveness of carbon pricing in the electricity sector and other key sectors could be helpful in determining carbon pricing's contribution to transforming emissions pathways. In addition, an exploration of the interaction and effectiveness of carbon pricing alongside different policies could be helpful in determining the policy mix which generates the greatest emissions reductions and co-benefits in the electricity sector without exacerbating distributional impacts.

[61] Exchange rate NZD 1 = EUR 0.58 (accessed on October 29 2022) (Google finance, 2022[353]).

4. Options for improving the contribution of carbon pricing in pathways to net zero

Key messages

Reaching net zero emissions will require comprehensive policy packages that simultaneously address both demand- and supply-side changes. Current climate policies mainly focus on the supply-side and there is significant potential to reduce GHG emissions through policies focused on influencing demand. Although challenging to implement, policies that enable demand-side changes can also contribute to improving living standards and support delivery of different SDGs.

Carbon pricing could play an important role in helping to incentivise demand-side shifts as part of a wider policy package. For example, carbon pricing can help to avoid/reduce consumption of carbon-intensive goods or improve the carbon intensity of existing goods through careful revenue recycling.

Sequencing the deployment of different policies so that preparatory policies, such as green industrial policies are deployed first, can help set the ground for subsequent policies such as carbon pricing. Careful sequencing of policies can help to reduce potential barriers, facilitate implementation and support a just transition to net zero.

Increased international co-operation and collaboration on carbon pricing could incentivise more ambitious carbon pricing by building political and public support, helping to overcome obstacles and encouraging exchanges of good practices.

Reaching net zero emissions will require a mix of policies that go beyond traditional climate mitigation policies to simultaneously address both supply-side and demand-side changes. Carbon pricing can help create enabling conditions to incentivise demand-side shifts for a just transition to net zero. Sequencing the deployment of specific policies as well as increased international co-operation and collaboration on carbon pricing could help to reduce or remove potential barriers in the path to net zero emissions. This section investigates different options for improving the contribution of carbon pricing in reaching net zero and limiting global warming to 1.5°C in this century.

4.1. The need for both demand- and supply-side changes to reach net zero

Policies targeting demand changes are needed alongside supply-side policies to reach net zero and limit global warming to 1.5°C (Creutzig et al., 2022[134]). However, current policies, including carbon pricing, mainly focus on the supply-side, and have contributed to numerous supply-chain innovations and efficiency measures (IPCC, 2018[135]). Despite these developments, the demand for energy and GHG-intensive products increased across all sectors from 2010-20, offsetting emission reductions in the supply-side and resulting in increased GHG emissions overall (IPCC, 2022[136]).

Policies focused on influencing demand could reduce overall emissions in end-use sectors[62] by 40-70% (Creutzig et al., 2022[134]) and help to relieve the decarbonisation pressure on the supply-side. According to the IPCC, "Rapid and deep changes in demand make it easier for every sector to reduce GHG emissions in the short- and medium-term" (Creutzig et al., 2022, p. 3[134]). Scenario analyses by (Grubler et al., 2018[137]), (Burke, 2020[138]) and (Keyßer and Lenzen, 2021[139]) indicate that reduced demand can contribute to meeting Decent Living Standards (DLS)[63] while keeping global warming below 2°C and limiting reliance on negative emissions technologies. The emissions reduction potential differs across sectors - 28.7% in the industry sector, 44.2% in the food sector, 66.8% in the land transport sector and 66% in the buildings sector (Creutzig et al., 2022[134]). The potential for demand-side emission reductions also differs between and within regions. Some regions have scope to reduce energy demand and resources, while others will require additional energy and resources to fulfil human well-being needs (IPCC, 2022[17]).

Enabling demand-side changes can also contribute to improving living standards and support the delivery of different SDGs. Using more efficient technologies and enabling conditions to shift demand to low-emission options (e.g. public transport, low GHG emissions diets), could reduce the average energy requirement for DLS in 2050 to 15.3 GJ/capita per year.[64] This significant reduction is more than 60% below current energy uses per capita globally (Millward-Hopkins et al., 2020[140]), and 75% lower than the energy use per capita in the IEA's 2050 Stated Policies estimate[65] (IEA, 2019[141]). Grubler et al (2018[137]) find that a low-emissions demand scenario could contribute to an improvement in fulfilling multiple SDGs.[66] OECD (2022[142]) finds that policies focused on systems transformation (or redesign) can allow people to meet their needs with low consumption levels, thus helping to align consumption patterns with planetary boundaries and resource availability (O'Neill et al., 2018[143]) (Toth and Szigeti, 2016[144]).

Table 4.1. Overview of the Avoid-Shift-Improve concept and potential role of carbon pricing

	Avoid	Shift	Improve
Description	**Avoid** or reduce consumption of carbon-intensive goods or services	**Shift** to less carbon-intensive goods or services	**Improve** the carbon-intensity of goods or services
Area of change	**Social-cultural factors** to modify individual choice, social norms, and culture	Changes in **infrastructure** and nudging can help create opportunity spaces that could shape and shift people's preferences	**Adoption** of technological improvements
Carbon pricing's role	Increase price of good or service that is to be avoided/reduced to discourage consumption	Increase price of carbon-intensive good to encourage the shift to low-carbon alternatives	Carbon pricing revenue could fund the improvement of existing technologies

Source : (Creutzig et al., 2018[145]) (Creutzig et al., 2022[134]).

[62] End-us sectors include transport (Mobility), buildings, industry, and agriculture (Pathak et al., 2022[404]).

[63] Decent living standards (DLS) refer to a set of universal service requirements deemed essential for basic human well-being, and includes aspects of nutrition, shelter, living conditions, clothing, health care, education and mobility (Rao and Min, 2017[376]) (Isobel et al., 2018[377]).

[64] To provide DLS, the average energy requirement is estimated to be between 13-18.4 GJ/capita per year.

[65] Outlining the expected trajectory if today's commitments are met in full and maintained.

[66] E.g. SDG 2 (reduced risks of hunger); SDG 3 (reduced air pollution); SDG 7 (improved access to modern energy forms as traditional biomass is phased-out[66]); SDG 13 (reduced emissions); SDG 14 (reduced ocean acidification); and SDG 15 (biodiversity benefits)

Carbon pricing could play an important role in incentivising demand-side shifts, yet its effect depends on the price level, the responsiveness of consumer demand (elasticity) to pricing levels, the availability of alternatives, and implementation of carbon pricing as part of wider policy packages (see Box 4.1). The Avoid-Shift-Improve (ASI) framework (Creutzig et al., 2018[145]) (European Environment Agency, 2009[146]) (Creutzig et al., 2022[134]) indicates the type of demand changes that carbon pricing could help to trigger (see Table 4.1). *Avoid* options focus on consumption avoidance or reduction. If this is not possible, *Shift* options could be considered, which focuses on the move to less carbon-intensive methods. If shifting is not an option, *Improve* options could be considered, focusing on reducing emissions of existing products or services (Creutzig et al., 2018[145]).

Box 4.1. Carbon pricing could contribute to Avoid-Shift-Improve policy options

Avoiding or reducing consumption

In the Swedish flight-shaming ("flygskam") movement, several factors, including an implicit carbon tax on flights, triggered changes in the social norm around flying. Before the Swedish flight tax was implemented in April 2018, it was heavily debated in the Swedish media, and at the same time, the flight shame movement on social media gained traction (Jacobson et al., 2020[147]). On 1 April 2018, the flight tax[67] (i.e. a distance-based ticket tax for commercial passenger planes departing from Sweden) was implemented. The tax started at EUR 6/passenger (SEK 60) for flights within Europe and ended at EUR 40/passenger (SEK 400) for flights beyond 6000 km (Regeringskansliet, 2017[148]) (Stråle, 2021[149]). In the first quarter after implementation, the number of international passengers flying from Swedish airports decreased by 4% (almost 200,000 fewer passengers). In the third quarter of the following year (2019), the effect of the tax continued, leading to an overall 11% decrease in international passengers (around 560,000 fewer passengers) (Stråle, 2021[149]). In 2018, the year the tax was implemented, Swedish train journeys increased by 1.5 million (SJ, 2018[150]), and by 2.5 million in 2019 (SJ, 2019[151]). Together, the flight-shaming movement, the tax and availability of a transport alternative in trains contributed to socio-cultural changes around flying (Jacobson et al., 2020[147]) (Stråle, 2021[149]).

Shifting people's preferences and behaviour

Redesigning urban areas can, together with a carbon price, contribute to more sustainable transport Redesigning urban areas can, together with a carbon price, contribute to more sustainable transport systems, in which active, public, and shared transport modes are the most attractive options, and lead to GHG emission reductions (OECD, 2021[92]). A process prioritising street redesign, reallocating road space to active and public transport modes, and shared mobility in urban areas could create alternatives to private vehicles in these areas. Such a redesign creates an opportunity space where lower-emissions alternatives are more appealing than higher-emission options. Thus, when combined with policies aimed at increasing the attractiveness of sustainable modes vis-à-vis private cars, carbon prices can accelerate transport modal shifts and the transition towards sustainable transport systems (OECD, 2021[92]; OECD, 2022[142]) (Gillingham and Munk-Nielsen, 2019[152]). If implemented in isolation, carbon pricing's effect on shifting demand away from private car use can be limited (Geman, 2019[153]) (Gillingham and Munk-Nielsen, 2019[154]). This is because, without policies that increase the availability and attractiveness of sustainable modes, private cars remain the most attractive, and sometimes the only option for people to meet their daily needs (OECD, 2021[92]). The political feasibility of introducing high pricing is also low in a context where convenient alternatives are not available.

[67] A flight tax is an implicit carbon tax that prices flights, which are associated with GHG emissions, yet not directly equivalent to the carbon content of the flight journey itself.

Improving carbon-intensity

Carbon pricing can foster the purchase of more efficient technologies[68] and improve technological performance through revenue recycling. Carbon pricing revenue could, for instance, be earmarked to research and development (R&D) for technology improvement within transport (e.g. vehicles and fuels) or the building sector (e.g. heating systems and energy efficiency) (Creutzig et al., 2022[134]). In Canada, carbon pricing revenue is recycled to citizens via the Climate Action Incentive Payment, encouraging households to invest in energy efficiency measures (Government of Canada, 2022[155]). Additional revenue amounts are also allocated to the Future Electricity Fund, that aims to support clean energy projects (Government of Canada, 2022[156]). An investigation of 40 OECD and G20 countries' revenue use in 2016 showed that several countries use ETS revenue to develop, improve and increase public accessibility to electrified mobility. Some countries use ETS revenues for subsidising the consumption of renewable energy, while others use it to subsidise the renovation and retrofitting of buildings and homes. Carbon tax revenues were also used to fund energy efficiency measure across the 40 countries, yet the amounts allocated for such purposes were much smaller (Marten and van Dender, 2019[157]).

Public support varies widely for different types of demand-side measures, as well as between different countries. The OECD surveyed more than 40,000 respondents in 20 high-income countries (HIC) and middle-income countries (MIC) (emitting 72% of global CO_2 emissions) about their attitudes toward climate policies. The survey showed that between the policy categories investigated, carbon or emissions pricing policies received the lowest amount of support[69] (see Figure 4.1). The survey also highlighted the importance of alternatives. For instance, support for a ban on fossil fuel car production was higher (by 8 pp)[70] when transportation alternatives were available (Dechezleprêtre et al., 2022[22]).

Incorporating carbon pricing in wider policy packages, and careful use of carbon pricing revenues, can increase effectiveness and help incentivise demand-side changes throughout society. Support for carbon pricing is highly correlated with decisions on how the revenue from carbon pricing is used (Maestre-Andrés, Drews and van den Bergh, 2019[100]) (Klenert et al., 2018[158]). In the OECD survey (Dechezleprêtre et al., 2022[22]), high support was found for using revenues to fund environmental infrastructure (HIC 63%, MIC 75%), subsidise low-carbon technologies (HIC 63%, MIC 73%) or reduce income taxes (HIC 57%, MIC 69%). Public support is lower for using revenue to reduce the public deficit (HIC 48%, MIC 63%) or corporate income taxes (HIC 37%, MIC 58%) and making equal transfers to all households (HIC 38%, MIC 61%). Certain factors increased respondents' willingness to change behaviour, such as receiving financial support to make the necessary changes (Dechezleprêtre et al., 2022[22]).

Policy packages that address multiple objectives could support low-carbon transitions in a socially just manner, by enabling synergies and minimising trade-offs (Nilsson et al., 2021[159]) (Lecocq et al., 2022[25]). For instance, implementing a carbon tax on transport without other policies or alternatives would have negative effects on suburban households, which may be car-dependent, and have little or no access to public transport. A policy package that includes an expansion of safe, low-carbon transport modes and affordable urban housing could improve the effectiveness of a carbon tax while contributing to increased

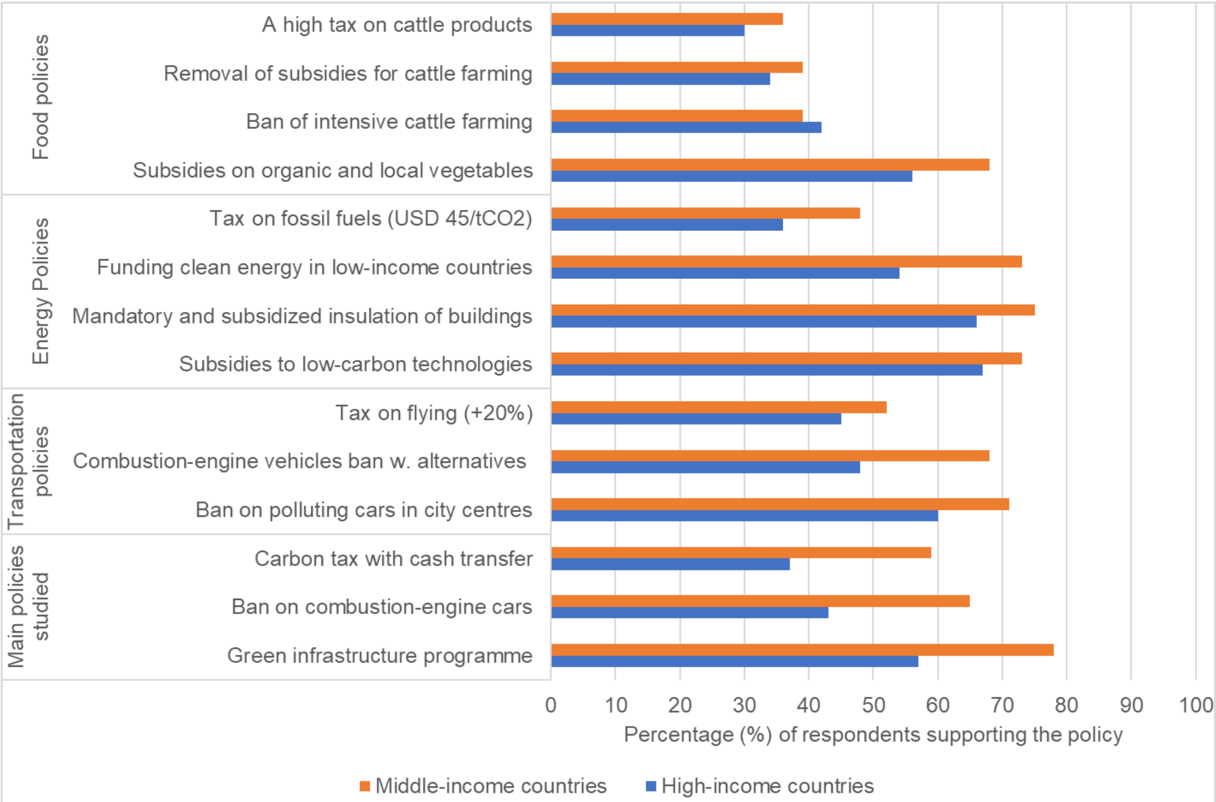
[68] In the transport sector, while carbon prices can accelerate the purchase of fuel-efficient or electric vehicles (Leard, Linn and Cleary, 2020[399]), evidence suggests that the emission reductions gained via better vehicles may be off-set by the rebound effect (Dimitropoulos, Oueslati and Sintek, 2016[419]).

[69] A high tax on cattle products (HIC 30%, MIC 36%), a fossil fuel tax of USD 45/tCO_2 (HIC 36%, MIC 48%), and a carbon tax with cash transfers (HIC 37%, MIC 59%) received lower levels of support.

[70] In the survey, 28% supported a ban on fossil fuel cars with an alternative, while this was 20% for a ban without alternatives (Dechezleprêtre et al., 2022[22]).

housing access and reduced air and noise pollution. Such an approach could mitigate emissions at lower economic and social costs (Lecocq et al., 2022[25]).

Figure 4.1. Levels of support for selected climate change policies



Source: Authors based on data from (Dechezleprêtre et al., 2022[22]).

4.2. Policy sequencing to improve carbon pricing's effectiveness

Sequencing the deployment of different policies can help reduce or remove potential barriers and support a just transition to net zero. Careful sequencing of different policies, as part of wider policy packages, can help to enhance policy effectiveness and increase overall policy stringency (Pahle et al., 2018[160]). For instance, as a first step, preparatory policies can be deployed to set the ground for the next round of policies in the package. Preparatory policies often help to reduce or remove barriers, lower the cost of specific policies, communicate the pros and cons of future policies to the public or help build coalitions around the subsequent policy, thus facilitating implementation (Meckling, Sterner and Wagner, 2017[161]).

Different types of policies have been used in several countries to prepare the ground for the subsequent introduction of carbon pricing. Linsenmeier, Mohommad and Schwerhoff (2022[162]) found that for 37 countries, including G20 economies, more than half used other policies[71] prior to implementing carbon pricing across five sectors (energy, buildings, industry, transport, agriculture). The largest number of policies applied before carbon pricing was found in the transport (3.09) and energy (1.55) sectors

[71] These instruments included regulatory instruments; grants, subsidies, and other financial incentives; procurement and investment; research, development, and deployment; voluntary agreements; information and education; and policy support (Linsenmeier, Mohommad and Schwerhoff, 2022[162]).

(Linsenmeier, Mohommad and Schwerhoff, 2022[162]). Meckling, Sterner and Wagner (2017[161]) found that of all jurisdictions that have carbon pricing in the power sector, most (65%) implemented green industrial policies (i.e., a feed-in tariff[72] or a renewable portfolio standard)[73] before carbon pricing to develop green alternatives. Similarly, 58% of jurisdictions that adopted carbon pricing in the transport sector implemented green industrial policies (e.g. electric vehicle incentives or biofuel mandates) before the introduction of carbon pricing (Meckling, Sterner and Wagner, 2017[161]). Once alternatives are in place, carbon pricing can help discourage the use of carbon-intensive goods or services and encourage the uptake of alternatives (see examples in Box 4.2).

Box 4.2. Country examples of observed policy sequencing with carbon pricing

EU

Policy sequencing with carbon pricing have been observed for the electricity sector within the EU. Preparatory policies were first deployed in 1996, in the form of measures to liberalise the EU electricity market (Ciucci, 2021[163]). These were followed by other preparatory policies in the form of two Directives. In 2001, Directive 2001/77/EC, promoting renewable energy in electricity generation (EC, 2001[164]) was adopted. In 2003, Directive 2003/30/EC, legislating the replacement of 5.75% of fossil fuels for transport (petrol and diesel) with biofuels by 2010 was approved (EC, 2003[165]). These directives helped lower barriers and created greener energy alternatives. This was followed in 2005 with the launch of the EU ETS phase I covering industry and power-generating units. The pricing scheme helped encourage the shift to green energy alternatives (EC, 2021[166]). Since then, the EU has ratcheted up its policies and the EU ETS. For instance, in 2008, the EU ETS entered phase II, which had new and stricter features (EC, 2021[166]). In 2009, the legislation of the 2020 Climate and Energy package increased energy requirements (20% renewable energy, 20% energy efficiency increase, and 20% GHG emissions reduction compared to 1990 levels by 2020). (EC, 2020[167]). In 2014, this policy was further ratcheted up with the 2030 Climate and Energy Framework (32.5% renewable energy, 32.5% energy efficiency increase, and 40% GHG emissions reductions compared to 1990 levels by 2030) (EC, 2021[168]) (Meckling, Sterner and Wagner, 2017[161]).

China

In China, policy sequencing with carbon pricing has also been detected for the energy sector during the mid-2000s. Preparatory policies were first deployed in 2005, as China passed a Renewable Energy Law. The law provided mandates and financial incentives (e.g. feed-in tariffs) to encourage renewable energy development and deployment in power generation, buildings and transportation. The feed-in tariffs helped significantly increase wind power capacity from 2005-10. In 2009, the Renewable Energy Law was amended to include a requirement of power generators to accept all wind power generated. Together, these policies helped develop clean energy alternatives resulting in lower economic barriers. This furthermore, created enabling conditions for effective carbon pricing in the power sector, which was implemented in the form of several pilot ETS in 2017 (Qi and Wu, 2013[169]) (Meckling, Sterner and Wagner, 2017[161]). Similar examples have also been seen with renewable energy policies and carbon pricing in California and Germany (Pahle et al., 2018[160])

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[72] Feed-in tariff (FIT) is a price-driven policy designed to accelerate investment in renewable energy technologies by offering guaranteed prices for electricity produced from renewable energy sources for fixed time periods (Mendonça, 2012[350]).

[73] Policies designed to increase the use of renewable energy sources for electricity generation (US EIA, 2021[349]).

4.3. Enhancing international co-operation and collaboration on carbon pricing

Enhanced international co-operation and collaboration could encourage more ambitious carbon pricing approaches in pathways to reach net zero. Enhanced co-operation between countries could help facilitate political and public support for carbon pricing, overcome obstacles to progress (e.g. competitiveness concerns), encourage exchanges of good practices, and support more efficient, effective and ambitious approaches (Withana et al., 2014[170]). A 2021 ex-ante study for the Carbon Market Platform (CMP) found that international co-operation on carbon pricing could deliver economic and environmental benefits while reducing carbon leakage. These benefits grow with the number of participating countries, emissions and sectoral coverage and alignment with ambitious climate goals (Nachtigall et al., 2021[171]).

To date, there are some limited examples of international co-operation and collaboration on carbon pricing, mainly among established ETS. One approach to increase international collaboration on carbon pricing is through linking ETS in different jurisdictions, e.g. Cap and Trade systems in California (USA) and Québec (Canada) (ICAP, 2022[172]), and the EU ETS and Switzerland's ETS (Council of the EU, 2019[173]). Linking different ETS can help to improve cost-efficiency by increasing the number of mitigation options with low abatement costs. It can also increase the liquidity through an extended market (Ranson and Stavins, 2015[174]). Moreover, linking can reduce negative competitiveness impacts as carbon leakage is likely to be smaller when sectors in other jurisdictions face the same carbon price (Kachi et al., 2015[175]). Another example of current collaboration on carbon pricing is the participation of Iceland, Norway and Liechtenstein in the EU ETS (EEA, 2007[176]).

Enhanced international co-ordination through a carbon price floor could help to address competitiveness concerns and reduce pressure for unilateral border carbon adjustments. A well-designed and flexible international carbon price floor (ICPF) for major emitting countries, as proposed in a 2021 IMF staff paper could effectively reduce emissions and address competitiveness concerns (Parry, Black and Roaf, 2021[177]). A 2022 IMF staff paper found that expanding an ICPF to all countries simultaneously, with tiered price floors based on income levels would reduce GHG emissions sufficiently to achieve the 2°C goal (Chateau, Jaumotte and Schwerhoff, 2022[178]). Such international co-ordination could help to avoid sub-optimal situations such as pressures to impose carbon border adjustments (CBA) to reduce competitiveness concerns (Chateau, Jaumotte and Schwerhoff, 2022[178]).

To increase international co-operation and collaboration on carbon pricing, informal partnerships could help to build political momentum, share best practices, and drive more ambitious action. For example, at COP26, Canadian Prime Minister Justin Trudeau issued a call for 60% of emissions to be covered by a carbon price by 2030. To achieve this goal, Canada, together with Chile launched the "Global Carbon Pricing Challenge" at COP27. This partnership, backed by the UK, New Zealand, Sweden, and others, aims to expand the use of carbon pricing (Government of Canada, 2022[179]) (Manuell, 2022[180]). The Global Carbon Pricing Challenge also intends to support countries in the adoption of carbon pricing approaches, including through the World Bank's Partnership for Market Implementation (PMI) initiative (PMI, 2021[181]). Other examples of international co-operation around carbon pricing include the G7's proposed "Climate club", which is built on three pillars including advancing ambitious climate mitigation policies, such as explicit carbon pricing (G7, 2022[182]).

5. Exploring the potential role of emissions pricing in pathways to net zero food systems

Key messages

Reducing GHG emissions from food systems will be an essential step towards net zero, however decarbonising all stages of food systems is far from straightforward. GHG emissions from food systems encompass multiple sectors and gases. To date, emissions pricing is only marginally used in global food systems, covering a small part of GHG emissions.

To help reduce GHG emissions from food systems, an emissions pricing scheme could be applied via the polluter-pays (ETS or carbon taxes) or the beneficiary-pays principle (abatement payments and offsets). Such a scheme could help to encourage production of individual food products to become more efficient and/or encourage shifts to foods with a lower GHG intensity.

There is a high variability in emissions intensity between and within different food products, both across and within geographies. It can be technically challenging to monitor, report and verify emissions for a specific food product across its lifecycle. This makes it technically and methodologically challenging to provide an accurate price signal, at least in the short term.

Implementing emissions pricing in food systems also has significant short-term political barriers and potentially some negative just transition implications, e.g. reductions in agricultural employment in specific sub-sectors such as livestock production, for farmer income, and for food security. Given the challenges of food systems emissions pricing, some countries are likely to prioritise the use of non-pricing policies in this area.

Reducing GHG emissions from food systems could also have many co-benefits and potential policy avenues on the supply- and the demand-side could usefully be further explored.

Producing sufficient and nutritious food for a growing population, while sustaining livelihoods in an environmentally sustainable manner represents the triple challenge of food systems (OECD, 2021[183]). In 2021, around 29.3% of the global population (2.3 billion) were moderately or severely food insecure, and almost 10% suffered from hunger (702-828 million) (FAO; IFAD; UNICEF; WFP; WHO, 2022[184]). These figures vary widely by region.[74] Food systems also support the livelihoods of millions of people around the world, employing more than half of the population in Sub-Saharan Africa (52.9%) (World Bank, 2022[185]), and accounting for around thirty million jobs in the agriculture sector in OECD countries (OECD, 2023[186]). Countries' policy priorities in food systems to date have often focused on maintaining food security and

[74] For example, around 23% of the African population suffered from severe food insecurity in 2021, compared to 1.5% of the population in Northern America and Europe[74] (FAO; IFAD; UNICEF; WFP; WHO, 2022[184])

supporting livelihoods but doing so has led to environmental impacts. While there has been important progress and emissions reductions in some areas, there is potential for additional action that could further help reduce emissions from food systems while also maintaining food security and livelihoods (OECD, 2022[187]).

Food systems are vulnerable to the effects of a changing climate while at the same time being a major contributor to climate change (Owino et al., 2022[188]). The IPCC estimates that the current policies in the agriculture, forest and other land-use (AFOLU) sector, along with the current policies of other sectors, could lead to global warming of 3°C by 2050. At the same time, the sector is estimated to be able to contribute 20% - 30% of emission reductions needed in 2050 to keep global warming between 1.5°C - 2°C (Nabuurs et al., 2022[189]). This section explores the current and potential role of emissions pricing in reducing GHG emissions from food systems, along with its challenges.[75]

5.1. Unpacking GHG emissions from food systems

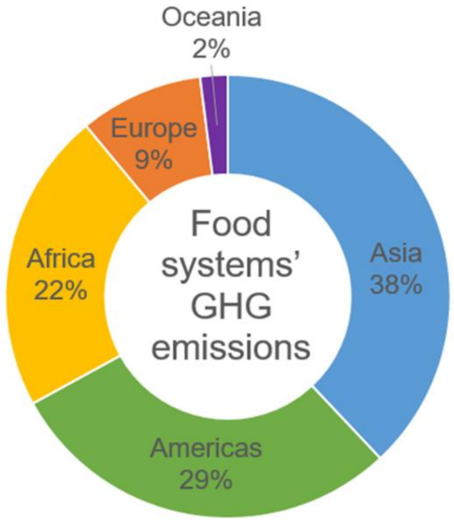
Direct and indirect emissions from food systems are increasing and are estimated to account for around one-third of global GHG emissions. Emissions from food systems increased by 16% from 14.5 $GtCO_2eq$ per year in 1990 to 16.8 $GtCO_2eq$ per year in 2018 (Crippa et al., 2021[190]) (FAO, 2021[191]) (Babiker et al., 2022[192]), accounting for 31% (range 23-42%) of global GHG emissions in 2018 (Babiker et al., 2022[192]).[76] More than a third of total agricultural emissions originated from Asia (38%), followed by the Americas (29%)(see Figure 5.1). Direct GHG emissions from agricultural production include methane (CH_4) and nitrous oxide (N_2O) emissions from food production, livestock, and manure management. Indirect emissions include CO_2 emissions from deforestation and energy-related emissions associated with transport, processing and packaging of foods. Of total food systems emissions globally, CO_2 represented 46%, CH_4 38%, and N_2O 13%, while fluorinated gases (F-gases)[77] represented 3% in 2018 (Babiker et al., 2022[192]) (Crippa et al., 2021[190]). This section explores only CO_2, CH_4 and N_2O emissions, and not F-gases. Between 2000-14, the emissions intensity of global food production decreased from 0.68 gCO_2eq per USD to 0.46 gCO_2eq per USD (Mrówczyńska-Kamińska et al., 2021[193]). This decrease could be due to structural shifts in various countries, such as replacing large shares of high emissions intensive ruminant livestock with lower-emissions intensive poultry as well as to technological advancements, and/or increases in the efficiencies of food production.

[75] This report analyses the potential for emissions pricing policies to support emissions reductions in food systems. While acknowledging that there are other possible policies to address GHG emissions from food systems, a detailed analysis of these are outside the scope of this paper. Given the technical and political challenges associated with implementing an emissions pricing system in food systems that accurately reflects the GHG-intensity of a specific food item, could mean that some countries are likely to prioritise the use of non-pricing policies to address emissions in this sector.

[76] This is down from 44% in 1990 (Babiker et al., 2022[192]), due mainly to an increase in non-food emissions as well as a decrease in the emissions intensity of food, in particular from land-use change (Crippa et al., 2021[190]).

[77] In food systems F-gases mostly originates from refrigeration in the retail stage (Crippa et al., 2021[190]).

Figure 5.1. Total emissions related to agriculture, forestry, and other land-use across world regions in 2019

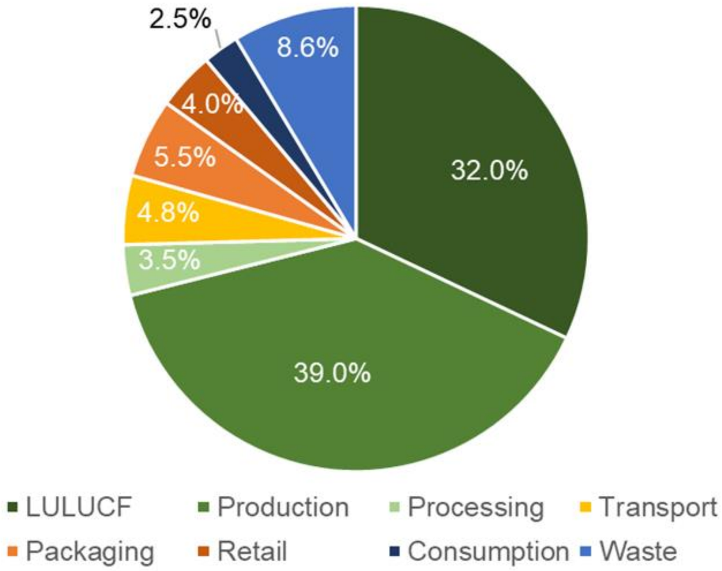


Note: Total emissions in this figure relate to parts of emissions from food systems, namely the ones from agriculture, forestry and other land-use across world regions.
Source: Authors based on data from (FAO, 2021[194]).

Both the level and the importance of GHG emissions from food systems varies across countries, as well as across different stages of the food system, with most emissions globally originating in the land use, land-use change and forestry (LULUCF) stage and the production stage. In 2015, 71% of food system emissions originated from the LULUCF (32%) and production (39%) stages, while processing, transport, packaging, retail, consumption and waste represented the remaining 29% of emissions from food systems (see Figure 5.2) (Poore and Nemecek, 2018[195]). In the LULUCF stage, emissions are mainly CO_2, originating from the degradation of organic soils and deforestation. Most LULUCF emissions occur in developing countries, but a substantial share is associated with food and feed exports (Henders, Persson and Kastner, 2015[196]). Moreover, in developing countries, 73% of GHG emissions from food systems stemmed from the LULUCF sector, while most GHG emissions in industrialised countries stemmed from downstream energy-related sectors (Crippa et al., 2021[190]). In the production stage, emissions are mainly CH_4 and N_2O.[78] In the processing, transport, packaging, retail and consumption stages, emissions are mainly CO_2, and originate from energy and fuel use. In the waste stage, the main emissions are CH_4, relating to waste decomposition (Crippa et al., 2021[190]).

[78] CH_4 mainly originates from livestock, rice production, and treatment of production waste (Crippa et al., 2021[190]). N_2O emissions arise from fertiliser use and manure management (Tian et al., 2020[378]).

Figure 5.2. Global emissions from food systems in 2015 attributed to various food system stages



Source: Authors based on numbers from (Crippa et al., 2021[190]).

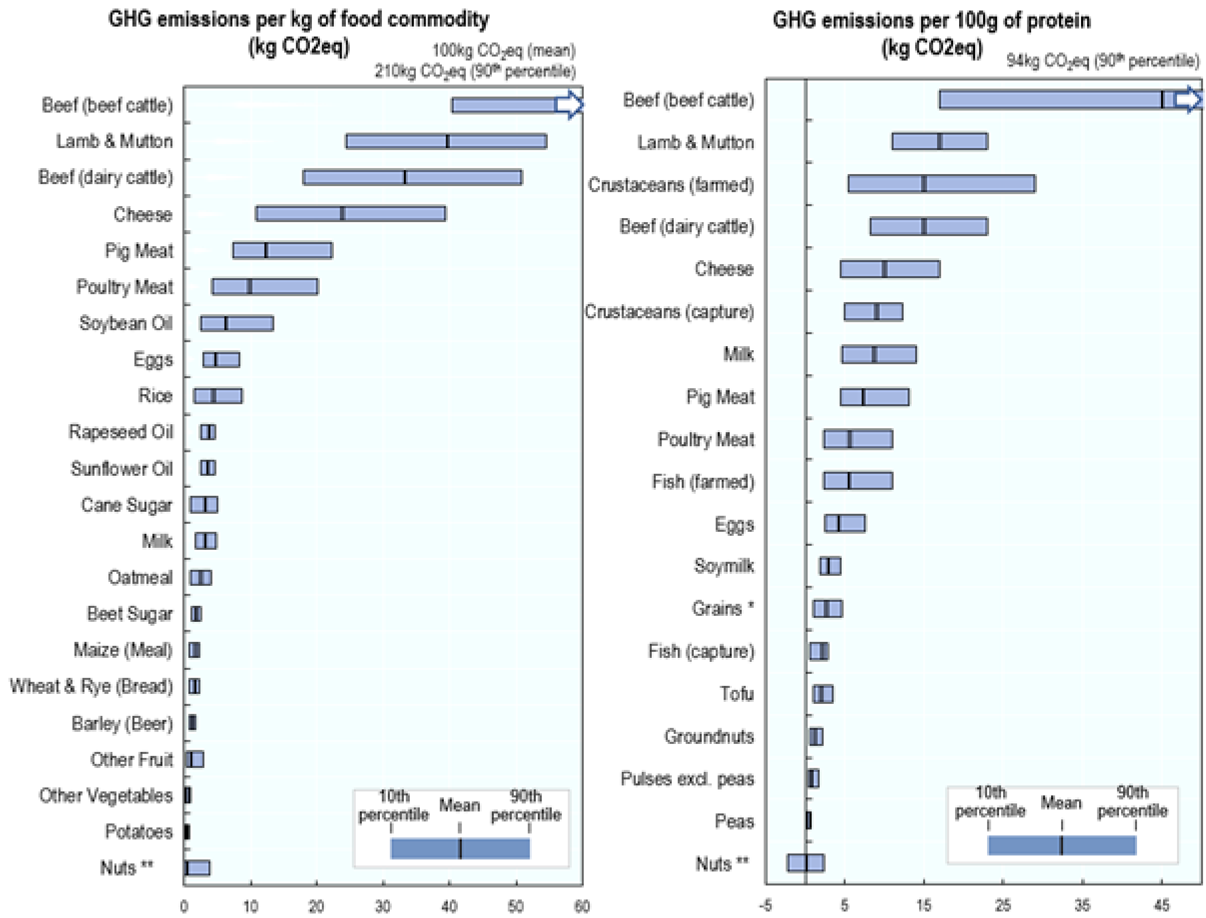
Different food products have widely different average GHG emission levels, with livestock (in particular beef and lamb) being the most GHG intensive, and accounts for a large share of total GHG emissions. The livestock supply-chain up until the point of retail is responsible for a minimum of 14.5% (Gerber, 2013[197]) to 16.5% (Twine, 2021[198]) of total anthropogenic GHG emissions.[79] Furthermore, nearly 65% (13.9 $GtCO_2eq$) of total food systems emissions, including emissions from retail, consumption and waste, can be attributed to four-value chains (beef, milk, rice, and maize) (Costa et al., 2022[199]). As set out in Figure 5.3 animal-based foods account, in most cases, for the largest share of GHG emissions per kg of food and per 100g of protein (OECD, 2022[187]).

Emissions intensity can also vary substantially within a given food product. Different production methods, including the use of different inputs and varying geographic locations, can affect the GHG emissions intensity as illustrated in Figure 5.3. For example, high (90th percentile) GHG emitting beef herd producers emit five times more emissions per kilo of beef produced (188 kg CO_2eq) than low (10th percentile) GHG emitting producers (34 kg CO_2eq) (OECD, 2022[187]). While there are also differences in the GHG emissions intensities in the production of staples, such as rice, wheat, and maize; average and even high GHG intensities are much lower for these than for animal-based food products (OECD, 2022[187]). However, opportunities do exist for the highest GHG intensity producers to reduce the emissions intensities of their products. For livestock, improving feed and pasture quality, strengthening farm, animal and manure management could help reduce the emissions intensity of livestock (see also section 5.1) (MacLeod et al., 2015[200]) (Blandford and Hassapoyannes, 2018[201]) (OECD, 2022[187]). For crops, the emissions intensity could be reduced by improving cultivation practices, using synthetic fertiliser and manure more efficiently, deploying integrated crop management as well as improved crop rotations. For producers with both livestock and crops, an integrated approach could also be pursued. For rice, water management could help reduce the emissions intensity (see also section 5.1) (MacLeod et al., 2015[200]) (Blandford and Hassapoyannes, 2018[201]) (OECD, 2022[187]).

[79] Of these livestock emissions (14.5% - 16.5%) up until the point of retail, more than half (62%) can be attributed to cattle, while pigs, poultry, buffalo and small ruminants contribute between 7% - 11% each (FAO, 2022[206]).

Figure 5.3. Variations in GHG emissions, between and within different food products

Mean, 10th and 90th percentile emissions intensities (per kg of food product and per 100g of protein)



Note: For Chart on GHG emissions per 100g of protein: Aggregation of CO_2, CH_4, and N_2O emissions in (Poore and Nemecek, 2018[195]) updated to use IPCC-AR6 100-year GWP. Data for capture fish, crustaceans, and cephalopods from (Parker et al., 2018[202]), with post-farm data from (Poore and Nemecek, 2018[195]), 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: (OECD, 2022[187]) building on (IPCC, 2022[136]), (Poore and Nemecek, 2018[195]).

Agricultural production is also an indirect driver of deforestation globally, in particular via expansion of land area for cattle pasture (Pendrill et al., 2022[203]). Expanding pastures for cattle meat production was assessed by (Pendrill et al., 2019[204]) to have led to more than 40% of deforestation globally between 2005-15, while (Goldman et al., 2020[205]) found this estimate to be 36.5%[80] between 2001-15. This is more than double the level of estimated global deforestation driven by cropland expansion for oilseeds (i.e.

[80] Calculated using the total deforestation area reported for 2001-15 (123 Mha) and the total deforestation area linked to cattle for the same period (45.1 Mha) (Goldman et al., 2020[205]).

soybean and palm oil), estimated at 15.2%[81] (Goldman et al., 2020[205]) to 18.4% (Pendrill et al., 2019[204]).[82] Compared to plant-based products, livestock production, especially ruminants, generally have more significant land use requirements per 100g of protein. Livestock production also exerts additional environmental pressures through increased terrestrial acidification, eutrophication, and freshwater use (Poore and Nemecek, 2018[195]). Yet, under some circumstances, livestock can also generate benefits (see section 5.4.1). As emissions from cattle represent the largest share of livestock emissions across all world regions (FAO, 2022[206]), there could be emissions reduction potential from reducing the global emissions from cattle, especially from beef herds.

Producers with above average GHG intensities are likely to have considerable mitigation potential. Although variations in the GHG emissions intensity of a given food product can be found across all food products, the largest numerical difference is between producers of animal-based products (i.e. livestock, dairy, and cheese). For ruminant production, producers with high GHG intensities are often those with extensive and less industrialised production – and are primarily located in developing countries, e.g. South Asia, Africa, Latin America and the Caribbean. These systems have large mitigation potentials through, for instance, improved feed, animal health and herd management. In countries where ruminant systems are more intensive and have lower GHG intensities (e.g. Europe and North America), mitigation potential at the production stage is relatively lower and related to improved feed and manure management and energy-saving measures (Gerber, 2013[197]).

Some GHG emissions are inherent by-products of the food production process and can only be reduced to a certain extent by improved practices. For example, in livestock, CH_4 emissions mainly arise from enteric fermentation (feed breakdown) in ruminant animals (e.g. cattle, buffalo, sheep and goats) and the decomposition of livestock manure. For manure, CH_4 (and N_2O) emissions could be reduced to some extent through the application of improved manure management practices.[83] For enteric fermentation, research indicates that feeding cattle seaweed could potentially lead to significant CH_4 emissions reductions (80%) (Roque et al., 2021[207]), yet no such solution has currently been applied at scale. Reducing the number of ruminant animals appears, for the moment, an important option for substantially reducing CH_4 emissions from enteric fermentation (Gerber, 2013[197]), yet such an approach could result in significant livelihood impacts, potentially rendering it politically difficult. After livestock, rice production is also a significant source of CH_4.[84] Reducing, or interrupting flooding could help reduce CH_4 emissions significantly (Wassmann, Papen and Rennenberg, 1993[208]) (Wassmann, Hosen and Sumfleth, 2009[209]) (Adhya et al., 2014[210]).

Food loss and waste accounts for a notable share of GHG emissions from food systems, and it would be important to address both the food lost on the supply-side, and the food wasted on the demand-side to reduce these emissions. In 2019, around 17% of the total food produced globally was wasted, equivalent to 931 million tonnes from households (61%), food services (26%) and retail (13%) (UNEP, 2021[211]). Approximately 8.6% of total GHG emissions from food systems are associated with food waste, while some, yet an undefined amount of emissions are also likely to originate from food loss (Crippa et al.,

[81] Calculated using the total deforestation area reported for 2001-15 (123 Mha) and the total deforestation area linked to oil palm (10.5 Mha) and soy (8.2 Mha) for the same period (Goldman et al., 2020[205]).

[82] However, the two studies did not clarify whether the soybean production linked to deforestation was intended for livestock feed or human consumption.

[83] Such practices could include e.g. solid manure or manure tank coverage, soil-liquid separation, as well as manure or slurry spreading (Nabuurs et al., 2022[189])

[84] CH_4 is released when rice fields are intentionally flooded; the longer the flooding, the higher the CH_4 emissions.

2021[190]).[85] Food losses often occur in developing countries due to the lack of functioning infrastructure and equipment on the agricultural production side. Of food produced globally in 2016, around 14% was lost from the farm and up to, yet excluding, the retail stage (FAO, 2019[212]).[86] Yet there is wide variation between the proportion of food loss indicated by different studies, however, these mainly cover developed countries. In California, on farm food loss was estimated to be 33.7% of food produced between 2016 and 2017 (Baker et al., 2019[213]), while in the Nordic countries (Finland, Sweden, Norway and Denmark) on farm food losses were between 1% - 3.7% of the total annual production (Hartikainen et al., 2018[214]). Food waste mainly occurs in developed countries at the retailer and consumer level.

5.2. Mitigation potential of food systems

Reducing GHG emissions from food systems will be an essential step towards net zero, however decarbonising all stages of food systems is far from straightforward. According to the IPCC, there is great mitigation potential for land-based and forestry actions (i.e. protection, improved management and restoration of forests,[87] peatlands,[88] and coastal wetlands),[89] followed by action in agriculture (e.g. soil carbon management,[90] agroforestry,[91] livestock management[92] and improved rice cultivation)[93] and some mitigation potential for demand-side actions (e.g. consuming less GHG intensive diets[94] and reducing food

[85] The FAO defines food loss and waste as “the decrease in the quantity or quality of food along the supply chain”. Food losses occur “along the supply chain from harvest/slaughter/catch up to, but not including the retail level”, while food waste “occurs at the retail and consumption level” (FAO, 2019, p. xii[212]).

[86] The food system emissions associated with these 14% are likely represented by (Crippa et al., 2021[190]) in the production stage as agricultural waste, yet it is unclear whether they are also accounted for in the processing and transport stages. Furthermore, these 14% excludes farm-level harvest losses (e.g. anything left in the field), for which there is limited data but which could be a similar order of magnitude. One report estimates that 15.3% of food produced globally in 2016 is lost at the farm level, and that 8.3% of this is related to harvest losses, (WWF-UK, 2021[420]).

[87] For more information on mitigation in forests, please see: (Busch et al., 2019[426]), (Griscom et al., 2020[424]), (Roe et al., 2021[425]), (Doelman et al., 2019[427]), (Bastin et al., 2019[428]), and (FAO, 2020[429]).

[88] For more information on mitigation using peatland, please see: (Page and Baird, 2016[430]), (Goldstein et al., 2020[431]), (Humpenöder et al., 2020[432]), (Leifeld and Menichetti, 2018[433]), (Günther et al., 2020[435]) and (Ojanen and Minkkinen, 2020[434]).

[89] For more information on mitigation through coastal wetland, please see: (Griscom et al., 2020[424]), (Bossio et al., 2020[436]), (Kauffman et al., 2020[437]), and (Goldstein et al., 2020[431]).

[90] For more information on mitigation using soil carbon management, please see: (Smith et al., 2022[438]), (Henderson et al., 2022[216]), (Smith et al., 2019[439]), (Bossio et al., 2020[436]), (Roe et al., 2021[425]) and (Lal et al., 2018[440]).

[91] For more information on mitigation using agroforestry, please see: (Chapman et al., 2020[441]), (Smith et al., 2019[439]), (Zomer et al., 2016[442]), and (Roe et al., 2019[443]).

[92] For more information on mitigation via livestock management, please see: (Blandford and Hassapoyannes, 2018[201]), (OECD, 2022[187]), (MacLeod et al., 2015[200]), and (Nabuurs et al., 2022[189]).

[93] For more information on mitigation via rice cultivation, please see: (Blandford and Hassapoyannes, 2018[201]), (OECD, 2022[187]), (Carrijo, Lundy and Linquist, 2017[444]), (Tran et al., 2017[445]), and (Tran et al., 2017[445]).

[94] For more information on mitigation via consuming less GHG intensive diets, please see: (FAO and WHO, 2019[446]), (Theurl et al., 2020[447]), (OECD, 2021[448]), (Ivanova et al., 2020[449]), (Springmann et al., 2016[274]), and (Poore and Nemecek, 2018[195])

loss and waste).[95] The majority of these mitigation options, especially land-based options, are readily available and could bring significant emissions reductions if deployed at greater scale globally (Nabuurs et al., 2022[189]). Yet, significantly mitigating other food system emissions, such as biogenic CH_4 and N_2O emissions from agricultural production, is more difficult given the lack of solutions and commercially available technologies for some aspects of production. For instance, reducing CH_4 emissions from enteric fermentation in ruminants remains difficult as no mitigation technology or method that can significantly reduce these emissions is currently commercially available at a large scale. Nevertheless, improving production practices with existing technology and methods, alongside land-use actions, could reduce the emissions intensity of production systems by 40-70% compared to average values (Costa et al., 2022[199]). Most of this mitigation potential is associated with reduced emissions from land-use change, improved animal feeding and breeding, improved management of manure and nutrients, improved water management in rice production and energy efficiency measures (Costa et al., 2022[199]).

Food systems can sequester CO_2 emissions through a variety of activities. In particular, reforestation/afforestation and agroforestry can increase carbon sequestration both above and below ground (OECD, 2022[215]). The potential of different parts of food systems to remove GHG emissions can be significant, particularly at the LULUCF and production stages. For example, recent analysis estimates that soil carbon sequestration on agricultural lands could offset 4% of annual global anthropogenic GHG emissions (Henderson et al., 2022[216]). However, such emission removals could potentially be reversed in the future (see section 5.4.1).

Despite the potential for emission reductions, few countries have established GHG mitigation targets for their food systems to date. In the latest NDCs,[96] several countries identify LULUCF (81%) and agriculture (77%) as priority areas for mitigation. For LULUCF, most countries mention afforestation, reforestation, and revegetation actions (54%). For agriculture, most countries mention cross-cutting actions (52%) and improved management of manure and herds (32%) (UNFCCC, 2021[217]). In countries' LT-LEDS,[97] agriculture is mentioned as a mitigation option by more than 40% of countries, while LULUCF is mentioned by less than 40% (UNFCCC, 2022[12]). Beyond this identification of priorities, only 16 out of 54 OECD and selected non-OECD countries[98] have established a mitigation target for their agricultural sector (OECD, 2022[215]). Uruguay is currently the only country to have highlighted a specific GHG mitigation target for food systems in its NDC (Eastern Republic of Uruguay, 2016[218]).[99] Nevertheless, New Zealand is in the process of creating a system to price agricultural emissions (see Box 5.1). Agricultural sub-sectors in some countries (e.g. Australia's red meat industry (CSIRO, 2022[219])), have also established their own GHG mitigation targets. As well as these initiatives, certain developed countries and emerging economies spend significant amounts on environmentally harmful agricultural support (see section 5.3), so there is therefore also some, yet limited, mitigation potential through policy reforms.

The link between food systems and climate change has been acknowledged in several recent initiatives. In the international climate negotiations, the Koronivia Joint Work Programme (KJWP) established in 2017

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[95] For more information on mitigation via food loss and waste, please see: (van Giesen and de Hooge, 2019[450]), (Galford et al., 2020[451]), (OECD, 2020[452]), (Smith et al., 2019[439]), (Ivanova et al., 2020[449]), and (Scherhaufer et al., 2018[453]).

[96] Information from the 166 latest available NDCs communicated by 193 Parties and recorded in the NDC registry as of 23 September 2022 (UNFCCC, 2021[217]).

[97] Information originates from the 53 latest available LT-LEDS communicated by 62 Parties to the UNFCCC secretariat as of 23 September 2022 (UNFCCC, 2022[12]).

[98] The 54 countries consist of 38 OECD countries, five non-OECD countries (EU Member States) and 11 emerging economies (OECD, 2022[215]).

[99] The target entails a 32% unconditional reduction (37% conditional) of CH_4 emissions per kg of beef and a 34% unconditional reduction (38% conditional) in N_2O levels per kg of beef by 2025 (Eastern Republic of Uruguay, 2016[218]).

was the only programme with a focus on agriculture and food systems (FAO, 2022[220]). At COP27, countries agreed on a four-year extension to the KJWP, to aid the implementation of climate action in food and agriculture (UNFCCC, 2022[221]). Other voluntary initiatives related to food systems have also come out of recent COPs (COP27 and COP26).[100] Beyond these, there are also other food systems-related initiatives that can help mitigate food systems emissions, for instance projects under REDD+.

Box 5.1. Plans to price agricultural emissions in New Zealand.

Engaging various actors to create an emissions pricing scheme

New Zealand's agricultural sector is responsible for more than half of total domestic GHG emissions (CO_2, CH_4, N_2O). Emissions from the agriculture sector will need to be reduced for New Zealand to meet its domestic and international climate mitigation targets. To incentivise agricultural producers to reduce their emissions, the Government launched a process to price agricultural emissions by January 2025, and for this process to be co-designed with various actors.

In 2022, the He Waka Eke Noa (Primary Sector Climate Action Partnership) presented a proposal for a pricing scheme based on different levies for short-lived GHG emissions (CH_4) and for long-lived GHG emissions (CO_2 and N_2O). The proposal included incentive payments for the uptake of approved emissions reduction actions and payments or credits for on-farm sequestration (He Waka Eke Noa, 2022[222]). The independent He Pou a Rangi (Climate Change Commission) also presented its advice as requested by the government in which it recommended producer assistance if high emissions prices are set, and the use of targeted assistance to specific groups (e.g. Māori) (He Pou a Rangi, 2022[223]).

Government's proposal on pricing agricultural emissions

Based on these inputs and a public consultation, in 2022 the Government proposed a farm-level split gas levy pricing system. This system proposes a split-gas levy at farm level for agricultural emissions of biogenic CH_4 and N_2O, with the exclusion of CO_2 emissions as the majority is covered by the domestic ETS. The price levels would be relatively low and follow price pathways for the first five years, yet with reviews after the first three years. The price of N_2O would, in addition, be capped for the first five years. Any generated revenue is recycled back into the system to cover operational costs, help further mitigate agricultural emissions (incentive and sequestration payments), and for a fund dedicated to Māori landowners (Ministry for the Environment and the Ministry for Primary Industries, 2022[224]).

In 2023, the Government is expected to decide on the type of emissions pricing system to be implemented to ensure an operational system is in place by January 2025. If an agreement on a pricing system cannot be reached or the system is not ready by January 2025, agricultural emissions are to be incorporated in the NZ ETS (Ministry for the Environment and Ministry for Primary Industries, 2022[225]).

[100] Relevant initiatives include the Global Methane Pledge, which encourages countries to reduce methane emissions including from the agriculture sector (Global Methane Pledge, 2022[380]); the Food and Agriculture Sustainable Transformation (FAST, which aims to increase climate finance at farm and country levels (FAO, 2022[386]); the Initiative on Climate Action and Nutrition (I-CAN) which targets an improvement in access to healthy food from sustainable food systems. (FAO, 2022[386]); and Action for Water Adaptation and Resilience, whose objective is to help improve water management for climate adaptation and resilience (FAO, 2022[386]).

5.3. Current application of emissions pricing and other climate-related policies in food systems

To help reduce food systems emissions, emissions pricing could be applied.[101] A range of different emissions pricing instruments and design options exists, each with advantages and disadvantages. Emissions pricing could be one potential way for countries to mitigate food systems' emissions, yet depending on the design, it could face several challenges, including public opposition (see section on Technical and methodological challenges related to applying emissions pricing in food systems). For that reason, some countries are also considering alternative mitigation approaches to tackle food systems emissions.

- Directly pricing emissions, thus applying the **polluter-pays principle**, can be implemented through an emissions tax or an ETS. An emissions tax, often implemented in the form of a carbon price equivalent, would set a certain price for GHG emissions, and thereby provide a stable price signal (and a stable revenue source). However, it would not provide certainty in terms of emissions reductions, as participants can chose to pay for an unlimited amount of emissions (World Bank, 2022[64]). An emissions tax could be applied on the supply- or the demand-side. An ETS, on the other hand, provides less economic certainty given fluctuations in the market-based price level[102] (Vaidyula and Alberola, 2015[102]). Yet, it does provide certainty regarding future emission levels as emissions within the scheme are capped (Black, Parry and Zhunussova, 2022[226]) (World Bank, 2022[64]). ETS have to date only been implemented on the supply-side, as the number of participating entities is limited and linked to an emissions cap. Both approaches generate government revenue, which could be used to compensate their impact or be invested in sector-specific mitigation technology and innovation (OECD, 2022[215]). Other emissions pricing design options could also be explored. For instance, emissions pricing could be applied to areas that are easy to measure and monitor, and the generated revenue could be used to support emissions reduction for hard-to-abate areas in other parts of food systems. However, it would be important to provide some benefits to stakeholders of taxed activities, to reduce any potential opposition. For all approaches, challenges could arise regarding the estimation, tracking and verification of emissions in food systems as there are a high number of diffuse emission sources and a high variability in food systems' emissions (see more in the section on Technical and methodological challenges related to applying emissions pricing in food systems).
- Countries could also apply the **beneficiary-pays principle**, which entails paying for emission reductions or avoidance in the form of abatement payments such as carbon/emissions reduction credits or offsets. In such a case, actors wishing to benefit from such credits or offsets would be eligible to purchase them. The beneficiary-pays approach does not add additional direct costs for producers or consumers, yet it is of a voluntary nature and thus tends to be less effective. Furthermore, if implemented on a large scale and if financed by governments, such beneficiary-pays schemes could entail large government costs (OECD, 2022[215]) (Denne, 2022[227]).

Using the social cost of carbon to calculate the price level of emissions pricing could furthermore help policy makers and stakeholders along food systems to understand the economic impacts of decisions that would increase or decrease emissions. Further producer and consumer impacts are investigated below (see section 5.4.1). Emissions pricing is currently only marginally applied in food systems in selected stages, using instruments under the beneficiary- and polluter-pays principle (see Table 5.1). However, the

[101] This paper explores the potential for systematic carbon pricing in food systems. It is also possible for carbon pricing in food systems to be implemented in a non-systematic manner, e.g. via offset programmes. A detailed analysis of these is outside the scope of this paper.

[102] The extent to which an ETS can provide revenue for governments is dependent on how the permits are allocated.

current emissions pricing application is patchy, not systematically applied, or only covers a small part of overall emissions from food systems (OECD, 2022[215]) as elaborated in Box 5.2 below.

Table 5.1. Selected overview of emissions pricing currently applied at different stages of food systems

Food system stage	Emissions pricing applied	Examples of existing schemes	Countries involved	Coverage of emissions	Details
LULUCF	To a small extent in certain countries*	REDD+	Multiple countries	Emissions coverage connected to specific activities in various schemes	REDD+ projects generate payments for avoided carbon emissions stemming from avoided deforestation, and degradation (Beneficiary-pays)
		CDM			CDM credits can be generated from re/afforestation (Beneficiary-pays)
		J-Credit Scheme	Japan		J-Credit Scheme certifies GHG emissions reduced or removed via forest management (Beneficiary-pays)
	To a medium extent in certain countries**	NZ ETS	New Zealand		NZ ETS is a voluntary scheme, which generates credits from re/afforestation and existing forests (Beneficiary-pays)
		ERF	Australia		Under the ERF, ACCUs can be earned for every tonne of CO_2eq reduced or stored in land via re/afforestation projects (Beneficiary-pays)
Production	To a small extent globally, with wide variation by country	Existing carbon pricing schemes and fuel excise taxes in some countries	Multiple countries	Some emissions are covered in certain carbon pricing schemes, with various exemptions	Existing carbon pricing schemes covering the energy sector are likely to cover energy-related CO_2 emissions, yet production emissions (CH_4 and N_2O) are not covered (Polluter-pays)
		CDM		Emissions coverage is connected to specific activities	CDM projects in this stage mainly covers CH_4 emissions reductions from certain production processes, while few projects cover N_2O emissions reductions (Beneficiary-pays)
		ERF	Australia	Emissions coverage connected to specific activities in various schemes	Under the ERF, ACCUs can be earned for every tonne of CO_2eq avoided in agricultural production (Beneficiary-pays)
		GHG Offset Credit System	Canada		Offset Credits can be earned for every tonne of CO_2eq reduced in the livestock sector or in agricultural soils (Offset protocols are in development) (Beneficiary-pays)
Processing, transport, packaging, retail	To some extent in certain countries	Existing carbon pricing schemes and fuel excise taxes	Multiple countries	Some emissions are covered, yet others are not	Some energy-related CO_2 emissions from the middle stages of food systems are covered by existing carbon pricing schemes and via fuel excise taxes, however exemptions exist (Polluter-pays)
Consumption	No				

Food waste	To some extent in some countries.	Volume-based food waste fee system	The Republic of Korea	Emissions associated with food waste are indirectly covered	A food waste tax in the Republic of Korea[103] requires households to pay for their food waste and thereby indirectly charges citizens for food waste emissions (Polluter-pays)
		General household waste tax	Multiple countries	Covers all waste, including food waste	General waste taxes exist in most EU countries[104], e.g. France, Finland, the Netherlands (Polluter-pays)

Note: *Emissions pricing applied to a small extent, refers to schemes where LULUCF projects are not widely applied within its context. For REDD+, less than 20 projects globally have been ongoing annually between 2018-2020 (UNFCCC, 2022[228]). For the CDM and the J-Credit scheme less than 10% of total cumulative carbon credits issued in compliance markets in 2019 were issued to forestry (Grafton et al., 2021[229]). Furthermore, 0.8% of CDM projects from 2013-22 were af/reforestation projects (UNEP, 2022[230]). ** Emissions pricing applied to a medium extent, refers to schemes where LULUCF projects represent a larger part. This is the case for the ERF and the NZ ETS where more than 50% of total cumulative carbon credits issued in compliance markets in 2019 have been issued to forestry (Grafton et al., 2021[229]).
REDD+ are projects under REDD (Reducing Emissions from Deforestation and Forest Degradations); CDM refers to Clean Development Mechanism; ERF stands for Emissions Reduction Fund, and activities under this fund can generate Australian Carbon Credit Units (ACCUs); Canada's GHG Offset Credit System has offset protocols for different activities. This table is not exhaustive.
Source: Authors.

Box 5.2. Examples of existing emissions pricing schemes at different stages of food systems

- **Land-use change stage:** Emissions pricing, or rather carbon pricing, is applied at the LULUCF stage in some countries through various international and national schemes that generate GHG credits for emission reductions (beneficiary-pays). These credits are obtained through different activities and practices such as sustainable forest management. For example, certain aspects of LULUCF activities are eligible under the Kyoto Protocol's Clean Development Mechanism (CDM) (re/afforestation and agriculture).[105] National level schemes include the J-Credit Scheme (forest management)[106] in Japan, the ETS in New Zealand (voluntary inclusion of forestry sector)[107] and the Emissions Reduction Fund (ERF) that generates Australian Carbon Credit Units (ACCUs) (re/afforestation projects).[108]
- **Production stage:** Emissions pricing is applied on CO_2 emissions associated with energy used in the production stage (e.g. fuel, heat and electricity). Yet most production emissions (CH_4 and N_2O) are only partially covered through specific projects aiming to offset or reduce the GHG intensity of certain agricultural production methods, which generates carbon credits that in some

[103] (Inno4sd, 2019[237]) and (Kim, 2022[238]).

[104] France (Government of France, 2022[405]), Finland (Finland's environmental administration, 2019[407]), and The Netherlands (Government of the Netherlands, 2022[406])

[105] Amongst many CDM projects from 2013-22, relatively few (0.8%) are af/reforestation projects (UNEP, 2022[230]).

[106] (Ministy of Economy, 2013[384]) (OECD, 2022[215]).

[107] Participating forest owners generate NZ units per tonne of CO_2 removed by expanding their forest, or through afforestation or reforestation activities. These NZ units can then be sold to other NZ ETS participants. However, the inclusion of forest is on a voluntary basis, thus forest owners can de-register from the NZ ETS (Ministry for Primary Industries, 2022[383]) (Carver et al., 2022[382]).

[108] In Australia, landowners can generate ACCUs by reducing or storing emissions in land via re- or afforestation projects and afterwards sell these to governments or third parties (Australian Government, 2022[385]) (OECD, 2022[215]).

instances can enter domestic carbon pricing schemes. Existing projects in the production stage include the Kyoto Protocol's CDM (e.g. manure management, rice production, fertiliser application)[109], Australia's ERF (agricultural production)[110], California's Compliance Offset Program (Livestock and rice production)[111] and Canada's GHG Offset Credit System (livestock and agricultural soils)[112]. Some energy-related CO_2 emissions from the production stage are covered by existing carbon pricing schemes or fuel excise taxes (polluter-pays). Yet the amount and relative importance of energy-related CO_2 emissions differs depending on the food product. For instance, CH_4 emissions are a more important component of total livestock emissions (44%) than energy-related CO_2 (27%) (Gerber et al., 2013[231]) (Gren et al., 2019[232]). In contrast, energy-related CO_2 emissions from vegetables grown in greenhouses accounts for a much higher share of their total GHG emissions (Gren et al., 2019[232]). For food security and affordability concerns, many governments apply exemptions or reduced rates for agricultural producers, e.g. under the EU's Energy Taxation Directive (EC, 2018[233]) (ECA, 2022[234]).

- **Processing, transport, packaging, and retail stages**: The majority of GHG emissions in this post-production stage of food systems relate to energy-related CO_2 emissions (including fuel). Such emissions are covered to some extent (e.g. depending on the total emissions of the entity) in certain countries under existing carbon pricing schemes or fuel excise taxes (polluter-pays approach) (Crippa et al., 2021[190]). Yet, small emitters are sometimes exempt from emissions pricing, such as in the EU ETS (Foucherot and Bellassen, 2013[235]) (Turner et al., 2021[236]).
- **Consumption stage**: There are currently no emissions pricing schemes applied on food products at the consumer level. However, some food-related taxes or pricing schemes have been applied to reduce various negative health impacts (see Box 5.3).
- **Waste stage**: Schemes pricing food waste emissions are few and these are mostly implicit emissions pricing, using a polluter-pays approach. In the Republic of Korea, a volume-based food waste fee system exists, which charges households for the food waste generated, indirectly pricing food waste emissions (Inno4sd, 2019[237]) (Kim, 2022[238]). In some countries, general household waste taxes include coverage of food waste, e.g. in several EU countries.

Source: Authors.

Many countries currently support climate action in food systems through various supply-side policies other than emissions pricing. In an assessment of agricultural mitigation policies in 20 countries, policies such as grants or income support schemes were more widely used than carbon pricing to reduce emissions (Henderson, Frezal and Flynn, 2020[239]). Countries often use a mix of policies to address climate impacts on agriculture. For example, dedicated agricultural support (e.g. EU's Common Agricultural Policy (CAP)), grants (e.g. Nature Climate Solution Fund in Canada) or preferential credit (e.g in the US[113]) (OECD, 2022[215]) are used to for instance promote reforestation/afforestation, revegetation and sustainable land

[109] Many CDM offset projects related to reducing CH_4 emissions from manure management exist, while very few CDM offset projects related to reducing CH_4 emissions from rice production and N_2O emissions from fertilisers application exist. Credits from CDM projects can in some instances enter domestic carbon pricing systems (UNEP, 2022[381]).

[110] The Emissions Reduction Fund generates Australian Carbon Credit Units (ACCUs) (Clean Energy Regulator, 2021[422]).

[111] California's Compliance Offset Program (California Air Resources Board, 2023[423]).

[112] Canada's GHG Offset Credit System (Government of Canada, 2023[421]).

[113] In the US both preferential credit and grants have been introduced as economic incentives to promote the adoption of GHG mitigation practices (OECD, 2022[215]).

management. Several countries also have environmental regulations (e.g. nitrogen regulations) (OECD, 2022[215]) to for example optimise nitrogen fertiliser use and reduce associated N_2O emissions (UNFCCC, 2022[240]). R&D and knowledge transfer are also supported by some countries (OECD, 2022[215]), for example on livestock and manure management, soil management and improved agricultural practices (UNFCCC, 2022[240]).

Some forms of agricultural support can encourage harmful practices and thereby result in negative environmental and nutritional outcomes. When governments provide agricultural support, this is for various reasons and can include to support farmers' livelihoods and improve food security. However, some forms of agricultural support can lead to environmentally harmful practices (e.g. overuse of agrochemicals and natural resources) (Henderson and Lankoski, 2019[241]) and reduce incentives for environmentally beneficial innovation (DeBoe, 2020[242]). Certain types of support can also lead to negative nutritional outcomes, such as encouraging the production of staple foods over the production of more diverse and/or nutritious foods, such as fruits and vegetables (FAO, UNDP and UNEP, 2021[243]).

While different support measures have different environmental impacts, at least one third of direct agricultural support currently provided encourages the production of high-emitting food products (OECD, 2022[215]).[114] In 2019-21, agricultural support across 54 countries[115] amounted to USD 817 billion per year. Most of this support was directed to individual producers (USD 611 billion/year) with specific amounts tied to certain commodities via market price support[116] or product-specific payments[117] (USD 361 billion/year)[118] Significant payments are also transferred for the provision of various forms of inputs[119] (USD 60 billion/year) (OECD, 2022[215]). Yet, the environmental impact of these and other measures differs. Market Price Support, output-based payments (most production-coupled support) and unconstrained input-based payments have been identified as potentially the most environmentally harmful types of support (Henderson and Lankoski, 2019[241]). Payments based on the cultivation area or the number of animals are potentially less harmful, yet, depending on their application, they can either encourage or discourage GHG emissions (OECD, 2022[215]). Other types of support, such as fully decoupled payments based on non-current crop area, are amongst the least potentially harmful support measures (Henderson and Lankoski, 2019[241]). Moreover, an analysis by (FAO, UNDP and UNEP, 2021[243]) of agricultural support across 88 countries found that emissions-intensive commodities (e.g. beef, milk, and rice) were among the ones benefitting from the highest rate of support between 2013-18.

Reorienting current agricultural support that is potentially most environmentally harmful could help mitigate climate change and encourage more environmentally friendly farming practices. Studies have estimated that eliminating both domestic agricultural support and trade barriers could lead to global GHG emission

[114] From 2019-21, large amounts of support, mainly market price support, was provided to high-emitting commodities including beef production (USD 25 billion/year, equivalent to USD 22/tCO_2eq), sheep and goat meat production (USD 7 billion/year, equivalent to USD 31/tCO_2eq) and rice production (USD 44 billion/year, equivalent to USD 115/tCO_2eq) (OECD, 2022[215]).

[115] The 54 countries consist of 38 OECD countries, five non-OECD countries (EU Member States) and 11 emerging economies (OECD, 2022[215]).

[116] Market price support increases the price received by producers for a specific commodity, thus it encourages additional production (OECD, 2022[215]).

[117] Product-specific payment is support linked to a specific element, such as output, cultivation area or the number of animals, and these typically also incentivise increased production (OECD, 2022[215]).

[118] Support can both be in the form of payments made directly to the farmer or in the form of market price support (OECD, 2022[215]).

[119] Input-based payments typically supports the unconstrained use of fertiliser, fossil fuels or irrigation (OECD, 2022[215])

reductions of 0.55% by 2040 (Gautam et al., 2022[244]) or 0.5% by 2050 (Guerrero et al., 2022[245]). Nevertheless, removing both agricultural support and trade barriers could have certain trade-offs, such as reductions in the global production of crops and livestock.[120] (Fell et al., 2022[246]) projects that with land expansion constraints, a removal of both domestic support and trade barriers, could decrease direct agricultural GHG emissions by 1.6% while increasing food production by 0.2%. Furthermore, (Fell et al., 2022[246]) estimates that a multilateral approach to reforming agricultural support could generate positive emissions and food security outcomes, compared to a partial reform (such as only removing domestic agricultural support or trade barriers) which could result in adverse outcomes. Although removing most potentially environmentally harmful support has the potential to decrease GHG emissions, emissions reductions would remain limited compared to the mitigation needs, as outlined by the studies above.

To support climate change mitigation policies, government support could be reoriented to the provision of public goods and key general services to improve the performance of the agricultural sector, and a package of approaches could be developed to ensure significant emissions reductions in agriculture. Some governments use agricultural support to encourage certain practices (e.g. agri-environmental payments provide economic incentives for the use of environmentally friendly inputs or practices) or invest in R&D and innovation within the agriculture sector. Other policies, e.g. land retirement policies (encourage retirement of cropland for permanent pasture or forests) are also used by some governments (DeBoe, 2020[242]). Such positive support schemes however remain limited. For example, of the USD 611 billion of support provided to individual producers per year between 2019-21, only USD 1.7 billion was provided for the production of agri-environmental public goods[121] (i.e. payments based on specific non-commodity outputs), while USD 26 billion was provided for R&D and innovation. Reorienting most market-distortive agricultural support towards more decoupled payments should also support reductions of GHG emissions, in particular when accompanied with environmental cross-compliance. However, decoupling is unlikely to be sufficient on its own, and the climate impact also depends on the type and effectiveness of mandatory management practices and environmental requirements that accompany payments (OECD, 2022[215]).

Demand-side policies have to date not frequently been used to mitigate emissions from food systems, and those that are used by countries are primarily information-based (e.g. voluntary food labels, flyers, and advertisements). The effectiveness of demand-side policies is currently unclear (Deconinck et al., 2021[247]). Examples of demand-side policies include the French Eco-Score, which uses scores and colours based on lifecycle assessments to indicate how environmentally friendly a given product is (Government of France, 2021[248]). Similar initiatives are being developed by other actors such as Foundation Earth – comprising of various international food corporations, supermarket chains, the European Institute of Innovation & Technology (EIT), food and environmental experts (Foundation Earth, 2021[249]). A handful of countries (e.g. Canada, Switzerland, Sweden, Qatar, Norway, Brazil and Germany) have included environmental considerations in their national dietary guidelines (Mouthful, 2020[250]). Some countries are also taking action to reduce consumer food waste. For example, the Scottish government adopted a target to reduce food waste per capita by 33% by 2025, compared with 2013 levels (Scottish Government, 2016[251]) by for instance reducing unnecessary food demand, reducing food loss and waste, and increasing food recycling (Zero Waste Scotland, 2019[252]).[122] Informational policy instruments are often complementary to economic or regulatory policy instruments (Dubash et al., 2022[253]). A combination of different types of instruments is likely needed to initiate demand-side changes to help reduce emissions

[120] (Gautam et al., 2022[244]) estimates this reduction to be 1.23% and 0.35% between 2020-40 for global crop and livestock productions. (FAO, UNDP and UNEP, 2021[243]) projects such reductions in global production to be 1.3% for crops and 0.2% for livestock, as well as a potential reduction in global farm employment by 1.27% between 2020-30.

[121] Agricultural support conditional on the provision of environmental public goods and services, refers to payments for outputs that provide public environmental benefits such as payment for carbon sequestration, afforestation, or for the restoration of marginal lands (OECD, 2022[215]).

[122] However, the Scottish food waste reduction target is currently off-track (The Scottish Government, 2022[412]).

from food systems, as also highlighted in the previous section on The need for both demand- and supply-side changes to reach net zero. To ensure that appropriate demand-side policies are deployed, additional research investigating the effectiveness of various demand-side policies would be beneficial.

5.4. Potential application of GHG emissions pricing in food systems

There is significant direct emission reduction potential from pricing GHG emissions on both the supply- and demand-side of food systems (see Figure 5.4), however, there are also significant challenges to its application (see Technical and methodological challenges related to applying emissions pricing in food systems). On the supply-side, both the polluter-pays and the beneficiary-pays principle could be deployed. For the former, pricing emissions on the supply-side of food systems could help improve the GHG intensity of production, yet it could also have several negative economic impacts (Isbasoiu, Jayet and De Cara, 2020[254]). The latter would entail paying for emission reductions or avoidance in the form of abatement payments such as carbon/emission reduction credits or offsets. On the demand-side, emissions pricing in the form of the polluter-pays principle, could encourage consumption changes, and thus have considerable emissions reduction potential. For instance, shifting 10% of ruminant product consumption (i.e. red meat and milk) to poultry and pork products in 2030 compared with 2017 levels could lower emissions by 0.9 GtCO_2eq per year by 2030 (OECD, 2019[255]). Emissions pricing could also encourage reduced demand for high GHG-intensity foods and thereby incentivise a decrease in the emissions intensity of high emitting products. It could also encourage an increase in low GHG-intensity foods (Ranganathan et al., 2016[256]). For instance, a shift towards plant-based diets could reduce the land area needed to feed the global population by 67% (Poore and Nemecek, 2018[195]). However, emissions pricing applied via the polluter-pays principle could also impose additional costs on producers and consumers, which could make the approach politically infeasible in some countries. To improve the political feasibility, and ensure a just transition, any adverse effects would need to be addressed. Utilising emissions pricing via a beneficiary-pays approach is likely to have fewer adverse effects on farmers given that they would be paid by beneficiaries to abate emissions, yet costs are shifted to stakeholders (and governments) paying for abatements (OECD, 2019[257]).

Figure 5.4. Potential emissions pricing opportunities across food systems stages

	LULUCF	Production	Transport, processing, packaging, retail	Consumption	Waste
Major GHG emissions	CO_2	CO_2, CH_4, N_2O	CO_2	CO_2	CH_4
Potential targeted emissions sources	• Forest cover **	• All agricultural emissions • Specific agricultural emissions (e.g. animal numbers)	• All production emissions	• All food products • Specific foods (e.g. high emitting foods)	• Food waste
Potential impacts (+/-)	+ Biodiversity, Ecosystems, water availability, soil erosion + Livelihoods + Land owner income - Reversibility risks	+ High GHG mitigation -/+ Farmer impact* - Food insecurity risks - Political resistance - Methodological challenges - Technical implementation issue	+ Small GHG emissions reductions - Technical implementation issue	+ Medium GHG mitigation - Food security risks	+ Low GHG mitigation - Technical implementation issue

Note: * The farmer impact could be positive or negative depending on the type of farm and the type of pricing (polluter-/beneficiary-pays). ** Forest cover entail multiple emissions sources such as re-/afforestation, avoided deforestation and forest degradation and increased forest regeneration.
Source: Authors.

Applying a polluter-pays GHG emission pricing scheme (e.g. carbon tax or ETS) at the production stage can be approached from different angles (see selected examples in Table 5.2). All modelling studies reviewed for this paper utilise a carbon tax, as an ETS can be more difficult to model given fluctuating price levels. These selected studies each display different emissions pricing approaches and the projected effects of each approach at the production stage in food systems. An EU study indicated that GHG emissions reductions from food systems could be reduced by 10-16%, 16-25%, and 25-39% at carbon prices applied at the production stage of EUR 50/tCO_2eq, EUR 100/tCO_2eq, and EUR 200/tCO_2eq, respectively (Isbasoiu, Jayet and De Cara, 2020[254]). Another study found a GHG emissions tax to be the most effective market-based instrument for reducing emissions from livestock and crop production in various farms in the EU. However, the study concluded that significantly reducing livestock GHG emissions is challenging without reducing the herd size (OECD, 2019[258]). Yet, reducing the herd size could have significant just transition impacts for livestock farmers and might not be politically feasible. A third scenario (OECD, 2019[259]) applied a global GHG tax of EUR 40/tCO_2eq (USD 40/tCO_2eq) in 2021-2030, EUR 60/tCO_2eq in 2031-2040, and EUR 100/tCO_2eq in 2041-2050. The study found that by 2050, a global GHG tax was most effective in reducing GHG emissions (28% non-CO_2, 35% land use change (LUC) CO_2). However, a global GHG tax also had the largest negative effect on food consumption levels and farm income. Applying the same GHG tax alongside a food subsidy to help maintain food consumption levels resulted in similar emission reductions (27% non-CO_2, 32% LUC CO_2) as the global GHG tax alone (OECD, 2019[259]).

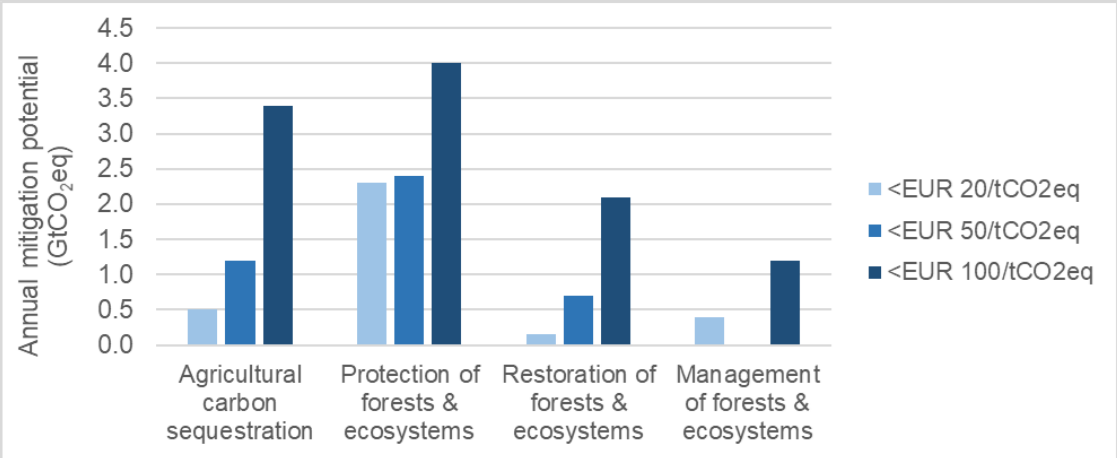
Table 5.2. Overview of selected studies modelling emissions pricing at the production stage in food systems

Study	Region	Emissions pricing policy coverage	Time period	Cost per unit of GHG emissions (EUR/tCO_2eq)	GHG emissions reduction (%)
(Isbasoiu, Jayet and De Cara, 2020[254]).	EU	Tax on agricultural production emissions (Polluter-pays)	2007-2012	50	10-16 (annually compared to BAU$^{\Delta}$)
				100	16-25 (annually compared to BAU$^{\Delta}$)
				200	25-39 (annually compared to BAU$^{\Delta}$)
(OECD, 2019[258])	EU	Tax on agricultural production emissions (Polluter-pays)	Unspecified time period	9	19.3* (total)
				30	45.8* (total)
				50	51.4* (total)
		Tax on number of ruminants (dairy herd) (Polluter-pays)		9	18.4* (total)
				30	45.6* (total)
				50	48.0* (total)
		Tax on nitrogen fertilizer (kg of application per hectare) (Polluter-pays)		9	1.8* (total)
				30	5.7* (total)
				50	9.8* (total)
(OECD, 2019[259])	Global	Tax on agricultural production emissions (Polluter-pays)	2020-2050	100 (Price increase every 10 years**)	28 (non-CO_2) 35 (LUC) (annually compared to BAU$^{\Delta}$)
		Tax on agricultural production emissions and food consumption subsidies (Polluter-pays)			27 (non-CO_2) 32 (LUC) (annually compared to BAU$^{\Delta}$)
		Abatement payment for agricultural production emissions (Beneficiary-pays)			14 (non-CO_2) 9 (LUC CO_2) (annually compared to BAU$^{\Delta}$)
	OECD	Tax on agricultural production emissions (Polluter-pays)			2 (non-CO_2) 2 (LUC CO_2)*** (annually compared to BAU$^{\Delta}$)
		Abatement payment for agricultural production emissions (Beneficiary-pays)			2 (non-CO_2) 2 (LUC CO_2)*** (annually compared to BAU$^{\Delta}$)

Note: $^{\Delta}$BAU refers to Business-as-Usual. *Average percentage emissions reductions across four different types of farms. ** Emissions pricing increases from EUR 40/tCO_2eq (USD 40/tCO_2eq) to EUR 60/tCO_2eq to 100/tCO_2eq in the periods from 2021-30, 2031-40, 2041-50, respectively. *** Global emissions reductions.
Few other studies modelling emissions pricing in food systems can also be found, such as (Henderson et al., 2021[260]), (Dumortier and Elobeid, 2021[261]), (Henderson et al., 2017[262]), (Dumortier et al., 2012[263]) and (Deybe and Fallot, 2003[264]). These have however not been included in the table as they either provide results in a different form or for different categories than those listed in the table or they are older publications.
Source: authors based on specified studies.

There is also mitigation potential in taking a beneficiary-pays approach, for instance via on-farm carbon sequestration or by protecting forests. On farms, carbon sequestration can occur in various ways, e.g. through agroforestry (Ghale et al., 2022[265]) or soils (Henderson et al., 2022[216]). (OECD, 2019[259]) found that providing abatement payments to farmers globally of EUR 40/tCO_2eq in 2021-2030, increasing to EUR 100/tCO_2eq in 2041-2050 could lead to GHG emission reductions. Such an approach would not impose additional cost burdens on farmers compared to a tax or ETS, yet the abatement payments would need to be financed. Nevertheless, the emissions reductions generated from abatement payments would be lower than those generated by a global emissions tax (see Table 5.2) (OECD, 2019[259]). (Nabuurs et al., 2022[189]) estimated mitigation potential at various emissions pricing levels. The highest annual mitigation potential was projected for the protection of forests and ecosystems and for carbon sequestration in agricultural production systems (see Figure 5.5). Yet, there can be important feedback loops and uncertainties with carbon sequestration, as the capacity to sequester carbon is also likely to be affected by climate change (Nabuurs et al., 2022[189]).

Figure 5.5. Annual mitigation potential of various emissions pricing levels applied from 2020-2050



Note: Price levels have been converted to EUR (USD 1 = EUR 1). No data is available for the annual mitigation potential of management of forests & ecosystems under price level of <EUR 50/tCO_2eq.
Source: Authors based on numbers from (Nabuurs et al., 2022[189]).

Polluter-pays GHG emissions pricing on food consumption could also generate significant emission reductions by e.g. leading consumers to make dietary changes and reduce their meat and dairy consumption, however, it could face implementation challenges.[123] (OECD, 2019[259]) also looked at the effect of a consumer GHG tax on ruminant meat and dairy products in OECD countries. It found such a GHG tax to be less effective than direct taxation of agricultural emissions on the supply-side, for instance, via a global GHG tax (OECD, 2019[259]). Another study simulating consumption taxes on animal products in the EU similarly found small but positive mitigation effects on agricultural emissions. It was estimated that by applying a tax of EUR 60/tCO_2eq and of EUR 290/tCO_2eq within the EU, the EU demand for animal products would decrease, potentially leading to a 1.5% and 4.9% decrease in EU agricultural emissions. Furthermore, the study estimated that if the EU demand for meat and dairy would decrease, so would the EU import and production of meat and animal feed and the associated emissions. This reduction in emissions within and beyond the EU could lead to decreases in global agricultural emissions of 0.75% and 0.2% for the respective taxes (Jansson and Säll, 2018[266]). Another study applying an emissions tax of EUR 60/tCO_2eq on animal foods in the EU found a higher reduction potential of 7% (32 million tonnes of GHG emissions) of EU agricultural emissions (Wirsenius, Hedenus and Mohlin, 2010[267]). A study in

[123] For example, aligning national diets in 54 high-income countries (Antigua and Barbuda, Australia Austria, Bahamas, Barbados, Brunei Darussalam, Canada, Chile, Cyprus, Denmark, Estonia, Finland, France, French Polynesia, Germany, Greece, China (Hong Kong, Macao, Taiwan), Hungary, Croatia, Iceland, Ireland, Israel, Italy, Japan, Republic of Korea, Kuwait, Latvia, Lithuania, Malta, Netherlands, New Caledonia, New Zealand, Norway, Pakistan, Panama, Czech Republic, Poland, Portugal, Saint Kitts and Nevis, Saudi Arabia, Slovenia, Spain, Sweden, Switzerland, Trinidad and Tobago, Oman, United Arab Emirates, UK, USA, Uruguay, Belgium, and Luxembourg) with the EAT-Lancet recommendations to reduce animal products intake, increase intakes of whole grains, fruits, vegetables, nuts and legumes (Willett et al., 2019[418]) is estimated to have the potential to reduce annual global agricultural emissions by 61.5% compared to 2010 levels. (Sun et al., 2022[391])

Sweden found a unit tax of EUR 0.16–2.9/kg[124] (SEK 1.8 – 32.5/kg) on domestic meat and dairy consumption could decrease emissions of GHG, nitrogen, ammonia, and phosphorus by 1.5% of total Swedish emissions and 12% of emissions just from the livestock sector (Säll and Gren, 2015[268]). Despite this mitigation potential, GHG emissions pricing applied to food consumption could have significant negative impacts on food security and nutrition – particularly for the most vulnerable populations - and it could therefore face significant political resistance, potentially hindering its feasibility.

Furthermore, leakage risks could be significant,[125] and limit the effectiveness of applying GHG emissions pricing in individual countries or food systems. Leakage would not only reduce agricultural production and have significant food security implications in affected countries, but it would also have a limited impact on reducing emissions from food systems globally as production in other potentially higher emitting systems are likely to increase resulting in any emissions reductions being fully or partly offset (Jansson et al., 2023[269]) (OECD, 2019[259]). Furthermore, such changes in production could also contribute to changes in trade flows (see more in section 5.4.1)

The impacts of implementing emissions pricing in food systems unilaterally could be reduced by putting in place a border tax adjustment. Introducing such a mechanism in the scenario above with a tax of EUR 120/tCO_2eq on EU agricultural emissions reduces carbon leakage from 76% to 36% (Jansson et al., 2023[269]). However, such a border tax may present administrative and legal challenges, as well as trade-offs between effectiveness, competitiveness and leakage (IMF&OECD, 2021[270]) (Arvanitopoulos, Garsous and Agnolucci, 2021[271]). Putting in place a global emissions price would have the benefit of not creating emissions leakage. However, the political feasibility of such a tax could be low. Emissions leakage is mainly related to the supply-side, and not the demand-side. This is likely because consumption taxes would be applied to food products from all sources, thus minimising leakage, and competitiveness concerns (Funke et al., 2022[272]). Instead, the demand-side is faced with the rebound effect. Typically, the taxation of products leads to reduced consumption, yet, money saved from the reduced consumption may be re-spent on other products. Depending on the GHG emissions of these other products, GHG emissions reductions may be counteracted partially or fully (Grabs, 2015[273]).

There are limited experiences with environmental polluter-pays consumption taxes, yet some similarities may exist with health-related food taxes (see Box 5.3). A study on the opportunity of GHG taxes to promote healthier diets found a global tax of USD 52/tCO_2eq on food emissions across food systems could generate emission reductions of up to 9% compared to 2020 emission levels (Springmann et al., 2016[274]). The study also estimated that this could generate health benefits from the reduction of red meat consumption induced by the tax. (Faccioli et al., 2022[275]) found that a combined carbon and health tax in the UK could decrease the consumption of snacks, sugary drinks, and alcohol while simultaneously boosting fruit and vegetable consumption. Yet, at the same time, household consumption patterns indicate that environmental considerations, such as the carbon footprint of food items, are less important in food purchases than other priorities such as affordability, freshness, taste, and nutritional value. Thus, any policies developed aiming at shifting diets towards low-emitting foods, could consider emphasising the cost co-benefits (as also effective in health-related food taxes, see Box 5.3) and the taste of products rather than environmental considerations (Lamhauge, Katherine Farrow and Lea Stapper, Forthcoming[276]).

Experiences with the implementation of the limited health-related taxes on food (see Box 5.3) could provide some insights for emission pricing policies, especially when it comes to potential implementation barriers

[124] Exchange rate SEK 1 = EUR 0.09 (assessed on 21 December 2022) (Google finance, 2022[353])

[125] A GHG production tax of USD 100/tCO_2eq applied to OECD countries by 2050 is estimated to lead to carbon leakage (34% of non-CO_2 emissions, 40% of LUC CO_2 emissions) (OECD, 2019[259]). Frank et al. (2021[387]) found leakage levels of 40% from unilateral pricing on agricultural emissions within the EU of USD 245/tCO_2eq (EUR 245/tCO_2eq), while (Jansson et al., 2023[269]) found leakage levels of 76% from emissions pricing of EUR 120/tCO_2eq within the EU.

and how they could be overcome. Many of the common barriers to implementing such policies, e.g. industry pushback, lobbying and low public acceptability, could also occur in the potential implementation of emissions pricing in food systems. Nevertheless, certain lessons on overcoming barriers could potentially be learned. These include that broad policy support from government and other actors is important for policy legislation and implementation. Such support can be built by focusing on improved health outcomes along with economic gains. Earmarking generated revenue for priorities that benefit the public or specific groups - and to publicise what the earmarked revenue is used for - can also be used to maintain or increase public support. Strong public support could also help address industry pushback but may need to be supplemented with policy modifications, exemptions or by providing other industrial advantages (e.g. no tax rate increases for a specific number of years). Furthermore, impact evaluations could also be helpful in confirming or rejecting the impact of industry claims.

Box 5.3. Country experiences with health-related food consumption taxes

Most taxes applied at the food consumption stage to date have been introduced because of health impacts. For example, in 2018, 12 OECD countries applied some form of taxes on foods high in sugar or saturated fats (Giner and Brooks, 2019[277]). Alcohol is also taxed or regulated in most countries for health reasons (WHO, 2022[278]). Some country experiences are set out below.

- A tax on sugar-sweetened beverages (SSB) in **Mexico** reduced the number of high-consumers from 50% to 43% and increased the number of non-consumers to 5% after three years (Sánchez-Romero et al., 2020[279]). The largest decrease was observed for low-income households (Ng et al., 2018[280]) (Colchero, Molina and Guerrero-López, 2017[281]). However, there were major barriers to the tax implementation including pushback and lobbying from transnational corporations and a lack of transparency regarding the final tax level. Several elements helped overcome some of these barriers. These included broad support for the tax across the government, civil society organisations and other societal actors. Furthermore, framing the tax as a health-related instrument to tackle high obesity rates[126] and as a revenue generating source to help further accelerate health, further increased public acceptance. Moreover, the Mexican president agreed with an industry consortium to not further increase the tax. However, some barriers remain unresolved, such as the justification for the implemented tax level that was lower than the recommended level. Moreover, the use of SSB-generated revenue remains unclear, and government-conducted policy evaluations are missing (Carriedo et al., 2021[282]).
- **South Africa** introduced a Health Promotion Levy in 2018 on all SSB. The level of the levy was equivalent to 10% of the litre price of the most popular soft drink. This was reduced from the originally-proposed tax rate of 20%, due to pushback from the sugar and beverage industry, arguing for the negative impact of potential job losses in an economy with high unemployment (Abdool Karim, Kruger and Hofman, 2020[283]). Despite a low tax rate, the sale of SSB decreased, with greater decreases among lower socio-economic groups (Hofman et al., 2021[284]).

[126] In 2012, around the time that the SSB tax was promoted, 72.2% of the adult population in Mexico were either obese or overweight (Carriedo et al., 2021[282]).

- In **Denmark**, a tax on saturated fats led to a reduction in the intake by 4% over a 13-month period after which it was repealed (Smed et al., 2016[285]). Reasons for the repeal was related to lobbying, lack of support across many actors, and a switch to low-priced brands (Vallgårda, Holm and Jensen, 2014[286]) (Jensen and Smed, 2012[287]). If the tax had been kept, it is estimated that it would have saved 120 lives annually via positive health impacts, e.g. reduction in cardiovascular disease (Smed et al., 2016[285]).

Note: The 12 OECD countries are Belgium, Chile, Finland, Estonia, France, Hungary, Ireland, Latvia, Mexico, Norway, Portugal, and the UK (Giner and Brooks, 2019[277]).

5.4.1. Technical and methodological challenges related to applying emissions pricing in food systems

There are significant methodological challenges to establishing GHG emissions pricing (ETS, emission tax or credits) in food systems. Methodological challenges include the high variability in emissions intensity between and within different food products (see Figure 5.3). These variations can be significant, even within a specific country. To provide an accurate price signal, GHG emissions pricing in food systems, whether on the supply- or demand-side, would need to reflect such GHG intensity variations in an emissions price. Such a task would be further complicated if different stages of food production for a given product occur in different countries.

These large variations in GHG intensity occur both for livestock, as well as for other agricultural outputs, and are due to many natural and anthropogenic factors. Natural factors affecting GHG intensity of agricultural outputs include soil moisture, temperature, pH, and type. Anthropogenic factors include the level of fertiliser application, whether livestock production is extensive or intensive, etc. (Liu et al., 2019[288]). For crop production, fertiliser use is often the most important source of GHG emissions, and the variation in the total GHG footprint within a given crop is often larger than the variation between different crops (Lam et al., 2021[289]). Moreover, for beef, there is a factor of two in the variation of the GHG intensity of Norwegian beef production (Samsonstuen et al., 2020[290]), as well as in Canadian beef production (Alemu et al., 2017[291]).[127]

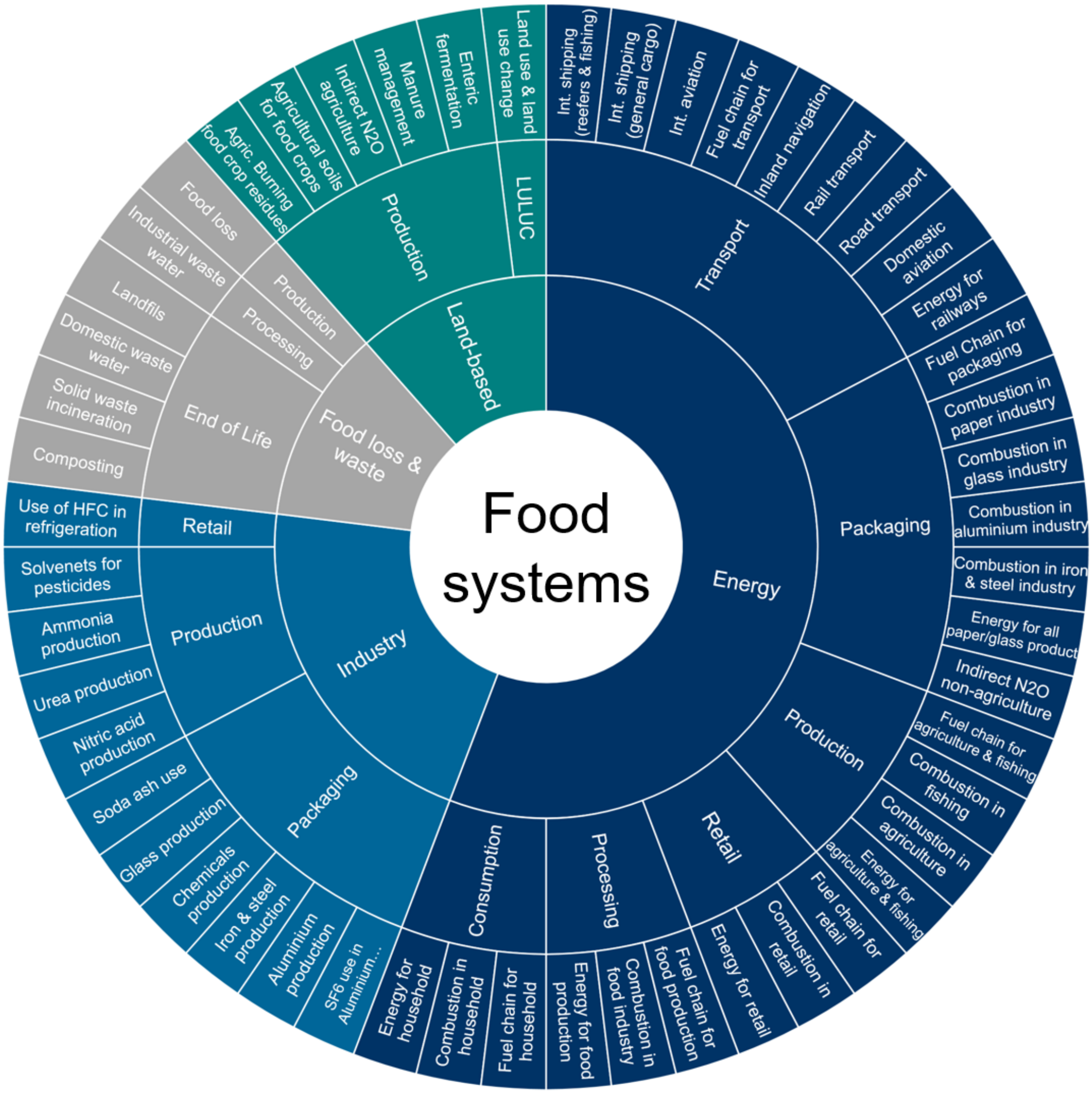
In addition to the variation of emissions intensity per food product, there can also be significant uncertainty in calculating GHG emission levels from specific sources. This includes particularly high uncertainties for N_2O, estimated at +/- 50% for N_2O emissions from soils (as a result of adding fertilisers) (Menegat, Ledo and Tirado, 2022[292]) and +/- 300% as a global average for N_2O from agriculture (Solazzo et al., 2021[293]). Nevertheless, it is most efficient to price emissions as close to the emissions source as possible. For food systems, this implies the supply-side.

There are also significant technical challenges associated with establishing an accurate carbon price signal for specific foods. These technical challenges include the extremely high number of diffuse emission sources in food systems, from a wide variety of actors (see Figure 5.6). If an accurate price signal requires information on GHG emissions from individual farms, as well as from post-production stages, this would imply a huge increase in monitoring emissions across different stages of food systems. This would include farm-level monitoring, and there are an estimated 570 million farms worldwide (Lowder, Skoet and Raney, 2016[294]), each with multiple emission sources and gases. It can take significant time and resources to develop farm-level monitoring systems, as demonstrated by the New Zealand “He Waka Eke Noa” programme (see Box 5.1) (New Zealand Agricultural Greenhouse Gas Research Centre, 2022[295]).

[127] For beef, the GHG-intensity of production is affected by several factors including the age of cattle at slaughter, the specific feed, and manure management (Business Wales, 2022[347]).

Establishing a pricing system on the supply- or demand-side that takes product-by-product, farm-by-farm, and post-production GHG variations into account would require a monitoring system that covers all stages of food systems, and that could track a particular food product from e.g. production to retail. Such a system could be spread over several countries – as well as across different actors, sectors and gases. If emissions pricing in food systems is introduced on the basis of average or benchmark GHG emission intensity levels, this could have implications for the accuracy of the price set, and consequent implications for the effectiveness of any GHG price signal in reducing emissions. In contrast, to date, ETS have focused on capping emissions from large point sources to reduce the administrative costs associated with establishing emissions caps and monitoring emissions. Countries could also make use of CBA, yet such systems may require a lot of resources to set up and run. Other technical challenges relate to the varying ability of individual farmers to reduce production-related GHG emissions, including how specific mitigation options are implemented, and the skill levels of individual farmers (Biological Emissions Reference Group, 2018[296]).

Figure 5.6. A large number of sectors contribute to total food system emissions



Note: The inner circle displays sectors, the second inner circle outlines life cycle food stages and the outer circle displays individual emission source categories (following the IPCC classification).
Source: Adapted from supplementary material in (Crippa et al., 2021[190]).

The economic feasibility of estimating emissions of a given food product across the food system is also likely to be challenging. In terms of the food production stage, requiring farmers to estimate, monitor and report their GHG emissions will increase their costs. The ability to meet such costs will vary widely depending on income and profit levels. These are influenced by many factors, including farm size – which differs greatly between different countries.[128] The relative importance of transaction costs – and therefore of the economic barrier to accurate monitoring – will increase as the size of farms and their profits decreases. Yet, farmers in OECD countries already have many reporting obligations and many also already adhere to quality schemes. Such efforts could potentially feed into GHG emissions monitoring and reporting requirements. However, this could also lead to market fragmentation, with some farmers being able to fulfil such requirements, while other farmers may not be able to, thus excluding them from the market. In the transport, packaging and processing stages, associated transaction costs could similarly also represent a significant barrier – particularly for smaller actors.[129]

Using an emissions pricing scheme to reduce the consumption and production of high-emitting foods, could also result in demand-side shifts and trade flow changes. There are limited studies investigating these aspects, and those which do mainly focus on the livestock sector. (Golub et al., 2012[297]) looked at the implications of a GHG emissions tax (USD 27/tCO_2eq) applied over a 20-year period to Annex I and non-Annex I countries on global livestock and agricultural production. Applying the emissions tax to Annex I countries alongside a forest carbon sequestration incentive was found to reduce production and emissions in Annex I countries by 3.9 GtCO_2eq for the entire period (around 46% of which could be generated in Annex I countries if the GHG tax is applied globally). (Henderson et al., 2017[262]) investigated the trade implications of a global GHG emissions tax of USD 20/tCO_2eq on ruminant emissions. The global tax projected increased costs of ruminant products from emissions-intensive regions and sectors.[130] This could potentially generate demand-side shifts, leading to a potential increase in the consumption of meat and ruminant products from less emissions-intensive regions and sectors. Given that meat from beef herds is more emissions intensive than meat from dairy herds[131], there could be an increase in the share of dairy meat and an equivalent decrease in beef meat. In South Asia and Sub-Saharan Africa, such a shift could be relatively large (8% each), while Latin America (5%) and East and Southeast Asia (2%) could also be affected by such a shift. In Oceania and Europe, beef production and exports could increase to compensate for the reduced cattle production in low-income regions, thus potentially altering trade flows. (Avetisyan et al., 2011[298]) also found that a global emissions tax could shift livestock production to more affluent countries with lower emissions intensities. Yet more research investigating potential demand-side shifts and trade flow changes onset by different types of emissions pricing in food systems would be beneficial.

Current levels of direct agricultural support encouraging the production of high-emitting food products could also undermine the effectiveness of a potential GHG emissions pricing scheme in food systems. Just as the positive impacts of carbon pricing have been counteracted by fossil fuel support (see section 3.1). Global agricultural support exceeded USD 817 billion per year in 54 countries from 2019-21 (see section 5.3), with high-emitting commodities receiving a large share of this support (OECD, 2022[299]). The

[128] For example, the average size of a farm in the US is more than 100 times that of an average farm in India, and the average income of a US farm is more than 20 times more than that in India (USDA ERS, 2022[408]) (Vinaykumar, 2022[409]). The average farm size in India dropped to 1.08 hectares in 2015/16 (Ministry of Agriculture & Farmers Welfare, 2020[410]) whereas in the US, the average farm size was 180 hectares in 2019 (USDA, 2020[411]).

[129] Tools exist to compare the environmental impact of different packaging designs. However, the cost of using these tools can be thousands of Euros per year – which would put this beyond reach for many

[130] (Henderson et al., 2017[262]) does not provide data for specific price increase. Yet (Glauber, 2018[454]) found that applying a GHG emissions tax of USD 20/tCO_2eq on beef in 2019 would generate significantly different price increases in different countries such as Australia (+11%), European Union (+8.2%), India (+54.4%), Ethiopia (+71.5) and US (+6%).

[131] Meat from dairy herds ("dairy meat") has lower GHG emissions intensities as emissions account for milk and meat production, whereas emissions from beef herds only account for meat production (OECD, 2022[215]).

provision of such support can influence farmers' decision on several aspects, such as what and how much to produce, and where and how to produce it (Mamun, Martin and Tokgoz, 2021[300]), and could potentially also disrupt any potential GHG emissions pricing signals.

The introduction of payments for emissions sequestration or removal in food systems is further complicated by the potential reversibility or non-permanence of these (Henderson et al., 2022[216]), which may require specific provisions to ensure that any reversals are accounted for. Reversal of sequestered emission can occur if the CO_2 that was stored either above ground (e.g. trees) or below it (e.g. roots, soils) is re-emitted in the future as a result of forest fires, harvesting, tilling of no-till soil, etc. The risk of potential reversal of the GHG benefits of selected activities – by either anthropogenic or natural causes – applies only to GHG sequestrations (and not to GHG reductions). Any such reversals that occurred on agricultural land and which were bought to generate a reduction in the GHG intensity of food products, would alter the previously calculated GHG intensity. To ensure that any reversals of GHG removals are accounted for in emissions pricing, safeguards may need to be developed. These safeguards could include: requiring extended monitoring periods (to track the level of any reversals); allocating temporary "lifetimes" of offsets from removal activities, with a requirement to replace an offset once it has expired (as was done under the Kyoto Protocol's Clean Development Mechanism); requiring any reversals to be addressed (as outlined in the framework agreed for Article 6.2 of the Paris Agreement).

5.4.2. Political and just transition challenges related to implementing carbon pricing in food systems

In addition to the technical and methodological challenges, there are significant political challenges in implementing GHG emissions pricing in food systems. Implementing emissions pricing on the supply-side could decrease the profitability of GHG-intensive foods, and it would thus impact specific groups. Livestock farmers (especially beef farmers) would risk being negatively affected and this could lead to strong opposition in countries with large livestock sectors. For example in 2022, farmers in Ireland (Murphy, Cannon and Walsh, 2022[301]), the Netherlands (Financial Times, 2022[302]), and New Zealand (potbsmap, 2022[303]) strongly opposed proposed government policies aiming to reduce GHG emissions in the agricultural sector. Yet such opposition could potentially be reduced by including farmers and other stakeholders in the design of an emissions pricing scheme or by for instance recycling the revenue generated to help farmers adjust, as seen in New Zealand (see Box 5.1). Governments could also gradually phase-in an emissions pricing scheme, allowing stakeholders to slowly transition. Financial aid or preferential grants could also be allocated to help ease a transition as seen suggested in the Netherlands (Holligan, 2022[304]). Moreover, governments could also use policy packages entailing policies that could reduce negative impacts and foster a transition.

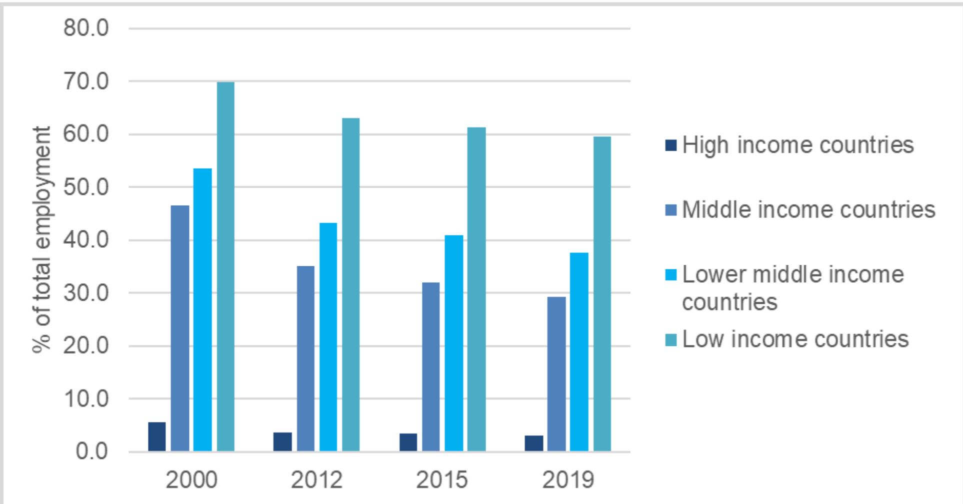
There could also be additional short-term political challenges in implementing GHG emissions pricing given the high rates of food inflation in most countries in 2022. More than 92% of low and middle-income countries, and 83% of high-income countries have experienced food inflation of more than 5% between Q2 in 2021 and 2022 (World Bank, 2022[305]). For example, costs of some basic food products such as wheat and palm oil have increased more than 50% between Q2 2021 and Q2 2022 (World Bank, 2022[306]). If the introduction of GHG emissions pricing in food systems would lead to further increases in food costs, there is likely to be significant political and societal pushback. Yet, if food inflation is tamed in a couple of years, political and societal pushback is expected to reduce.

Although emissions reduction in the livestock sector is key to lower the food systems' contribution to climate change, adjustment of herd management may also come with some trade-offs. Livestock can play an important role for farmers' livelihoods in some countries and animal grazing on land unsuitable for reforestation/afforestation or crop production could, under specific circumstances, generate potential benefits for biodiversity (OECD, 2015[307]), pasture productivity (via manure), soil carbon sequestration (Henderson et al., 2015[308]) (FAO, 2006[309]) as well as help control and prevent weeds and invasive plant species (OECD, 2021[310]). Nonetheless, overgrazing can also lead to soil degradation, thereby increasing the risk of erosion and increasing GHG emissions (FAO, 2020[311]), and soil sequestration rarely offsets the climate impact from animal non-CO2 emissions (Garnett et al., 2017[312]). However, livestock may be important to farmers for other reasons, such as capital insurance and animal traction in agricultural labour (e.g. ploughing) (OECD, 2021[310]).

Implementing emissions pricing in food systems, could also lead to just transition concerns, given the potential negative knock-on effects of introducing emissions pricing on employment, farmer income, and food security. In 2020, 27% of the global workforce (874 million people) were employed in agriculture (FAO, 2021[194]). However, there are wide geographical variations in the proportion of the population employed in agriculture, ranging from 5% in Europe to almost 50% in Africa (FAO, 2021[194]). Applying GHG emissions pricing in food systems could change both the structure of employment within the agricultural sector (and particularly in the subsectors producing the most GHG-intensive foods), as well as the total number of people employed. Furthermore, it could also contribute to increased food prices and thereby to a rise in food insecurity. Due to limited data on potential just transition effects of emissions pricing (e.g. on employment, farmer income and food security) across different stages of food systems, the following sections focus on potential just transition impacts in the production and the consumption stages.

Emissions pricing in agricultural production could have just transition impacts on employment and farmer income. As can be seen in Figure 5.7, employment in agriculture has decreased across all country groups from 2000 to 2019. In 2019, the highest percentage of agricultural employment was found within low-income countries, with 59.5% of the total employment, while this was 2.8% and 4.8% for high-income countries and OECD countries, respectively (World Bank, 2022[185]). Putting in place emissions pricing policies would not only affect sectoral employment overall, but there could also be concentrated impacts on farmers producing high GHG-intensity products, e.g. livestock farmers. Although data on livestock employment are scare, satellite data shows that cattle remain the most common ruminant livestock, mainly present in South America, Middle and East Africa, Asia, and Europe. Many sheep and goats are also found in Middle and East Africa, Asia, and in parts of Europe (Gilbert et al., 2018[313]).

Figure 5.7. Employment in agriculture across country income groups



Note: Included years are based on data availability.
Source: based on data from (World Bank, 2022[185]).

A GHG food tax of EUR 16/tCO_2eq, EUR 60/tCO_2eq, and EUR 290/tCO_2eq in the EU could result in average producer losses for all farmers of EUR 2, EUR 7, and EUR 9 billion annually, respectively. This is equivalent to 0.89% to 11% of the average farm income across all EU farms, yet economic losses are estimated to be even greater for beef and dairy farms (Jansson and Säll, 2018[266]). Emissions pricing, based on individual farm emissions, would generate larger income reductions for livestock farmers with high GHG intensity-systems (e.g. South Asia, Africa, Latin America and the Caribbean) compared to farmers with low GHG intensity-systems (e.g. Europe and North America) (Gerber, 2013[197]). Thus, from a just transition perspective (see section 2.2), it is essential that governments engage with affected stakeholders. Developing a social dialogue between governments and affected stakeholders could improve the understanding of the impacts and needs of different groups, as well as outline how targeted and proactive measures, e.g. social protection policies, could help ensure farmers' and workers' livelihoods in a transition to low-carbon farming. Utilising a beneficiary-pays approach, where farmers are paid for their emissions reductions (e.g. soil carbon sequestration), could generate additional farm income and thus contribute to a just transition. Such an approach could help reduce negative income and employment impacts, although, it may be unable to offset the total loss of farm income alone, especially given its voluntary nature (Lehtonen, Huan-Niemi and Niemi, 2022[314]). Furthermore, such abatement payments may be most effective if they are results-based (Sidemo-Holm, Smith and Brady, 2018[315]).

On the consumption side, there could be just transition impacts related to the regressive effect of polluter-pays emissions pricing as it could increase food prices, which could have regressive impacts[132] and increase risk of hunger and poverty for parts of the populations ((Hasegawa et al., 2018[316]) and (Hussein, Hertel and Golub, 2013[317]). A GHG food tax of EUR 16/tCO_2e, EUR 60/tCO_2eq, and EUR 290/tCO_2eq in the EU would increase food prices and could result in consumer losses of EUR 9, EUR 33 and EUR 159 billion annually, equal to 0.04% to 0.78% of total expenditure in the EU (Jansson and Säll, 2018[266]). . An emissions price on foods could furthermore increase the relative prices of some processed foods, e.g. those containing beef compared to poultry or plant-based alternatives (Clark et al., 2022[318]). Consumers in regions with higher-GHG intensities per output are also likely to face higher food insecurity risks as price increases will likely be larger than in places with low-GHG intensities per output (Frank et al., 2017[319]). A uniform carbon price aligned with a scenario limiting warming to 1.5°C could result in a calorie loss of 110-285 kcal per capita per day by 2050. Such a loss would be equivalent to an increase in undernourishment of 80-300 million people by 2050 (Frank et al., 2017[320]). Furthermore, emissions pricing could also results in reduced animal-protein intake, and unless replaced by plant-based protein, nutritional risks related to inadequate protein intake could occur (Grant, Lusk and Caputo, 2020[321]). There are, therefore, also just transition consumer implications to implementing emissions pricing on the demand-side of food systems.

5.4.3. Opportunities for applying emissions pricing in food systems

There are significant short-term challenges related to introducing emissions pricing in food systems, but, nevertheless, there are potential policy avenues, both on the supply- and the demand-side. Reducing GHG emissions from food systems could have many co-benefits (e.g. reducing deforestation, biodiversity losses, air and water pollution), as well as being needed in order to reach net zero GHG emissions (UNEP, 2022[8]).

At present, the agricultural sector benefits from significant levels of support – some of which increases GHG emissions from this sector (OECD, 2022[215]), and reforming agricultural support could help encourage more environmentally friendly farming practices. Phasing out most environmentally harmful support and investing more in general services and public goods (e.g. rural landscape preservation, habitat provision, and invasive species control) could generate positive environmental outcomes (OECD, 2022[215]) (Gautam et al., 2022[244]). These positive environmental outcomes could both entail GHG emissions reductions as well as biodiversity benefits (OECD, 2022[187]). However, the outcome of such an approach also depends on the type and effectiveness of management practices and the environmental requirements

[132] Studies in various countries have found the application of emission pricing on food could lead to regressive impacts, e.g. in France (Caillavet, Fadhuile and Nichèle, 2019[388]); Sweden (Röös, Säll and Moberg, 2021[348]), Scotland (Chalmers, Revoredo-Giha and Shackley, 2016[390]) and in the US (Tiboldo et al., 2022[389]).

that accompany payments. There is also a need for more support going towards R&D and innovative practices to help develop new technologies and improve the emissions efficiency of production systems (OECD, 2022[187]) Reforming agricultural support might be politically difficult or too slow in some contexts, but transformation of the sector is needed to ensure that agriculture contributes to the long-term climate commitments under the Paris Agreement.

Implementing demand-side policies together with polluter-pays emissions pricing could help reduce potential distributional effects on consumers. Potential regressive impacts of a polluter-pays approach could be addressed through the application of revenue recycling, lump sum payments (Maestre-Andrés, Drews and van den Bergh, 2019[100]), or subsidisation of certain food groups (Andreyeva et al., 2022[322]). In scenarios where consumers are not compensated for the increase in taxation level, there is an overall decrease in total kilojoules (kJ) consumed daily. On the other hand, if consumers are compensated, there is an increase in total kJ consumed daily (Edjabou and Smed, 2013[323]). Andreyeva et al. (2022[322]) found that fruit and vegetable subsidies for low-income populations were associated with increased sales and thereby an increase in consumption. Communicating the possible health and nutritional benefits of taxing high-emitting low-nutritious foods, could potentially also help address a consumer backlash (Clark et al., 2022[318]). Further information on plant-based protein sources and subsidisation of these could also contribute to increasing the intake of plant-based protein, and reducing nutritional risks related to inadequate protein intake.

The use of other policies such as “feebates” (Batini, Parry and Wingender, 2020[324]) could also be explored to influence the relative demand for different food items. Feebates could increase the price of more GHG-intensive food products while at the same time reduce the price of less GHG-intensive food products (Batini, Parry and Wingender, 2020[324]).[133] However, the fruit and vegetable subsidy was estimated to increase emissions by 4.6% (Broeks et al., 2020[325]). Thus, combining taxes and subsidies through a feebates approach could potentially reduce any emissions increases, while still providing societal benefits. Furthermore, combining feebates with nudging measures could also stimulate healthy purchases (Hoenink et al., 2020[326]), which are often also environmentally friendly (Clark et al., 2022[318]). Yet, as with other pricing approaches, there could be implementation challenges and opposition similar to those of emissions pricing (see Technical and methodological challenges related to applying emissions pricing in food systems), which could potentially render feebates politically infeasible.

Demand for low-GHG foods could be further encouraged by focusing on their co-benefits, including often their lower cost. In countries where there are low levels of food insecurity, governments could focus on emphasising the link between healthy and low-GHG diets (Clark et al., 2022[318]). There are significant potential health improvements from substituting animal food products with whole grains, cereals, fruits, vegetables, nuts and certain vegetable oils. For example, the use of feebates could potentially help prevent 10-39% of cancers in Europe over a 20-year risk period (Laine et al., 2021[327]). A UK study also found health benefits from reduced meat and processed food consumption (Aston, Smith and Powles, 2012[328]). However, health and environmental impacts from reducing meat consumption depends on the food consumed instead of meat (Guasch-Ferré et al., 2019[329]). Low-GHG diets are also associated with lower impacts on other environmental parameters. For instance, current dietary patterns use 87% of total agricultural land including cropland and pasture to feed livestock (Poore and Nemecek, 2018[195]). Replacing half of the animal-based food consumption with vegetable equivalents could decrease land use by 67%, reduce acidification by 64% and eutrophication by 55% (Poore and Nemecek, 2018[195]). Such

[133] An ex-ante study in Spain found that taxing red meat and subsidising carbohydrates (e.g. vegetables, fruits, and pulses) at up to 30-40% of the market price could reduce GHG emissions and improve the Spanish diet (Markandya et al., 2016[414]). In the Netherlands, a 15% or 30% meat tax or a 10% fruit and vegetable subsidy could, over a 30-year period, lead to societal benefits, at a value of EUR 3.1-7.4 billion, EUR 4.1-12.3 or EUR 1.8-3.3 billion, respectively. These would for the meat tax scenarios include both environmental (GHG emissions reductions) and health benefits (reduced healthcare costs and increased quality of life due to higher intake of fruits and vegetables).

changes could result in additional space for (and reduced pressure on) nature, thus helping to create better conditions for many ecosystems.

Sufficient protein is important for a nutritionally adequate diet, and plant-based protein generally has lower GHG emissions, and water, land and energy requirements compared to animal-based protein, yet there are also differences between different kinds of plant-based proteins. (Reijnders and Soret, 2003[330]) compared meat and soybean protein production and found that meat protein production had 6-17 times larger land requirements, 4-26 times greater water requirements, seven times larger use of phosphate rock (used for synthetic fertiliser and feed additives) and produced seven times more acidifying compounds. (Santo et al., 2020[331]), (Frezal, Nenert and Gay, 2022[332]), and (Zhu and van Ierland, 2004[333]) similarly reported lower GHG emissions, eutrophication, acidification, and energy, water and land requirements for plant-based meat substitutes compared to meat. There are also differences in the environmental impact of various plant-based protein sources. Yet, (Fresán et al., 2019[334]) and (Detzel et al., 2021[335]) found plant-based meat substitutes to have water requirements and environmental impacts similar to unprocessed or minimally processed animal-sourced products. Nevertheless, animal protein remains an essential source of nutrition in certain countries, and it will be important to ensure the availability of sufficient protein to meet the nutritional demands of a growing global population.

In countries where there is a higher level of food insecurity, co-benefits of reducing high-GHG food are also potentially important. For example, animal-based foods mostly have higher water footprints (Clark et al., 2022[318]), thus growing less water-intensive crops could be helpful in areas prone to droughts. This may become a key consideration given that climate change is expected to exacerbate water scarcity. Moreover, partly reducing the herd size, while increasing low-land use crops could contribute to both plant diversification and to the sparing of land that could be used to provide other ecosystem services (Isbell et al., 2017[336]) (Benton et al., 2018[337]). However, livestock may be used by famers for other purposes than food, e.g. as capital insurance and animal traction.

Further exploration of the potential applicability of different types of GHG emissions pricing approaches to various food systems in different contexts is needed. Additional research could help identify which approaches could be used to help transform various food systems across the globe in a just manner to ones that are low-GHG emitting, and that deliver adequate food for all. Although challenging, it is a key task to ensure that food systems are put onto a more sustainable footing in the medium- and long-term.

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