Enhancing Climate Change Mitigation through Agriculture

OECD

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BETTER POLICIES FOR BETTER LIVES

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Foreword

Climate change is the world's greatest environmental challenge, but progress on delivering a proportionate policy response has been slow and inadequate. Climate change has quickly moved from being a future concern to a present and evident crisis. High temperature records are now routinely broken around the world and the growing frequency of catastrophic climatic events has heightened awareness about the urgency to stabilise global temperatures.

Transforming humankind's means of production and patterns of consumption to lower greenhouse gas emissions are essential to this goal. Recent scientific research, together with growing public awareness, imply that agriculture will need to be at the forefront of global strategies to keep global warming well below 2°C. Much of this rests on the sector's large contribution to climate change; however, concerns about global food security and the importance of agriculture to national economies create policy challenges that are unique to agriculture.

The pricing of greenhouse gas emissions from agriculture, according to the polluter pays principle, is likely be more challenging than it has been for other sectors, despite its efficiency in correcting the market failures responsible for climate change. Much more policy guidance is needed to manage the multiple and often competing objectives of mitigation, and to improve food security and farm incomes. The present research takes an important step in this direction by showing how different mitigation policies can affect these objectives, and which can help countries identify and further develop policy approaches that are suitable to different national circumstances. It relies on multiple economic models, applied at different scales for a broad range of mitigation policy scenarios.

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Abbreviations

2DS	2-Degree Scenario
ABC	Low Carbon Emission Agriculture program (Brazil)
AC-2DS	AGLINK-COSIMO 2-degree scenario
AFOLU	Agriculture, Forestry and Other Land Use
AEOS	Alberta Emission Offset System (Canada)
CARB	California Air Resources Board
CCS	Carbon Capture and Storage
CCU	Carbon Capture and Utilisation
CGE	Computable general equilibrium
CH_4	Methane
CO_2	Carbon Dioxide
EPA	Environmental Protection Agency (United States)
ERF	Emissions Reduction Fund (Australia)
ETS	Energy Technology Perspective (IEA)
ETS	Emissions Trading Scheme
EU ETS	EU Emissions Trading System
EV	Equivalent Variation
FAO	United Nation's Food and Agriculture Organisation
GHGs	Greenhouse Gases
GTAP	Global Trade Analysis Project
GTAP-E	GTAP Energy-Environmental Database
$GtCO_2eq$	Gigatonnes of equivalent carbon dioxide
IIASA	International Institute of Applied System Analysis
IEA	International Energy Agency
IPCC	Intergovernmental Panel on Climate Change
LDCs	Less Developed Countries
LUC	Land Use Changes
LULUCF	Land Use, Land-Use Change and Forestry
MACs	Marginal Abatement Costs
MAGNET	Modular Applied GeNeral Equilibrium Tool
MENA	Middle East and North Africa
MRV	Measurement Reporting and Verification
NAMA	National Appropriate Mitigation Action (Brazil)
N_2O	Nitrous Oxide
NDC	Nationally Determined Contributions
OECD	Organisation for Economic Co-operation and Development
PCF	Pan-Canadian Framework
SCS	Soil Carbon Sequestration
UN	United Nations
UNFCCC	United Nations Framework Conventions on Climate Change
WTW	Well-to-Wheel

Executive Summary

Agriculture is one of the main sectors responsible for climate change. Between 2007 and 2016, the sector directly contributed approximately 12% of global anthropogenic greenhouse gas (GHG) emissions (6.2 ± 1.4 $GtCO_2eq$), and was responsible for an additional 9% of global GHG emissions each year (4.9 ± 2.5 $GtCO_2eq$) from changes in land use, i.e. the conversion of forestland to cropland and grassland.

The collective global effort to mitigate GHG emissions in the agricultural sector has been weak. A continued lack of progress could lead to direct and indirect emissions from agriculture, becoming the largest source of global emissions by mid-century as more rapid decarbonisation in other sectors is anticipated. With growing recognition about the importance of agricultural emissions, it is imperative to assess how alternative policies in the agricultural sector could contribute to ambitious global mitigation efforts and to build knowledge about their potential socio-economic impacts.

The present research assesses the potential of different policies and options to unlock the large mitigation potential of the agricultural sector, while quantifying the economic impacts of these policies on agricultural producers and food consumers. To perform these assessments, two global models and one farm-scale model are used, and the findings are discussed within the context of the existing literature on the global mitigation potential of the agricultural sector.

Given that climate change reflects a failure of markets, whereby emitters do not pay for damages to others, pricing all global GHG emissions according to the polluter-pays-principle is in principle the most economically efficient approach to limiting global warming. For the agricultural sector, the analysis shows that a global emission tax that prices carbon at USD 40-60 per tonne of equivalent carbon dioxide per year (tCO_2eq[1]) could reduce annual non-CO_2 emissions by about 0.85 gigatonnes of equivalent carbon dioxide ($GtCO_2eq$) by 2030. The overall mitigation potential increases to 1.4 GtCO2eq once the induced conversion of agricultural land to forests is taken into account. However, despite their efficiency, the economic burden that such carbon pricing policies could place on some agricultural producers and consumers can make them politically difficult to introduce. Further, much of the mitigation from an emissions tax would be driven by the reallocation of production away from *more* towards *less* emission-intensive sectors and regions. While this lowers the overall emission intensity of agricultural production, it could cause a large decline in ruminant production in developing countries, raising concerns about food security among poor producers and consumers.

A limitation when carbon pricing policies are applied by a single country or small groups of countries stems from the fact that producers subject to such policies would lose their competitiveness relative to producers in countries where these policies are not applied. Hence, some of the reductions in agricultural emissions by countries that apply carbon pricing policies may “leak” in the form of higher emissions in countries that do not apply such policies. The present research suggests that about one-third of the mitigation resulting from a tax applied only to OECD country emissions would be leaked in other countries not applying such a tax.

Perhaps in response to these drawbacks, the few market-based mitigation policies that have been implemented involve paying farmers to mitigate emissions via either a subsidy or the creation of an offset market. While these policies largely avoid imposing costs on producers, inflating food prices, and creating emission leakages, they are less effective in lowering emissions than policies that apply the polluter-pays-principle, partly because they do not disincentive production. Mitigation is further limited if abatement payments support low emission practices, but do not support a switch of production to lower emission commodities. For these reasons, a globally applied abatement subsidy was found to be about half as effective at lowering agricultural emissions for the same carbon price. Furthermore, if payments to farmers are offered via a subsidy, the resulting increase in taxes could reduce economic welfare compared to polluter-pays policies. Although if such payments were funded by changing the focus of market distorting support payments to agriculture, they could improve economic welfare relative to the status quo.

Another important challenge given the large number and heterogeneity of farms is to better measure and verify mitigation efforts. Levying taxes on high emission inputs, instead of targeting emissions more directly, would be one option to meeting this challenge. This was, nevertheless, found to be far less effective and cost-effective than policies that targeted emissions more directly, even after considering their transaction cost savings. This is because taxes that target emissions can incentivise the deployment of all available mitigation measures, whereas taxes on inputs encourage only the reduction of emission intensive inputs.

Growing attention is also being given to the very large potential of demand-side mitigation options (0.7-8.6 $GtCO_2eq\ yr^{-1}$ by 2030), including measures that encourage consumers to shift to lower emission diets and reduce food waste. However, the potential of policies to achieve this is unknown and is likely to be much smaller than the more optimistic estimates cited in the literature, which typically ignore the costs associated with reducing food waste.

The agriculture sector can also potentially contribute to global GHG mitigation by supplying biofuels derived from food and feed to the transport sector. However, this is expected to play a minor role only in reducing such emissions due to constraints on the availability of feedstocks and policy settings that are unlikely to encourage further expansion at the expense of food production. Consideration of the effects of land use change would likely further lower this minor potential.

The importance of sending clear and consistent policy signals to the agricultural sector cannot be overstated as the high levels of support to agriculture in many countries are likely to counteract the effectiveness of mitigation policies in many instances, raising concerns with regard to policy coherence. Clear signals are also necessary to allow farmers to make investment decisions that can facilitate the transition to low carbon agriculture, particularly in farming systems with high fixed investment costs.

1 Potential for mitigation policies in agriculture: Summary insights

The global ambition to reduce greenhouse gas (GHG) emissions in agriculture is currently weak, and the lack of progress will stifle efforts to meet the goals of the Paris Agreement to limit global warming to 1.5°C, or well below 2°C. Although there are policy implementation barriers, policy solutions exist. These include selecting policy options that can navigate trade-offs in economic impacts between different interest groups, and those that can address the practical challenges and transaction costs related to measurement, reporting, and verification of GHG emission reductions.

The need to reduce agricultural emissions

Agriculture continues to contribute substantially to climate change by directly emitting non-carbon dioxide (non-CO_2) emissions, including methane (CH_4) and nitrous oxide (N_2O), from crop and livestock production, and by affecting net CO_2 emissions from agricultural soils, forestry and other land use. Average annual emissions from agriculture amounted to 6.2 ± 1.4 $GtCO_2eq$ of GHG emissions, between 2007 and 2016, representing approximately 12% of global anthropogenic GHGs. There were a further 4.9 ± 2.5 $GtCO_2eq$ of average annual emissions from land use change caused by agriculture during this period, contributing a further 9% to global emissions (IPCC, 2019[1]).

Developing countries are, in general, the largest and fastest growing source. Between 1990 and 2014, they were responsible for the 15% increase in global non-CO_2 emissions from agriculture (Blandford and Hassapoyannes, 2018[2]), while OECD countries as a whole experienced a slight reduction in non-CO_2 emissions over this period. Production efficiency improvements have helped contribute to this reduction by lowering the emission intensity of agricultural output (McLeod et al., 2015[3]). However, the rate of decline in intensity appears to be slowing down. Net CO_2 emissions from forestry and other land use[1] have fallen in both developed and developing countries due to progress on deforestation rates and increased afforestation in several regions of the world (Blandford and Hassapoyannes, 2018[2]; Smith et al., 2014[4]).

A recent Intergovernmental Panel on Climate Change (IPCC) Special Report on Global Warming of 1.5ºC confirms there is an important role for land use sectors in stabilising global temperatures (IPCC, 2018). Four broad options could be implemented in the agriculture sector to mitigate GHG emissions. The first two encompass supply-side measures and the latter two cover demand-side measures:

- Introduce farm practices that reduce agricultural non-carbon dioxide (non-CO_2) emissions; including methane (CH_4) and nitrous oxide (N_2O).
- Introduce practices to remove CO_2 from the atmosphere and accumulate as carbon in vegetation and soils, or that reduce emissions from the degradation and removal of these carbon stocks
- Introduce measures that encourage consumers to shift to healthier, lower emission diets.
- Introduce measures that reduce product losses along food supply chains and food waste by consumers.

This chapter is primarily concerned with the mitigation potential from measures and policies to reduce agricultural non-CO_2 emissions and net CO_2 emissions from grassland and cropland soils. Reductions in net CO_2 emissions from avoided deforestation and afforestation are reported under forestry and other land use in the IPCC Guidelines for National Greenhouse Gas Inventories (IPPC, 2008[5]). While these do not count as reductions in agricultural emissions, agriculture is a driver of deforestation and there are opportunity costs associated with the use of land for forestry instead of agricultural production. Given these interactions, some policies that affect land use change are also included in the overview of mitigation policies implemented by countries.

OECD research presented in Chapter 5 shows that biofuels derived from food and feed are expected to play a minor role only in climate change mitigation, in particular because the supply of ethanol and biodiesel is constrained by the availability of feedstock and rising agricultural production costs. Consideration of the effects of land use change would most likely decrease this minor potential. Yet according to the IPCC Fifth Assessment report, bioenergy could play an important role in mitigating emissions, although “the scientific debate about the overall climate impact related to land use competition effects of specific bioenergy pathways remains unresolved” (Smith et al., 2014[4]).

Technical and economic potential for supply-side mitigation in agriculture

The mitigation potential of supply-side options in the agriculture sector can be decomposed into four components: technical, economic, market, and socially/politically constrained potential (Figure 1.1). The technical potential is defined as the maximum mitigation possible in the sector with the full implementation of all available supply-side mitigation options, ignoring all barriers to adoption. The economic potential takes the costs and benefits associated with different mitigation measures into account, indicating what can be achieved for a given carbon price. Institutional capacity constraints, political and social barriers related to the distributional impacts of policy options can also erode the potential of supply-side mitigation measures. Finally, these barriers along with practical implementation challenges, particularly those related to measurement reporting and verification (MRV) of emission reductions, combine to markedly lower the “market” potential for mitigation. That is, the collection of mitigation measures that farmers find worthy of adopting, given the existing incentives and constraints they face. This section is mainly concerned with the technical and economic potential to mitigate agricultural GHG emissions.

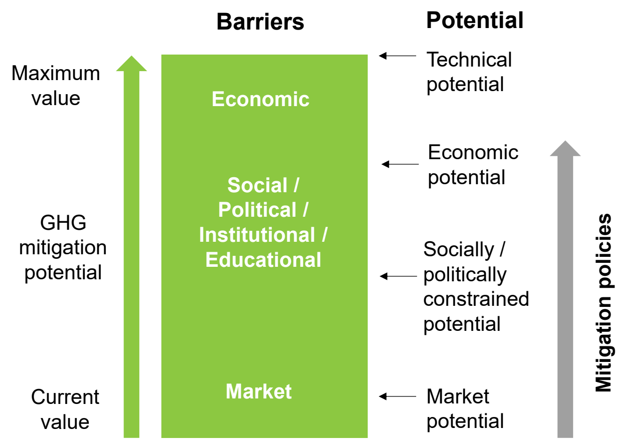
With full deployment of available emission reduction and carbon sequestration opportunities, the global technical mitigation potential of the agricultural sector in 2030 is estimated to be 5 500-6 000 $MtCO_2eq\ yr^1$, with a 95% confidence interval around this mean value of 300-11 400 $MtCO_2eq\ yr^1$ (Smith, 2012). This demonstrates that it is technically feasible for agriculture to become close to carbon neutral, relying on supply-side mitigation measures alone, although this depends on optimistic assumptions about the potential of soil carbon sequestration (SCS).

As shown in Figure 1.1, there are several barriers to the uptake of mitigation measures which will reduce its overall technical potential. The economic potential of these measures is reflected by their potential at a given carbon price, taking into account their costs and benefits. The most recent Assessment Report of the IPPC report, based on results from several studies (Rose et al., 2012[6]; McKinsey & Company, 2009[7]; Golub et al., 2009[8]; Smith, P., et al., 2008[9]), found that emission reductions for agriculture of 0.03-2.6 $GtCO_2eq$ are possible at USD 50 tCO_2eq^{-1}, and 0.2-4.6 $GtCO_2eq$ at USD 100 tCO_2eq^{-1} in 2030 (Smith et al., 2014[4]). This wide range reflects the different coverage of mitigation sources and methodologies used. For example, the higher figures in these ranges include SCS measures.

A recent partial equilibrium assessment by Frank et al. (2018[10]) calculated non-CO_2 mitigation potential in 2030 of 1 $GtCO_2eq$ at USD 25 tCO_2eq^{-1}, and 2.6 $GtCO_2eq$ in 2050 at USD 100/tCO_2eq. Recent assessments presented Chapters 2 and 4 have found comparable economic mitigation potential. As shown in Chapter 2, global non-CO_2 emission reductions of 0.84 $GtCO_2eq$ in 2030 at USD 40 tCO_2eq^{-1} and 2.7 $GtCO_2eq$ at USD 100 tCO_2eq^{-1} found using the MAGNET computable general equilibrium (CGE) model. The increase in carbon stocks from changes in land use (from agriculture to forestry) substantially raised the economic potential, from 0.84 to 1.4 $GtCO_2eq$ in 2030 and from 2.7 to 4.4 $GtCO_2eq$ in 2050. In the global[2] assessment made using AGLINK-COSIMO (Chapter 4), non-CO_2 emission reductions of 0.85 $GtCO_2eq$ in 2030 at USD 60 tCO_2eq^{-1} were calculated.

The global potential for SCS is estimated to be large but highly uncertain. Smith (2016[11]) reports the mean global potential for SCS in agricultural soils as 1.5 Gt $CO_2\ yr^{-1}$ and 2.6 $GtCO_2\ yr^1$, at carbon prices of USD 20/tCO_2eq and USD 100/tCO_2eq, respectively. In addition, the mitigation potential from increasing carbon stocks in vegetation is substantial. For example, Golub et al. (2012[12]) found that a policy package that combined a tax on agricultural emissions and a subsidy to sequester carbon in forest biomass at USD 27/tCO_2eq could mitigate 5.3 $GtCO_2eq\ yr^{-1}$ of emissions. Most of this potential was attributed to the sequestration of carbon in forest biomass, particularly from avoided deforestation and the conversion of forest to agricultural land (i.e. from changes in the extensive margin between forestry and agriculture).

Figure 1.1. Supply-side mitigation potential in the context of barriers to implementation and policy ambition



Source: Adapted from (Smith et al., 2014[4]).

Two requirements are necessary for policy measures to achieve any given level of mitigation at minimum economic cost. The first is that market-based policy instruments that achieve a common price for GHG emissions (such as an emissions tax or emissions trading scheme) be used. The second is that coverage of the market-based policy includes the largest possible share of global emissions from all regions and sectors. Given the large heterogeneity in marginal abatement costs amongst agents, sectors and regions, these two policy requirements will ensure that the lowest cost mitigation measures are adopted. The least cost characteristic of market-based policies stems from the flexibility with which they provide agents to select mitigation measures and technologies that provide them with the lowest net costs (or highest net benefits) for a given carbon price (Baumol and Oates, 1988[13]). Given this flexibility, it is possible it will not be economical to reduce at current or expected carbon market prices mitigation from some agricultural emission sources. According to some studies, mitigation measures that rely on the addition of lipids or nitrates in animal diets to lower CH_4 emissions from enteric fermentation fall into this category (Van Middelaar et al., 2014[14]; Henderson et al., 2015[15]).

The economic mitigation potentials described above are in many respects upper-bound estimates because they ignore several important constraints. These include political constraints related to sensitivities about food security, distributional impact on producers, and emissions leakages which can affect the type and strength of coverage of the policy measures implemented (Wreford, Ignaciuk and Gruère, 2017[16]). In addition, transaction costs, particularly those associated with overcoming complexities for MRV, can be problematic for some sources of mitigation, thereby reducing their cost effectiveness and hindering implementation. Given these constraints, achieving the necessary global uptake of mitigation policies in the agricultural sector to deliver the economic potentials outlined above, is likely to be an enormous political and technical challenge. Policy responses and solutions to address these barriers are elaborated below.

The mitigation potential of demand-side waste reduction measures

Recent research has highlighted the large potential for GHG emission reductions by replacing red meat and milk with less emission-intensive food products in human diets. Smith et al. (2014[4]) report that the potential from such dietary changes and reduced losses in the food supply chain is uncertain, but could mitigate a substantial 0.76-8.55 $GtCO_2eq\ yr^{-1}$ in 2050. Herrero et al. (2016[17]) report a similar mitigation potential of 0.7-7.3 $GtCO_2eq\ yr^{-1}$ in 2050 from a moderation in demand for livestock production, with the potential depending on the size of the moderation and the assumptions on the use of "spared land".

OECD research presented in Chapter 4 has shown that a dietary shift away from ruminant products (a 10% reduction in red meat and milk consumption, offset by a commensurate increase in the consumption of pig and poultry products) could lower emissions by 0.9 $GtCO_2eq\ yr^{-1}$ in 2030. This is substantial, but it is at the lower range of estimates from the literature for this type of measure because of the modest size of the assumed dietary shift and because its scope was limited to changes in non-CO_2 emissions. Large emission reductions of between 0.4 and 0.8 $GtCO_2eq\ yr^{-1}$ were found to be possible if food waste was eliminated by 2030.

Despite the significant potential from these demand-side mitigation approaches, specific policy mechanisms to incentivise dietary adjustments and waste reduction have not been identified. Their policy potential is therefore unknown and likely to be much smaller than the upper range of estimates in the literature. In addition to identifying plausible policy options to reduce food waste, their potential to cause farm income to decrease and impose costs on consumers are typically ignored, and yet their inclusion has the potential to increase food prices and exacerbate farm income losses (Chapter 4).

Policy progress in mitigating GHG emissions originating from agricultural activities

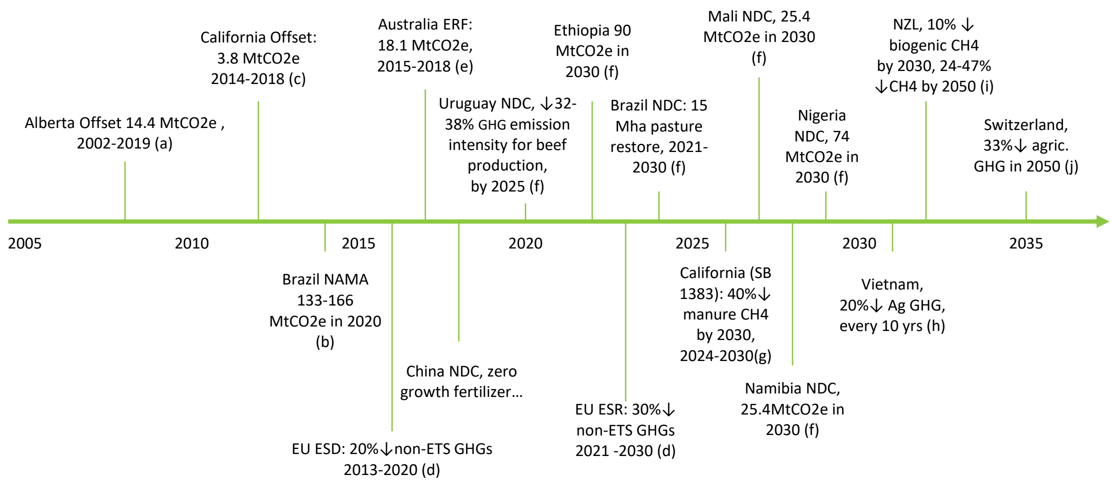
185 states and the European Union have ratified the Paris Agreement and have outlined their commitments to mitigate GHG emissions in their Nationally Determined Contributions (NDCs), submitted under the Agreement. One hundred and three of these NDCs mention agriculture as a contributing sector but, with the exception of a few developing countries, they do not commit to specific targets for that sector. Moreover, progress in implementing concrete mitigation policy incentives and regulations lags behind other sectors, such as energy and transport, including in countries that have implemented or are scheduled to implement national level carbon pricing instruments for GHG emissions (World Bank and Ecofys, 2018[18]; Sense Partners, 2018[19]).

A snapshot of the main mitigation policies and targets to date can be seen in the timeline displayed in Figure 1.2. This non-exhaustive list was selected on the basis of two steps. First, a review of NDCs was conducted, and some countries with ambitious emission reduction targets specific to agriculture were selected. Second, a review of countries was carried out to identify mitigation policies with explicit incentives that either establish a price on emissions or make substantial funds available for investment in mitigation measures in agriculture. For consistency, the figure includes agricultural emission reductions and reduction targets, and excludes emission reductions from forestry and other land use. European Union–wide targets pertinent to agriculture are presented in Figure 1.2, whereas agriculture-specific targets set by some individual Member States, which can contribute to the broader targets, are described below.

Reflecting concerns about imposing costs on producers, emission leakages and MRV challenges, concrete mitigation outcomes have mainly been delivered by a small number of voluntary policies on the basis of paying farmers to mitigate emissions by adopting management improvements that are considered to be more than "business as usual". These include Australia's Emissions Reduction Fund (ERF) and offset schemes within regulated emission reduction systems, such as the Alberta Emission Offset System (AEOS), and the California Air Resources Board (ARB) Compliance Offset Program (CARB, 2019[20]). The ERF is notable for its relatively large government budget and the scale of its emission reductions, the overwhelming share of which have come from vegetation projects that enhance or protect carbon stocks,[3] mostly on farmland (Regulator, 2019[21])(Clean Energy Regulator, 2019). The AEOS offsets are purchased using private funds, with the majority from increased soil carbon sequestration as a result of reduced and zero tillage. These AEOS offsets also include new uses of the Anaerobic Decomposition of Agricultural Materials protocol and the Reducing Emissions from Fed Cattle protocols (AEOR, 2019[22]). The ARB

Compliance Offset Program is narrower in scope, including only protocols for measures to reduce methane from livestock manure and rice production. Policy action to address emissions in California has been substantially augmented with the Senate Bill No. 1383 on Climate Short-Lived Pollutants (2016), which sets the target of cutting dairy and livestock manure methane by 40% by 2030 from 2013 levels (equal to a reduction of about 12 $MtCO_2eq\ yr^{-1}$ in 2030) (Lee and Sumner, 2018[23]).

Figure 1.2. A snapshot of mitigation policy progress in the agriculture sector



Note: Includes a selection of mitigation policies, realised reductions, and national mitigation targets. Within the European Union, several countries have set specific targets and budgets for agriculture to contribute to Effort Sharing goals. An attempt was made to place the policies/targets at midpoint of their timeframes.
Sources: : (a) AEOR, 2019; (b) Gebara and Thuault, 2013; (c) CARB, 2019; (d) European Commission, 2019; (e) Clean Energy Regulator, 2019; (f) UNFCCC, 2019; (g) Meat & Livestock Australia, 2017; (h) CARB, 2017; (i) Bundesamt für Landwirtschaft, 2019.

In countries and jurisdictions where carbon pricing policies have been implemented, they have exempted non-CO_2 emissions from agriculture, including in the EU Emissions Trading System (EU ETS), the New Zealand Emissions Trading Scheme (NZ ETS), the carbon pricing component of Canada's recently introduced Pan-Canadian Framework (PCF) policy, and the California Air Resources Board (CARB) Cap-and-Trade Program.

In its 2015 NDC submission under the Paris Agreement, Brazil pledged to strengthen its Low Carbon Emission Agriculture (ABC) program, including actions to restore an additional 15 million hectares of degraded pastureland UNFCCC (2019). This along with other mitigation measures that were initially set in its National Appropriate Mitigation Action (NAMA) submitted to the UNFCCC in 2010 are mostly incentivised by a substantial line of credit as part of its ABC program. However, the level of progress in meeting the mitigation goals from agriculture in its NAMA are unclear. Brazil's major mitigation ambition comes from its efforts to curb deforestation. Although these do not count as reductions in agricultural emissions, agriculture is a driver of deforestation.

A few other countries have pledged ambitious future mitigation targets specific to agriculture (including Ethiopia and Nigeria), although some of these targets are conditional on external support and details about the policy instruments are unknown at present. Despite the current exemption of agriculture from the NZ ETS, New Zealand recently announced the Zero Carbon Amendment Bill[4] which targets the reduction of all national GHG emissions[5] – with the exception of biogenic methane[6] – to net zero by 2050. The Bill sets separate targets to reduce biogenic methane emissions (which come mainly from ruminant livestock) by 10% by 2030, and to between 24% and 47% by 2050, which is below 2017 levels (Ministry for the Environment, 2019). The New Zealand government is currently considering the policies required to reach these methane reduction targets. In the European Union, the EU Effort sharing legislation sets binding targets for non-ETS sectors (transport,[7] agriculture, buildings, and waste) of 10% by 2020 and 30% by 2030 (European Commission, 2019a). Member States have flexibility regarding the contribution from non-ETS sectors, with some banking/borrowing and trading allowances, as well as the possibility to offset some emissions with reductions from Land Use, Land-Use Change and Forestry (LULUCF) measures.

The targets set under the EU Effort sharing legislation can vary slightly among Member States; while most do not have agriculture-specific targets, there are exceptions. For example, the Netherlands have proposed an emission reduction target of 3.5 $MtCO_2eq$ yr^{-1} by 2030 (~18% of agricultural emissions in 2016), to be achieved through government and industry collaboration and co-funding of mitigation solutions, in their National Climate Agreement (Klimaatakkoord, 2018[24]). Other Member States have established carbon budgets for agriculture. France, for example, has targeted a reduction of GHG emissions of 8% by 2023, 13% by 2028, and 20% by 2033, compared to 2015 levels, in their National Low-Carbon Strategy (Ministère de l'Ecologie, du Développement Durable, 2015[25]). The United Kingdom has also developed carbon budgets with strategic targets for sectors, including a 20% reduction in Agriculture, Forestry and Other Land Use (AFOLU) emissions between 2016 and 2030 (CCC, 2019[26]). Germany has more ambitious reduction goals of 31-34% for agricultural emissions by 2030, compared to 1990 levels, in its Climate Action Plan 2050 (BMUB, 2016[27]). The Climate Change Programme for Finnish Agriculture includes a national reduction goal of 13% for agricultural emissions between 2005 and 2020 (Ministry of Agriculture and Forestry (Finland), 2014[28])). Ireland's Climate Action Plan sets out a decarbonisation pathway to 2030, which is consistent with the adoption of a net zero emission target by 2050.[8] With cumulative CH_4 and N_2O emission reductions of 16.5 to 18.5 Mt CO_2-e between 2021 and 2030, the agricultural sector is expected to deliver 17% of total emission reductions set by this plan. A larger contribution of 26.8 Mt CO_2-e from LULUCF actions, mainly in the forestry sector, is targeted by the Plan over this same period. It is expected that the next CAP, beyond 2020, will be the main driver of emission reductions in agriculture (Ireland, Government of, 2019[29]).

Without direct and strong policy incentives in place to drive GHG mitigation in the agricultural sectors of EU Member States, some of their targets are strategic or aspirational. There are, however, strong regulatory frameworks in place in the European Union for other pollutants, which can have a synergistic impact on mitigating GHG emissions in agriculture, including the Nitrates Directive and the National Emission Ceilings Directive (NEC).

There has been a slow emergence of industry-led initiatives to move ahead with GHG mitigation initiatives in the absence of strong policy action by governments, and in recognition of growing consumer preferences for low-emission products. An example is the Australian red meat and livestock industry's ambitious goal to become carbon neutral by 2030.[9] The National Farmers Union of the United Kingdom announced they would try to achieve "net zero" agricultural emissions from UK Agriculture by 2040 through a combination of improvements in efficiency, increased carbon storage, and bioenergy production (NFU, 2019[30]). Emerging public policy frameworks incentivising mitigation in agriculture could create more favourable returns to such industry-led investments and initiatives. There are nevertheless limitations to all voluntary approaches such as these, and in most countries stronger incentives will be needed to underpin large-scale mitigation ambitions that are commensurate with the targets of the Paris Agreement.

The small number of policies that have been implemented to date have generated useful knowledge on the feasibility and costs of certain practices. The viability of measures based on reducing manure methane emissions from confined livestock operations, such as piggeries and dairy farms, is apparent from their level of enrolment in offset schemes, e.g. Australia's ERF and the California's Offset Credit Scheme. This is not surprising given that manure methane from intensive systems is the closest of all agricultural GHGs to a point source, which tend to be much easier to manage than diffuse sources that dominate the sector's profile of GHG emissions. Thus, the MRV challenges raised earlier have played out to some extent in the types of abatement measures that are permitted and that have been enrolled in these schemes. However, manure CH_4 emissions represent only a small share of agriculture's overall emissions at the global level.

When considering the pledges and policy outcomes of the land sector as whole, it is not uncommon for higher mitigation ambition and outcomes to be found in the forestry component of AFOLU. In Ireland, for example, the state-funded Afforestation Scheme, which incentivises land owners to convert land from agricultural production to forestry has been instrumental in increasing the country's forest cover. Since 1990, as a consequence of this scheme, Ireland is expected to remove a net 4.5 Mt CO_2e per year from the atmosphere over the period 2021-30 (DCCAE, 2017[31]) In general, the forestry sector is more widely cited than agriculture as a contributor to emission reduction targets in countries' NDC commitments (Richards et al., 2016).

In order to evaluate a country's mitigation efforts in agriculture fairly, it is important to take an economy-wide perspective. The European Union may, for instance, lack strong and direct GHG mitigation policy incentives for agriculture but it has committed to an ambitious economy-wide mitigation goal in its NDC, and has a large-scale carbon pricing policy (EU ETS) which covers 45% of its GHG emissions (European Commission, 2019[32]).

In summary, the progress to-date on GHG mitigation policy in agriculture has been uneven across countries, relying on a combination of voluntary policies including beneficiary-pays approaches, green finance, and modest target setting. This amounts to an aggregate global level of policy ambition that is out of step with the agricultural sector's potential to address climate change. Richards et al. (2016[33]) calculate that the mitigation potential of countries providing specific targets for agriculture in their NDCs is 15%. However, given that most countries have submitted agriculture-specific targets, it is not possible to gauge the overall impact of NDCs submissions on global agricultural emissions (Richards et al., 2016[33]). Continued lack of progress in agriculture could stifle efforts to limit global warming, with some model scenarios showing that non-CO_2 emissions from agriculture could become the largest sectoral source of global GHG emissions by mid-century if other sectors succeed in their decarbonisation (Gernaat et al., 2015[34]; Wollenberg et al., 2016[35]). Alarmingly, recent research by Nisbet et al. (2019[36]) shows that methane emissions (the second most important anthropogenic GHG), of which agriculture is a major contributor, have risen much faster than expected. This reduces the timeframe needed to achieve net-zero CO_2 emissions to meet the goals of the Paris Agreement. With most national level targets set for 2030, there is still time for countries to develop more concrete policies for the agricultural sector, but recent analysis by the UN (UNEP, 2018[37]) shows that without full implementation of the NDC commitments global temperature will increase by 3^{o}C by 2100, well in excess of the 1.5^{o}C and 2^{o}C targets of the Paris Agreement.

Possible responses to the mitigation policy challenges for agriculture

Managing the trade-offs between mitigation effectiveness and the distributional impacts of mitigation policies

To mitigate GHG emissions from agriculture as cost effectively as possible, global action and a reliance on market-based mitigation policies are required. The barriers mentioned above – political constraints related to sensitivities about food security, distributional impacts on producers, emission leakages, and the challenges related to institutional capacity and the MRV of emission reductions – need to be addressed to enable the widespread implementation of effective mitigation policies in the agricultural sector. In this section, several policy design options are explored.

OECD research presented in the subsequent chapters, shows that the choice of policy used to mobilise mitigation efforts in agriculture can induce profoundly different trade-offs in terms of impact on mitigation, farm income, food consumption, government finances, cost effectiveness, and overall economic welfare. These policy choices can have very different impacts on the competitiveness of producers in different sectors and regions.

The economic potential described above includes the assumption that a carbon pricing policy is applied to all agricultural emissions. Such policies are based on the “beneficiary pays principle” and are very effective at reducing emissions for a given carbon price. In addition to incentivising the uptake of mitigation measures, much of their effectiveness stems from the contraction of output they induce by reducing profits and causing farms to exit the sector. This is particularly the case for farmers producing emission-intensive commodities, which lack affordable solutions to reduce the bulk of their emissions. This is typical of some biological emission sources in agriculture, such as enteric methane from ruminants. Policies base on the beneficiary pays principle are the most economically efficient policy instruments (Baumol and Oates, 1988[13]), and these adjustments are a necessary part of the process to attain efficiency benefits. However, their distributional impacts can pose significant political challenges. Although many of the same concerns could be expressed for producers of emission-intensive commodities in other sectors of the economy, there are unique sensitivities in several regions that are associated with food production and agricultural development which can amplify the political challenge of securing support.

Where these concerns prevail, market-based policies that are based on the “beneficiary pays principle” can pay for emission reductions either via a subsidy or the creation of an offset market. These policies can provide the same marginal abatement incentives as a GHG tax, while they avoid imposing costs on producers and tend to have negligible impacts on food consumption and prices compared to a GHG emission tax. For these reasons, paying farmers to abate emissions is less of a political challenge than making them pay for their emissions. It is, however, a less effective way to reduce agricultural GHG emissions. As shown in Chapter 2, a global-level abatement payment would be half as effective as a GHG tax for a given carbon price. This is the case because abatement payments do not reduce farm profits and, unlike taxes, do not induce farmers to exit the sector. Indeed, if payments overcompensate farmers for their abatement costs, this can encourage new farmers to enter the sector, thereby reducing the effectiveness of the policy (Baumol and Oates, 1988[13]).

There are other limitations to using a beneficiary-pays policy, least of which is the need to raise funds to finance it and the associated opportunity costs. Where the policy is financed by government, a competitive market-based mechanism such as an auction would be required to deliver cost-effective mitigation outcomes. Unless the payments replace existing distortionary forms of support, they risk reducing the overall economic welfare compared to a GHG tax or emission-trading scheme in countries that have highly supported agricultural sectors. Chapter 2 demonstrates that the economic welfare (or efficiency of economic resource allocation) costs of using additional government funds to pay for a global abatement subsidy would be higher on a per unit of mitigation basis than a comparably priced GHG tax. A potentially more economically efficient option is to redirect existing coupled distortionary forms of farm support for this purpose. The level of funds needed to support a global abatement payment (assuming a carbon price of USD 100 tCO_2eq^{-1}) represents a small proportion of agricultural producer support currently provided by countries for non-environmental purposes. However, given that the level of support among countries is so variable, some countries could easily fund abatement this way, while others could not, thus limiting the widespread applicability of this option. More analysis is needed to assess the welfare impacts and feasibility of this approach to finance abatement payment policies.

There are alternative market-based policy designs that could draw on the strengths of both the GHG tax and abatement payment policies to achieve a more desirable blend of trade-offs. Hybrid tax-subsidy mechanisms that recycle emission tax revenue back to producers to subsidise the adoption of low emission technologies offer a potential compromise (Pezzey, 2003[38]). These type of instruments have been used with some success in Europe to control nitrous oxide and sulphur dioxide emissions from industrial facilities (Millock and Nauges, 2006[39]). There are a range of ETS designs that also merit further attention, including the provision of free but binding permit allocations to agriculture which could help adjust the balance between mitigation effectiveness and the impact on farm incomes.

Policies to improve agricultural productivity have the potential to substantially mitigate emissions without compromising food security, although some types of productivity improvement can have unintended negative impacts on producers. The assessment presented in Chapter 4 shows that a 10% increase in the total factor productivity of agricultural production by 2030 could reduce annual emissions by 330 $MtCO_2eq$. However, this could also cause significant reductions in consumer food prices and agricultural incomes. While productivity improvements are often correctly framed as win-win options for the environment and the economy, their potential impact on some producers should be evaluated.

With respect to the substantial mitigation potential that could arise from changing human diets, the taxing of GHG emissions could facilitate a shift away from emission-intensive food products. It is possible to obtain the mitigation benefits of a GHG tax and maintain baseline food consumption levels by combining this policy with a food consumption subsidy (Chapter 2). However, a significant shift in consumer preferences to less emission intensive diets would be required to achieve the large mitigation potential that has been reported for this measure in the literature. Raising awareness about the climate change, health and other environmental impacts associated with ruminant products could help with this transition and is indeed already be happening in some developed countries. However, such approaches are likely to have a gradual impact and make a significant contribution over the long term only.

Another major policy challenge is that the largest populations of ruminant animals are found in developing regions, including in India and sub-Saharan Africa, where reducing the consumption of ruminant-based food products is likely to adversely impact food security and nutrition.

Mitigation policy options for managing the impacts of leakage

The spectre of emission leakages has been a potential deterrent for countries seeking to implement mitigation policies in agriculture. As with the distributional impacts, policy choice matters a lot with regard to managing trade-offs between leakage and mitigation effectiveness. Scenarios restricting a GHG tax on agricultural emissions in OECD countries, assessed in an *ex ante* global model presented in Chapter 2, showed that more than a third of emission reductions in these countries could be leaked as emission increases in other countries. If the number of countries applying this policy were smaller, the rate of leakage could be even larger. However, results from the *ex ante* modelling literature may overestimate leakage results due to difficulties in representing some policies that restrict market access and trade flows, including sanitary regulations and other non-trade barriers (Grosjean et al., 2016[40]). There are alternative policy options that can address leakage impacts. For example, the global assessment presented in Chapter 2 shows that an abatement payment to reduce emissions could deliver similar reductions to a GHG tax for a given carbon price without inducing emission leakages. The challenge of funding such a payment, however, would need to be resolved.

It is possible that expanding abatement payments to incentivise carbon sequestration on agricultural land could also cause some emission leakage. While most SCS measures should increase long term agricultural productivity and not create any obvious trade-offs with production, some measures such as protecting and restoring degraded peatlands would displace agricultural production. The displacement of production from policies that subsidise an increase in forest and shrub land biomass on agricultural land is more direct and likely to cause larger rates of leakage (Montserrat and Sohngen, 2009[41]). More research is needed to estimate the mitigation potential of such policies, net of these leakage impacts.

The use of a GHG-based tax levied on emission-intensive consumer products (red meat and dairy products) within OECD countries from domestic and imported sources could also eliminate leakage impacts. However, as with all policies that exclude non-OECD countries, this would have a very small impact on global agricultural emissions (Chapter 2). Furthermore, failure to take into account the emission intensities of products from different sources would reduce the economic efficiency of this policy option.

The importance of policy coherence and policy certainty

The absence of policy coherence can hamper the effectiveness of mitigation policies as the agriculture sector is subject to a wide range of regulations and policies which can have intended and unintended effects on its GHG emissions. For instance, subsidies for emission-intensive inputs such as nitrogen fertilisers and fossil fuels can cause agricultural emissions to increase (OECD, 2015[42]). Policies that affect or promote agricultural production can pose further challenges to reducing GHG emissions, as seen recently in the Irish dairy sector: the abolition of the EU milk quota regime in 2015, combined with Ireland's comparative advantage in dairy and policies to increase its national milk production have led to an increase in dairy output and emissions (EPA, 2017[43]). However, dairy output has increased by more than emissions since this time, reducing the GHG emission intensity of the sector's output.

It is important to send clear and consistent policy signals to the agricultural sector. The presence of high fixed investment costs in some production systems such as dairy can significantly lower the effectiveness of mitigation policies (Chapter 3). In the short run, investment costs are sunk and farms will continue to operate as long as market revenues exceed the variable costs of production rather than make new investments in response to mitigation policy incentives. Fixed investment costs are thus likely to slow the transition to lower-carbon agriculture. The transition will take longer where investments are more recent and where their costs are higher. Thus, there is a need for governments to avoid uncertainty in their long-term GHG mitigation objectives and policies so that farmers can make the appropriate investment decisions.

Policy options for MRV and other challenges related to SCS measures

MRV challenges and mitigation policy solutions for agriculture in general

Barriers related to the measurement reporting and verification (MRV) of emission reductions and institutional and education capacity constraints need to be taken into consideration as they limit the mitigation potential of the agricultural sector. The agricultural sector is comprised of a very large number of heterogeneous producers with mostly diffuse sources of emissions. This presents large MRV-related challenges to implementing mitigation policies in the sector, given that a significant proportion of the transaction costs related to MRV are considered to be fixed costs that are invariant to farm size (Bellassen et al., 2015[44]). However, there is a paucity of transaction cost estimates in the literature and the size of the available cost estimates vary widely, from as little as EUR 0.2-0.7 tCO_2eq^{-1} for CDM projects in Latin America to 65-85% of the total costs of credits in an offset scheme in Western Canada (Grosjean et al., 2016[40]). These costs should decrease over time as farmers and agencies learn new procedures and find new ways to minimise the time and resources needed to comply with and administer new policies (Grosjean et al., 2016[40]).

Despite the tendency of MRV challenges and costs to decrease over time, weaknesses in the institutional capacity of many developing countries is a significant constraint for accurate MRV and large-scale policy implementation. Evidence of this is the dominant reliance on IPCC Tier 1 emission factors (IPPC, 2008[5]) to calculate and report national level GHG emissions from agriculture in developing countries. For example, Wilkes et al. (2017[45]) found that 118 of the 140 developing countries they assessed used the Tier 1 approach to calculate enteric CH_4 emissions from ruminant livestock. This simple calculation approach involves multiplying animal numbers default emission factors, which vary by species and region, but not according to feed quality, productivity improvements, and management practices which can lower emission levels. Consequently, it is not possible to reflect emission reductions from mitigation practices other than from the reduction of animal numbers in national GHG inventories, which lowers the recognition governments can gain from implementing mitigation policies in this sector. This occurs despite the fact that over half of developing countries have identified the potential to reduce livestock-related GHG emissions in their communications to the UNFCCC (Wilkes et al., 2017[45]). In contrast, a review of OECD countries' national GHG inventory reports revealed that 33 of 36 OECD countries use more complex Tier 2 or Tier 3 approaches, which are better able to reflect the impact of changes in management on emission levels (IPPC, 2008[5]). Furthermore, some countries may have institutional frameworks linked to administering other environmental policies that can bring down the transaction costs associated with adopting new MRV protocols for new environmental policies (Coggan, Whittten and Bennett, 2010[46]). This is true of the European Union, where existing regulations – such as the National Emissions Ceilings directive, the Nitrate Directive and MRV tools linked to the CAP – can provide synergies to lower the transactions costs of regulating GHG emissions (Grosjean et al., 2016[40]).

The use of emission proxies, which are easier and cheaper than more direct forms of emission measurement, can lower these MRV-related transaction costs. According to the global assessment presented in Chapter 2, applying a GHG-based tax to ruminant animal numbers and quantity of nitrogen fertiliser would be far less effective than policies that directly taxed emissions[10] or that issued payments for emission reductions. Farm-level assessment results also show it is more cost effective to target GHG emissions directly than to rely on simplistic emission proxies, even when transaction costs are accounted for (Chapter 3). A major problem with relying on simplistic emission proxies is that they severely limit the available options for mitigation, which means they require much higher carbon prices and therefore much higher costs to achieve the same mitigation outcomes as policies that target emissions more accurately and directly.

Given that a high share of MRV-related costs are fixed costs that are invariant to farm size (Bellassen et al., 2015[44]), it is possible to reduce some of these costs on a per-farm or per-emission basis with mechanisms to aggregate farms into larger units for MRV purposes.

Policy implementation challenges and solutions specific to SCS

The uncertainty and complexity of measuring some sources of emission reductions, including N_2O emission reductions from soils and SCS, are greater than other sources, introducing stronger trade-offs between MRV accuracy and cost (Grosjean et al., 2016[40]). An inherent MRV challenge for SCS is that the changes in soil carbon are often small relative to the size of the carbon stocks in the soil, and relative to the large area over which these changes occur. In addition, concerns about the permanence of carbon stocks, the finite capacity of soil carbon storage, and difficulties in demonstrating additionality have led to skepticism about the policy potential of SCS measures (MacLeod et al., 2018[47]). However, avoiding CO_2 emissions from the cultivation of soils with high organic matter content and from preventing their degradation through restoration can deliver high rates of mitigation over small areas (Lal, 2004[48]; Smith et al., 2014[4]; Griscom et al., 2017[49]). This may create viable mitigation opportunities despite the challenges.

Permanence is problematic because sequestration can easily be reversed at any point in time by poor soil management (Smith, 2012). Policy solutions are available to deal with this issue. One such approach is the creation of buffer pools to manage the risk of impermanence, whereby projects contribute a share of their offsets (based on the risk of reversal) to the pool, which can then be used to replace unforeseen losses of carbon stocks. Buffer pools were a feature of six of the ten carbon-offset protocols reviewed by Richards and Huebner (2014). In addition, accounting systems that record both carbon gains and carbon losses from storage pools are needed. Finally, policies that place greater value on temporary over permanent carbon sequestration are sometimes favoured as they are politically convenient, but are ultimately inefficient (Gramig, 2011[50]).

Conclusions

Progress on GHG mitigation policy in agriculture has been uneven across countries, relying on a combination of voluntary policies including beneficiary-pays approaches, green finance, and modest target setting. Collectively, they imply an aggregate level of ambition that is out of step with the sector's potential to address climate change. Continued lack of progress will stifle efforts to meet the goals of the Paris Agreement to limit global warming to between 1.5°C and 2°C. The modest assembly of policies and targets are, in some ways, a testament to the policy implementation constraints faced by the sector. However, the evidence reported here demonstrates there are policy design solutions to overcome the most serious of these challenges, and bring the agriculture sector closer to fulfilling its substantial GHG mitigation potential.

Given that the vast majority of agricultural production and emissions is outside the OECD area, any mitigation policy restricted to OECD countries will have a limited impact on global emissions. However, reaching a global scale of uptake in mitigation policies, while managing the distributional impacts on producers and consumers in regions where food security and development objectives predominate, is a significant political challenge. In this context, the choice of policy used to mobilise mitigation is profoundly important as the distributional impacts and effectiveness for a given carbon price vary considerably among the main market-based mitigation policy options.

For example, polluter-pays policies, including the taxation of producer-level GHG emissions, are the most effective options available but they can impose relatively high costs on farmers and create emission leakages. Where these concerns stifle progress on mitigation, beneficiary-pays policies that pay for emission reductions by either a subsidy or the creation of an offset market could be a useful alternative. These policies are less effective however and unless they replace existing distortionary forms of support to agriculture, they risk reducing economic welfare compared to polluter-pays policies. There are alternative hybrid policy designs which could draw on the strengths of both types of market-based policy options and potentially achieve a more politically acceptable blend of trade-offs. A hybrid tax-subsidy mechanism which recycles emission tax revenue back to producers in order to subsidise the adoption of low emission technologies is one example. Free, but binding, allocations of permits in emission trading schemes are another.

There are also policy design solutions to address the practical challenges and transaction costs related to MRV. Simple emission proxies can be used instead of more direct forms of measurement to reduce these costs, but they are less effective and less cost-effective than policies that target emissions more directly, even after considering their transaction cost savings. The use of process-based models, supplemented with measurements, is another approach that can lower MRV costs, especially for SCS measures. However, there are serious questions about the policy feasibility of SCS measures, which is of concern given that they comprise such a large share of agriculture's global mitigation potential.

Growing attention is being given to the important technical mitigation potential of demand-side mitigation options (including measures that encourage consumers to switch to lower emission diets and reduce food waste). However, the potential of such policies to achieve this is remains untested.

Whatever option is chosen, it is important to send clear and consistent policy signals to the agricultural sector. The presence of high-fixed investment costs in some production systems can significantly lower the effectiveness of mitigation policies, especially in the short run when investment costs are sunk. By avoiding uncertainty in their long-term GHG mitigation objectives and policies, governments allow farmers to make the appropriate investment decisions to facilitate the transition to low carbon agriculture.

Notes

[1] As per Smith et al. (2014[4]), the term “forestry and other land use” here is consistent with the non-agricultural component of the term Agriculture, Forestry and Other Land Use (AFOLU) from the IPCC (2006) Guidelines and is also consistent with the term Land Use, Land-Use Change and Forestry (IPCC, 2003).

[2] Carbon tax applied to all global regions except to least developed countries.

[3] Between 2015 and 2019, 125.5 MtCO_2e of abatement was achieved with vegetation projects compared to 18.1 MtCO_2e from agricultural projects (Clean Energy Regulator, 2019).

[4] New Zealand Climate Change Response (Zero Carbon) Amendment Bill (amendment to the current Climate Change Response Act 2002).

[5] Includes carbon dioxide, nitrous oxide, hydrofluorocarbons, perfluorocarbons, sulphur hexafluoride, and nitrogen trifluoride.

[6] Biogenic methane emissions refer to methane emissions produced by the agriculture and waste sectors.

[7] With the exceptions of aviation and international maritime shipping.

[8] This is in line with the EU 2050 carbon neutrality objective outlined in the European strategic long-term vision for a climate neutral economy (European Commission, 2018b).

[9] https://www.mla.com.au/news-and-events/industry-news/red-meat-industry-can-be-carbon-neutral-by-2030

[10] Although emissions are rarely measured directly, some approaches such as those that rely on the accurate monitoring of production inputs, processes and outputs, coupled with detailed process-based models carefully calibrated to local conditions (i.e. IPCC Tier 3 measurement methods) are much closer to direct measurement than are approaches based on monitoring more simplistic emission proxies, such as number of cattle (i.e. IPCC Tier 1 measurement methods).

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2 Global analysis of mitigation policies for agriculture: Impacts and trade-offs

The Modular Applied GeNeral Equilibrium Tool (MAGNET) model, a multi-sector, multi-region computable general equilibrium model that covers the global economy, is used to evaluate several market-based mitigation policies to limit GHG emissions in agriculture. The policies analysed differ considerably in terms of the trade-offs they generate between mitigation outcomes and their associated impact on agricultural income, competitiveness, food consumption and government finances. This assessment provides policy makers with quantitative information about different policy design options that could deliver an acceptable blend of trade-offs, given their country-specific objectives and constraints.

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.

The importance of agriculture to global mitigation efforts

As discussed in Chapter 1, agriculture contributes substantially to climate change and to mitigate global GHG emissions effectively and efficiently, agriculture must do its part. This will become increasingly important over time, given that agriculture has so far received less consideration in GHG mitigation policies compared with energy and other sectors (Bajželj et al., 2014[1]).

In the past, concerns about emissions leakage and loss of competiveness may have prevented countries from taking independent and early action. Such leakage occurs when mitigation policies in one region raise agricultural production costs and prices, causing supply from that region to fall, which creates incentives for increases in production and emissions elsewhere to partially fill the shortfall in supply.

In the absence of ambitious targets and policies for reducing agricultural emissions in most countries, this chapter explores how agriculture could make a substantial contribution to global mitigation efforts with a range of market-based policies. The potential economic consequences of policies to deliver ambitious emission reductions in agriculture, including their possible impact on competitiveness, food security and agricultural income, are also assessed.

A global tax on agricultural GHG emissions is the most ambitious policy option assessed, which assumes a willingness by all countries to apply an equally strong GHG tax rate, irrespective of their development status and priorities. This policy represents a high mitigation benchmark, which is then compared to a range of arguably more feasible but less effective policy options. These options include changing the burden of mitigation responsibility to exclude non-OECD countries, as well as applying the "beneficiary pays" principle rather than the "polluter pays" principle to incentivise ambitious mitigation outcomes for the agriculture sector. In recognition of the challenges and costs associated with measuring agricultural emissions, the efficacy of GHG-based payments on emission-intensive producer inputs and products is also examined.

With respect to evaluating the mitigation performance of different policy instruments, it is helpful to have in mind a reasonable or "fair" global emissions reduction target for agriculture globally. Taking into account relative mitigation costs and considerations about food security, Wollenberg et al. (2016[2]) suggest a non-CO_2 emission reduction goal of 1 $GtCO_2eq\ yr^{-1}$ by 2030 for agriculture to contribute to the 2^oC warming target by the end of the century. This represents an 11-18% reduction relative to the business-as-usual baselines assumed in their study and an allowable non-CO_2 emissions budget of 6.15 to 7.78 $GtCO_2eq$ yr^1. By comparison, a 1 $GtCO_2eq\ yr^{-1}$ emission reduction in this assessment represents a 14% reduction of the baseline emissions, bringing the baseline non-CO_2 emissions in 2030 down from 7.33 to 6.33 $GtCO_2eq\ yr^1$. Wollenberg et al. (2016[2]) also propose a stronger longer-term target of and 2.5 $GtCO_2eq\ yr^{-1}$ by 2050 for agriculture's contribution to meeting the 2^oC target. These emission reduction targets have since become used as benchmarks in global mitigation assessments for agriculture, including in Frank et al. (2018[3]). Accordingly, they are used throughout this chapter as one of the performance benchmarks of the assessed policies.

In the next section, the model and data used for the analysis are described, along with the scope of the analysis and the selected mitigation policy instruments. Following this, the quantitative policy findings are presented. In the final section, the key policy messages and recommendations are explained, along with the main limitations of the analysis.

Modelling mitigation policies in agriculture for OECD countries and the world

The MAGNET model and scope of analysis

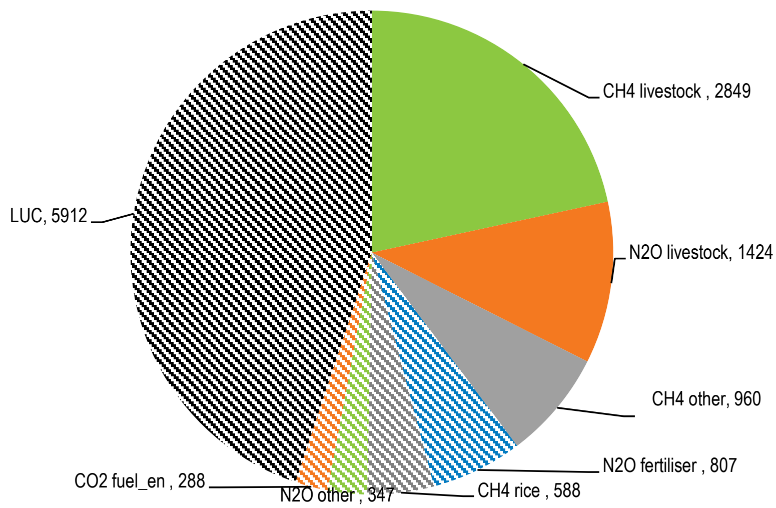
A computable general equilibrium (CGE) model is well suited to address many of the policy questions required to quantitatively assess the economic, competitiveness, and food security consequences of ambitious GHG mitigation targets for agriculture. A key strength of the CGE framework is its capacity to capture inter-sectoral relationships within agriculture, and between agriculture and other sectors, including other land use sectors. Other identified strengths include its ability to track trade relationships that influence competitiveness and leakage outcomes of mitigation policies, and the flow of costs and benefits to different sectors of the economy, including government, consumers and producers.

Given the utility of using a CGE model that can capture land use interactions with an acceptable degree of realism, the Modular Applied GeNeral Equilibrium Tool (MAGNET) model was selected (Woltjer and Kuiper, 2014[4]). This model has a long history of use within Wageningen University to assess the global impacts of policies in agriculture. It is a recursive dynamic multi-sector, multi-region Computable General Equilibrium (CGE) model that covers the global economy (Woltjer and Kuiper, 2014[4]). MAGNET is based on the Global Trade Analysis Project (GTAP) database and model developed by Purdue University in the United States (Hertel and Tsigas, 1997[5]). MAGNET and GTAP were originally designed to model the effects of trade policies, such as the Uruguay Round of multilateral trade negotiations, especially on the agricultural sectors. MAGNET has been extended and updated with several modules to improve the modelling of land markets and agricultural policies, biofuel policies, and the socio-economic and environmental impacts of environmental policies. There are eleven primary production sectors in agriculture, including eight crop sectors and three livestock sectors, and a total of 50 sectors in the model.

The version of MAGNET in this chapter uses the GTAP 9.2 database (Aguiar, Narayanan and McDougall, 2016[6]), which has a base year of 2011, but is updated in this assessment to create a dynamic baseline, from 2011 to 2050, with yield and economic growth assumptions that conform to the "middle of the road" Shared Socioeconomic Pathway (SSP2) (Fricko et al., 2017[7]). The model also incorporates emissions from the GTAP non-CO_2 database (Irfanoglu and van der Mensbrugghe, 2015[8]), including methane (CH_4) and nitrous oxide (N_2O). This is complemented by CO_2 emissions from the GTAP Energy-Environmental database (GTAP-E). Livestock non-CO_2 emissions and Rice CH_4 emissions are tied to the output variables of these respective sectors within the MAGNET model, whereas N_2O emissions from crop fertiliser use are tied to the fertiliser input variable in these sectors. In addition, data on the marginal abatement costs (MACs) associated with practices and technologies that can be used to reduce GHG emissions are incorporated. These data are from the US Environmental Protection Agency (EPA) (2013) and they cover measures lowering the main non-CO_2 emission sources, including methane (CH_4) from enteric fermentation by ruminants (i.e. cattle, sheep and goats), nitrous oxide (N_2O) and CH_4 from livestock manure, CH_4 emissions from paddy rice and N_2O emissions from soil associated with fertiliser use by crops. Accordingly, it is these emission sources that are targeted by mitigation policies in this assessment. It should be noted that the MACs used in this assessment do not include assumptions about technological change from the development and adoption of new technologies which lower the costs of mitigation over time. Consequently, the MAC data used in this assessment are conservative with respect to their assumed GHG mitigation potential, especially over the longer term. The CO_2 emissions associated with land use change (LUC) include changes in above and below ground carbon stocks between three aggregate types of land cover: cropland, grazing land, forest shrub land, and savannah land. The coefficients determining these changes in carbon stocks and CO_2 emissions are drawn from the Agro-ecological Zone Emission Factor (AEZ-EF) model described in Plevin et al. (2014[9]).

In this assessment, GHG mitigation policies are only applied to non-CO_2 emissions in the agriculture sector and not GHG emissions in other sectors of the economy. The possible implications of this modelling assumption are discussed below. Within agriculture, the vast majority of GHG emissions are targeted by most of the global mitigation policies considered in this assessment (78% of total agricultural GHG emissions in 2020, excluding LUC emissions). With reference to Figure 2.1, these include: CH_4 from enteric fermentation and livestock manure management; N_2O from livestock manure; N_2O from fertiliser applied to crops; and CH_4 from rice production. The remaining 22% of the emissions include CH_4 and N_2O emissions from the burning of biomass, and from fuel and energy use, and CO_2 emissions from fuel and energy use. The emission sources that are not targeted by mitigation options considered here are still included in MAGNET and the changes in these emissions can be reported including, for example, changes in LUC emissions due to the expansion or contraction of agricultural land.

Figure 2.1. The agricultural GHGs in the MAGNET model (MtCO2eq), 2020



The economic impacts of mitigation policies on the different agricultural sectors and regions depend on the mitigation opportunities embedded in their MACs, and on the economic emission intensity of the sector's output (i.e. the amount of GHG emissions from a sector divided by the economic value of its output). While there is a large variation in emission intensities across countries within a given sector, they are highest in the ruminant sector (OECD, 2019[10]). A GHG tax is therefore expected to have a relatively large impact on this sector.

Designing policies to unlock agriculture's mitigation potential

Based on considerations about relevant and feasible mitigation policy options for agriculture, a set of eight mitigation policies was selected for assessment. These policies are considered sufficiently broad in scope to address the primary objective of identifying policy solutions that can unlock the large mitigation potential of the agricultural sector, without compromising food security in low-income regions while helping regions maintain their competitiveness. The first five policy options directly target agricultural emissions, whereas the last options target emission-intensive production inputs or consumer products.

The assessed policy instruments are listed below.

Policies that directly target emissions

- Global tax on agricultural GHG emissions.
- OECD tax on agricultural GHG emissions.
- Global tax on agricultural GHG emissions combined with a food consumption subsidy
- Global abatement payment for agricultural GHG emission reductions.
- OECD abatement payment for agricultural GHG emission reductions.

Policies that target emission intensive production inputs and consumer products

- Consumer-level GHG tax on ruminant meat and dairy products consumed within OECD countries.
- Global GHG-based tax on emission intensive agricultural inputs, including ruminant animals and fertiliser
- OECD GHG-based tax on emission intensive agricultural inputs, including ruminant animals and fertiliser

The first five policy scenarios listed above are assessed under dynamic settings, whereby the policies are applied from 2020 through to 2050. In each of these scenarios, the same increasing carbon price pathway is applied: with GHG prices of USD 40/tCO_2eq, USD 60/tCO_2eq, and USD 100/tCO_2eq for the 2021-2030, 2031-2040, and 2041-2050 periods, respectively. These prices were considered to represent a reasonably high level of mitigation ambition compared to the much lower carbon market prices that have been observed to-date, where such markets exist. The USD 60/tCO_2eq price approximately corresponds to the value that some modelling studies suggest will be required to limit temperature increases to 1.5°C (Rogelj et al., 2015[11]). For technical reasons related to the fact that the final three scenarios impose a GHG-based tax on consumer products or producer inputs, it was necessary to assess these scenarios in static mode.[1] For these cases, 2050 was selected as the simulation year and a GHG price of USD 100/tCO_2eq was applied in order to be consistent with the prices used in the other scenarios for this same year. The mitigation performance of the policies simulated under dynamic settings is evaluated with respect to their capacity to achieve the non-CO_2 emission reduction targets of 1 GtCO_2eq yr^{-1} by 2030, and 2.5 GtCO_2eq yr^{-1} by 2050, proposed by Wollenberg et al. (2016[2]).

Beginning with the policies that directly target emissions, the first three follow the “polluter pays” principle by imposing a tax on emissions. The global taxes on GHG emissions, with and without the food consumption subsidy are the most ambitious policy options, as they assume a willingness by all countries to apply an equally strong GHG tax rate, irrespective of their development status and concerns about food production and food security. As mentioned above, the purpose of the first policy – the global tax on emissions – is to provide a high mitigation benchmark which can then be compared to a range of more feasible, but potentially less effective, mitigation policy options. In an attempt to address concerns that low-income countries may have about negative impacts on food production and agricultural incomes, a second scenario is defined where the tax on GHG emissions is limited to OECD countries. This option is, however, likely to erode the competitiveness of agriculture in OECD countries and cause a leakage of emissions mitigated by OECD countries into non-OECD countries. The third policy is a hybrid instrument that attempts to exploit the large mitigation potential that a global tax on agricultural emissions can provide by driving the restructuring of agricultural production in favour of sectors with lower GHG emissions, while at the same time providing a subsidy to consumers to maintain their baseline levels of food consumption.

The fourth and fifth policy options differ from the previous options by applying the “beneficiary pays” principle and providing an abatement payment to cover the mitigation costs of agricultural producers. This provides the same marginal abatement incentives as the GHG tax, but does not impose any tax burden on agricultural producers. The abatement payment is paid by the government to producers, and it precisely compensates producers for the costs they incur to reduce emissions at the selected carbon prices.

The final three scenarios are based on polices that attempt to circumvent the substantial challenge of measuring and monitoring GHG emissions from agricultural producers by applying a GHG-based tax to either emission-intensive production inputs (ruminant animals and fertiliser) or emission intensive consumer products (processed ruminant meat and dairy products). These policies would allow a saving in transaction costs (not quantified in this assessment) related to the measurement of emissions, but they would result in a loss of economic efficiency by failing to reward producers who lower their emissions by adopting mitigation practices that aim to lower emission intensities. The consumer-level GHG tax translates the value of emissions for the given tax rate into an equivalent tax set at the same rate for both domestic and imported consumer products within each OECD country or region, based on the economic emission intensity of the domestically-produced product. This tax is applied to ruminant meat and dairy products only. The motivation behind this policy is to address competitiveness and leakage issues that would typically emerge from the non-global application of a GHG tax by preserving the competitive position of domestic and imported products by taxing them at the same rate. This removes the onerous challenge of applying different tax rates to consumer products sourced from different destinations according to their emission intensities.

A notable omission from the above policy options is an emission-trading scheme. It is worth mentioning that an emission-trading scheme could be designed to provide similar mitigation and economic outcomes for agriculture as does the GHG tax and abatement payment mechanisms. According to economic theory, the auctioning of emission permits can provide the same mitigation incentives as a GHG tax, while the provision of free emission permits to agriculture could provide similar mitigation incentives as the abatement payment. Consequently, many of the insights on the mitigation effectiveness and economic impacts from the assessed instruments can be generalised to a broader range of market-based mitigation instruments than those assessed here.

GHG emission reductions and economic consequences of mitigation policies in agriculture

The quantitative impacts of the assessed policy instruments on emission reductions, agricultural producers and food consumers are presented. A more detailed regional breakdown of the modelling results is provided in the appendix of (OECD, 2019[10]).

The global GHG taxes, with and without the food subsidy, appear to be the most effective mitigation policies, narrowly missing the 1 $GtCO_2eq$, non-CO_2, 2030 mitigation target, and slightly exceeding the 2.5 $GtCO_2eq$ 2050 targets described in the previous sections (Figure 2.2 to Figure 2.4, Table 2.1). The global abatement payment is less effective, but still able to go about halfway towards achieving these targets. Although the GHG tax and abatement payments provide the same marginal mitigation incentives, the cost and price increases from the tax cause a contraction in the supply and demand for agricultural products in aggregate, but particularly from more emission-intensive sectors. This contraction is a major contributor to the overall reduction in emissions induced by this policy in some regions. For the ruminant sector aggregated across non-OECD countries, falls in production account for 42%, 43%, 46% of emission reductions of the global GHG tax in 2030, 2040 and 2050, respectively. Globally, the contribution of falling ruminant output to the total emission reductions of the ruminant sector is more muted at 28%, 26%, and 15%, respectively, as overall ruminant production in OECD countries increases over all three simulation periods. Accounting for the changes in LUC emissions reveals that the taxation policies could be substantially more effective by 2050 (Figure 2.3, Table 2.1). This results from a global shift in land cover from pasture to forest and shrub land, which will increase global carbon stocks over time as the ruminant grazing footprint contracts, particularly in Sub Saharan Africa and Latin America. Following the global abatement payment, LUC emissions increase relative to the baseline (Figure 2.4, Table 2.1), mainly due an increase in cropland at the expense of forest and shrub land in South East Asia and Latin America. However, these changes in land cover are one to two orders of magnitude smaller than the changes in land cover caused by the GHG tax. This nevertheless illustrates the potential importance of coupling this policy option with regulations to prevent the clearing of non-agricultural land containing comparatively high carbon stocks. Note that the consumer-level tax and tax on input policies are not displayed in Figure 2.3 and Figure 2.4 because they were only conducted for 2050.

As expected, the OECD GHG tax leads to the leakage of or increases in emissions in non-OECD countries, partially reducing its effectiveness (Table 2.1). The OECD GHG abatement payment is able to eliminate these leakage effects and provide a similar level of global mitigation as the OECD GHG tax, without the same negative consequences for agricultural production. Nevertheless, the policies confined to OECD countries make only small progress towards the proposed mitigation targets at the selected carbon prices (Figure 2.2 and Figure 2.3, Table 2.1).

The results of the consumer-level tax and tax on input policies that were assessed in static mode are presented in Table 2.2. For the purposes of comparison, the global GHG tax and the global abatement payment were also assessed in static mode for the year 2050 because dynamic and static scenario results cannot be meaningfully compared.[2] The global tax on ruminants and fertilisers generated less than one-fifth of the emission reductions achieved by the global GHG tax and about two-fifths of the reductions from the global abatement payment. This is partly because the global tax on ruminants and fertilisers targets a smaller volume (86%) of the emissions than the global GHG tax and the abatement payment. When limited to OECD countries, its impact is naturally much smaller, with leakage effects further weakening its effectiveness.

Figure 2.2. Global reductions in agricultural non-CO2 emissions for dynamic policy scenarios

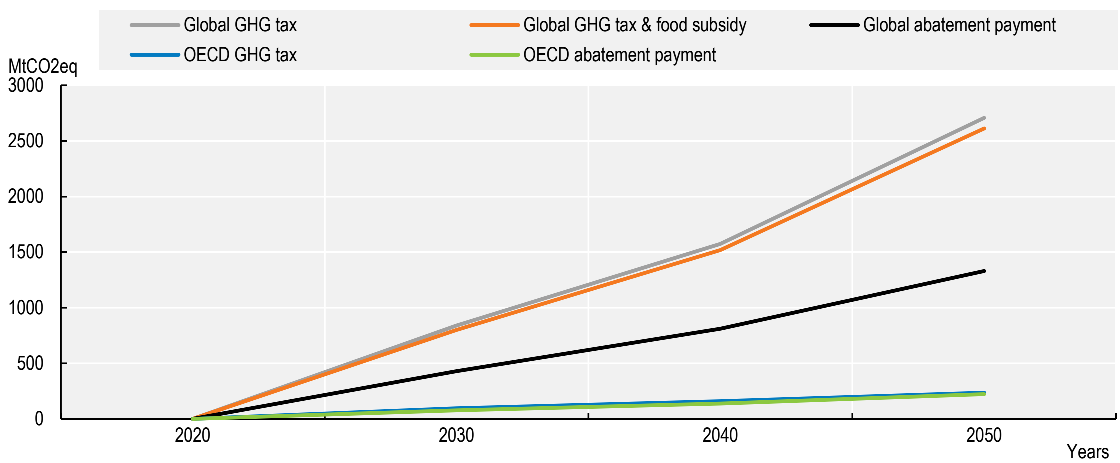


Figure 2.3. Global reductions in agricultural non-CO_2 and land use change emissions for dynamic policy scenarios

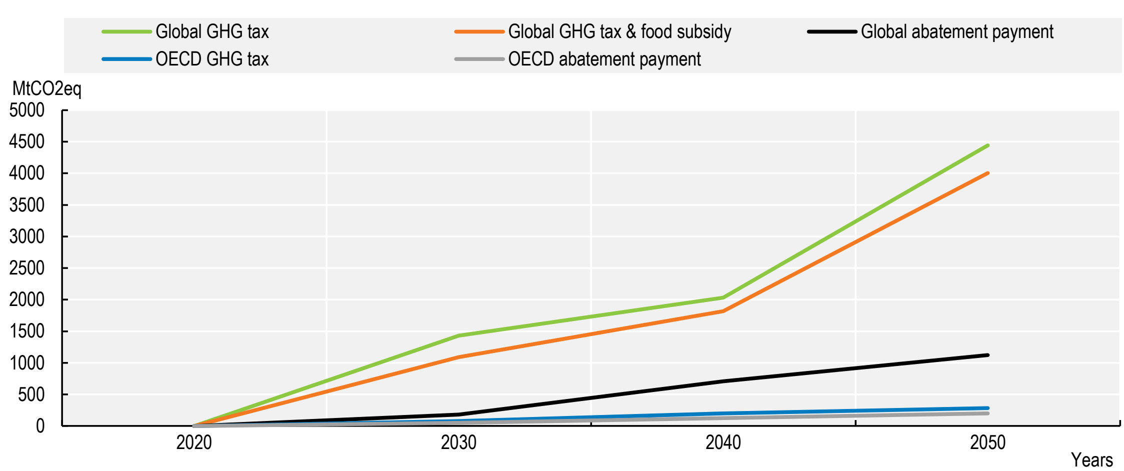


Figure 2.4. Global reductions in agricultural non-CO2 and land use change emissions for the global abatement payment

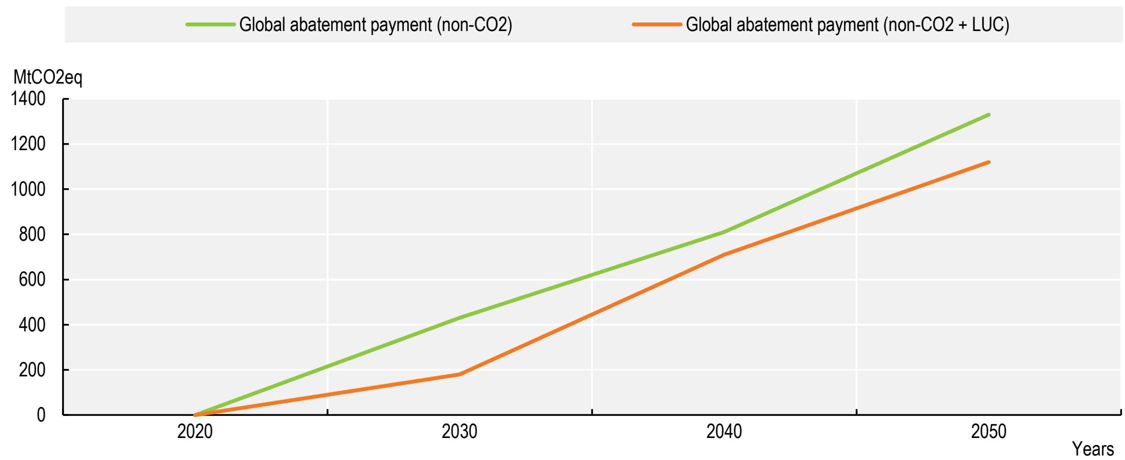


Table 2.1. Summary of annual agricultural non-CO_2 and LUC emission reductions policy instruments assessed under dynamic settings ($MtCO_2eq$), in 2050

		OECD	Non-OECD	Global	Leakage*
Global GHG tax	Non-CO_2	213	2,492	2,706	0%
		(15%)	(31%)	(28%)	
	LUC change	-70	1,806	1,736	
	Total	143	4,299	4,442	0%
		(8%)	(39%)	(35%)	
Global GHG abatement payment	Non-CO_2	224	1,106	1,330	0%
		(15%)	(14%)	(14%)	
	LUC change	-29	-180	-210	
	Total	194	926	1,120	0%
		(12%)	(8%)	(9%)	
OECD GHG tax	Non-CO_2	357	-122	235	34%
		(25%)	(-2%)	(2%)	
	LUC change	119	-69	49	
	Total	477	-192	284	40%
		(29%)	(-2%)	(2%)	
OECD GHG abatement payment	Non-CO2	228	-6	223	0%
		(16%)	(0%)	(2%)	
	LUC change	-12	-13	-25	
	Total	217	-19	197	0%
		(13%)	(0%)	(2%)	
Global GHG tax & food subsidy	Non-CO_2	199	2,413	2,611	0%
		(14%)	(30%)	(27%)	
	LUC change	-58	1,411	1,353	
	Total	144	3,861	4,005	0%
		(9%)	(35%)	(32%)	

* The leakage rate is calculated as the sum of the increases in agricultural GHG emissions in non-OECD countries, divided by the sum of the reductions in agricultural GHG emissions in OECD countries.
The percentages of the baseline non-CO_2 emissions reduced in each broad region are provided in parentheses.

The OECD consumer-level tax can negate the leakage of emissions, but as with the OECD ruminant and fertiliser tax, it is one of the least effective instruments for lowering emissions. The ineffectiveness of these less targeted approaches appears to worsen when the tax is levied at the consumer rather than at the input stage. This is because the impact of the tax is further weakened by the diversion of affected farm commodities from domestic to export markets, and by the diluting effect of intermediate inputs in the final processed food products.

The global GHG tax, abatement payment, and GHG tax with food subsidy, each have differing impacts not only on emission levels, but also on agricultural producers and food consumers. While the GHG tax leads to the largest emission reductions, it has the most detrimental effect on farm income (measured as value-added or returns to the land, capital and labour endowments, at agents prices), particularly in non-OECD regions. It also causes the largest reduction in food consumption (weighted by value at constant 2020 world prices), though not nearly as large as its impact on producers (Table 2.3). Conversely, it generates the largest increases in government revenue (Table 2.4).

Table 2.2. Summary of annual agricultural non-CO_2 emission reductions for policy instruments assessed under static settings ($MtCO_2eq$), 2050

	OECD	Non-OECD	Global	Leakage*
Global GHG tax	215	1 380	1 595	0%
Global GHG abatement payment	146	579	725	0%
OECD meat & milk consumer-level tax	33	18	51	0%
Global GHG tax on ruminants & fertilisers	16	285	301	0%
OECD GHG tax on ruminants & fertilisers	59	-13	46	22%

Table 2.3 Changes in agricultural value-added and household food consumption from policies, 2050

	Global GHG tax		Global GHG tax and food subsidy		Global abatement payment	
Region*	Value-added	Consumption	Value added	Consumption	Value added	Consumption
North America	-2%	-2%	3%	0%	3%	0%
Australia-New Zealand	3%	-3%	8%	0%	3%	0%
Europe	0%	-2%	5%	0%	3%	0%
Mexico-Chile	-9%	-1%	-5%	0%	2%	0%
Other OECD	1%	-1%	5%	0%	2%	0%
MENA-Caspian	0%	-2%	4%	0%	2%	0%
South Asia	-13%	-1%	-9%	0%	3%	0%
Sub-Saharan Africa	-36%	1%	-34%	2%	5%	0%
East & South East Asia	-2%	-1%	0%	0%	2%	0%
Latin America	-9%	-3%	-4%	1%	4%	0%
OECD	**-1%**	**-2%**	**4%**	**0%**	**3%**	**0%**
Non-OECD	**-14%**	**-1%**	**-10%**	**1%**	**3%**	**0%**
Global	**-11%**	**-1%**	**-8%**	**1%**	**3%**	**0%**

Note: OECD regions are indicated in bold. North America consists of the United States and Canada. Europe covers all OECD European countries. Other OECD includes Japan, Korea, Israel, and Turkey. MENA-Caspian includes the Middle East, North Africa and countries of the Caspian region. East and South East Asia include China, South East Asia, and non-OECD countries in East Asia. Latin America includes all non-OECD Latin American countries.

Table 2.4. Annual changes to government budget from selected global GHG mitigation policies, 2050 (USD million)

	Global GHG tax revenue	Global GHG tax and food subsidy net	Global GHG abatement payment cost
North America	36 915	13 945	-1 863
Australia-New Zealand	17 096	13 462	-1 228
Europe	39 754	4 054	-1 658
Mexico-Chile	7 349	844	-455
Other OECD	7 859	342	-471
MENA-Caspian	37 873	-3 647	-1 170
South Asia	111 530	67 633	-6 909
Sub-Saharan Africa	111 092	113 781	-3 575
East and South East Asia	108 710	75 096	-8 485
Latin America	100 760	36 757	-4 859

A different but somewhat improved assembly of trade-offs emerges from the addition of a food consumption subsidy to the GHG tax. The combined policies have similar impacts on reducing emissions and on producers, but this time consumption is maintained and raises a smaller but still positive amount of government revenue in all regions apart from one. However, given the substantial negative impact of this policy on producers in low-income countries, it would be very likely to reduce food security for the rural poor in these same countries. Note that in Sub-Saharan Africa, the global GHG tax does not cause aggregate food consumption to fall. In this region, the crop sector benefits from the reduction in input prices that ensue from the substantial fall in emission intensive livestock production, expanding its production (OECD, 2019[10]). On balance, this has a positive net impact on aggregate, value-weighted, food consumption in 2050. Consequently, in this year, this region does not receive a food consumption subsidy in the GHG tax with food subsidy scenario. In all other simulation periods, aggregate food consumption weighted by value declines in all regions.[3]

The global abatement payment offers the prospect of appreciable global emission reductions (Table 2.1) without harming agricultural producers or food consumption at the aggregate regional level (Table 2.3). However, in contrast to the GHG tax policies, the abatement payment needs to be paid for. In this assessment, the cost of the abatement payment is paid by governments within each region. These policies not only differ in terms of who incurs the cost of abatement, but also with respect to the size of these costs, with costs of the abatement payment to government being much smaller than the cost of the GHG tax to producers. This asymmetry occurs because the abatement payment covers only the cost of reducing emissions, whereas the GHG tax is levied on the entire stream of producers' non-CO_2 emissions (i.e. both the abated and unabated portion of emissions).

In addition to their impact of food consumption, producer income, and government budgets, these instruments generate different economic welfare impacts. To assess these impacts, the welfare measure known as equivalent variation (EV) was used. This approach uses government expenditures as a proxy for welfare obtained from public goods (Keller, 1980[12]). It is also often used in CGE analyses to approximate changes in the efficiency with which economic resources are allocated within the economy. Global EV for the global GHG tax and the global abatement payment is USD -27 944 million and USD -18 430 million, respectively, in 2050. These figures are negative, indicating there is a loss of welfare associated with these policies that attempt to mitigate GHG emissions (World Bank, 2018[13]). The welfare loss from the tax is about 50% larger than the loss associated with the abatement payment; however, the tax generates about 100% and 300% higher non-CO_2 reductions and total emission reductions (non-CO_2 + LUC emissions), respectively (Table 2.4). Therefore, from an economic welfare perspective, the abatement payment performed worse than the GHG tax relative to the quantity of emissions reduced. However, this is a partial evaluation of the economic welfare because it does not consider welfare benefits in terms of the avoided damage costs associated with emission reductions achieved by each policy. If these benefits were considered, both policies could deliver an improvement in net welfare.

Another more policy targeted option, but which is not assessed in this chapter, would be to redirect part of the existing producer support provided to the sector for non-environmental purposes to pay for the abatement payment instrument. This approach to lower the sector's carbon footprint is presently gathering support among international experts and agencies, including the World Bank (2018[13]). With 2015-2017 agricultural support for the 51 countries considered in the OECD's *Agricultural Policy Monitoring and Evaluation 2018* (2018[14]) calculated to be USD 484 billion, there are arguably sufficient resources to easily cover annual abatement payments for OECD and non-OECD countries, which are projected to reach USD 2 312 and USD 9 022 million, respectively, by 2030, and USD 5 675 and USD 25 117 million, respectively, by 2050.[4] The financial burden of this instrument would increase further if a more ambitious carbon price path capable of reaching the sector's 2030 and 2050 mitigation targets was assumed.

Other funding arrangements may be feasible, for example the purchasing of agricultural emission reduction credits by other sectors that are required to pay for emitting GHGs, notwithstanding the political challenges that may be associated with initiating such transfers. This approach would be possible in the few locations with operational emission trading schemes (e.g. the European Union and New Zealand, although more countries are expected to adopt national carbon pricing schemes in future).

To provide some validation of the model results it is useful to compare the magnitudes of emission reductions from this assessment with comparable global studies. The non-CO_2 emission reduction potentials of 0.43-0.84 $GtCO_2eq$ at USD 40/tCO_2eq in 2030, 0.81-1.57 $GtCO_2eq$ at USD 60/tCO_2eq in 2040, and 1.33-2.71 $GtCO_2eq$ at USD 100/tCO_2eq in 2050, from the global GHG tax and abatement payment policies assessed in this chapter, are well within the range of potentials from existing studies in the literature. According to the most recent Assessment Report of the Intergovernmental Panel on Climate Change (Smith et al., 2014[15]), annual emission reductions for agriculture of 0.03-2.6 $GtCO_2eq$ – at USD 50/tCO_2eq,and 0.2-4.6 $GtCO_2eq$ at USD 100/tCO_2eq in 2030 – are based on results from different studies (Rose et al., 2012[16]; McKinsey & Company, 2009[17]; Golub et al., 2009[18]; Smith et al., 2007[19]) and include soil carbon sequestration as well as non-CO_2 emission reductions. A more recent partial equilibrium assessment by Frank et al. (2018[3])(2018) calculated higher non-CO_2 mitigation potential in 2030 of 1 $GtCO_2eq$ at only USD 25/tCO_2eq, but with a slightly lower mitigation potential of 2.6 $GtCO_2eq$ at USD 100/tCO_2eq in 2050. These figures are comparable to those in this assessment, although the models differ significantly in structure and emission baselines and in the way they integrate abatement options.

There may also be substantial additional mitigation from changing consumers' dietary choices to include a less emission intensive basket of food commodities (Bajželj et al., 2014[1]; Wollenberg et al., 2016[2]; Poore and Nemecek, 2018[20]). However, no clear or effective policy options have been proposed to achieve this. The hybrid policy assessed in this chapter, which combined a GHG tax with a food consumption subsidy, provides one option for incentivising such a dietary shift without sacrificing total food consumption. The assessment of this policy could, however, be improved by focusing on maintaining the nutritional value of consumption rather than its value at constant world prices.

As with all modelling assessments, there are caveats. For instance, the mitigation potential of the policies calculated in this chapter may be lower than the agricultural sector's full potential because the mitigation policies only target 78% of the sector's non-CO_2 emissions. Moreover, the mitigation potentials for these emissions that are included are also conservative, because the MACs used do not consider technological changes that lower the costs of mitigation over time. In addition, options to sequester soil carbon in grasslands and croplands were not considered. This omission was due to the absence of reliable global data on the marginal costs of soil carbon sequestration.

Including mitigation policies in non-agricultural sectors, particularly land use sectors, could have important implications for the performance of mitigation policies in agriculture. Competition for land between agriculture and forestry can be particularly influential for agricultural production and emissions. Research by Golub et al. (2009[18]), also using CGE model, showed that subsidising carbon sequestration in the forestry sector can increase forest area at the expense of grazing land, causing extensive ruminant production and emissions to contract. When combined with a GHG tax on agricultural emissions, this contraction intensified. Considering mitigation more broadly for the land use sector as whole would be a useful extension to the assessment in this chapter.

Another caveat is the absence of climate change impacts in the baseline and policy scenarios. However, the policy insights from the assessment, in terms of the relative magnitudes of the different policies and the types of trade-offs they induce, are unlikely to change very much if these impacts were taken into account. At the global level, most studies assessing climate change impacts over time do not predict very large changes in agricultural production between now and 2050. For instance, Nelson et al. (2013[21]) project a mean global decline in crop production of only 2% by 2050, and van Meijl et al. (2018[22]) simulate a similar small decline in agricultural production of between 0.5 and 2.5% by 2050. Still, there will be larger impacts in some regions. Importantly, however, Meijl et al (2018[22]) found, in a model inter-comparison study covering five global models (IMAGE, CAPRI, GLOBIOM, MAgPIE, MAGNET) that non-CO_2 emission taxes and land-based mitigation policies in agriculture, commensurate with the sector's contribution to a 2^oC global warming target, would have a much larger negative impact on agricultural production than the effects of climate change.

It would have also been instructive to assess the impact of transferring a portion of existing coupled support payments to agriculture to fund the GHG abatement payment. However, given that the level of support among countries is so variable, some countries could easily fund abatement this way, while others could not. Consequently, this approach could result in differentiated impacts, with countries that are able to transfer coupled support to abatement activities possibly experiencing stronger reductions in emissions and output as a consequence of removing support. Further work on quantifying these impacts is recommended, including the calculation of possible emission leakage effects that may arise from the ensuing adjustments in competiveness.

Summary of findings

There is growing recognition of the importance of reducing GHG emissions from agriculture to meet the ambitious targets of the Paris Agreement goal to limit global average temperatures to well below 2°C and pursue efforts to limit the increase to 1.5°C above pre-industrial levels. The challenge for policy makers is to find ways to reduce agricultural emissions in a way that also minimises the negative consequences of mitigation policies on food security, agricultural income, and competitiveness.

The policies assessed in this chapter differed considerably in terms of the trade-offs they generated between mitigation outcomes and their associated impacts on agricultural income, competitiveness, food consumption, and government finances. The mitigation effectiveness is assessed with reference to annual non-CO_2 emission reduction targets of 1 and 2.5 $GtCO_2eq$ by 2030 and 2050. These are not official targets, but have been proposed by some analysts as being commensurate with agriculture's global emission contribution and capacity to mitigate.

The global GHG tax-induced large emission reductions are more or less aligned with the above targets, but imposed the largest economic costs on agricultural producers, particularly in the emission intensive ruminant sectors of many developing countries. They also slightly reduce household food consumption, although it should be possible to insulate consumers from the associated negative impact linked to the resulting higher food prices by combining the tax with a food subsidy, which could be financed by the GHG tax. The global abatement payment offers the prospect of appreciable global emission reductions without harming agricultural producers or food consumers, although only half as effective as the tax in reducing non-CO_2 emissions. The effectiveness of the abatement payment could fall further if emissions from land use change are also taken into account due to the small expansion of agricultural land that can result from this policy. From an economic welfare (or efficiency of economic resource allocation) perspective, the abatement payment performed worse than the GHG tax, relative to the quantity of emissions reduced.

Moreover, unlike the GHG tax policies, which generate government revenue, the global abatement payment would need to be funded. However, the level of payment needed globally represents a small proportion of the agricultural producer support currently provided by countries for non-environmental purposes.

The policy options which levy GHG taxes on emission proxies, such as more easily measurable and emission intensive production inputs or consumer products, were found to be far less effective than directly taxing emissions. Their ineffectiveness appears to worsen when the tax is levied at the consumer stage compared to the input stage.

The geographical scale of policies is critical to their mitigation effectiveness. More than a third of the GHG emission reductions from a GHG tax that is limited to OECD countries could be leaked as increases in emissions in non-OECD countries. If an abatement payment to OECD country emissions were applied instead, these leakage impacts could be controlled while delivering a similar level of global mitigation. However, it is clear that OECD countries alone cannot make a meaningful contribution to lowering global agricultural emissions given the dominant share of non-OECD countries in global agricultural emissions.

Notes

[1] For policies vi, vii and viii, small policy shocks are used to restrict the tax revenue generated by taxing either consumption or inputs to match the revenue that would be collected GHG tax on the emissions that are associated with these inputs and outputs. Given the small policy shocks, interaction with dynamic features is expected to be limited, so these shocks were implemented in static mode for the year 2050.

[2] The reason is that the agriculture sector is exposed to mitigation incentives over a sustained period (2020-2050) in the dynamic scenarios, causing emissions to diverge quite considerably with the dynamic baseline. In contrast, the emission reductions achieved with a specific mitigation policy applied for single year to the 2050 baseline, as is done with the static simulations, are smaller than the emission reductions achieved in 2050 under dynamic settings.

[3] Latin America experiences a similar pattern of production effects. However, the substitution effect between crops and livestock is not as strong and the share of the food that is derived from crop-based sources is lower than in Sub Saharan Africa. For this region, the global GHG tax causes aggregate food consumption to decline.

[4] Note these 2050 figures for the non-OECD countries do not equal those presented in the table because the Russian Federation and non-OECD European countries are not included in the latter.

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3 Farm-level analysis of mitigation policies for agriculture

A quantitative bio-economic farm model was developed to assess the cost-effectiveness of key GHG mitigation policy instruments to reduce emissions from crop and livestock production. Six policy instruments are examined and applied to farm cases in the European Union. The results show high abatement costs in mixed dairy and crop production when aiming for large GHG emission reductions and confirm that market-based policy instruments (GHG emission tax, GHG abatement subsidy, and cap-and-trade scheme) are the most cost-effective policy options. The results also show that policy instruments that target all GHG emissions from farms are more cost-effective than those that target only a subset of emissions or proxies of emissions, even when higher transaction costs of those policy instruments targeting all GHG emissions are accounted for.

Introduction

Many greenhouse gas (GHG) mitigation options are readily available to the agriculture sector, including reducing nitrogen fertiliser use, adopting reduced or no tillage methods, conversion of arable land to grassland, and changing livestock diet (MacLeod et al., 2015[1]). Few of these options can be considered win-win, i.e. increasing farm profits while reducing GHG emissions, so their adoption often requires policy instruments or markets that incentivise and accelerate their uptake.

Most GHG mitigation studies in the agriculture sector focus on ranking the cost-effectiveness of technical mitigation options and deriving marginal abatement cost curves with various methodologies, including bottom-up cost engineering, micro-economic modelling with exogenous prices, and equilibrium models with endogenous prices (MacLeod et al., 2015[1]).

This chapter focuses on the question of mitigation policy design with the objective to assess the relative mitigation effectiveness and cost effectiveness of key GHG mitigation policy instruments to reduce emissions from crop and livestock production. It analyses six policy instruments: an emission constraint; an emission tax; an abatement subsidy; an input tax on nitrogen fertiliser; an input tax on ruminant heads; and emissions trading. These are applied to farm cases in the European Union. To do this, a model covering crop and livestock production activities was developed and applied to farms representing a diversity of situations at the regional-level.

A bio-economic framework for a mixed farming system made up of crop and dairy production was developed and implemented as a detailed bio-economic optimization model for arable-dairy farms with non-linear crop and milk yield functions, and a detailed accounting of GHG emissions parameterised to four regional production systems. In addition to adjusting the crop land allocation, herd size, feed mix, and mineral fertiliser and manure application levels, the model incorporates technological changes regarding manure storage (from non-covered to covered manure storage) and manure spreading (from broadcast spreading to injection) as GHG abatement options. Manure nitrogen excretion response to dietary changes is also modelled.

A bio-economic framework for dairy and crop production

Overview of the bio-economic framework

The bio-economic framework accounts for the interactions between decisions on livestock (milk production) and crop choices associated with on-farm fodder production, manure use as a source of nutrients, and competition for quasi-fixed resources such as land and labour between crop and dairy production. The model baseline depicts interrelated, profit-maximising choices of herd size, milk and crop yields, diet, fertilization, and land allocation between grass silage and crop production under current market and policy conditions. Under different greenhouse gas (GHG) mitigation policy instruments, the farmer adjusts decision variables to reach new profit-maximising levels.

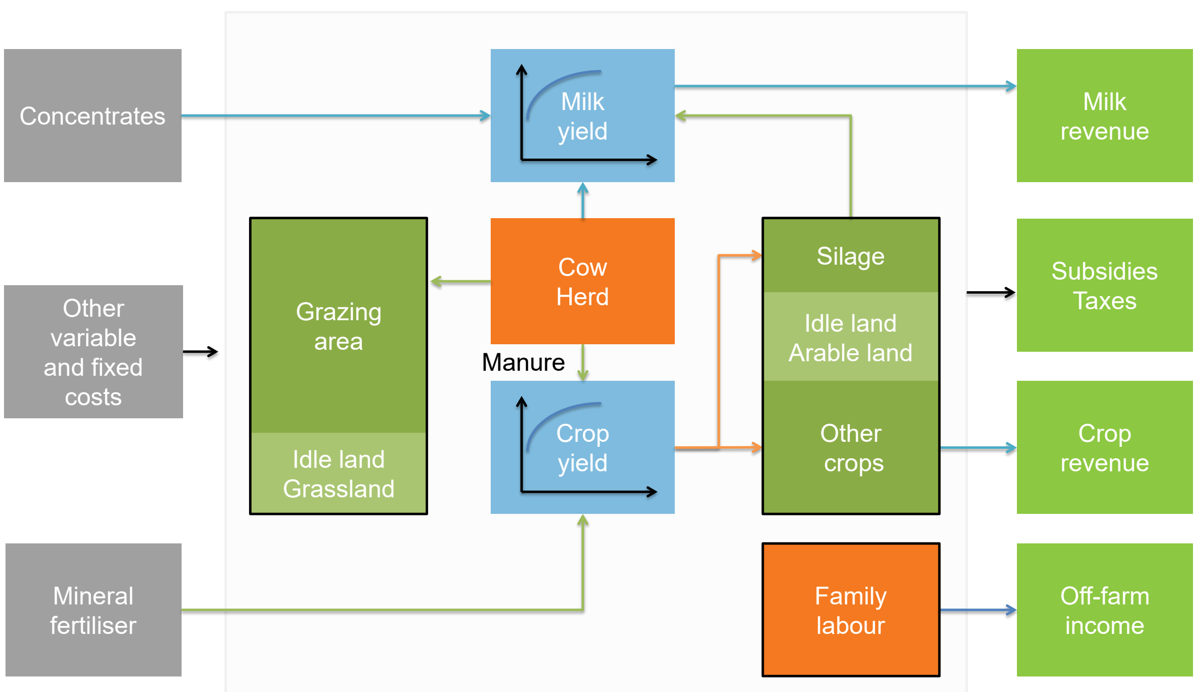
For dairy production, the impact of diet composition on milk yield, manure excretion, and manure composition is modelled. Increased intake of concentrates increases milk yield per cow at decreasing rates up to a maximal yield level, while the intake of fodder decreases in parallel. Fodder sources include grass silage that is produced on-farm and grazing. The replacement of animals for the milk herd is based on heifers raised on the farm. The number of lactations is modelled as a function of the milk yield. All other revenues and costs are expressed per dairy cow.

Fodder and other crops compete for arable land. Their yields depend on the applied fertiliser. Mineral fertiliser and manure – reflecting plant available nutrients – are assumed to be perfect substitutes in the relevant simulation range. The marginal crop yield response decreases with increasing fertiliser application rates, up to a maximal crop yield. All activities compete for farm labour, which can alternatively be employed off-farm at a given reservation wage. As grass silage is not marketed, its costs reflect production costs, opportunity costs of labour, and land on the one hand, and the substitution value against feed concentrates on the other. Similarly, the value of manure reflects differences in application costs relative to mineral fertiliser and the content of plant-available nutrients.

Various decision variables affect GHG emissions. Different types of policy instruments can be modelled (emission constraint, emission tax, abatement subsidy, input and output taxes, and carbon trading).

A non-linear programming approach was used to simulate the optimal decision making of a farmer (Figure 3.1) under different endowments and technology, as well as different market and policy environments. The farmer manages three fixed endowments, indicated by the black-outlined boxes: grasslands, arable lands, and family labour. The latter can be used on- or off-farm. The interdependent and simultaneously determined decision variables in the comparative-static framework are the cowherd and acreages, crop and milk yields, mineral and organic fertilization levels, and the feed mix. The costs of mineral fertiliser and concentrates are explicitly included; other costs are summarised. Revenues stem from selling milk and arable crops. Costs for animal replacement and revenues from selling old cows are accounted for as well. Besides grassland and silage, the model includes the arable crops wheat, barley and rape. Input and output prices are considered exogenous.

Figure 3.1. Main interactions in bio-economic modelling framework



Crop yields are endogenously depicted by nitrogen dependent yield functions of either the Mitcherlich or quadratic functional form. The different crops including grass silage compete for arable land, while pastureland per cow is fixed. The model maximizes either profits or utility when production risks for crops are considered. Compared to other bio-economic models, this model differentiates itself by using non-linear crop and milk yield functions and by its endogenous use of IPCC Tier 3 emission accounting.[1]

While a vast body of literature analyses the interaction between yields, fertilization and climate-change-relevant emissions based on biophysical models (Britz and Leip, 2009[2]), application of non-linear yield functions in more complex farm-scale bio-economic models is still scarce. For example, Lengers et al. (2013[3]) use a purely linear model in their analysis, which includes similar abatement options as here, but does not consider yield response or non-linear substitution between concentrates and fodder, focusing on different GHG emission indicators. Similarly, De Cara et al. (2005[4]) use more aggregate linear single farm models in their European-wide analysis, however fixing crop and milk yields.

The following GHG emissions are accounted for:

- methane emissions from enteric fermentation and from manure storage,
- direct N_2O emissions from manure storage
- indirect N_2O emissions from manure storage and spreading (NH_3 emissions from manure storage and spreading cause indirect N_2O emissions)
- GHG emissions from cultivated land including nitrogen fertiliser use and autonomous soil emissions
- emissions from cultivation practices, crop yield transportation and grain drying. Furthermore, soil carbon sequestration is taken into account when arable land is put under green set-aside.

Methane emissions from enteric fermentation per cow reflect milk yields and the digestibility of the cow's diet. Methane and nitrous oxide emissions from manure storage depend on feeding and manure storage technologies (e.g. uncovered manure storage and manure storage with a floating cover). Feeding practices only impact emissions from uncovered manure storage, while a floating manure storage cover decreases emissions by about 30%, independent of concentrate feeding levels.

Annexes 3.A and 3.B detail all equations and parameters.

Data and model calibration

The model application draws on the Common Agricultural Policy Regional Impact (CAPRI) database.[2, 3, 4] In order to derive stylised regional cases for this bio-economic model, data encompassing 23 European countries were taken from the CAPRI database for the year 2012. These data relate to, for example, regional crop acreage and dairy cow numbers, crop and milk yields, application of nitrogen and phosphorus in chemical fertiliser and manure, value of outputs and production inputs, various GHG emissions and total global warming potential of crop and milk production, ammonia emissions, and feed inputs.

To apply the model, four representative farms combining dairy and crop production were parameterised to illustrate the impact of productivity differences in both crop and milk production. The following data from the CAPRI database were used to calibrate these four representative farms: crop yields, milk yields, crop-specific production costs, milk-specific production costs, mineral and organic nitrogen application, dairy cow diet (the amount of feed cereals and protein) and output value for crops and milk. Based on these data, crop and milk yield functions for each farm were calibrated so that yield levels corresponded to input use given in the CAPRI database. In addition, production costs were calibrated based on the CAPRI database for each farm.

All four-farm cases were represented through a standard farm layout: a dairy and crop production farm with 60 hectares of arable land and up to 40 hectares of pastureland, reflecting the EU-15 average of about 60 dairy cows. Farm A represents a high milk and low crop yield situation;[5] Farm B has both low milk and crop yields; Farm C represents a low milk and high crop yield situation; and Farm D features both high milk and crop yields. The yields for the four farms relate to four regions from the CAPRI database, and represent mean yields and production costs in a given region for both milk and crop production, and were selected to illustrate how productivity and profitability differences affect the mitigation and cost-effectiveness of GHG abatement policy instruments across regions.

Results

Baseline scenario

Table 3.1 presents the baseline situation for the four farms representing different conditions across four regions. The baseline assumes that all farms receive support payments of EUR 190/hectare under the first pillar of the Common Agricultural Policy (CAP). The following paragraphs briefly describe the situation of each farm type under the baseline scenario.

Under current market and policy conditions, Farm A receives EUR 19 000 per year as CAP support based on non-current production. With milk price of EUR 0.45/kg, the herd generates market revenues of around EUR 250 000 yearly. Dairy production is labour intensive, with more than 100 hours per dairy cow (including fodder production), such that the labour input totals 7 000 working hours per year. Significant costs are spent on concentrates, close to EUR 56 000, and nitrogen fertiliser, around EUR 30 000.

Profit before taxes and social security is around EUR 125 000, which suggests returns to labour of EUR 18/hour, potentially competitive to wages outside agriculture. Note that, by assumption, only 50% of fixed costs in dairy production are included in the medium-term calculation, with the remaining 50% considered sunk and not decision-dependent. It should be noted that in the real-world farm population, some farmers will have invested (or re-invested) recently in dairy operations and are likely to continue for several years despite relatively low returns to labour, while others will need to decide whether to re-invest in dairy production over the medium-term, or to continue crop production as well as work off-farm.

Table 3.1. Baseline scenario

	Farm A High milk and low crop yields	**Farm B** Low milk and low crop yields	**Farm C** Low milk and high crop yields	**Farm D** High milk and high crop yields
Herd size, dairy cows	57	42	63	57
Land allocation, ha: CO:Si:P:Se:GSe[1]	18-42-40-0-0	31-29-32-0-8	37-23-21-0-19	40-20-40-0-0
Wheat yield, kg/ha	4371	4866	6406	6760
Nitrogen fertiliser application, kg/ha	151	159	181	184
Milk, kg/dairy cow/ year	9098	8074	8070	9128
Concentrates, kg DM/dairy cow/day[2]	10.3	7.1	6.0	14.0
Silage, kg DM/dairy cow/day[2]	5.3	6.9	7.3	3.0
Total GHG emissions, kg CO_2eq./year	642 600	515 504	649 996	671 539
GHG emission shares: cultivation: fertiliser: soil: livestock	7-22-13-58	11-26-15-48	9-22-10-58	11-22-13-55
GHG Emission intensity for wheat (kg CO_2eq per value of output)	4.3	3.9	3.1	2.9
GHG Emission intensity for milk (kg CO_2eq per value of output)	1.1	1.2	1.2	1.1
Profit, EUR/year	125 606	99 234	159 140	146 159

1 Cereals and oilseeds (CO), Grass silage (Si), Pasture (P), Set-aside (Se) and Green set-aside (GSe).
2 DM refers to dry matter.

The total fixed costs amount to around EUR 800 per cow per year. Moving to a long-term perspective, and including the sunk part of these costs, profits would decrease by EUR 23 000 (50% of EUR 46 000 in total fixed costs). EUR 19 000 of support payments are decoupled and thus do not impact production decisions on the farm.[6] If profit net of decoupled payment is decreased by the fixed costs, the total long-run decision-dependent profits of the farm amount to EUR 83 000, or to around EUR 12/hour. This illustrates that sunk costs of investment play a key role in farm adjustment and management response to different policy instruments.[7]

The majority of GHG emissions stems from enteric fermentation (representing 58% of total CO_2eq emissions and 90% of livestock-related emissions), followed by nitrogen fertilisation (22% of total CO_2eq emissions). These numbers are similar to findings in other studies for intensive dairy systems in the temperate zone. Due to the relatively low productivity of wheat production on Farm A, the greenhouse gas (GHG) emission intensity of wheat is high relative to the value of output, while GHG emission intensity is low for milk production owing to high milk yields.

Due to lower milk yields on Farm B, dairy cow feeding is less intensive than on Farm A, and thus requires higher grass silage areas per cow, while marginal returns per unit of silage produced are lower. Combined with moderate crop yields, this results in the smallest herd of the four farms as land is allocated to cash crops. The smaller dairy herd size results in low methane emissions from enteric fermentation and as a result the total GHG emissions are smaller than in Farm A. The emission intensity of milk production is however higher, as emissions linked to the energy maintenance needs of the herd are distributed over a smaller milk quantity. The GHG emission intensity for wheat is lower than on Farm A, reflecting slightly higher yields.

Although milk yields are relatively low on Farm C, higher crop yields (and in particular grass silage production) helps to push the optimal herd size of dairy cows to 63. High productivity grass silage production allows less land to be allocated to silage and more to wheat production. Although total CO_2eq emissions are higher than on Farms A or B, the GHG emission intensity for wheat on Farm C is lower due to high wheat yields.

Farm D represents the case of both high milk and crop yields. This leads to high fertiliser intensity in wheat production and high concentrate feeding for dairy cows. Due to intensive production, total CO_2eq emissions at the farm level are high, but the emission intensities for wheat and milk are relatively low.

GHG emission constraint: Abatement cost function and marginal abatement costs

Table 3.2 presents how the different farms respond to decreasing GHG emission ceilings. Calculations assume that farmers have no off-farm employment opportunities and face fixed investment costs of 50%.[8] An enforced uniform 10% GHG emission reduction for each farm considerably affects input (reduced concentrate feeding and nitrogen fertiliser application) and land use (allocation of land from pasture and grass silage towards wheat and green set-aside). The latter reflects that most of the GHG savings stems from reduced methane emissions related to enteric fermentation, which decrease on average by about 15%, reflecting a reduction in herd size. This, in turn, decreases fodder needs and drives the land allocation from pasture and silage towards cereals and green set-aside. Note that except for Farm A, all farms increase concentrate intake per cow and thus milk yields. Farm A is a special case as its baseline shows the highest share of grass silage in land use of all farms because of the low productivity and profitability of wheat production relative to silage. Nitrogen fertiliser application per hectare is another way to respond to the emission ceiling, but its contribution to reducing GHG emissions is limited.

At higher GHG emissions reduction levels (20%-40%), farms adjust with the same mechanisms: they reduce the number of dairy cows and shift land allocation towards cereals and green set-aside, away from pasture and silage.

Table 3.3 shows the marginal abatement costs (MACs) of GHG emission reductions for the four farms. Marginal abatement costs represent the shadow price of emission constraint for each emission reduction level. The estimates made assume that farmers have no off-farm employment opportunity (thus zero opportunity cost of farm labour) in order to show the agricultural cost of adjustment on a given farm for each emission reduction level.[9] These marginal abatement costs coincide with the literature. For an 8% GHG emission reduction, De Cara et al. (2005[4]) estimate an average marginal abatement cost of EUR 123/ton of CO_2eq for the EU15, and Pérez Dominguez et al. (2003) an average of EUR 95/ton of CO_2eq for the EU27 (with regional variation of EUR 30-230/ton of CO_2eq). As Table 3.3 shows, the average marginal abatement cost is EUR 114/ton of CO_2eq for 10% GHG emission reduction. For 8% GHG emission reduction, it would be EUR 107/ton of CO_2eq. Note that off-farm employment opportunities are accounted for in the simulations presented in the other sections of this chapter.

The main GHG abatement technologies included in the model for livestock-related emissions are covered manure storage (so-called floating cover) to reduce GHG (especially methane) emissions from manure storage, and injection spreading of manure in the field parcels to reduce GHG (especially nitrous oxide) emissions from manure spreading. Table 3.4 presents the GHG reduction capacity and abatement costs related to these two technological options from dairy farming. Results are presented for Farm A relative to the baseline scenario (open manure storage and broadcast spreading of manure in the field parcels).

Table 3.2. Response of farmers to decreasing GHG emission ceilings

Percentage change from baseline under 10% to 40% emission reductions

Farms	Baseline and GHG emissions reduction levels, %	Profit, % change	Milk production, % change	Concentrate feeding, % change	Nitrogen application for wheat, % change	Cereals acreage, % change	Share of green set-aside of total area, %	Methane emissions, % change
Farm A	**Base**	**125 606**	**519 303**	**10.3**	**151**	**18**	**0**	**363 015**
(High milk	10	-10.0	-15.9	-2.8	-2.6	+26.6	0	-15.8
and low	20	-21.1	-32.8	-4.8	-4.1	+61.6	7.0	-39.7
crop yields)	30	-32.8	-50.3	-6.6	-5.4	+100.1	15.0	-50.1
	40	-45.2	-68.2	-8.4	-6.5	+144.6	22.0	-68.1
Farm B	**Base**	**99 234**	**336 777**	**7.1**	**159**	**31**	**8**	**240 147**
(Low milk	10	-10.0	-18.3	+0.1	-0.3	+17.0	14.0	-18.3
and low	20	-20.1	-36.6	+0.3	-0.6	+34.0	20.0	-36.7
crop yields)	30	-30.3	-55.1	+0.4	-0.9	+51.2	26.0	-55.1
	40	-40.5	-73.5	+0.5	-1.2	+68.6	32.0	-73.6
Farm C	**Base**	**159 140**	**509 974**	**6.0**	**181**	**37**	**19**	**365 986**
(Low milk	10	-9.0	-14.1	+0.8	-8.0	+8.8	24.0	-15.5
and high	20	-18.9	-30.0	+0.8	-8.3	+18.5	27.0	-31.8
crop yields)	30	-29.0	-46.1	+0.8	-8.6	+28.4	30.0	-47.7
	40	-39.2	-62.3	+0.9	-8.9	+38.3	33.0	-63.4
Farm D	**Base**	**146 184**	**521 667**	**14.0**	**184**	**40**	**3**	**358 750**
(High milk	10	-8.7	-15.4	+0.2	-1.1	+7.5	10.0	-15.4
and high	20	-17.7	-31.2	+0.2	-1.4	+15.2	16.0	-31.2
crop yields)	30	-26.8	-47.1	+0.2	-1.7	+23.1	21.0	-47.1
	40	-36.0	-63.0	+0.2	-2.1	+30.9	27.0	-63.0

Table 3.3. Marginal abatement costs (MACs) for farms

Reductions in tons of CO_2eq, MAC in EUR per ton of CO_2eq

	10%		20%		30%		40%	
	Reduction	MAC	Reduction	MAC	Reduction	MAC	Reduction	MAC
Farm A	64.26	50.2	128.52	88.2	192.78	109.0	257.04	125.7
Farm B	51.55	192.1	103.10	193.1	154.65	194.1	206.20	195.0
Farm C	65.00	108.0	130.00	119.7	195.00	148.6	260.00	168.0
Farm D	66.94	107.0	133.89	133.4	200.83	149.2	267.77	159.9

Table 3.4. GHG abatement technologies for Farm A

	Manure storage with floating cover	Manure injection spreading
Total GHG reduction, tons (%)	3.65 (0.57%)	7.96 (1.24%)
Abatement cost, EUR per ton of CO_2eq	59	208
Reduction of total livestock emissions (%)	0.9	1.2

The GHG reduction capacity appears relatively limited for both technologies as covered manure storage reduces total GHG emissions from the farm only by about 0.6% and livestock GHG emissions by 0.9%. Manure injection spreading has slightly higher GHG abatement capacity as it reduces both total GHG emissions and livestock GHG emissions by about 1.2%. The abatement cost per ton of CO_2eq is lower for covered manure storage than for manure injection spreading, but the bottom line is that both technologies have relatively limited capacity for GHG abatement.

This chapter analyses only a limited set of technological abatement options. Other technological options, such as fat supplementation in ruminant diets to reduce enteric methane emissions or anaerobic digestion to reduce methane emissions from manure storage, exist and may be not only more effective but also more cost-effective. For example, MacLeod et al. (2015[1]) review the literature and reports, e.g. EUR 70/ton of CO_2eq for fat supplementation diet (EU15) and EUR 77-214/ton of CO_2eq for on-farm digesters (EU27). More significant dairy production system changes were analysed, for example, in a report published by the French Ministry of Agriculture in 2016,[10] in which low-input, low-emission-intensive and economically viable dairy production systems were identified.

GHG emission tax, abatement subsidy, and cap-and-trade

Compared to command-and-control measures – such as enforcing specific abatement technologies by law – a tax, abatement subsidy, and cap-and-trade on GHG emissions give farmers the freedom to adjust to the tax, subsidy or permit price. Because of their cost-effectiveness, these tools should be considered as preferred policy options. However, they require monitoring of all GHG emissions on-farm including, for example, herd size, milk yields, fertiliser application levels of different crops, manure storage, and manure application techniques.[11] Taking into account these potential implementation issues, this chapter considers other more practical but less cost-effective policy instruments.

The effect of three GHG emission tax and abatement subsidy rates are tested in the following analysis: a rate of EUR 9/ton of CO_2eq, which corresponds to the European Emission Allowance price in January 2018; a rate of EUR 30/ton of CO_2eq, which is a lower-end estimate of climate damage cost of CO_2 emissions according to the OECD (2016[5]); and a rate of EUR 50/ton of CO_2eq, which models indicate is the required value to limit the temperature increase to 1.5°C, in line with the more ambitious target of the Paris Agreement (Rogelj et al., 2015[6]).

Results from the simulations of the three emission tax rates (EUR 9, EUR 30, and EUR 50/ton of CO_2eq) show that the mitigation effectiveness of the tax varies with the specificities of each farm, i.e. how dairy production responds to the tax level (Table 3.5).

Farm C keeps the herd unchanged up to a tax level of EUR 30/ton of CO_2eq and pays the tax (EUR 19 401) rather than reduce the number of dairy cows. Only the intensity of fertiliser use is somewhat adjusted, leaving overall GHG emissions almost unchanged. A tax level of EUR 50/ton of CO_2eq pushes the farmer to reduce somewhat dairy herd size, thus decreasing GHG emissions. The farm nevertheless pays a significant amount of tax (EUR 25 745). Farm B is characterised by both low milk and crop yields and has the lowest profit in the baseline, which implies low returns to labour and land. The lowest tax rate of EUR 9/ton of CO_2eq renders milk production unprofitable and triggers the farm to switch to crop production only. In Farms A and D, EUR 30/ton of CO_2eq is the critical tax rate where milk production is abandoned. Reduction of the number of dairy cows or switching the production line not only have a significant impact on GHG emissions, but on the profitability of production as seen from profits without tax payments. Moreover, the reduction of milk production or a switch to only crop production significantly reduces the labour input requirements. Depending on employment opportunities, the reduced labour input for farming creates the possibility to work part-time off-farm and to earn off-farm income in order to compensate for the decrease in farm income.

Table 3.5. Impact of three emission tax-rates on production, GHG emissions and profits

EUR 9, 30 and 50/ton of CO_2

	Dairy cows, number	Nitrogen application for wheat, kg/ha	Land allocation, CO:Si:P:Se:GSe[1]	GHG emissions, g CO_2eq	Profit, EUR	Tax payments, EUR	Profit without tax payments, EUR
Farm A (high milk, low crop yields)							
Base	57	151	18-42-40-0-0	642 600	125 606	0	125 606
9	52	148	21-39-40-0-0	602 795	112 629	5 425	118 054
30	0	141	60-0-0-0-40	228 979	32 902	6 869	39 771
50	0	136	60-0-0-0-40	226 363	28 348	11 318	39 667
Farm B (low milk, low crop yields)							
Base	42	159	31-29-32-0-8	515 504	99 234	0	99 234
9	0	156	60-0-0-0-40	235 648	42 169	2 121	44 290
30	0	150	60-0-0-0-40	232 721	37 251	6 982	44 233
50	0	144	60-0-0-0-40	230 105	32 623	11 505	44 128
Farm C (low milk, high crop yields)							
Base	63	181	37-23-21-0-19	649 996	159 140	0	159 140
9	63	178	37-23-20-0-20	648 939	153 295	5 840	159 135
30	63	171	37-23-19-0-21	646 698	139 691	19 401	159 092
50	43	166	44-16-13-0-27	514 899	102 116	25 745	127 861
Farm D (high milk, high crop yields)							
Base	57	184	40-20-40-0-0	671 539	146 159	0	146 159
9	43	182	45-15-26-0-14	559 610	119 996	5 036	125 033
30	0	175	60-0-0-0-40	244 510	54 439	7 335	61 774
50	0	170	60-0-0-0-40	241 894	49 575	12 095	61 670

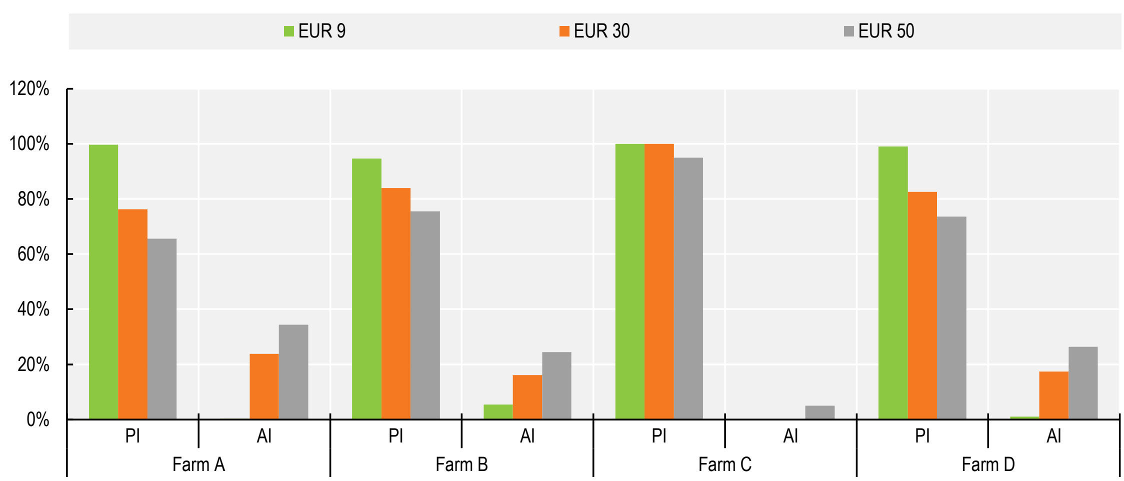
Note: [1] Cereals and oilseeds (CO), Grass silage (Si), Pasture (P), Set-aside (Se) and Green set-aside (GSe).

A GHG abatement subsidy provides the same incentives for emission reduction as a tax, which also implies exactly the same adjustments and results. Input use, land allocation and dairy herd size are exactly as those presented for GHG emission tax in Table 3.5. The only difference is that an abatement subsidy increases farm profits while an emission tax reduces them. In the long run, however, their GHG mitigation impacts are not equivalent as they have totally different impacts on the entry-exit margin of production: the abatement subsidy induces entry to the sector while a tax induces exit from the sector. A tax and a subsidy are also naturally different from the viewpoint of net government revenue, as a tax increases net revenues whereas a subsidy decreases them. Furthermore, a subsidy violates the "beneficiary pays" principle and might be considered unfair if other agents in the economy are subject to environmental taxes or costly command-and-control measures.

A GHG abatement subsidy changes the income portfolio of the farms given that in addition to income from production there is also income from GHG abatement. Figure 3.2 illustrates the shares of production income (PI) and GHG abatement income (AI) at different abatement subsidy levels (EUR 9, EUR 30, and EUR 50/ton of CO_2eq). This figure does not account for potential off-farm employment income and includes only the total income from production and GHG abatement. At the highest abatement subsidy level (EUR 50/ton of CO_2eq), income from abatement represents between 24% and 34% of total income for Farms A, B, and D, while for Farm C it represents only 5%. Hence, income shares of production and GHG abatement depend on how farms respond to the abatement subsidy by adjusting dairy herd size, land use, and nitrogen application.

Figure 3.2. Shares of production and GHG abatement income under different abatement subsidy levels

EUR 9, EUR 30 and EUR 50/ton of CO_2eq



Note: Production income (PI) and abatement income (AI).

A cap-and-trade scheme (emission trading) is often presented as a third cost-effective mitigation policy instrument as it provides the same incentives for emission reductions and implies exactly the same adjustments regarding input use, land allocation, and dairy herd size. Compared to a GHG abatement subsidy, emission trading has no impact on a government's budget, as emission permits are distributed freely and are not auctioned, while net-buyers of permits in the agriculture sector finance GHG abatement rather than taxpayers, as is the case for an abatement subsidy.

This analysis tests a cap-and-trade scheme to illustrate the efficiency gains of market-based policy instruments in their ability to target GHG abatement to those farms that can mitigate GHGs at the least cost. In our example, a government caps the total GHG emissions and allocates a certain amount of tradable emissions permits to each farm free of charge (i.e. grandfathering). The original permit allocation is based on each farm's baseline emissions minus 20%. Thus, emission reductions and permit allocations are equal in relative terms across the four farms under study.

The effects of this trading scheme are compared to those of the uniform 20% emission constraint to identify the efficiency gains of a cap-and-trade system (Table 3.6). The results show that the trading scheme generates an average efficiency gain of 17%; that is, the average cost of meeting the emission target is 17% lower with trading than with a uniform emission constraint. Large net-sellers (Farm A) reduce their compliance costs by 35%, and large net-buyers reduce them by 34%. Farms C and D, which have marginal abatement costs close to the equilibrium permit price (EUR 127.5/tons of CO_2eq), gain only slightly (1%) from trading relative to a uniform emission restriction.

Table 3.6. Gains from GHG emission trading relative to uniform emission constraint

Farms	Baseline emissions, tons of CO_2eq	Allocation of permits, tons of CO_2eq	Abatement cost without trading, EUR	Permit sales (-)/ purchases (+)	Net cost with trading, EUR	Gains from trading, EUR
Farm A	643	514	11 336	-101	7 350	3 985
Farm B	516	412	19 909	103	13 145	6 763
Farm C	650	520	15 560	-24	15 371	189
Farm D	669	536	17 860	22	17 727	133
SUM	2 478	1 982	64 665	0	53 593	11 071

Input taxes on ruminant heads and nitrogen fertiliser

Because detailed reporting on farm processes responsible for GHG emissions might be costly for both farmers and programme administrators, policies that would tax emission drivers (inputs) that can be more easily observed than emissions themselves are analysed. This more easily implemented tax comes at the cost of reduced economic efficiency as not all emissions are taxed and emissions linked to a driver such as herd size vary from farm to farm, such that the marginal emission cost carried by the farmer might be different from the (implicit) tax rate per CO_2eq targeted.

Two input taxes on GHG emissions drivers were analysed: an input tax on nitrogen fertiliser and an input tax on ruminant heads.[12] The same three levels of CO_2eq tax rates (EUR 9, EUR 30, and EUR 50/ton of CO_2eq) were used to analyse the second-best policy instruments, mapped into a tax per unit of fertiliser or cow.

The input tax on nitrogen fertiliser is based on CO2eq emissions from nitrogen fertiliser application. The tax rates applied to nitrogen fertiliser corresponding to the three levels of CO_2eq taxes are: 3.1% (EUR 9/ton of CO_2eq), 10.3% (EUR 30/ton of CO_2eq), and 17.1% (EUR 50/ton of CO_2eq). The input tax on ruminant heads targets GHG emissions from dairy herd, including enteric fermentation and manure management.

The input tax on nitrogen fertiliser has a relatively strong mitigation impact on Farm B only, where total GHG emissions reductions are 6%, 20%, and 33% under the respective tax rates (Table 3.7). This is due to the reduced herd size despite the fact that the fertiliser tax does not directly target dairy-production-related emissions. Here, the low concentrate and high grass silage use per dairy cow leads to a strong pass-through of the somewhat increased silage production costs on the profitability of milk production.

An input tax on ruminant heads directly targets GHG emissions from dairy production and has a strong mitigation impact in all other cases except Farm C, which has the highest MAC. Here, even an implicit tax rate of EUR 30/ton of CO_2eq does not reduce the herd size. This is similar to the effect of a tax on emissions, except that at EUR 50/ton of CO_2eq. the effectiveness of the tax on ruminant heads is only half that of a tax on emissions for Farm C. This is explained by the inability of the former to induce mitigation in crop activities.

Table 3.7. Detailed impact of a tax on nitrogen fertiliser and ruminant heads

EUR 9, EUR 30 and EUR 50/ton of CO_2eq

	Input tax on nitrogen fertiliser			Input tax on ruminant heads		
	Dairy cows, number	Profit, EUR	GHG emissions, kg CO_2eq	Dairy cows, number	Profit, EUR	GHG emissions, kg CO_2eq
Farm A (high milk, low crop yields) - Base	**57**	**125 606**	**642 600**	**57**	**125 606**	**642 600**
9 EUR/ton of CO_2eq	57	124 675	640 268	52	113 387	602 795
30 EUR/ton of CO_2eq	57	122 223	632 899	0	34 419	228 979
50 EUR/ton of CO_2eq	53	117 318	612 485	0	34 419	228 979
Farm B (low milk, low crop yields) - Base	**42**	**99 234**	**515 504**	**42**	**99 234**	**515 504**
9 EUR/ton of CO_2eq	37	92 511	483 579	0	42 624	235 648
30 EUR/ton of CO_2eq	27	77 377	411 497	0	42 624	235 648
50 EUR/ton of CO_2eq	17	63 725	346 317	0	42 624	235 648
Farm C (low milk, high crop yields) - Base	**63**	**159 140**	**649 996**	**63**	**159 140**	**649 996**
9 EUR/ton of CO_2eq	63	158 143	649 295	63	153 905	649 311
30 EUR/ton of CO_2eq	63	155 862	647 750	63	141 720	647 804
50 EUR/ton of CO_2eq	63	153 749	646 385	54	119 095	585 950
Farm D (high milk, high crop yields) - Base	**57**	**146 159**	**671 539**	**57**	**146 159**	**671 539**
9 EUR/ton of CO_2eq	57	145 184	668 314	46	125 162	584 081
30 EUR/ton of CO_2eq	57	142 911	665 509	0	55 957	244 510
50 EUR/ton of CO_2eq	57	140 826	662 963	0	55 957	244 510

In other cases, especially on Farms C and D, the number of dairy cows is not affected even with the highest tax rate of EUR 50/ton of CO_2eq. As a result, GHG emissions reductions are relatively modest, ranging from less than 1% (Farms C and D) to 5% (Farm A), even under the highest tax rate.

Table 3.8 provides a summary of the impacts of the emissions- and input-based taxes and subsidies on GHG emissions and farm income with and without tax or subsidy payments for an equivalent of EUR 30/ton of CO_2eq. Results show that the GHG emission tax and abatement subsidy have the same impact on GHG emissions and farm income without tax or subsidy payments. The only difference is found with farm income, which is lower with a GHG tax than with an abatement subsidy. An input tax on ruminant heads closely resembles the emissions-based instruments at EUR 30/ton of CO_2eq emission price, since it leads to the same adjustments regarding dairy herd size, which is a key driver for changes in both emission reduction and farm income. The input tax on nitrogen fertiliser has relatively modest impacts on GHG emissions and income, except in the case of Farm B.

Table 3.8. Impacts of the emissions-based and the input-based policy instruments on GHG emissions and profit

	GHG emissions, kg CO2eq	Profit with tax or subsidy payment, EUR	Profit without tax or subsidy payments, EUR
Farm A (high milk, low crop yields) - Base	**642 600**	**125 606**	**125 606**
GHG emission tax	-64%	-74%	-68%
GHG abatement subsidy	-64%	-58%	-68%
Input tax on nitrogen fertiliser	-2%	-3%	0%
Input tax on ruminant heads	-64%	-73%	-68%
Farm B (low milk, low crop yields) - Base	**515 504**	**99 234**	**99 234**
GHG emission tax	-55%	-62%	-55%

	GHG emissions, kg CO2eq	Profit with tax or subsidy payment, EUR	Profit without tax or subsidy payments, EUR
GHG abatement subsidy	-55%	-47%	-55%
Input tax on nitrogen fertiliser	-20%	-22%	-19%
Input tax on ruminant heads	-54%	-57%	-55%
Farm C (low milk, high crop yields) - Base	**649 996**	**159 140**	**159 140**
GHG emission tax	-1%	-12%	0%
GHG abatement subsidy	-1%	0%	0%
Input tax on nitrogen fertiliser	0%	-2%	0%
Input tax on ruminant heads	0%	-11%	0%
Farm D (high milk, high crop yields) - Base	**671 539**	**146 159**	**146 159**
GHG emission tax	-64%	-63%	-58%
GHG abatement subsidy	-64%	-50%	-58%
Input tax on nitrogen fertiliser	-1%	-2%	0%
Input tax on ruminant heads	-64%	-62%	-58%

Note: Impacts evaluated with a tax rate of EUR 30/ton of CO_2eq.

Afforestation of agricultural land for carbon sequestration[13]

Carbon sequestration practices on agricultural lands may hold large potential and need to be considered in the overall mix of GHG mitigation options.

Compared with measures that reduce annual GHG emission flows, carbon sequestration measures face several policy design challenges: dynamics, additionality, permanence, and leakage. Carbon sequestration practices increase carbon storage with diminishing rate until they plateau at a new equilibrium, which may take 20-100 years. Policy needs to encourage sequestration practices that are additional (i.e. that would not have happened without a specific policy). Some sequestration practices, such as no-till and green set-aside, are relatively easily reversed, which would lead to a loss of the sequestration benefits. Because of potential impermanence, soil carbon sequestration practices may not have the same climate protection benefits as technological changes that permanently reduce GHG emissions. Finally, leakage occurs when a soil carbon sequestration project increases GHG emissions elsewhere due to production displacements.

Considering the potential of carbon sequestration practices on agricultural lands, Lal (2004[7]) estimates that agricultural soils can offset 15% of global GHG emissions. However, the global potential for carbon sequestration in agriculture, forestry, and land use sectors (AFOLU) remains uncertain. Recent estimates show that technical global carbon sequestration potential ranges between 2.6 and 4.8 Gt CO_2 at carbon prices between USD 20 and USD 100/ton of CO_2eq (Smith et al., 2015[8]).

This chapter presents a policy scenario supporting carbon sequestration through afforestation. Afforestation as a carbon sequestration practice is less easily reversible and its additionality is clearer than, for example, no-till adoption. Like many other soil carbon sequestration practices, afforestation provides ancillary environmental benefits, including improved water quality. This scenario introduces a subsidy for carbon sequestration (EUR 9, EUR 30, and EUR 50/ton of CO_2eq) applied to GHG mitigation induced by the afforestation of agricultural land.[14] Afforestation provides abatement benefits through above (trees crown) and below ground (roots) carbon sequestration. Kolari et al. (2004[9]) find that Scots pine (Pinus sylvestris) forests on mineral soils act as a sink with an average sequestration of 5 085 kg CO_2/ha/year over an 80-year rotation period. The annual net-revenue over one rotation period (80 years) of afforested land is assumed to be EUR 47.8/hectare.

This afforestation subsidy does not induce much change. Afforestation of agricultural land is a profitable option to mitigate GHG emissions only in Farm A, and in this case afforestation only takes place at the highest subsidy level of EUR 50/ton of CO_2eq. Other farms do not adopt afforestation as a mitigation option even at the highest subsidy level. Relative to the baseline for Farm A, under the afforestation subsidy, land allocation shifts from cereals and grass silage towards afforestation (20 hectares of land is afforested) and dairy herd size reduces from 57 to 55. The total emissions of Farm A decrease by 26% and this reduction is mainly due to afforestation through conversion of land from emissions source to sink. The decrease in the size of the dairy herd plays a small role only in decreasing emissions relative to the baseline.

In view of the poor GHG abatement performance of afforestation subsidies under carbon prices (EUR 9, EUR 30, and EUR 50/ton of CO_2eq), it is not considered further as a mitigation option.

Mitigation policy instruments and the role of sunk investment costs

Sunk investment costs[15] play a key role in farm adjustments and production response to different mitigation policy instruments, and can lead to substantially different reactions in the short and long run. This is especially the case in dairy farming, which is characterised by long-lasting investments in stables and milking parlours that account for a larger share of overall production cost. The following analysis tests the case of all fixed costs assumed as sunk (instead of the 50% assumed so far), simulating the short-run response. This simulates a situation where farms continue dairy operations as long as market revenues exceed the variable costs of milk production (including any GHG taxes).

As expected, the mitigation responses of all farms are drastically reduced when all fixed costs are assumed as sunk. Comparing results in Table 3.9 with corresponding results in Table 3.5 (GHG emission tax) and Table 3.7 (input tax on ruminants) shows the significant impact of fixed investment costs on adjustment possibilities of dairy farms under GHG mitigation policies. If investment costs for dairy operations are sunk and the opportunity costs of farm labour are low (the farmer has no opportunity to work off-farm), the GHG mitigation effectiveness of an emission tax or an input tax on ruminants is drastically lower in the short run than the medium-term adjustments shown in Tables 3.5 and 3.7. If farmers view all their investments in dairy production as sunk costs, they will keep the dairy herd and continue production even with high tax rates. As a result, reductions in GHG emissions are modest. In a real-world situation this would correspond to farmers who have invested recently and will continue to produce for several years rather than make adjustments. Farmers who will need to invest over the medium term would rather adjust their herd size as discussed above.

Table 3.9. Performance of GHG emission and input taxes on ruminant heads (EUR 30/ton of CO2eq) under the assumption that all dairy investments are sunk costs

	Dairy cows, number	Milk production, kg	GHG emissions, kg CO_2eq	Profit, EUR	Tax payments, EUR
Farm A (high milk, low crop yields) – Base	**58**	**517 843**	**639 431**	**148 572**	**0**
GHG emission tax (EUR 30/ton of CO_2eq)	58	513 966	624 002	129 608	18 720
Input tax on ruminant heads (EUR 30/ton of CO_2eq)	58	513 966	624 002	132 137	16 191
Farm B (low milk, low crop yields) – Base	**63**	**499 669**	**642 970**	**149 270**	**0**
GHG emission tax (EUR 30/ton of CO_2eq)	63	498 639	631 464	130 148	18 944
Input tax on ruminant heads (EUR 30/ton of CO_2eq)	63	498 639	631 464	132 677	16 415
Farm C (low milk, high crop yields) – Base	**62**	**510 752**	**661 683**	**169 232**	**0**
GHG emission tax (EUR 30/ton of CO_2eq)	62	511 059	654 127	149 495	19 624
Input tax on ruminant heads (EUR 30/ton of CO_2eq)	62	510 643	655 373	151 764	17 359
Farm D (high milk, high crop yields) – Base	**57**	**520 322**	**667 369**	**169 087**	**0**
GHG emission tax (EUR 30/ton of CO_2eq)	57	520 322	661 067	149 160	19 832
Input tax on ruminant heads (EUR 30/ton of CO_2eq)	57	520 322	662 472	151 505	17 493

Ancillary environmental costs and benefits of GHG mitigation policies

GHG mitigation policy instruments incentivise farmers to change input use, land allocation and production technologies. These adjustments may have significant ancillary environmental benefits or costs.

The indirect effects of mitigation efforts on nitrogen runoff (reflecting water quality impacts) are not straightforward and may vary according to the stringency of the mitigation policy applied (as seen for Farm A in Table 3.10). Results indicate that how this impacts water quality may depend on the chosen level of tax rate. For a GHG abatement subsidy, a GHG emission tax and an input tax on ruminants, the impact on water quality is first positive at the low tax rate of EUR 9/ton of CO_2eq. The level of nitrogen runoff decreases by about 5% and counts as an environmental co-benefit of the GHG mitigation policy. This stems from the fact that at a low tax rate fertilization levels are adjusted, but the dairy herd is not reduced (by much), such that the grass silage area with lower nitrogen losses than wheat is more or less constant. At higher tax rates, however, the impact on water quality is negative (increased nitrogen runoff by about 11%). This is because higher tax rates reduce dairy herd size, and more land is allocated to cereals production and away from pasture and grass silage that have lower propensity for nutrient runoff. Moreover, the negative water quality impact is larger under a tax rate of EUR 30/ton of CO_2eq than EUR 50/ton of CO_2eq. For Farm A, under the three instruments (CO_2 tax or subsidy, ruminant tax), the strongest adjustment in terms of dairy cow numbers and land allocation towards cereals takes place when the tax rate is EUR 30/ton of CO_2eq. Under a tax rate of EUR 50/ton of CO_2eq there is no additional change in dairy cow numbers or land allocation, but the tax induces reduced application of nitrogen fertiliser leading to lower nitrogen runoff, and thus a lower negative impact on water quality relative to a lower tax rate of EUR 30/ton of CO_2eq. Potential ancillary environmental effects depend not only on a given type of GHG mitigation policy instrument, but also on its intensity. The input tax on nitrogen provides environmental co-benefits with all CO_2eq tax rates.

Table 3.10. Impact of GHG mitigation instruments on nitrogen runoff for Farm A

Percentage change from base

Instrument	EUR 9 /ton CO_2eq	EUR 30 /ton CO_2eq	EUR 50 /ton CO_2eq
Baseline N runoff	1351	1351	1351
GHG abatement subsidy	-5.1%	+11.1%	+7.5%
GHG emission tax	-5.1%	+11.1%	+7.5%
Input tax ruminants	-5.1%	+11.1%	+7.5%
Input tax fertiliser	-2.0%	-6.5%	-5.1%

Ranking alternative policy instruments by cost effectiveness

While the previous sections point to significant variations in cost-effectiveness across policy instruments, they also highlight that some are less dependent on GHG emission monitoring than others. Since the latter may have consequences on the cost of policy implementation, this section reviews the relative cost-effectiveness of GHG mitigation policy instruments with and without transaction costs. Accounting for transaction costs: improves comparison among and screening of alternative policy instruments; can help the design and implementation of effective policy instruments to achieve objectives; improves the evaluation of policy instruments; and helps track budgetary costs of policy instruments over their entire life cycle (McCann et al., 2005[10]).

The literature on transaction costs of GHG mitigation policy instruments in the context of European agriculture is scarce. De Cara et al. (2018[11]) analyse optimal coverage of GHG emission tax in the presence of monitoring, reporting, and verification costs in the context of European agriculture. To calculate the magnitude of these for GHG emission tax in their “medium” scenario, they use EUR 2.5/ton of CO_2eq based on Ancev (2011[12]). Bakam et al. (2012[13]) assess the cost-effectiveness of a fertiliser tax, a GHG emission tax, and a permit-trading scheme based on data from Scottish agriculture. They assume zero transaction costs for the fertiliser tax, since the tax is included in the price of fertiliser. For medium-size farms, the transaction costs of the emissions tax are calculated to be 29% lower than those part of a permit-trading scheme (GBP 2 000 and GBP 2 825, respectively). Pérez Domínguez and Britz (2003[14]) analyse GHG emission trading for European agriculture and adopt a transaction cost (paid by permit buyers) of EUR 5/ton of CO_2eq for trades within EU Member States and EUR 10/ton of CO_2eq for trade between member states.

Based on the literature cited, the analysis undertaken here uses the following transaction cost estimates: EUR 3.5/tons of CO_2eq for the GHG emission tax and GHG abatement subsidy; EUR 5.0/ton of CO_2eq for GHG emission trading; and EUR 2.0/ton of CO_2eq for ruminant tax.

As expected, when comparing cost effectiveness without considering transaction costs, all the emissions-based policy instruments continue to have highest level of cost effectiveness at 10% emissions reduction as they provide exactly the same marginal incentives (Table 3.11).[16] As an input-based policy instrument, the tax on ruminant heads (which addresses only a subset of GHG emissions stemming from farms) induces slightly higher abatement costs, and thus is slightly less cost effective than the emissions-based policy instruments.

Including transaction costs improves slightly the relative performance of the tax on ruminant heads and worsens the relative performance of GHG emissions trading. The last row of Table 3.11 shows targeting gains to be about 30% for the GHG emission tax and GHG abatement subsidy relative to the input tax on ruminant heads. These targeting gains show that every euro spent on better monitoring, reporting and verification – which increases policy-related transaction costs – brings EUR 1.3 in improved cost-effectiveness. For GHG emission trading, the ratio of these targeting gains is less than one because of the relatively high transaction costs of this instrument in comparison to other emissions-based instruments, and because an input tax on ruminant heads is, as an input-based policy instrument, relatively cost-effective.

Table 3.11. Cost effectiveness of policy instruments with and without transaction costs (Farm A)

	GHG abatement subsidy, EUR/ton of CO_2eq	GHG emission tax, EUR/ton of CO_2eq	Tax on ruminant heads, EUR/ton of CO_2eq	GHG emission trading, EUR/ton of CO_2eq
Cost effectiveness	50.2	50.2	53.6	50.2
Cost effectiveness with transaction costs	53.7	53.7	55.6	55.2
Targeting gains ratio	1.32	1.32	-	0.15

Note: All instruments evaluated at 10% reduction of GHG emissions.

Discussion of results and caveats

Consistent with other studies of GHG mitigation in European agriculture, the results show rather high abatement costs in mixed dairy and crop production, at least when a high GHG emission reduction (20-40%) is targeted (Lengers, Britz and Holm-Müller, 2014[15]). The marginal abatement costs for the farms analysed in this study are in the range of results from other European studies. For an 8% GHG emission standard, De Cara et al. (2005[4]) estimate average EU15 marginal abatement costs of EUR 123/ton of CO_2eq, and Pérez Domínguez et al (2003[16]) estimate average EU27 costs of EUR 95/ton of CO_2eq (with regional variation of EUR 30-230/ton of CO_2eq.). In this study, an 8% GHG emission reduction results in average marginal abatement costs of EUR 113/ton of CO_2eq.

This study also confirms that the market-based instruments based on emissions (GHG emission tax, GHG abatement subsidy, and cap-and-trade scheme) are the most cost-effective options for GHG mitigation in agriculture. For example, they show that gains from emissions trading relative to uniform emission constraints average 17% for analysed farms. Pérez Domínguez and Britz (2003[14]) find similar gains (23%) from emissions trading relative to uniform emissions constraints for the EU27. De Cara et al. (2005[4]) compare GHG emissions tax with uniform emission constraints and show that for an 8% GHG abatement target, the cost saving ratio of GHG emission tax is 2.2. That is, meeting the abatement target is more than twice as expensive for the uniform emission constraint than for the emission tax.

Moreover, results confirm that it pays to target emissions broadly, as the emissions-based policy instruments (GHG emission tax and GHG abatement subsidy in particular) are more cost-effective, even when transaction costs related to monitoring, reporting, and verification are included. Targeting gains are about 30% for the GHG emission tax and GHG abatement subsidy relative to the input tax on ruminant heads. These targeting gains show that, for a GHG emission tax or a GHG abatement subsidy, every euro spent on better monitoring, reporting and verification, which increases policy-related transaction costs, brings EUR 1.3 through improved cost-effectiveness. Bakam et al. (2012[13]) get similar results for GHG emission trading over an input tax for GHG emission reduction targets that are over 29%. De Cara et al. (2018[11]) show that the social welfare of covering all farms or only the largest emitters of GHG emissions in European agriculture depends on the marginal social damage of GHG emissions and monitoring, reporting and verification costs. If the marginal damage of emissions is high (EUR 100/ton of CO_2eq) and monitoring, reporting and verification costs per farm stay below EUR 1 220, then full coverage is welfare improving; that is, it pays to target all farms and not only the largest emitters.

These results are subject to two main caveats. First, data and the calibration of the bio-economic model focus on four regions in Europe drawing on the database of the CAPRI model. The four farms chosen represent regional differences in crop and milk yields, crop-specific costs, milk-specific costs, mineral and organic fertiliser use, and dairy cow feeding practices. However, calibration of a bio-economic model also requires a relatively large amount of detailed bio-physical data and considering that these data were not available for certain regions, the analysis aims to illustrate the performance of policy instruments under heterogeneous productivity and profitability of production. Thus, results cannot be considered representative of either Europe or OECD countries in general. A modification of the bio-economic model structure is ongoing to facilitate the representation of diverse agricultural production structures and technologies, environmental conditions, and policy contexts in different OECD and non-OECD countries

Secondly, as always in supply-side farm-level models, the prices of outputs and inputs are considered exogenous. The assumption of exogenous output prices causes a large decrease in output, especially in dairy production, under those policy instruments that reduce the profitability of production. Since demand for agricultural products is inelastic, market prices for these products would increase when product supply decreases, and this would at least partly offset the decrease in profitability due to mitigation policies, moderating production and income losses.

Conclusions

There is increasing policy interest in exploring means to reduce GHG emissions in agriculture. Studies have shown there is a diversity of technical measures that farms could undertake with varying cost-effectiveness, but the policy levers to encourage their uptake has not been studied as much. This chapter addresses this gap by assessing the relative effectiveness and cost effectiveness of key GHG mitigation policy instruments in reducing emissions from crop and livestock (focusing on dairy) production. It looks at six policy instruments: an emission constraint, emission tax, abatement subsidy, input tax on nitrogen fertiliser, input tax on ruminant heads, and emissions trading, and applies these to the European Union. To do this, a detailed quantitative bio-economic farm model covering both production activities was developed and applied to farms representing a diversity of regional-level situations in Europe, drawing on data from the CAPRI database. On the basis of these data, four representative farms from four EU countries were developed to illustrate how differential crop and milk productivity and production costs affect the GHG mitigation effectiveness and costs of policies.

Consistent with other studies about GHG mitigation in European agriculture, the results show rather high abatement costs in mixed dairy and crop production, at least when targeting large GHG emission reduction. Study results confirm that the market-based instruments based on all GHG emissions (GHG emission tax, GHG abatement subsidy, and cap-and-trade scheme) are the most cost-effective options for GHG mitigation in agriculture. Moreover, results show it pays to target GHG emissions broadly, since the policy instruments that target all GHG emissions from farms are more cost-effective than the instruments targeting only a subset of emissions or proxies of emissions (e.g. input tax on nitrogen fertiliser or input tax on ruminant heads). This is the case even when higher policy-related transaction costs (related to monitoring, reporting and verification of emissions) are accounted for, in particular for a GHG emission tax and a GHG abatement subsidy.

The results underline the importance of investment costs and the planning horizon when evaluating GHG abatement strategies and costs in crop and dairy production. Investment costs lead to substantially different reactions by farms in the short and long run. In the short run, investment costs are sunk and farms continue dairy operations as long as market revenues exceed variable costs of milk production (including any GHG tax payments or abatement subsidies). Dairy farming is labour intensive, and the impact of sunk investment costs is intensified if farm labour input has low or zero opportunity costs (that is, no off-farm employment opportunities). As a result, in the short run, reductions in GHG emissions are likely to be modest. Long-run calculations assume that farms face fixed costs of investment in dairy production. Under this assumption, the mitigation effectiveness of policy instruments increases. This, however, varies across farm situations. For example, in the case of a GHG emission tax, some farms start reducing the number of dairy cows with the lowest emission tax level of EUR 9/ton of CO_2eq. Other farms start to adjust the size of their dairy operation only when the emission tax is EUR 50/ton of CO_2eq. The latter farms find it more profitable to pay a large amount of tax rather than to reduce GHG emissions by decreasing the number of dairy cows.

Fixed investment costs are thus likely to slow the transition to lower-carbon agriculture. Transition will take longer where fixed investment costs are higher and investments recent. This may call for temporary policy packages to facilitate transition when needed, or at least there is the need for governments to avoid uncertainty in their long-term GHG mitigation objectives and policies so that farmers can make appropriate investment decisions.

The availability of off-farm sources of income plays an important role in facilitating the development of mitigation solutions, as it helps to reduce the economic cost for farmers. Economic policies and conditions that facilitate job mobility and flexibility are likely to favour more effective mitigation policies.

The livestock sector is likely to be the most affected by mitigation policies. This has three major implications: this is where research on cost-effective mitigation practices and technologies will be needed; transition policy packages are needed for this sector; and competitiveness issues, while not discussed in this chapter, will be more important for the agriculture sector.

The effects of GHG mitigation policies may indirectly affect other environmental dimensions, such as the impact on water quality of nitrogen and phosphorus runoff through changes in input use (application of chemical fertiliser and manure) and land use (land allocation between cereals and grasslands). The results presented in this chapter show that land use change driven by mitigation policies from grasslands (grass silage and pasture) to cereals and oilseeds could increase nutrient runoff. This calls for considering ancillary benefits and trade-offs with regard to other environmental dimensions in the design of GHG mitigation policies in order to improve policy coherence. This study shows, in particular, that effective policy instruments for water quality will be important when introducing climate policies in regions where livestock is a major activity.

Overall, the results confirm that it is difficult to significantly reduce livestock GHG emissions without reducing dairy herd size. However, the real effect of reduced herd size is difficult to judge in a supply-side model as the one here. If demand is inelastic, resulting price increases could trigger production elsewhere and simply shift GHG emissions (emissions leakage) from one region to another. The type of detailed, farm-level analysis presented in this chapter must therefore be complemented with a large-scale market-equilibrium framework analysis.

Notes

[1] (Durandeau et al., 2009[34]) also adopt non-linear yield functions in their analysis of the first-best and the second-best taxation of GHG emissions from agriculture in northern France.

[2] www.capri-model.org/docs/capri_documentation.pdf.

[3] As an agricultural sector model, CAPRI combines a global partial equilibrium model for agri-food products (employing the Armington assumption to depict bi-lateral trade) with non-linear programming models for 280 NUTS2 regions, or about 3 000 farm group models, to detail agricultural production decisions in the European Union, EU candidate countries and Norway. Eurostat is the key data source for the European part of the model providing, for instance, crop and animal production statistics, land use statistics, market balances and Economic Accounts for Agriculture (EAA). The CAPRI database at national level integrates the EAA (valued output and input use) with, for instance, market balances, and trade and production statistics. The country data are subsequently used to derive a regionalised database at NUTS2 level that depicts the allocation of inputs across activities and regions, as well as acreages, herd sizes and yields. The regional data are subsequently further disaggregated to farm-type level, mainly based on farm structural data.

[4] CAPRI is used in other GHG mitigation studies as well. For example, Pèrez Dominguez et al. (2003) derive marginal abatement costs of GHG emissions from CAPRI at regional level for analysing an EU-wide trading scheme of GHG emission permits for agriculture while explicitly including transaction costs in permit trading.

[5] Terminology used referring to low and high milk and crop yields is relative to other farms modelled.

[6] Farmers are assumed to be risk-neutral, in which case fully decoupled support does not affect farmer choices regarding input use and land use.

[7] Their impact will be illustrated later using a sensitivity test, assuming that all fixed costs are sunk instead of only 50% of fixed costs of investment, reflecting a situation where the farm would continue dairy operation as long as market revenues exceed the variable costs of milk production (including taxes).

[8] Other calculations assume that farmers have off-farm employment opportunities with wage-rate of EUR 13/hour. However, off-farm income is not reported in any of the result tables as it is important to show

how farm income changes due to adjustments driven by policy instruments.

[9] If farmers have off-farm employment opportunity and they use saved labour input from farming (e.g. due to reduction in dairy herd size) to work part-time off-farm to earn off-farm income, the forgone income (sum of farm income and off-farm income) at a given emission constraint level would not reflect fully the cost of adjustments in agriculture due to emission constraint.

[10] Entitled "Les exploitations d'elevage herbivore economes en intrants (ou autonomes): quelles sont leurs caracteristiques? Comment accompagner leur developpement?"

[11] (Grosjean et al., 2016[35]) discuss potential barriers to pricing agricultural GHG emissions in Europe. Since transaction costs also depend on the existing institutional frameworks and because agriculture sector is already regulated (such as the Nitrate directive) and subsidised through the Common Agriculture Policy (CAP), various monitoring, reporting, and verification tools are already in place, which would help to implement GHG emissions based policy instruments.

[12] (De Cara and Jayet, 2000[36]) show that taxation of animals and their feed, the second-best policies for methane reduction, produces significant results in terms GHG emission reductions. However, their analysis finds subsidy for afforestation of set-aside land to be even more effective.

[13] Carbon sequestration is dealt relatively briefly here as it will be analysed in the synthesis report of the whole project. Moreover, soil carbon sequestration and voluntary provision of carbon offsets to carbon credit markets have been analysed in detail in (Lankoski et al., 2015[37]).

[14] In addition to afforestation of agricultural land there are other options, which have less impact on the amount of agricultural land, such as agroforestry or planting trees on field boundaries.

[15] Sunk cost refers to a cost that has been incurred and cannot be recovered. Thus, it should not affect rational decision-making.

[16] Marginal incentives and cost-effectiveness are same for these three instruments. They do differ in terms of distributional impacts (farm income and government net-revenues).

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Annex 3.A. Key parametric equations of the empirical model

This annex provides a brief description of the key parametric equations of the model.

Total intake function of dairy cows, kg DM/animal/year

$$intake(v) = (v - 0.163v - 0.0188v^2 + 13.4) * 365$$

In the intake function *v* denotes concentrate feed intake, and silage intake is given be *intake(v) – v*. Intake function is from Huhtanen et al. (2008[17]).

Milk production function, kg/animal/year

$$g(v) = (20.09 + 1.252\, v - 0.04\, v^2) * 300$$

The quadratic milk production function is based on Lehtonen (2001[18]). By assumption each cow has 300 milking days and cows are dry the rest of the year. Milk production peaks roughly at 16-17 kg of the concentrate feeding.

Manure excretion to cowshed, m^3/animal/year

To determine the manure excretion, one needs to define the following shares of animals in the farm. A notion of *production animal* refers to the steady-state process needed to maintain one lactating cow and is technically a composition of one lactating cow, 1/3 calf and 1/3 heifer. Thus,

$$shareD = 1,\ shareH = \frac{1}{slaught},\ shareC = \frac{1}{slaught}$$

where $slaught$ is the number of milking seasons before a dairy cow is slaughtered, and the ending D stands for dairy cows, H for heifers and C for calves. Using this notation, the manure excretion to cowshed and pasture, respectively, can be defined as follows (Nennich et al., 2005[19]):

Manure excretion to cowshed, m^3/production animal/year

$$w(v) = \frac{[(intake(v)*w1+w0)\ shareD+wC*shareC+wH*shareH](1-wp)365\ scale}{1000} + h2o$$

Manure excretion to pasture, m^3/ production animal/year

$$wpa(v) = \frac{[(intake(v)*w1+w0)\ shareD+wC*shareC+wH*shareH]wp\ 365\ scale}{1000}$$

Manure N content

To determine the manure N content, one needs to account for the share of manure N evaporated as NH_3-N. This evaporation is affected by manure storage and spreading technologies and defined as $ammonia^{ij} = emstor^i * emsperad^j$, where $i = \{1,2\} = \{no\ cover, floating\ cover\}$ and $j = \{1,2\} = \{broadcast, injection\}$. Manure N content, kg N/m^3 manure/year (ThetaN) and total N excretion in manure, kg N/animal/year (Nexcr), respectively, are based on Nennick et al. (2005[19]) and given by

$$ThetaN(v) = \frac{(1-ammonia^{ij})\ scale\left(intake(v)\left(\frac{v}{intake(v)}*vcp+\left(1-\frac{v}{intake(v)}\right)scp\right)N1+BWD*N2\right)365}{(w(v)+wpa(v))1000}$$

$$Nexcr(v) = \frac{(1-ammonia^{ij})\, scale\left(intake(v)\left(\frac{v}{intake(v)}*vcp+\left(1-\frac{v}{intake(v)}\right)scp\right)N1+BWD*N2\right)365}{1000}$$

Manure transport and application cost

Costs for manure spreading and transportation are determined based on the distance, amount of manure, spreading technology, gear capacity, and contractor charge as follows.

$$em(r,m) = \frac{m}{spcap}\left(\frac{2*r}{trsp} + load * spcap + \frac{spread^{i}}{60} * spcap\right) * spp^{i} + ctran * r * m$$

where $i = \{1, 2\} = \{broadcast, injection\}$, m is m^3/manure and r is distance in km.

CH_4 emissions from enteric fermentation

Methane emissions are calculated applying a procedure from GHG inventory calculations that follow IPCC's recommendations. Calculation is based on the following set of equations, which also account for the possible abatement through fat supplementation. Equations are based on the inventory reporting of Statistic Finland (2016[20]). When estimating CH_4 emissions from enteric fermentation, the diet's gross energy digestibility is calculated only for dairy cows, and the same value is used for calves and heifers for simplification. The same set of equations are thus used for calculating the emissions for dairy cows, heifers and calves. Dry cows are not accounted for separately.

$$EFem(v) = \frac{(GED(v)*shareD+GEH(v)*shareH+GEC(v)*shareC)*Ym*365}{55.65} * \frac{(100-fatinc*4)}{100}$$, where

$$GEX(v) = \frac{\frac{NEmX+NEaX+NE1(v)+NEpX}{REM(v)} + \frac{NEgX}{REG(v)}}{DE(v)/100}$$

$$REM(v) = 1.123 - \left(4.092 * 10^{-3} * DE(v)\right) + (1.126 * 10^{-5} * DE(v)^2) - \frac{25.4}{DE(v)}$$

$$REV(v) = 1.164 - \left(5.160 * 10^{-3} * DE(v)\right) + (1.308 * 10^{-5} * DE(v)^2) - \frac{37.4}{DE(v)}$$

$$NEgX = 22.02 * \frac{BWX}{(CoX*MW)^{0.75}} * WGX^{1.097}$$

$NEpD = CpD * NEmD$ (only for dairy cows)

$NE1(v) = \frac{g(v)}{300} * (1.47 + 0.40 * fat)$ (only for dairy cows)

$$NEaX = \left(cap * \frac{tpX}{365} + cao * \left(1 - \frac{tpX}{365}\right)\right) * NEmX$$

$$NEmX = CfiX * BWX^{0.75}$$

$DE(V) = -11.3 + 0.977 * \frac{seosoas(v)}{10}$, where $seosoas(v)$ is the share of digestible organic matter of the total organic matter as g/kg DM

$X = \{D, H, C\}$

GHG emissions from manure storage

Manure storage is a source of both methane and nitrous dioxide emissions. They are defined using the following equations.

***CH_4 emissions from storage*, kg CH_4/animal/year** (based on Statistic Finland (2016[20])

$$EFmm(v) = \left(GED(v) * \left(1 - \frac{DE(v)}{100}\right) + 0.04\right) * \left(\frac{1-ash}{18.45}\right) * 365 * chmax * 0.67 * mcf^{i}$$

where mcf^1 is storage without cover and mcf^2 is storage with floating cover

Direct N_2O emissions from storage, kg N_2O/animal/year (based on Statistic Finland (2016[20])

$$EFmn(v) = \frac{Nexcr(v)*wp}{(1-ammonia^{ij})} * ef^i * \frac{44}{28}$$

where ef^1 is storage without cover and ef^2 is storage with floating cover

GHG emissions from manure management

Manure storage and spreading cause NH_3 emissions and based on those indirectly N_2O emissions. Drawing on Statistic Finland (2016[20]) and Grönroos (2015[21]) they can be expressed using the following equations.

NH_3 emissions from manure management, kg NH_3-N/m^3 manure/year

$$ThetaNvol(v) = \frac{ThetaN(v)}{(1-ammonia^{ij})} * ammonia^{ij}$$

Indirect N_2O emissions from manure managements, kg N_2O/m^3 manure/year

$$EFmni(v) = ThetaNvol(v) * 0.01 * \frac{44}{28} + ThetaN(v) * 0.01 * \frac{44}{28}$$

Emissions from fertiliser use, machinery and soil (kg CO_2eq./ha) from cultivated land

The GHG emissions from cultivated land comprise autonomous soil emissions (soil N_2O emissions due to fertilization are assumed to be included here, i.e. they are not accounted for separately) and emissions from cultivation practices, yield transportation to processing, crop drying and manufacturing mineral fertilisers.

$$ghgX(N) = autoX + cultX + emtrans + emdry * y^X(N) + emprod * N$$

where X is {s, c}={silage, barley}

Emissions from fertiliser use, machinery and soil (kg CO_2eq./ha) from pasture land

The GHG emissions from pasture land are calculated based on Statistic Finland (2016[20]) with additional terms for autonomous soil emissions, cultivation practices and mineral fertiliser manufacture.

$$empas(v,H) = \left(H * wpa(v) * \frac{\frac{\frac{Nexcr(v)}{wpa(v)}}{1000}}{(1-ammonia^{ij})} \right) * 0.02 * \frac{44}{28} * N2O + (autop + cultp + emprod * lp)Ap(H)$$

Crop yield response functions

The crop nitrogen response function for rape seed and silage (kg DM yield/ha) is given by quadratic response function

$$y(N) = a + bN + cN^2$$

where *a*, *b* and *c* are parameters of a quadratic nitrogen response function. And for wheat and barley by Mitscherlich nitrogen response function

$$y^c(N) = \varphi(1 - \sigma \exp(-\rho N))$$

where and φ, σ and *ρ* are parameters of a Mitscherlich response function. The parameters of the quadratic crop yield functions are taken from Lehtonen (2001[18]) and those of the Mitscherlich yield function have been estimated by Bäckman et al. (1997[22]) on the basis of Finnish field experiments.

Nitrogen and phosphorus runoff functions

Both nitrogen and phosphorus runoff are included and in the case of phosphorus both dissolved reactive phosphorus (DRP) and particulate phosphorus (PP) runoff is estimated. Because in compound fertiliser (NPK) the three main nutrients are in fixed proportions, nitrogen fertiliser intensity determines also the amount of phosphorus used. Part of this phosphorus is taken up by the crop, while the rest accumulates and builds up soil P. Drawing on Finnish field experiment studies it is assumed that 1 kg increase in soil phosphorus reserve increases the soil P status (i.e. ammonium acetate-extractable P) by 0.01 mg/l soil. Uusitalo and Jansson (2002[23]) estimated the following linear equation between soil P and the concentration of dissolved phosphorus (DRP) in runoff: *water soluble P in runoff (mg/l) = 0.021*soil_P (mg/l soil) – 0.015 (mg/l)*. The surface runoff of potentially bioavailable particulate phosphorus is approximated from the rate of soil loss and the concentration of potentially bioavailable phosphorus in eroded soil material as follows: *potentially bioavailable particulate phosphorus PP (mg/kg eroded soil) = 250 * ln [soil_P (mg/l soil)]-150* (Uusitalo, 2004[24]). Thus, the parametric description of surface phosphorus runoff is given by

$$Z_{PP}^{i} = \alpha^{t}\left[\varsigma^{t}\left\{\frac{250\ln(\theta + 0.01P_{i}) - 150}{1000000}\right\}\right]$$

For particulate phosphorus PP runoff function, ζ^{i} is erosion rate (kg/ha), θ the amount of soil phosphorus (mg/l). *Soil_P* is fixed at 10.6 mg/l, which is the average for Finnish FADN farms situated in southern and south-western Finland (Myyrä et al., 2005[25]).

$$Z_{DRP}^{i} = \beta^{t}\left[\frac{\psi(0.021(\theta + 0.01P_{i})) - 0.015}{100}\right]$$

For DRP runoff function ψ is the amount of surface runoff (mm/ha). P_i is in both equations the phosphorus application rate (kg/ha). Runoff and erosion differ between different tillage methods (no-till versus conventional tillage) or land cover types (grasslands versus cereals) and technology specific factors, α^{i} and $\beta^{i,}$ describe the distinctive characters of the different tillage methods and land cover types.

For nitrogen runoff following runoff function, estimated by Simmelsgaard (1991[26]), is employed

$$Z_{l}^{i} = \varpi * Exp\left[b_{0} + b_{1} * \frac{l_{i}}{100}\right]$$

where Z_{l}^{i} = nitrogen runoff at fertiliser intensity level l_i, kg/ha, ϖ = nitrogen runoff at average nitrogen application, $b_0 < 0$ and $b_1 > 0$ are constants and l_i = nitrogen fertilization in relation to the normal fertiliser intensity for the crop, $0.5 \leq N \leq 1.5$. This runoff function represents nitrogen runoff generated by a nitrogen application rate of l_i per hectare, and the parameter ϖ reflects differences in tillage methods and land cover types.

Annex 3.B. List of parameter values

Parameter	Symbol	Value	Reference
Market price, EUR/kg			
Milk	pM	0.4455	(OSF, 2014[27])
Concentrate, domestic	pv	0.183	(Tuottopehtori, 2017[28]))
Concentrate, soybean meal	psoy	0.3507	IndexMundi (2014)
Mineral fertiliser, YaraMila Y2	pl	0.45	Tuottopehtori (2014)
Meat	pmeat	2.1	Tuottopehtori (2014)
Calf (selling), EUR/animal	pcalf	115	Tuottopehtori (2014)
Mineral fertiliser, YaraMila Y2			
N-content	eN	0.24	Tuottopehtori (2014)
P-content	eP	0.04	Tuottopehtori (2014)
Variable cost in barley production,EUR/kg	hC	0.056	Tuottopehtori (2011)
Variable costs in silage production			
EUR/kg yield	h0	0.0918	Tuottopehtori (2014)
Silage dry matter %	dmpc	25	
Silage density, kg/m3	dens	250	
Loading capacity, m3	trcap	20	
Transport speed, km/h	trsp	15	
Transport price, EUR/h	trp	63.1	(Palva, 2015[29])
Cost of floating storage cover, EUR/m2/year	float	2	
Capacity of manure spreader, m3	spcap	16	Palva (2015)
Contractor charge for spreading,E UR/h			
Broadcast spreading	spp1	77.9	Palva (2015)
Injection	spp2	102.5	Palva (2015)
Time for loading, h/m3	load	0.004	
Time for spreading, min/m3			
Broadcast spreading	spread1	0.5	
Injection	spread2	1.5	
Transport cost interrelated to spreading, EUR/m3/km	ctran	0.4	Palva (2015)
Damage from GHG emissions, EUR/kg CO2eq.	?G	0.05	
Animal body weight, kg			
Dairy cow	BWD	600	
Heifer	BWH	400	VTT (2000)
Calf	BWC	150	VTT (2000)
Number of milking seasons	slaught	3	
Number of dairy cows	H	60	Chosen
Share of animals per dairy cow			
Dairy cow	shareD	1	
Heifer	shareH	1/3	
Calf	shareC	1/3	

Parameter	Symbol	Value	Reference
Total intake, kg DM/animal/day			
Heifer	inH	5	
Calf	inC	10	
Share of concentrates in H and C diet, %		50	
Manure excretion, m3/animal/year			
Heifer	wH	8.5	(Finlex, 2014[30])
Calf	wC	6.25	Finlex 1250/2014
Water in liquid manure, m3/animal/year	h2o	10	
Manure density, kg/m3	kgm3	1000	
Share of manure excreted on pasture	wp	0.15	
Scaling factor to match Finnish statistics	scale	0.65	Chosen
Parameter for manure excretion	w0	9.4	(Nennich et al., 2005[19])
Parameter for manure excretion	w1	2.63	Nennich et al. (2005)
Parameter for manure N content	N1	84.1	Nennich et al. (2005)
Parameter for manure N content	N2	0.196	Nennich et al. (2005)
Feed nutrition values			
Concentrate (barley 54-62 kg/hl)	v		
Dry matter, g/kg	cka	860	
Organic matter, g/kg DM	coa	971	
Organic matter digestibility	coas	0.82	
Crude protein content of DM	vcp	0.126	
P content of DM	vp	0.0041	
Concentrate (soybean meal)	v		
Dry matter, g/kg	cka	880	
Organic matter, g/kg DM	coa	821	
Organic matter digestibility	coas	0.88	
Crude protein content of DM	vcp	0.520	
P content of DM	vp	0.007	
Silage feed (grass silage)	s		
Dry matter, g/kg	ska	1000	
Organic matter, g/kg DM	soa	911	
Organic matter digestibility	soas	0.74	
Crude protein content of DM	scp	0.161	
P content of DM	sp	0.0031	

Parameters for nitrogen response functions	Symbol	Value	Reference
Quadratic response function for silage	a	1182.9	Bäckman et al. (1997) and Lehtonen (2001)
	b	24.24	
	c	-0.0394	
Quadratic response function for rape seed	a	890.0	
	b	9.95	
	c	-0.0354	
Mitscherlich response function for wheat	φ	4956	
	σ	0.7624	
	ρ	0.011	
Mitscherlich response function for barley	φ	5218	
	σ	0.8280	
	ρ	0.017	

Parameters for GHG emissions	Symbol	Value	Reference
Conversion factors			
N_2O to CO_2eq.	*N2O*	298	
CH_4 to CO_2eq.	*CH4*	21	
Enteric fermentation			
Coefficients			
Dairy cow	*CfiD*	0.379	(Statistic Finland, 2016[20])
Heifer	*CfiH*	0.322	Statistics Finland (2016)
Calf	*CfiC*	0.322	Statistics Finland (2016)
Pasture	*cap*	0.17	Statistics Finland (2016)
Stall	*cao*	0.00	Statistics Finland (2016)
Pregnancy (dairy cows)	*CpD*	0.10	Statistics Finland (2016)
Growth			
Dairy cow	*CoD*	0.00	Statistics Finland (2016)
Heifer	*CoH*	0.80	Statistics Finland (2016)
Calf	*CoC*	1.00	Statistics Finland (2016)
Average weight gain, kg/day			
Dairy cow	*WGD*	0.05	Statistics Finland (2016)
Heifer	*WGH*	0.45	Statistics Finland (2016)
Calf	*WGC*	0.90	Statistics Finland (2016)
Pasture season, days			
Dairy cow	*tpD*	125	(OSF, 2010[31])
Heifer	*tpH*	135	OSF (2010)
Calf	*tpC*	115	OSF (2010)
Milk fat content, %	*fat*	4.3	Statistics Finland (2016)
CH_4 conversion rate	*Ym*	0.065	Statistics Finland (2016)
Manure storage			
No cover			
Emission factor N_2O	*ef1*	0	Statistics Finland (2016)
Emission factor CH_4	*mcf1*	0.17	Statistics Finland (2016)
Manure N evaporated as NH_3, %	*emstor1*	10	Grönroos (2014)
Floating cover			
Emission factor N_2O	*ef2*	0.005	Statistics Finland (2016)
Emission factor CH_4	*mcf2*	0.10	Statistics Finland (2016)
Manure N evaporated as NH_3, %	*emstor2*	6	(Grönroos, 2015[21])
Manure ash content	*ash*	0.08	(IPCC, 2006[32])
Max. CH_4 producing capacity, m^3/kg vs	*chmax*	0.24	Statistics Finland (2016)
Manure spreading			
Broadcast spreading			
Manure N evaporated as NH_3, %	*emspread1*	40	Grönroos (2015)
Injection			
Manure N evaporated as NH_3, %	*emspread2*	9	Grönroos (2015)
Autonomous soil emissions, kg CO_2eq./ha			
Barley	*autoc*	1535	
Silage	*autos*	426	
Pasture	*autop*	1535	
Cultivation practices, kg CO_2eq./ha			
Barley	*cultc*	362	
Silage	*cults*	136.5	
Pasture	*cultp*	362	
N applied to pasture land, kg N/ha	*lp*	220	

Parameter	Symbol	Value	Reference
Other parameters			
Yield transport to processing, kg CO_2eq./ha	*emtrans*	0.00696	
Crop drying, kg CO_2eq./kg yield	*emdry*	0.028	
Mineral fertiliser manufacture, kg CO_2eq./kg N	*emprod*	4.32	
Soybean meal manufacture, kg CO_2eq./kg	*emsoy*	5.35	(Opio et al., 2013[33])
Parameters for nutrient functions			
Constant	$b0$	-0.7	(Simmelsgaard, 1991[26]; Uusitalo and Jansson, 2002[23]; Uusitalo, 2004[24])
Constant	b	0.7	
Average runoff from fertilisation	ϖ	15	
Erosion	ζ	800	
Surface runoff	Ψ	234	
Soil phosphorus	θ	10.6	
Phosphorus rate	P	0.143	
Technology factor, PP	α	2.4	
Technology factor, DRP	β	0.77	

4 Global potential of supply-side and demand-side mitigation options

This chapter analyses how agriculture could moderate changes to the climate by simulating supply- and demand-side mitigation strategies. Based on the Aglink-Cosimo model as used for the *OECD-FAO Agricultural Outlook 2018-2027* baseline, only direct emissions that result from agricultural crop and livestock production activities are taken into consideration.

Introduction

FAOSTAT reports that 11% of global GHG emissions in 2010 came directly from agricultural production. The subcategories of agricultural emissions distinguished in FAOSTAT are:

- Enteric fermentation
- Manure management
- Rice cultivation
- Synthetic fertilizers
- Manure applied to soils
- Manure left on pasture
- Crop residues
- Cultivation of organic soils
- Burning – crop residues
- Burning – savannah

This percentage does not include emissions that result from converting land, e.g. forest areas to grass or cropland that are categorised in the Land Use, Land-Use Change and Forestry (LULUCF) sector, which accounts for an additional 11% of global GHG-Emissions.

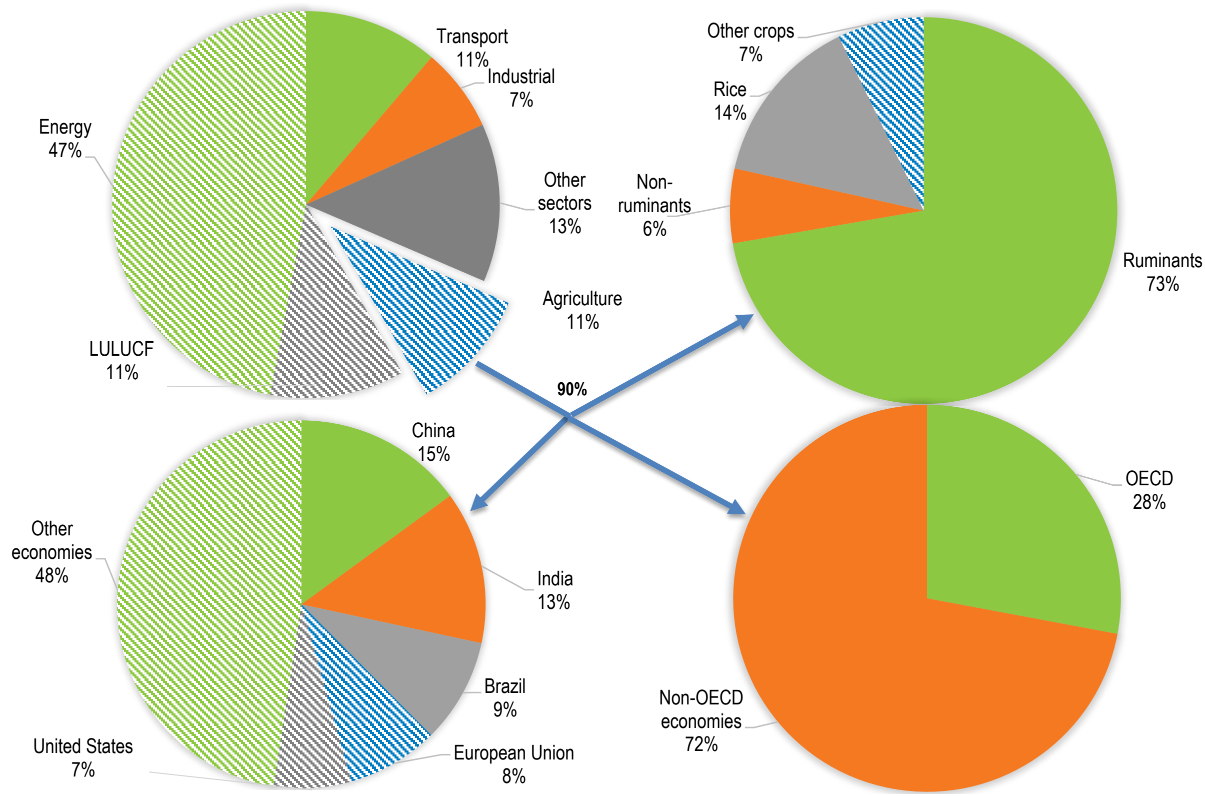
At present, the Aglink-Cosimo model can only capture the first category of emissions.[1] About 90% of those direct emissions are captured, as not all production activities are represented and emissions from burning crop residues and savannah, as well as organic soils and crop residues are not accounted for.

Of the emissions studied in this chapter, 73% can be attributed to the ruminant sector, 6% to non-ruminant meat production, 14% to rice cultivation, and 7% to other crops. Geographically, the highest absolute emissions are located in the People's Republic of China (hereafter "China") (15%), India (13%), Brazil (9%), the European Union (8%), and the United States (7%). These four countries and the European Union account for over 50% of agricultural GHG-emissions. OECD countries are responsible for about 30% of global emissions.

Figure 4.1 reveals the importance of the ruminant sector in global GHG emissions, implying that there is also significant potential to reduce emissions in this sector. Nevertheless, the 2018 edition of the *OECD-FAO Agricultural Outlook* (OECD/FAO, 2018[2]) projects that demand for products produced by ruminants will continue to increase up to 2027 for most countries. Global meat production is projected to be 15% higher in 2027 relative to the base period, and the projected output growth is expected to occur predominantly in developing countries. The most rapid expansion is expected to occur in the poultry sector. Consumers in developing countries are expected to increase and diversify their consumption towards more expensive meats, including beef and sheep meat, which have higher emissions per unit of output.

World milk production is projected to increase by 22% over the projection period, with over half the increase originating in Pakistan and India, two countries where emissions per litre of milk are relatively high. Both countries are expected to jointly account for 32% of global milk production by 2027.

Figure 4.1. Composition of GHG emissions, 2010



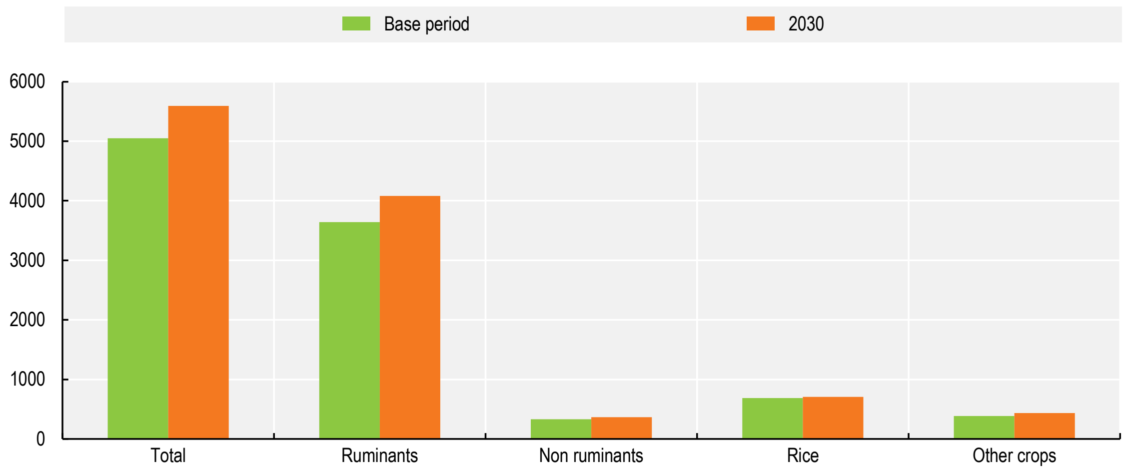
Note: Energy" includes energy, manufacturing and construction industries and fugitive emissions, "Other" includes residential, commercial and institutional emissions, and emissions from industrial processes and product use, waste international bunkers, and other non-specified sources.
Source: FAOSTAT and Aglink-Cosimo database.

These projections lead to an increase in total GHG Emissions throughout the baseline projection period as shown in Figure 4.2. Total emissions from agriculture are projected to increase by 540 $MtCO_2$ equivalents (CO_2eq) between the outlook base period (average 2015-2017) and 2030, with about 80% of this increase stemming from the ruminant sector (440 $MtCO_2eq$), and within that sector methane (CH4) emissions from enteric fermentation will account for 300 $MtCO_2eq$ (70%).

The distribution of total emissions from agriculture varies widely across countries (Figure 4.3). China, India and the rest of South-East Asia account for over 40% of global agricultural GHG emissions at present and are projected to account for 70% of the increase in global GHG emissions. Large increases in emissions are also expected from Sub-Saharan Africa and Brazil.

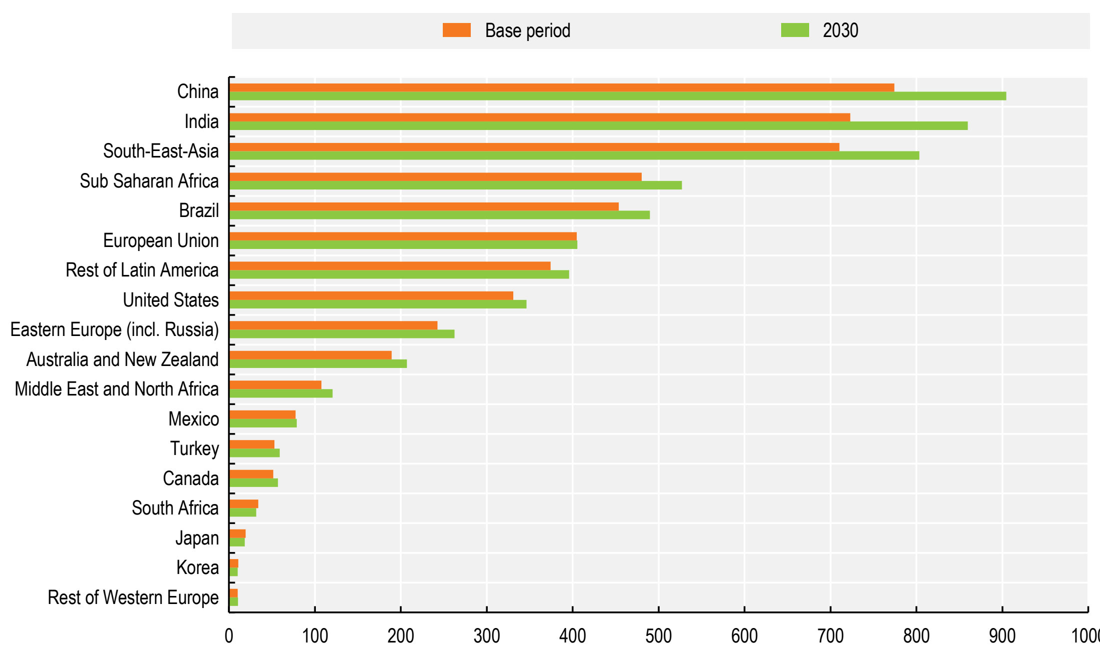
This baseline represents a *business as usual* scenario where no additional actions towards emission savings are undertaken, although existing mitigation efforts that are visible in past trends of emissions per production unit are taken into account. This report does not deal with the impact of climate change on the agricultural sector but the overall impact over the analysed time horizon is limited. Hasegawa et al. (2018[3]) showed that even by 2050, the average impact of climate change is expected to be low compared to the potential mitigation impact of agriculture. It should be noted, however, that the impact of potential increases in extreme events has not taken into account.

Figure 4.2. Total GHG emissions from agriculture ($MtCO_2eq$)



Total emissions do not include emissions from products not covered in Aglink-Cosimo
Source: Calculations based on OECD/FAO (2018), "OECD-FAO Agricultural Outlook", OECD Agriculture statistics (database), http://dx.doi.org/10.1787/agr-outl-data-en.

Figure 4.3. Regional differences in total GHG emissions from agriculture ($MtCO_2eq$)



Total emissions do not include emissions from products not covered in Aglink-Cosimo
Source: Calculations based on OECD/FAO (2018), "OECD-FAO Agricultural Outlook", OECD Agriculture statistics (database), http://dx.doi.org/10.1787/agr-outl-data-en.

Box 4.1. Aglink-Cosimo's contribution to climate change analysis

The main strength of the Aglink-Cosimo model is its level of detail on agricultural commodity markets and ability to capture interactions with market policies. It can outline the implications of changes in exogenous drivers and policy assumptions for market outcomes. This is exploited here from a new perspective. The Aglink-Cosimo baseline projections are carefully linked to historical developments, so that future projections start from where the world is now (a feature absent in many models). This helps provide a clearer benchmark to assess future contributions to climate change mitigation.

In order to obtain estimates of the emissions that are directly produced by the agricultural sector from each model run, emission factors per commodity produced were inherited from IIASA's Globiom model which are based on the IPCC guidelines at the tier 1 level and broadly consistent with the FAOSTAT database. The resulting coefficients are not static, but include the most recent trends of emission intensities at the country/region level.

Technological mitigation options are incorporated into the analysis (Scenario 5) using regional marginal abatement cost curves (MACC) for the different direct agricultural emissions. These MACCs were obtained from the technological adjustment behaviour of the Globiom model.

The combination of those two recent developments allow reporting on direct agricultural emissions, as well as analysis of supply-side mitigation scenarios. This analysis will, however, be partial as it does not include the effects on emissions created during the processing of food products, nor those arising from producing inputs for agriculture. Furthermore, changes to input quantities are only implicitly accounted for.

The Joint Research Centre of the European Commission recently published a report on the economic impacts of a low carbon economy on global agriculture using a similar approach to the Aglink-Cosimo model (Jensen et al., 2019[4]). These two methodologies will be merged in future studies.

The baseline scenario is then compared to alternative scenarios, which address three possible ways to mitigate emissions: reduce the share of food consumed from ruminants; reduce food waste; and impose carbon taxes and improve productivity on the production side.

This analysis, however, also addresses potential trade-offs between food security and emission reduction. Insofar as measures to reduce GHG emissions from agricultural production lead to lower food output or increases in food prices, there may be trade-offs between the dual goals of guaranteeing food security and reducing GHG-emissions.

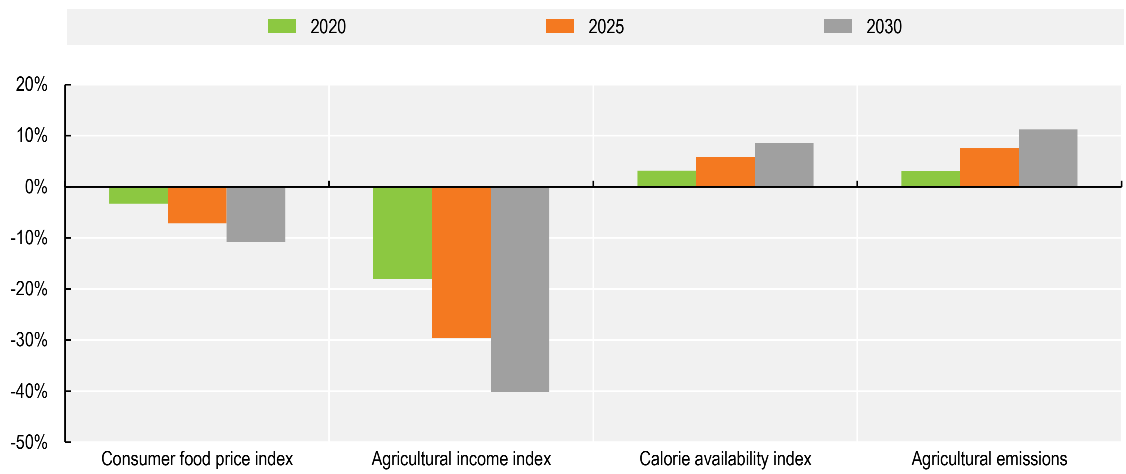
The FAO definition of food security – encompassing the four dimensions of availability, accessibility, utilization, and stability – is used here.[2] Aglink-Cosimo partially addresses the availability and access dimensions via projections for national availability and food prices. Three indicators can be calculated based on the scenario outputs.

- *Calorie Availability Index (Availability):* The average amount of calories available per capita in each country for the subset of the food basket represented in the model.[3]
- *Consumer Food Price Index (Accessibility):* Calculated as a fixed weight index of national consumer prices in real terms. Food consumption quantities in 2015 are used as weights. Higher consumer prices are assumed to lead to lower access to food for parts of the population.
- *Agricultural Gross Income Index (Accessibility):* Calculated as a fixed weight index of a combination of producer prices and subsidies normalised by an input price index. Agricultural production quantities in 2015 are used as weights. This indicator is most relevant in countries where the agricultural sector is a large contributor to the national GDP.[4]

Figure 4.4 shows how under the *baseline scenario* the three food security indicators and emissions from the agricultural sector are projected to evolve relative to their 2015 values. Two of the three indicators show a positive development over the projection period. Prices for consumers, measured by the consumer price index, are projected to decrease in real terms indicating improvements in accessibility. Calorie availability is also projected to improve over the next decades. However, the agricultural income index (in real terms) is expected to decline strongly due to the projected increase in input prices over the outlook period, while real output prices decrease in general. This corresponds to the historical tendency for real agricultural prices to decline over time, exerting income pressure on farmers who are not participating in the productivity gains that drive lower prices. Emissions from agriculture are projected to be 11% higher in 2030, as compared to 2015.

Figure 4.4. Baseline development of global food security indicators and emissions

Percentage change relative to 2015



Source: Calculations based on OECD/FAO (2018[2])

Scenarios to reduce GHG emission

Reducing the consumption share of food produced from ruminants

Ruminant products covered in this analysis contain beef, sheep meat, as well as butter, cheese, fresh dairy milk products, skimmed milk and whole milk powders. Ruminant products account for 70% of the emissions analysed in this report. Strategies to reduce emissions from these products are more promising than for many other food commodities. This section examines the food security and emission impacts of two scenarios that reduce the level of final food consumption of these products.

Scenario definition

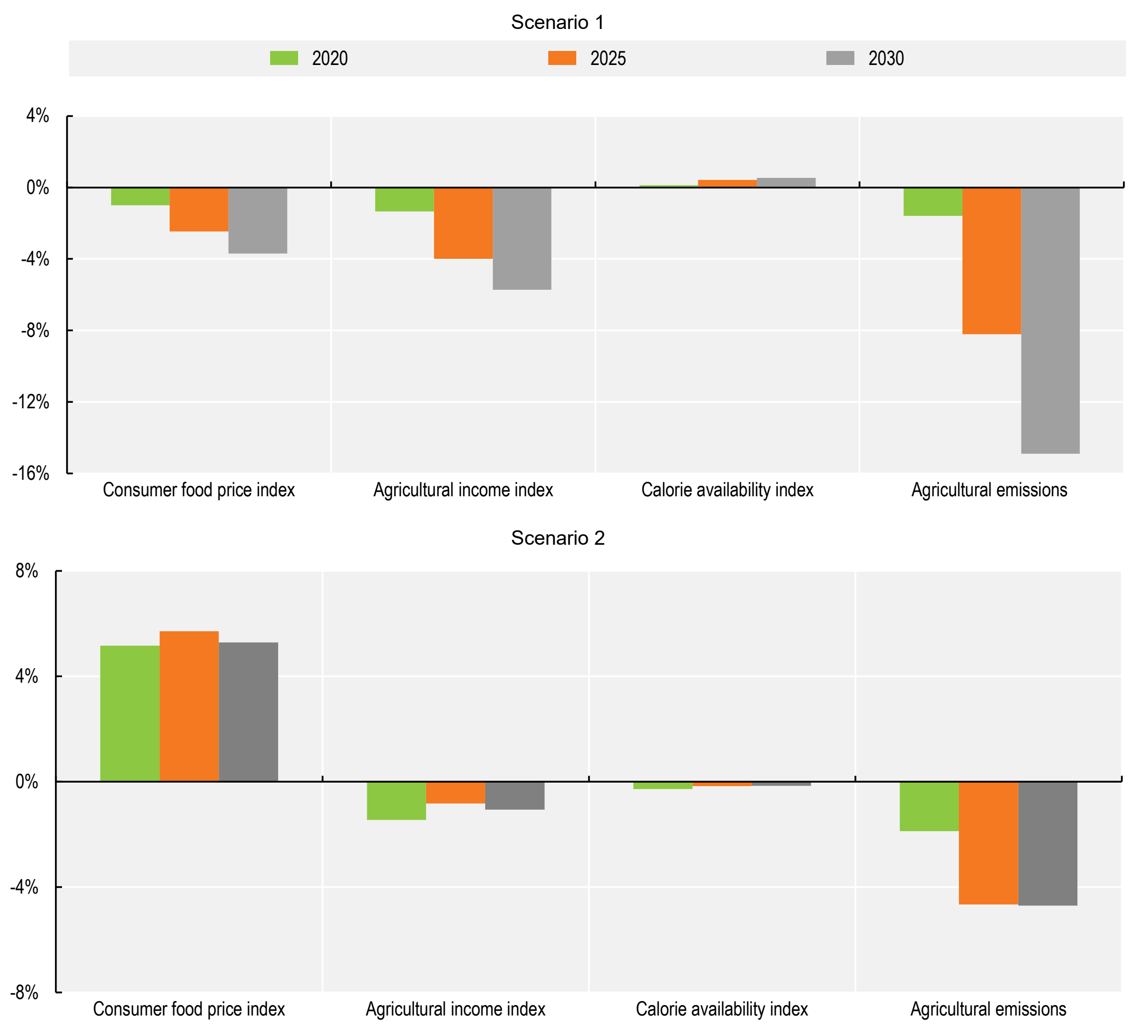
Scenario 1 assumes that the average per capita consumption of each of the ruminant products is gradually reduced to reach levels in 2030 that are 10% below the values of 2017. This scenario also assumes that consumption of non-ruminant products increases by the same proportion across all commodities, such that over that same period the same average per capita consumption of total calories is maintained as under the baseline scenario.

In *Scenario 2* a consumer demand tax of USD 60[5] (in year 2000 real USD) per ton of CO_2eq emitted by each product is applied globally, taking country specific emission factors for primary agricultural products into account.[6] Given their higher emissions intensity, this measure will result in much higher consumer price increases on ruminant products than for the other commodities. However, Scenario 2 does not address emissions that occur between the farm gate and the final consumer,[7] and contrary to Scenario 1, its implementation does not target only the ruminant sector. In addition, the analysis does not include the potential effects arising from redistribution of the collected tax money.[8] Both scenarios are applied to all countries with the exception of the Least Developed Countries (LDCs).

Results

Figure 4.5 compares the food security indicators with agricultural emissions saving between Scenarios 1 and 2. The GHG emission savings by 2030 are projected to be much higher under Scenario 1 (870 $MtCO_2eq$) than under Scenario 2 (160 $MtCO_2eq$). This can be explained by a lower reduction in calorie consumption of ruminant products. In Scenario 1, the average global per capita consumption of ruminant products in 2030 is reduced from 265 to 197 kcal/cap/day, a reduction of roughly 70 kcal/cap/day. In Scenario 2, this is reduced to 12 kcal/cap/day.

Figure 4.5. Emission savings versus food security in relation to the baseline scenario

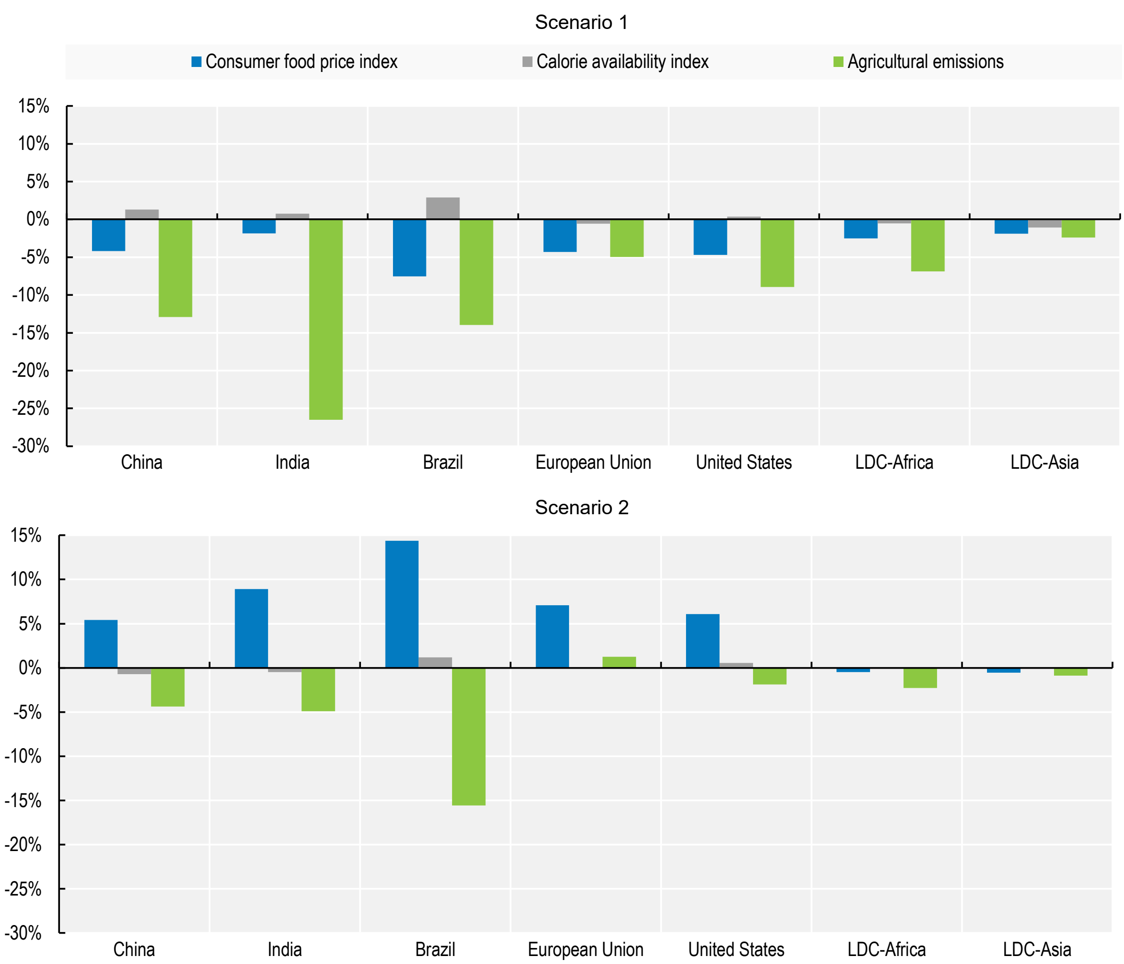


Source: Aglink-Cosimo simulation results.

Scenario 1 also performs better in terms of the consumer price index. This index is projected to increase strongly in Scenario 2 because taxing food globally makes food more expensive, putting accessibility to food at risk. Under Scenario 1, the index is expected to decrease because lower demand leads to lower consumer prices, not only for ruminant products but also for food commodities used to feed ruminants. Agricultural income is more affected in Scenario 1 because reduced demand for ruminant products is more pronounced here than under Scenario 2. This decrease not only reduces the producer price of ruminant products, but feed demand as well. Global cereal demand under both scenarios is, for example, lower than under the baseline, implying that in Scenario 1 the effect of reducing feed demand through lower ruminant product consumption dominates the increase in food consumptions of cereals.

Figure 4.6 illustrates the projected relative changes in the consumer price index, the calorie availability index, and agricultural emissions under both scenarios for selected countries. These countries were chosen based on their importance in global agricultural GHG emissions. The two LDC aggregates are included even though these countries were exempt from any mitigation obligations under the two scenarios; nevertheless, they will be affected via global market impacts. Note that calorie availability is not subject to significant changes in either scenario.[9]

Figure 4.6. Regional emission savings versus food security in relation to the baseline scenario, 2030



Source: Aglink-Cosimo simulation results.

A consumption tax (Scenario 2) is projected to reduce emissions to a lesser extent than Scenario 1 in all selected countries, except for Brazil. The main reasons for the relatively stronger emission-reducing effect of Scenario 2 in Brazil are: it has relatively high emission coefficients for beef production, which translate into higher taxes; beef prices are relatively low so that the new tax accounts for more than 50% of the consumer price (the share would be only 10% in the United States); and, demand in emerging economies such as Brazil is more elastic to price changes than in developed countries.

Emission savings in Scenario 2 are associated with a large cost burden for consumers in terms of consumer prices. Even though small, the spillover effects to least developed countries are positive in terms of cheaper food and reduced emissions. The latter effect is due to lower international producer prices for the ruminant products, which lead to a decrease in production of those products in LDC countries and an increase in imports.[10] It is clear that a shift in preferences as simulated in Scenario 1 would lead to lower concerns regarding food security and to higher emission savings than would the introduction of a consumption tax (Scenario 2).

The impact of food waste on GHG emissions

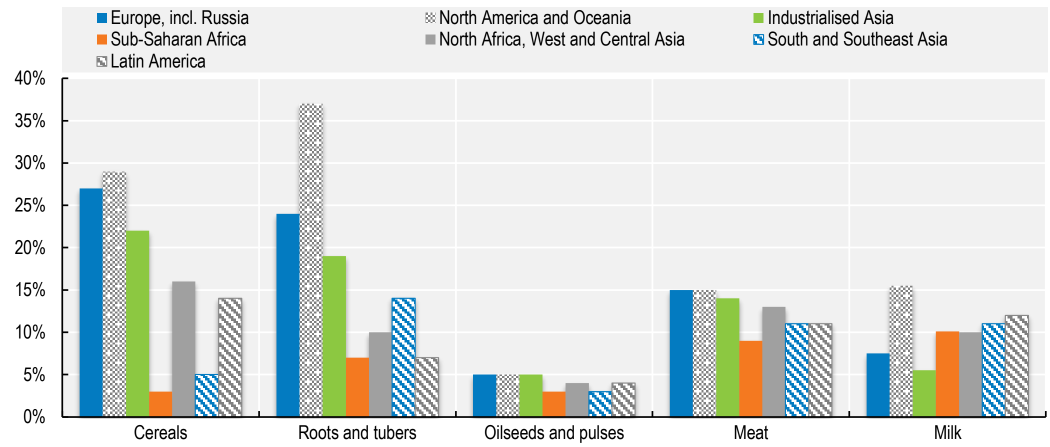
Throughout the supply chain, food is lost or wasted. In general, a distinction is made between the two. Food loss occurring on the supply side is defined by FAO (2011[5]) as food that gets spilled or spoilt before it reaches its final product or retail stage. Food waste occurs at the retail and household levels.

This chapter focuses on the food waste issue, which is considered a part of food demand. Reducing food waste has many benefits but also carries costs. For example, a restaurant owner knows how much food is thrown away every day and knows there are ways to reduce the amount of food wasted, but the costs of applying those measures might be higher than the economic benefits. Similarly, a retail store could reduce the amount of fruits and vegetables thrown away at the end of a day by investing in better cooling systems, but might consider it more profitable not to do so. Finally, a family could reduce food that is thrown away by buying less food during each grocery trip at the cost of shopping more frequently.

Only a few studies examine the impact of reducing food waste and loss at the global level. In Okawa (2015[6]), the medium-term market impacts of reducing food waste and food loss are examined based on the OECD-*FAO Agricultural Outlook 2014-2023* projections for world and national agricultural markets. The study applies FAO's region-specific estimates of producer loss and consumer waste, which are reduced by 20% over ten years on the assumption that these reductions can be achieved without cost. The study finds a greater impact on international markets due to contractions in demand via reduced waste than from the stimulus to supply from lower losses. Savings to consumers total more than USD 2.5 trillion over ten years and reduced crop losses in developing countries lead to higher crop supplies in these countries, with reduced prices from efficiency gains benefiting both developing and developed countries. However, the analysis in this study does not consider the potential environmental impact of reducing food loss and waste.

In order to asses this maximal GHG emissions abatement potential of reducing food waste, the analysis in Okawa (2015[6]) was repeated, focussing on the food waste aspect on the demand side and using the 2018 version of the Aglink-Cosimo model. Okawa (2015[6]) uses the food loss and food waste estimates published in FAO (2011[5]), which are the only estimates currently available on a global scale. The estimates in the FAO study show the shares that are lost or wasted for seven agricultural product groups (cereals, roots and tubers, oilseeds and pulses, fruits and vegetables, meat, fish and seafood, and milk) at seven regional aggregates (Europe including the Russian Federation, North America, and Oceania, Industrialised Asia, West and Central Asia, South and Southeast Asia, Sub-Saharan Africa, North Africa, and Latin America). Those shares are defined for five levels of the supply chain (agricultural production, post-harvest handling and storage, processing and packaging, distribution, and consumption at household level). Figure 4.7 shows the shares of food waste in the distribution and consumption at household levels from that study.

Figure 4.7. Food waste shares



Note: Only waste shares = Sum of distribution and household in FAO (2011[5]). Fish, seafood, fruits and vegetables are not included since they cannot be addressed in this chapter.
Source: FAO (2011[5]).

As indicated above, this section considers the food waste issue as defined by the FAO. As such, only two supply chain levels matter: distribution and consumption at the household level. The food use variable in Aglink-Cosimo implicitly includes waste at these two levels and can therefore be interpreted as food availability. It is assumed that food availability and final food demand are the same, as waste no longer occurs. But since actual waste levels are unknowns in the Aglink-Cosimo database, two assumptions are necessary:

- the waste rates from FAO (2011[5]) are applied to all countries within the seven regional aggregates
- the waste rates from FAO (2011[5]) are applied to all commodities within the five product groups

The LDCs are exempted from all mitigation efforts.

Scenario definition

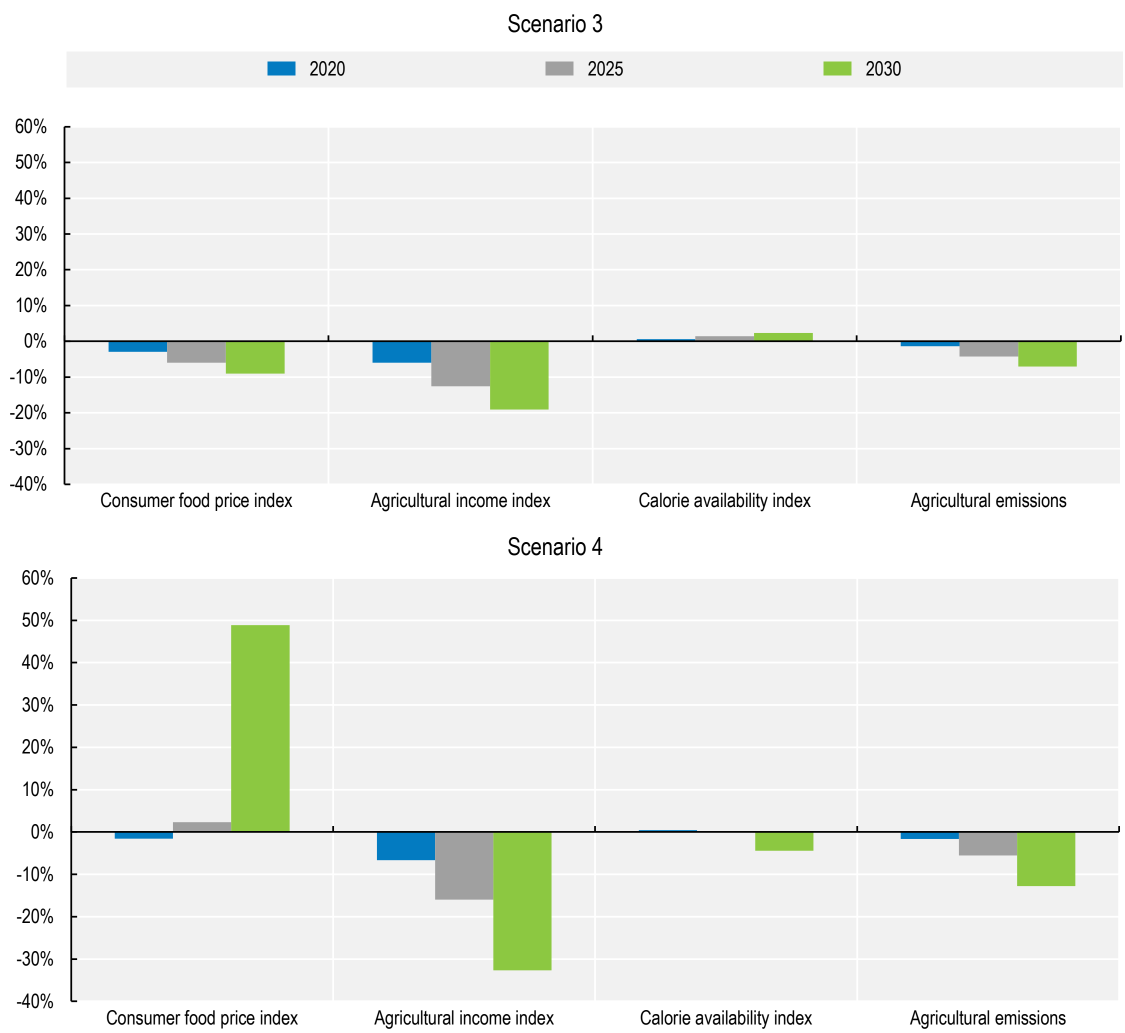
Scenario 3 assumes that the wasted quantities as defined above disappear linearly between 2018 and 2030 by shifting the product-specific food demand equations to the left (as in the left graph of Figure 4.9). This scenario makes no assumptions about the costs that may be associated with reducing waste. Even though the assumption of zero costs for reducing food loss and waste is unrealistic, such a scenario can be seen as the theoretical upper bound of the potential impacts.

Scenario 4 has a set-up similar to Scenario 3, but incorporates the cost aspects of reducing waste. Under this scenario, it is assumed that the costs of waste reduction increase exponentially, i.e. it costs relatively less to reduce the first rather than the final units of waste, and the cost of eliminating the final unit of waste is set equal to the amount the consumer pays for one unit of the respective commodity in the baseline. It is further assumed that the costs to reduce waste will be manifested in higher prices for consumers. This assumption can be justified as follows: a restaurant will transfer its waste reduction costs to its clients, or improved (and more expensive) packaging at the retail level that improves the storage lifetime of products would have to be paid by consumers. Reducing waste at the household level will not lead to higher observed consumer prices. However, it can be assumed that the perceived consumer prices are indeed higher as the household has to pay an implicit mark-up on each consumption unit to reduce waste. For example, if a household reduces waste by increasing shopping frequency to avoid the deterioration of fresh food, this imposes costs (e.g. fuel costs to drive to the supermarket, opportunity costs of the additional time spent in the supermarket), and for simplification these costs are added to consumer food prices. This assumption illustrates that incorporating the costs of reducing waste significantly changes the cost-benefit calculation of reducing emissions by reducing waste.[11]

Results

Under Scenario 3, eliminating food waste would reduce the agricultural part of GHG-emissions by 8% or 440 $MtCO_2eq$ by 2030. The reduction would be even higher in Scenario 4: 14% or 800 $MtCO_2eq$. At a first glance, it appears surprising that reflecting costs to reduce waste increases the emission reduction potential. However, as those costs increase consumer prices, the demand-reducing effect of higher expenditures for food reduces production beyond the levels of Scenario 3.

Figure 4.8. Emission savings versus food security in relation to the baseline scenario



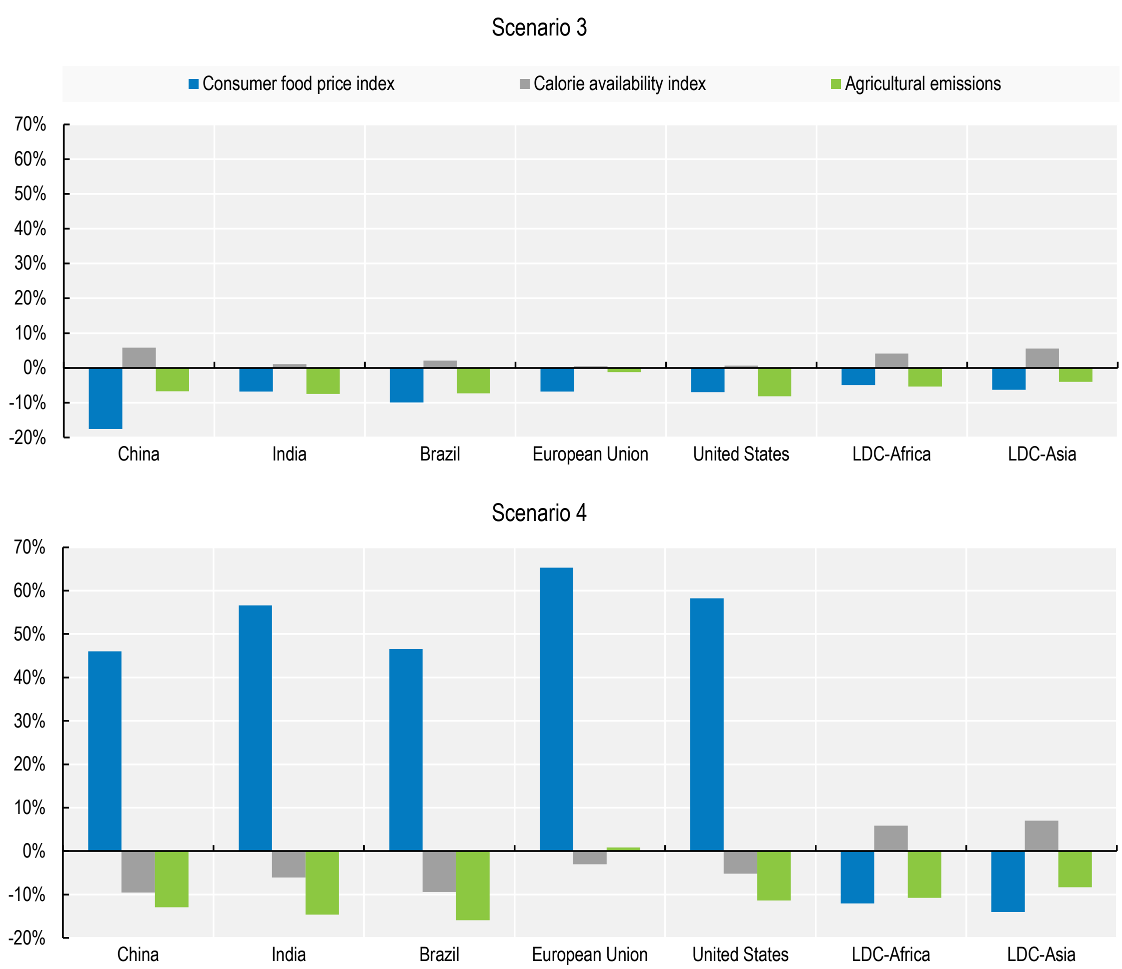
Source: Aglink-Cosimo simulation results.

Figure 4.8 shows how the three food security indexes and the emission savings are projected to evolve under Scenarios 3 and 4 relative to the baseline scenario. Under Scenario 3, consumer prices in 2030 are projected to be 10% lower than under the baseline indicating a positive impact on food accessibility. However, a strong negative effect on agricultural incomes can be observed as producer prices decrease strongly because of lower demand. The calorie availability index indicates that slightly more calories are available for final consumption as compared to the baseline. However, Scenario 3 assumes that the consumer does not have to pay for waste reduction. In Scenario 4, where this assumption is abandoned, the results are very different: the consumer price index increases strongly over time,[12] while the agricultural income index and calorie availability decrease.

Clearly the different outcomes under Scenarios 3 and 4 are a direct result of the assumptions regarding the cost of food waste reduction. Even though Scenario 3 is less realistic, it does illustrate that reducing waste without keeping the cost aspect in mind underestimates the emission-saving potential as well as the negative impact on food security. It further underlines that a better understanding of the costs of waste reduction is needed in order to assess the trade-offs between emission-reduction targets and food security issues.

The significant difference between the two scenarios is also apparent in comparisons across major countries. Calorie availability is projected to be much lower under Scenario 4 and accessibility is at risk with much higher consumer prices. (Figure 4.9). The spill-over effect to LDCs is positive in terms of reduced consumer prices and reduced emissions. However, it should be noted that Scenario 4 would perform better in terms of the food security indicators if it were assumed that waste is not reduced to the final unit. The first units of waste reduction are assumed relatively "cheap" and therefore will not affect demand and prices as much as shown in this chapter.

Figure 4.9. Regional emission savings versus food security in relation to the baseline scenario, 2030



Source: Aglink-Cosimo simulation results.

Production-side mitigation

Blanford et al. (2018[1]) identify supply-side mitigation options in agriculture, excluding output reduction, in the form of changes in farming practices and land management that target emissions per unit of input (land or animal), as well as changes in technical and managerial efficiency and technology that lower emissions per unit of output (productivity improvements). The potential for improvement with respect to the practices and management aspects in particular can be found in the heterogeneity of livestock systems across farms (Box 4.2). The major assumption here is that if carbon emissions were priced, this would act as an incentive for farmers to transfer their systems to use less GHG-emitting practices.

Based on the work described in Box 4.2, the Aglink-Cosimo model was adjusted in several. The revised model not only captures the consumption change component of mitigation, but also incorporates technological options and structural changes by introducing dynamic emission coefficients depending on the applied carbon tax level.

Scenario definition

Scenario 5: A production tax of USD 60 (real USD, 2000) per tonne of CO_2eq emitted is imposed on the agricultural production activities by shifting the supply curves for each commodity upwards by the amount of the tax. The individual tax rates per tonne of product thereby differ because beef production has higher emission coefficients compared to wheat for example. The applied emission coefficients in this scenario differ from those in the baseline due to the assumption that a carbon tax will lead to technological and structural mitigation (Figure 4.11) that cannot be explicitly modelled with Aglink-Cosimo, thus reducing the emissions per production unit. The choice of the carbon tax level is the same as that applied in Scenario 2.

Scenario 6: A productivity shift of 10% by 2030 is implemented for all products, linearly increasing from 2018. This means that yields for crops are increasing as well as the output of meat and dairy products per animal. This increase is assumed to be achieved at no cost.

Box 4.2. Heterogeneity of the production system as a source of climate change mitigation in agriculture

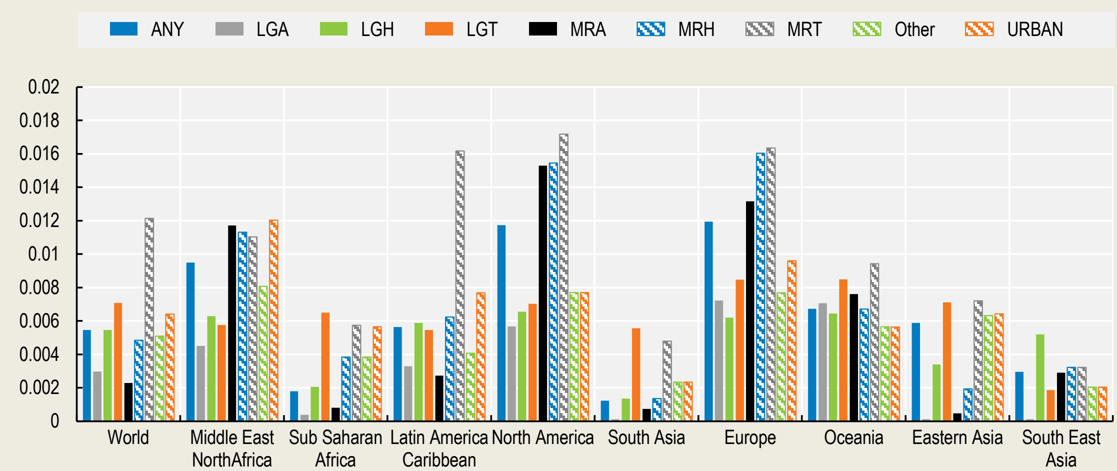
Climate change mitigation in agriculture can be modelled through changes on the consumer or producer side. If the agricultural sector were represented in a simplified way whereby each production activity is associated with the same greenhouse gas (GHG) emission coefficient, then reducing the consumption of GHG intensive products would be the only mitigation option. In order to include mitigation options related to the production side, technological options have been incorporated. Examples of technological options are propionate precursors and antimethanogen vaccinations to reduce CH4 emissions from enteric fermentation, or anaerobic digesters to reduce CH4 emissions from manure management. These options are sometimes referred to as add-on technologies since their mitigation potential is typically calculated in addition to current production activities. Another option is to change production to more efficient systems. This structural change option was overlooked for a long time in large-scale assessments, but is valuable as it incorporates the heterogeneity in production systems in terms of their GHG efficiency.

This box presents an overview of recent studies on the structural change option. Most of these studies focus on the livestock sector as it is responsible for 65% of agricultural non-CO_2 emissions. In 2013, Herrero et al. published a detailed dataset describing livestock production systems worldwide in terms of their productivity, feed rations, and GHG emissions. The dataset quantifies the differences in GHG efficiency across individual products, from the relatively high efficiency of poultry production to the relatively low efficiency of beef production. In addition, the study shows that large differences exist for the same product across alternative production systems within the same region and across regions. For example, in

Europe 17 kg protein of beef per tonne CO_2eq can be produced in the temperate agro-ecological zone with sufficient concentrate feed supplementation, while only 8 kg protein of beef per tonne CO_2eq can be produced in the grazing systems in the same agro-ecological zone. Similarly, in a comparable system with concentrate supplementation in Australia, typically only 10 kg protein of beef per tonne CO_2eq is produced, 7 kg less than in Europe (Figure 4.10).

Figure 4.10. GHG efficiency for bovine meat from non-dairy cattle

Regional and production system differences (t protein per tCO_2eq)



Notes: Non-dairy cattle include here all cattle heads other than dairy cows and replacement heifers. Production systems: ANY – average across all production systems; LGA – grazing systems arid; LGH – grazing systems humid; LGT – grazing systems temperate/tropical highlands; MRA – mixed crop-livestock systems arid; MRH – mixed crop-livestock systems humid; MRT – mixed crop-livestock systems temperate/tropical highlands; URBAN – urban systems; Other – other systems.
Source: Herrero et al (2013[8]).

A key driver of the differences in GHG efficiencies is feed quality. Herrero et al. (2013[8]) found that increasing the metabolisable energy content in feed, for example from 9 to 10 MJ per kg dry matter feed, reduced the emissions related to beef production from about 0.250 to 0.10 tonne CO_2eq per kg protein. This could be achieved through feeding practices that include less grazing and better quality feeds. These transitions are generally induced by changes in relative factor prices: because of the increased population density, land values are growing faster than the economic opportunity cost of labour.

Havlík et al. (2014[9]) implemented two scenarios in the Globiom model using the Herrero et al. (2013[8]) dataset to analyse the future dynamics of livestock production systems and their contribution to reducing GHG emissions. The first scenario is a dynamic one representing business as usual adaptation of the structure of livestock production systems to the future economic conditions. The second scenario is counterfactual whereby the structure of livestock production systems is fixed around the year 2000. In the dynamic scenario, 64% of all ruminants would be reared in mixed systems with feed supplementation in 2030 compared to 56% of ruminants in 2000 (the counterfactual scenario), representing an intensification in feeding strategies.

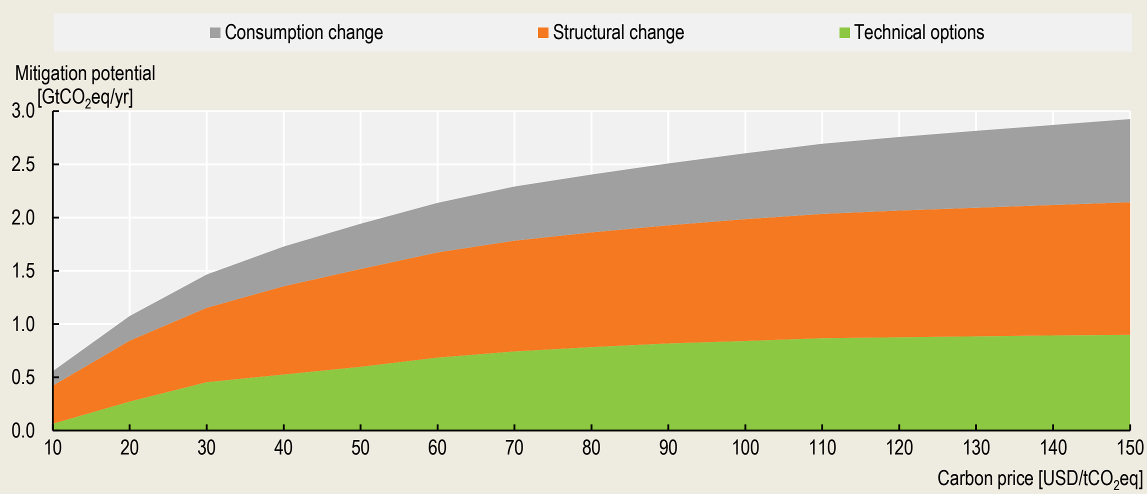
Under the dynamic scenario, total Agriculture, Forestry, and Other Land Uses (AFOLU) emissions over the period 2010-2030 would be 9% lower than in the counterfactual scenario, indicating that individual adjustments in the production system structure would lead to an average annual saving of 736 $MtCO_2$eq. In the dynamic scenario, the majority of GHG reductions came from changes in land use (-23%), while agricultural non-CO_2 emissions were reduced by less than 5%. Increased feed use efficiency in mixed systems with concentrate supplementation saved 176 million ha from pasture expansion, and limited cropland expansion to only 14 million ha in the dynamic scenario as compared to the counterfactual scenario.

The role of structural change as a mitigation option under a carbon price policy was also analysed by Frank et al. (2018[7]) in an integrated framework which considered the technological options mentioned above and the consumption side response to the increased production cost. This study found that at a carbon price of USD 100 per tCO_2eq, the agricultural sector could decrease by 2.6 billion tCO_2eq annually non-CO_2 emissions originating from this sector.

At this level, GHG reduction through structural change, including the transition of livestock production systems, and the technological options would contribute 38% each to the total reduction, while a decrease of consumption in response to increased producer prices would provide the remaining 24% (Figure 4.11). This study also finds indirect benefits of GHG reduction in the agricultural sector from land use change (-0.7 billion tCO_2eq). These examples show that it is important to take into account production system heterogeneity for the baseline emission profile development and the agriculture sector GHG mitigation potential assessment.

Figure 4.11. Economic mitigation potential of non-CO_2 GHG emissions in agriculture

By mitigation option in 2050 at increasing global carbon price levels

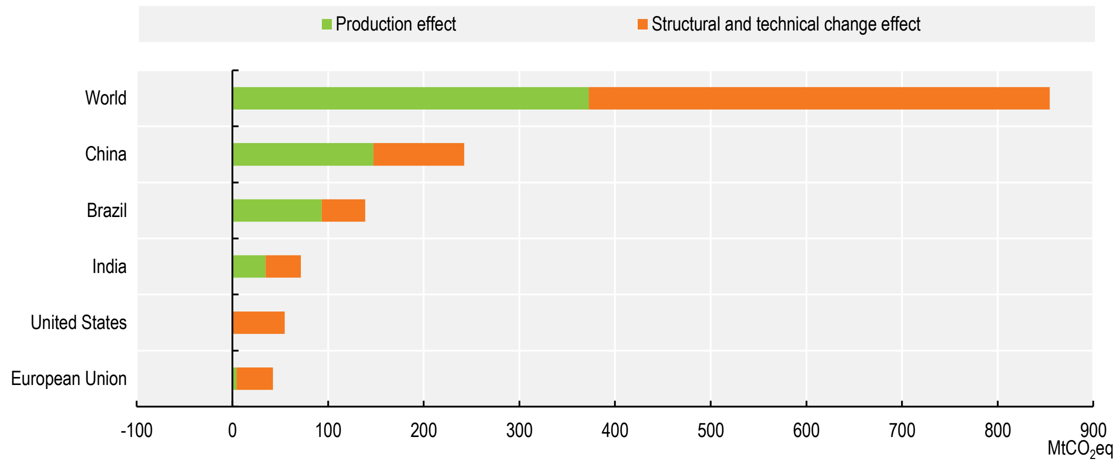


Source Frank et al (2018[7]).

Results

Applying a carbon tax to production reduces emissions from agriculture by about 850 $MtCO_2eq$ in 2030. Globally, more than half of this reduction comes from the assumed improvements in technology adaptation and sub-national reallocation of production (Figure 4.12), while the rest results from changes in production and consumption levels. This reduction is considerably lower than what is presented in Box 4.2 for several reasons. The mitigation potential in Frank et al. (2018[7]) was reported for 2050, hence the baseline level of emissions to mitigate from was higher, and the lead time of the carbon tax was longer which allowed for more pronounced structural change and larger diffusion of mitigation technologies. Finally, the applied carbon tax was higher (USD 100/tCO_2eq) and the LDCs were not excluded from mitigation.

Figure 4.12. Global emission savings in Scenario 5



Source: Aglink-Cosimo simulation results.

The global emission reduction in Scenario 6 amounts to 340 $MtCO_2eq$ (-6%). As expected, the effect on emission savings is lower than the introduced supply shock, as markets adjust to equilibria with lower commodity prices and higher consumed quantities compared to the baseline.

In Figure 4.13, the trade-off between food security indicators and emissions is shown for Scenarios 5 and 6. In Scenario 5 there are more dynamic effects in place than in the scenarios analysed so far because of the lagged response of production to price changes. This is the case for equilibrium prices and the agricultural income index which decreases significantly in the first part of the projection period, increases in the middle part, and levels out towards the end. In the first years, production decisions are already locked in (herd sizes, land allocation) based on expectations that did not include the policy change. Therefore prices do not change much; however, the additional costs in terms of the applied tax strongly reduce income during those years. In the following years, farmers react to the higher costs by reducing production, which then drives prices up and in turn leads to a recovery in production in the next period. By 2030, the increase in prices matches on average the tax applied to each product. Again, it is the consumer who pays most for the GHG emissions reduction. This is clear in the increased consumer price index, which reaches a similar level to that in the consumption tax scenario (Scenario 2).

It seems that once again average calorie availability is not affected strongly; however, the consumer price index increases significantly, putting accessibility at risk especially for the poor. This is different to Scenario 6, where emission reductions are relatively modest but calorie availability shows the strongest increase across all six scenarios. Consumer food prices decrease strongly, putting consumers in a more secure position. The downside of lower prices is visible in the agricultural income index that also decreases significantly, as the effect of additional production quantities through the productivity boost is overcompensated by reduced market prices.

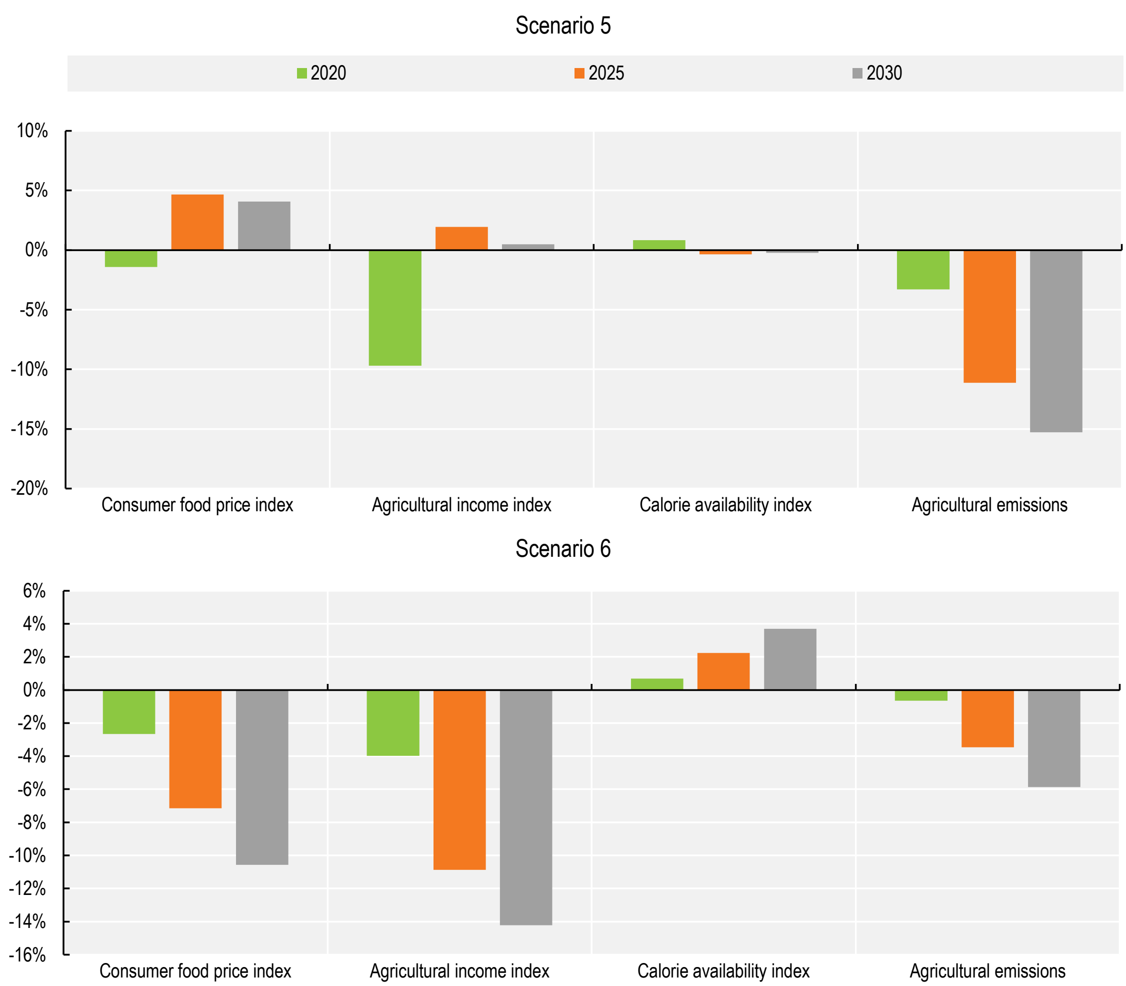
The country comparison (Figure 4.14) shows that the gains in terms of emission abatement in Scenario 5 are higher in Brazil and China compared to the European Union and the United States. This is due not only to larger emission saving potential in the first two countries through technological and structural adjustments, but also because the initial emissions per tonne of product are much higher in these two countries. For example, emissions per produced tonne of beef in 2030 in the baseline amount to over 30 $kgCO_2eq/kg$ in Brazil and China, while the European Union and the United States only emit about 10 $kgCO_2eq/kg$. The emission reduction in India is projected to be relatively low because the emission reduction potential through technological and structural change in the milk-producing sector, which dominates emissions in India, is limited within the simulated horizon.

Figure 4.13 shows the breakdown of emission savings for several major countries/regions in absolute terms. It illustrates that mitigation in the United States and the European Union is almost entirely based on technical and structural adjustments while production levels hardly change compared to the baseline in 2030. In contrast, technical and structural mitigation options in Brazil, China and India appear to be more costly than in the United States or the European Union, and large parts of the emissions saved come through decreases in production.

Contrary to the other scenarios, the spill-over effect to the groups of LDCs tends to be negative as agricultural prices, especially for ruminant products, increase significantly making food on world markets more expensive. At the same time it stimulates local production which leads to increases in GHG emissions in those regions.

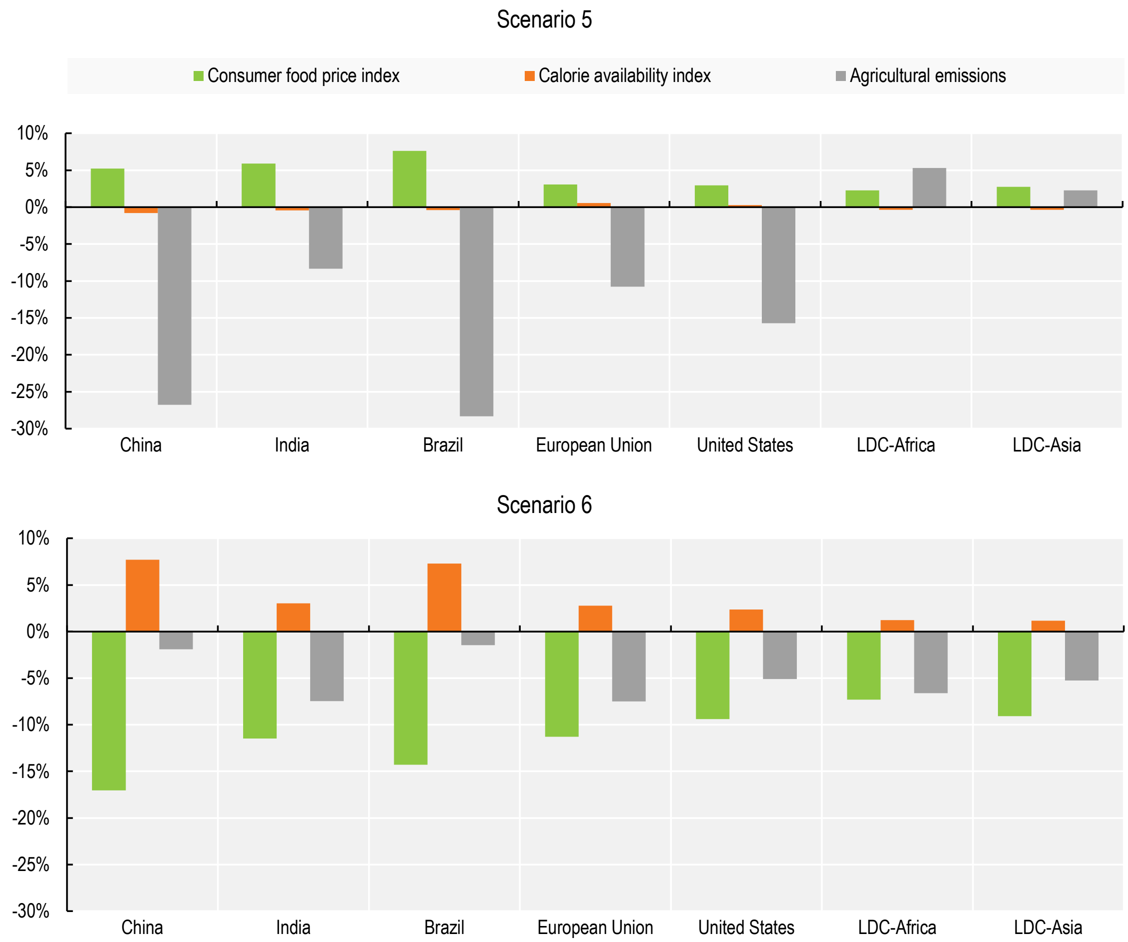
The cross-country comparison for Scenario 6 illustrates how emission savings and calorie availability are interlinked. In Brazil and China, calorie availability is increasing stronger than in India, the European Union, and the United states. Emission savings are lower in the former two countries, however. Obviously, the demand response is more elastic in those countries so that lower prices lead to stronger consumption increases. As a consequence, the emission savings are smaller as higher demand is associated with higher emissions. The spill-over effect is again positive for consumers in LDCs, but negative for net surplus agricultural producers.

Figure 4.13. Emission savings versus food security in relation to the baseline scenario



Source: Aglink-Cosimo simulation results.

Figure 4.14. Regional emission savings versus food security in relation to the baseline scenario, 2030



Source: Aglink-Cosimo simulation results.

Comparison across scenarios

Each of the analysed scenarios reveals potential to reduce GHG-emissions by 2030 as summarised in Figure 4.15. Ranked by emission reduction, the highest potential is found in Scenarios 1 (preference shift in food demand) and 5 (carbon tax on production) where emissions are reduced by about 850 $MtCO_2eq$ (-15%). A reduction by 13% is obtained in Scenario 4 (food waste reduction with costs) followed by Scenario 3 (food waste reduction) (-8%) and the productivity increase Scenario 6 (-6%). The lowest emission reduction is projected under the consumption tax Scenario 2 (-5%). Interestingly, applying a demand tax at the consumer price level appears to be much less efficient than applying the same tax at producer level. This is in line with the general observation that, especially in high income regions, consumers are not so responsive to the final consumer price and so it is not easy to change consumption patterns by implementing taxes on food products. Furthermore, the applied tax is decoupled from the actual carbon produced and thus the only opportunity to reduce emissions is provided through reductions in outputs.

Figure 4.15. Emission savings versus food security across scenarios in relation to baseline scenario, 2030

Source: Aglink-Cosimo simulation results

When including the food security indicators, Scenarios 1, 3 and 6 appear to consistently improve both emissions and food security. However, Scenarios 1 and 3 are also the most difficult to achieve. Reducing food waste without incurring any costs is unrealistic, while influencing consumer preferences such that they consume less ruminant products would most likely prove to be very challenging. Increasing productivity is generally promising, especially given the strong evidence of high returns to research and development, although the cost of implementing such policies could to some extent dampen the price benefits as pointed out by Alston (2010[10]) and Hurley et al. (2016[11]). Scenario 4 clearly illustrates that under the assumption of increasing marginal costs of food waste reduction, the food security goals might be at risk if the objective is to eliminate the final unit of waste. Food security in Scenario 5 might be threatened by higher consumer prices, but still appears to be a promising option for reducing GHG-emissions.

The feasibility of a global carbon tax is questionable. However, production-side mitigation can also be achieved by subsidising emission-reducing technologies, which could reduce emissions without major influences on supply and demand quantities.

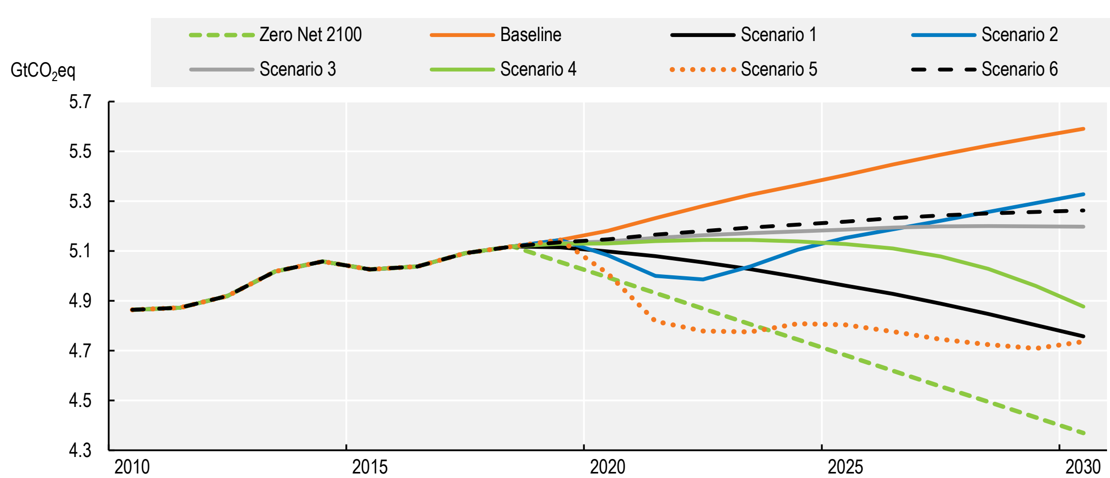
It should be noted that comparability across scenarios is limited as the applied measures and scenario assumptions are not always consistent. Figure 4.16 relates the three food security indices to one percentage point of emissions saved. In this figure the signs are chosen such that a positive number is associated with an increase of food security. For example, in Scenario 1 each percent point of reduced emissions has a positive impact on consumer prices of 0.25 percentage points while it impacts negatively on agricultural income (-0.4 percentage points) and slightly reduces calorie availability. It becomes apparent that in most cases, emission savings and food security measures impact in opposite directions. Scenario 3 and 6 are the only cases where two of the indicators – consumer prices and calorie availability – show positive developments, while Scenarios 2 and 4 impact negatively on all food security indices. This perspective generally favours Scenarios 1, 3 and 6 if the agricultural income effect is set aside.

Figure 4.16. Food security indicators per one percentage point of emissions saved

Source: Aglink-Cosimo simulation results.

Figure 4.17 illustrates how emission reductions in the six scenarios benchmarked against the necessary reductions to comply with the 2-degree target specified in the COP 21 agreement. Rogelj et al. (2016[12]) argue that *"limiting warming to any level requires net CO_2 emissions to become zero at some point in time and, given the small remaining carbon budget, this moment is estimated to be before the end of this century for a 2 °C limit"*. The path to get to that level is, however, not specified. Assuming that all sectors contribute at the same proportionate rate to that target, Figure 4.13 shows the savings that are necessary to reach zero net emissions by 2100 under the assumption that the annual reduction and the emission trajectories of the six scenarios are constant.[13]

Figure 4.17. Emission pathways under scenarios compared to a linear path to net zero emissions, 2100



Source: Aglink-Cosimo simulation results.

It is clear that none of the six scenarios alone could reduce emissions enough to turn onto a path that would lead to zero net emissions of the sector by 2100. The baseline scenario, under which no additional action towards climate change mitigation is assumed, would increase climate change risks considerably. If that series were to continue towards 2100, emissions from agriculture would be 60% higher than in 2020.[14] If all sectors were to perform like this, global warming would be about 4 degrees over that of the current century (Rogelj et al., 2016[12]). To avoid this, climate change mitigation action is needed. Alhough none of the analysed scenarios alone can reduce emissions enough to get on the 2-degree trend, a combination is perhaps sufficient. Given the general positive assessments of Scenarios 1 and 5, a combination of these two scenarios has been carried out. This shows that the emission-savings effect is slightly lower than the sum of those effects in the two single scenarios, leading to an emission level below 4 $GtCO_2eq$, a stronger mitigation effort than required by the 2-degree target pathway and achieved without increasing significant food security concerns.

Implicit trade of emissions in food products is another layer of the mitigation task. Without global agreement and implementation, countries that implement GHG-abatement strategies run the risk of reducing their emissions, e.g. by a decrease in production, but then importing products responsible for even more emissions to replace the shortfall in supply (carbon leakage). However, trade can also allow production to shift to locations with lower emissions intensity (carbon reallocation). This analysis cannot investigate this aspect in detail because it would require that imports be distinguished by origin, which cannot be done in Aglink-Cosimo. Nevertheless, the analysis does illustrate the global changes in carbon reallocation via trade in the analysed scenarios. The trade quantities recorded in the *OECD-FAO Agricultural Outlook 2018-2027* project that traded emissions will increase from 460 in 2017 to 500 $MtCO_2eq$ in 2030. Eighty per cent of these carbon exports can be attributed to trade in beef (55%), sheep meat (10%), and rice (15%). As illustrated in the left graph of Figure 4.18, trade in emissions decreases in all but Scenario 2. In general, rising exports of emissions are not a problem if overall emissions are decreasing, therefore the sign of those exports is not very important. The interesting aspect is a decomposition of the change in emissions trade into changes in trade quantities and emission intensities (Figure 4.18, right column), which reveals that the increase in Scenario 2 is predominantly a source of increase in the emission intensity of trade. This means that the composition of global exports is moving towards countries with higher emission intensities which happen to also be lower-cost producers. For beef, these shifts are dominated by increases of beef exports from Brazil, where beef production emits about 30 $kgCO_2eq/kg$ and decreases in exports from the United States and Canada, where about 10 $kgCO_2eq$ are emitted per kg of beef produced. For sheep meat, global traded quantities decrease, while emissions trade increases. This is mainly caused by export shares that move from the LDCs in Africa, where emissions of producing sheepmeat are estimated at 22 $kgCO_2eq/kg$ in 2020 to Australia (25 $kgCO_2eq/kg$).

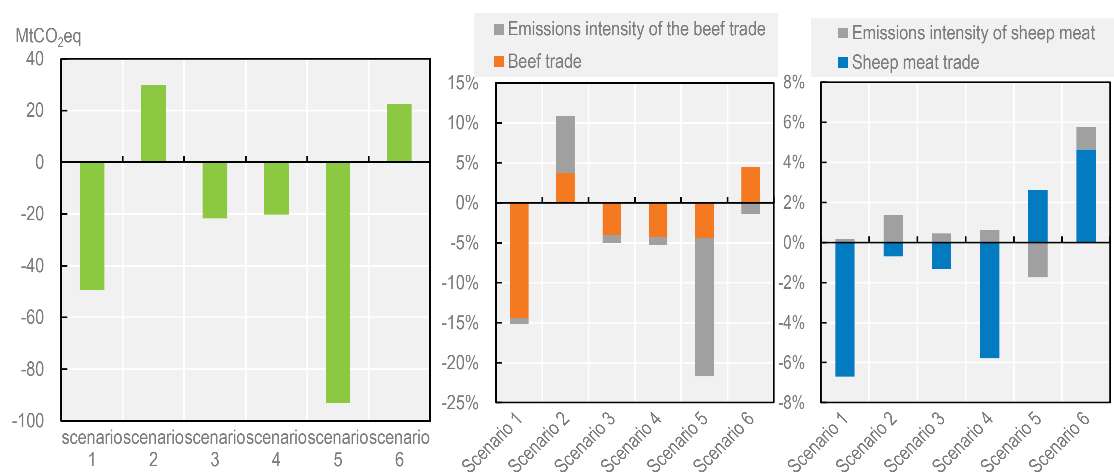
A second observation is that the trade quantity effect dominates in Scenarios 1 and 6. This is intuitive because the scenario setup changes demand (Scenario 5) and supply (Scenario 6) structures in countries through similar shifts, leading to a more equal distribution of the scenario shock across countries so that the relative trade pattern is little affected.

For beef in the food waste scenarios (3 and 4), quantity and intensity changes lead to reductions of traded emissions, with a reduced intensity dimension accounting for the larger shares. Here, the emission intensity reduction stems from export shares moving from Brazil to the European Union. This happens due to higher initial levels of food waste in the meat sectors in the European Union than in Latin America (Figure 4.18), which lead to a stronger shift of beef production to export markets in the European Union. For sheep meat, total emissions traded decrease although emission intensity increases for the same reasons as in Scenario 2, i.e. a shift of export shares towards regions with higher emissions.

The opposite can be observed in Scenario 5 where emissions traded in sheep meat increase and emission intensity decreases. The decrease in emission intensity stems mainly from the reduction of national emissions through technological adjustments and structural change.

At the national level, each of the above mitigation instruments can lead to carbon reallocation through trade. For example, if a country's emissions reductions were to be achieved primarily through supply side mitigation, with consumption relatively unchanged (as would happen under Scenario 4 border measure to contain imports or raise domestic prices), then final emissions would increase if the emissions intensity of those imports were higher than the emissions intensity of domestic production. Conversely, emissions would fall if the emissions intensity of imports were lower than the emissions intensity of domestic production. A global approach to mitigation is needed to ensure that national mitigation efforts are complementary and lead to positive carbon reallocations rather than carbon leakage.

Figure 4.18. Changes in global emission trade (left graph) and changes in emission intensity of trade (right graphs) in relation to baseline scenario, 2030



Source: Aglink-Cosimo simulation results

Conclusions

Agriculture contributes a significant share of greenhouse gas (GHG) emissions at the global level (about 11% excluding the impact of land use changes), but also holds substantial potential to limit the increase in global warming over the next decades. In a 2018 report prepared for OECD, Blandford and Hassapoyannes (2018[1]) observed that the agricultural sector could limit this increase by reducing direct emissions in crop and livestock production systems and in indirect emissions that are associated with changes in land use, as well as by increasing carbon sequestration. They found that technological advancements on the supply side and changes in consumer preferences on the demand side that result in land-sparing are promising options, particularly in view of global food security concerns.

This report has analysed agriculture's potential contribution to climate change mitigation by simulating supply and demand side mitigation strategies. It uses the Aglink-Cosimo model based on the baseline of the *OECD-FAO Agricultural Outlook 2018-2027* and takes into consideration only direct emissions that result from agricultural crop and livestock production activities. It does not analyse the most efficient way agriculture can contribute to overall mitigation as this also depends on changes in land use. However, all scenarios presented here should reduce the pressure on land and thus the results underestimate the full mitigation potential of the sector

Six scenarios have been analysed: four relate to demand-side mitigation measures (a preference shift towards less ruminant products, taxing food according to related emissions, and two variants of food waste reduction) and two address supply-side mitigation (carbon tax and productivity shift). The effects on emissions have been compared with the impact on food security as captured by indicators of food availability, food prices and on farm incomes. The analysis finds the following.

- Influencing consumer preferences so that more calories are obtained from non-ruminant animal sources has the highest benefits among the analysed scenarios. However, the mechanism by which such a change could be achieved is not specified.
- Consumption taxes are the least effective measure to reduce greenhouse gases, especially when these are decoupled from the actual carbon produced, owing to the inelasticity of demand for broad food groups, and would raise food prices, potentially leading to food security risks for low income consumers.
- Reducing food waste can be a strategy to mitigate climate change, but it is important to take into consideration that the potentially high costs to reduce waste could raise food prices, and potentially lead to food security concerns.
- Supply side mitigation via carbon taxes has a high potential to reduce emissions from agriculture with limited risks in terms of food security.
- Increasing productivity in agricultural production systems could potentially reduce emissions and increase food availability, in addition to improving access via lower prices.
- A global approach to mitigation is needed to ensure that national mitigation efforts are complementary and lead to positive carbon reallocations rather than carbon leakage.

Notes

[1] Ongoing investments in Aglink-Cosimo should allow emissions from LULUCF to be included in future analysis. The exclusion of the land use sector from this partial analysis should not, however, be viewed as a significant shortcoming. Instead, it allows to focus on the analysis on direct emissions. The six scenarios developed for this chapter should reduce pressure on land, whereas including the land use sector would not lead to a compensation of the analysed emission reduction potential. Relative potentials might change slightly depending on differences in the impact of total cropland.

[2] According to the FAO definition, “food security exists when all people at all times have physical and economic access to sufficient, safe and nutritious food to meet their dietary needs and food preferences for an active and healthy life”.

[3] Hasegawa et al (2018[3]) illustrate that food availability is also correlated with prevalence of undernourishment, so an increase of that availability should also improve the undernourishment situation. This study does not, however, go deeper into that issue.

[4] The Aglink-Cosimo model does not include explicit costs, and this index only covers implicit cost increases via input price changes, but not via input quantity adjustments.

[5] The choice of the carbon tax level was based on the fact that this price corresponds to the value that some modelling studies suggest will be required to limit temperature increases to 1.5°C (Rogelj et al., 2015[14])

[6] This assumption is simplified as the emissions that are implicitly contained in final consumption depend on the origin of each product. A large share of imported commodities could change the emission coefficient significantly. This effect could, however, only be captured by a model capturing bilateral trade.

[7] This tax is applied at the primary consumer level as emissions arising from the processing industry are not accounted for.

[8] Indeed, if the effect of the collection of taxes was included, the various possibilities of use could impact income or final prices, and the conclusions drawn from this scenario might change. This is also true for Scenario 5. However, one principle of this report is not to have too many overlaying parameters changed in one scenario in order to assess the pure effect of the instrument in place.

[9] This is by construction in Scenario 1, while it is an endogenous outcome in Scenario 2.

[10] Trade in emissions is not covered in this analysis, but it is fair to assume that imports which are consumed instead of goods domestically produced, the former would be produced with lower emission intensity in the exporting countries as emission intensities are higher in the LDCs.

[11] The scenario assumptions are a strongly simplified representation of the mechanisms that would be in place when applying different waste reduction technologies. This scenario is primarily meant to illustrate that neglecting waste reduction costs, as in Scenario 3, is dangerous. The *EU Agricultural Outlook 2018-2030* includes a box on the effects of reducing food waste in European households. Using a CGE model, the authors attribute waste avoidance costs mainly to improved packaging, and they estimate those costs to be between 1% and 5% of sales.

[12] The evolution of the consumer price index in Scenario 4 cannot be interpreted directly as increasing consumer prices. It does indicate that other (opportunity) costs of acquiring food by households increases significantly when waste is eliminated.

[13] The two-degree compatible emission reduction from agriculture by 2030 using this method amounts to about 650 $MtCO_2eq$. This is below the range of estimates reported in Wollenberg et al (2016[13]). They estimate that on average it would require an annualized reduction of 1 $GtCO_2eq$ in 2030. It should also be noted that such an individual trajectory, for a given sector, has only very limited scope and relevance. On the one hand, the various sectors do not have the same mitigation potential, and on the other hand, these potentials differ according to the boundaries of the various sectors. For example, the relative reduction potential for agriculture would not be the same if the effect of the substitution of fossil energy by biomass was attributed to it, or if it was accounted for in the industry or transport sector, as is currently the case. It is likely that agriculture will continue to have net positive emissions, but other sectors, especially LULUCF, could compensate for this.

[14] A linear extrapolation is most likely no proper approximation of a long-term business as usual scenario. Slowing population growth and increasing saturation of demand will lead to lower growth in emissions. Havlik et al (2014[9]) estimate that agricultural emissions will be only 30% higher in 2100 compared to 2020.

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Annex 4.A. Methodology

Extensions to Aglink-Cosimo

In the course of this project, the Aglink-Cosimo model was extended to capture the necessary aspects to analyse emission-related scenarios.

According to FAOSTAT, the agricultural sector accounts for about 10% of Global Greenhouse Gas (GHG) emissions. Although the database reports this share to be decreasing over time, absolute emissions have been increasing over the past decades. It can be observed, however, that in some countries emissions continue to increase with rising production levels, and in other countries they remain stable or decrease despite increasing production, thus implying decreasing emissions per unit produced over time. The development of such emission factors over time are the result of more efficient input use as well as technological changes and conversion to different production systems.

Earlier work that aimed to include emission factors into the Aglink-Cosimo model did not reflect these dynamics over time, which are essential for comprehensive reporting. The model reflects production technologies and agricultural inputs not explicitly, wherefore a direct link of emissions to their sources is not possible.

During the first half of 2017, collaboration between the OECD and the International Institute of Applied System Analysis (IIASA) was established in order to obtain – as a first step in the direction of analysing climate change related questions with Aglink-Cosimo – dynamic emission coefficients for a baseline scenario. IIASA was chosen because its Globiom model is among the leading models that can capture some of those aspects driving emission factors over time. The idea of this project was therefore to align the production quantities assumed in the *OECD-FAO Agricultural Outlook* with those of the Globiom baseline and incorporate the resulting emission factors into the Aglink-Cosimo model to calculate total emissions from agriculture.

Since the two models differ in regional and commodity scope, a mapping between the two model codes was developed, including possible conversion/aggregation factors that reflect possible differences at the processing stage of the two models. The OECD then provided the GLOBOIM team with detailed time series and assumptions needed to align the Globiom baseline with the *Outlook*, and a time series of emission factors from 2000 to 2030 was calculated. These factors were then added to the post-model calculation of Aglink-Cosimo. They are available disaggregated according to the categories available in FAOSTAT:

- CropSoil_N2O N2O emissions from applying mineral fertilizer to crop soil
- Rice_CH4 CH4 emissions from cultivating Rice
- ManmgtTot_N2O N2O emissions from Manure management
- ManaplTot_N2O N2O emissions from applying manure on the field
- ManprpTot_N2O N2O emissions from manure left on pasture
- ManmgtTot_CH4 CH4 emissions from manure management
- Entferm_CH4 CH4 emissions from enteric fermentation

These positions sum up to the

- Total_CH4N2O Total direct non-CO_2 from agriculture

All coefficients have been converted to CO_2 equivalents using the conversion coefficients reported in the IPCC Fourth Assessment Report and are available for the following countries/country aggregates:

- USA United States
- EUN European Union
- BRA Brazil
- CAN Canada
- CHN China
- JPN Japan
- MEX Mexico
- KOR Korea
- TUR Turkey
- ZAF South Africa
- IND India
- RLAM Rest of Latin America
- AUNZ Australia and New Zealand
- MNAF Middle East and North Africa
- OCEL Other Oceania
- REUW Rest of Western Europe
- SSAF Sub Saharan Africa
- SSEA South-East-Asia
- ECSI Eastern Europe (including the Russian Federation)

As Aglink-Cosimo does not cover all food commodities, it currently captures the emissions of the agricultural sector partially. Emission coefficients exist for the following commodities:

- BV Beef and veal
- CT Cotton
- EG Eggs
- MA Maize
- MK Milk
- OCG Other coarse grains
- OOS Other oilseeds
- PL Palm oil
- PK Pork
- PT Poultry
- RI Rice
- RT Roots and tubers
- SB Soybeans
- SCA Sugar cane
- SH Sheep and goat meat
- WT Wheat

This subset of products covers about 80% of global emissions caused by the agricultural sector. Although it would be clearer to link the emission coefficients to the activity levels (e.g. land use and herd sizes), due to a simplified presentation of the animal sector in Aglink-Cosimo in particular, this is not done and emissions are calculated as a function of production quantities:

$$\mathrm{Emis}_{c,p,e,t} = \frac{\beta_{c,p,e,t}\mathrm{QP}_{c,p,e,t}}{1000}$$

The Emission Emis of type e in region c for product p in year t measured in $MtCO_2eq$ per year are calculated by multiplying the production QP (measured in 1000 t) of a product by the emission coefficient β measured in $kgCO_2eq$ by kg of product.

Total emissions are then the sum over the single emission types e.

This enhancement makes it possible to assess the emissions path which is inherent to the OECD-FAO agricultural outlooks, but also in most scenarios that can be analysed with Aglink-Cosimo. It does not allow to address supply side mitigation policies, as the average emission coefficient of a country would adjust endogenously as soon as those technologies change. The Globiom Team at IIASA was asked to address this problem.

Globiom has been used in the past to develop the MACCs (Marginal Abatement Cost Curves) for the agricultural sector e.g. for the UK DECC model GLOCAF, and for integrated assessment models such as MESSAGE (IIASA), POLES (JRC), or WITCH (FEEM). A detailed agricultural non-CO_2 MACC analysis is published in Frank et al (2018[7]) However, MAC curves cannot be used directly in Aglink-Cosimo, and an alternative approach was developed.

Globiom explicitly covers the following non-CO_2 emission sources: N2O from application of synthetic fertilizer, CH4 from rice cultivation, N2O from manure dropped on pastures, N2O from manure application, CH4 and N2O from manure management, and CH4 from enteric fermentation. The global amount of emissions in a mitigation scenario will be the result of three endogenous mechanisms.

Regional GHG intensity change

- Management / production system change
- Spatial relocation within a region/country (crops)
- Technological mitigation options (e.g. biodigesters)

Global GHG intensity change

- Average regional GHG intensities for individual products differ substantially across regions, hence worldwide relocation of the production through international trade to more or less GHG intensive regions will change the global average GHG intensity

Global production volume change

- Result of a change in food and feed consumption potentially related to increased market prices, themselves being result of the additional production cost related to mitigation policies

Aglink-Cosimo has the internal capacity to deal with Scenarios 2 and 3 – reduction of GHG emissions via international trade and consumption side adjustments – based on the GHG emissions coefficients derived. For example, for a simple climate policy implemented as a carbon tax, the supply curve of a given product in a given region would be shifted upwards by the product of the emission factor and the carbon price and the model would endogenously adapt demand and international trade.

The only missing mitigation mechanism that needed to be added to the Aglink-Cosimo based on additional Globiom input were the “Regional GHG intensity” improvements as a response to climate change policy. Similarly, as for the above mentioned MACCs, the parameters from Globiom to Aglink-Cosimo would be derived by running a series of scenarios covering the whole relevant range of carbon prices representing the cost efficient contribution of the agricultural sector to a wide range of overall mitigation effort levels. In order to avoid double counting, since Aglink-Cosimo represents endogenously consumption side and international trade adjustments, these mechanisms were fixed in Globiom to the baseline levels. Globiom being a standard economic model, endogenous adjustments in management systems, intra-regional spatial allocation of activities, and adoption of mitigation technologies take place as long as the marginal cost of emissions reduction are less or equal to the carbon tax. This approach allowed to derive a new set of emission coefficients corresponding to each carbon price level. The new coefficients can now be implemented in Aglink-Cosimo depending on the carbon price / mitigation effort scenario considered allowing the model to capture the production side adjustment to mitigation incentives.

Given that in the Globiom scenarios the marginal cost of reduction (abatement) of the emissions coefficient is always equal to the carbon price, the Aglink-Cosimo supply curves would need to be shifted by the “baseline” emission coefficient multiplied by the carbon price. Potential alternative effects of the management change on the necessary supply curve shift could be included in Aglink-Cosimo for specific additional climate policies. For example, the farm-level mitigation measures were implemented in the form of a subsidy rather than a tax. In this case, if the region-level management adjustment led to a halving of the emission intensity of a given product, and the adoption of such management were supported by a program fully covering the cost difference between the “baseline” and “mitigation” management system, the supply curve would be shifted only by half compared to the scenario without a subsidy.

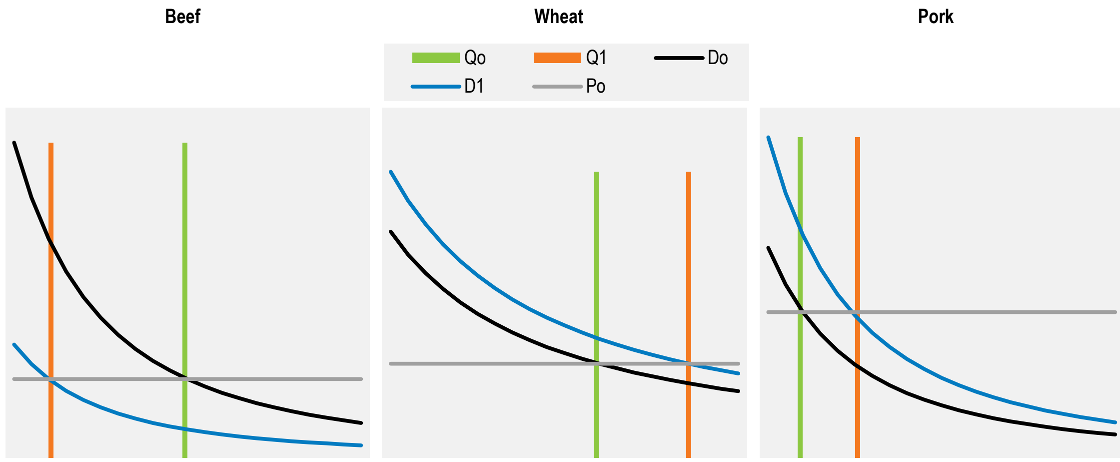
The missing elements to get substantially closer to the complete AFOLU GHG emissions accounting are emissions from other land uses and carbon sequestration. Since there are potential interactions between non-CO_2 emissions and CO_2 emissions, it would be useful to account for both to avoid unintentional negative effects, such as reduction of nitrogen fertilizer use leading to reduced N_2O emissions but also larger CO_2 emissions from additional land cover change due to lower yields. This aspect will be added in the future.

Technical scenario implementation

Scenario 1 reflects a preference shift in demand away from ruminant production such that demand for ruminant products decreases 10% below the base year values. This scenario was implemented by shifting the demand curves for different food items. Figure 4.A.1 illustrates how Scenario 1 was implemented using a simplified example whereby only three food commodities exist in the consumption basket: beef, wheat and pork. The graph gives the initial demand curves for the three products in calorie equivalents (Do) with the corresponding price quantity combinations (Po and Qo) in the baseline. The demand function of beef is moved to the left (D1) so that the new quantity Q1 would be demanded at the baseline price Po. Q1 would be derived from the assumption described above (10% below current per capita values). The difference between Q1 and Qo in the left graph is then distributed between the other two commodities with wheat taking larger shares, because the initial demand quantities were higher for wheat than for pork. Consequently, the new demand curves for wheat and pork are located to the right of the original ones.

Naturally, as demand remains an endogenous scenario output, the simulated demand quantities will not be exactly equal to the initial assumptions, as prices will move away from the baseline equilibrium as supply adjusts. Such a shift in demand behaviour could be achieved by influencing consumer preferences. This shift in preferences was applied globally, except in the LDCs.

Annex Figure 4.A.1. Shifting demand preferences in Scenario 1



Scenario 2: In this scenario, all products are taxed at the consumption level based on their primary emissions. The consumption tax is set at USD 60 (in year 2000 real USD) per ton of CO_2eq emitted by each product. It is applied globally, again except in the LDCs. Technically this tax is added to consumer prices that enter the food demand equations.

Scenario 3: In this scenario, wasted food at consumption level is reduced to zero without any costs. The food demand variable in Aglink-Cosimo includes wasted quantities implicitly. Those implicit values were estimated based on the FAO (2011[5]) study and then gradually deducted from the food demand variable by shifting the demand curves to the left. Technically this is done by adjusting the R-factors of the food demand equation:

$$\log(FO_{r,c,t}) = \alpha + \sum_{c1(food)} \beta_{c1} * \log\left(\frac{CP_{r,c1,t}}{CPI_{r,t}}\right) + \beta_1 * \log\left(\frac{GDPI_{r,t}}{POP_{r,t}/POP_{r,2005}}\right) + \log(POP_{r,t}) + \beta_2 * TRD + \log(R)$$

Where:

FO	=	Food use
CP	=	Consumer price
CPI	=	Consumer price index
GDPI	=	Nominal GDP index
c1(food)	=	Commodities with food use
β_{c1}	=	Cross- and own-price elasticities
POP	=	Population
TRD	=	Trend
R	=	Residual calibration factor
r	=	Regions
c	=	Commodities
t	=	Years

An additive logged variable in this double log representation corresponds to a multiplicative variable in the un-logged version. Therefore, the R-factor is a multiplicative scaler to the food use variable. Since the idea in this scenario is that if no waste exists demand goes down by the wasted amounts, the R-factors can be used to define those. For example, FAO data says that in the European Union, 25% of wheat is wasted at consumption (and retail) level. If those quantities vanish, 25% less of the product would be needed to achieve the same final consumption level. Therefore, the original R-factor of the food use equation of wheat in the European Union was multiplied by 0.75 in the final simulation year (2030). Between the first and the final simulation year this factor has been gradually decreased from 1 to 0.75.

Scenario 4: This scenario is implemented in the same way as Scenario 3 but it increases consumer prices in relation to waste abatement levels. A crucial assumption is the level of consumer price increase for the final unit of waste abatement. Since data on this does not exist, assumptions were made to illustrate the importance of reflecting waste abatement costs. The assumption that the final unit of waste abatement costs as much as the consumer pays for the respective product in the baseline was taken. This means the consumer must effectively pay in the final simulation year twice as much for one unit of product as in the baseline. Between the first and the final simulation year, these costs are not assumed to increase linearly, but exponentially. This reflects the assumption that the first units of waste abatement are relatively cheaper than the following ones.

Scenario 5: A production tax of USD 60 (real USD, 2000) per tonne CO_2eq emitted was imposed on agricultural production activities by shifting the supply curves for each commodity upwards by the amount of the tax. In other words, marginal production costs increase by the amount of tax applied. Based on the collaboration with IIASAA described above, the emission coefficients that were derived from Globiom simulations with the same tax level were also applied.

Scenario 6: A productivity shift in the crop sector was simulated by shifting the yield equations using the R-factor in a similar way as described for Scenario 3. For animal products, where yields are not explicit variables but inherent from the production – and herd size equations, production equations were shifted by the applied 10% and it was ensured that the resulting higher production quantities were achieved with the same herd sizes c.p. Note that by doing this, the yield increase came at no additional cost.

5 Global mitigation potential of biofuels in the transport sector

Based on the *OECD-FAO Agricultural Outlook 2018-2027* baseline, this chapter examines the potential contribution of biofuels to climate change mitigation in the transport sector

Introduction

The International Energy Agency's (IEA) *Energy Technology Perspectives* (IEA, 2017[1]) foresees that bioenergy will play an important role in climate change mitigation. The IEA has defined a mitigation scenario, the 2-degree scenario (2DS), that is consistent with a 50% chance of limiting future global average temperature to an increase of 2°C by 2100. This scenario is developed on assumptions of the future evolution of crude oil prices, the macroeconomic environment, and policies that concern the whole energy system (including the transport sector), including associated intended impacts on transportation fuel demand. In the 2DS, carbon taxes are applied according to the well-to-wheel (WTW) greenhouse gas (GHG) emission profiles of the different types of fuel which increase over time.

The IEA 2DS defines a target for energy-related GHG emissions reduction; which involves lower transport demand and more transport fuel coming from lower GHG emitting fuels. The IEA 2DS does not assess the capacity of agriculture to deliver the ambitious volume of biofuels foreseen in this pathway over the coming decade and is not able to estimate implications for agricultural markets.

This chapter presents the results of the AGLINK-COSIMO 2-degree scenario (AC-2DS) simulated with the AGLINK-COSIMO model. The AC-2DS uses assumptions consistent with the IEA 2DS assumptions. In particular, the IEA 2DS increasing path for carbon taxes is replicated. The AC-2DS can assess the potential impacts of a mitigation scenario on both biofuel and agricultural markets across the globe up to 2030.

The analysis complements and assesses the path developed by the IEA by including an agricultural sector perspective. Indeed, the AGLINK-COSIMO model is a partial equilibrium model that is able to take into account the interconnection between agricultural and biofuel markets[1] and the ability of agricultural activities to supply the amount of bioenergy the IEA has determined is needed to meet climate change targets.[2] This chapter also proposes ways to enrich and develop the analysis of those linkages.[3]

Biofuels and greenhouse gas emission savings in the transport sector

Greenhouse gas emissions associated with biofuels

Biofuels are fuels produced by the transformation of biomass. They can be blended with or replace conventional fossil fuels.[4] This chapter focuses on two kinds of biofuels: ethanol and biodiesel.[5] Ethanol and biodiesel are used, respectively, as gasoline and diesel substitutes or complements. Similar to conventional fossil fuels, biofuels are composed of hydrocarbon chains. Emissions of greenhouse gases (GHGs) occur at each step of the biofuel value chain: during plant growth,[6] crop harvest, transportation of the feedstock to the processing plant, conversion process, distribution to the fuel terminal, WTW GHGs emissions for biofuels[7]. This chapter does not consider specific GHG individually, but looks at the aggregate of all GHGs identified, expressed in CO_2eq.

The WTW carbon intensity of biofuels can be represented by an "emission factor",[8] expressed in kgCO_2eq per unit of energy content. A literature review (Annex 5.A) was conducted to gather WTW emission factors for different biofuel pathways (Box 5.1). The WTW emission factors collected in this literature review were then averaged to derive a set of WTW emission factors for each biofuel type to be used in the present analysis. Most studies reviewed take into account the by-product use of biofuel food and feed feedstock when allocating WTW GHG emissions to the different biofuel feedstock. The literature review does not cover the growing discussion concerning carbon sinks, i.e. carbon capture by oceans and soils when any types of fuel is combusted.

Biofuel-related greenhouse gas emission savings at the 2030 horizon in the transport sector

Table 5.1 provides an overview of WTW emission savings using the baseline of the 2018 *OECD-FAO Agricultural Outlook* for the period 2015-2017 and by 2030 for major biofuel consuming countries. Box 5.2 describes major trends in biofuel baseline projections.

Table 5.1. Biofuel blending in transportation fuels and associated WTW emission savings

	Ethanol volume share in gasoline type fuels (%)		Ethanol-related WTW emission savings in gasoline type fuels (%) (1)		Biodiesel volume share in diesel type fuels (%)		Biodiesel-related WTW emission savings in diesel-type fuels (%) (2)	
	Average 2015-17	2030	Average 2015-17	2030	Average 2015-17	2030	Average 2015-17	2030
World	7.7	8.2	2.0	2.3	3.5	3.8	1.6	1.7
North America								
Canada	5.6	5.6	0.9	0.9	1.2	2.0	0.4	0.5
United States	9.6	11.8	1.6	2.1	4.0	4.4	2.1	2.3
Latin America and Caribbean								
Argentina	10.0	12.1	3.1	4.0	9.0	12.0	3.3	4.4
Brazil	46.6	51.1	33.6	38.3	7.4	10.0	3.4	4.6
Colombia	7.4	7.1	3.3	3.1	6.4	6.2	2.1	2.0
Paraguay	21.8	26.6	9.8	12.4	0.9	1.0	0.3	0.2
Europe								
European Union	4.8	4.7	1.0	1.0	6.1	6.1	2.8	2.9
Asia								
China (3)	2.0	2.0	0.3	0.4	0.4	0.7	0.3	0.6
India	2.6	2.4	1.0	0.9	0.1	0.1	0.1	0.1
Indonesia	0.1	0.1	0.1	0.1	5.9	8.9	1.9	3.0
Japan	1.4	1.7	0.7	0.8	0	0	0	0
Malaysia	..	..	..	..	2.1	3.2	0.7	1.0
Philippines	9.6	9.5	3.4	3.7	2.1	1.6	0.9	0.7
Thailand	12.8	15.0	6.2	7.5	5.5	6.0	1.8	2.0
Oceania								
Australia	1.1	1.2	0.2	0.2	1.7	1.1	0.6	0.8

Note: Not available

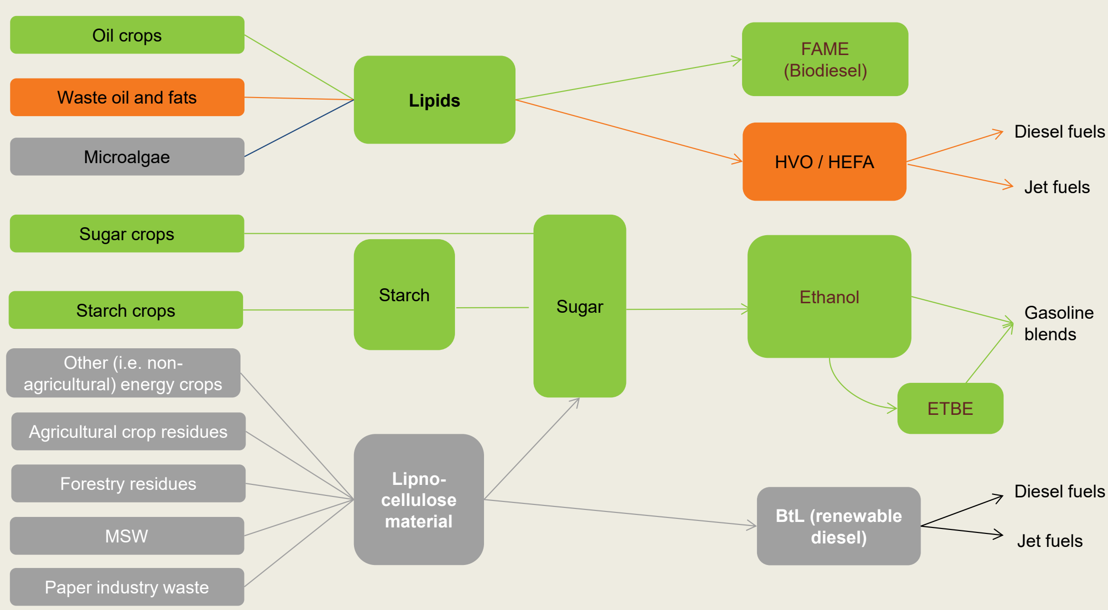
1. The WTW ethanol percentage savings were calculated in a given country as the ratio of the difference between the WTW reference emissions that would have been associated with gasoline if ethanol was not used to replace gasoline, and the WTW emissions associated with the mix of ethanol and gasoline use over the total emissions associated with gasoline use in the country.
2. The WTW biodiesel percentage savings were calculated in a given country as the ratio of the difference between the WTW reference emissions that would have been associated with diesel if biodiesel was not used to replace diesel and the WTW emissions associated with the mix of biodiesel and diesel use and over the total emissions associated with diesel use in the country.
3. Refers to mainland only. The economies of Chinese Taipei, Hong Kong, China (China) and Macau, China (China) are included in the Other Asia Pacific aggregate.

Box 5.1. The pathways of biofuels production

Terms commonly used to classify biofuels relate to feedstock used, the production process, and their capacity to reduce GHGs emissions or policies in place. Biofuels derived from food and feed crops are commonly classified as "conventional biofuels" or "first-generation biofuels". Another classification is "advanced biofuels". Identical biofuels may be considered advanced in some countries and not in others. The IEA (IEA, 2017[3]) defines "advanced biofuels" as sustainable fuels produced from non-food crop feedstocks which are capable of delivering significant life-cycle emissions savings compared with fossil fuel alternatives, and which do not directly compete with food and feed crops for agricultural land or cause adverse sustainability impacts. The terms "second-" and "third-generation" biofuels are used without any agreement on their definitions.

It is not intend here to classify biofuels into categories, but to use the IEA framework to focus on biofuel pathways which combine a source of biomass, a production process, and an end product. Figure 5.1 presents the major liquid biofuel pathways in the transport sector and provides an overall view of the sources of the relevant biomass. Depending on the biofuel production process, certain molecules such as lipid or sugar solutions are extracted from the feedstock. In the most recent conversion processes, ligno-cellulosic material is used directly.

Figure 5.1. Major biofuel pathways in the transport sector



Note: A biofuel pathway is the combination of a source of biomass, a conversion process and an end product.
1. Biofuel pathways not modelled endogenously in AGLINK-COSIMO appear in grey.
2. MSW corresponds to Municipal solid Waste.
3. Some companies are also planning to produce biodiesel from recycled plastics.

WTW GHGs emissions savings from the use of biofuels were calculated as the difference between the amounts of GHGs actually emitted by the use of a mix of biofuels and fossil fuels and the emissions that would have arisen from an equivalent amount of energy being supplied by fossil fuels only[9]. Annex 4.B presents how WTW GHGs emissions are taken into account in the AGLINK-COSIMO modelling framework for the *Agricultural Outlook* (OECD/FAO, 2018[1]).

Box 5.2. Biofuel market prospects towards 2030

Main findings from the 2018 OECD-FAO Agricultural Outlook baseline

The biofuel industry is relatively recent, with the volumes consumed becoming significant only in the 1990s. Biofuel policies continue to play a major role on biofuel markets. In 2017, ethanol accounted for 8.1% in volume of global gasoline-type fuels consumed in the road transport sector while biodiesel accounted for 3% in volume of global diesel-type fuels (OECD/FAO, 2018[1]).[1]

Consumption is highly concentrated among several key players with 12 countries representing 97% of the biodiesel and ethanol fuel use. The United States and Brazil dominate the ethanol market, representing respectively 50% and 27% of global ethanol production. The European Union and the United States, representing 39% and 19%, respectively, of global volumes, lead world biofuel production.

The *OECD-FAO Agricultural Outlook 2018-2027* describes in detail the expected developments on biofuels markets. The *Outlook* is based on projections established at the 2030 horizon with the AGLINK-COSIMO model (see Annex 5.B for the main features of the biofuel component of the AGLINK-COSIMO model).

The 2018 *Outlook* assumes a continuation of current policies, although some general policy targets especially in developing countries would not be met due to the absence of the necessary policy instruments to achieve them. The announcement concerning the Chinese E10 program[2] and the Brazilian RenovaBio program[3] have not been included in the *Outlook*.

At the global level, ethanol production (including ethanol used for industrial purposes and beverage) is projected to expand to 133 Bln L by 2030 (compared to 120 Bln L in 2017), while biodiesel production is projected to increase to 40.3 Bln L by 2030 (compared to 36 Bln L in 2017). By 2030, 55% of global ethanol production is expected to be based on maize and 27% on sugarcane. About 21% of global biodiesel production is projected to be based on waste vegetable oils and animal fat. Lignocellulosic biofuels are not expected to take off over the projection period as production costs are likely to remain high.

Biofuel trade is likely to remain limited. Potential ethanol exporters are the United States, as constraints associated with vehicle suitability for high blends and fuel distribution infrastructure will likely limit a further increase in domestic demand, and Brazil. On the biodiesel side, Argentina will likely be the major player, but with limited import demand.

Notes:

1. The *OECD-FAO Agricultural Outlook* provides ten-year projections for world agricultural markets. However, the model used for projections in the 2018 report has been extended to 2030 for purposes of scenario analysis.
2. In September 2017, the Chinese government proposed a new nationwide ethanol mandate expanding the mandatory use of E10 fuel from 11 trial provinces to the entire country by 2020. The underlying rationale has not been clearly stated but could be related to abundant grains stocks and to environmental concerns. The mechanisms for the implementation and enforcement have not been announced. If fully implemented these policies could have important impacts on biofuel and agricultural markets.
3. The RenovaBio program was officially signed in January 2018 as a follow up to the Brazilian commitment to reduce greenhouse gas emissions by 37% in 2025 and 43% in 2030 compared to 2005. Its implementation plan is not yet defined. The program defines a minimum blending target for anhydrous fuel ethanol that should reach 30% by 2022 and 40% by 2030 as expressed in volume terms. The fuel ethanol share in the fuels matrix should reach 55% by 2030.

At the global level, while the share of ethanol volume in gasoline type fuels is projected to be 8.2% in 2030, ethanol-related WTW emission savings in gasoline-type fuels are estimated at around 2.3%. For biodiesel, the volume share in diesel type fuels is projected to be 3.8% at the global level in 2030 and biodiesel-related WTW emission savings in diesel-type fuels at around 1.7%. The disparities across countries are significant and depend on the blending mandates or targets in place, as well as the type of biofuels used.

Emissions related to land use change

Concerns about the increasing pressure placed on natural resources and land use changes (LUC) effects created by biofuel production and associated GHG emissions arose in the late 2000s, along with other concerns on the sustainability and potential negative impact of biofuels.

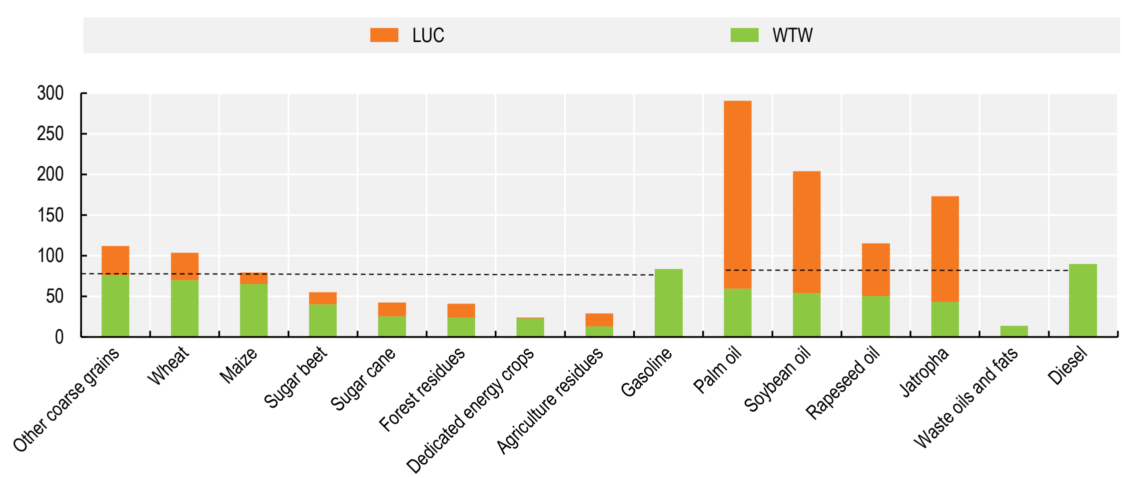
Multiple studies since 2009 examine the extent and consequences of LUC (LCFS, 2009[3]) (Laborde, 2011[4]) (De Cara, 2012[5]) (ECOFYS, 2015[6]), (Overmars et al., 2015[7]), (European Parliament and European Council, 2015[8]). The majority use economic models to estimate LUC impacts: they compare land uses in the baseline situation with a scenario that assumes a different path for biofuel demand. Most evaluations find that the development of biofuels leads to changes in land use[10] which result in substantial GHGs emission impacts.

In this context, the overall carbon intensity of biofuels would have to take into account two components: the CO2 emitted along the value chain (WTW emissions) and the CO2 emitted because of LUC changes associated with biofuel use (LUC emissions).

Figure 5.2 provides an overview of potential WTW and LUC carbon intensity of the most important biofuel pathways compared to the fossil fuels they replace. Estimates for LUC emission factors were derived from the study commissioned by the European Commission and conducted by the Ecofys, International Institute for Applied Systems Analysis (IIASA), and E4tech consortium in 2015 based on the GLOBIOM economic model (ECOFYS, 2015[6]). It is the most comprehensive study undertaken to date.

With these estimates, the results in terms of GHGs emission savings from the blending of biofuels in conventional transportation fuels up to 2030 based on the *Agricultural Outlook* baseline are very different from those presented in the previous section.

Figure 5.2. Carbon intensity (WTW and LUC) of different categories of biofuels in $kgCO_2e/GJ$



Note: LUC emission factors include direct land use changes and indirect land use changes
Source: Literature review undertaken by the OECD (Annex B) for WTW emission factors and Ecofys (2015) for LUC emission factors.

At the global level, calculations based on the *Agricultural Outlook* baseline quantity projections establish total ethanol-related emission savings in gasoline-type fuels at 0.7% by 2030, compared to 2.3% when only WTW emissions are taken into account. For biodiesel, the picture even shows negative savings (-3.4% in 2030 compared to 1.7% when only WTW emissions are taken into account). This means that the blending of biodiesel in diesel type fuels could lead to an increase in cumulative carbon dioxide emissions throughout the projection period when emissions related to land use changes are taken into account based on the ECOFYS study.

The 2018 version of the AGLINK-COSIMO model includes a GHG component for the agricultural sector,[11] which is used in the analysis of agriculture's potential contribution to climate change mitigation. To date, however, emissions arising from LUC are not included in the AGLINK-COSIMO biofuel component. In a future version of the GHG component developed in collaboration with the International Institute for Applied Systems Analysis (IIASA), it will be possible to assess GHGs emissions associated with LUC.

The present chapter adopts an approach similar to the IEA by focusing on WTW emissions in the transport sector and uses the GHG component of the agricultural part of AGLINK-COSIMO to measure GHGs emissions related to the agricultural sector. In the scenario analysis the assumptions of IEA 2DS regarding crude oil prices, the macroeconomic environment, taxes applied to fuels according to their GHGs profiles, and future demand of gasoline-type and diesel-type fuels are inputted into AGLINK-COSIMO. The resulting emissions are compared to those projected by the IEA and the baseline presented in the *Agricultural Outlook 2018-2027*, with the latter projections extended to 2030.

Assessing the potential contribution of biofuels in the decarbonisation of the transport sector: Scenario definition

The IEA perspective

The *Energy Technology Perspective* (ETP) (IEA, 2017[1]) and the *Technology Roadmap* (IEA, 2017[9]) focus on the opportunities and challenges of scaling up and accelerating the deployment of clean energy technologies in different sectors. The IEA 2DS sets a path for the energy sector at the 2060 horizon that is consistent with a 50% chance of limiting future global average temperature increases to 2°C by 2100. In the 2DS, carbon taxes increase over time, which partly offsets lower fossil fuel prices occurring due to lower demand.[12]

The 2DS assumes progressive improvements in the following areas: vehicle technical efficiencies, "avoid-shift measures" for passenger cars,[13] systemic and logistic efficiency gains in road-freight, and electrification. Both IEA reports see an important role for bioenergy. They emphasise that the future role of bioenergy will need to be contingent on unambiguous and significant carbon savings (and hence rely on a rapid transition to advanced biofuels), and will need to be consistent with improvements in environmental and social sustainability.

In the 2DS, 17% of the energy consumed in 2060 is derived from bioenergy compared to 4.5% in 2015 and bioenergy is responsible for 17% of the cumulative reductions in emissions to 2060. In the transport sector, fossil fuel consumption is sharply reduced and bioenergy would provide 29% of the total transport final energy demand by 2060.

Towards 2030, the IEA sees gasoline demand retracting more than diesel demand; gasoline is mostly used by passenger road vehicles and thus more easily offset.

By 2030, ethanol and biodiesel use in the 2DS is projected to be respectively 40% and 110% higher than in the OECD-FAO baseline projections. The use of conventional biodiesel derived from vegetable oil is set to be phased out in favour of waste-based biofuels for the diesel pool which offer stronger GHGs emission saving. For ethanol, conventional ethanol mainly derived from sugarcane (that has a stronger GHGs reduction profile than ethanol produced from starch feedstocks) is produced at the expense of other food crops. Advanced ethanol, based on agricultural residues or energy crops, is expected to become widely available as of 2025.

Definition of an AGLINK-COSIMO scenario

The IEA 2DS is based on simulations undertaken with the MoMo model,[14] a simulation model that uses detailed projections of transport activity and vehicle activity, energy demand, and WTW GHGs and pollutant emissions to 2060 under alternative policy scenarios.

The IEA sees a role for biofuels in transport sector mitigation within a 2060 horizon. As part of its 2DS, it describes a ten-year transition period, where the use of currently available biofuels would increase before it is replaced by more sustainable biofuels with lower carbon emission profiles. However, the IEA did not take into account the interconnection between agricultural and biofuel markets, and the ability of agricultural activities to supply the amount of bioenergy foreseen in its 2DS to meet climate change targets.

The present AGLINK-COSIMO scenario attempts to fill that need and is a further step in an enhanced collaboration between the OECD and IEA to better capture the potential role of biofuels in climate-change mitigation.[15]

AGLINK-COSIMO has a medium-term horizon with projections until 2030 and a detailed production, use and trade modelling framework (presented in Annex 5.B) for most categories of biofuels currently available on the market with a direct connection to agricultural markets. It is capable of taking into account alternative assumptions than those used to produce the *Agricultural Outlook 2018-2027* baseline.

In particular, the AGLINK-COSIMO 2-degree scenario (AC-2DS) is defined using assumptions consistent with the IEA 2DS assumptions. Assumptions are summarised in Table 5.1. They differ from the *Outlook* baseline assumptions regarding the future evolution of crude oil prices, the macroeconomic environment, carbon taxes applied to fuels according to their GHGs profiles, as well as the future demand of gasoline-type and diesel-type fuels.

A key difference between the AC-2DS and the *2018 Agricultural Outlook* assumptions is that the former projects lower transportation fuel demand. This is crucial given that the AGLINK-COSIMO models biofuel demand as a share of transportation fuel demand. This share is defined as the maximum value between a market-driven share and a mandate-driven share. The market-driven share reacts to the price difference between the biofuel and the conventional transportation fuel it replaces. When the relative consumer price of conventional transportation fuel increases compared to that of biofuel, i.e. when biofuel becomes more competitive, the market-driven share increases. Under AC-2DS, carbon taxes are applied according to the WTW GHGs profiles of the different fuels and encourage or discourage the use of specific biofuels in the transportation mix.[16]

An additional assumption is made to increase the ethanol blend wall to 15% across all countries to allow additional ethanol to be blended with gasoline[17] within the framework of the climate change mitigation scenario, similar to what is included in the IEA 2DS.

To date ethanol production from agricultural and forest residues and specific energy crops is not included endogenously in the AGLINK-COSIMO modelling framework. This is due to the fact that current production levels are low as there is a limited number of commercial scale plants (IEA, 2017[9]) and the fact that information on production costs is not widely available. The AC-2DS scenario setup therefore shows the impact of a mitigation scenario in the transport sector where the availability of biofuels based on ligno-cellulosic material remains limited at the level expected in the 2018 *OECD-FAO* Agricultural *Outlook*. In future analysis, additional assumptions concerning the development of advanced ethanol production costs could be included.

Table 5.2. AC-2DS main assumptions

		AC-2DS	AC-2DS	AC-2DS	Baseline	% difference of scenarios vs baseline
		2020	2025	2030	2030	2030
Crude oil prices	USD/barrel	87.6	111.9	140.1	79.7	76%
Additional carbon taxes applied to fuels						
Expressed in terms of carbon tax equivalent						
Gasoline-type fuels	USD/tCO_2eq	24.4	54.6	91.7	..	..
Diesel-type fuels	USD/tCO_2eq	18.5	41.5	69.8	..	..
Expressed as a WTW emission based tax						
Gasoline	USD/hl	6.5	14.6	24.5	..	..
Diesel	USD/hl	6.6	14.8	24.9	..	..
Sugarcane based ethanol	USD/hl	1.3	2.9	4.9	..	..
Maize based ethanol	USD/hl	3.3	7.5	12.5	..	..
Agriculture residues-based ethanol	USD/hl	0.7	1.5	2.5	..	..
Palm oil based biodiesel	USD/hl	4.0	8.9	14.9	..	..
Soybean oil based biodiesel	USD/hl	3.8	8.5	14.2	..	..
Rapeseed oil based biodiesel	USD/hl	3.4	7.5	12.7	..	..
Waste oil based biodiesel	USD/hl	1.2	2.6	4.3	..	..
Demand for transportation fuels in key countries						
Gasoline-type fuels						
World	Bln l	1 268	1 140	998	1 318	-24%
United States	Bln l	542	455	367	454	-19%
European Union	Bln l	117	94	76	103	-26%
Brazil	Bln l	47	47	45	51	-11%
China	Bln l	173	176	164	229	-28%
India	Bln l	37	42	48	87	-45%
Diesel-type fuels						
World	Bln l	996	1 021	1 024	1 047	-2%
United States	Bln l	217	204	186	199	-7%
European Union	Bln l	229	209	186	204	-9%
Brazil	Bln l	52	53	54	56	-3%
China	Bln l	125	132	134	126	6%
Indonesia	Bln l	38	46	52	52	0%
Argentina	Bln l	13	14	14	16	-13%

Note: It is assumed that WTW emission factors associated with biofuels are constant over the period leading to 2030. If technologies associated with conventional biofuels were to change over the medium-tem – due to new conversion processes, better use of co-products, technical innovation – this could well modify downward the WTW emission factors and change the level of carbon taxes assumed in the scenario

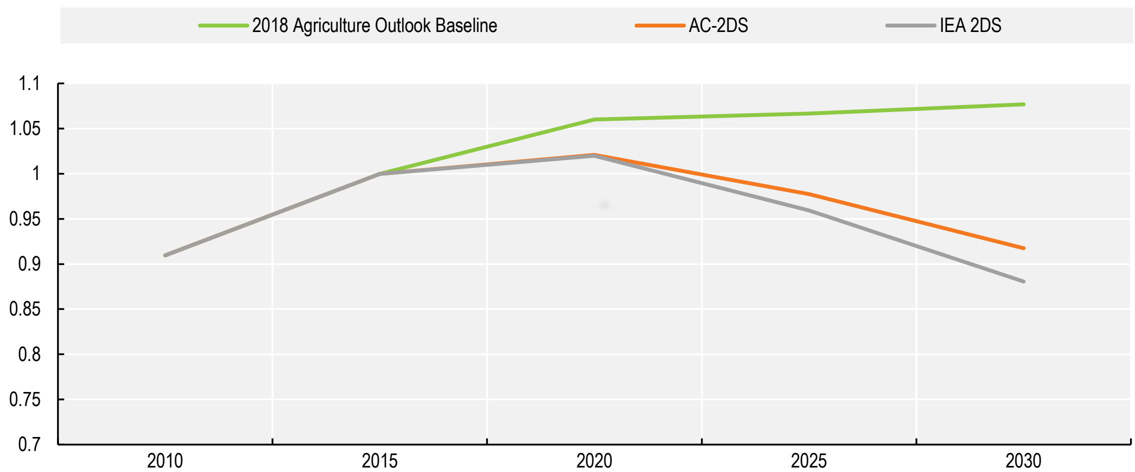
Scenario results

All scenario results, unless otherwise specified, are compared to the 2018 *OECD-FAO Agricultural Outlook* baseline projections, henceforth referred to as the "baseline".

Biofuel markets

Under AC-2DS, global WTW GHGs emissions in the transport sector are 15% lower by 2030 than under the baseline (Figure 5.3). The most important factor behind this decrease is the reduction in transportation fuel use due to "avoid and shift measures" and vehicle efficiency gains, while biofuels contribute only marginally.[18] Figure 5.3 shows a divergence between the evolution of WTW emissions in the transport sector in the IEA 2DS and in the AC-2DS. This is related to a more limited response of biofuel markets to the policy stimuli in the AC-2DS as compared to IEA 2DS. This is explained in detail below.

Figure 5.3. Evolution of WTW emissions in the transport sector in AC-2DS compared to the baseline and to IEA 2DS



Note: WTW emissions in the transport sector are indexed at 1 in 2015.
Source: OECD for the baseline and AC-2DS, IEA for 2DS, https://www.iea.org/topics/transport/.

Whereas in 2030 gasoline and diesel use are projected to be considerably lower under the AC-2DS compared to the baseline, ethanol fuel and biodiesel use would be 0.9% and 1.2% stronger, respectively, under the AC-2DS than under the baseline. Ethanol-related savings of WTW GHG emissions of gasoline-type fuels would reach 3.1% by 2030 (versus 2.3% in the baseline) and biodiesel related savings in diesel-type fuels would reached 1.8% (versus 1.7% in the baseline) (Figure 5.4).

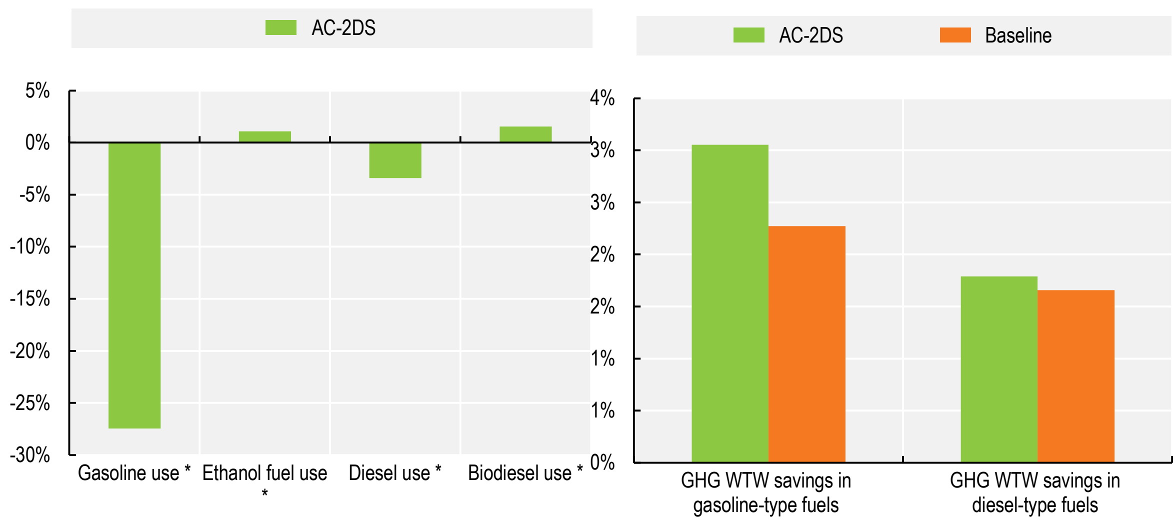
Expressed in terms of the blending of biofuels in conventional fuels, this means that the volume share of ethanol in gasoline-type fuels at the global level would reach 11% by 2030 in the AC-2DS (versus 8.2% in the baseline) and that the volume share of biodiesel in diesel-type fuels would reach 4% in the AC-2DS (versus 3.8% in the baseline).

Figure 5.5 compares biofuel blending shares between the baseline, the AC-2DS and the IEA 2DS. The development of biofuel blending in transportation fuels is less pronounced in the AC-2DS. The IEA 2DS expects a strong development of sugar cane based ethanol production in the period leading up to 2025 (+130% at the expense of maize based ethanol production) and then an uptake of advanced ethanol production. For biodiesel, the IEA 2DS sees an important increase of vegetable oil based biodiesel in the period leading to 2025 (+39%) and then a take-off of waste-oil and animal-fat based biodiesel and other types of biofuels used for the diesel pool (such as synthetic fuels or animal fats). The AC-2DS foresees much lower biofuel use growth over the period leading to 2030.

Under IEA 2DS and AC-2DS, biofuel use is promoted by taxes applied to fuels according to their GHGs emission profiles. However, in AC-2DS, the production of conventional biofuels (such as sugarcane-based ethanol or vegetable-oil based biodiesel) is constrained by the availability of agricultural feedstock and the potential of agricultural markets to supply more feedstock for biofuels while meeting demand for food and feed. A doubling of sugarcane-based ethanol use as forecast in IEA-2DS would produce a very strong shock on sugar markets and most probably important land use impacts.[19]

Figure 5.4. Biofuel use and GHGWTW savings, 2030

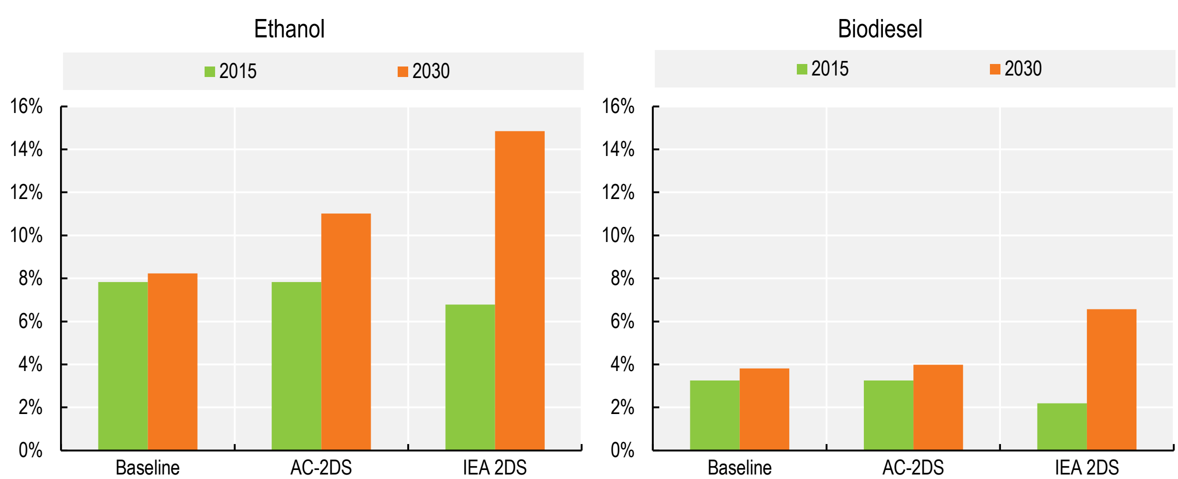
AC-2DS main results



* Percentage change with respect to the baseline.
Source: OECD, based on Aglink-Cosimo simulations.

Figure 5.5. Comparison of biofuel blending shares in volume, 2015 and 2030

AC-2DS, baseline and IEA 2DS

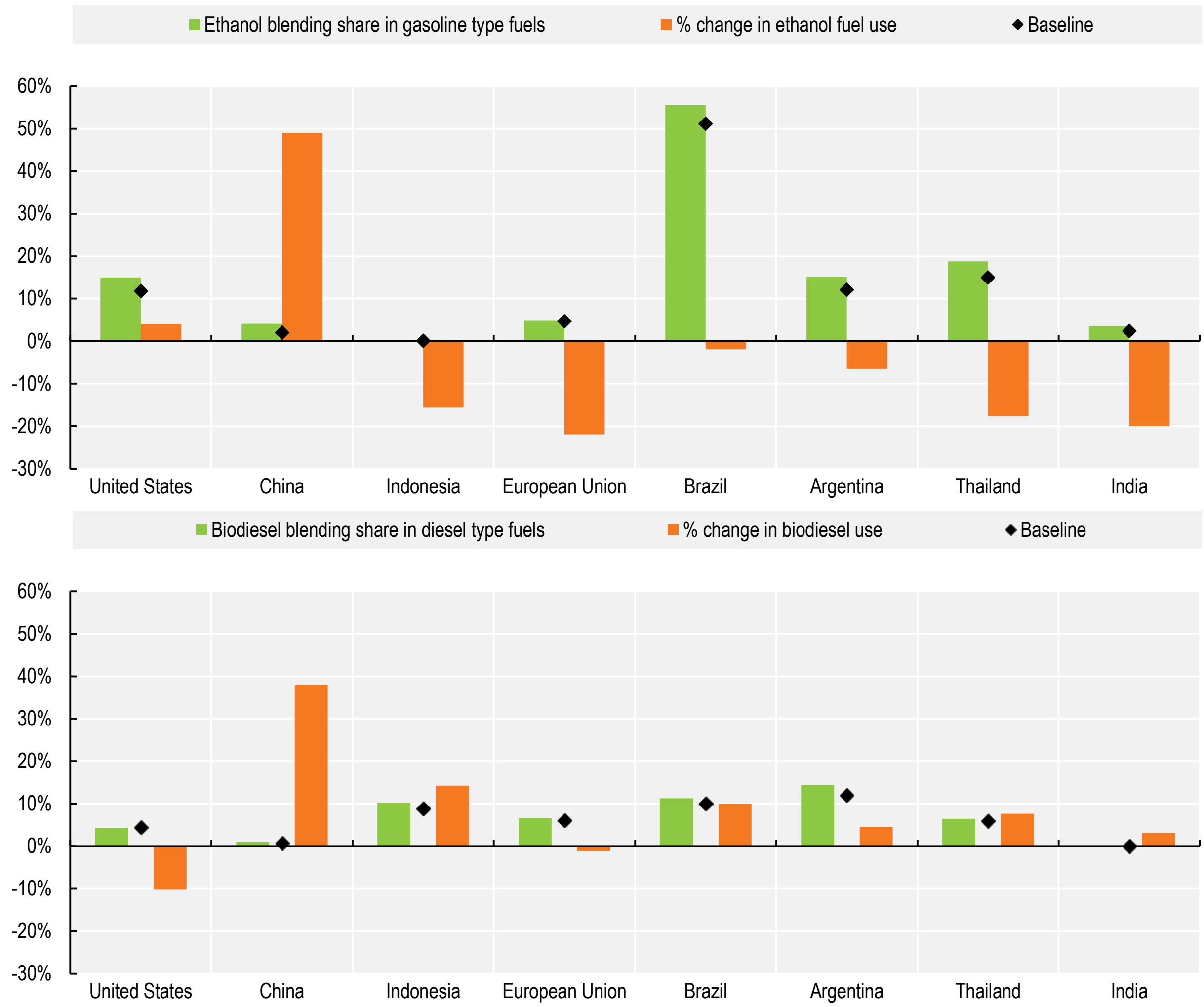


Source: OECD, based on AGLINK-COSIMO simulations and IEA (2017).

Figure 5.6 provides an overview of major AC-2DS results in terms of biofuel blending shares and biofuel use at the country level. In all countries, the ethanol and biodiesel blending shares are higher under AC-2DS than under the baseline, as the taxes applied to the different fuels according to their WTW GHGs profiles and the assumed developments of crude oil prices decrease the price ratio between biofuels and conventional fuels, thus encouraging the market-driven use of biofuels.[20]

Figure 5.6. Changes in biofuel blending share and fuel use by 2030 for major countries

AC-2DS compared to baseline projections



Note: The percentage change in biofuel fuel use is calculated as the change between the AC-2DS results and the baseline value.
Source: OECD based on Aglink-Cosimo simulations.

Biofuel use does not exceed mandates in the United States; those mandates, expressed in volume terms, are kept at the same volumes as the baseline case (with lower gasoline and diesel use). The assumption of a blend wall gradually increasing to 15% implies that some of the advanced mandate can be met with sugarcane-based ethanol.

In the absence of strong nation-wide ethanol mandate, the market-driven effect is particularly strong in the People's Republic of China (hereafter "China") where the ethanol share in gasoline-type fuels doubles to 4.1% by 2030.[21] Further increases in ethanol blending is constrained by this country's ethanol production capacity and the strong domestic demand in major ethanol-producing countries. In Brazil, the use of hydrous ethanol (pure ethanol that can be used by flex-fuel cars) strongly increases in response to the carbon tax stimulus.

The shares of ethanol produced from maize and sugarcane remains relatively stable when compared to the baseline at 51% and 30% respectively. As described above, the current model does not allow the take-off of advanced ethanol based on agricultural residues or energy crops production in the medium term. The picture differs for biodiesel where waste oil-based biodiesel production is 24% stronger than in the baseline, at the expense of vegetable oil based biodiesel.

There continues to be little trade of biofuels compared to global production levels as the biofuel policies in place[22] and the taxes applied to fuels based on WTW emissions mostly encourage the consumption of domestically produced biofuels. However, the trade share of biofuels with lower GHGs emission profiles in total biofuel trade increases strongly (+25% for sugarcane based ethanol and +75% for waste oil and animal fat-based biodiesel when compared to the baseline).

Agricultural markets

The OECD-FAO *Agricultural Outlook 2018-2027* (OECD/FAO, 2018[2]) reports an alternative scenario to the baseline where crude oil prices and macro-economic assumptions would follow a similar path to what was included in the AC-2DS (see Chapter 1). It highlights that higher oil prices increase agricultural production costs through higher prices for fuel and fertiliser, as well as through general cost increases induced by higher inflation. They can also affect demand for agricultural commodities through biofuels markets. This is also the case in the AC-2DS.

In addition, in the AC-2DS, global demand for agricultural commodities used as biofuel feedstock is affected by the taxes applied to the different fuels according to their WTW GHGs profiles and by the different crude oil price and macroeconomic assumptions. Global maize and sugar cane use for biofuels are supposed to increase by 1% and 0.3%, respectively, by 2030 when compared to the baseline while the demand for vegetable oil to be used for biofuels would be 4% lower than in the baseline (due to the development of waste-based biodiesel).

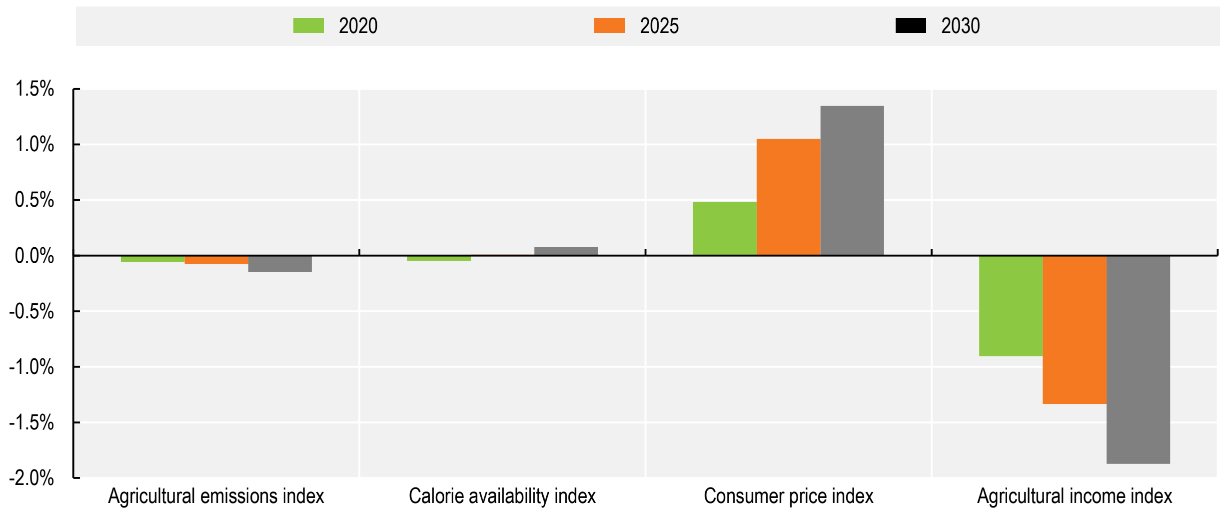
The overall effects on emissions and food security indicators are presented in Figure 5.7, using the same three indicators as in Chapter 3.

- The Calorie Availability Index represents the average amount of calories available per capita in each country for the subset of the food basket represented in the model.
- The Consumer Food Price Index is calculated as a fixed weight index of the national consumer prices in real terms. The food consumption quantities of 2015 are used as weights. Higher consumer prices are assumed to lead to lower access to food for parts of the population.
- The Agricultural Income Index is calculated as a fixed weight index of a combination of producer prices, subsidies and a cost index. As weights, the production quantities in 2015 are used. This indicator can be used in countries where the agricultural sector is a large contributor to the GDP.

The AC-2DS would imply by 2030 a stronger consumer food price index by about 1.4% when compared to the baseline, while the calorie availability index would remain stable. The agricultural income index would decrease by 1.9%, reflecting higher production costs. Agricultural-related emissions would decrease by 0.15%.

Overall effects of the AC-2DS on agricultural markets are relatively moderate as the increase in biofuel use when expressed, as a share of conventional fuel use does not lead to a strong increase in demand for agricultural feedstock. This is due to the lower demand for gasoline and diesel in the medium term.

Figure 5.7. Impact of the AC-2DS scenario on agricultural markets



Source: OECD based on AGLINK-COSIMO simulations.

Summary of main findings

The main results of the AC-2DS are as follows.

- WTW GHGs emissions in the transport sector would be 16% lower by 2030 than compared to the baseline (Figure 5.3). This is mostly related to the assumption of slower growth of gasoline and diesel use up to 2030.
- The volume share of ethanol in gasoline-type fuels would reach 11% by 2030 and that of biodiesel in diesel-type fuels would reach 4% (versus 8.1% and 3.8% respectively in the *Outlook* baseline) (Figure 5.5).
- The assumptions regarding the future evolution of crude oil prices coupled with the taxes applied to the different fuels according to their WTW GHGs profiles decrease the price ratio between biofuels and conventional fuels. Market-driven biofuel use is thus encouraged and mandates are not binding in most countries, as is the case in the baseline (Figure 5.6).
- Waste oil-based biodiesel production is set to develop strongly (+24%) by 2030 when compared to the baseline due to its lower WTW GHGs profile as opposed to vegetable oil-based biodiesel.
- The trade share of biofuels with lower GHGs emission profiles, such as sugarcane-based ethanol and waste oil-based biodiesel, in total biofuel trade will increase by 25% and 75%, respectively, by 2030 when compared to the baseline.
- The AC-2DS foresees only moderate increases in the volumes of agricultural feedstock used to produce biofuels despite stronger blending shares of biofuels in volume terms.
- Developments in terms of volume shares are lower than those foreseen in the IEA 2DS. In the AC-2DS, the production of biofuels based on agricultural feedstock (such as sugarcane-based ethanol or vegetable oil-based biodiesel) is constrained by the availability of feedstock and rising agricultural production costs.
- The impact on agricultural markets will be relatively small with a stronger food consumer price index of about 1.4% and a lower agricultural income index by about 1.9% when compared to the baseline. The higher oil prices assumed in the AC-2DS increase agricultural production costs (Figure 5.7).

- The current AGLINK-COSIMO model cannot analyse the effects in terms of global land use, but this should be possible in future work.

Conclusions

This chapter presents the results of a quantitative analysis based on the AGLINK-COSIMO model of a climate change mitigation scenario for the transport sector, namely the AC-2DS; the decarbonisation of the transport sector being an often-stated argument behind biofuel policies.[23] The analysis shows that the role of biofuels in the transport sector on climate change mitigation depends, in part, on the ability of the agricultural sector to provide in the medium-term agricultural feedstock to produce biofuels and on the set of policy incentives.

Gasoline substitutes classified as “advanced” according to the IEA definition (Boxes 5.1 and 5.2) are likely to be increasingly produced on an industrial scale. To date, however, the biofuel with the lowest WTW GHGs emission profile able to replace gasoline is sugarcane-based ethanol. The same potential is not seen for the further development in global use of sugarcane-based ethanol in the medium term, as opposed to the IEA. This is due to the position of Brazil as a major supplier of sugar, its own strong use of ethanol, as well as constraints related to the expansion of sugarcane production in the country. In addition, the AC-2DS carbon tax differential between sugarcane-based ethanol and maize-based ethanol is not high enough to promote a massive shift towards the use of sugarcane-based ethanol in the United States, a strong user of ethanol.

On the biodiesel side, waste-based biodiesel offers important WTW GHGs savings when compared to vegetable oil-based biodiesel. The use of waste oil-and animal fat based biodiesel is widespread in the United States and the European Union, constrained mostly by the ability to collect and recycle vegetable oil and animal fats. Production costs are similar to those of vegetable oil-based biodiesel and the policy incentives implemented in AC-2DS lead to an expansion of its production by almost 30%. Further growth is possible, but it could be supported more widely across the globe with other policy incentives, such as targets for the recycling industry and the implementation of traceability measures.

Simulations using the AGLINK-COSIMO model suggest the policy incentives described in the IEA 2DS (mostly carbon taxes according to WTW emissions) may not be sufficient to elicit the expected response in terms of the production of biofuels. Given such policies and constraints on feedstock supplies, biofuels derived from food and feed feedstock have no more than a minor role to play in delivering climate change mitigation from the agricultural sector. A substantially increased role of biofuels in the decarbonisation of the transport sector would require a different set of policy incentives that would need to be cost-effective and take into account effects on food security and the sustainable use of resources.

At present, the AGLINK-COSIMO model cannot evaluate GHGs emissions associated with LUC in the agricultural sector, but this should be possible soon. This would allow for a new scenario where carbon taxes would be applied to the different fuels according to their total emissions (WTW + LUC).

Notes

[1] The main features of the AGLINK-COSIMO model are presented in Annex 5.B.

[2] Over the coming decade, the level of food- and feed-based biofuel use in the IEA 2DS is related to their comparative costs (including carbon taxes) compared to the cost of conventional fuels.

[3] This chapter does not deal with climate change mitigation in the agricultural sector and does not assume any change in agricultural production costs related to climate change in the medium term compared to the baseline scenario presented in the 2018 *Agricultural Outlook*.

[4] This analysis and more generally all previous analysis undertaken by OECD have focused on liquid fuels produced from biomass (OECD/FAO, 2018[2]), (OECD, 2008[12])). Biogas – gaseous fuels produced from biomass – are not included in the presented analysis principally due to the lack of information on their current development; however, their potential for transport vehicles is clear (Renewable Energy Agency, 2018[13]).

[5] Other types of biofuels exist on the market such as jet fuels, a drop-in fuel with the same characteristic as kerosene.

[6] Plant growth may lead to GHG emissions despite GHG capture by the plant itself due, for example, to the use of fertilisers.

[7] The development of Carbon Capture and Storage (CCS) or Carbon Capture and Utilisation (CCU) could be associated to biofuel production processes to lower CO_2 emissions. CCS and CCU are presently at the prototype or demonstration stages (IEA, 2017[9]).

[8] The Fifth Assessment Report of the IPCC defines an emission factor as "the emissions released per unit of activity" (Allwood et al., 2014[10]). The units of activity considered in this study correspond to the consumption of a certain amount of biofuel, expressed in volume or in energy content.

[9] Based on the literature, it is calculated to be 2674 gCO_2/l for gasoline and 3208 gCO_2/l for diesel.

[10] For example, the conversion of existing cropland to the cultivation of biofuel feedstocks could lead to the expansion of cropland on natural land elsewhere for food production.

[11] This component covers GHGs emissions from the following activities: enteric fermentation (linked to ruminant production systems), manure management, rice cultivation, synthetic fertilisers, manure applied

to soils, and manure left on pasture. It does not include emissions arising from the burning of crop residues and savannah, from the cultivation of organic soils, nor from crop residues.

[12] Carbon taxes applied to transportation fuels according to their GHG profiles are in use in several countries. Nevertheless, most of the taxes applied to transportation fuels take the form of excise taxes, with often the lower rates applied to biofuels (OECD, 2018[14]).

[13] "Avoid-shift" measures for passenger cars correspond to measures that promote a decrease in the demand for passenger-car transport and the length of trips, and transport modes that induce fewer emissions.

[14] https://www.iea.org/etp/etpmodel/transport/.

[15] In 2017, the OECD collaborated with the IEA to review biofuel production costs to be implemented in their respective models (AGLINK-COSIMO and MoMo models).

[16] The scenario would yield different results if taxes were applied according to a GHG profile that included LUC-associated emissions.

[17] This 15% constraint applies to all countries except Brazil, Paraguay and Thailand, where the use of high-blend ethanol has been developed.

[18] In 2030, biofuels use in AC-2DS are behind about one eighth of the WTW GHGs emissions decrease when compared to the baseline. This share is calculated by comparing the WTW GHGs emissions in AC-2DS and in a scenario equivalent in terms of energy demand for the transport sector where only conventional fossil fuels would be used with baseline WTW GHGs emissions.

[19] The impact of a shock of this magnitude has not been evaluated with the AGLINK-COSIMO model. The AC-2DS foresees developments on the Brazilian ethanol market that are in line with the RenovaBio program announcements (a fuel ethanol share of 55% by 2030).

[20] The market-driven use of biofuel develops when the relative consumer price of conventional transportation fuel increases more than that of biofuel. It occurs either through an increase in high-blend biofuel use or an increase in the share of biofuels being blended in the transportation fuel mix.

[21] The baseline does not take into account the 2017 Chinese announcement concerning an E10 mandate (see the biofuel chapter of the *OECD-FAO Agricultural Outlook* 2018-2027).

[22] The AC-2DS assumes a continuation of biofuel policies that are currently in place.

[23] There are different types of policy motivations behind the implementation of biofuel policies including climate change mitigation, energy security, rural development, and agricultural market support.

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Annex 5.A. Literature review on WTW emissions

Annex Table 5.A.1. Summary of the literature review

	CARB, 2009	EU, RED 2009	JEC, 2014	Havlik, 2010	Koga et al, 2010	Nguyen et al, 2007	Hoefnagels et al, 2010
Presentation and context							
Title	California's Low Carbon Fuel Standard - Final Statement Of Reasons		WTT report, version 4a and Appendix 4	Global land-use implications of first and second generation biofuel targets	Assessing energy efficiencies and greenhouse gas emissions under bioethanol-oriented paddy rice production in northern Japan	Energy balance and GHG-abatement cost of cassava utilization for fuel ethanol in Thailand	GHG footprint biofuels
Publication date	December 2009	2009	2014	2010	2010	2007	2010
Links	CARB, California's LCFS Final Statement Of Reasons, 2009.pdf	http://eur-lex.europa.eu/eli/dir/2009/28/oj	http://iet.jrc.ec.europa.eu/about-jec/sites/iet.jrc.ec.europa.eu.about-jec/files/documents/report_2014/wtt_report_v4a.pdf ; http://iet.jrc.ec.europa.eu/about-jec/sites/iet.jrc.ec.europa.eu.about-jec/files/documents/report_2013/wtt_appendix_4_v4_july_2013_final.pdf	http://users.auth.gr/~akontses/%CE%A4%CE%A0%CE%A0_biofuels/Global%20land-use%20implications%20of%20first%20and%20second%20generation%20biofuel%20targets.pdf	http://www.pubfacts.com/detail/21126818/Assessing-energy-efficiencies-and-greenhouse-gas-emissions-under-bioethanol-oriented-paddy-rice-prod		
Focus/scope		Default values for biofuels and bioliquids. Total for cultivation, processing, transport and distribution. "el" (annualised emissions from carbon stock changes caused by land-use	EU crops (except Soya and palm oil which are imported)	Lifecycle GHG savings from substitution of fuels by biofuels, without LUC related emissions.	Ethanol production from rice (Japan)	Ethanol production from cassava (Thailand)	GHG emissions from biofuel production, No LUC. Allocation of co-products by EU default (energy allocation for co-products, subtraction for co-generation of electricity and heat), energy, mass

	CARB, 2009	EU, RED 2009	JEC, 2014	Havlik, 2010	Koga et al, 2010	Nguyen et al, 2007	Hoefnagels et al, 2010
		change) are excluded. /!\ these are "default" values", whereas "actual" values must be used (actual values include notably emissions from land use)					and market value. JRC DNDC model for N2O emissions from sugar cane, wheat, sugar beet, maize and rapeseed. IPCC model for miscanthus, palm fruit, soy beans, switchgrass, eucalyptus and jatropha.
Remarks	California averages			Reference: - for ET from woody biomass: EF of gasoline = 85.9 - for other biofuels: EF of gasoline = 85, diesel = 86			- Includes co-products. To account for co-products, both allocation (by market value) and system expansion approach are applied. Results with allocation by energy content of mass are also available. - Includes emissions from transport to the EU
		Annex V			Figure 2 and Table 1	Table 4	Figure 2 & Appendix B

Source: OECD literature review.

Annex Table 5.A.2. Comparison of WTW emission factors

	CARB, 2009	EU, RED (2009)	JEC, 2014	Havlik, 2010	Koga et al, 2010	Nguyen et al, 2007	Hoefnagels et al, 2010	Average
WTW emissions (gCO_2eq/MJ)								
Conventional ethanol								
Wheat ethanol		70	69.4					69.7
Maize ethanol	65.66		80.3	49.42				65.1
Barley ethanol			76					76
Rye ethanol			76					76
Sugar beet ethanol		40	40.3					40.2
Sugar cane ethanol	27.4	24	24.8 (excess bagasse used for electricity production)	25.01				25.3
Sweet sorghum ethanol								
Rice ethanol					66.3			66.3
Cassava ethanol						45.9		45.9
Conventional biodiesel								
Sunflower oil biodiesel		41	45.9					43.5
Palm oil biodiesel		68	50.8				52.5	59.4
Rapeseed oil biodiesel		52	53.9	44.82			45.7	50.2
Soybean oil biodiesel		58	55.1	47.21			52.4 (Brazil), 59.2 (US)	54
Canola oil biodiesel							51	51
Jatropha oil biodiesel							43	43
Biodiesel from waste vegetable oil (UCO)		14	13.8					13.9
Biodiesel from animal fats		14	26.3					20.2
HVO								
HVO from rapeseed		44	50.2					47.1
HVO from sunflower		32	44.8					38.4
HVO from palm oil		62	48.6					55.3
HVO from soybean			55.1 (imported soy to the EU)					55.1
HVO from UCO			8.1					8.1
HVO from animal fat			24.5					24.5
ETBE								
The part from renewable sources of ETBE		Equal to that of the ethanol production pathway used						
Advanced biofuels								
Cereal straw ethanol			9.2					9.2
Wheat straw ethanol		13						13
Woody biomass ethanol				22.8				22.8
Switchgrass ethanol (herbaceous)							24	24
Miscanthus ethanol (herbaceous)							17.8	17.8
Eucalyptus ethanol (woody)							4.9	4.9
Farmed wood ethanol		25	22.8					23.9
Waste wood ethanol		22	19.5					20.8

	CARB, 2009	EU, RED (2009)	JEC, 2014	Havlik, 2010	Koga et al, 2010	Nguyen et al, 2007	Hoefnagels et al, 2010	Average
Switchgrass FT diesel (herbaceous)							13.9	13.9
Miscanthus FT diesel (herbaceous)							10.3	10.3
Eucalyptus FT diesel (woody)							7.7	7.7
SRP FT diesel								
Forest residues FT diesel								
Farmed wood FT diesel		6	7					6.5
Waste wood FT diesel		4						4
Paper & pulp industry waste FT diesel ("waste wood via black liquor")			2.5					2.5

Source: OECD literature review.

Annex 5.B. An overview of the AGLINK-COSIMO biofuel model

The biofuels component of the AGLINK-COSIMO model is a structural partial equilibrium economic model that analyses the world supply and demand of biofuels. The biofuels module, similar to other components of the AGLINK-COSIMO model, is recursive and dynamic. It simulates annual market balances and prices for the production, consumption and traded quantity of ethanol and biodiesel worldwide.

This biofuel model is completely integrated with the cereals, oilseeds and sugar components of the AGLINK-COSIMO model and produces the baseline presented in the annual *OECD-FAO Agricultural Outlook*. The production of biofuels drives the additional demand for agricultural commodities, in particular for coarse grains, vegetable oil, and sugar.

The AGLINK-COSIMO model has been adapted to explore the environmental impacts of biofuels use in the medium-term. An add-in module to the biofuel model was developed to assess well-to-wheels (WTW) emissions associated with biofuels whereas land-use-changes (LUC) emissions are assessed with the GHG add-in to the global AGLINK-COSIMO model. It is thus possible to compare total GHGs emissions associated with the baseline and different alternative scenarios.

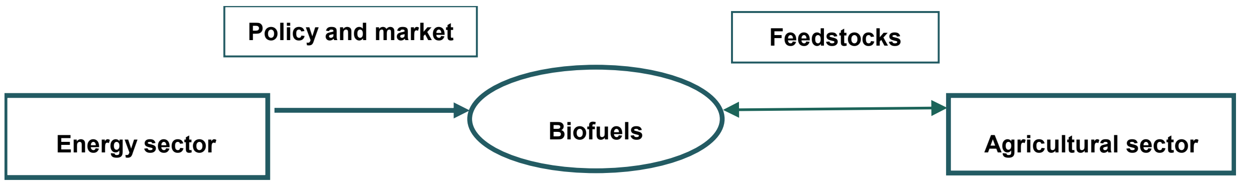
A major update of the AGLINK-COSIMO biofuels (BFL) module was undertaken between 2016 and 2018. In particular, it included:

- The full revision of the model with the introduction of a template for the following countries: Argentina, Australia, Brazil, Canada, China, European Union, Japan, Korea, Mexico, New Zealand, Norway, Russian Federation, Switzerland, United States, Colombia, Chile, India, Indonesia, Malaysia, Paraguay, Philippines and Thailand). Ethanol and biodiesel are modelled separately for each country. The new biofuels module also includes a separate demand and supply for ethanol and biodiesel for the Rest of the World (ROW).
- The introduction of separate fossil fuel (gasoline and diesel) demand equations for each of these countries.
- The modelling of high blend substitute use of ethanol to gasoline and biodiesel use to diesel.
- A revisit of the linkage between the BFL module and other component of the AGLINK-COSIMO model.
- The development of an add-in that allows to calculate WTW GHGs emission and savings derived directly from the use of the biofuels.
- The development of endogenous production functions for biodiesel based on used cooking oil and tallow. This implies to establish price linkages between used cooking oil and vegetable oil.

The documentation on the AGLINK-COSIMO biofuel model is available on www.agri-outlook.org. This annex provides an overview of the main features of the BFL module. In addition to generating baseline outlook reports, the BFL module is used to stimulate policies in the context of global climate change mitigation scenarios.

Significant improvement has been made to the supply side of the biofuels module. The most important was reinforcing the link between the BFL module and other components of the AGLINK-COSIMO model.

Annex Figure 5.B.1. Linkage between the energy, biofuels and agricultural



The production of biofuels (ethanol and biodiesel) derived from each type of feedstock ($FS^{BF}_{q,i,c,t}$) are modelled separately, using the following equation:

$$\log FS^{BF}_{q,i,c,t} = v_0 + v_1 {*} \log RM^{BF}_{q,i,c,t} + v_2 {*} \log RM^{BF}_{q,i,c,t-1} + v_3 {*} \log RM^{BF}_{q,i,c,t-2} + v_4 {*} \log RM^{BF}_{q,i,c,t-3} + v_5 {*} \log FS^{ET}_{q,i,c,t-1} + \varepsilon^{BF}_{q,i,c,t} \quad (4)$$

where

$RM^{BF}_{q,i,c,t}$ = the profit derived from utilising i^{th} feedstocks for the production of biofuels (ethanol or biodiesel) to be blend in corresponding q type of fossil fuel (petroleum or diesel) in country c and year t.

The $RM^{ET}_{q,i,c,t}$ is derived based on the following identity:

$$RM^{BF}_{q,i,c,t} = PP^{BF}_{q,i,c,t} + DP^{BF}_{q,i,c,t} + VL^{BF}_{q,i,c,t} \quad (5)$$

$PI^{BF}_{q,i,c,t}$

where

$PP^{BF}_{q,i,c,t}$ = the biofuels (ethanol or biodiesel) producer price in country c in year t

$DP^{BF}_{q,i,c,t}$ = direct government support for biofuels (ethanol or biodiesel) production tied with the use of feedstock i in country c and year t

$VL^{BF}_{q,i,c,t}$ = the value of by-products derived from the use of feedstock i in biofuels (ethanol or biodiesel) production in country c and year t

$PI^{BF}_{q,i,c,t}$ = production cost index associated with the use of feedstock i in the production of biofuels (ethanol or biodiesel) in country c and in year t

$\varepsilon^{BF}_{q,i,c,t}$ = the corresponding error term.

$PI^{BF}_{q,i,c,t}$ depends on the i^{th} feedstocks producers price ($PP^{i}_{c,t}$) as shown below:

$$PI^{BF}_{q,i,c,t} = f(PP^{i}_{c,t}) \quad (6)$$

The purpose of including separate production functions specific to feedstocks is to track changes in greenhouse gas emission, and the direct and indirect consequences of changes in land use due to the use of a variety of food and non-food-based feedstocks in the production of biofuels.

An additional add-in to the biofuel component was developed to calculate GHGs emissions associated with the consumption of biofuels. A review of WTW biofuel emission factors was undertaken in the scope of this project. These emission factors are applied to biofuel use to obtain an estimate of WTW emissions associated with biofuels.

To be able to make that calculation, however, it is necessary to assess the consumption volumes of biofuel by each feedstock types used to produce biofuels on a country basis. This can be done residually if biofuel imports and exports are allocated to the different types of biofuel feedstocks that were used to produce them; this information is known for production as described above. This allocation is made on the assumption that a country's biofuel export can be divided in exports produced from different feedstocks in the same proportion as its domestic production of biofuels. It is then possible to calculate world exports and also imports by feedstock.

For imports at the country level, the add-in assumes that import shares for major importers (i.e. the European Union and the United States for biodiesel and Canada, and Japan and the United States for ethanol) are fixed and use industry information or US GAINS report to quantify them. For less important countries, it is assumed that national biofuel imports can be split in the same proportion as the world biofuel imports minus the import from the three major importers. It is important to note that biofuel trade remains limited anyway in the baseline projection.

For a given country, WTW emissions associated with ethanol and biodiesel can then be quantified. It is possible to calculate the WTW GHGs emission savings induced by the replacement of conventional fuels by biofuels by comparing the total level of WTW emissions associated by the mix of conventional fuels and biofuel (for example the mix of gasoline and ethanol) in a given country with the emissions that would have occurred if only conventional fossil-based fuels (for the same total energy content) were used.

This can be modelled for a specific country in a given year as:

$$\overline{GHG\_WTW}_{BF} = \sum_f GHG\_WTW_{BF_f} \times V_{BF_f} \tag{7}$$

$$\overline{GHG}_{transfuel,sub} = GHG_{transfuel} \times V_{transfuel,sub} \tag{8}$$

$$\overline{GHG}_{transfuel,cons} = GHG_{transfuel} \times V_{transfuel,cons} \tag{9}$$

It is thus possible to derive the emissions savings expressed in percentage as:

$$WTW\ GHG\ savings = \frac{\overline{GHG\_WTW}_{BF} - \overline{GHG}_{transfuel,sub}}{\overline{GHG}_{transfuel,cons}} \tag{10}$$

Where

V_{BF_f} is the volume of biofuels (ethanol or biodiesel) consumed which was produced from feedstock of type *f*

$GHG_{transfuel}$ is the emission factor of transportation fuels (gasoline or diesel)

$V_{transfuel,sub}$ is the volume of transportation fuels (gasoline or diesel) substituted by biofuels (ethanol or biodiesel)

$$V_{transfuel,sub} = V_{BF} \times \varepsilon_{transfuel,BF} \tag{11}$$

$V_{transfuel,cons}$ is the volume of transportation fuels of gasoline- or diesel- type consumed

$\varepsilon_{transfuel,BF}$ is the energy content ratio between biofuels (ethanol or biodiesel) and transportation fuels (gasoline or diesel)

$$\varepsilon_{transfuel,BF} = \frac{[GJ/m^3 BF]}{[GJ/m^3 transfuel]}$$

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Enhancing Climate Change Mitigation through Agriculture

Agriculture, with its growing contribution to global greenhouse gas emissions and opportunities to mitigate emissions, can help close the gap between existing global mitigation efforts and those that are needed to keep global warming to between 1.5 °C and 2 °C by the end of the century. Global scale and farm scale analyses are used to evaluate both the effectiveness of different policy options to reduce agricultural emissions, and the impact on competitiveness, farm income, food security, and government finances. In order to contribute to global mitigation efforts, countries will need to design agricultural policy measures that can navigate these trade-offs within the context of their national policy priorities and objectives. As most countries have not yet implemented policies to reduce emissions from agriculture, the analyses provided here come at an opportune time to inform this policy development.

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