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Pursuing Higher Environmental Goals for Agriculture in an Interconnected World: Climate Change and Pesticides

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Governments in many countries are pursuing higher environmental goals for agriculture. However, in an interconnected world, the unilateral adoption of environmental policies for agriculture can reduce the producers' competitiveness and induce pollution leakage. This report analyses these challenges and discusses policy solutions, focusing on two examples: climate change mitigation policies and policies limiting the environmental impacts of pesticides. The extent of competitiveness and leakage effects is found to depend on market conditions, differences in pollution intensity, and the type of environmental policy adopted. Two policy routes are identified to improve agriculture's environmental performance while maintaining the benefits of global markets. The first route relies on "direct" environmental policies, such as market-based instruments or regulations, which are rapidly effective in limiting environmental impacts but may require additional complementary policies to limit their potential competitiveness and leakage impacts. The second route involves alternative policies acting on agricultural supply, demand, or through private sector engagement, which limit competitiveness and leakage impacts but may require time to be environmentally effective.

Key words: Agriculture policy, environmental policy, trade policy, climate change, pesticides, competitiveness, pollution leakage

JEL codes Q17, Q18, Q58

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* E. Migliaccio, E. Ellis, W. Kodama, and L. Roffredi were working at the OECD when contributing to this report.

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Key messages

- In an interconnected world, the unilateral adoption of environmental policies for agriculture can reduce the international competitiveness of producers and induce environmental impacts in other countries not applying such policies, a phenomenon called pollution leakage.
- While subject to some limitations, this paper examines available evidence on both agricultural greenhouse gas (GHG) mitigation and policies limiting the environmental impacts of pesticides. It first suggests that the extent of competitiveness and leakage challenges depends on domestic and international market conditions, differences in pollution intensity across trading partners, and the type of environmental policy adopted.
- There are two broad policy routes that countries can take to improve environmental performance in their agricultural sector while maintaining the benefits of open and predictable global markets. Both have benefits and challenges.
- The first route relies on the use of "direct" policies, such as pricing GHG emissions or pesticides bans or use regulations, which are the most immediately effective instruments in improving environmental performance. However, they may induce pollution leakage and competitiveness loss.
- These effects can be mitigated by additional, complementary policies which can be pursued multilaterally or unilaterally. Multilateral approaches enable global solutions to global challenges and limit leakage and competitiveness concerns, but they require significant co-ordination efforts. The extent to which unilateral approaches are effective depends on their specific design and, while they do not require co-ordination, they may pose implementation challenges and could lead to trade disputes if not designed in accordance with WTO rules.
- The second route, relatively more indirect in improving environmental performance, involves employing alternative policies. These include supply-side options (such as agri-environmental payments), demand-side options (such as green public procurement), as well as private sector engagement mechanisms (such as due diligence mechanisms). These policies can be cost-effective options with limited environmental and trade-related challenges, but they may require more time to achieve desired environmental outcomes. In particular, research and development can play a key role, generating both competitiveness gains and environmental benefits but may require time to be effective.

Executive Summary

Governments in many countries are elevating their environmental goals for the agriculture and food sector. For instance, several major economies have adopted net zero greenhouse gas (GHG) emission or zero biodiversity loss targets, including in the agriculture, forestry and other land use (AFOLU) sector. G20 members and participants in the UN Food System Summit have also pledged broader efforts towards the sustainability of agriculture.

In the context of interconnected markets, however, the unilateral adoption of more stringent environmental policies by some countries could have unintended consequences. Specifically, in agriculture and food, such efforts could potentially reduce the international competitiveness of domestic agricultural producers, at least in the short to medium term. They could also increase production and generate environmental impacts in other countries that are not implementing such measures, a mechanism called pollution leakage, thereby limiting the overall efficacy of the new environmental policy.

Are these environmental and economic concerns present for all types of policies and agricultural markets? What factors may increase their importance? What can be done to limit the identified environmental and trade-related challenges? The report addresses these questions through an assessment of the existing literature and evidence, focused on two illustrative, but highly relevant cases: GHG mitigation policies in agriculture and policies to reduce the environmental impact of pesticides. It should be noted that several important caveats apply to this analysis, from the choice of policy instruments to the limited evidence used to review the selected policy instruments. Further research is needed to ascertain the effects of certain policy approaches, particularly those emerging in policy debates.

The reviewed evidence shows that challenges related to competitiveness and pollution leakage are not always present in the case of unilateral policy action to improve the environmental performance of domestic agriculture. The analysis suggests that competitiveness loss and pollution leakage will only emerge or be significant for specific environmental policies in the short to medium term. Four main factors affect the likelihood that new unilateral policy adoption will create competitiveness losses and pollution leakage in the short to medium term: (1) the impact of the new policy on agricultural production, with direct environmental policies much more likely to raise these challenges; (2) the responsiveness of domestic demand to price and the rate of substitution between domestic and foreign products; (3) differences in pollution intensity (defined as pollution per unit of product) between the regulating country and its trade partners; and (4) international market conditions for the affected agricultural commodities.

These four policy and market factors apply differently depending on the environmental policy domain:

- In the case of *GHG mitigation*, policies like market-based instruments that impose a burden on emitters, such as pricing mechanisms, are the most likely to lead to GHG leakage and competitiveness loss of domestic producers. Livestock activities are expected to be the most affected by new pricing policies, due to their high GHG intensity and the higher rigidity of their production structure. Rice production could also be affected, but to a lesser extent. These effects are larger when carbon pricing is adopted in a country with low GHG intensity of production that trades with countries with higher GHG intensity.
- In the case of *pesticides*, the analysis suggests that regulations, and particularly banning an important pesticide, could lead to significant loss in competitiveness, especially in the case of staple crops with relatively thin international markets. Pesticide taxes and pesticide use regulations, if not overly stringent, may be less likely to create environmental and trade-related challenges.

Against this background, the paper highlights the choice between two possible routes to reduce environmental harm in the agricultural sector while enabling the benefits of open and predictable international markets. A first route involves the rapid adoption of direct environmental policy instruments that are the most effective in achieving their environmental aims domestically but may induce other environmental and trade-related challenges and may require additional policy measures to limit these challenges. The second route involves the implementation of alternative policies that may be less immediately effective in achieving the environmental aim domestically but pose limited other environmental and trade-related challenges and therefore do not require additional measures.

Under the first route, governments can employ direct environmental policies, such as market-based instruments or environmental regulations. While they are the most immediately effective instruments to tackle environmental challenges, they might induce environmental and trade-related issues and may require additional, complementary policies to mitigate them.

Such complementary policies can be pursued multilaterally or unilaterally. Multilateral approaches, through different types of international regulatory co-operation mechanisms, enable global solutions to global problems and reduce environmental and trade-related challenges associated with direct policy instruments, but they require significant co-ordination effort. The attenuating effects of unilateral approaches on leakage and competitiveness losses depend on their specific design. Additionally, unilateral responses pose implementation challenges and could lead to trade disputes if not appropriately designed, including in accordance with WTO rules.

Under the second route, alternative policies can be implemented to reduce environmental harm in the agricultural sector. These options include supply-side policies (such as agri-environmental payments), demand-side policies (such as green public procurement), as well as private sector engagement mechanisms (such as due diligence mechanisms). These policies can be cost-effective options, with limited negative environmental and trade-related effects, although their effectiveness in achieving domestic goals varies by instrument, and they can be slower to have impact. Some of the most viable options, such as research and development, can play a key role, generating both competitiveness gains and global environmental benefits, but they may require time to be effective.

1. Pursuing higher domestic environmental goals in an interconnected world

Following the development of international environmental objectives, including the UN Sustainable Development Goals, the 2015 Paris Agreement on Climate Change and the Global Methane Pledge, many countries have raised the ambitions of their national environmental objectives. In particular, major economies have adopted net zero greenhouse gas (GHG) emissions, biodiversity or oceans targets.[1] Several countries have also assigned significant portions of their COVID-19 recovery policy packages towards environmental sustainability. For example, the European Union has earmarked 30% of its funds to fighting climate change.

These trends also apply to the agriculture and food sector. In particular, OECD, G7 and G20 agricultural ministers have committed to improving the environmental sustainability of agriculture and food (G7, 2021[1]; G20, 2021[2]; OECD, 2022[3]). For example, as part of the European Green Deal, the European Union has set environmental sustainability goals including a reduction in fertiliser and pesticides use by 20% and 50% by 2030, respectively, and an expansion of organic agriculture to be 25% of all farmland in their Farm-to-Fork strategy (European Commission, 2020[4]). Japan's Strategy for Sustainable Food System targets zero GHG emission from agriculture and reduced use of fertiliser (20%) and pesticides (50%) by 2050, along with 25% of farmland under organic production (Ministry of Agriculture, Fisheries and Forestries of Japan, 2020[5]). New Zealand has adopted legally binding biogenic methane mitigation targets (-24% to -47% compared to 2017 by 2050) (Henderson, Frezal and Flynn, 2020[6]).

This growing but nationally driven environmental policy ambition, however, has raised environmental and trade-related concerns, including in the agricultural sector. Specifically, in an interconnected world, heterogeneous domestic policy targets, efforts and pacing towards environmental sustainability could lead to unwanted economic and environmental consequences (Fuchs, Brown and Rounsevell, 2020[7]; Baylis, Heckelei and Hertel, 2021[8]; OECD, 2020[9]). On the economic side, the costs of complying with new environmental measures and the potential productivity losses producers may face domestically could put them at a competitive disadvantage with producers in other countries. On the environmental side, stronger domestic environmental efforts could result in pollution leakage in foreign countries adjusting their production to occupy vacated international market shares (Tamiotti et al., 2009[10]; IPCC WGIII, 2014[11]).[2]

Are these environmental and economic concerns present for varying types of policies and agricultural markets? What factors may increase their importance? What can be done to limit the identified economic and environmental concerns? This report aims to address these questions looking in particular at policies to mitigate GHGs and reduce the environmental harm of pesticides. Understanding these complex environmental and trade-related issues is necessary to identify economically viable policy options that allow countries and international communities to achieve environmental goals for the agricultural sector while maintaining the benefits of open and predictable global markets.

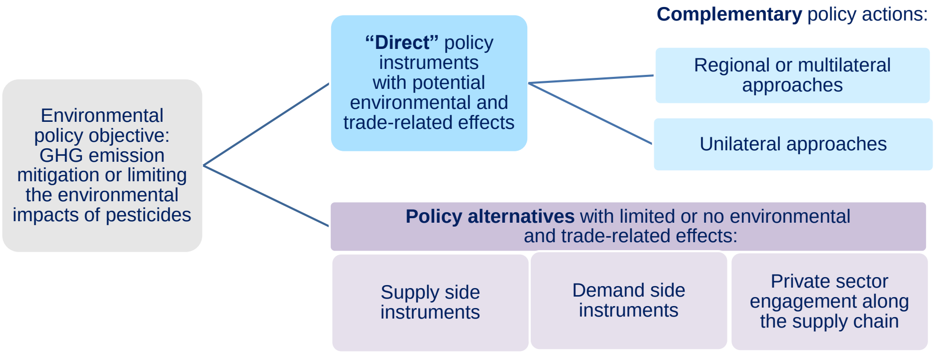
The analysis identifies two broad routes to progress on environmental objectives while facilitating international markets that involve two sets of environmental policies, as shown in Figure 1. The first set of policies, that will be called "direct" throughout the paper, aims to tackle directly its targeted environmental objective.[3] These policies are generally effective in the short to medium run, but they could generate environmental and trade-related challenges and require therefore "complementary actions" to mitigate these challenges. The second set of policies, that will be called "alternatives" throughout the paper, would induce limited or no pollution leakage and competitiveness losses, but may require more time to be effective.

[1] Within this context, net-zero is defined using Carbon Dioxide equivalents (CO2-eq), a metric measure used to compare the global warming potential of different GHGs (Eurostat, 2017[310]). GHGs amounts are converted to the amount of CO_2 with the same global warming potential.

[2] The phrase "environmental and trade related concerns" will be used to stand for competitiveness and/or pollution leakage effects.

[3] This does not imply however that they will always be more effective than other policy instruments in particular contexts.

Figure 1. Decision tree to address environmental and trade-related challenges



Section 2 of this report analyses the challenges of environmental policies in agricultural markets, which includes potential competitiveness loss and pollution leakage, and identifies four policy and market factors affecting their presence and magnitude.

Section 3 focuses in particular on the first route, which employs "direct" policy instruments, such as carbon taxes for GHG mitigation, pesticides bans, pesticide use regulations and other stringent regulations. It defines their scope and evidence of their creating environmental and trade-related challenges. The section also analyses multiple complementary policy actions to mitigate the challenges. These encompass multilateral, regional and bilateral approaches to enhance international co-operation, as well as unilateral policies that countries can adopt to limit pollution leakage or competitiveness losses.

The second route to approach environmental challenges in agriculture is through indirect policies that might require time to be effective. Section 4 of the report reviews potential alternative policies to the direct instruments, which would induce limited or no pollution leakage and competitiveness losses, and thus could serve as substitutes or complements. This includes instruments acting on the supply side, targeting changes in agricultural production; instruments targeting food demand; and approaches engaging the private sector along the agriculture and food supply chain. Section 4 explores whether the reviewed alternatives are cost-effective and whether they entail any possibility of leakage or competitiveness losses.

Several caveats apply to this analysis. First, not all plausible policy instruments are covered. Second, it should be noted that any challenges resulting from policy implementation would be part of a broader net cost-benefits analysis. While the report describes the benefits of certain policies, a comprehensive environmental cost-benefit analysis is beyond its scope. Third, the assessment is based on a first review of the literature, which does not claim to be complete. As such, while comparisons can be made, it is difficult to draw robust conclusions as to the relative merit of each policy instrument. Nor does the study identify the best policy package from an economic or environmental perspective.

2. Challenges of environmental policies in agricultural trade: Leakage and competitiveness

There are two main environmental and trade-related concerns related to the adoption of domestic environmental policies in a globalised market (Tamiotti et al., 2009[10]; OECD, 2020[9]). The first is economic: a growing difference in regulation may create a wedge in farm competitiveness internationally. This leads farms in less regulated countries to produce more and export towards more highly regulated countries, to the detriment of their producers. The second concern is environmental. It postulates that international regulatory differences will limit the effectiveness of environmental policy efforts because of the pollution leakage.

Competitiveness loss or competitive disadvantage occurs when domestic producers become less competitive due to domestic (environmental) regulations or policies (Mulatu, Florax and Withagen, 2003[12]; Arvanitopoulos, Garsous and Agnolucci, 2021[13]). Domestic producers in implementing countries face higher production costs and pass on some of these costs to consumers. Buyers in response may switch to relatively cheaper products (e.g. imported goods) of similar quality.[4] In an international trade context, competitiveness loss can be measured by a change in trade flow or domestic output of a specific product (Jaffe, Peterson and Portney, 1995[14]; Arvanitopoulos, Garsous and Agnolucci, 2021[13]) even though its definition remains relatively vague in the literature (Mulatu, Florax and Withagen, 2003[12]). From a producer welfare perspective, a change in output may be an acceptable measure of competitiveness, so this is the one that will be mainly used here.

The concept of pollution leakage was first discussed in the 1990s (Copeland and Taylor, 1994[15]; Grossman and Krueger, 1991[16]) in the context of the environmental impact of trade liberalisation, in particular the North American Free Trade Agreement (NAFTA) (Arvanitopoulos, Garsous and Agnolucci, 2021[13]). A pollution leakage occurs when stringent environmental policies in implementing countries lead to an increase in pollution in countries with weaker environmental regulations (OECD, 2019[17]; OECD, 2020[9]). The two main paths for this are via movement of firms or loss of competitiveness by domestic producers. Under the first path, environmental regulations and policies that internalise the external costs of pollution will often affect production processes and costs in implementing countries. Domestic firms may respond by moving to a country where they may produce with lower costs.[5] Under the second path, local producers are less competitive due to their higher production costs and lose domestic or international market share. Producers in non-implementing countries become relatively more competitive, produce more goods, and emit more pollution (although overall pollution may not necessarily rise). In agriculture, the first path is unlikely because farmers do not usually migrate to other countries. Therefore, loss of competitiveness is of most concern for agri-environmental policies.

The economic mechanism for those two effects to occur can be understood with a simple representation of trade (as detailed in Box 1). The application of a strict regulation in an importing country imposes costs on producers and potentially declines their production, which can be defined as a competitiveness loss. The decline in production can lead to higher commodity prices depending on elasticities. Net import can increase, assuming demand is relatively price inelastic, as would be the case for food staples. The decline in production is compensated by an increase in production in exporting countries. Environmental damages decrease because of the decline in domestic production. However, exporters increase their production and therefore their potential additional environmental damages. This increase in damage partially offset the reduction in the importing country, which is a case of pollution leakage. The degree of leakage (or leakage rate) is defined as the ratio of increased damage in the exporting country to the reduced damage in the importing country and depends on the respective pollution intensities.

Box 1. Simple economic representation of trade-related effect of unilateral environmental policies

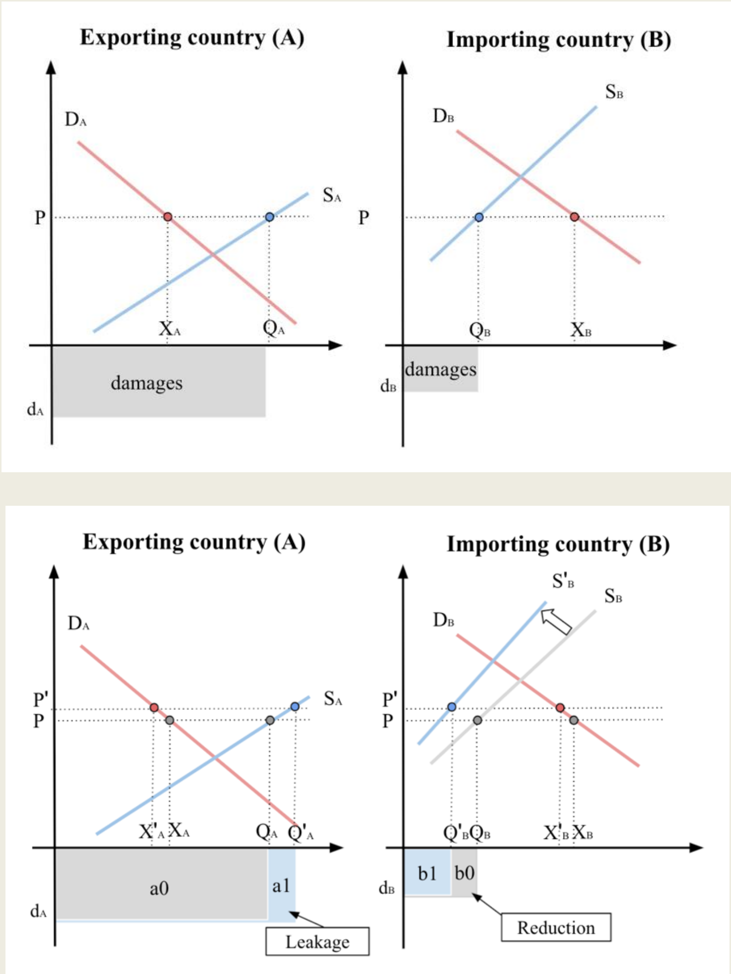
Consider the simplified example of two-country open economy with an exporting and importing country, denoted as A and B, respectively. These two countries trade a specific product (e.g. corn, rice, meat, or milk) as depicted in the higher panel of Figure 2. Demand and supply curves are shown as red lines (D_A and D_B) and blue lines (S_A and S_B), respectively. The environmental damage of existing production processes is the product of domestic production (Q_A and Q_B) and the damage parameter (d_A and d_B) and shown on the below part of the figure. In this example, the exporting country (A) emits more pollution to produce a unit of output than does the importing country (B) – i.e. dA>dB. X_A and X_B represent the two countries' domestic consumption. The equilibrium price (P) balances exports (Q_A– X_A) and imports (X_B– Q_B). Without regulation, environmental damages in country A and B are a0 and b0, respectively.

[4] If the costs are not passed on to the consumers or the consumers do not switch to substitutes, these effects would not occur.

[5] This mechanism is frequently called "pollution haven effects" in the literature.

Figure 2. Policy, trade and environmental damages

Upper panel: No change in policy   Lower panel: New environmental policy in B



Source: Authors, based on an illustration developed by Beghin, Disdier and Marette.

If a strict regulation is implemented in the importing country (B), producers there face higher production costs, and their supply curve moves upward (S'_B) (lower panel of Figure 2). This shift leads to an increase in equilibrium price (P') which influences consumers, producers, and environments in both countries, as follows.

- Domestic and foreign consumers face higher commodity prices and consume less. Producers in the importing country produce less (from Q_B to Q'_B), which we define as a competitiveness loss.
- Net import changes from ($X_B - Q_B$) to ($X'_B - Q'_B$), which is positive in most cases assuming demand is relatively inelastic with respect to price increases, as would be the case for food staples. The decrease in the domestic production in country B is substituted by an increased production in the exporting country (from Q_A to Q'_A). These changes increase the traded quantity between the two countries.
- Environmental damages in country B decrease by b0 – b1 (= $d_B*(Q_B - Q'_B)$) because of the decline in domestic production. However, producers in country A produce more, which leads to additional environmental damages amounting to a1 – a0 (= $d_A*(Q'_A - Q_A)$). A part of the environmental damages reduced in country B is offset by an increase in country A; pollution leakage. The degree of leakage (or leakage rate) is defined as the ratio of increased damage to reduced damage ((a1 – a0)/(b0 – b1)) and depends on the respective pollution intensities (d_A and d_B).

Introducing an environmental policy in a net exporting country often reduces production and trade. As a result, importing countries produce more or find other sources of purchase. Environmental damages are reduced in the exporting country and may increase in the importing country or in the alternative supplier depending on changes in trade positions and respective pollution intensities.

This simple representation helps identify four factors that can affect the presence and severity of environmental leakage and loss of competitiveness:

- The effect of policy intervention on production and costs determines the shift in the supply curve, and indirectly impacts on the environment.
- The price responsiveness of demand in the regulating country and the rate of substitution between domestic and foreign products affect the magnitude of changes in net trade position and therefore affect the degree of potential leakage.[6]
- The pollution intensity of importing and exporting countries significantly affect the degree of leakage and net reduction of environmental damages.
- International market conditions will influence the overall changes in world prices, which may amplify the supply shifts.

The above four factors may affect the presence and magnitude of potential environmental and trade-related challenges, as explained in Box 2. Two factors depend on existing and new environmental policies and two are related to markets and the status of commodities most affected by these policies.[7]

This chain of expected effects is presented irrespective of other policies, however the presence of trade distorting policies in concerned countries could impact the magnitude of these factors, by affecting the effect of the new environmental policy on production and costs, demand responsiveness or international market conditions. In particular, the presence of policies limiting imports in the regulating country would likely limit pollution leakages. In contrast, policies enhancing exports in non-regulating countries may increase the leakage effect.

Box 2. Different factors affect the magnitude of environmental and trade-related challenges

The likelihood and magnitude of environmental and trade-related challenges when applying domestic environmental policies on certain agricultural goods depend on many factors. These include: the effect of policy intervention on production costs, the pollution intensity of importing and exporting countries, the thickness of the international commodity market and demand factors such as the price elasticity of demand in the regulating country and the Armington elasticity between domestic and foreign goods.

The effect of policy intervention on domestic production costs determines an upward shift of the domestic supply curve. Depending on the other factors listed below, this could cause an increase in domestic prices (competitiveness loss) and an increase in imports from a country which has higher pollution intensity (environmental leakage).

Demand factors include:

- *The price elasticity of demand for a good* is the ratio of the percentage change in quantity demanded of the good to the percentage change in its price, and measures how much consumers are sensitive to price changes of the good. The higher is the price elasticity of demand for a good, the more sensitive is the consumers' demand for it. When an environmental policy is implemented in an importing country, producers face higher production costs, and their supply curve moves upward. The price of the good increases and consumers with high elasticity of demand for the good will reduce their demand for it. The difference between supply and demand may remain the same in the importing country and therefore there may not be to substantive pollution leakage.

[6] Indeed, in the case of a good with relatively elastic demand, a change in price will result in decline in demand, and therefore the difference between supply and demand may remain the same.
[7] Table A A.2 and Table A A.3 in the Annex provide information on own-price demand elasticities and on the importance of trade for selected commodities and countries.

However, the equilibrium quantity sold in the importing country could now be lower, causing a certain competitiveness loss of domestic producers.

- The *Armington elasticity* is the rate of substitution between domestic and foreign goods. When the Armington elasticity is high domestic consumers would consider the domestic and foreign goods as substitutes. Therefore, when an environmental policy is implemented in an importing country, ceteris paribus, the higher is the Armington elasticity the higher might be both competitiveness loss and pollution leakage.

The pollution intensity of importing and exporting countries influences the magnitude of the pollution leakage. Ceteris paribus, the higher is the pollution intensity in the exporting country, the higher will be the pollution leakage and the lower will be the net overall environmental benefit of a domestic policy.

An international commodity market is considered thin when it represents only a small proportion of global production. Thin markets are often subject to large swings in traded volumes, since relatively small changes in production in an important producing country may result in large increases in exports or imports, should that country resort to the international market to dispose of a sudden increase in domestic supplies or to cover a shortfall. Given the high fluctuations in thin markets, such as rice, they might be exposed to higher associated risks.

3. Addressing environmental objectives directly: Environmental and trade-related implications and resolutions

This section analyses the first proposed route to improve environmental performance while facilitating the functioning of international markets. Direct policy instruments are used to mitigate GHG emissions or to limit the environmental impacts of pesticides. These are market-based instruments and regulations that aim to tackle directly their targeted environmental objective. Section 3.1 reviews these instruments and their potential economic and environmental challenges in the cases of GHG mitigation and policies to limit the environmental impact of pesticides. Section 3.2 investigates the merit of complementary policy actions that can attenuate the competitiveness and pollution leakage challenges that direct instruments can induce.

3.1. Direct policy instruments

Given that different environmental policy objectives require specific policy instruments and affect different agricultural commodities (e.g. crops or livestock), it is important to consider the cases of specific environmental concerns to get a better sense of the degree of environmental and trade-related challenges. The two case studies analysed in this report, climate mitigation policies and policies to limit the environmental effects of pesticides,[8] offer distinct examples. They involve the global and emerging issue of GHG leakage and the existing and increasingly important policies to reduce the environmental harm from use of pesticides. As the externalities of GHG emissions are geographically agnostic whereas the externalities of pesticides are more localised, there are different policy objectives related to each case study. The next two subsections discuss these cases separately. These two case studies highlight differences in the nature of the potential challenge and current policy developments as shown in Table 1.

OECD countries are setting increasingly ambitious climate mitigation targets, including in the agricultural sector. There is no internationally harmonised GHG mitigation policy; instead, countries are setting up nationally determined contributions (NDCs) under the Paris Agreement. As they start developing policies to curb GHG emissions in agriculture, there are concerns that GHG (carbon) leakage may undermine unilateral and therefore global mitigation efforts.

In contrast, discussions around pesticides invoke trade-offs between input use and potential risks that vary across countries. Pesticides are key inputs into crop production decisions in many regions and pesticide

[8] The term pesticide covers insecticides, fungicides, herbicides, rodenticides, molluscicides, or nematicides.

regulations have been in place for years in many countries to limit their environmental and health risks. There are, however, major differences in the use of substances and their regulations across countries and jurisdictions. While there are international standards on maximum residue limits of certain pesticide substances, as agreed at the FAO-WHO Codex Alimentarius, countries apply different levels of regulation based on their own situation and risk management processes (see Box 3 and Box 4 in this section). The anticipated adoption of stricter regulation on pesticides in some countries could further induce potential competitiveness and leakage challenges.

The remainder of this sub-section is organised as follows. Section 3.1.1 analyses the case of GHG mitigation policies, focussing in particular on direct policy instruments such as GHG pricing or environmental regulations. Section 3.1.2 analyses direct policy measures to limit pesticides' environmental externalities such as authorisations measures, regulations, and economic incentives.

Table 1. Comparing the two case studies

	Mitigation of GHG emissions in agriculture	Limiting environmental impact of pesticides
Environmental challenge	Climate change	Biodiversity, air, soil, and water quality
Scope of the challenge	Global	Local, regional, and indirectly global
Agriculture activities concerned	Livestock, dairy, rice, the use of N fertilisers and fossil fuel based energy, and indirectly all activities affecting land use change	Field crops, fruits and vegetables in uncontrolled environments
Link with agriculture production	Related to the use of inputs (land, energy, fertilisers), and the production process (cows, paddy rice).	Related to the use of inputs. Pesticides are key inputs to control damages for pest and disease loss
Objective of policies	Reducing total GHG emissions (GHG intensity and associated production volume)	Limiting environmental and health damages associated with pesticide use (toxicity and exposure of used substances)
Presence of international standard or target	No- differentiated national objectives under the Paris Agreement	Codex Alimentarius offers harmonised maximum residue limits for many substances.
Phase in policy development in the sector in OECD countries	Emerging and growing policy efforts to address broader mitigation targets	Policies applied in a number of countries for decades with more ambitious targets

3.1.1. The case of GHG emission mitigation

Problem definition and summary of the findings

Agricultural, forestry and other land use (AFOLU) activities are important sources of GHG emissions globally and are expected to make up a larger share of global GHG emissions over time as emissions from other sectors diminish (OECD, 2019[17]; OECD, 2022[18]). During the period of 2007–16, emissions in the AFOLU sector accounted for an estimated 22% of global GHG emissions. Specifically, AFOLU accounts for about 13% of carbon dioxide (CO_2) emissions, 44% of methane (CH_4) emissions, and 81% of nitrous oxide (N_2O) emissions.[9] Agriculture accounts for half of non-CO_2 AFOLU emissions (Intergovernmental Panel on Climate Change, 2019[19]). Different agricultural activities generate specific GHG emissions.

- Livestock is the largest source of methane (CH_4) emissions, followed by rice cultivation. The expansion of livestock production and paddy rice cultivation is the main factor explaining the observed 5.34% increase in global agricultural methane emissions during 2008–17 (Saunois et al., 2020[20]).
- Nitrous oxide (N_2O) emission is associated with the use of nitrogen fertilisers and livestock manure. N_2O emissions have increased by 30% between 1980 and 2017 and agriculture currently accounts for two-thirds of these emissions (Tian et al., 2020[21]).
- Carbon dioxide (CO_2) stems mainly from land use change activities (including deforestation) and fuel use for agricultural activities such as machinery.

[9] Methane and nitrous oxides are the two most important contributors to the non-CO_2 GHG. In 2019 carbon dioxide accounted for 64% of global GHG emissions, followed by methane (18%) and nitrous oxide (4%) (Intergovernmental Panel on Climate Change, 2022[309]).

Agricultural GHG emissions vary significantly by country, including among OECD countries, due mostly to differences in agricultural production activities (OECD, 2021[22]). These differences further translate into different GHG (carbon) intensities when accounting for productivity differences (Table A A.1).

Many governments have introduced or are introducing policy measures to mitigate agricultural related non-CO_2 GHG emissions (OECD, 2022[18]). At least 80% of the signatories of the Paris Agreement include agriculture in their National Determined Contributions (NDCs) (Richards et al., 2015[23]), though a limited number of governments have set AFOLU or agriculture specific GHG reduction targets (Henderson, Frezal and Flynn, 2020[6]).

While these developments suggest that agriculture is bound to play an important role in climate change solutions, widely divergent targets lead to concerns about the effectiveness of overall GHG reduction efforts due to potential GHG leakage (OECD, 2020[9]). In addition, negative competitive effects could raise concerns on the viability of stringent mitigation policy approaches for the sector.

Table 1 shows the summary of the findings with regards to direct policy, noting that other policies will be reviewed in Section 4.[10] The analysis of GHG pricing policies combined with market factors suggests that GHG pricing policies are likely to result in some GHG leakage and loss of competitiveness. The degree of this effect depends on the number of actors, their trade positions, and existing mitigation practices. The impact on competitiveness will likely vary among subsectors and types of farmers. Livestock and dairy producers have relatively higher amounts of GHG emissions and are generally less flexible than crop producers. Environmental regulations can directly (land use) or indirectly (pollution) affect GHG emissions and could have an impact on competitiveness, depending on the cost of implementation, and GHG leakage.

Table 2. Potential environmental and trade-related challenges: The case of GHG mitigation

<table>
<tr><th>Introduced policy</th><th>Potential domestic production impacts</th><th>Responsiveness of domestic demand</th><th>Pollution intensity in domestic and foreign markets</th><th>International market conditions</th><th>Potential competitiveness loss</th><th>Potential pollution leakage</th></tr>
<tr><td>Carbon pricing</td><td>High</td><td rowspan="2">Cereals and oilseeds: Low

Dairy and livestock: Medium to high</td><td rowspan="2">Generally higher GHG intensity in low- and middle-income countries

Higher average GHG intensity in livestock producing countries</td><td rowspan="2">Largely traded commodities like staple crops or animal products are less likely to change prices than others.

Concentration of imports or exports can increase price sensitivity</td><td>High for rice or beef especially if imposed at sufficiently high level</td><td>High especially for beef or dairy products introduced in low GHG intensity importers trading with higher GHG intensity producers</td></tr>
<tr><td>Environmental regulations</td><td>Medium</td><td colspan="2">Low to high, depending on impact on production</td></tr>
</table>

Note: Impacts are shown in the short to medium run.

Some of the key commodities associated with GHG emissions in agriculture are livestock, dairy, and rice. The GHG intensity of agriculture production is expected to be larger for countries producing mainly these commodities.[11] The ratio of pollution intensity by exporters and importers will influence the amplitude of leakage rates, so introducing a domestic GHG mitigation measure in a low GHG intensity country that

[10] In particular, instruments that pay farmers to reduce emissions, whether practice or outcome-based, are much less likely to affect competitiveness and are less likely to induce GHG leakage (Section 4.1.1). Investments in research and development or transfer of knowledge could increase productivity and competitiveness while limiting emissions domestically and abroad (Section 4.1.3).

[11] This could be the case in particular for animal production in emerging economies that tend to have larger GHG intensity.

imports meat or dairy products from a higher GHG intensity country would likely lead to more leakage than in other situations.

Livestock and dairy activities have relatively higher price elasticities of demand (Annex Table A A.2) and are relatively less traded than cereals compared to other agricultural goods (Annex Table A A.3). The environmental and trade-related impacts of introducing mitigation policies on livestock and dairy activities might therefore be tempered by their relative high price elasticity and low international market responsiveness. In contrast, rice, which is the most GHG intensive cereal, has a relatively low demand elasticity and thin international market that might expose it to price fluctuations and therefore potential competitiveness losses.

The remainder of this subsection focuses on direct policy instruments used to mitigate GHG, i.e. pricing mechanisms and environmental regulations, and reviews their effects.

GHG mitigation policy targets and instruments

Existing policy instruments mitigating GHG emissions from AFOLU production activities fall into four categories (Henderson, Frezal and Flynn, 2020[6]) (Annex Table A A.5) market-based instruments, environmental regulations, supporting policy instruments that offer financial support for AFOLU activities mitigating GHG emissions and R&D and information approaches. This section focuses on the analysis of direct policy instruments, i.e. GHG pricing mechanisms and environmental regulations, which are the ones that are most likely to result in some GHG leakage and loss of competitiveness.

Market based instruments have been adopted in many countries in the energy or industry sectors but remain limited in the AFOLU sector (ICAP, 2021[24]; European Court of Auditors, 2021[25]). Polluter-pays approaches essentially price emissions (taxes) or restrict the quantity of emission (e.g. emission trading systems). These two approaches have been shown to be the most cost-effective policy instruments to reduce GHG emissions (Arvanitopoulos, Garsous and Agnolucci, 2021[13]). Fuel-related agricultural CO_2 emissions are priced in some countries (Annex Figure A A.1). However, these emissions account for only 4% of agricultural GHG emissions. In contrast, non-CO_2 GHG from agriculture have been generally exempted from existing pricing or ETS schemes. At the time of writing, there was no GHG tax on agricultural non-CO_2 GHG. That said, Denmark has been discussing the possibility of introducing a GHG tax that could include agriculture, while New Zealand plans to introduce a GHG price for individual livestock operator and fertiliser companies (Danish Economic Councils, 2020[26]; Henderson, Frezal and Flynn, 2020[6]).

At the same time, pricing on GHG may lead to higher production costs and thereby competitiveness losses for producers.[12] Additionally, if only few countries impose a price on GHG, producers in non-implementing exporting countries could produce more and emit even more GHG (i.e. inducing GHG leakage).

Environmental regulations can play a significant complementary role in GHG mitigation. Deforestation regulations protect specific zones, preventing GHG emissions and preserving carbon sinks and ecosystems in these areas. Existing regulations, while potentially effective, face enforcement challenges which threatens their effectiveness (Henderson, Frezal and Flynn, 2020[6]). Regulations on pollution may also contribute to GHG mitigation especially if they relate to GHG emitting activities, such as limitation of nitrogen runoff from fertiliser uses. The EU's Nitrates Directive (ND) is an example of such regulation, whereby farmers comply with discharge limits in identified Nutrient Vulnerable Zones (NVZs). While it has likely helped to reduce overall nitrate pollution in Europe, implementation has been difficult in some regions of the European Union (Gruère, Ashley and Cadilhon, 2018[27]).

Regulations could be effective policy measures to limit the impact of the AFOLU sectors on land, water, and pollution. Where these also address GHG inducing activities, they will help mitigate GHG emissions. However, these measures can increase costs (if related to inputs) and therefore GHG leakage (Henders and Ostwald, 2014[28]). These consequences can be prevented by harmonising regulations across countries.

[12] Higher prices they generate can also have negative impacts on food security (Henderson et al., 2021[82]; OECD, 2019[29]).

Potential environmental and trade-related challenges associated with direct climate mitigation policies

There are a limited number of *ex ante* studies estimating GHG leakage, either for agriculture or overall GHG emissions (Annex Table A A.7). These studies typically take advantage of Computable General Equilibrium (CGE) models' ability to capture changes in bilateral trade flows to estimate the impacts of introducing different GHG pricing schemes.[13] In particular, OECD (2019[29]) and Henderson and Verma (2021[30]) used the MAGNET model to estimate impacts of a GHG tax on non-CO_2 GHG from the agricultural sector under different policy scenarios. These analyses were based on a GHG tax rate of USD 40 per tonne emitted in 2020–30, USD 60 in 2030–40, and USD 100 in the 2040–50 period. There are several points to note from the results of these exercises:

- The estimates suggest that introducing a GHG tax substantially reduces domestic GHG emissions, but there is some leakage to non-taxed regions depending on the geographical coverage of the tax. When the tax coverage includes only OECD countries, for example, the estimated leakage rate, defined as a ratio of GHG increased in non-taxed region to the GHG decreased in taxed region, is 31–34% (OECD, 2019[29]; Henderson and Verma, 2021[30]).
- The availability of abatement technologies largely determines the degree of leakage. The leakage rate doubles when there is no access to abatement technologies (Henderson and Verma, 2021[30]).
- Taxation on ruminants and fertilisers are less effective than a GHG tax partly because they do not cover all emissions (OECD, 2019[29]). Comprehensive pricing schemes are the most-cost effective policy instrument (OECD, 2019[31]).
- Agricultural output in taxed region declines but the overall production impact is limited in most scenarios (Henderson and Verma, 2021[30]).

Two other *ex ante* assessments of the potential impacts of key environmental policy targets for agriculture in the EU Farm-to-Fork and Biodiversity strategies consider its effect on GHG leakage (Barreiro-Hurle et al., 2021[32]; Henning and Witzke, 2021[33]).[14] The results of these modelling analyses suggest that the proposed policy targets could greatly reduce GHG emissions while also resulting in significant loss of competitiveness and GHG leakage. Barreiro-Hurle et al found a 15–17% non-CO_2 and 20–28% CO_2 emission reduction (2021[32]) and an estimated leakage rate ranging from 51–61% (non-CO_2 GHG) and 50% (GHG) if the European Union unilaterally commits to mitigation efforts. These high rates are mainly because of the strong leakage effects within animal production (Henning and Witzke, 2021[33]). The expected output loss is relatively high, e.g. 13–21% for cereals and 12–20% for oilseeds. Moreover, net imports could increase by 130% for cereals and 102% for beef according to Henning and Witzke (2021[33]).[15]

One study looks at a particular country. Denmark has an overall GHG mitigation target of 70% by 2030 compared with 1990. The Danish Economic Council (2020[26]) estimated the economic impact of imposing a GHG tax along with outcome-based abatement subsidies in Denmark and found significant welfare loss for the agricultural sector. They recommend exempting the sector from the comprehensive GHG taxation schemes, suggesting that abatement subsidies are more cost-effective.

[13] For example, the Modular Applied General Equilibrium Tool (MAGNET) model can take intra- and inter-sector relations into account to compute several indicators including GHG leakage and output loss (Arvanitopoulos, Garsous and Agnolucci, 2021[13]).

[14] Barreiro-Hurle et al. (2021[32]) model a 50% and 20% reduction in use of pesticide and fertiliser, respectively; the fact that at least 25% of the European Union's agricultural land to be under organic farming with 10% of land under high-diversity landscape; and a 50% reduction of nitrogen-balance surplus. Henning and Witzke (2021[33]) also consider additional scenarios including an incorporation of the agricultural sector to the EU-ETS, whereas Barreiro-Hurle et al. (2021[32]) study the impacts of agricultural support on sustainable practice under 2014-2020 CAP and post 2020 CAP scheme. It should be noted that the report's findings are established based off findings from the Common Agriculture Regionalised Impact Model, which has a number of limitations in its representation of the supply chain and consumption habits. As such, the study's estimates of leakage and competitiveness effects are merely rough indications of potential policy outcomes.

[15] Modelling assumptions are discussed and compared in European Commission (2021[307]).

While they do not capture international spillover effects, farm level models can also illustrate the extent of the production effects of different instruments. Some agricultural activities are more GHG intensive than others and these differences translate into heterogeneous farm impacts. OECD (2019[31]) estimated the impact of a GHG tax of between EUR 9 and EUR 50 per tonne on four types of mixed dairy and crop farms: farm A – high milk and low crop yield, farm B – low milk and low crop yield, farm C – low milk and high crop yield, and farm D – high milk and high crop yield (determined based on an EU farm level database). The study found substantial differences in impact (Annex Table A A.6). Profit is affected more for larger farmers with lower crop productivity (farms A and B) because they cannot effectively shift from dairy to crop production.

Ex post studies assessing the agricultural GHG leakage and competitiveness concerns are rare as the sector is usually exempted from pricing schemes. The 2008–12 CAD 10–30 per tonne carbon tax on fossil fuels in British Columbia offered a quasi-natural-experiment of a carbon-pricing policy. Rivers and Schaufele (2014[34]) found that the carbon tax had negligible impact on agricultural trade flows mainly because fossil fuels represented a small fraction of the total agricultural production costs. On the other hand, Olale et al. (2019[35]) investigated the farm-level income impact of the same policy and found modest but significant effects on producers' income. These studies suggest that the British Columbia's GHG tax on fossil fuels did not severely affect producers' competitiveness and led to only limited GHG leakage. Therefore, compensating those affected via lump sum or output-based rebates, as proposed by Rivers and Schaufele (2014[34]) and Olale et al. (2019[35]) would minimise any adverse impact.

3.1.2. The case of direct policies to limit the environmental impacts of inappropriate or excessive pesticide use

Problem definition and summary of the findings

Pesticides are essential inputs for crop production that limit damage from weeds and pests and maintain relatively stable crop yields. Pesticides help limit food supply losses, extend the shelf life of produce, inhibit microbial contamination, reduce foodborne illnesses, reduce the labour and fuel use needed for weeding, and limit soil disturbances (Sud, 2020[36]; Dobhal S, 2014[37]).

However, inappropriate or excessive use of specific pesticide substances can result in environmental damage and affect the health of exposed individuals. There are three main types of observed environmental impacts (Annex Table A A.8). First, inappropriate or excessive use of specific pesticide substances can reduce biodiversity and ecosystem services (Sud, 2020[36]; IPBES, 2016[38]).[16] Second, such use of pesticides can significantly contribute to soil and water pollution. Third, the inappropriate, excessive or continued use of the same substances can generate resistant pests and weeds, requiring additional applications of the same or potentially more environmentally harmful pesticides. Inappropriate or excessive use of pesticides can also have some risks for human health. For instance, exposure to certain pesticide substances is associated with certain types of cancers, cognitive and neurodevelopmental disorders, reproductive and endocrine disruptions (Trasande et al., 2015[39]). Poor pesticide management (e.g. excessive or inappropriate application) and remaining residue due to an early harvest can cause food contamination.[17] Finally, unintended acute pesticide poisoning (UAPP) is a worldwide concern (Handford, Elliott and Campbell, 2015[40]).[18] Compliance with appropriate regulations and good agronomic practices or technologies can help reduce pesticide risks without sacrificing productivity and farmers' income (Sud, 2020[36]; OECD, 2016[41]). Research and development play a significant role in this respect.

[16] Pesticide exposure has been associated with population loss in several key species of birds, bees, and amphibians (Green et al., 2005[303]; Sud, 2020[36]). Pimentel (2005[305]) estimated that the pesticide caused loss of birds, a beneficial pest predator on farmlands, is equivalent to USD 2.2 billion annual economic losses in the United States alone. Neonicotinoids, a class of widely used insecticides, are among the pesticides that have been linked to this issue.

[17] Though limited research suggests dietary exposure to pesticides may adversely affect human health (such as through the development of metabolic diseases) (Lukowicz et al., 2018[308]), it is quite rare for consumers to be exposed to residue levels high enough to cause health complications (Chen et al., 2011[302]; Mesnage et al., 2020[319]; Wendie L. Claeys, 2011[320]).

[18] A systematic review suggests that worldwide UAPP cases total about 385 million, leading to 11 000 annual deaths, often as a result of incorrect pesticide use or suicide (Boedeker et al., 2020[291]).

Pesticide use depends on several factors such as economic conditions, climatic conditions (Olfert and Weiss, 2006[42]) and crop composition (Popp, Pető and Nagy, 2012[43]). Pesticide use per hectare has increased significantly since 1990 in the Asian and American continents, remained flatter in European and Oceanian countries, and is stable at a low base in the African continent.[19] Empirical evidence suggests that pesticides are often overused and can be reduced without affecting productivity (OECD, 2021[44]; Lechenet et al., 2017[45]).[20] This highlights the potential to improve pesticide management practices to minimise the adverse impact of pesticide applications.

While reflecting different contexts and approaches, differing pesticide regulations across countries may lead to pollution leakage and loss of competitiveness. Governments have adopted a variety of policy measures to approve and regulate the use of specific pesticide substances based on different risk management approaches. Moreover, risk profiles of pesticides vary which necessitates a range of regulation on differing pesticides depending on evaluated risks and perceived trade-offs.

Just like in the case of GHG mitigation policies, however, reviewed evidence suggests that not all policies will have the same effect, as shown in Table 3. Governments should consider the most appropriate set of policies that best address each country's context and specificities. More precisely, reviewed evidence in this section and market factors suggests that:

- Banning specific pesticide substance can lead to significant loss in domestic producers' competitiveness, but the severity of the loss will depend on alternative pest management practices and technologies.[21] In addition, the literature implies a possibility of pollution leakage after banning specific pesticides under certain market conditions.
- When applied at a sufficiently high level and on a risk differentiated basis, pesticide taxes can be effective policy measures to limit pesticide risks. Depending on the range of pesticides available, they can have relatively limited impacts on competitiveness and thus pollution leakage, given that pesticides are still overused in many regions.

Unlike in the case of GHG mitigation, most countries have already introduced policies, so a change in approach may not be as significant on production as for some of the most important policies to reduce GHG emissions from agriculture.

Pesticide pollution intensities are not easily available. Pesticide use per crop and country are poor predictors of potential risks, as a farm spraying large volume of low toxicity herbicide may induce much less pollution by area of land or product than one using limited volumes of recognised highly toxic pesticides. Ghimire and Woodward (2013[46]) study under and over use of pesticides compared to agronomic requirements by countries and find that low income countries tend to under-use pesticides, middle income and top income countries over- use pesticides. Yet this broad characterisation does not capture risks and would not explain all differences, given that agriculture intensity also depends on land, type of production etc.

Pesticides are used on most crops and produce, but they are particularly important for some crops (e.g. cotton) and for some fruits and vegetable produce (e.g. strawberry and tomato). While price elasticities of demand vary across agricultural products, produce will face generally higher price elasticity

[19] More specifically, OECD countries and emerging economies have followed different trends in pesticide use (Annex Table A A.9). Pesticide use per hectare increased rapidly from the 1990s to the last decade especially in some emerging economies. In contrast, most OECD countries had more limited evolution (FAOSTAT, 2021[292]). These trends however do not necessarily indicate the change in the risks arising from pesticide use as the types of chemicals used changes over time.

[20] For instance, in the People's Republic of China (hereafter "China"), Zhang et al. (2015[304]) found that 57% and 64% of pesticide applications on rice and cotton production, respectively, are excessive and do not affect productivity. On many Chinese farms, pesticides are used as insurance by producers that are not full time farmers and do not monitor pests (OECD, 2018[318]). Skevas et al. (2014[77]) report that 100% of Dutch producers overused herbicides, 86% overused fungicides, and 67% overused insecticides from a profit maximisation perspective. A study in France found that 59% of farmers could reduce pesticide on average by 42% while maintaining their productivities (Lechenet et al., 2017[45]), while another study suggests that France could achieve an overall 30% pesticide reduction without sacrificing farmers' income (Jacquet, Butault and Guichard, 2011[301]).

[21] ESPR (2021[306]) offers a quantitative assessment of the alternative practices in EU agriculture.

of demand than cereals or oilseeds (Annex Table A A.2). This implies that cereals may be more exposed to price changes induced by a pesticide regulation, especially in relatively thin international markets like that of rice (Calpe, 2006[47]). A pesticide regulation on rice might therefore be more likely to reduce competitiveness than one on produce.

Table 3. Potential environmental and trade-related challenges: The case of pesticides

Introduced policy	Potential production impacts	Responsiveness of domestic demand	Pollution intensity in domestic and foreign markets	International market conditions	Potential competitiveness loss	Potential pollution leakage
Ban on key substance	High	Cereals and oilseeds: Low Fruits and vegetables: Low to medium	Likely low for low income countries, varying for others	Largely traded commodity less responsive to price changes than others	High especially for substance used on less traded staple crops like rice or some produce	Medium depending on access to pesticides in exporters (especially if introduced in large importer with low pollution intensity)
Pesticide use regulations	Low to medium				Limited	Limited
Pesticide Tax	Low to medium				Medium if tax is sufficiently high and covers critical pesticides	Low to medium leakage depending on existing pesticide use

Note: Impacts are shown in the short to medium run.

The following subsection reviews direct policy measures in the case of pesticides and analyses existing evidence on their potential environmental and trade-related effects.

Direct policy measures to limit pesticides' environmental externalities

Several governments have announced plans to increase the stringency of their regulatory instruments related to pesticides in order to reduce adverse environmental and health impacts. For example, the EU's Farm-to-Fork strategy aims to achieve a 50% reduction in pesticide use and risks by 2030 (European Commission, 2020[4]). Japan's Strategy for Sustainable Food System establishes a similar reduction target of 50% in the risk-weighted use of pesticides by 2050 through the implementation of integrated pest management practices (Ministry of Agriculture, Fisheries and Forestries of Japan, 2020[5]).

Governments have employed various instruments to limit the potential environmental or health impacts of pesticides (Böcker and Finger, 2016[48]; Möhring et al., 2020[49]). Some involve qualitative regulations, which authorise or deny the use of specific substances, others are quantitative regulations, aiming to reduce pesticide use. More specifically, policy instruments can be categorised into five groups: authorisation measures, pesticide use regulations, economic incentive measures, informative measures, and food regulations (Annex Table A A.9). The following subsections discuss the effectiveness and modalities of the three direct policy instruments mentioned before (authorisation measures, pesticide use regulations, economic incentive measures) and thereby their likelihood to cause environmental and trade-related challenges, in the form of pollution leakage and loss of competitiveness. Informative measures and food regulations will be discussed in Section 4.

Authorisation measures for pesticide substances are direct policy interventions constraining the type and amount of pesticide use. New pesticides are first registered upon passing approval stages. Approved pesticides can then be restricted or banned typically due to the identification of new risks. These decisions have a critical effect on the use of pesticides on farm, as banning of a previously approved substance can deter productivity in the short to medium term, in the absence of cost-effective alternatives.

There are several possible statuses for pesticides in a given country: approved, never approved, once approved but later banned, and never approved and officially banned. As governments have different registration processes and criteria to manage potential risks there is a wide variance in approval status across countries. In general, developed countries have more stringent registration processes than developing countries (OECD, 2020[50]).

Differences in approval decisions also exist among OECD countries. For example, several neonicotinoids-related substances which have been associated with risks for invertebrates, continue to be used as insecticides in some European countries which allow its use on seed dressings for emergency purposes (European Commission, 2021[51]). France forbade the use of all neonicotinoids-family pesticides in 2018 due to reports of its effects on honeybees. However, the measure was partially and temporarily lifted in response to requests by sugar beet producers who were about to lose their production in 2020 due to the widespread yellow virus (USDA FAS, 2020[52]). Regulations vary among different jurisdictions within states as well, such as in Canada, where provincial, territorial, and local governments have the authority to further regulate pesticide use at a stringency beyond what is determined at the federal level (Canada, 2019[53]).

Differences in approval status also depend on national risk management procedures.[22] In particular, while decisions to ban are typically justified based on new evidence on the presence of risks, pesticide bans on the same substances vary considerably across countries. These variances can be due to different use, land and environmental condition, and exposure potential. According to PAN (2021[54]), of the 434 main pesticide substances in 163 reported countries, as much as 144 substances (33%) are banned in only one country. On the other hand, some 20 toxic substances are banned in more than 100 countries. A quarter of active ingredients approved in the United States are banned in the European Union (Donley, 2019[55]).[23] The number of bans, however, does not necessarily indicate that a regulatory approach is more stringent. Bans may be due to different risk management approaches and may be more frequent in countries with more approved pesticide substances than others.

Different approval status and regulatory measures in pesticide use, while aimed at balancing risks and benefits according to local conditions, may create environmental and trade-related challenges. For instance, a ban of an important pesticide can lead to higher pest pressures in the implementing country and lead to more imported products made in a foreign country with banned pesticides. The environmental burden of pesticide use can be offshored to the exporting country (Lynch, Malcolm and Zilberman, 2005[56]).

Pesticide use regulations are on-farm restrictions concerning pesticide applications. They are applied in most OECD countries via the use of licences or permits. In particular, aerial spraying has been banned in some parts of the EU in response to growing concerns on health risks (regulations on application) (Vojtech, 2010[57]). In many OECD countries, pesticide uses near water sources are restricted (distance limits) (ibid). Moreover, producers are usually restricted in the storage of pesticides (e.g. place, quantity, duration).

Pesticide use regulations may provide effective means to reduce environmental or health risks. At the same time, their aggregate effects might be more limited on production than pesticide bans or qualitative regulations as they do not prevent the use of pesticide substances. As such they may be less likely to create environmental and trade-related challenges.

Well-designed pesticide taxes can be efficient instruments to achieve the environmental optimum level of pesticide use (Finger et al., 2017[58]; Böcker and Finger, 2016[48]). Only a few OECD countries have adopted pesticide taxes and the tax rates vary across countries (Annex Table A A.10).[24] Taxes in Denmark and Norway are relatively high, and they may have contributed to the reduction in impacts of pesticide use. The Danish tax was introduced in 1996, as a general *ad valorem* rate (Dasgupta, 2021[59]). Although *ex ante* studies suggested the tax would result in a high level of pesticide reduction (18-20%), it had a more limited impact. In 2013, the tax was changed to a differentiated rate tax with higher levies on substances with a higher pesticide load, with predicted greater effect (Dasgupta, 2021[59]). Pesticide load (PL) is calculated for three sub-indicators (human health, environmental fate, and ecotoxicity) and expressed as the PL per unit of commercial product) e.g. litre, kilogram, standard dose, capsule, or tablet). Tax revenues are refunded to the agricultural sector to compensate farmers' incurred loss due to the regulation, which has enhanced public acceptance (UNEP, 2020[60]). Subsequent evaluations show that

[22] While risk assessment processes may be similar across countries, risk management decisions that are taken at least in part related to the risk assessment may differ.

[23] Interpretation of differences in risk management will not be discussed in this paper.

[24] Several countries (e.g. Switzerland, the Netherlands, Germany) are considering using pesticide taxes (Böcker and Finger, 2016[48]; OECD, 2017[63]).

the differentiated pesticide tax has caused the average user to buy pesticides with a lower PL (lower scores on the three above mentioned sub-indicators) (Ørum, 2018[61]; Nielsen, 2020[62]).In France, a tax was first introduced in 1999 as a part of the tax on general polluting activities (TGAP) and replaced by a tax on diffuse agricultural pollution in 2008. Its relatively low rate has however limited the effectiveness in reducing pesticide use (OECD, 2017[63]).

The effectiveness of pesticide taxes depends on the responsiveness to tax rates. A meta-analysis by Böcker and Finger (2016[48]) estimated that the price elasticity of pesticide demand in the Europe and North America is about -0.28, explaining why low tax rates will be relatively ineffective. Finger et al. (2017[58]) still find a persistent impact of taxes on pesticide use in the long run and that differentiated taxes are more effective based on the experience in Denmark, France, Norway, and Sweden. Moreover, taxes are most effective when implemented in tandem with a range of policy instruments designed to reduce pesticide usage, such as tradable permits, direct environmental regulation, public financial support, payments for ecosystem services, information measures, and voluntary schemes (Section 4).

The fact that pesticide taxes need to be raised at sufficiently high rates suggest that effective taxes may increase production costs, with potential impact on targeted producers' competitiveness (in the absence of tax rebates or compensating payments). Imports from countries with less stringent policy approaches could then generate pollution leakages. These issues would be even more severe if trade partners offer input subsidies to encourage pesticide use in their agricultural production.

Potential environmental and trade-related challenges associated with direct policies to limit pesticides environmental impact

Several important dimensions determine the impacts of banning a pesticide, including the past use of pesticides; the availability of substitutes; price elasticity of demand; international competitiveness; and research and development efforts (Zilberman et al., 1991[64]). In particular, the availability of alternative pesticides, pest-management practices or technologies largely determine the potential impact on producers' output and revenue, and international impacts depend on whether it applies to domestic and imported products.

A few studies have estimated the potential domestic and international impacts of banning certain pesticides in specific regions (Annex Table A A.12). Lynch, Malcolm, and Zilberman (2005[56]) estimated the effects of banning Methyl Bromide (MB) in the United States. MB was widely used by strawberry farmers in California and tomato and pepper farmers in Florida. However, it was listed as a Class I ozone-depleting substance in the 1992 Montreal Protocol. The United States phased out MB in 2005 (with a critical use exception granted to existing stocks which expired in 2016), whereas Mexico, a trade competitor, did not agree with the phase out.[25] The study projected a substantial shift in production regions both between the United States and Mexico and within the United States. Strawberry fields in California were expected to decline by more than 40% and the US strawberry production would be largely substituted by Mexico. Carter et al. (2005[65]) projected a 6–17% sectoral revenue loss for strawberry producers in California after banning the MB. Regarding pollution leakage, Lynch, Malcolm, and Zilberman (2005[56]) suggested that most Mexican farmers would not use MB for additional production because the pesticide was too expensive to adopt. The leakage rate was estimated to be 11.5%, i.e. more than one-tenth of the MB reduced in the United States would be applied in Mexico. However, if the pesticide was 20% cheaper than the assumed price of USD 500 per acre, more Mexican farmers would use MB, and the leakage rate would reach 61%.

Other *ex ante* studies demonstrate the importance of substituting technologies to minimise the impact of phasing out certain pesticides. Garcia-German et al. (2014[66]) considered a ban of an important herbicide in Spain and assumed no alternative herbicide and hand weeding as a main weed management technology. Thus, farmers were predicted to lose substantial revenue with the phase out. Zilberman et al. (1991[64]) assessed a potential impact of phasing out several pesticides due to California's Proposition 128. They predicted a significant decline in output partly because 30% of the pesticides lacked alternatives (Zilberman et al., 1991[64]). The plan to move to 100% organic farming in Bhutan, which would, among

[25] California strawberry farmers have access to limited methyl bromide use consistent with the critical use exceptions provided for by the Montreal Protocol.

other things, forbid the use of pesticides was estimated to generate significant welfare losses and food insecurity (Feuerbacher et al., 2018[67]).

Research suggests that there is yield gap between organic and chemically intensive conventional agriculture, with organic production typically less productive (Ponisio et al., 2015[68]; Schrama et al., 2018[69]; Seufert, 2012[70]; Ponti, Rijk and Ittersum, 2012[71]). However, the size of this gap is highly contextual, depending on a range of factors (management practices, crop type, climatic conditions, etc.). Although there is greater research on organic yields in developed countries, research suggest the yield gap does not vary between less developed and more developed nations (Ponisio et al., 2015[68]).

Expected production effects of pesticide bans have reportedly affected regulatory proposals. In Switzerland, two initiatives, which proposed to ban the use of any chemical pesticide to improve water quality, were subject to a national referendum in June 2021. They were both rejected due in part to the concerns on the expected impacts of the proposed measures on domestic production and their potential leakage effects (Finger, 2021[72]; Illien, 2021[73]). Another example is from the United States where California's proposition 128, at the non-federal level, was rejected due in part to concerns related to production losses.

When export-destined crops are affected by a pesticide ban, the potential loss in market share depends on the demand characteristics in importing countries. In Garcia-German et al. (2014[66]), for instance, the estimated impact of a pesticide ban on lettuce export from Spain was estimated to be –7.24% to the United Kingdom and –15.84% to Germany when part of the resulting increased cost of production was passed to the final price. These differences are related to the availability of alternative lettuce suppliers and consumer preferences.

Competitiveness and leakage concerns can also emerge when countries are allowed to export pesticides banned for domestic use to countries with less stringent pesticide policies (Fuchs, Brown and Rounsevell, 2020[7]). By encouraging the use of banned pesticides abroad, they encourage leakage, and if imports are not subject to the same requirement, this can lower competitiveness. European countries have recently sought to reduce such inconsistencies. In 2020, the EU Commission acknowledged its international commitment under its Chemicals Strategy for Sustainability to ensure that hazardous chemicals banned in the European Union, including pesticides, are not produced for export, and stated that various options are being considered, including a revision of the relevant legislation (Watson, 2020[74]). In 2018, France officially banned the production, distribution, and export of pesticides that include substances not approved in the European Union. The German agricultural ministry announced a similar ban on the export of plant protection products banned in the EU as well (Dahm, 2022[75]).

Pesticide use regulations and taxes are the main direct approaches to regulate the quantity of farm-level pesticide use. Although pesticide taxes can be cost-effective policy tools to mitigate the pesticide related risks, many governments thus far do not adopt the policy measures in part due to competitiveness concerns (UNEP, 2020[60]).

Some *ex ante* studies assess the impact of pesticide taxes. Chen, McIntosh, and Epperson (1994[76]) estimated the potential impact of imposing pesticide taxes in Alabama on production and profit of local producers. They found modest impact: 1% tax on pesticide would lead to 2.12–2.71% decrease in pesticide use but only incur loss of 0.24–0.55% (field crop output), 0.12–0.19% (vegetable and fruits output), and 0.075% (profit). Similarly pesticide taxes with higher rates (e.g. 30–50%) seem to have limited impact on production and income perhaps because of farmers' overuse of pesticides as observed in specific European countries (Lechenet et al., 2017[45]; Skevas, Stefanou and Oude Lansink, 2014[77]). Compensating farmers subject to pesticide taxes, as done in Denmark, would minimise income losses. When the concerns on competitiveness loss is modest, pollution leakage would not be a significant policy challenge as noted above.

However, environmental and trade-related challenges could be more substantial under quantitative pesticide restrictions such as that envisioned in the EU's Farm to Fork strategy. Barreiro-Hurle et al. (2021[32]) estimate that the policy packages of the European Green Deal and post 2020 CAP could be associated with significant losses. It considers the impact of implementing environmental regulations to meet the several policy goals including a 50% and 20% reduction in use of pesticides and fertilisers, respectively, and an expansion of organic farmland to account for a quarter of the European Union's agricultural land. These policy interventions would lead to significant decline in production (e.g. 13–15% production loss of cereals and 12–15% loss of oilseeds) and variations in prices at local markets. Beckman et al. (2020[78]) estimations of the policy effects of the Food to Fork strategy came to similar conclusions. If the proposed input reductions were met, the report estimated a decline in EU agricultural production of around 7-10%. It also estimated a reduction in trade with the worst impacts affecting the world's most food-insecure populations. The prevalence of global food insecurity, which is defined as the number of people who lack access to 2 100 daily calories, may also increase by 0.5%, or 22 million people worldwide.[26]

3.2. Complementary policy actions

The previous sub-section discussed the possibility and magnitude of competitiveness loss and pollution leakage induced by the adoption of direct environmental policies in agriculture such as carbon taxes or pesticides bans. This sub-section reviews possible complementary policy actions to limit these challenges.

Two types of approaches are analysed: multilateral and unilateral. Section 3.2.1 reviews the role of multilateral or regional approaches in limiting the potential environmental and trade-related effects of market-based policies or regulations. Section 3.2.2 discusses the potential application of selected unilateral approaches, including trade policy instruments, and their propensity to limit pollution leakage and competitiveness issues.

Findings from this section are shown in Table 4. While the previous sub-section analysed environmental and trade-related challenges in the context of GHG mitigation and pesticides reduction separately, this sub-section presents each policy separately and groups considerations for the two case studies, as most actions could be applied to both cases, and evidence is limited for each of the two case studies.

The evidence reviewed suggests that multilateral and regional approaches can be effective at enabling global solutions to global problems and reducing environmental and trade-related challenges, but they require time and effort, as well as voluntary mechanisms, and the buy-in of trading partners. The unilateral approaches reviewed are faster and easier to introduce, but their attenuating effects on pollution leakage and competitiveness depends on the design of the specific measure used. These measures also raise implementation challenges and, potentially, risks of trade disputes if not appropriately designed.

Table 4. Comparison of reviewed complementary policy actions

Complementary policy actions	Effectiveness in limiting pollution leakages and competitiveness losses	Economic and political viability and other considerations
	3.2.1 Multilateral and regional approaches	
International regulatory co-operation	Effectiveness depends on mechanism and participation. (H, M) Ex: climate coalition or mutual recognition agreement	Takes time and effort, potential exclusion of non-participants
Regional trade agreements	Effectiveness depends on the scope of provisions and their enforcement, can result in leakage in non-participants. (L,M)	Negotiated agreement mutually beneficial, limited to partners
Sectoral agreements	Effective only if applied by key supply chain actors. (L,L)	Not determined
	3.2.2 Unilateral approaches	
Carbon border adjustment mechanisms	Effectiveness depends on design. (H,L)	Potential implementation costs, risks of trade disputes
Mirror clauses	Limiting pollution leakage is not guaranteed if applied only to exported products. Moderate effectiveness on reducing competitiveness losses depending on participation and enforcement. (L,L)	Potential implementation costs, trade legality risks
Other mechanisms: exemption of allowance, VAT, MRL set to zero, import bans	Effectiveness depends on mechanism's impact on trade. (M,-)	Depends on mechanism: potential implementation costs, and trade legality risks

Note: Volume and Agreement of the reviewed literature determined within brackets (Volume, Agreement).
Volume: 0-5 articles = low (L); 5-10 = moderate (M); 10+ = high (H). Agreement determined by divergence in findings. Agreement is not determined for "other mechanisms," signified by (-).
Source: Authors based on reviewed literature.

[26] It should be noted that these studies did not evaluate all aspects of the plan or its potential benefits and costs (such as to the environment or human health) outside of its reductions of agricultural inputs.

3.2.1. Multilateral and regional policy approaches

The most obvious way to limit the international environmental and trade-related effects of heterogeneous domestic environmental policies is international co-operation mechanisms. Such mechanisms can either limit international differences, for instance through agreements around a common reference, or limit the international effects of domestic policies.

There are a number of means to do so, whether at the regional or multilateral level, that all belong to the broad umbrella of international regulatory co-operation mechanisms (IRCs). Two specific IRC mechanisms, the use of environmental provisions in regional trade agreements (RTAs) and sectoral agreements, have the potential to address the environmental and trade related challenges associated with direct policy instruments targeting GHG instruments and pesticide regulations.

International regulatory co-operation

In general, IRCs refer to any formal or informal agreement among countries which aim at promoting various forms of co-operation to develop, manage or enforce regulations. OECD (2013[79]) classified the most common IRC mechanisms at the global level, identifying 11 main types of IRCs, ranging from complete harmonisation to more flexible options, such as dialogue or information exchange (Table 5). Mechanisms vary in political feasibility and efficacy, especially depending on their legal enforceability. Given the limited available enforcement mechanisms in the environmental domain, effective environmental agreements sometimes rely upon "issue linkage" or the tying of environmental goals to policy tools in other domains. For example, environmental provisions in trade agreements can utilize binding measures which commit members to specific actions with environmental and trade-related remedies if not taken (Abman, Lundberg and Ruta, 2021[80]).

Several IRC mechanisms have been employed in the area of GHG mitigation, including for GHG emissions from agriculture. These are, among others, negotiated agreements, such as the UNFCCC (1992), the Kyoto Protocol (1997), the Paris Agreement (2015), or the Global Methane Pledge (2021), the harmonisation of rules through supranational institutions, such as the EU Green Deal (2019), but also formal and informal dialogue, for instance through the UNFCCC Conference of Parties (COP).

Any IRC mechanism that can help progress towards a more unified application of carbon tax or equivalent mechanisms helps limit environmental and trade-related challenges (Rosenbloom et al., 2020[81]). Indeed, a global carbon tax, including in agriculture, forestry and other land use sectors (AFOLU) would eliminate carbon leakage (Henderson et al., 2021[82]; OECD, 2019[29]). For instance, simulations of the agricultural sector suggest that the potential carbon leakage of a carbon tax could be halved if it was extended from the northern European Union to the OECD as a whole (Henderson and Verma, 2021[30]). Böhringer, Carbone and Rutherford (2012[83]) and Frank et al. (2019[84]) simulate the multilateral harmonisation of carbon taxes and also find that international co-operation agreements, via the re-distribution of tax revenues, can reduce competitiveness losses. More generally, Nachtigall et al. (2021[85]) show that different forms of international co-ordination on carbon pricing, such as linked carbon markets or forming climate coalitions, can deliver economic and environmental benefits including lowering the cost of GHG mitigation and reducing carbon leakage.

"Climate clubs" or "climate coalitions" are one type of IRC applied to GHG mitigation. There are different types of climate clubs, but they share five criteria: limited membership, climate co-operation, benefits for members, economic contribution from members, and the involvement of a monitoring mechanism (van Asselt, 2017[86]). Nordhaus (2021[87]) discusses the potential role of a dynamic climate club, which would limit free-riding by combining tariff penalties and rapid investments in technological change. Such mechanisms can reduce the risk of carbon leakage associated with carbon price differences, especially if the club expands and applies a harmonised carbon price (Nachtigall et al., 2021[85]). Unlike voluntary agreements, climate clubs can facilitate the application and enforcement of GHG mitigation requirements, for example through penalties or the establishment of a strong common vision (Hovi et al., 2016[88]). Nevertheless, by excluding other countries, for instance via tariffs, they may go against the international goals for climate and create other trade-offs on trade and food security. In addition, they may be at odds with the principle of common but differentiated responsibility and respective capabilities (CBDR-RC).

Table 5. Types of international regulatory co-operation (IRC) mechanisms

Typology of mechanism	Definition	Example
Integration or harmonisation through supranational or joint institutions	Harmonisation of laws is facilitated by the partial substitution of national regulatory competences with supra-national law making and institutions	-EU institutions and directives. -Joint Food Standards Australia and New Zealand (FSANZ)
Negotiated agreements (treaties/conventions)	Formal regulatory co-operation signed by states and binding with respect to international law.	-Montreal protocol -Paris Agreement
Regulatory partnerships between countries	Formal political co-operation agreements between countries, minimising regulatory divergences	-Trans-Tasman co-operation -Canada-United States Regulatory Cooperation Council (comprising agriculture)
Inter-governmental organisations (IOs)	Memberships to IOs (based on a treaty) promoting IRC	- FAO, ILO, OECD, WTO, etc.
Regional Agreements with regulatory provisions	Formal regional trade agreements comprising regulatory provisions and enhancing economic co-operation	Regional trade agreements (RTAs) as reported in the WTO database
Mutual recognition agreements (MRAs)	Principle of international law whereby States uphold legal decisions taken by the authorities of another State.	-EU technico harmonisation and standardisation. -Trans-Tasman Mutual Recognition Arrangement
Transgovernmental networks	Technical co-operation based on frequent interaction of officials (e.g. through structured dialogues, or MoUs)	- Asia-Pacific Economic Cooperation Network (APEC)
Formal requirements to consider IRC when developing regulations	Cross-sectoral and cross-government requirements imposed on responsible authorities to consider all relevant international standards and frameworks for co-operation in the same field	-COAG Best Practice Regulation
International standards recognition	Incorporation of international standards in legislative instruments	-Adoption of ISO standards - Transnational regulation of the private sector
Soft-law	Co-operation based on non-legally binding instruments.	-OECD Guidelines and Principles
Dialogue/Informal exchange of information	Conferences, forums and other settings where regulators and other stakeholders have an exchange on regulatory issues.	-Food Systems Summit -IUCN World Conservation Congress

Source: Adapted from OECD (2013[79]).

In the context of pesticides, regulatory heterogeneity can generate international transaction costs, limit market access and competition (OECD, 2013[79]). For instance, different regulatory systems can affect the price of agricultural inputs for farmers in different countries, resulting in potential competitiveness losses. Heterogeneous regulations defining organic products and those pertaining to the pesticide approvals can generate important transaction costs for farmers and other supply chain actors (OECD, 2013[79]).

Different types of IRCs can help limit regulatory heterogeneity and thereby reduce these transaction costs and market constraints, including in the case of pesticides:

- The recognition of international standards or mutual recognition mechanisms can reduce regulatory heterogeneity by promoting global regulatory convergence (OECD, 2013[79]).
- The transparent systematic application of standardised regulatory practices can alleviate environmental and trade-related challenges (von Lampe, Deconinck and Bastien, 2016[89]). A successful example of IRC applied to the pesticide sector is the FAO/WHO Codex Alimentarius, an international body whose mandate includes the development of harmonised international health-related maximum residue limits (MRLs) for many chemical substances including pesticides (FAO/WHO, 2022[90])(Box 3). These guidelines have been adopted by 188 countries and have been used as reference in WTO disputes. While they are established based on food safety assessments in accordance with good agricultural practices related, there are discussions as to whether the Codex Alimentarius (or a similar institution) could also develop harmonised environmental sustainability standards for agriculture and food (Box 4).
- Addressing pesticide management differences across borders also raises the potential to address environmental and trade-related challenges (OECD, 2013[79]). The Canada-US Regulatory

Cooperation Council provides a good example of how sectoral ministries and specialised agencies of neighbouring and trade partner countries can design IRC mechanisms, for example for common labelling procedures.

- Finally, enabling international co-operation through research and knowledge exchanges can also reduce potential pollution leakages due to the heterogeneous application of pesticide bans or regulations (FAO, 2020[91]). For instance, the OECD's Chemical Committee's work on risk assessment, which includes pesticides, helped establish a common language for risk assessment frameworks.

Box 3. Maximum residue limits (MRL)

Maximum residue limits are defined as "the highest level of a given pesticide's residue in a given crop that is legally tolerated in the government jurisdiction" (USITC, 2021[92]). The Codex Maximum Limit for pesticide residues (expressed in mg/kg) is the maximum concentration of pesticide that is recommended by the Codex Alimentarius Commission (after undergoing dietary risks assessments by the Joint FAO/WHO Meeting on Pesticide Residues) to be legally permitted or recognised as acceptable in or on a food or feed, based on good agricultural practice (GAP) (FAO/WHO, 2019[93]). The FAO defines good agricultural practice, in the use of pesticides, as "the officially recommended or nationally authorized uses of pesticides under actual conditions necessary for effective and reliable pest control" (FAO, 2003[94]). As GAPs are determined at the national level, countries frequently update MRLs not only in response to food safety risk assessments, but to support new uses of pesticides, facilitate trade, and to respond to public health, occupational health, and environmental safety concerns as well.

Under the WTO Agreement on the Application of Sanitary and Phytosanitary Measures (SPS Agreement), WTO Members can set MRLs at any level as long as the standards reflect scientific risk assessment decisions. That said, Members are encouraged to use internationally harmonised MRLs standards set up by the WHO/FAO Codex Alimentarius (Wilson and Otsuki, 2003[95]). While MRL standards are adopted worldwide to control pesticide residues through the use of labels, which are intended mainly to provide information for the enforcement of GAPs, their introduction also indirectly affects the use of pesticides as agricultural inputs.

As MRLs are applied to domestic and imported food products, harmonised standards can limit the impacts of MRLs on international agricultural trade (Handford, Elliott and Campbell, 2015[40]). The maximum residue limits (CXLs) defined by the Codex Alimentarius Commission act as an international reference for a wide range of pesticide substances. According to a WHO probabilistic dietary exposure assessment conducted in 8 countries, CXLs provided a high level of protection to children and adults for 36 of the 38 studied pesticides, including higher than 99% for 29 pesticides and above 90% the remaining pesticides (Crépet et al., 2021[96]).

Maximum residue limits (MRLs) and trade

The adoption of heterogeneous and lower MRLs in importing countries can potentially impose production costs domestically and on foreign suppliers. As such, stricter MRLs in importing countries are sometimes regarded as non-tariff trade barriers (Wilson and Otsuki, 2003[95]; Möhring et al., 2020[49]). The net effect on competitiveness and the environment depends on domestic and foreign production systems, their trade position (net importers or net exporters), and their required use of the targeted pesticide substances. Heterogeneous and low MRLs may also amplify the costs imposed on producers from changing climatic conditions, such as through increased pest abundance and range (USITC, 2021[92]; Delcour et al., 2014[97]). At the same time, environmental damage may decline if regulatory changes encourage a change in production practices (Matthews, 2022[98]).

Much of the literature discusses the competitiveness loss of exporting countries as the main trade concern associated with the use of MRLs. This is in part because stricter MRLs can be adopted by developed countries and affect products imported from developing countries located in tropical areas with high pest pressures (USITC, 2021[92]). For example, Wilson and Otsuki (2004[99]) estimated that a

1% percent increase in MRL stringency on the pesticide chlorpyrifos in OECD countries would result in a 0.15% decrease in the trade value of banana exports.

Although the literature focuses on the effects of MRLs on producers in exporting countries, these policies also raise potential concerns on the competitiveness of producers in regulating countries, especially when abatement costs are relatively high in implementing countries. When lower MRLs are implemented in an exporting country they may increase production costs of domestic producers. This may result in competitiveness loss and export decrease, potentially leading to additional environmental damages in other countries (pollution leakage). In practice, however, this scenario is less likely to happen as lower MRLs are typically set in importing countries (USITC, 2021[92]).

Box 4. EU discussion on possible expansion of the Codex Alimentarius to sustainability concerns

In February 2022, the European Union's Agriculture and Fisheries Council (2022[100]) discussed the possibility that Codex Alimentarius consider developing standards based on environmental sustainability criteria. Currently, the primary channels through which the Codex Alimentarius supports the Sustainable Development Goals (SDGs) are SDG 1, 2, 3, 8 and 12, without directly targeting SDG 13 (climate action) (Price, 2020[101]).

During the same month, the General Secretariat of the Council of the European Union (2022[102]) "highlighted the EU's willingness to explore, together with its partners, all pragmatic ways of integrating sustainability considerations into the work of the CAC, in line with commitments made by its members at international level".

An extension of the Codex Alimentarius to sustainability standards could presumably also imply their recognition in international trade legislation, as seen with the recognition of Codex food standards as an international reference under the Agreement on the Application of Sanitary and Phytosanitary (SPS) Measures (Negi, Blankenbach and Pérez-Pineda, 2020[103]). However, any change in Codex would require the agreement of all 188 Members, which face different priorities and contexts and may have different views on what sustainability standards should be. Additionally, implementing sustainability into the setting of Codex standards may prove challenging, as it would add complexity into the already resource intensive process of MRL standard making.

IRCs are voluntary, so their effectiveness is determined to a large extent by the interests and willingness of its parties to co-operate, promote, comply with and enforce the IRC scheme and associated economic, or trade related incentives (OECD, 2017[104]). Existing mutual recognition agreements have demonstrated that the successful application of IRC depends on a number of conditions and policy contexts (Correia de Brito, Kauffmann and Pelkmans, 2016[105]).

In general, IRCs have been more successful in policy areas relying on technical standards, targeting for instance environmental degradation or green growth. They are more challenging to apply in sectors dependent on specific local conditions, such as agriculture (OECD, 2013[79]).

Specific IRC mechanisms can also generate economic trade-offs. For instance, international harmonisation efforts can eliminate potential leakage, but they may impose costs on some of the countries concerned depending on their production standards, thereby potentially resulting in short term competitiveness losses for some trade partners.

International private standards, like Global G.A.P., could also play a role, but their overall effectiveness can be reduced when many different schemes coexist (Prag, Lyon and Russillo, 2016[106]).[27] Public and private institutions may play a role in increasing the standardisation of regulatory practices, ensuring effectiveness, enforcement, and compliance (Cafaggi, 2010[107]).

[27] GlobalG.A.P. is a global system to certify good agricultural practices. See https://www.globalgap.org/uk_en/.

Regional Trade Agreements

A regional trade agreement (RTA) is "any reciprocal trade agreement between two or more partners, not necessarily belonging to the same region" (WTO, 2022[108]). RTAs can be seen as a type of IRC as they fulfil the following criteria: (1) the promotion of the convergence of international standards, mutual recognition and transparency; (2) the inclusion of sector-specific provisions; (3) the spread of information, good practice and knowledge exchanges (OECD, 2017[104]). The number of regional trade agreements (RTAs) has been increasing throughout the last three decades (WTO, 2016[109]; Xiong, 2017[110]; Monteiro and Trachtman, 2020[111]).

OECD (2017[104]) reviewed the factors influencing the effectiveness of RTAs and the potential obstacles in implementing them. It was found that including sector-specific detail and the active involvement of different stakeholders are critical to the success of these agreements (Figure 3).

Figure 3. Summary of benefits, costs and success factors of RTAs

Benefits	Costs / challenges	Success factors
• Integration of trade and regulatory matters • Formalisation of transparency and enhanced dialogue on regulation among RTA parties • Promote sector specific ambitious co-operation	• Make IRC part of trade negotiations • Administrative burden and manageability of multiple agreements at national level • Limited enforcement	• Co-ordination between agencies • Political support • Consultation with private sector • Hybrid of horizontal approach and sector-specific outcomes

Source: OECD (2017, p. 48[104]).

While regional trade agreements generally do not cover environmental concerns as a primary objective, they may impact the environment through their general provisions or through specific environmental chapters or provisions.[28] General provisions can both address environmental objectives and preferential market access clauses that can benefit the environment (Martínez-Zarzoso, 2018[112]) or can include conditioning preferential tariffs on adherence to environmental norms (European Commission, 2022[113]) By adopting environmental provisions, RTA partners can specifically aim to progress on their international environmental and climate goals (George, 2014[114]).

More specifically, environmental provisions typically appear in RTAs in preambles, in the "exceptions" sections concerning the protection of human, animal and plant life, but also as commitments to uphold environmental legislation. Notably, according to a review of the 270 RTAs in force in 2016, environmental exceptions and environmental co-operation provisions remain the two more frequent types of environmental provision included in RTAs (WTO, 2016[109]). The content of such provisions includes, among others, dispute settlement mechanisms in environmental matters, the coverage of specific environmental issues, or implementation mechanisms (George, 2014[114]).

The use of environmental provisions in RTAs can help limit pollution leakages and competitiveness losses through convergence towards international environmental standards, mutual recognition of environmental principles, increased transparency, and the encouragement of exchanging sectoral knowledge (OECD, 2017[104]). The most common goals pursued by RTA partners that have used environmental provisions in RTAs are promises that current environmental commitments will not be relaxed, and that trade liberalisation will not reduce environmental protection levels (George, 2014[114]). The second and third most common objectives are to pursue coherence between climate and trade objectives, environmental co-operation and enforcement of climate laws.

[28] The overall sustainability of RTAs can be ascertained by conducting sustainable impact assessments. These assessments can help promote environmental and social objectives and offer an opportunity for dialogue with different stakeholders. Different methods exist, each with strength challenges and limitations (Moïsé and Rubínová, 2021[312]).

However, most approaches or instruments to reduce pollution leakage in RTAs are implicit rather than explicit. Most recently, the EU-UK Trade and Cooperation agreement includes a provision on carbon pricing, stating that countries will engage in defining carbon pricing instruments reducing "greenhouse gas emissions from electricity generation, heat generation, industry and aviation" (Pirlot, 2021[115]), without discussing how these would work together.

RTAs also offer a vehicle towards mutual recognition or equivalence, which can effectively limit or prevent pollution leakages or competitiveness losses, including in the case of pesticide use. These mechanisms may require parties to consider technical regulations and standards already present in other countries as equivalent, requiring justification for any exceptions (OECD, 2017[104]).

Finally, RTAs may include sectoral measures, tailoring commitments to the specificities of the agricultural sector, complementing the effects associated with increased market access. An example is given by the increasing number of RTAs that include nutritional objectives such as nutritional labelling consideration, along with the general increase in environmental provisions (Zimmermann and Rapsomanikis, 2021[116]).

Very little is known about the implementation of environmental provisions and therefore their effects, as they rarely include monitoring or evaluation requirements. George and Yamaguchi (2018[117]) surveyed 177 environmental provisions in RTAs and found that only 18 requested documentations on implementation or some type of evaluation. Still several RTAs, such as those implemented by the United States, include commitments on enforcement of provisions or dispute settlement procedures. A few empirical studies have been conducted on the impact of RTAs' environmental provisions, which face difficulties in disentangling the RTA or environmental provision effect from other domestic factors. Abman, Lundberg and Ruta found that environmental provisions were effective in avoiding deforestation caused by the agricultural expansion which typically follows the implementation of RTAs without specific environmental provisions (2021[80]). RTAs may also induce indirect losses of competitiveness and subsequent potential environmental leakages in third countries. Looking at 54 agricultural commodities, He (2021[118]) found that RTAs should be accompanied by assistance to developing countries excluded from the deals to avoid competitiveness losses.[29]

International sectoral agreements

In the context of climate change mitigation, international sectoral agreements are defined generally as transnational engagements by public and private actors to reduce the intensity of emissions in a co-ordinated manner and from specific sectors (OECD, 2020[9]). International sectoral agreements may reduce GHG emissions and limit carbon leakages in high emission sectors that are not covered by the Paris Agreement or market-based instruments.

Relevant examples of international sectoral agreements are the Carbon Offsetting and Reduction Scheme for International Aviation (CORSIA), adopted by the International Civil Aviation Organization (ICAO) to offset airlines emissions.[30] Other examples are the International Maritime Organization (IMO) sectoral agreement on maritime shipping,[31] the Global Methane Pledge, or the Global Cement and Concrete Association Sustainability Charter.[32] At this date, on the climate side, no evidence could be found of a formal sectoral agreement specifically dedicated to agriculture at the international level, though private standards on GHG emission reduction targets or measurements have been introduced by specific subsectors and multinational companies (Henderson, Frezal and Flynn, 2020[6]).

[29] At the same time, Song (2019[313]) finds that emerging economies are increasingly adopting environmental protection in the trade agreements due to the combination of local and international pressure and the fact that environmental provisions can be cost-effective.

[30] Concerning the Carbon Offsetting and Reduction Scheme for International Aviation (CORSIA), more information at https://www.icao.int/environmental-protection/CORSIA/Pages/default.aspx.

[31] Concerning IMO's sectoral agreement on international shipping, more information at http://www.imo.org/en/MediaCentre/PressBriefings/Pages/06GHGinitialstrategy.aspx.

[32] Concerning the Global Cement and Concrete Sustainability Charter, more information at https://gccassociation.org/sustainability-innovation/sustainability-charter-and-guidelines/.

Sectoral agreements could help limit potential emission leakages, as they can broaden the participation of key international players, and simplify negotiations of common objectives and actions, for example by creating effective discussion platforms (Bodansky, 2007[119]). At the same time, if they act as coalition, these agreements may also set relatively less ambitious targets, than those set up by a larger set of actors. Sectoral agreements can help limit competitiveness issues by ensuring that international competitors adopt coherent and comparable mitigation approaches, for instance by setting sector-wide targets.

Sector agreements require the definition of common standards acceptable to private and public actors, which may limit their scope and effects. As discussed by OECD (2020[9]), these agreements can be adopted in parallel with other key instruments and responses, but their effectiveness is strictly linked to the sectoral willingness to respect the agreements, as well as the commitments of governments from developed and developing countries.

3.2.2. Unilateral policy approaches

The diverse environmental targets pursued by different countries, as well as the expected unilateral adoption of market-based and regulatory instruments may induce pollution leakage and competitiveness concerns. Unlike the approaches discussed in the previous subsection, the policy instruments presented in this section are adopted unilaterally by countries or regions with more ambitious environmental policies to limit leakage or competitiveness losses.[33] They do not strictly require an agreement or co-operation mechanism with trade partners, although international co-operation is generally necessary to facilitate their international acceptance and ensure their effective implementation.

This subsection focuses on two instruments that are most often discussed in relation to GHG mitigation policies and pesticide regulations: border adjustments and reciprocal requirements. Border adjustment mechanisms are currently discussed in the context of GHG emissions,[34] while regulatory reciprocity requirements or so called "mirror clauses" are being discussed in the case of environmental regulations for agriculture, including for pesticides. A few additional options are briefly evoked at the end of the subsection.

Border Carbon Adjustments

Border Carbon Adjustments (BCAs, also referred to as Carbon Border Adjustment Mechanisms or CBAMs) can be defined as "measures applied to traded products that seek to make their prices in destination markets reflect the costs they would have incurred had they been regulated under the destination market's greenhouse gas emission regime" (Cosbey et al., 2019[120]; OECD, 2020[9]). BCAs are designed to reduce carbon leakage occurring between producers in regions implementing carbon pricing schemes and those operating in regions with lower or no carbon pricing scheme by imposing a similar extra cost on goods that are not produced under the domestic carbon pricing schemes (European Commission, 2021[121]; Meyer and Tucker, 2022[122]). As such, BCAs can limit the impact induced by the adoption of carbon pricing policies on competitiveness and emission leakages (Martin, 2021[123]; OECD, 2020[9]).

There are multiple types of BCAs, varying in their design and application (OECD, 2020[9]). In particular, they can differ according to the instrument they rely on, such as border taxes, trading permits, or the obligation to purchase carbon certificates (European Commission, 2021[121]; IPCC, 2014[124]) and according to whether they apply to imports, exports, or both (Eicke et al., 2021[125]). They also differ in their sectoral application; current proposals in the EU have focused on energy-intensive trade-exposed (EITE) goods, defined as cement, iron and steel, aluminium, fertilisers and electricity (European Commission, 2021[121]).

Economically, BCAs shift carbon costs from domestic suppliers to buyers (Martin, 2021[123]). By imposing a cost adjustment (such as via an import duty) of carbon-intensive products, they aim to ensure that all the goods entering a market receive equal environmental treatment - provided other measures are also equal (i.e. no free emissions allocations) - as signalled by their carbon prices (Morsdörf, 2022[126]). In order to avoid discrimination, BCAs would need to implement a framework to impose an equal price signal on both

[33] The term "unilateral" is used here in contrast to "multilateral".

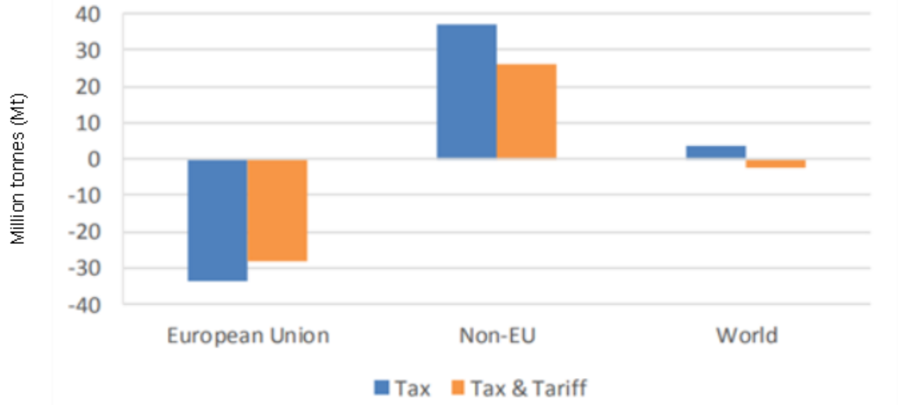
[34] Several governments have declared the intention to consider or promote some forms of CBAMs, comprising the European Union, the United States, Canada and Japan (Takeda, Tetsuya and Arimura, 2012[128]; European Commission, 2020[4]; Meyer and Tucker, 2022[122]).

domestic and foreign producers based on the carbon content of the products. According to Young (2022[127]), an estimation of producer subsidy equivalents would evaluate the effects of both direct and indirect subsidies and GHG reduction policy effects on the cost of production. The result would be an objectively determined price that would not favour one GHG reduction policy mechanism over another.

Most studies and simulations on the economic and environmental impacts of BCAs focus on the EITE sectors, which are the first targeted by existing proposals (Böhringer, Carbone and Rutherford, 2012[83]; Takeda, Tetsuya and Arimura, 2012[128]; Morsdörf, 2022[126]). These proposed BCAs would have limited effects on GHG emissions from the AFOLU sector, though they may affect the sector indirectly through energy and other agricultural inputs (Martin, 2021[123]). Indeed, the inclusion of fertiliser production in the EU proposal could affect agricultural producers in the European Union (European Commission, 2021[121]).

Nordin et al. (2019[129]) explicitly focused on the application of BCA to agriculture. They simulated the introduction of a carbon tax covering agriculture at the EU level, concluding that the introduction of a BCA - modelled as an import tariff based on carbon content imposed on top of existing tariffs - would partially offset emission leakages (which exceed 100%), as shown in Figure 4. The study also finds that a BCA may change the net increase in global GHG emissions associated with a unilateral carbon tax into a net global reduction of emissions. More studies are needed to evaluate, in particular, the role of existing trade and domestic support policies in the agricultural sector on the potential effect of BCA in different regions.

Figure 4. GHG emission changes with an EU carbon tax and with a combined EU carbon tax and tariff (BCA) in the agricultural sector



Notes: the "tax" columns indicate the increase/decrease in $MtCO_2eq$ following the implementation of a carbon tax scheme in the European Union. The "tax & tariff" columns indicate the expected increase/decrease in $MtCO_2eq$ combining a carbon tax scheme with an import BCA in the European Union. The model demonstrates how a BCA may reverse the negative overall GHG impact of a unilateral carbon tax, as evident comparing the two "world" columns.
Source: Nordin et al. (2019[129]).

More generally, beyond agriculture, the potential leakage reduction impact of BCA policies has been extensively studied based on models covering different scenarios and geographical contexts. Economic modelling has found that 5% to 25% of the GHG emissions saved by implementing economy wide carbon pricing schemes would be offset due to carbon leakage (Böhringer, Balistreri and Rutherford, 2012[130]; Branger and Quirion, 2014[131]; Morsdörf, 2022[126]; IMF, 2021[132]).[35] OECD (2020[9]) reviewed the literature and found, with the introduction of a BCA, the carbon leakage ratio would decrease to between 2% and 12%. A similar meta-analysis of 25 studies found that BCAs would decrease the carbon leakage ratio by 6% (Branger and Quirion, 2014[131]).

Fewer studies investigate the practical impact of BCAs based on existing carbon pricing systems. Fowlie, Petersen and Reguant (2021[133]), for example, try to evaluate how different carbon adjustment

[35] As noted above, these results might differ for agriculture given the presence of existing tariffs and other market distorting policies.

mechanisms may apply to California's GHG pricing scheme on the electricity sector. They conclude that parameters such as the inelasticity of electricity demand and the presence of subnational markets may make BCAs only partially effective.[36]

While BCAs may reduce potential competitiveness losses in countries with carbon pricing policies, they could induce competitiveness losses in other countries without carbon pricing schemes (Martin, 2021[123]; IPCC, 2014[124]). This will vary according to the BCA design, sector and geographical area concerned (IPCC, 2014[124]). The introduction of different BCAs, leading to inconsistent standards, may also create challenges. International collaboration towards a standardisation of BCA programmes could avoid such a problem (Nordhaus, 2021[87]; Meyer and Tucker, 2022[122]). Moreover, BCAs require significant data on GHG emissions for various commodities and the infrastructure to record that data, which makes implementation complex.

More generally, the creation of BCAs involves significant issues concerning welfare and fairness (Zhong and Pei, 2022[134]; Eicke et al., 2021[125]; OECD, 2020[9]). There are three main criticisms of BCAs: their potential ineffectiveness in limiting emission leakages; their potential to act as protectionist measures; and the fact that they may infringe equity principles enshrined in multilateral agreements (OECD, 2020[9]). There might also be competitiveness risks for third countries (Box 5). Governments interested in introducing such instruments must carefully consider their design in consultation with trading partners. They must take into account trade-offs, such as those between environmental effectiveness and administrative feasibility.

Moreover, Martin (2021[123]) argues that the inclusion of agriculture in a BCA could be problematic because of the importance of indirect land use changes which could unintentionally increase GHG emissions. Further, any mechanism would have to reflect the fact that emissions from agriculture vary substantially by production method which are also influenced by agriculture policy support measures (Poore and Nemecek, 2018[135]; OECD, 2021[44]).

Box 5. Third country competitiveness risks associated with BCAs

Reports in the literature note that the uneven distribution of trade losses associated with BCAs could induce competitiveness losses in specific geographical regions (Zhong and Pei, 2022[134]). Developing countries may be most affected by this, considering their high exposure to trade and potential higher emission intensities (Blandford and Hassapoyannes, 2018[136]; Eicke et al., 2021[125]). To take account of the development priorities of these countries and in line with the principle of common but differentiated responsibility and respective capabilities (CBDR-RC), governments implementing BCAs may adopt exemptions for least developed and/or other developing countries (Carbon Market Watch, 2021[137]; OECD, 2020[50]). Furthermore, the revenues generated by these mechanisms could be used to support innovation and international climate finance, contributing to achieve sustainable development goals (Zhong and Pei, 2022[134]; Morsdörf, 2022[126]).

Mirror clauses

"Mirror clauses" have been used since the 1950s in the field of bilateral investment treaties (Yannaca-Small, 2006[138]). In this context, these instruments are commonly referred to in the literature also as "umbrella clauses", "parallel effect", "*pacta sunt servanda*", "observations of commitments clause", or "clauses with a mirror effect'' (Weissenfels, 2005[139]; Yannaca-Small, 2006[138]). Within the scope of investment treaties, they are generally defined as provisions encompassing the agreement to "cover any contractual commitments and other obligations of the host state to the foreign investor" (OECD, 2020[140]). In other words, they "require a state party to observe any obligation or commitments it enters into with respect to investments of the other state party" (Li, 2018[141]). Mirror clauses may serve several purposes according to their design (Samson and Ugale, 2021[142]) including to safeguard the interests of private economic actors operating in international markets (OECD, 2020[140]).

[36] More generally, carbon adjustment mechanisms are expected to be beneficial only for goods for which the elasticity of demand is consistently lower than the elasticity of supply (Martin, 2021[123]).

Introducing the concept of mirror clauses in the agricultural trade policy context could in principle be a potential response to the environmental leakages and domestic competitiveness losses generated by the adoption of specific unilateral environmental regulations, including in relation to pesticides. In this case, mirror clauses would entail imposing the same requirements on foreign products entering in markets as those faced by domestic producers (Carles and Kirsch, 2021[143]).

Although the scope and context are different, there are examples of established regulatory import mechanisms in the safety and phytosanitary area. Certain non-tariff measures in agriculture are governed under the WTO Agreement on the Application of Sanitary and Phytosanitary Measures (SPS Agreement). The SPS Agreement encourages countries to use international standards, guidelines, and recommendations but countries may adopt higher levels of protection if there is scientific justification for it, or if they are based on appropriate assessment of risks. The Agreement also notes that SPS measures should be applied only to the extent necessary to protect human, animal or plant life or health, and should not arbitrarily or unjustifiably discriminate between countries where identical or similar conditions prevail (WTO, 1994[144]). It is argued that regulatory approval of chemical substances *de facto* imposes regulatory requirement on manufacturers of these substances. EU legislation has also applied a ban on imports of animal products derived from animals having been administered antimicrobial drugs for growth promotion purposes reflecting a domestic ban of these products (Council of the European Union, 2022[145]; European Commission, 2022[113]).[37]

Although this view is not agreed by all observers and governments, it has been argued that the same principle could apply to environmental purposes, while remaining in line with countries commitments under the WTO (Galindo, 2021[146]). Given the absence of such approaches in these areas, no evidence is available to demonstrate their possible economic and environmental effects.

There are several possible practical and legal concerns around the use of mirror clause mechanisms for environmental purposes. Carles and Kirsch (2021[143]) identify three primary risks involving the adoption of mirror clauses for environmental purposes. First, the fact that they would interfere with the food production sovereignty of other countries, and risk to generate trade conflicts if not appropriately designed, including in accordance with WTO rules. Second, enforcing these measures implies significantly increasing administrative and border technical controls on food imports, with associated administrative burdens. This is especially the case if the measure requires additional control of process and production methods. Finally, the unilateral adoption of mirror clauses risks losing trust in trade, economic and political partners, and subsequently increasing the risk of retaliatory measures.

Moïsé and Steenblik (2011[147])identified technical, administrative, economic and trade feasibility constraints limiting the potential application of trade-related measures based on processes and production methods (PPMs) to enhance climate change mitigation. Similar issues may occur in the case of mirror clauses, such as the risk of imposing on trading partners PPM requirements that favour excessively expensive or unavailable technologies (Moïsé and Steenblik, 2011[147]).

Governments advocating mirror clauses will also be called upon to demonstrate how these are not disguised protectionist measures and do not violate the WTO principles of non-discrimination. The use of mirror clauses would need to take into account the different environmental and climate conditions of the trading partners affected, which may further enhance the technical complexity of this instrument and limit its capacity to address environmental leakages and competitiveness losses.

[37] Regulation (EU) No 2019/6 of the European Parliament and of the Council of 11 December 2018 on veterinary medicinal products.

Other unilateral actions

Several other proposals to help limit environmental and trade-related challenges associated with the direct policy instruments are emerging in the literature or entering political debates. These include:

- Exemptions or allocation of GHG emission allowances determined on the basis of historical output levels or benchmarks for specific products. These policy instruments, which offer credits to exporters exposed to international competition, effectively limit leakage but are generally less effective, at least with respect to EITE goods (Böhringer, Carbone and Rutherford, 2012[83]; Takeda, Tetsuya and Arimura, 2012[128]; Morsdörf, 2022[126]).
- *Import bans for products issued from illegal deforestation:* there have been proposals to ban products issued from illegal deforestation, which could potentially limit leakage related to land use change and associated competitiveness effects. Such schemes would require evidence of the link between production and illegal deforestation at the farm level for the purposes of identifying relevant exports, implying robust monitoring and enforcement mechanisms (Wolosing, 2022[148]). Implementing countries may also need to reassure trading partners that such mechanisms would be applied in a non-discriminatory manner and would not be used as disguised protectionism. That said, illegal deforestation related import bans by consuming countries could potentially help incentivise improved governance of forest management in producing nations, especially when paired with appropriate technical and financial support.
- *Reducing MRLs to zero tolerance levels for selected non-approved pesticides,* on case-by-case basis, has been suggested in the European Union as a possible means to mitigate environmental and trade related challenges associated with using locally prohibited pesticides.[38,39] As MRLs are currently established based on food safety assessments in accordance with good agricultural practices (Box 3), zero tolerance MRLs would need to be justified in accordance with WTO rules. While it would in theory limit the use of specific substances in other countries, not all countries require the same substances, so the effect it could have on pesticide type and use in other countries may not be significant. Furthermore, applying zero tolerance could be costly to implement for all supply chain actors, given that some minimum quantity of residue may be present in containers and adjacent products and the measure could generate additional trade tensions if not appropriately designed (Nitzko, Enno and Spiller, 2022[149]; Drogué and DeMaria, 2012[150]).

[38] The EU submitted a proposal to the WTO in June 2022 to delete the MRLs of clothianidin and thiamethoxam, pesticides already banned in the European Union, effectively banning the import of products with any traceable residue of these substances (WTO Committee on Technical Barriers to Trade, 2022[311]). Following the risk assessments performed by European Food Safety Authority concluding that due to their intrinsic properties, the exposure from outdoor use of clothianidin and thiamethoxam leads to unacceptable risks for bees, or such risks could not be excluded based on the available data.

[39] It should be noted, however, that not all countries support this approach on the basis of MRLs as food safety, rather than environmental standards.

4. Employing alternative domestic policy instruments to address environmental and trade-related challenges

Direct policy instruments, as discussed in Section 3 are only a subset of possible environmental policy instruments used in the agriculture and food sector. This section looks at the use of alternative instruments which could be employed in a second route to improve environmental performance while facilitating international markets.

There is a wide range of domestic policy instruments to mitigate GHG emissions from agriculture or to reduce the environmental harm from inappropriate or excessive pesticide use in agriculture. They can act as substitute or complement to the direct policy instruments. Three categories of instruments are reviewed: (1) supply-side policy instruments, primarily targeting producers; (2) demand-side policy instruments focusing on consumers; and (3) policy instruments leveraging the role of agriculture and food supply chain actors.[40]

The main findings of this section are summarised in Table 6. Specifically:

- Supply-side policy instruments offer relatively more direct and cost-effective policy options to address environmental challenges, including GHG mitigation and the limitation of environmental harm from pesticides than other alternatives. While some reviewed options can improve competitiveness and global environmental benefits, others may induce some generally limited pollution leakage or competitiveness losses in specific circumstances.
- In contrast, demand-side policy instruments present limited environmental and trade-related challenges, but their environmental effects are indirect and they may not always be cost-effective. There is also less information available on some of these options that are generally less used.
- Limited evidence was found on the potential effects of policy alternatives relying on supply chain engagement. They may increase costs for supply chain actors, albeit with limited effect on the competitiveness of domestic producers. Their environmental benefits depend on multiple factors, but they tend to avoid creating pollution leakage.

[40] In particular, "beneficiary pays" approaches and consumer taxes are substitutes to "polluter pays" approaches like taxes, while reforms of potentially environmentally harmful support is limiting incentives to produce, hence can serve as an important complement.

Table 6. Comparison of reviewed policy alternatives

Subsection and instruments	Relationship with direct policy instruments	Potential to limit environmental harm in a cost-effective way	Potential to generate Pollution leakage and Competitiveness losses	Primary conditions of success
	4.1. Supply-based alternative policy instruments			
4.1.1. Beneficiary pays approaches	Substitute	Medium effectiveness depending on adoption, type, and associated costs (H,M)	Limited depending on scale and scheme (M,M)	-Compensation levels -Quality of information provided
4.1.2. Land use policies	Substitute or complement	Medium effectiveness, potential costs (M,M)	Limited to moderate impact depending on program design and scale, and associated costs. (M,M)	-Enforcement procedure, monitoring & evaluation
4.1.3. Research and development	Substitute or complement	Potentially high effectiveness per unit of expenditure (M,M)	Can generate competitiveness gains and potential environmental gains in other countries in the medium to long term when paired with appropriate environmental and land-use policies. (M,H)	-Stability of funding -Long-term vision -Technology of adoption
4.1.4. Reforming potentially environmentally harmful support	Complement	Effectiveness depends on the type of support, subsidy reform generates revenue. (L,M)	Moderate pollution leakage depending on specific support measures. Limited competitive loss but possible in the short run in some cases. (L,M)	-Reform process management -Coupling with environmental regulations
4.1.5. Reducing food loss and waste[1]	Substitute or complement	High effectiveness and low costs (L,M)	Can generate competitiveness gains. (L,M)	-Efficient technologies
	4.2. Demand-based alternative policy instruments			
4.2.1. Green public procurement	Substitute or complement	Moderate effectiveness rate, depending on impact on non-providers, debatable costs (L,M)	Depending on non-adopters unless local companies are privileged. (L,M)	-Open and transparent procedures -Combination with other instruments
4.2.2. Environmental labelling and information schemes	Substitute or complement	Effectiveness depends on label design and adoption (L,L)	Uncertain due to limited research. (L,M)	-Guidance to avoid multiplication of labels -Quantifying impact of different approaches -Consumer education campaign
4.2.3. Behaviorally informed policies	Substitute or complement	Low costs and potentially high effectiveness, depending on techniques (M,M)	Unlikely unless specific products are favoured (L,L)	-Larger scale adoption -Establishing PPPs with key sectoral actors
4.2.4. Consumption taxes	Substitute	Indirect effect compared to direct incentive (L,M)	Limited effects (M,M)	-Targeting key products, limiting other food system effects
	4.3. Private sector engagement alternatives			
4.3.1. Enhanced due diligence	Substitute or complement	Depending on enforcement and effects on non-committing companies (L,M)	Depending on enforcement and non-adopters. Costs may incur for the whole supply chain. (L,M)	-Identification of risks across the supply chain -Enforcement procedure monitoring & evaluation
4.3.2. Mandatory labelling	Substitute or complement	Medium enforcement costs. Effectiveness proven in other policy fields, not yet in environment (L,M)	Not available, costs may incur for the whole supply chain	-Balance between specificity, clarity and adoption costs
4.3.3. Taxonomy classification systems	Substitute or complement	Not available	Not available however, costs may incur for the whole supply chain	-Clear, transparent and science-based categorisation

Notes: 1. Reducing food loss and waste can also be considered a demand side instrument.
2. Volume and Agreement determined within brackets (Volume, Agreement). Volume: 0-5 articles = low (L); 5-10 = moderate (M); 10+ = high (H). Agreement determined by divergence in findings.
Source: Authors based on reviewed literature.

4.1. Supply-side alternative policy instruments

Five types of instruments are discussed in this section: (1) beneficiary pays approaches incentivising farmers to change practices; (2) land use policies; (3) reforming potentially environmentally harmful subsidies; (4) R&D investment; and (5) limiting agriculture and food loss and waste.[41]

4.1.1. Beneficiary pays approaches

Beneficiary pays approaches can be defined as payment mechanisms to reward farmers for enhancing their environmental performance (Ezzine-de-Blas et al., 2016[151]). Four categories of payments are considered here: agri-environmental payments, payments for ecosystem services (PES),[42] pollution abatement payments, and offset mechanisms. While they all share the common goal of using funding to incentivise practice change, they differ in payment scope, modalities and performance metrics.

Potential to limit environmental harm

Agri-environmental payments are public support designed to support environmental improvement of agricultural activities. They are widely used in OECD countries with different objectives, related to climate change or limiting pollution, although their overall budget is limited compared to mainstream agriculture support (OECD, 2021[44]). Their design characteristics, including performance metrics, targeting and tailoring, which can be difficult to measure, are key conditions to their cost-effectiveness (Guerrero, 2021[152]; OECD, 2010[153]). They generally support the use of specific farming practices, which has limited their environmental effectiveness (DeBoe, 2020[154]). The use of these payments has covered climate mitigation, in particular in the European Union, with limited results thus far (European Court of Auditors, 2021[155]). Some payment schemes have been introduced for farmers that require avoiding the use of specific pesticides, as seen in Switzerland or Japan (Finger, 2021[72]; OECD, 2021[156]). There is an increasing interest in shifting payment basis towards results-based or hybrid mechanisms that would consider results and practices, including thanks to digital solutions (OECD, 2019[157]).

Payments for ecosystem services (PES) compensate producers or landowners for the additional costs of providing environmental services (OECD, 2013[158]). They can involve private or public funders, with government often involved in the setting up of the schemes. The success of these schemes requires the establishment of flexible conditions for securing the service provision and disincentives to breaching PES agreements (OECD, 2010[159]; Lankoski et al., 2015[160]). Forms of PES are commonly used in many OECD countries, comprising EU countries, Norway, Switzerland or the United States (OECD, 2013[79]).

Abatement payments aim at rewarding producers based on the quantity of GHG emissions reduced, at carbon market price levels (in CO_2 equivalent). Australia's Emission Reduction Fund (ERF) is an example of abatement subsidy scheme whereby landowners and farmers can earn Australian Carbon Credit Unit which can be sold at the carbon market to obtain compensation. Abatement payments can be effective instruments in limiting emissions at a sufficiently high carbon price, although their global mitigation potential was estimated to be half that of an equivalent carbon tax (Henderson and Verma, 2021[30]; OECD, 2019[29]).

Offset mechanisms enable voluntary commitments from private firms to compensate environmental harm, like CO_2 emissions, through investments in projects that reduce emissions, such as carbon sequestration or afforestation (Box 6). There is an increasing number of programmes to develop carbon farming (carbon sequestration on farms) schemes, such as the *Label Bas Carbone* in France. Offsets can be traded and, in some cases, be integrated in ETS, as in California (Elliott et al., 2022[161]; Henderson, Frezal and Flynn, 2020[6]). Other examples include the Alberta Emission Offset System (AEOS) and the UK Farm Soil Carbon Code (Elliott et al., 2022[161]; Arvanitopoulos, Garsous and Agnolucci, 2021[13]). Biodiversity offset mechanisms have also been applied in many countries, with various design and effectiveness, though their effect on pesticide damage is more difficult to gauge (OECD, 2016[162]).

[41] Discussions in this subsection is based on the reviewed literature, with volume and agreement of reviewed reports compared in Table 6.

[42] The term can be used for all types of beneficiary pays approaches, here it is taken in a more restrictive interpretation, considering schemes set for payments by public or private entities to reward the provision of ecosystem services.

Evidence suggests that the overall impact in terms of these different payment mechanisms has been limited thus far, though they do hold significant potential. Given their voluntary nature, the overall effectiveness of these schemes largely depends on their adoption by farms where the most benefits would accrue or at a sufficiently large scale, in addition to their design and monitoring and enforcement systems.

Box 6. Potential of payments toward soil carbon sequestration (SCS)

Agricultural land has the potential to sequester on average 2.5 Gt C per year, which would offset around 20-35% of current global fossil fuel GHG emissions (Henderson et al., 2021[82]). Soil carbon sequestration potential from agriculture also vary widely across countries, which may call for targeted measures (Rodrigues et al., 2021[163])

More generally, this would require significant innovations in production methods and incentive designs to achieve such potential due to three challenges (Henderson et al., 2022[164]). First, the non-permanence of soil carbon stocks poses challenges associated with the risk of paying for abatement that is lost at some future point. Second, transaction costs in general, including financial transaction expenses (such as legal and brokerage fees) and measuring reporting and verification mechanisms, can raise the costs of contracting carbon credits (from 3% to 85% of total credit value) and reduce landholders' willingness to participate in carbon markets. Third, non-additionality is an important issue that can affect the environmental integrity of carbon credits generated by net SCS practices.

Future research into policy design options for contracting solutions to address the issues of non-permanence, lower transaction costs and provide greater assurance of additionality would help to improve the feasibility of using market-based policy measures to incentivise net SCS.

Potential to generate environmental and trade-related concerns and possible success factors

Beneficiary-pays approaches are expected to generate limited environmental and trade-related challenges (OECD, 2019[29]), in particular because these programmes are voluntary (Lankoski et al., 2015[160]; OECD, 2013[158]). This is especially the case of GHG emission abatement payments (OECD, 2019[29]). By paying farmers willing to reduce GHG emissions, they theoretically compensate the costs of reducing emissions, therefore not inducing competitiveness and subsequent leakages a priori. The exception may be if emissions are reduced on the extensive margin, with reduced production not compensated by payments. In this case, carbon leakage may occur due to the displacement of production by the voluntary farmer.

Other types of payments rewarding better practices generally result in limited or no losses in competitiveness for farmers, who may decide to fetch price premiums by upgrading the quality of their products. These payment schemes could induce a leakage effect depending on their scale of application. For instance, beneficiary pays approaches incentivising the compliance with chemical inputs use restrictions can be successful in reducing direct GHG emissions (Henderson and Lankoski, 2019[165]). However, they may induce land use changes favouring emission leakages, for example by stimulating the expansion of agricultural land and thus inducing biodiversity losses. Broader adoption of payment schemes through generally less productive practices to limit pesticide use and harm could also imply pollution leakages.

At the same time, beneficiary pays mechanisms can also result in efficiency gains, thereby offering both environmental improvement and competitiveness gains if they encourage a change in production systems. For instance, producers adhering to PES or other beneficiary-pays programmes could improve their productivity per hectare, by combining production practices through agro-silvo pastoralism (Abdul-Salam, Ovando and Roberts, 2021[166]; Kragt, Dumbrell and Blackmore, 2017[167]).

Several factors influence the success rate of these approaches. These include the level of compensation (Rico Garcia-Amado et al., 2011[168]; Ma et al., 2012[169]; Kragt, Dumbrell and Blackmore, 2017[167]), the perceived stability and viability of payments programmes (Dumbrell, Kragt and Gibson, 2016[170]; Evans, 2018[171]), the degree of trust in the information received, and uncertainties around co-benefits (Evans, 2018[171]; Baumber et al., 2020[172]). The question of additionality also matters; environmentally beneficial practices ought to be designed to ensure farmers adopt practices that deliver their full environmental potential (Sidemo-Holm, Smith and Brady, 2018[173]). This is especially the case in the context of soil carbon sequestration, where benefits from specific practices are generally neither guaranteed nor permanent, thereby requiring long-term contracts (Henderson et al., 2022[164]).

4.1.2. Land use policies

Land use policies, defined as policies to encourage conservation, can provide other levers to limit environmental harm. They can act as complements to beneficiary pays approaches. Different approaches may exist to protect or conserve land, targeting active or non-active land. First, government agencies can protect specific sensitive areas, such as establishing conservation or protected areas to prevent agricultural expansion (Wolfe and Elizondo, 2020[174]). For example, Costa Rica established ambitious conservation area programmes over the last decades, harnessing the cross-sectoral potential of these policies by using conservation areas to promote the country as leading eco-touristic location (Valverde, 2018[175]). Farmers may also need to comply with specific protection practices, for example in the cases of peatland drainage bans, or for norms protecting endangered plant or animal species (DeBoe, 2020[176]; Arvanitopoulos, Garsous and Agnolucci, 2021[13]). Second, governments can restore the potential of an agricultural landscape to act as a natural sink or enhance its biodiversity through the financing of ecosystem restoration practices (Malak et al., 2021[177]). This is the case, for example, of forest afforestation or reforestation programmes, such as payments under the EU Forest Strategy for 2030. Third, conservation easements can help preserve land, such as the US Agricultural Conservation Easement Program (ACEP) (DeBoe, 2020[176]).

Potential to limit environmental harm

Preserving or restoring peatlands, agroforestry techniques or crop rotation have high GHG mitigation potential (Lamb et al., 2016[178]; Rodrigues et al., 2021[163]). Peatlands and coastal wetlands alone, while only accounting for 3% of soils, store 600 GtCO2eq, twice the carbon stock of all forest biomass, which accounts for 30% of soils (Kirpotin et al., 2021[179]).

Afforestation, reforestation and deforestation regulation also present relatively cost-effective sequestration options (MacLeod et al., 2015[180]). For example, Nilsson and Shopfhauser (1995[181]) estimated that a global afforestation programme of 345 million hectares would entail the potential of sequestering 1.48 Gt CO_2eq per year at its maximal annual sequestration rate, to be achieved 60 years following the initiation of the programme. At the same time, the issue of non-permanence can prevent to achieve this potential, and it may be more effective to combine a forest biomass growth with increased use of long-lived wood products.

Land conservation measures targeting active land can reduce input use, preventing agricultural input infiltrations in soils, such as Nitrate leaching to aquifers (Henderson and Lankoski, 2019[165]; OECD, 2019[17]). In general, larger conservation programmes are more effective in preventing pesticide and other types of runoffs in wetlands and other sensitive ecosystems near croplands (Belden et al., 2012[182]).

Protected areas can cost effectively limit environmental harm. For example, Grafton et al. (2021[183]) calculated that the global average costs of sequestration practices from forestry may be negative for conservation, afforestation and reforestation. Adequately estimating the costs of ecosystem restoration is key, particularly in the case of forest restoration, as these are not always calculated realistically (Brancalion et al., 2019[184]). However, the ecosystem services benefits of restoration projects usually outweigh their costs (Taillardat et al., 2020[185]; De Groot et al., 2013[186]).

Potential to generate environmental and trade-related concerns and possible success factors

Specific land use policies can induce limited environmental leakages or competitiveness losses. Setting protected areas limits land for production and may result in production losses in the short-term that could create leakage; however it may also push farmers to increase agriculture productivity on remaining active land, as seen for instance in a study in Brazil (Koch et al., 2019[187]).[43] Policies encouraging the restoration of active land face similar trade related risks as the beneficiary pays policies discussed above.

[43] Agriculture productivity growth can in turn lead to land expansion in some contexts (Byerlee, Stevenson and Villoria, 2014[200]).

Conservation easement programmes typically target the less productive land, with limited or no expected effect on competitiveness or leakage. Environmental and trade related challenges may be limited if land exchanges are done under formal or informal international coordination mechanisms, such as the UNREDD+ (FAO, 2022[188]), the UN Decade for Ecosystem Restoration (UNEP, 2020[189]) or the International Union for Conservation of nature (IUCN, 2022[190]).

A key determinant for success of land conservation regulation is enforcement (Haupt et al., 2020[191]; IPCC, 2014[124]). Public-private partnerships can improve enforcement, such as in Costa Rica where the local public sector works with reliable local private sector organisations to administer conservation areas (Wolfe and Elizondo, 2020[174]; Arvanitopoulos, Garsous and Agnolucci, 2021[13]). Improved unmanned aerial vehicles (UAVS), such as drones or satellite technology can reduce the cost of enforcing regulations by improved remote sensing to monitor the state of forests and other ecosystems (Kinaneva et al., 2019[192]; Wang et al., 2019[193]; Arvanitopoulos, Garsous and Agnolucci, 2021[13]).

4.1.3. Research and development

Research and development (R&D) is one of main drivers of agricultural innovations underpinning sustainable productivity growth (OECD, 2013[194]; OECD, 2019[195]). Governments can support R&D by providing funding (especially long-term, stable funding), offering tax credits for private R&D investments, and other incentives (IPCC, 2014[124]; OECD, 2019[17]). Second, they can strengthen intellectual property rights (IPR), for example through patents (IPCC, 2014[124]). Third, they can facilitate innovation by improving co-ordination and linkages across research institutes, by fostering international co-operation or by facilitating the development of public private partnerships (PPPs) (Wreford, Ignaciuk and Gruère, 2017[196]; OECD, 2019[17]).

Cost-effectiveness to limit environmental harm

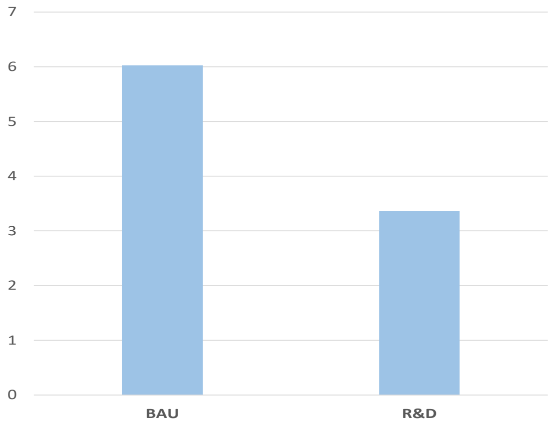
R&D can significantly contribute to reduced GHG emissions through increased total factor productivity and the development of more effective methods to reduce GHG emissions (OECD, 2019[197]; Fuglie et al., 2022[198]). Burney, Davis and Lobell (2010[199]) estimated that increase in agriculture productivity, measured in terms of crop yields, contributed to reduce GHG emissions from agriculture by 590 $GtCO_2eq$ or 13 $GtCO_2eq$ per year from 1961 to 2005. As such it was one of the most effective mitigation efforts in that period. Looking forward, Fuglie, Hertel and Baldos (2022[198]) projected that accelerated R&D investments could reduce cropland expansion by 2050 by almost half via increased productivity (Figure 5). Similarly, a modelling study showed that an increasing agricultural productivity by 10% by 2030 with no increase in inputs use would lead to a GHG emission reduction of 340 $MtCO_2eq$, equivalent to 6% of global sectoral emissions (OECD, 2019[197]).

At the same time, land use saving will require regulatory barriers to be effective in countries facing risks of land use change. Indeed, increased productivity may lead to land use related rebound effects by inducing the expansion of the production in specific regions (Byerlee, Stevenson and Villoria, 2014[200]). The risk that this offsets the GHG emission gains of productivity investment is particularly high in developing regions exposed to the risk of deforestation (Hertel, Ramankutty and Baldos, 2014[201]).

R&D investments also help limit the use and environmental impact of agricultural inputs. Research can develop more efficient, targeted and less environmentally harmful pesticides, and improved seed varieties that need less pesticides (OECD, 2016[41]). At the farm level, R&D can develop improved input application techniques, such as precision agriculture techniques or improved agronomic practices (OECD, 2019[157]). At the supply chain level, R&D may improve traceability and transparency, stimulating in turn more responsible pesticides application practices. Finally, at the consumption stage, technological innovation can inform consumers, for instance via more reliable data on the origin of products and their treatment (OECD, 2019[157]). While these investments are critical for limiting the use and impact of agricultural inputs, these innovations can also take years, if not decades, from conception and development to field adoption.

Evenson (2001[202]) estimates the median economic returns from agricultural research often exceed 40%. Similarly, a literature review by Alston (2010[203]) concludes that productivity gains following R&D investments exceed by many times the total expenditure in research, irrespective of the method of measurement. While R&D processes are slow to pay off, their impact on productivity can last for many decades (Alston, Beddow and Pardey, 2009[204]). The availability of skilled researchers (scientists, engineers) and the high costs of new technologies remain the highest barriers to investment (OECD, 2019[205]).

Figure 5. Projected effects of accelerated agriculture R&D on global cropland change to 2050



Note: y-axis: Areas in million hectares; BAU: Business as usual; R&D: Accelerated research and development.
Source: Fuglie, Hertel and Baldos (2022[198]).

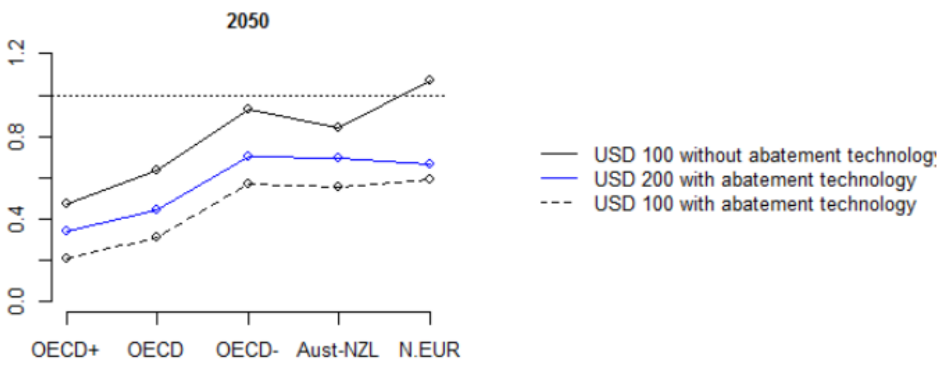
Potential to generate environmental and trade-related concerns

Investments in agriculture R&D directed towards sustainable productivity growth can generate competitiveness gains instead of losses. Furthermore, international co-operation, through linkages across national innovation systems and associated networks can accelerate the diffusion of productivity gains across borders (OECD, 2019[195]).

R&D investment on GHG abatement technologies for agriculture can also reduce the risk of pollution leakages. Henderson and Verma (2021[30]) estimated carbon leakages associated with different carbon tax rates in the agricultural sector, in the presence or absence of abatement technology development. They find that mitigation policy packages including new abatement technologies can significantly enhance the effectiveness of carbon pricing policies and help minimise carbon leakage regardless of the region applying the carbon tax (Figure 6). At the same time, the effect of R&D on agriculture productivity could also generate leakage if it results in land use change in foreign countries. This indirect effect might happen in countries with partial or imperfectly applied land use regulations.

Figure 6. Percentage leakage rates for different carbon tax scenarios in agriculture by 2050

Comparative results for the application of USD 100/tCO_2e or USD 200/tCO_2e carbon taxes with and without abatement technology innovation



Notes: C taxes applied by the different regions or countries shown below the figure. OECD+: OECD, Brazil and China; OECD-: Australia-New Zealand, Northern Europe, Canada; Aust-NZL: Australian-New Zealand; N.EUR: Northern Europe.
Source: Henderson and Verma (2021[30]).

4.1.4. Removing potentially environmentally harmful agricultural support

Certain types of agricultural support can incentivise farmers to produce more or use less sustainable practices (OECD, 2013[206]). Phasing out these potentially environmentally harmful forms of support (which are also the most trade distorting) while prioritising investments for public goods, could improve agriculture's environmental performance and could improve the overall performance of food systems (OECD, 2021[44]).

Three types of agriculture support have been identified as potentially most environmentally harmful: price support measures, payments based on commodity outputs and payments based on unconstrained variable input use (Henderson and Lankoski, 2019[165]; Mamun, Martin and Tokgoz, 2021[207]; DeBoe, 2020[176]; OECD, 2013[206]). These support policies have been associated with increased domestic GHG emissions and nutrient surpluses (Henderson and Lankoski, 2019[165]). These forms of support may also negatively affect biodiversity by reducing crop diversity depending on their design (DeBoe, 2020[176]).

Potential to limit environmental harm

The GHG mitigation potential of removing potentially harmful agricultural support measures is directly linked to the extra GHG emissions induced by each specific measure. Some agricultural subsidies may directly increase sectoral emissions, for instance, when they encourage the use of emission-intensive inputs such as nitrogen fertilisers or fossil fuels (OECD, 2019[17]). Existing literature suggests that market price support and coupled payments contribute the most to increased GHG emissions from domestic agriculture production, as these encourage intensification in fertile areas (DeBoe, 2020[176]). Some of the most common negative environmental effects induced by support policies in terms of GHG emissions are linked to the increased number of livestock units, the expansion of agricultural land and its reallocation across outputs, and biodiversity losses (Henderson and Lankoski, 2019[165]; DeBoe, 2020[176]). The extent of the effect of these various support measures depends on the existing structure of farming system (OECD, 2020[208]). The effects of other measures, such as payments based on area, are more ambiguous (Henderson and Lankoski, 2019[165]).

Few studies looked at the effects of agriculture support reform on pesticides, as how support affects pesticide use depends on the specific production system. That said, when support leads to intensive crop production or maintains pesticide intensive activities, reform is most likely to generate gains (DeBoe, 2020[176]).

Potential to generate environmental and trade-related concerns

In general, some studies find a negative relationship between agricultural support and productivity (DeBoe, 2020[176]), suggesting that removing harmful subsidies can actually help competitiveness. Structural change occurring after deep reforms has helped some countries to increase the overall productivity of their agriculture. But not all support is harmful; some forms of support may push farmers to invest in innovation and improve production processes.

Unilaterally removing environmentally harmful support measures may in some cases lead to GHG emission leakages that reduce the environmental benefits of reform. For instance, Jansson et al. (2021[209]) used the CAPRI model to simulate reforms of EU output payments. They find significant emissions reductions, but also that most of these emissions are offset by GHG leakages outside Europe as imports increase. Other modelling studies have found that global agricultural trade liberalisation could have limited net effects on global GHG emissions as production moves to more emission-intensive producing regions or provokes land use change (Laborde et al., 2021[210]; Guerrero et al., 2022[211]). The effect would however depend on the type of agriculture policy reform: removing border measures could slightly increase GHG emissions as agricultural production relocates and increases., removing commodity-specific coupled payments, would have the opposite effects. Better land use regulations would help increase the net GHG benefit of such reforms (Guerrero et al., 2022[211]).

In the case of partial trade liberalisation, leakage would vary according to the countries implementing reforms, the responsiveness of their farming sector, the type of product and level of support, as well as the state of the international market, as mentioned in Section 3 of this report. Furthermore, reforming potentially environmentally harmful support in a particular country can reduce domestic GHG emissions with international benefits (OECD, 2021[44]; Jansson et al., 2021[209]).

4.1.5. Reducing food loss and waste

More than 20% of world food production is lost to food losses and waste (FLW) (FAO, 2019[212]).[44] This implies not only reduced food availability, but also increased waste disposal and unnecessary GHG emissions (Slorach et al., 2019[213]). Food losses is generally taken to mean losses throughout the supply chain, and food waste is food unsold at the retail level or purchased but not consumed by households (OECD, 2019[214]).

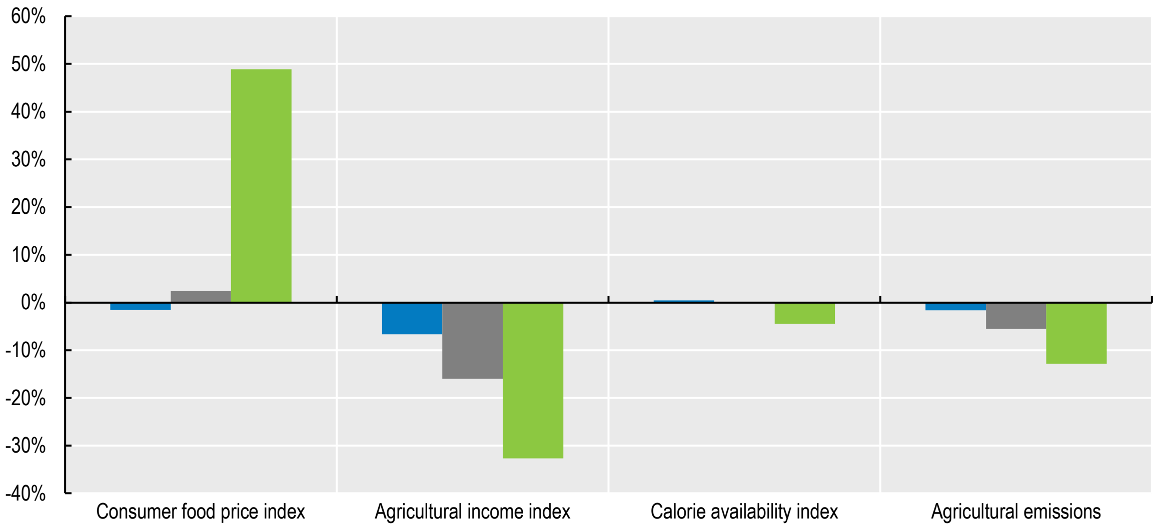
Current agriculture and food supply chains are far from achieving zero FLW (OECD, 2019[214]). While FLW prevention has become a priority in OECD countries, few governments have conducted economic assessments of their food waste prevention policies. An example of FLW policies include Canada's Food Waste Reduction Challenge, a USD 15 million initiative promoting innovative business models for reduced waste across the supply chain. Another example is Türkiye's national strategy on the Prevention, Reduction and Monitoring of FLW, including sectoral goals and targets (OECD, 2021[44]).

Potential to limit environmental harm

The total carbon footprint of FLW, including losses from land use change, was estimated to be 4.4 $GtCO_2eq$ per year; if FLW was a country, it would be the third largest emitter in the world (Scialabba, 2015[215]). Considering the reduction potential in different regions (Figure A A.1), the FAO estimated that it is possible to avoid 1.4 $GtCO_2eq$ per year of emissions from FLW (Scialabba, 2015[215]).

Studies have used *ex ante* modelling to measure the potential of FLW reduction on GHG emissions and other economic outcomes. OECD foresight scenarios on the food waste component of FLW concluded that progressively eliminating food waste from 2018 to 2030 globally, accounting for the cost of waste, would lead to a 14% reduction of sectoral GHG emissions (800 $MtCO_2eq$ by 2030) (OECD, 2019[197]) (Figure 7). Under this scenario, the lower the waste levels, the higher the expected disposal costs; conversely, where large quantities of food waste are present, it is expected to be easier and cost-effective to reduce these, the first units of waste being "cheaper" to dispose (OECD, 2019[197]).

Figure 7. Projected effects of eliminating food waste on GHG emissions and food security by 2030



Note: The presented scenario accounts for the cost of waste. Columns indicate the intermediate effects of a progressive elimination of food waste over a ten-year period, showing its potential effects on various indexes and emissions in 2020 (blue), 2025 (grey) and 2030 (green). This scenario assumes the consumer must pay for waste reduction which negatively impacts the agricultural income index and calorie availability over time.
Source: OECD (2019[17]).

[44] According to HLPE (2014[317]), FLW can be defined as "a decrease, at all stages of the food chain from harvest to consumption in mass, of food that was originally intended for human consumption, regardless of the cause".

FAO estimated that 50% of FLW in the distribution and consumption phases would be avoidable in both developing and developed countries, while a 5% reduction in food losses could be prevented in developed countries (Scialabba, 2015[215]). Furthermore, the analysis argued that the greatest reduction potential of food losses lied in developing countries, where more than 50% of food losses could be avoided at the production and post-harvest stage, respectively.

Improved food loss and waste management practices with high emission reduction potential include the reduction of post-harvest losses, storage losses, waste from food processing, trade and consumption practices (Henderson, Frezal and Flynn, 2020[6]). A number of examples involves recycling or reuse of organic matter. In the United States, for instance, the state of California introduced a programme on mandatory collection of organic waste from residential households and businesses in 2022. Organic materials are then composted or turned into biogas instead of being deposited in landfills. It is estimated that organic waste in landfills is responsible for 20% of the state's methane emissions.[45]

Slorach et al. (2019[213]) recently ranked the cost and environmental benefits associated with the most discussed FLW treatment options, quantifying the potential emission and cost reductions in the United Kingdom. They find that preventing avoidable FLW, for example via economic incentives, regulations or behavioural approaches, is the best option, generating estimated reductions of 14 $MtCO_2eq$ and GBP 10.7 billion annually in the United Kingdom. By comparison, they find that anaerobic digestion is the food waste treatment option with the largest environmental potential in terms of tCO_2eq/year saved (490 000 tonnes in the United Kingdom) though it is also more costly than other options (GBP 251 million/year) (Slorach et al., 2019[213]).

The pesticide-related effects of FLW reduction are more ambiguous. Pesticides can be an important tool to prevent post-harvest losses (FAO, 2019[212]). But reducing FLW may also reduce the use of pesticides and could indirectly affect their potential environmental harm. The net effects depend on the dynamics of supply and demand shifts associated with the reduction of FLW. For instance, a more efficient supply chain for fruits and vegetables with costly FLW could result in lower area of production, with lower pesticide use overall (Conrad et al., 2018[216]). In general, the targeted use of agricultural inputs as well as the optimal access to the most efficient pesticides is regarded as the best combination to reduce FLW and especially food losses while reducing the environmental impact of pesticides (Neff, Kanter and Vandevijvere, 2015[217]; Thompson, 2003[218]).

Potential to generate environmental and trade-related concerns

Although reducing FLW can be an effective strategy to reduce GHG emissions, it may have different economic implications. Costs associated with food loss prevention may impact producers and increase food prices (OECD, 2019[197]). The adoption of FLW processing technologies, which are environmentally effective but relatively costly, such as anaerobic digestion, may have implications on producers' competitiveness (Slorach et al., 2019[213]).

These potential losses may be offset by efficiency gains, Okawa (2015[219]), for example, suggests that efficiency gains from FLW reduction could lead to increased trade, benefitting both developed and developing countries. A transition to an economy that can efficiently reuse waste may also radically transform agriculture policy and trade, creating demand for new skills, opening new markets, and increasing the volume of international trade (OECD, 2019[214]).

Studies are missing on FLW and pollution leakage and the absence of effective international mechanisms to measure FLW makes it difficult to estimate their impact (Lopez Barrera and Hertel, 2021[220]). The effects of FLW reduction on carbon leakage depends on the overall effect on production and consumption responses. A more efficient food chain may actually reduce GHG emissions in multiple countries (Slorach et al., 2019[213]; OECD, 2019[214]). So long as adopting FLW reduction policies unilaterally does not result in large production shifts, it is unlikely to create significant leakage (OECD, 2019[197]; Thyberg and Tonjes, 2016[221]).

[45]More information on this programme is available at:https://calrecycle.ca.gov/organics/slcp/collection/#:~:text=Beginning%20in%202022%2C%20SB%201383,provides%20solid%20waste%20collection%20services.

4.2. Demand-side alternative policy instruments

Governments can leverage consumers' demand for sustainable products domestically, therefore encouraging a greening of products on markets regardless of their country of origin. Demand-side instruments may not only result in positive environmental outcomes, but also encourage food chain actors to play a role and deliver more sustainable food products.[46]

According to the IPCC (2014[124]), three broad categories of demand side instruments can be used for environmental purposes: the provision by governments of public services, information schemes, and regulatory approaches. This subsection focuses on the potential application of policy instruments of the first two categories on agri-food consumption policies. It discusses the provision of public services through green public procurement (discussed under 4.2.1.), which aims at harnessing the direct purchasing capacity of public bodies to influence the demand for more sustainable products. Second, it discusses the potential of environmental labelling and information schemes (ELIS) to guide consumers towards more environmentally sustainable food products (4.2.2.). Third, it discusses behaviourally informed policies as a means to educate and encourage more sustainable consumption (4.2.3).

4.2.1. Green Public Procurement

Public procurement generally refers to the purchase of goods, services and works by public actors, such as governments or state-owned enterprises (Arvanitopoulos, Garsous and Agnolucci, 2021[13]). Harnessing the fact that public bodies' purchases account for around 12% of the GDP in OECD countries, green public procurement (GPP) can play a role in complementing traditional environmental policy tools (OECD, 2020[222]; Olave and Staropoli, 2021[223]; Arvanitopoulos, Garsous and Agnolucci, 2021[13]). This also encompass public procurement on food products and associated services (Deconinck and Hobeika, 2022[224]). Food procurement typically covers the provision of food to schools, health and social care, higher education, government office canteens, sports and leisure arenas, prisons and defence services, underlining the potential of this policy instrument at the subnational or local level (Neto and Caldas, 2018[225]).

Guidelines have been published on GPP for food and catering services, including by the European Commission (European Commission, 2019[226]). Still, only 44% of countries adopting GPP practices have included food and catering as priority area for GPP action (UNEP, 2017[227]). FAO (2021[228]) reviewed the potential of GPP for sustainable food and healthy diets across food systems, identifying conditions for effective GPP schemes, such as a sound monitoring and evaluation framework or appeals to promote adherence to voluntary schemes.

Potential to limit environmental harm

The partial adoption of GPP programmes, primarily due to high costs of adherence for local actors, makes it difficult to estimate their potential positive environmental impact at the large scale (Czarnetzki, 2019[229]; Deconinck and Hobeika, 2022[224]). The IPCC recognises the important role of GPP in climate change mitigation at the national and sub-national level in the AFOLU sector and noted that GPP is compatible with carbon taxes (IPCC, 2014[124]). FAO (2021[228]) highlights the potential spill over effect GPP practices may have, as the increase in demand for sustainable products they would induce may go beyond public sector purchases. For example, if consumers enjoy products consumed in public canteens, they may purchase more sustainable products for their own consumption.

Several studies emphasised that the greatest GHG mitigation potential of GPP practices lies in production, processing and upstream transportation of food products, especially in the case of meat products (FAO/Alliance of Bioversity International and CIAT/Editora da UFRGS, 2021[228]; Cerutti et al., 2016[230]). More specifically, they emphasise the role GPP may have in amending the composition of public canteen menus towards low GHG emission alternatives (FAO/Alliance of Bioversity International and CIAT/Editora da UFRGS, 2021[228]) as a key effective strategy. While some studies attempt to quantify GHG savings at

[46] Discussions in this subsection is based on the reviewed literature, with volume and agreement of reviewed reports compared in Table 6.

the local or subnational level (Cerutti et al., 2016[230]; Lindström, Lundberg and Marklund, 2020[231]), no quantification of GHG savings in food GPP could be found at the national level.

Public procurement may have a limited role in the selection and use of pesticides in agriculture, as these remain the decision of private agents (WHO, 2012[232]) (with the exception of gardening in cities etc.). GPP schemes can drive the demand for food produced through responsible pesticide practices, integrated pest management (IPM), or organic farming.[47] For example, a series of voluntary organic purchasing programme for public food services in Denmark led to a 24% increase in the purchase of organic food three years after the beginning of the project (Sørensen et al., 2016[233]). Similarly, a study led in Sweden demonstrated how targeted GPP expenditures are significantly positively correlated with organic farmland area (Lindström, Lundberg and Marklund, 2020[231]). Other studies suggest that the public sector can influence input procurement processes by sharing guidelines and criteria on appropriate selection of pesticides (Van Der Berg et al., 2020[234]). However, given the variation within both conventional and organic production, these types of schemes may not always reduce environmental impact.

More generally, one of the potential limitations of the effectiveness of GPP on environmental harm relies in the fact that while it encourages environmental improvement for suppliers, it will not affect other food suppliers who may continue using the same practices and could in some case fetch other market niches. In the case of pollution that may translate into a positive or limited effect but in the case of natural resource management, such as land use, any non-committed supplier may continue and even increase its land expansion, thanks to lower competition.

Potential to generate environmental and trade-related concerns

The large-scale implementation of GPP schemes for food is unlikely to induce significant competitiveness losses or environmental leakages, so long as it rewards environmental performance on a voluntary basis. In fact, international competitors may have additional incentives to make their production more sustainable, to participate to the public tenders associated with GPP schemes.

At the same time, two circumstances may lead to environmental and trade related concerns. First, generalised GPP requirements could encourage improved environmental practices with low productivity, which might trigger imports with subpar environmental standards in other markets to fill the gap. Second, as discussed above, if a broad GPP is conditioned on the absence of land use change, non-participants might actually take their place and could potentially increase overall land use change.

Differentiating between conventional or environmentally friendly products at the public procurement level may create concerns in the international trade arena (Malumfashi, 2010[235]). The plurilateral Agreement on Government Procurement (GPA), which ensures open, fair and transparent conditions of competition in the government procurement markets for WTO member signatories, recognises the value of the sustainable procurement concept (WTO, 2012[236]).

GPP can promote the design of improved labelling criteria and encourage innovation (Czarnetzki, 2019[229]). Indeed, according to the IPCC, R&D “technology push” policies are most effective when complemented by “demand-pull” policies, like GPP (IPCC, 2014[124]). Combining the use of these two policy options could be a win-win outcome for GHG mitigation and potentially in the case of pesticides damage reduction.

4.2.2. Environmental Labelling and Information Schemes

Environmental labelling and information schemes (ELIS), which provide information concerning one or more aspects of the environmental performance of a product or service to external users, have been multiplying over the past 25 years (Gruère, 2013[237]). Environmental labels in particular have been the most used on food products, often based on some type of certification scheme or seals. Yet less than 2% of agricultural land was certified under a sustainability standard scheme in 2018 (Meier et al., 2020[238]), suggesting a potential for further growth.

[47] Organic farming however may be associated with higher GHG emissions per unit of production (OECD, 2016[41]).

Governments can support ELIS in many ways, from offering a guidance on private claims, to designing harmonised labels or standards (Table A A.13). Many ELIS are managed by private entities operating within guidelines defined by governments, who therefore play the role of framing institutions.

A large majority of ELIS are voluntary schemes displaying specific environmental attributes (Gruère, 2014[239]).[48] They allow producers to distinguish themselves from competitors, but also empower consumers by allowing them to make more informed consumption choices (Djekic et al., 2021[240]). ELIS targeting environmental components of food production may be observable in multiple forms, such as certifications disclosing information on production techniques (e.g. organic labels), single issue labels like those looking at biodiversity (or deforestation-free certifications), or environmental declaration displaying the performance of a product (Keller, 2013[241]; Pistorius and Foote, 2022[242]; Gruère, 2013[237]; Deconinck and Hobeika, 2022[224]).

Sustainability standards and certification schemes are key elements of voluntary labelling practices, as they have the role of ensuring the reliability of the indications provided by labels, distinguishing labelling initiatives from greenwashing practices (OECD, 2011[243]). The role of governments in voluntary food standards lies in the protection of consumers and preventing fraud, as well as in the generation of functioning food markets and improving the efficiency, the design, the implementation and the monitoring of these instruments (Rousset et al., 2015[244]). Governments can also ensure the transparency on the definition of the label and associated measurements, which is critical to the effectiveness of ELIS.

Cost-effectiveness to limit environmental harm

Initially focused on organics and other farming practices, agriculture and food-related ELIS have increasingly covered other attributes including GHG emissions, based on carbon footprint measurements. Organic labels were the dominant form (Meemken, 2020[245]) of ELIS in the 1970s and still represented the 15% of total ELIS in 2012 (Gruère, 2013[237]). This mode of production prohibits the use of synthetic pesticides, which is one of the reasons behind the positive direct effects of organic farming on local biodiversity (OECD, 2016[41]). Public efforts to harmonise labelling, such as the USDA National Organic Program, or the EU organic programme, have helped develop consumer awareness and understanding. Several other labelling initiatives promoting responsible use of pesticides or low pesticides residual products aimed at rewarding producers engaging in such agronomic practices. For instance, France recently decided to consider integrated agriculture ("*agriculture raisonnée*") production methods eligible to obtain the "high environmental value" (HVE) (Ministère de l'Agriculture et de l'Alimentation, 2022[246]).

Different types of labels can be used to disclose information and GHG emissions (Table 7) (Deconinck and Hobeika, 2022[224]). Product carbon footprint are usually based on a limited range of GHG emissions standards, such as the British PAS2050, or the WRI Greenhouse Gas Protocol.[49] Due to the vast range of initiatives in the food certification field, their voluntary nature and the wide difference in the sustainability requirements associated, quantifying the overall potential reduction in terms of GHGs from the adoption of labelling or certification schemes is complex (Prag, Lyon and Russillo, 2016[247]; Keller, 2013[241]).

Evidence on the environmental effectiveness of ELIS is mixed (Prag, Lyon and Russillo, 2016[106]; Deconinck and Hobeika, 2022[224]; Traldi, 2021[248]). While some of the core practice-based labels have increased the use of specific farming practices, outcome-based performance measurements are rare. For instance, a comprehensive empirical study of voluntary sustainability standards for coffee and cocoa found that it was more effective at improving economic and social sustainability than environmental sustainability (COSA, 2013[249]). Still, there is evidence of the fact that labels can contribute to ensure the success of and raise awareness of carbon sinks, biodiversity conservation programmes or other instruments aimed at reducing sectoral GHG emissions (IPCC, 2014[124]).

[48] Mandatory labelling is discussed together with other information disclosure requirements in Section 4.3.2.

[49] More information on the PAS2050 Carbon Footprint standard is available at https://ghgprotocol.org/sites/default/files/standards_supporting/GHG%20Protocol%20PAS%202050%20Factsheet.pdf. More information about the WRI Greenhouse Gas Protocol is available at https://ghgprotocol.org/.

Table 7. Consumer focused carbon footprint labelling concepts

	What is labelled	Claim	Demonstration of GHG reduction across categories	Evaluation
Compensation label	Suppliers' purchase of compensation certificates equal to GHG emissions	'Climate-neutral'	No	Supply of compensation schemes is limited; no communication of CO2eq associated with product category; transition option for suppliers
Reduction label	Reduction of GHG emissions by a certain percentage	'X% decrease in GHG emissions'	No	Can incentivise product improvements; no communication of CO2 eq. associated with a product category
Best-in-class label	Significant lower GHG emission than average of food category or market leader	'particularly climate friendly'	No	Can incentivise product improvements; no communication of CO2 eq associated with a product category
Absolute CO_2 eq. label	CO2 footprint, the absolute value of GHG emissions per kg	GHG in kg CO2eq per kg of product	Yes	Promotes dietary change; accurate, but demands high consumer involvement
Multi-level categorical label	Normative rating of absolute GHG emissions through colour-coding	Green equals a low CO2 footprint	Yes	Promotes dietary change; simple; sensitive to scaling decisions; does not incentivise producers to demonstrate small improvements
Categorical label with absolute CO_2 eq. values	Colour coding in combination with the absolute value of GHG emissions	Absolute CO2eq value with a normative colour coding	Yes	Simple, accurate and can promote dietary change; incentivises producers to demonstrate small improvements

Note: "Demonstration of GHG emission" refers to the possibility of allowing consumers to verify the emissions associated to a product, for example by including GHG measurements verified by reliable third parties.
Source: Lemken, Zühlsdorf and Spiller (2021[250]).

Potential to generate environmental and trade-related concerns

The implications of ELIS on competitiveness remain limited, as certified products, that are adopted on a voluntary basis, are usually sold at higher prices on the market (Meemken, 2020[245]). Beyond informing consumers, they can enhance the competitiveness of producers who have invested in sustainable production practices (Meemken, 2020[245]). Simpler ELIS may be more effective, but they also risk over-simplification, potentially favouring certain producers (Taufique et al., 2018[251]). Governments can promote reliable, science-based labelling and certification schemes, to prevent unfair or discriminatory practices.

While studies are missing in this area, the non-constraining nature of voluntary labelling schemes makes it unlikely to lead to environmental leakages. On the contrary, when these schemes are effective in increasing the demand for sustainable products, new producers or new markets may start providing sustainable food products.

4.2.3. Behaviourally informed policies

"Behaviourally informed policies", or "nudges", are techniques aimed at influencing the choices of consumers without creating mandatory requirements or the use of financial incentives (Leonard, 2008[252]; Sunstein, 2014[253]; Thaler, 2018[254]). Communication strategies, education programmes, as well as programmes encouraging the promotion of food products which have a limited impact on the environment, are all strategies that the public sector can deploy to affect consumption choices. Concrete examples include reminders, warnings, information about the consequence of a specific choice, simplification, or instituting default rules (Reisch et al., 2021[255]). These practices may be particularly effective in the agri-food sector, as behavioural experiments suggest that environmental sustainability is not one of main driving factor for food purchases (OECD, 2017[256]).

Awareness campaigns, education initiatives, nudges and other behavioural insights techniques can have a low-cost, relatively significant potential to mitigate GHG emissions in agriculture and food (Table 8). Large-scale studies led in the United States, for example, demonstrated that specific forms of nudges incentivising the consumption of plant-based food in restaurant menus brought to double the number of times these dishes were ordered (Blondin et al., 2022[257]).

Table 8. Behavioural interventions to reduce GHG emissions in the food sector

Intervention	Examples
Default rules	Introducing "Meatless Mondays" in public canteens
Simplification	Simplifying access to vegetarian menu choices
Use of social norms	Emphasising what most people are doing and eating
Increase in ease and convenience	Making low-carbon options more visible, their access easier and more convenient
Priming	Using visual, or spatial –or other forms- of primes (e.g. store design, or signs in shops)
Disclosure	Disclosure of environmental costs associated with meat consumption on a menu
Warnings	Coloured carbon warning labels on meat products
Pre-commitment strategies	Self-pledge to reduce food waste by a certain percentage
Reminders	Reminding people of their plans, for example via email or text message
Eliciting implementation intentions	Asking "do you plan to eat meat?"
Informing people of the nature and consequences of their own choices	Disclosing what earlier food choices meant, e.g. in terms of GHG savings
Physical or digital micro-environment changes altering the context of a choice	Ordering products on shelf spaces in supermarkets or of choices on a website; changing the affordances and signalling atmosphere of a building
Other	Applying other nudges not covered (e.g. framing, herding, feedback, and praise)

Source: Adapted from Reisch et al. (2021[255]).

Default nudges (e.g. "Meatless Mondays") are considered in the literature as the most effective type of behavioural economic instrument to mitigate GHG emission (Hummel and Maedche, 2019[258]). Reliable quantifications of the impact of behavioural insights in the food environmental sustainability domain are missing. Instead many studies demonstrated the effectiveness of nudges in other related fields, such as nutrition and health (Hummel and Maedche, 2019[258]; Reisch et al., 2021[255]; Vecchio and Cavallo, 2019[259]).[50]

Governments may play a role in incentivising the use of priming techniques in food retail stores, markets, the hospitality industry, or canteens through public-private partnership initiatives (e.g. store design, or signs in shops and menus), thereby promoting more sustainable products (Wilson et al., 2016[260]). These nudges can be combined with other policy instruments, such as carbon taxes, to enhance their effectiveness.

Education initiatives promoting the nutritional and environmental benefits of buying sustainable food products including food produced using responsible pesticide management practices can play an important role in changing purchasing behaviour (OECD, 2011[261]). Like nudges, there is a lack of evidence around the mitigation potential of education on the environmental impact of food, though evidence from the nutrition and health field confirms how food literacy programmes have statistically significant effects on food consumption choices (Vardanjani et al., 2015[262]; Bailey, Drummond and Ward, 2019[263]). For example, the perceived health benefits of organic foods, despite mixed epidemiological evidence, has greatly influenced its growth in demand among consumers in OECD nations (OECD, 2011[261]).

Awareness campaigns, education initiatives and other behavioural insights techniques are unlikely to generate environmental and trade-related challenges, given their indirect and voluntary nature. So long as they remain positive, they encourage the adoption of farm practices driven by increased demand. The non-constraining nature of behavioural programmes is not likely to lead to environmental leakages. On the contrary, additional demand may encourage producers from other markets to improve their environmental performance.

[50] Some experiences have been led also on the supply-side, testing how farmers would respond to a series of nudges stimulating the reduction in the use of pesticides (Kuhfuss et al., 2016[314]; Buchholz and Musshoff, 2021[315]; Kuhfuss et al., 2015[316])

4.2.4. Consumer taxes

In the case of climate change mitigation, a market-based alternative to carbon tax, which would directly affect pollution at the source, would be to tax consumers for those products with embedded carbon (Deconinck and Hobeika, 2022[224]). Multiple strategies have been discussed in the literature; however, there has not been any large-scale tax based on GHG emissions in food thus far.

Much of the discussion in the literature has focused on the high GHG mitigation potential of changing diets, particularly limiting food based on ruminants. Foresight studies have shown that limiting their consumption would have potential; for instance, OECD (2019[17]) finds that a reduction of 10% of the per capita consumption of ruminant based products would lead to a 15% GHG reduction in agriculture. However, the impacts of policies to incentivise dietary changes may not be as important; the same modelling study found that a USD 60 tCO_2e tax on carbon for consumer demand would lead to 5% GHG emission reduction. Dietary changes may also not be as effective as a direct carbon tax on production. Henderson et al. (2021[82]) find that reducing 50% of consumption of livestock products, except in India and less developed countries,[51] would be only half as effective as a USD 70 tCO_2e carbon tax on the AFOLU sector.

Martin (2021[123]) argues for taxing GHG emissions through a carbon specific value added tax. A "carbon-added tax", for example, would convey market signals directly to consumers in importing markets, instead of to farmers (Courchene and Allan, 2008[264]). Agriculture producers would adjust to market signals from the demand shifts following the changed price of the products sold on the market. However, McLure Jr (2010[265]) argued that if based on emission accounting, this option might be associated with significant implementation and enforcement costs. There are also different views over how to measure the carbon content of goods, which might undermine the potential application of these policy instruments.

While these strategies would indeed theoretically eliminate leakage, their cost-effectiveness still needs to be studied. They also may have multiple other effects on food systems, from nutrition to livelihoods that will vary in different contexts and would warrant being explored.

4.3. Private sector engagement

A third way to improve the environmental performance of the sector, in between supply and demand policy alternatives, is to engage directly with large food supply chain actors that participate in international agriculture and food markets (Deconinck and Hobeika, 2022[224]). This channel of action is increasingly being discussed at the international level (G20, 2021[2]; G7, 2021[1]). Engaging with private sector actors can be beneficial for governments for three main reasons. First, due to their size, corporate strategies of large firms affect the food supply chain at all levels, thereby influencing farmers, food processors, retailers and distributors (OECD/FAO, 2016[266]). Second, while they generally have adopted corporate responsibility strategies, there are still margins of improvements; for instance, only 26 of the 350 largest food and agriculture MNEs had adopted climate targets in line with the Paris Agreement in 2021 (World Benchmarking Alliance, 2021[267]). Third, considering their large economic returns, multinational enterprises and more generally private financing actors could help finance the long-term investments needed to promote the sustainable transition of the sector (FAO, 2021[268]).[52]

Designing effective policy frameworks to engage with the private sector is challenging, especially considering the international dimension of its most influential actors. The 2016 OECD-FAO Guidance for Responsible Agricultural Supply Chains (2016[266]) offers a set of recommendations for food systems companies to progress. Building on this work, this subsection briefly reviews how government could engage with supply chain actors to improve the environmental performance the agriculture food sector. Three approaches covered here are due diligence processes, mandatory labelling, and the setting of green taxonomies (see Deconinck and Hobeika (2022[224]) for a more extensive discussion of initiatives taking a supply chain approach).

[51] As these are protein deficient countries.

[52] Discussions in this subsection is based on the reviewed literature, with volume and agreement of reviewed reports compared in Table 6.

4.3.1. Due diligence processes

In this context, due diligence means fulfilling the duty to prevent and mitigate (environmental) harm (Krieger, Peters and Kreuzer, 2020[269]).[53] Practically, this may take the form of information disclosure requirements, mandatory consultations, impact assessment of environmental risks, sharing benefits, or grievance mechanisms (OECD/FAO, 2016[266]).

Due diligence requirements can be an effective means to encourage companies communicate their climate impacts. Furthermore, when backed by enforcement and sanctions, mandatory disclosure measures may concretely contribute to reduce the negative environmental impacts of the targeted companies.[54]

OECD and FAO (2016[266]) designed a framework to promote responsible due diligence along agricultural supply chains, based on the following five steps: (1) establishing effective enterprise management systems ensuring climate responsibility; (2) identifying, evaluating and prioritising risks along supply chains; (3) designing and implementing strategies responding to the identified risks; (4) monitoring and evaluation of the due diligence process; (5) reporting supply chain due diligence. The OECD also developed more general provisions on MNEs disclosure practices, including in the case of environment as part of the OECD Guideline for MNEs (Boxes A A.1 and A A.2).

The potential impact of mandatory due diligence likely depends on two main factors: the presence of enforcement procedures, and the nature of sanctions (Villiers, 2019[270]). In those cases in which sanctions are not applied, the success would depend instead primarily on factors such as the reputational risk associated with the agricultural or food product, or on the type of company concerned, its visibility on the market, its visibility on the media, and the premium it gains from marketing itself as sustainable (Michelon, 2011[271]; Lydgate et al., 2022[272]). At the same time, as for green procurement programmes, the effectiveness of voluntary due diligence schemes will also depend on which companies do not adopt these schemes and their potential additional environmental footprint (a sort of rebound effect, including on land use).[55]

While no evidence could be found on environmental and trade-related challenges, these mechanisms may impose costs on supply chain actors, but these are likely to cover both domestic and imported products, without significant effect on competitiveness. Leakage is also expected to be limited, although this will depend on whether schemes result in changes in behaviour for non-participating companies.

4.3.2. Mandatory labelling

A specific form of disclosure particularly affecting the food sector is that of mandatory labelling requirements (Dannenberg, Scatasta and Sturm, 2011[273]; Gracia, Loureiro and Nayga, 2007[274]). This specific form of disclosure can be distinguished from due diligence because, like many other labels and standards, it aims at providing information to consumers (Djekic et al., 2021[240]). In practice, however, it remains an instrument directed to the supply chain as it affects intermediate actors, particularly food manufacturers first. Indeed, faced with a new mandatory label, food manufacturers decide whether to change their procurement strategy, modify ingredients, and revise their manufacturing processes for labels on their products to appear more appealing to consumers (Gruère, Carter and Farzin, 2008[275]). In the case of declarations, which display quantitative attributes, labels then may lead to first demand shifts, and reactions from other suppliers. Mandatory labelling can therefore induce impacts on supply chain actors and on the demand-side (Dannenberg, Scatasta and Sturm, 2011[273]).

While competitiveness and leakage concerns are unlikely to emerge, mandatory labelling schemes can limit market access and competition on international markets. For instance, a complex product carbon footprint requirement, which would rely on life cycle assessment may be difficult or costly for exporters, and depending on the design, penalise their products compared to the domestic one (Gruère, 2013[276];

[53] While this section focuses on the environment, in due diligence, harm can refer to firms' impacts on the environment as well as other dimensions such as human rights, health and safety of workers, animal welfare, etc.

[54] In particular, there are efforts in some OECD countries to require companies to perform due diligence in an aim to prevent numerous products, such as cocoa, coffee, and palm oil, sourced from deforested land from entering their markets.

[55] For further discussion on the effectiveness of due diligence, see Deconinck and Hobeika (2022[224]).

Prag, Lyon and Russillo, 2016[106]). Design of labels therefore matters; there is a trade-off between the quality of information on the environmental attribute and potential trade costs. The challenge is to ensure that the environmental information is accurate and robust without generating international costs.[56]

4.3.3. Green finance classification systems

Public and private investors and financing companies are increasingly interested in investing in sustainable assets (OECD, 2021[277]). Using a taxonomy to define what economic activity is sustainable is an instrument that can offer an indirect incentive for food companies to change their sourcing strategy, and indirectly improve the environmental performance of agriculture.

A taxonomy for sustainable activities classifies and lists activities that are seen to be sustainable according to a set of criteria (Schütze et al., 2020[278]). An important example is the European Union's Green Taxonomy, which offers a classification system associated with standardised reporting mechanisms (European Commission, 2022[279]). The EU Green Taxonomy comprehensively covers the most emitting sectors, including the AFOLU sector (Table 9).

Taxonomies do not directly affect environmental performance but help determine which activities are sustainable under a standardised, uniform and science-based method (Schütze et al., 2020[278]). Taxonomies can help improve transparency on the climate impact of investments and they can also encourage the private sector to adopt more accurate systems and methos to measure GHG impact. The standardisation they offer can also serve a reference for private investors' sustainability labels.

These classification systems may facilitate green finance, allowing the best environmental performers to attract investments as well as obtaining reputational rewards or risks (Pacces, 2021[280]). Taxonomies can also help government and public agencies prepare the ground for introducing measurable –thus enforceable and accountable- environmental regulatory schemes.

Table 9. EU Green Taxonomy AFOLU Sector technical screening criteria

Classification	Environmental contributions						
Activity	Climate Change Mitigation		Climate change Adaptation	Water	Circular Economy	Pollution	Eco-systems
	Own performance	Transitional activities					
Afforestation	Yes		Yes	Yes		Yes	Yes
Ecosystem rehabilitation	Yes		Yes	Yes		Yes	Yes
Reforestation	Yes		Yes	Yes		Yes	Yes
Existing forest management	Yes		Yes	Yes		Yes	Yes
Conservation forest	Yes		Yes	Yes		Yes	Yes
Growing of perennial crops	Yes	Yes	Yes	Yes	Yes	Yes	Yes
Growing of non-perennial crops	Yes	Yes	Yes	Yes	Yes	Yes	Yes
Livestock production	Yes	Yes	Yes	Yes	Yes	Yes	Yes

Note: Further information on table methodology can be found within the technical annex of the report (EU Technical Expert Group on Sustainable Finance., 2020[281]). "Transitional activities" refer to these activities which increase GHG emissions but are crucial to achieve sustainable development objectives, in this case by ensuring food security. Livestock production activity section considers the mitigation potential of the maintenance and further sequestration of carbon stocks, and the avoidance of GHG emissions (such as through animal management).
Source: Adapted from EU Technical Expert Group on Sustainable Finance (2020[282]).

[56] Country of origin labels, which have been discussed in this area, have a low cost of implementation, but may bias consumer decision regardless of the environmental attribute of the products.

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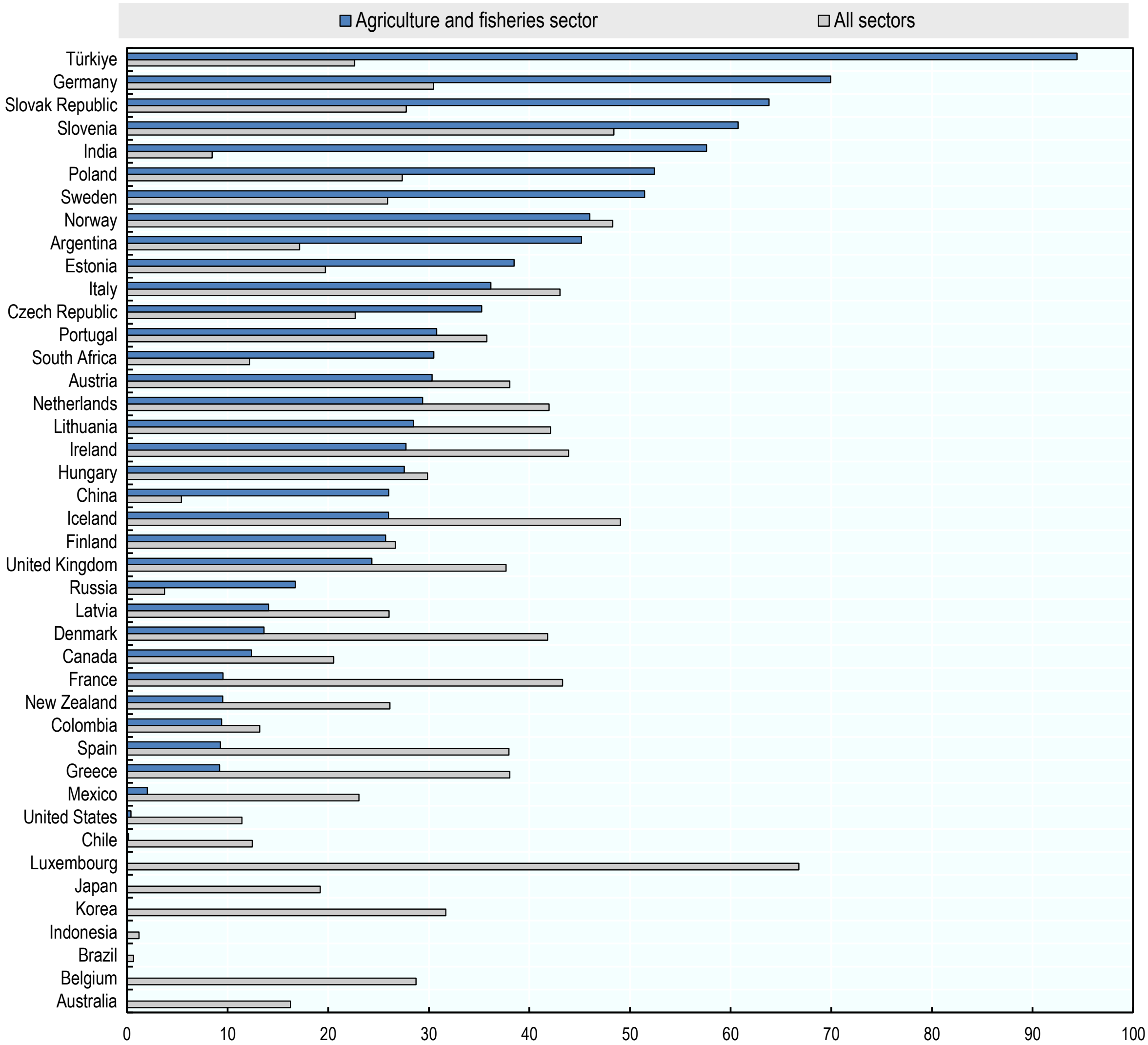
Annex A. Additional tables and figures

Figure A A.1. Effective carbon rate (ECR) of agriculture and fisheries sector and for all sectors

Carbon scores, in percentage, expressing how close the sector is to a EUR 120/tCO_2 pricing target for energy



Source: OECD (2021[283]) data.

Table A A.1. GHG intensities of different products in selected OECD and G20 countries

Country	AUS	BRA	CAN	CHN	FRA	DEU	IND	IDN	JPN	KOR	NZL	UK	US	Averages		
Products	Carbon/GHG intensity (kgCO$_2$eq./kg), 2015-2017 average													All	OECD	EE
Cereals (w/o rice)	0.2	0.2	0.3	0.2	0.1	0.2	0.3	0.2	0.2	0.5	0.2	0.2	0.2	0.2	0.2	0.2
Rice	0.7	0.5		0.8			0.7	1.1	0.8	0.8			1.1	0.8	0.9	0.8
Beef	22	35	19	16	20	13	108	46	10	17	15	16	12	27	16	51
Pig meat	2.5	2.5	2.1	1.0	1.5	1.2	5.0	4.7	0.9	1.0	2.1	1.3	2.0	2.1	1.7	3.3
Milk	0.6	1.2	0.5	0.8	0.6	0.5	1.1	1.5	0.3	0.2	0.8	0.5	0.4	0.7	0.5	1.1

Notes: AUS: Australia, BRA: Brazil, CAN: Canada, CHN: People's Republic of China, FRA: France, DEU: Germany, IND: India, IDN: Indonesia, JPN: Japan, KOR: Korea, NZL: New Zealand; EE: Emerging economies
Source: FAOSTAT (2021[284]).

Table A A.2. Own-price elasticities of demand of different products in major regions

Region / Product category	North America	Latin America	East Asia	Asia other	EU	Europe other	Former Soviet Union	Middle East	North Africa	Sub Saharan Africa	Oceania	International sector aggregate
Cereals	-0.68	-0.36	-0.63	-0.59	-0.19	-0.42	-0.32	-0.58	-0.33	-0.50	-0.16	-0.33
Dairy	-0.41	-0.58	-0.69	-0.53	-0.55	-0.62	-0.59	-0.66	-0.57	-0.68	-0.42	-0.57
Fruits & Vegetables	-0.75	-0.50	-0.67	-0.64	-0.49	-0.70	-0.43	-0.62	-0.42	-0.56	-0.30	-0.50
Meat	-0.62	-0.54	-0.66	-0.53	-0.49	-0.54	-0.55	-0.59	-0.52	-0.60	-0.39	-0.50
Oils and fat	-0.32	-0.37	-0.64	-0.59	-0.17	-0.42	-0.34	-0.56	-0.34	-0.44	-0.19	-0.36
Other food	-0.41	-0.61	-0.64	-0.71	-0.53	-0.77	-0.68	-0.77	-0.70	-0.92	-0.48	-0.68

Note: the estimates are weighted averages of existing estimates using the sample size as weights.
Source: Femenia (2019[285]).

Table A A.3. Share of total imports/exports in total productions (%) of different products as in 2018

Product	Traded quantity (million metric tonnes)	Production quantity (million metric tonnes)	Share (%)
Apple	14 563	124 972	11.65
Banana	23 916	125 405	19.07
Beans	3 870	30 565	12.66
Beef	14 204	77 804	18.26
Egg	2 962	115 101	2.57
Maize	183 239	1 374 303	13.33
Milk	62 403	882 621	7.07
Nuts	7 384	24 254	30.44
Palm oil	52 393	71 633	73.14
Pork	18 256	175 665	10.39
Rice	68 287	991 358	6.89
Soyabeans	244 889	359 735	68.07
Wheat	223 257	863 922	25.84

Source: Calculation based on FAOSTAT (2018[286]).

Table A A.4. AFOLU specific GHG mitigation targets by 2030 in selected countries and regions

	Economy-wide emissions reduction targets		Long-term strategy submitted to UNFCCC	Agriculture-specific target (base year/level)	Global methane pledge (reduce global CH_4 -30% from 2020 levels by 2030)
	2030 target (base year/level)	2050 target			
Argentina	Max 359 $MtCO_2eq$	None	No	None	Yes
Australia	-26-28% (2005)	Net zero	Yes	None	No
Brazil	-50% (2005)	Net zero	No	None	Yes
Canada	-40-45% (2005)	Net zero	Yes	-30% fertiliser emissions by 2030 (2020)	Yes
Chile	Max 95 $MtCO_2eq$	Net zero	Yes	None	Yes
China	Peak CO_2 ; -65% GDP emission intensity (2005)	Net zero by 2060	Yes	None	No
Colombia	-51% (BAU)	Net zero	Yes	None	Yes
Costa Rica	Max 9.11 $MtCO_2eq$	Net zero	Yes	None	Yes
European Union	-55% (1990)	Net zero	Yes	None at EU level	Yes
EU Member States			18 out of 27 countries (except BGR, CYP,[1] EST, GRC, HRV, IRL, ITA, POL, ROU)	2030 targets: BEL -25% (2005); DNK -55% (1990); DEU -31-34% (1990); FRA -18% (2015); IRL -22-30% (2018) PRT -11% (2005)	19 out of 27 countries (except AUT, CZE, HUN, LVA, LTU, POL, ROU, SVK)
Iceland	-55% (1990)	"Largely neutral" by 2040	Yes	None	Yes
India	-45% GDP emission intensity (2005)	Net zero by 2070	No	None	No
Indonesia	-29% from BAU; up to -41% conditional on int. support	Net zero by 2060	Yes	None	Yes
Israel	-27% (2015)	-85% from 2015 levels	No	None	Yes
Japan	-46% (2013)	Net zero	Yes	49.5 $MtCO_2eq$ by 2030	Yes
Kazakhstan	-15% (1990)	None	No	None	No
Korea	-40% (2018)	Net zero	Yes	-27.1% by 2030[1]; -37.7% by 2050 (2018)	Yes
Mexico	-22% (BAU); up to -36% conditional on int. support	None	Yes	-8% by 2030 (BAU)	Yes
New Zealand	-50% (2005)	Net zero except methane	Yes	-24-47% reduction in biogenic methane by 2050	Yes
Norway	-50-55% (1990)	-90-95% (1990)	Yes	Voluntary agreement with agricultural sector: -5 $MtCO_2eq$ by 2030	Yes
Philippines	-2.7% (2020); up to -75% conditional on int. support	None	No	-29.4% by 2030 (BAU) conditional on int. support	Yes
Russia	-30% (1990)	Net zero by 2060	No	None	No
South Africa	350-420 $MtCO_2eq$ (BAU 398-614 $MtCO_2eq$)	None	Yes	None	No
Switzerland	-50% (1990)	Net zero	Yes	-40% by 2050 (1990)	Yes
Türkiye	-21% (BAU)	Net zero by 2053	No	None	No
Ukraine	-65% (1990)	Net zero by 2060	Yes	None	Yes
United Kingdom	-68% (1990)	Net zero	Yes	-17-30% by 2030;	Yes

	Economy-wide emissions reduction targets		Long-term strategy submitted to UNFCCC	Agriculture-specific target (base year/level)	Global methane pledge (reduce global CH_4 -30% from 2020 levels by 2030)
	2030 target (base year/level)	2050 target			
				-24-40% by 2035 (2019)	
United States	-50-52% (2005)	Net zero	Yes	None	Yes
Viet Nam	-9% (BAU); up to -27% conditional on int. support	Net zero	No	-20% every 10 years	Yes

Note: 1. Set for Agriculture, livestock farming and fisheries.
Note by the Republic of Türkiye: The information in this document with reference to "Cyprus" relates to the southern part of the Island. There is no single authority representing both Turkish and Greek Cypriot people on the Island. Türkiye recognises the Turkish Republic of Northern Cyprus (TRNC). Until a lasting and equitable solution is found within the context of the United Nations, Türkiye shall preserve its position concerning the "Cyprus issue".
Note by all the European Union Member States of the OECD and the European Union: The Republic of Cyprus is recognised by all members of the United Nations with the exception of Türkiye. The information in this document relates to the area under the effective control of the Government of the Republic of Cyprus.
Source*:* Adapted from OECD (2022[18]).

Table A A.5. List of policy instruments to mitigate GHG emissions

Policy instrument	Application to AFOLU sector	Observed impacts
	Market based instruments	
GHG tax	Canada: Fuel charge at federal level, tax on fossil fuels in British Columbia, Output-Based Pricing System.	Modest due to its limited coverages
Emission trading systems (ETS)	New Zealand (horizon 2025): market price applied per farm (CH4) and fertiliser tax applied to industry (N2O)	TBD- transition phases with pilot of a sophisticated system of monitoring of emissions
Abatement subsides	Emission reduction fund (ERF) in Australia (auctioned emission credits)	Observed progress but disputable efficacy
Carbon offsets	Alberta and Quebec, soon Canada, California, China (potentially linked to the respective emission trading systems)	Observed progress with private buyers
	Agricultural support, grants, and preferential credits	
Agricultural support	Common Agricultural Policy in the European Union (EU), Canada and other OECD countries	Insufficiently studied effect (3.5% of reduction according to one study)
Forestation programmes	Ireland, New Zealand, China (Grains for Green)	Varying efficacy, observable forest area growth in some cases
Grants	United States (biogas), China (fertilisers), Australia (energy)	–
Preferential credits	Brazil (ABC program)	Growing effect, about to reach its objectives
REDD+ (payments linked to land use)	Several developing countries are developing their strategies	–
	Environmental regulations	
Deforestation regulation	Brazil (Forest code) and Indonesia (Forest-clearing ban)	Implementation issues
Pollution regulations	Nitrates Directive and pollution control (EU)	Potentially effective, but not systematically implemented
	R&D and information approaches	
R&D	Many countries– Global Research Alliance	Contribution to GHG monitoring and mitigation practices
Knowledge transfer	Ireland, France, and others	Increases the adoption of sustainable practices

Source: Adapted from Henderson, Frezal, and Flynn (2020[6]).

Table A A.6. Farm-level impacts of GHG tax

Tax rate (EUR/t)	Farm-level GHG emission reduction (%)				Farm-level net profit change (%) (profit before tax in parenthesis)			
	Farm A (high milk and low crop yield)	Farm B (low milk and low crop yield)	Farm C (low milk and high crop yield)	Farm D (high milk and high crop yield)	Farm A (high milk and low crop yield)	Farm B low milk and low crop yield)	Farm C (low milk and high crop yield)	Farm D (high milk and high crop yield)
9	6.19	54.29	0.16	16.67	–10.33 (–6.01)	–57.51 (–55.37)	–3.67 (–0.003)	–17.9 (–14.45)
30	64.37	54.86	0.51	63.59	–73.81 (–68.34)	–62.46 (–55.43)	–12.22 (–0.03)	–62.75 (–57.74)
50	64.77	55.36	20.78	63.98	–77.43 (–68.42)	–67.13 (–55.53)	–35.83 (–19.66)	–66.08 (–57.81)

Source: Adapted from OECD (2019[31]).

Table A A.7. List of existing studies assessing the leakage rates of GHG mitigation policies

Study	Model	Policy	Scenario	Regional coverage	Net GHG emission reduction	Leakage rate	Agricultural output loss
Study assessing agricultural sector-specific impacts:							
Henderson and Verma (2021[30])	CGE (MAG-NET)	Tax on non-CO2 emission from agriculture	Tax rate of USD40/Mt (2020–2030), USD60 (2030–2040) and USD100 (2040–2050); impacts in 2050	OECD, Brazil, and China	605Mt	21%	2%
				OECD	268Mt	31%	3%
				Australia, NZ, Northern Europe and Canada	60Mt	57%	10%
			Same policy but without access to abatement technology; impacts in 2050	OECD	83Mt	64%	–
OECD (2019[29])	CGE (MAG-NET)	Tax on non-CO2 emission from agriculture	Same tax rate as OECD (2021); impacts in 2050	Global	2 706Mt / 23%	0%	–
				OECD	235Mt[1]	34%	–
		Tax on non-CO2 emission from ruminants & fertiliser	Same tax rate as OECD (2021); impacts in 2050	Global	301Mt	0%	–
				OECD countries	46Mt[1]	22%	–
		Abatement payment for non-CO2 reduction in agriculture	Provide the same marginal abatement incentives as tax; impacts in 2050	Global	1 330Mt	0%	–
				OECD	223Mt[1]	0%	–
Barreiro-Hurle et al. (2021[32])	CAPRI	F2F and BDS targets & CAP 2014–2020	Meet the several policy targets in F2F and BDS & agricultural support in CAP; impacts in 2030	EU	15% (Non-CO2)	66%	15% (Cereals) 15% (Oilseeds) 10% (Dairy) 14% (Beef)
					20% (GHG)	–	

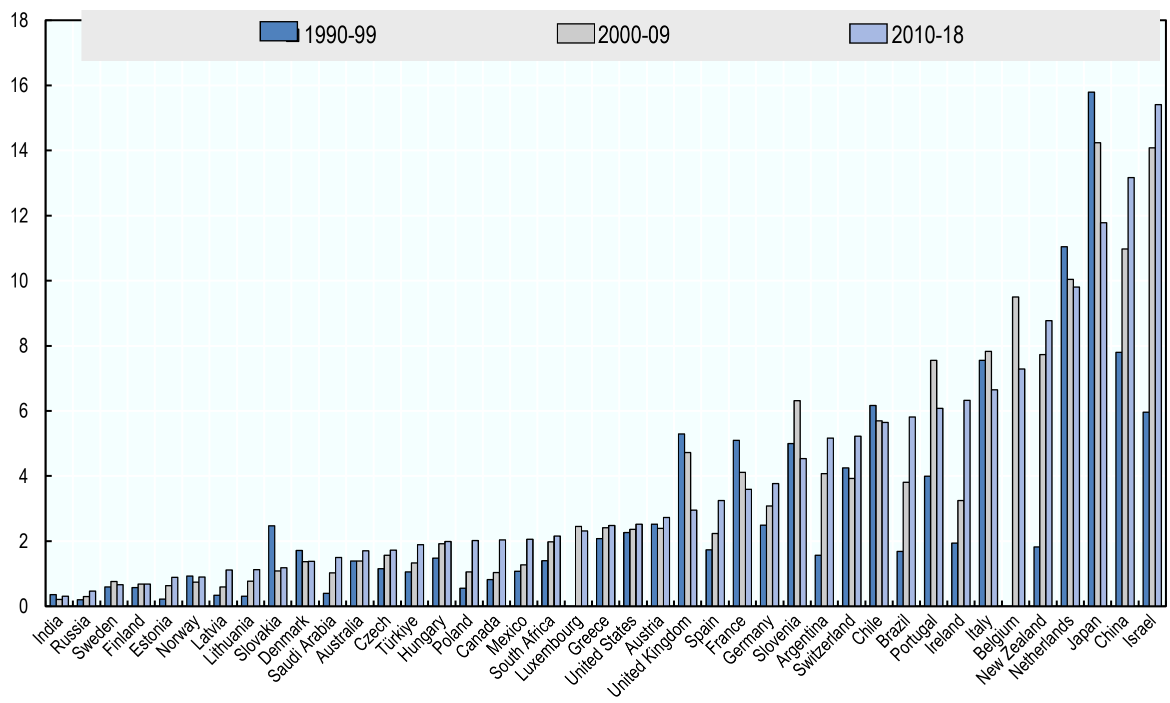
Study	Model	Policy	Scenario	Regional coverage	Net GHG emission reduction	Leakage rate	Agricultural output loss
		F2F and BDS targets & Post 2020 CAP	Meet the several policy targets in F2F and BDS & agricultural support in CAP LP; impacts in 2030	EU	17% (Non-CO_2)	51%	15% (Cereals) 15% (Oilseeds) 10% (Dairy) 14% (Beef)
					28% (GHG)	–	
Henning and Witzke (2021[33])	CAPRI	F2F and BDS targets	Meet the several policy targets in F2F and BDS; addition to the EU-ETS	EU	37.5% (N_2O)	–	Output loss and net import increase 21; 130% (Cereals) 20; 29% (Oilseeds) 6.3; 17% (Milk) 20; 102% (Beef)
					22.7% (CH_4)	–	
					29% (GHG)	50%	
Danish Economic Council (2020[26])	CGE	Tax on GHG emission to all sectors	Tax rate to meet the 2030 policy target of 70% GHG reduction compared to 1990.	Denmark	70%	–	25% employment loss in agriculture

Notes: 1. Net Non-CO2 emissions accounting for leakages. Annex I country– Australia, Austria, Belarus, Belgium, Bulgaria, Canada, Croatia, Czech Republic, Denmark, European Union, Estonia, Finland, France, Germany, Greece, Hungary, Iceland, Ireland, Italy, Japan, Latvia, Liechtenstein, Lithuania, Luxembourg, Monaco, the Netherlands, New Zealand, Norway, Poland, Portugal, Romania, Russia, Slovak Republic, Slovenia, Spain, Sweden, Switzerland, Türkiye, Ukraine, the United Kingdom, the United States.

Table A A.8. Summary of environmental and health concerns of pesticide use

Adverse effects	Details	Evidence
Environmental concerns		
Biodiversity loss	Population reduction of key species in ecosystem; Birds are beneficial pest predators; Loss of bee colonies result in pollination losses	Study in the Netherlands found neonicotinoids exposures caused increase in bird loss (Hallmann et al., 2014[287]); 75% of honey samples across the world were found to contain neonicotinoids (IPBES, 2016[38])
Soil and water pollutions	Threaten soil and aquatic organisms and may indirectly affect human health	About a half of rivers and lakes in Europe contained high level of pesticides that could harm aquatic organisms (Malaj et al., 2014[288])
Health concerns		
Food contaminations	Inappropriate or excessive pesticide residues could adversely affect human health	137 out of 216 breads sold at supermarkets in the UK found to contain more than one pesticide residue (PAN, 2013[289])
Non-fatal pesticide poisoning	Exposures are associated certain types of cancers (lung and breast), cognitive and neurodevelopmental disorders, and reproductive and endocrine disruptions	Non-fatal pesticide poisoning is estimated to cause 3 million hospitalisations, 220 000 deaths and 750 000 chronic illness at global scale (Hart and Pimentel, 2002[290])
Acute pesticide poisoning	Particularly the case in developing countries where some toxic pesticides are	Worldwide UAPP cases are about 385 million and they cause 11 000 annual deaths (Boedeker et al., 2020[291])

Figure A A.2. Pesticide use in OECD and G20 countries (kg per ha of cropland)



Notes: Some countries are excluded due to data availabilities and outliers. Pesticide use does not account for concentration or toxicity and is therefore not a good indicator of pesticide risks.
The statistical data for Israel are supplied by and under the responsibility of the relevant Israeli authorities. The use of such data by the OECD is without prejudice to the status of the Golan Heights, East Jerusalem and Israeli settlements in the West Bank under the terms of international law.
Source: FAOSTAT (2021[292]) data.

Table A A.9. Policy instruments used to limit the environmental and health effects of pesticides

Policy Instrument	Description	Example
Authorisation processes		
Approval	Approval and usage standard for specific pesticide	Registration of new pesticide and setting its standard
Renewal	Change in the regulation on (dis)approved pesticides	Adoption of more stringent standard on farm-level pesticide use
Ban	Disapproval of certain pesticide	Restriction of DDT use in agriculture
Pesticide use regulations		
Applications	Regulation on the application of pesticides	Bans of aerial spraying
Distance limits	Limitation of sprays in certain geographical areas	Regulation of distance with water courses or houses
Storage	Regulations on storage of pesticide substances	Regulatory requirements on containers or storage location
Economic incentives		
Tax	Financial charge on per unit pesticide use, reduction of tax exemptions	Tax on wholesale value
Agri-environmental subsidies	Compensation for reduced pesticide use	Payment to producers who halved the pesticide use
Information measures		
Training	Share of knowledge and management practices to reduce pesticide	Provision of information of non-chemical measures to control pests
Certificate	Compulsory disclosure of information on pesticide usage	Pesticide labels
Advisory service	Voluntary share of information for farmers' decision on pesticide use	
Food regulations		
Safety standards	Food and feed safety related standards	Maximum residue limits (MRLs)

Note: Pesticide use does not account for concentration or toxicity and is therefore not a good indicator of pesticide risks.

Table A A.10. Pesticide tax schemes in selected countries

Country	Tax base	Tax rate	Year introduced
Denmark	Active ingredients in pesticide Pesticides	The differentiated tax on plant protection products is based on environmental and human health effects of the plant protection products. This will mean I differentiated tax mainly depending on the constituents of each plant protection product and risk for human health and the environment. The tax rates are at present: 113 DKK/kg or litre active substance multiplied by the score of the environmental effect 113 DKK/kg or litre active substance multiplied by the score of the environmental fate and behaviour effect 113 DKK/kg plant protection product multiplied by the score of the human health effect	1996 (1986–1996 3% tax) 2013 – introduction of the differentiated pesticide tax
France	Pesticides	EUR 5.1per kg – substances very toxic, toxic, carcinogenic, mutagenic toxic to reproduction EUR 2 per kg – substances hazardous for environment EUR 2 per kg – mineral chemical hazardous for environment	2008 (1999–2008 TGAP)
Italy	Pesticides	2% of wholesale value	2001
Norway	Pesticides	Banded tax system, determined by a complex formula	1988
Sweden	Pesticides	EUR 3.64 per kg of volume sold	1984
Florida (US)	Pesticide	USD 0.0500 (EUR 0.0452) per barrel of sales	
Maine (US)	Aquatic pesticides	USD 200.0 (EUR 180.8) per year	2020
Washington State (US)	Pesticide	0.7% of wholesale value	

Source: OECD (2021[283]) data and OECD (2017[63]).

Table A A.11. MRLs of selected pesticides in different countries – grain maize and apples

Pesticide	Codex	EU	Australia	Canada	Japan	India	US
Maize/corn							
2,4-D	0.05	0.05	0.2	0.05	0.05	0.01	0.05
Atrazine	–	0.05	0.1	0.2	0.2	–	0.2
Carboxin	–	0.03	0.1	0.2	0.2	–	0.2
Chlorpyrifos	0.05	0.01	0.1	0.05	0.05	0.05	0.05
Deltamethrin	2	2	2	–	2	0.5	1
EPTC	–	0.01	0.04	0.1	0.1	–	0.08
Fluoride ion	–	2	–	–	–	–	10
Glyphosate	5	1	5	3	5	5	5
Malathion	0.05	8	8	8	2	4	8
Piperonyl Butoxide	30	–	20	20	24	30	20
Prothioconazole	0.1	0.1	0.3	0.35	–	0.1	0.35
Spinosad	1	2	1	1.5	2	1	1.5
Tefluthrin	–	0.05	–	0.06	0.1	–	0.06
Zeta-Cypermethrin	0.3	0.3	1	0.05	0.2	0.3	0.05
Apples							
Acetamiprid	0.8	0.4	0.2	1	2	0.8	1
Captan	15	10	10	5	15	15	25
Chlorantraniliprole	0.4	0.5	1.2	0.4	1	0.4	1.2
Deltamethrin	0.2	0.2	0.05	0.4	0.3	0.2	0.2
Fenbutatin oxide	5	0.01	3	3	5	5	15
Fludioxonil	5	5	5	5	5	5	5
Hexythiazox	0.4	1	1	0.4	0.7	0.4	0.4
Malathion	0.5	0.02	2	2	0.5	4	8
Oxamyl	–	0.01	–	–	2	–	2
Permethrin	2	0.05	–	1	2	2	0.05
Pyrethrins	–	1	1	1	1	1	1
Spinosad	0.1	0.3	0.5	0.2	0.5	0.1	0.2
Thiacloprid	0.7	0.3	1	0.3	2	0.7	0.3
Thiram	5	5	3	7	5	3	5
Ziram	5	0.1	3	7	5	5	7

Source: Codex– Codex (2021[293]), EU–European Commission (2021[294]), Australia– Australian Government (2021[295]), Canada– Health Canada (n.d.[296]), Japan–The Japan Food Chemical Research Foundation (2021[297]), India– Food Safety and Standards Authority in India (2018[298]), US–Federal Government of the United States (2021[299]).

Table A A.12. List of studies estimating effects of banning specific pesticide

Study	Model	Policy	Detail	Crop	Competitiveness change	Leakage rate
Lynch et al. (2005[56])	CGE	Ban of a fumigant (MB: Methyl Bromide) in the United States	Farmers in California and Florida are main MB users; Mexico is a competitor; Alternative substance considered	Strawberry	Planted acreage: –41.7% (CA); –8.5% (FL) –32.7% (US); 1021.6% (MX)	11.5% (limited) US: 10330 Mt to nil MX: 117Mt to 1309Mt
				Tomato	–12.5% (CA); –5.3% (FL) –5.0% (US); 28.6% (MX)	
				Pepper	–7.3% (FL); –7.6% (US) 40.9% (MX)	
			Assume MB is 20% cheaper in Mexico			60.9% MX: 117Mt to 6404Mt
Garcia-German et al. (2014[66])	Monte-Carlo simulation	Ban of a herbicide (Pendimethalin) in Europe	Spanish farmers are considered; No alternative herbicide; Hand weeding as an alternative technology	Lettuce	Yield: –7.9% Revenue: –165%	NA
				Celery	Yield: –16.8% Revenue: –45.9%	NA
Zilberman et al. (1991[64])	CGE	Ban of a several pesticides in California	Ban of pesticides related to cancer risks was considered; no alternative substance for 30% of the pesticides	Almond	Output: –15% Revenue: –0.2%	NA
				Grape	Output: –19% Revenue: –3.4%	
				Lettuce	Output: –9% Revenue: 15.6%	
				Orange	Output: –21% Revenue: –11.4%	
				Strawberry	Output: –25% Revenue: –12.0%	

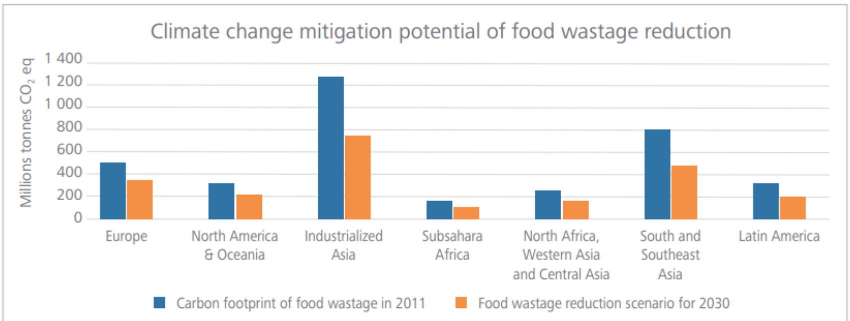
Notes: Competitiveness effects show the average impact for the considered sample. CA– California, FL–Florida, MX–Mexico.

Figure A A.3. Climate change mitigation potential of FLW reduction

Assumptions for food wastage reduction ratios achievable by 2030
Phases "Agricultural production" and "Processing"
• 5% reduction of 2011 food wastage in developed countries • 15% reduction of 2011 food wastage in developing countries (a larger progress margin is assumed for developing countries)
Phase "Post-harvest handling and storage"
• 5% reduction of 2011 of food wastage in developed countries • 54% reduction of 2011 food wastage in developing countries (reduction estimated to be needed to reach the average percentage of wastage observed in developed countries for most commodity groups)
Phases "Distribution" and "Consumption"
• 50% reduction of 2011 food wastage amounts in all regions

The proposed scenario would lead to a reduction of the carbon footprint of food wastage by 38%, or 1.4 $GtCO_2$ eq per year (see chart below); this would be equivalent to the GHG emissions of the Japanese economy. Considering that post-harvest handling reductions are feasible in developing countries through improvements in their food systems (e.g. adopting improved technologies, better handling practices, efficient markets), investment in reducing post-harvest losses represent an important climate mitigation strategy.

Despite data and modelling uncertainties, the magnitude of the figures above suggest that a reduction of food losses and waste at global, regional and national levels would have a substantial positive effect on societal resources and in particular, climate change.



Source: FAO data (Scialabba, 2015[215]).

Table A A.13. Observed roles of government in environmental labelling and information schemes (ELIS)

Institutional role	Identified specific role	Observed examples in OECD countries
As ELIS supplier	1. Setting Standard 2. Managing Standard 3. Certification 4. Promotion	Public ELIS examples Type I labels
As ELIS user	5. Public procurement 6. Regulatory fulfilment	EU procurement program EU biofuel directive
With supporting institutions	7. Member of group 8. Contributing to activities of the group 9. Funding activities of the group 10. General awareness and education	GEN membership, Mutual recognition of organic schemes, Agencies funding IFOAM. General consumer education programs
With inventorying institutions	11. Lead public inventory 12. Promoting inventories and guides 13. Funding activities	Danish guidance forbrug.dk Support of ITC Standard Map, Ecolabel.be in Belgium.
With policy support institutions	14. As part of international organisations 15. Leading analysis or dialogue internally 16. Funding analysis or dialogue externally 17. Funding academic research	UNFSS, OECD, others Ademe (France) ENTWINED (Sweden), PEF project (Germany) EU Research Frameworks,
With platforms and consortiums	18. Member of a platform 19. Funding a platform	Product Sustainability Forum (UK) Global Report Initiative (IADB)
As framing institutions	20. Regulating and guiding claims 21. Regulating labels or information systems 22. Obligatory disclosure 23. Contributing to international guidelines and standard-setting 24. Actors in a trade dispute	US Green Codes, France's practical guide on environmental claims EU Energy label, Korea's Carbon footprint label, France (Grenelle 1 et 2 laws) Canada's EnerGuide ISO members, Contribution to FAO and ITTO guidelines US and Mexico: Dolphin-Tuna WTO disputes

Notes: FAO: Food and Agriculture Organization; ITTO: International Tropical Timber Organization; PEF: Product Environmental Footprinting; IADB: Inter-American Development Bank
Source: Gruère (2013[237]).

Box A A.1. Key provisions of Chapter VI of the OECD MNE Guidelines, on Environment

Establish and maintain a system of environmental management – including (art1):

a) the collection and evaluation of information regarding the environmental, health and safety impacts of their activities

b) establishment of measurable objectives and, where appropriate, targets for improved environmental performance that are periodically reviewed and consistent with relevant national policies and international environmental commitments

c) regular monitoring and verification of progress toward environmental, health and safety objectives or targets.

Provide adequate, measurable, verifiable and timely information on the potential environment, health and safety impacts of the activities of the enterprise; and engage in communication and consultation with the communities directly affected (art 2).

Assess, and address the foreseeable environmental, health, and safety-related impacts associated with the processes, goods and services of the enterprise over their full life cycle. Where appropriate, prepare an environmental impact assessment (art 3).

Not use the lack of full scientific certainty as a reason for postponing cost-effective measures to prevent or minimise damage (art 4).

Maintain contingency plans for preventing, mitigating, and controlling serious environmental and health damage from their operations, including accidents and emergencies; and mechanisms for immediate reporting to the competent authorities (art 5).

Continually seek to improve corporate environmental performance by encouraging such activities as (art 6):

a) Adoption of technologies and operating procedures.
b) Development and provision of products or services that have no undue environmental impacts; are safe in their intended use; reduce greenhouse gas emissions; are efficient in their consumption of energy and natural resources; can be reused, recycled, or disposed of safely.
c) Promoting higher levels of awareness among customers of the environmental implications of using the products and services of the enterprise, including, by providing accurate information on their products (for example, on greenhouse gas emissions, biodiversity, resource efficiency, or other environmental issues).
d) Exploring and assessing ways of improving the environmental performance of the enterprise over the longer term, for instance by developing strategies for emission reduction, efficient resource utilisation and recycling, substitution or reduction of use of toxic substances, or strategies on biodiversity.

Provide adequate education and training to workers in environmental health and safety matters, including environmental impact assessment procedures, public relations, and environmental technologies (art 7).

Contribute to the development of environmentally meaningful and economically efficient public policy, for example, by means of partnerships or initiatives that will enhance environmental awareness and protection (art 8).

Source: OECD (2021[300]).

Box A A.2. Key provisions of Chapter VI of the OECD MNE Guidelines, on Disclosure

Enterprises should ensure that timely and accurate information is disclosed regarding their activities, structure financial situation and performance. This information should be disclosed for the enterprise as a whole, and, where appropriate, along business lines or geographic areas (art. 1).

Disclosure policies of enterprises should include, but not be limited to, material information on, amongst others, foreseeable risk factors, issues regarding workers and other stakeholders, and governance structures and policies (art. 2)

Enterprises are encouraged to communicate additional information that could include (art. 3):

Value statement or statements of business conduct intended for public disclosure including information on the social, ethical and other codes of conduct to which the company subscribes.

Information on systems for managing risks and complying with laws, and on statements or codes of business conduct.

Information on relationships with employees and other stakeholders.

Enterprises should apply high quality standards for accounting, and financial as well as non-financial disclosure, including environmental and social reporting where they exist. The standards or policies under which information is compiled and published should be reported (art4).

Source: OECD (2021[300]).

OECD FOOD, AGRICULTURE AND FISHERIES PAPERS

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Comments are welcome and can be sent to tad.contact@oecd.org.