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Environmental Impacts Along Food Supply Chains: Methods, Findings, and Evidence Gaps

Koen Deconinck and Lucinda Toyama

Food systems exert major pressures on the environment. This paper reviews what is known and not known about environmental impacts along food supply chains, looking at the contribution of different stages of the supply chain, the impact of different products, heterogeneity among producers, and the role of international trade. This review shows that most environmental impacts in food supply chains occur through land use change or at the stage of agricultural production. Livestock (especially ruminant livestock) has a higher footprint than plant-based food. However, there is also important heterogeneity among producers, even within the same region. A significant share of total environmental impacts is "embodied" in international trade, although considerably less than half. In terms of evidence gaps, some impacts (e.g. biodiversity, soil carbon) have been less studied, and there are geographic and product blind spots. Moreover, existing evidence is not sufficiently granular. While important evidence gaps thus exist, the overall picture that emerges is one of a rapidly growing evidence base, which can inform innovative supply chain initiatives to reduce impacts.

This is one of four papers developing work on addressing evidence gaps on food systems in OECD countries (*OECD Food, Agriculture and Fisheries Papers 183 to 186*).

Key words: Life Cycle Assessment, input-output analysis, agricultural trade, global value chains, sustainability

JEL codes: Q51, Q56, Q17, Q27, Q37

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Key messages

- Food systems exert major pressures on the environment. For effective public and private action we need to understand environmental impacts along food supply chains.
- Most of the environmental impacts of food occur through land use change or in agricultural production.
- Livestock products typically have a greater environmental footprint than plant-based foods, and ruminant products typically have a greater footprint than other livestock products.
- However, there is large heterogeneity among producers: 25% of global production often accounts for more than half of total environmental impacts.
- A significant share of environmental impacts is "embodied" in international trade – but this share is considerably less than half.
- There are however important evidence gaps, as some impacts, products, and geographies have not been studied sufficiently.
- Yet, there is rapid progress in recent years, with new data sources and techniques rapidly increasing the evidence base

Executive summary

Food systems account for a significant share of global environmental pressures such as greenhouse gas emissions, water pollution, and biodiversity loss. Addressing these pressures will require action not only by agricultural producers, but also by other supply chain actors, consumers, and policy makers. Informing these decisions requires evidence on environmental impacts along food supply chains.

This paper reviews evidence from downscaling approaches, life cycle assessments (LCAs), and trade-based analyses. For each method, the paper first introduces the main methodological characteristics before discussing main findings, evidence gaps, as well as strengths and limitations.

The vast majority of environmental impacts occur either through land use change or at the agricultural production stage. Studies generally show a greater environmental footprint for livestock products than plant-based products, and a greater footprint for ruminant products than for other livestock products. However, there is a large heterogeneity among producers, with 25% of global production often responsible for more than half of all environmental impacts. The evidence also shows that the share of food systems' environmental impacts linked to international trade is significant, but considerably less than half.

Greenhouse gas emissions, land use, and freshwater use are generally the most studied impacts; by contrast, biodiversity and soil carbon have been much less studied. There are also major gaps in terms of coverage of products. This is reinforced by geographic blind spots, as there are notable gaps for LCAs conducted in Africa and Central Asia. Moreover, given the evidence of heterogeneity, it is clear that existing information sources are not yet sufficiently granular. While important evidence gaps exist, the overall picture that emerges is one of rapid progress in recent years. The availability of new data sources and techniques as well as the creative combination of existing data sources are rapidly increasing the evidence base.

The different methodological approaches reviewed here are complementary. The strength of the downscaling approach is that it provides relative magnitudes of environmental impacts (e.g. GHG

emissions) for different stages of food supply chains, which is useful in prioritising efforts. By construction, however, the approach cannot illuminate impacts of products or individual producers.

LCAs are an essential tool, in part because they can rely on an increasingly harmonised approach codified in standards and handbooks, complemented by widely used databases and models. On the other hand, LCAs take a "bottom-up" approach, often providing in-depth assessments of a single product only. To understand system-wide impacts thus requires a creative re-analysis and complementary approaches to answer questions which are harder to answer through LCA, such as to what extent imports of one country are linked to environmental impacts in another.

Several trade-based approaches map environmental impacts of food production through international trade to final consumption, with recent progress in modelling physical (as opposed to monetary) flows, and in shedding light on the structure of sub-national supply chains. Some studies explicitly model how consumers and producers adjust their behaviour in response to policy interventions and changing market conditions. These approaches make it possible to capture substitution and other feedback effects which would occur if policies are implemented on a large scale. However, this typically requires additional assumptions.

The coming years may see new developments combining the strengths of the different approaches outlined in this paper. Future studies could include a much broader range of environmental (and social) impacts and could use data obtained directly from producers. Governments can also play an important role in enabling better analysis, e.g. by publishing detailed customs data and subnational production statistics and by helping to harmonise methods.

1. Introduction

Food systems account for a significant share of global environmental pressures such as greenhouse gas (GHG) emissions, water pollution, and biodiversity loss (IPCC, 2019; IPBES, 2019; Poore and Nemecek, 2018). It is now widely recognised that addressing these environmental pressures will require action not only by agricultural producers, but also by other supply chain actors, consumers, and policy makers (Poore and Nemecek, 2018; Hodson et al., 2021). Informing these decisions requires evidence on environmental impacts along food supply chains, as well as on the effectiveness of various initiatives to improve these impacts.

This paper reviews what is known about environmental impacts along supply chains. A companion paper discusses evidence around the effectiveness of voluntary and mandatory supply chain initiatives to improve environmental impacts along food supply chains.[1]

A number of distinct but complementary methodological approaches can be used to assess environmental impacts along food supply chains, such as downscaling, life cycle assessments, and trade-based approaches. For each method, the paper first introduces the main methodological characteristics before discussing main findings, evidence gaps, as well as strengths and limitations.

While important evidence gaps exist, the overall picture that emerges is one of rapid progress in recent years. The availability of new data sources and techniques as well as the creative combination of existing data sources are rapidly increasing the evidence base.

The remainder of this introduction explains why evidence is needed on environmental impacts along supply chains, and previews a number of recurring themes of the paper: the role of international trade, the importance of taking into account the heterogeneity and dynamic nature of food supply chains, the different levels of detail needed for different decision-making contexts, and the role of interests and values.

1.1. Why do we need to know environmental impacts along supply chains?

Agri-food products pass through a number of stages during their life cycle, with different environmental impacts at each stage. A full accounting of the environmental effects of food systems should therefore consider impacts at each stage, including those indirectly caused by input use (e.g. GHG emissions related to energy used in food production); potential land use effects (e.g. when greater demand for a product contributes to deforestation); and the role of waste (including food loss and waste, as well as waste of, for example, packaging materials). Moreover, a full accounting should cover as many environmental impacts as possible (i.e.,not only GHG emissions but also eutrophication, acidification, biodiversity impacts, etc.).

Detailed evidence about environmental impacts along food supply chains is useful for several reasons:

- Such evidence can show which specific *stages* of food supply chains contribute a greater share of environmental impacts and should therefore be the focus of efforts to improve environmental performance.
- Detailed evidence could also show whether specific *products* on average contribute a greater share of environmental impacts, as well as the heterogeneity of results.

[1] The work in this paper forms part of broader OECD work assessing evidence gaps for food systems (Deconinck et al., 2021). This is one of four papers developing work on addressing evidence gaps on food systems in OECD countries (*OECD Food, Agriculture and Fisheries Papers 183 to 186*).

- If evidence is sufficiently granular, it becomes possible to identify differences in environmental impacts for specific *producers, technologies* or *production systems*. This information can help firms in benchmarking their own performance against competitors and can help them in adopting more sustainable production practices. It can also help governments develop more targeted policies.
- With sufficiently detailed information, governments might be able to directly incentivise firms for the environmental performance of their products.
- Moreover, transparency on environmental impacts along food supply chains allows consumers to make more informed decisions, and allows civil society stakeholders to hold firms accountable and monitor progress against commitments.

A further reason for understanding environmental impacts along food supply chains is that consumers are themselves increasingly demanding more information on this topic. At the same time, structural changes in food supply chains suggest that supply chain actors may be well placed to help improve environmental outcomes. As the companion paper on the effectiveness of supply chain initiatives shows, there is indeed growing pressure on firms to take responsibility for environmental impacts not only in their own operations, but along their entire supply chain (Deconinck and Hobeika, 2022).

1.2. The role of international trade

The past decades have seen strong growth in agro-food trade, and in particular in global agro-food value chains (OECD, 2020a; FAO, 2020; Barrett et al., 2020). Where food supply chains span multiple countries, decisions in one country may influence environmental impacts in other countries.

Since 1995, international trade in food and agriculture has more than doubled in volume terms, with emerging and developing economies accounting for one-third of total exports. About one-third of trade in agricultural and food products crosses borders at least twice, as primary products are exported for processing and then re-exported (FAO, 2020). The importance of trade differs depending on the type of agricultural commodity: it is particularly high for tropical commodities such as cocoa, coffee or tea (where almost all of production is exported), but considerably lower for most other agricultural commodities. Data from the *OECD-FAO Agricultural Outlook* (OECD/FAO, 2021) indicates that the share of global production traded is about 10% or less for rice, pork, and poultry, and less than 20% for maize and beef. Similarly, less than half of the global production of cotton, sugar, or soybeans is traded internationally. Most agricultural commodities have a low value-to-weight ratio, and perishability creates an additional challenge to long-distance transport. While trade plays an indispensable role in the global food system, an important share of food supply chains thus operate within a single country.

The relative role of international and domestic value chains matters for discussions on the environmental impact of food systems. Historically, environmental impacts have typically been measured and reported for the country where production occurs, not where final consumption takes place. However, due to international trade, countries may have a good *production-based* environmental performance but a worse *consumption-based* environmental performance if significant environmental impacts are "embodied" in imported products. Conversely, some countries' production-based environmental performance may be driven to an important extent by consumers abroad.

This distinction matters, as it determines which instruments are most effective to improve environmental performance. Policies focusing on domestic production practices will be insufficient if a country's environmental impacts are mostly linked to imported products; conversely, efforts to inform more sustainable consumption choices will have a limited impact if most demand originates abroad. Where international trade plays a major role, environmental performance in the exporting country could be improved not only through policy initiatives in that country, but potentially also through interventions in importing countries, e.g. through greater consumer awareness, sustainability labelling, greater due diligence by importing firms, etc.; some of these initiatives are reviewed in the companion paper (Deconinck and Hobeika, 2022).[2] The structure of food supply chains and the importance of international trade thus influence the relevant policy instruments.

1.3. The heterogeneity and dynamism of food supply chains

As the evidence reviewed in this paper shows, producers around the world (and even within a single country) differ strongly in their environmental performance. As products may be re-combined in complex production processes and re-exported, this underscores the importance of understanding trade flows and the environmental performance of producers at different stages in these processes. Food supply chains are also dynamic, however, as firms are constantly in search of better economic opportunities offered by new suppliers or customers, including in response to policy changes or changing market conditions.

The heterogeneity and dynamism of food supply chains have important implications for the interpretation of evidence on environmental impacts along food supply chains. While averages and aggregates may be useful for some purposes, they might not be representative of specific products or producers. A targeted approach (focusing on the highest-impact products and producers on the basis of granular data) or an approach based on measuring actual impacts (rather than relying on historical averages) might be effective, provided that the relevant information can be obtained in a reliable and low-cost way.

At the same time, historical data might underestimate the potential for actors along the food supply chain to improve their environmental impacts, or might not capture system-wide effects as consumers and producers adapt their behaviour. One implication here is that *ex ante* assessments of the likely effects of policy interventions may need to take into account dynamic responses in food supply chains, e.g. through changes in international trade flows. Such approaches exist but, as shown in this paper, they are less common, in part because they require many additional assumptions.

1.4. Required levels of detail

Understanding environmental performance along food supply chains thus holds the promise of unlocking new possibilities for reducing the environmental damage of food systems. But as the discussion above suggests, different objectives may require different levels of detail. If the goal is to find out which stage of the supply chain deserves most attention, then it is sufficient to have information on environmental impacts segmented by supply chain stage, without necessarily being able to trace a product from one stage to the next. Comparing the environmental performance of products, by contrast, requires a life cycle assessment. The required level of detail also depends on the policy implementation it is supposed to support: if policy instruments (e.g. taxes, subsidies) are directly linked to environmental impacts, the evidence required will need to be especially accurate (Rajagopal et al., 2017; Wardenaar et al., 2012).

These different levels of detail broadly correspond to different analytical approaches which can be taken to understand environmental impacts along supply chains:

- Studies which provide ‘downscaled’ estimates of environmental impacts segmented by supply chain stage, without being able to trace specific products;
- Life cycle assessments of specific products, and studies synthesising the findings from such assessments (including recent developments around the “true cost” of food);
- Studies leveraging various trade data sources and models to track international trade flows and/or shed light on specific supply chains.

These approaches are discussed in sections 2, 3, and 4 respectively.

[2] For a review of the broader literature on trade and agricultural sustainability, see Baylis et al. (2021).

1.5. Interests and values

Policy debates on food systems are often characterised by interlocking discussions over facts, interests, and values (OECD, 2021). This is equally true for the discussion regarding environmental impacts along food supply chains. As the focus of this paper is on facts regarding the extent of environmental impacts, it is worth highlighting different ways in which interests and values come into play.

At its core, environmental sustainability is a policy concern precisely because the interests of market participants normally do not align with the public good. If firms maximise their profits, they will refrain from using more sustainable production methods that are more expensive and have no other benefits to the firm. Similarly, even though consumers may place a value on environmental sustainability in the abstract, these may not play a big role in their personal consumption choices. The resulting market failure may require government intervention, but firms which stand to lose from stricter environmental standards may lobby against such measures.

Yet, it is also possible to think of a more positive dynamic which could at least partly overcome these issues. Growing concerns about environmental sustainability might influence the behaviour of consumers and (potential) employees. This in turn creates incentives for firms to invest in improving the environmental sustainability of their operations. In parallel, growing concerns also raise the demand on governments to act, which may in turn cause firms to try to proactively address issues to pre-empt onerous regulation (e.g. under pressure from investors). Depending on the circumstances, firms with more environmentally friendly operations may even have an incentive to lobby in favour of stricter standards, as these would give it an edge over competitors, giving rise to a “green spiral” (Kelsey, 2021).

Whether or not such dynamics are important enough to significantly reduce environmental damage is an empirical question. For a virtuous circle to emerge, there would have to be sufficient transparency and traceability to avoid “greenwashing” – the practice of pretending to improve environmental sustainability while making little effort to substantially reduce negative environmental impacts. Sound methodological approaches are therefore an important enabling factor in allowing positive dynamics to take hold.

However, all the methodological approaches covered in this paper require assumptions, and different assumptions may change results in ways which matter for market participants. For example, for production processes with several outputs there is a question of how to allocate environmental impacts across these co-products, and different choices will influence how sustainable each product appears. Relative environmental performance may also be influenced by the number and type of environmental impacts included in the analysis, and other methodological choices. For this reason, empirical studies on environmental impacts along food supply chains should be transparent about definitions, data sources, assumptions, and sensitivity of results to various methodological choices. Moreover, it may be useful to see evidence generation and stakeholder engagement as two parts of the same process, allowing more evidence-based engagement with stakeholders while also allowing for robust discussion of methodological choices.

Values and interests also come into play in interpreting disparate environmental impacts. Food systems affect many environmental outcomes, such as soil health, water use, water pollution, greenhouse gas emissions, and biodiversity. While synergies often exist between these outcomes, in many other cases there will be trade-offs: a specific product or producer or a specific policy proposal might score better in some categories and worse in others. Some method is needed for comparing these disparate impacts and evaluating overall environmental sustainability; otherwise, it is difficult to inform better decisions by consumers, producers, and governments. As the discussion below shows, several approaches exist, but these necessarily involve value judgments. An important question is therefore how society can arrive at such value judgments in a way that is both evidence-based and democratically accountable. The principles outlined in OECD (2021) for overcoming obstacles related to facts, interests, and values can be useful here.

2. Downscaled estimates of impacts by supply chain stage

2.1. Overview

A first way in which information on environmental impacts along food supply chains can be constructed is by "downscaling" other estimates. For example, if overall GHG emissions from the transport sector are known, then an estimate of food's share of transport activities could be used to estimate transport-related GHG emissions for food. Repeating this for different stages of the supply chain can then result in a view of food-related impacts along different supply chain stages, which in turn can provide useful insights into the relative contribution of different stages. To date, such estimates have mostly been conducted for GHG emissions and water use.

Recent work by Crippa et al. (2021) has led to the development of EDGAR-FOOD, the first database which consistently covers each stage of the food supply chain for every country for the period 1990-2015. The methodology behind this paper uses detailed estimates of GHG emissions by activity (e.g. "fuel production for transport", "electricity and heat for retail", "solid waste incineration") and combines these with estimates of the food-related shares of those activities. Importantly, the estimates of Crippa et al. (2021) also cover post-retail GHG emissions, such as those related to cooking and waste disposal, as well as emissions of fuels used across the food system. A similar approach was taken by Tubiello et al. (2021), while an earlier estimate of GHG emissions by food supply chain stage is provided by IPCC (2019).

2.2. Findings from downscaled estimates

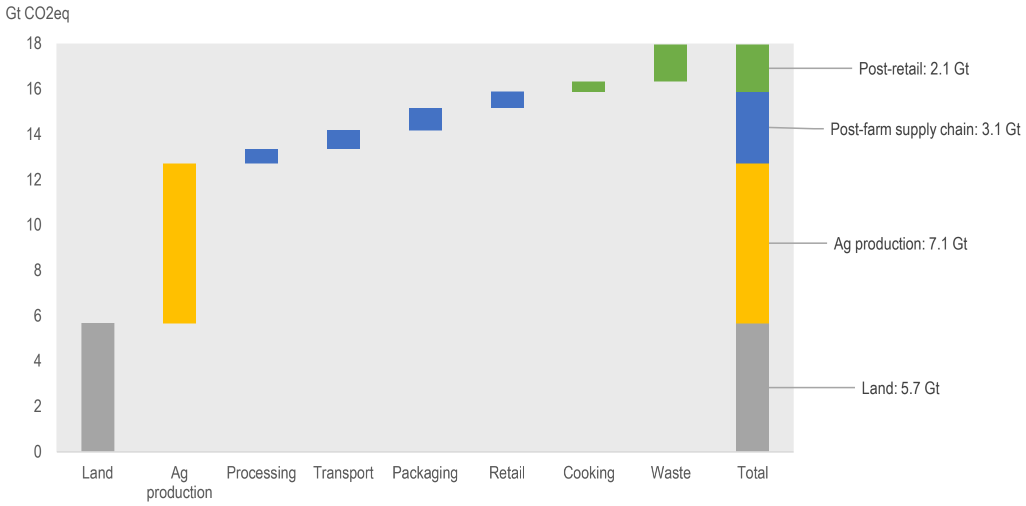
Crippa et al. (2021) calculate that total estimated GHG emissions related to food systems amount to 18 gigatonnes (Gt) of CO_2-equivalent per year in 2015. While this represents an absolute increase by 12.5% from 1990 levels, emissions from other sources grew faster over this period, leading to a decline in the food system's share of GHG emissions from 44% in 1990 to 34% in 2015.

Figure 1 summarises the global estimates of Crippa et al. (2021) by supply chain stage.[3] Globally, 71% of food systems GHG emissions occur either through land use or agricultural production. However, the relative importance of different supply chain stages varies considerably across countries. For example, among 59 industrialised countries and territories, the post-farm supply chain (processing, transport, packaging, retail) accounted for 33% of food GHG emissions, compared to 12% for a set of 163 developing countries and territories.

Research by Tubiello et al. (2021) establishes that emissions generated outside of agricultural land (i.e. pre- and post-production processes) are increasingly important as drivers of overall food systems emissions. This trend is observed at global, regional and national scales.

[3] The presentation of the data here follows that of Ritchie (2021), who compares the Crippa et al. (2021) estimates with those of Poore and Nemecek (2018) discussed in the next section.

Figure 1. Food systems GHG emissions by supply chain stage, 2015



Source: Crippa et al. (2021), using the mapping developed by Ritchie (2021).

2.3. Strengths and limitations

One strength of the downscaling approach is that it can provide relative magnitudes of different stages of food supply chains, which helps in prioritising efforts. For example, at the global level, these estimates underscore the importance of addressing land use change for climate change mitigation. Moreover, the Crippa et al. (2021) estimates cover most of the world, and their taxonomy maps directly into the accounting frameworks used for official GHG emissions inventories.

However, the downscaling approach has several important limitations. By construction, many questions cannot be answered using this approach, such as the environmental impacts of individual food products or production systems. It also misses heterogeneity among producers. The estimates also account for emissions based on where the relevant activity occurs, not where the resulting products are consumed. Other approaches such as life cycle assessments and trade-based analyses can shed light on these aspects.

3. Life Cycle Assessments

3.1. Overview

Life Cycle Assessment (LCA) is a commonly used methodology for assessing the environmental impact of a product over its entire life cycle.[4] The basic principles for conducting an LCA are defined in the ISO 14040 and ISO 14044 standards, which identify four main phases of an LCA: the definition of the goal and

[4] For a comprehensive overview, see Hauschild et al. (2018). For a food systems-specific primer on LCA, see Cucurachi et al. (2019), from which some of the examples in this section are drawn. Historically, LCA focused on environmental issues, but recent advances are also incorporating social aspects, with LCA potentially evolving into a broader Life Cycle Sustainability Assessment; see Moltesen et al. (2018) and UNEP/SETAC (2011). The "true cost of food" approach discussed below can be seen as an example of this trend. The discussion in this section focuses on environmental aspects.

scope of the assessment; the Life Cycle Inventory (LCI) analysis; the Life Cycle Impact Assessment (LCIA); and the interpretation of results.

In addition to being a widely used approach in its own right, many of the concepts and principles of LCA are also used in other approaches, including several of the trade-based approaches discussed in the next section. LCAs also underpin some of the initiatives covered later in the paper. For this reason, it will be useful to review this approach in some detail.

The first phase of an LCA is the definition of the goal and scope. To allow meaningful comparisons across different products or systems, the environmental impacts in an LCA are linked to the relevant *function* delivered by the system being assessed. This function is quantitatively expressed in a *functional unit*. In food LCAs, the functional unit may be based on a quantity of food (e.g. “the delivery of 1,000 litres of drinking milk to consumers”) or on the quality or nutrient content of food (e.g. “the supply of the recommended dietary intake of vitamin C”) (Cucurachi et al., 2019). The definition of the scope then includes a delineation of the *product system* under investigation and the *system boundaries* (i.e. a discussion of which processes are out of scope). For example, in conducting an LCA of milk, the analyst needs to decide whether the production of fertilisers used in growing animal feed is part of the scope.

The next phase in an LCA is a Life Cycle Inventory (LCI), which can be thought of as a large flowchart documenting and quantifying all inputs (natural resources extracted from the environment, e.g. energy and material use) and outputs (all products, co-products, waste, and emissions to air, water, and soil) across the various activities and sub-processes under assessment (Cucurachi et al., 2019). Information on LCI flows can be obtained through primary research, from secondary data such as scientific or technical literature, or from existing databases created for this purpose.

The third phase of an LCA is the Life Cycle Impact Assessment (LCIA), where the various flows quantified in the LCI stage are translated into impacts. A distinction exists between midpoint and endpoint (or damage) approaches. A midpoint approach assesses impacts in terms of a midway point in an impact pathway (for example, ozone depletion), while an endpoint approach follows the impact pathway all the way to final outcomes which matter to the analyst (e.g. human health) (Jolliet et al., 2004). In translating LCI flows into impacts, multiple flows may contribute to the same impact category, as is the case when emissions of different greenhouse gases all contribute to global warming. Conversely, a single flow may contribute to multiple impact categories, as is the case with agrochemicals, which can affect both ecosystems and human health. The translation of flows into impacts is done using so-called “characterisation factors”, such as the Global Warming Potentials used to compare the impact of different greenhouse gases.

The final phase of an LCA is the interpretation of the results. In a comparative assessment of different products, LCAs can indicate which products have better or worse environmental impacts. LCAs can also show which stage of the life cycle contributes most to the total environmental impact. Moreover, where multiple impacts are considered, LCAs could shed light on possible trade-offs, e.g. where a product scores better on one dimension but worse on another compared to a reference product. In some cases, final results of an LCA are weighted (or assigned monetary values) which makes it possible to compare different products on a single metric; this is the approach taken by recent work on the “True Cost of Food” (Box 1).

3.2. Attributional versus consequential LCAs

A major distinction in LCA is between attributional and consequential LCAs (Rajagopal et al 2017). An attributional LCA can be thought of as a “snapshot” detailing the flows and impacts that can be attributed to a product or system at a given point in time. A consequential LCA, by contrast, asks what the consequences would be of a marginal change (e.g. increasing or decreasing total output by one functional unit of a product), considering economic and behavioural feedback and substitution effects. This type of analysis can inform assessments of different interventions (e.g. new policies or technologies) which would have large effects on markets, leading to substitution and other feedback effects. For example, a policy to stimulate biofuels may raise prices of agricultural commodities, which could in turn increase the demand for agricultural land. Such land use change effects would not be captured by an attributional LCA comparing biofuels with conventional fuels, but should be included in a consequential LCA.[5] For this reason, consequential LCAs are more challenging, as they may require specifying multi-market or economy-wide economic models, which inevitably requires many additional assumptions. The vast majority of LCAs on food are attributional (Gava et al., 2020).

Historically, most LCAs were built around an in-depth assessment of a particular process (Finnveden et al., 2009; Guinée et al., 2011). One difficulty with such process-based assessments is that practitioners' definitions of system boundaries may be arbitrary. To overcome this, approaches have been developed which incorporate economic input-output data taken from national accounts. Input-output models are discussed in more detail in the section on trade-based approaches, as they are also commonly used to account for international trade flows. Hybrid attributional approaches (combining in-depth information on the process of interest, with input-output data for other processes) also exist. These are all attributional approaches (Rajagopal et al., 2017).

Both attributional and consequential LCAs are useful, although they provide different types of information. An attributional LCA can be helpful for accounting purposes and in identifying which stages of a product life cycle have particularly high impacts (commonly referred to as "hotspots" in the LCA literature), or in comparing the performance of different producers for otherwise similar products. Moreover, in "micro-level" decision contexts (where an intervention is not expected to lead to large effects on markets), attributional LCAs are likely to be sufficient to support decision-making. Attributional LCAs are also used in determining whether a producer is in compliance with standards (Rajagopal et al., 2017). In discussing major policy interventions, a consequential LCA is needed to understand the full effect.[6] However, as consequential LCAs require many additional assumptions on economic and behavioural responses, these tend to have a large uncertainty range. For this reason, Rajagopal (2014) has argued that the most important contribution of consequential LCAs might be in warning policy makers early in the policy cycle about potential weaknesses in a proposal, rather than in providing precise estimates for uncertain variables.

3.3. Findings from food LCAs

LCAs are becoming increasingly common in efforts to understand the environmental impacts of food supply chains. Rather than reviewing the numerous individual studies reporting LCAs on food products, this report will illustrate findings of food LCAs through the work of Poore and Nemecek (2018), who provide an ambitious synthesis of existing food LCAs. They draw on a comprehensive meta-analysis of 570 studies covering some 38 700 regional- or farm-level assessments in 119 countries (including around 1 000 estimates of postfarm processes), with a focus on five environmental impacts: land use; freshwater withdrawals (weighted by local water scarcity); GHG emissions; acidifying emissions; and eutrophying emissions. To ensure comparability of estimates across studies, the authors used the underlying data on

[5] For a detailed discussion, see Chapter 5 in OECD (2019). For a simulation of land use effects of various biofuels scenarios for the European Union, see Valin et al. (2015).

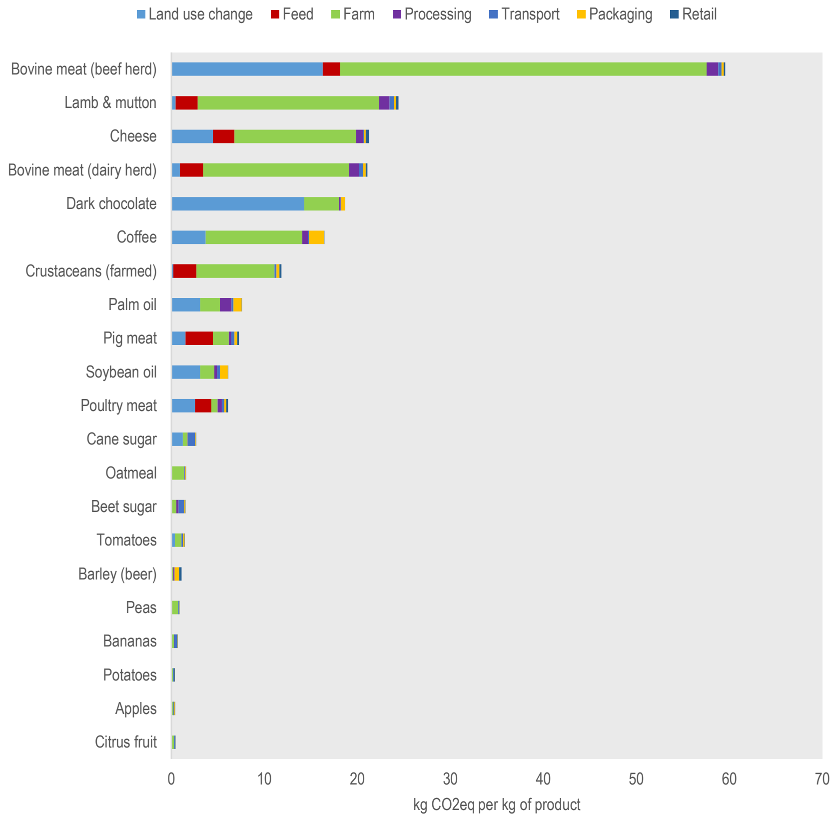
[6] The International Reference Life Cycle Data System (ILCD) Handbook, a widely used reference work developed by the European Commission, distinguishes between three decision contexts: Situation A (micro-level decision support), Situation B (meso/macro-level decision support, i.e. where large-scale effects in other systems are foreseen), and Situation C (accounting, i.e. not intended to support a specific decision). The latter is further disaggregated depending on whether or not interactions with other systems are taken into account. In Situation B, consequential LCAs are needed; attributional LCAs are likely to be sufficient for Situation A and C (European Commission, 2010).

inputs and outputs in each LCA study to re-calculate environmental impacts using a consistent methodology.[7]

In addition to providing new, bottom-up estimates of the environmental impact of food systems, the data in Poore and Nemecek (2018) make it possible to compare impacts across products, as well as across stages of the supply chain (Figure 2), in addition to shedding some light on the heterogeneity across producers (Figure 3). The example here focuses on GHG emissions, although, as noted, Poore and Nemecek (2018) cover several other environmental outcomes as well.

Figure 2 shows median GHG emissions per kg of food for selected products, including a breakdown per stage of the supply chain. As this figure shows, products differ greatly in their GHG emissions intensity, with beef derived from pure beef herds showing a considerably higher GHG emissions intensity than other products. Figure 2 also shows that land use change (LUC) and on-farm GHG emissions typically account for the vast majority of food-related GHG emissions. (Interestingly, rankings of products are fairly similar for the other environmental impacts covered by Poore and Nemecek (2018): livestock products tend to have higher acidification, eutrophication, land use, and scarcity-weighted water use compared to crop products, and ruminant livestock products generally have a worse performance than non-ruminant livestock products).

Figure 2. GHG emissions intensity for selected food products

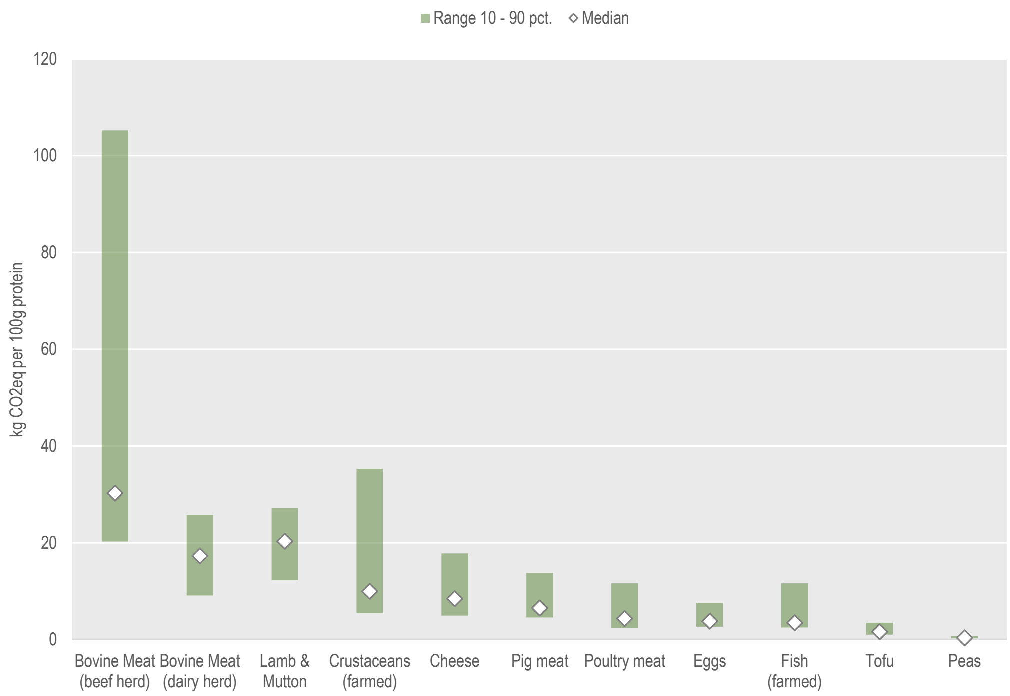


Note: Data represent global median greenhouse gas emissions in kg CO2eq per kg of food (retail weight, excluding waste).
Source: Poore and Nemecek (2018).

[7] Where data were missing, the authors used study coordinates to estimate the missing values from external spatial data sources. An additional meta-analysis of 153 studies and information from global datasets were similarly used to fill in gaps on postfarm processes. Observations were weighted according to the share of national production they represented, and national estimates were subsequently weighted by their share of global production to arrive at global estimates.

While Figure 2 shows the large variation in GHG emissions across products, the Poore and Nemecek estimates also show that GHG emissions intensities vary considerably for the same product. Figure 3 focuses on protein-rich products and expresses GHG emissions per 100g of protein. The chart shows the median GHG emission intensity globally, in addition to a range indicating the variation between the bottom 10% of producers and the top 10% of producers.[8] As Poore and Nemecek (2018) note, high variation across and between products is also found for other environmental indicators such as land use, acidification, eutrophication, and water use. For the three main field crops (maize, wheat, rice), impacts from producers at the 90th percentile are more than three times as large as those from producers at the 10th percentile. With the exception of land use, the same heterogeneity is found even within major producing regions. Across all products, Poore and Nemecek (2018) report that 25% of global production is responsible for more than half of all environmental impacts.

Figure 3. Variation in global GHG emissions intensities of protein-rich products



Note: Figure shows the median and 10th to 90th percentile range of greenhouse gas emissions intensities in kg CO2eq per 100g of protein.
Source: Poore and Nemecek (2018).

Poore and Nemecek's synthesis of food LCAs highlights the wealth of information that can be provided via LCAs. However, despite extensive research efforts, important evidence gaps persist. These can be assessed along the different stages of an LCA, i.e. goal and scope definition; life cycle inventory; life cycle impact assessment; and interpretation.

[8] In presenting their results, Poore and Nemecek (2018) talk about "producers", but this should be seen as shorthand for production-weighted impacts. Thus, the "top 10% of producers" refers to an estimate representing the top 10% of *production*.

3.4. Goal and scope definition

As per ISO standard 14040 and 14044, an LCA begins with defining the goal and scope of the study. The goal of a study should clearly state a number of factors such as intended application, target audience, limitations, and who commissioned the study (Bjørn et al., 2018a). These and other elements serve to put the study in context, so that proper interpretations can be made. The *scope* of the study, in turn, should define both the product system being assessed and how the assessment should take place. The aim here is "to ensure and document the consistency of methods, assumptions and data and strengthen the reproducibility of the study" (Bjørn et al., 2018b).[9]

The goal and scope stages of the LCA are important for transparency and comparability. When methods, assumptions, and data are consistently and adequately documented, this greatly facilitates efforts to synthesize insights from different studies. However, a survey of recent food LCAs by Vidergar, Perc, and Lukman (2021) found that only 10% of the 49 research and review papers included in their analysis documented the intended application of the results, while none of the studies included justifications or explanations for the system boundaries or cut-off criteria used. This lack of documentation limits the potential for synthesis work along the lines of Poore and Nemecek (2018).

Reviews of food LCAs have also identified evidence gaps in terms of products and impacts covered, and in terms of system boundaries, both with regard to geospatial coverage and supply chain stage. For instance, product systems involving livestock have been most commonly studied, with global warming potential or climate change as the predominant impact category analysed (Vidergar, Perc and Lukman, 2021, Gava et al., 2020; Cucuracchi et al., 2019). Looking at reviews published between 2008 and 2018, Halpern et al. (2019) similarly demonstrate that significant evidence gaps exist both in terms of products and impacts (Figure 4).[10] Halpern et al. (2019) calculate that "underassessed" foods represent more than half of animal production in 76 countries, and more than 25% of total food in 40 countries.

Among the impact categories, energy use and biodiversity impacts are particularly under-studied. For biodiversity, this applies both to on-farm biodiversity and to biodiversity effects of induced land use change. Moreover, the analysis of greenhouse gas emissions in LCAs typically does not capture changes in soil carbon, as there is currently no consensus on how this should be quantified (Joensuu et al., 2021).

Studies are also concentrated in high-income countries, with notable gaps for Africa and Central Asia (Poore and Nemecek, 2018, Gava et al., 2020, and Cucuracchi et al., 2019).

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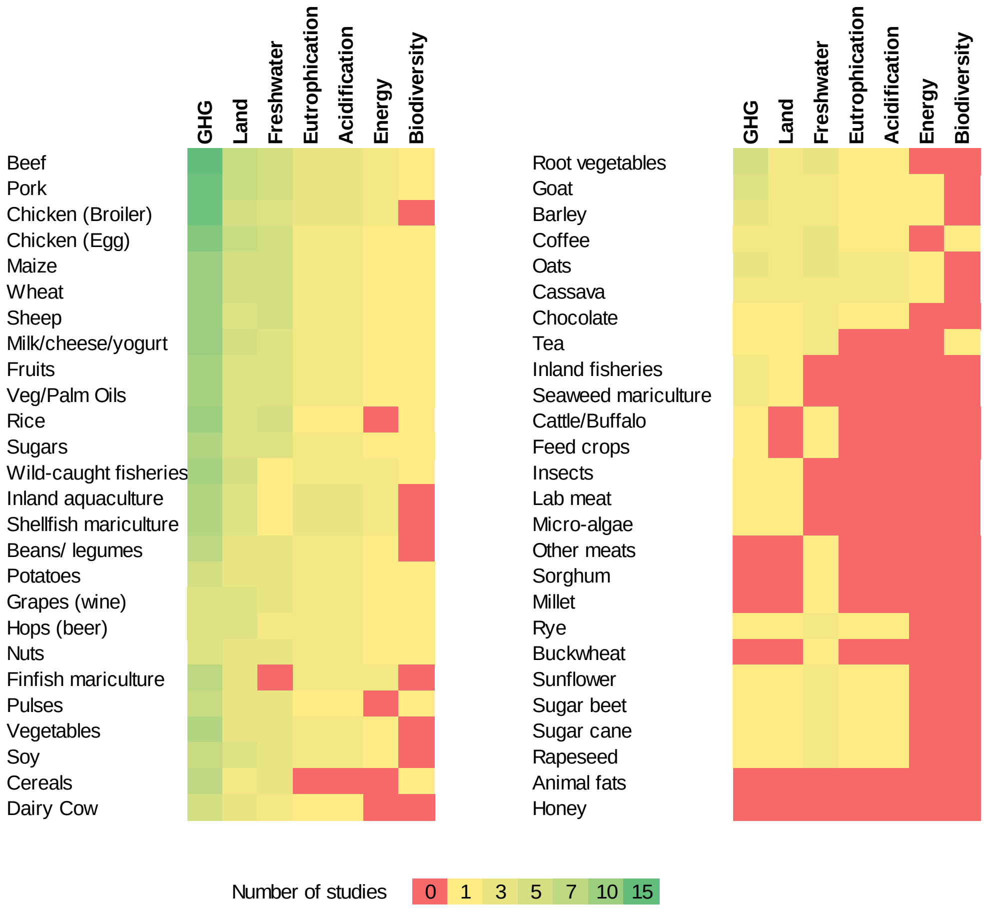
[9] While the scope definition consists of nine items, including some relating to the LCI and LCIA phase, discussion here will centre on evidence gaps as they relate to product systems and their boundaries, leaving a discussion of LCI and LCIA to the next sections.

[10] Halpern et al. (2019) look at sixteen major review papers on the environmental impacts of food products (including Poore & Nemecek, 2018). These papers in turn use a variety of methods, including LCA. A fortiori, gaps in the table signify that gaps exist in the LCA studies included in review papers. The existence of these gaps may not always be obvious from looking at existing databases, which typically cover a large number of products and impacts. But many of these estimates may be based on a mix of extrapolations, older studies, recombinations of earlier estimates made for other purposes, or expert judgment. As discussed below in the context of Life Cycle Inventory data, the use of a "data quality" indicator may be helpful to alert users to the reliability of the underlying data.

Figure 4. Gaps in coverage of products and impacts



Note: Figure shows the number of times each product and impact was included in sixteen review papers published between 2008 and 2018.
Source: Adapted from Halpern et al. (2019).

Lastly, while a life cycle assessment by definition intends to take a view of the entire life cycle of a product or process, only 26% of the studies examined by Vidergar, Perc and Lukman (2021) included use and end-of-life stages. However, it is well known that a large share of food production is wasted at the use stage. A review by UNEP (2021) suggests that globally 16% of all food is wasted by households or in the food service sector (in addition to waste at retail level, and losses along the supply chain). Food waste has direct environmental impacts (e.g. methane emissions from landfills) which ought to be included in a life cycle assessment. In addition, if food loss and waste are not properly taken into account in LCAs, this will lead to an underestimate of actual environmental impacts associated with providing nutrition to consumers. This raises the broader question of how nutritional considerations can be included in food LCAs (e.g. whether the functional unit of food LCAs should be defined in terms of nutritional benefits to consumers rather than in terms of physical products). Recent work by FAO is exploring methodological options for *nutritional* LCA (nLCA) studies, defined as LCA studies where the provision of nutrient(s) is considered as either the main function or one of the main functions of a food item (McLaren et al., 2021).

3.5. Life Cycle Inventory: Data and methodological issues

The LCI phase draws up an inventory of all inputs and outputs related to the product system being studied. Such information can be gathered through primary research, but often researchers rely on scientific or technical literature, or specialised databases. Sources of LCI data can have a very high degree of specificity (when primary data are used, for example measurements at specific sites for the specific product/process being assessed), an intermediate degree (for example, using secondary data from existing LCI databases or from literature on the same product/process), a low degree (for example, extrapolating from LCI databases or literature covering different but similar products/processes), or a very low degree of specificity (expert judgment) (Bjørn et al., 2018b). Some commonly used LCI databases include ecoinvent, ELCD, Agri-footprint, LCA Food, GaBi, and US Federal LCA Commons.[11]

Primary data on the specific product or process being studied clearly have the advantage of being able to capture the heterogeneity in impacts across different producers and production regions, and greatly increase the validity of the study. Databases, by contrast, often face difficulties in capturing the heterogeneity of food systems. The findings of Poore and Nemecek (2018) discussed above highlighted the importance of understanding this heterogeneity in the environmental impacts of food products. However, in their review of food LCAs, Notarnicola et al. (2017) point out that LCI databases used in food LCAs, as well as models used for impact assessment in the LCIA phase, tend to lack spatial or temporal detail. Similarly, despite studies showing heterogeneity *within* specific agricultural practices, these are often grouped together into broad categories such as “organic” (Notarnicola et al., 2017). Moreover, food supply chains may change frequently in response to, for example, weather, policy changes, or changing consumer preferences, which means LCAs may become outdated. Even where individual supply chain actors have data on their own processes, these are often not shared with other actors.[12]

The lack of representation of spatial, temporal and practice-based heterogeneity in current LCA databases limits the quality of evidence that can currently be generated by LCAs. Several initiatives have emerged to respond to these problems. For example, the UNEP-SETAC Life Cycle Initiative stimulates the development of national LCA databases (UNEP, 2020). Specifically for food, the University of Oxford and the World Wildlife Foundation have developed the HESTIA platform, providing a standardised and structured format for users to upload data that is then validated and made available in open access.[13]

Other methodological considerations in the LCI phase can affect the quality of evidence. One important issue is how to account for joint production. When the same production process results in multiple products, there is a question of how the environmental impacts are assigned to these products. Using different methods can lead to different results, and can even be the main driver of those results (Rajagopal et al., 2017).

Wardenaar et al. (2012) illustrate this using the example of bio-electricity from rapeseed oil. In producing rapeseed oil, the by-product of rapeseed cake is produced, which is used as animal feed. Environmental effects could be allocated between these two products using physical partitioning (based on the energy content), economic partitioning (on the basis of the relative economic value of the products), or using a substitution method. The latter method assumes that rapeseed cake displaces other products that would serve as animal feed, and uses the environmental effects of reduced production of these substitutes as

[11] For an overview, see Bjørn et al. (2018b). The Global LCA Data Access network (GLAD, https://www.globallcadataaccess.org/) provides a directory of 80 000 LCA datasets (consulted 3 Nov 2021).

[12] There are, however, new initiatives which try to facilitate such information sharing. In the European Union, for example, there are efforts to create a Common Agricultural Data Space (e.g. https://digital-strategy.ec.europa.eu/en/library/expert-workshop-common-european-agricultural-data-space, consulted 19 July 2021).

[13] See https://www.oxfordmartin.ox.ac.uk/food-sustainability-analytics/ (consulted 3 Nov 2021).

relevant measures.[14] Depending on the method used, the estimated GHG emissions of bio-electricity from rapeseed oil vary from 0.3 to 0.6 kg CO_2 equivalent per kWh. Without properly disclosing and justifying such methodological choices, it is difficult to interpret resulting findings.

As noted earlier, there are considerable gaps in coverage of LCAs, whether in terms of products, impacts, geographies, or life cycle stages. This may not always be clear from the various databases: for example, the French Agribalyse database offers LCA estimates for nearly 2 500 food products commonly consumed in France. Yet, estimates in databases may be based in part on extrapolation (e.g. from different production methods or a different geography) or expert judgment, or may be based on old studies. In the context of the development of the EU Product Environmental Footprint methodology, a "data quality rating" (DQR) assessment has been developed, which can help users of LCAs better understand the reliability of LCA estimates. The score is based on the technological, geographical and time-related representativeness of the underlying data, as well as its completeness, precision/uncertainty, and methodological appropriateness and consistency. As an illustration, Table 1 summarises this score for the products in the Agribalyse database. Assessing data quality in this way can help prioritise for which products new LCA estimates should be developed.

Table 1. Data quality rating of food products in the French Agribalyse database

Data quality level	Data quality rating	Number of products	%
Excellent quality	≤ 1.6	1	0%
Very good quality	1.6 to 2.0	128	5%
Good quality	2.0 to 3.0	1554	63%
Fair quality	3.0 to 4.0	728	29%
Poor quality	> 4.0	68	3%
Total	(median: 2.76)	2479	100%

Note: OECD analysis based on the Agribalyse 3.0 dataset (accessed 14 May 2022), using the mapping to "data quality levels" provided by Manfredi et al. (2012).

3.6. Life Cycle Impact Assessment: Data and methodological issues

The impact assessment phase translates the flows quantified in the LCI stage into impacts, either expressed as "endpoints" (e.g. human health) or "midpoints" (a point along an impact pathway, e.g. ozone depletion). Such conversions are done through so-called characterisation factors or characterisation models. This phase is often highly automated using dedicated software packages.

Rosenbaum et al. (2018b) review different methods in detail. These differ in the number of impact categories considered (e.g. climate change, ozone depletion, particulate matter formation, human toxicity, eutrophication, etc.) and in the specifics of how these impacts are modelled (e.g. whether eutrophication effects consider freshwater, marine, and/or terrestrial ecosystems; whether marginal or average effects are used; what time horizon is used).

As food systems potentially contribute to many different impacts, methods that cover multiple categories together are important to arrive at a comprehensive overview of the environmental impact of food supply chains (Vidergar, Perc and Lukman, 2021). For instance, if one product system outperforms others with respect to one impact category, it is important to know whether there are any other impact categories where its performance is less satisfactory (and if so, to what extent). Weighting, aggregation or grouping are sometimes used when multiple impact categories are assessed; this is discussed in more detail below. As noted above, however, many food LCA studies currently focus on a limited number of impact categories

[14] The ISO 14044 standard recommends that allocation or partitioning should be avoided wherever possible. When it cannot be avoided, physical partitioning is preferred to economic partitioning (ISO 14044, 2006).

(notably climate change). Covering a wider range of impact categories would be important to better understand trade-offs and synergies across environmental impacts.

Impact assessment methods differ in how they model impacts, which raises the question of harmonisation across these methods. There have been efforts to harmonise approaches, such as the European Commission's International Reference Life Cycle Data System (ILCD) Handbook (European Commission, 2010), as well as the Life Cycle Initiative's Global Guidance on Environmental Life Cycle Impact Assessment Indicators (GLAM). The latter aims to create global consensus on impact assessment indicators and methods.[15] In comparing the results of different LCAs, it may be necessary to recalculate impact assessments to ensure comparability, as is done by Poore and Nemecek (2018) in their review of food LCAs.

As in the LCI phase, questions of variability and heterogeneity arise in the LCIA phase. This is particularly important for food LCAs. For example, the same amount of water extraction might be unproblematic or highly problematic depending on local water scarcity. Accounting for such geospatial heterogeneity is an important area of work in impact assessment methods. Work conducted under the auspices of the Life Cycle Initiative led to the creation of a consensus characterisation model for water scarcity based on the concept of "available water remaining" (i.e. after needs of humans and ecosystems are met) (Boulay et al., 2018).

3.7. Interpretation of LCA results

Interpretation is the final phase of an LCA, where the findings of the previous phases are considered together, analysed in light of uncertainties and assumptions, and translated into conclusions or recommendations (Hauschild et al., 2018b). Three steps can be distinguished:

- Identification of significant issues – i.e. methodological choices, assumptions, inventory data, characterisation factors etc. which are most likely to change the final results of the LCA
- Evaluation of the significant issues through *completeness* checks (i.e. checking whether information is missing or incomplete), *sensitivity* checks (e.g. by assessing whether different methodological choices would change the results, or by using simulation techniques to assess the extent to which data uncertainty influences the results), and *consistency* checks (i.e. investigating whether the assumptions, methods, and data used in the study are consistent with the stated goal and scope of the LCA).
- Stating conclusions, limitations, and recommendations on the basis of the findings.

In the context of food LCAs, the interpretation phase should therefore pay particular attention to the inherent heterogeneity and uncertainty of agricultural production processes and environmental impacts.

When multiple impact categories are evaluated in an LCA, it is possible to identify synergies (when one product system performs better on several dimensions) or trade-offs (when one product system is superior on one dimension but inferior on another). To facilitate the interpretation of multiple impacts, these are sometimes weighted, aggregated, or grouped together.[16] It is important to note that these procedures always implicitly or explicitly involve assigning weights to the different impacts, and that this inevitably involves ethical value choices (Rosenbaum et al., 2018b). For this reason, weighting is only an optional step in the ISO 14040/14044 Life Cycle Assessment standard.

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[15] See https://www.lifecycleinitiative.org/activities/key-programme-areas/life-cycle-knowledge-consensus-and-platform/global-guidance-for-life-cycle-impact-assessment-indicators-and-methods-glam/ (consulted 3 Nov 2021).

[16] In the ISO 14040/14044 framework, these are considered optional steps as part of the Life Cycle Impact Assessment.

Several approaches to weighting have been developed (Rosenbaum et al., 2018b); such as:

- Placing a monetary value on different impacts, based on e.g. how much it would cost to prevent the impact, or on monetary estimates of the impact on e.g. human health (for example, based on society's willingness to pay for healthcare costs);
- Weights assigned by experts or stakeholders, or weights based on the distance to a politically or scientifically defined target;
- Assessing the different impacts from the perspective of a number of archetypal profiles of ethical values and preferences.

The "True Cost of Food" approach (discussed below) can be seen as an attempt to express the diverse impacts (including health and social aspects) of food systems using monetary values.

3.8. To what extent is LCA used in policy?

Life cycle assessment is widely used in OECD countries, and governments support its use through a variety of instruments. This includes in-house use by, for example, environmental agencies, as well as information campaigns, financial incentives, or regulations promoting the use of life cycle assessment (CIRAIG, 2020; Sonnemann et al., 2018). For example, within the EU, LCA is promoted as a tool to support initiatives such as the Green Deal, the Circular Economy Action Plan, the Farm2Fork Strategy, the Biodiversity Strategy, and the Chemical Strategy; and a detailed common methodology for "Product Environmental Footprints" and "Organisational Environmental Footprints" based on the LCA methodology have been developed and tested (European Commission, 2021a; 2017). The Product Environmental Footprint and related environmental impact labelling initiatives are discussed in more detail in the companion paper.

Governments and international organisations have also funded much of the research which helped to create the knowledge base for LCA. Several governments have set up publicly available LCA databases (Gava et al., 2019), including the French Agribalyse database of food LCAs highlighted earlier. The European Commission also coordinates the European Platform on Life Cycle Assessment (European Commission, 2021b). The UNEP-SETAC Life Cycle Initiative was mentioned earlier (UNEP/SETAC, 2011). FAO coordinates the Livestock Environmental Assessment and Performance (LEAP) Partnership, a multi-stakeholder initiative to harmonise methods, metrics, and data to measure the environmental performance of the livestock sector (FAO, 2022).

Rajagopal et al. (2017) distinguish three main ways in which life cycle assessment can be used as a policy tool:

- *Information provision*: For example, a public agency might provide life cycle information to stimulate efforts by consumers and businesses to reduce their footprints. Alternatively, firms may be required to report such information themselves, as is increasingly the case (see the discussion on mandatory disclosure in the companion paper).
- *Passive environmental regulation*: For example, the results of LCAs may be used to inform decisions around public support for specific products, technologies, or end-of-life management practices (e.g. recycling), but without the policy instruments itself referring to the quantitative findings of an LCA.
- *Active environmental regulation*: For example, when estimates from LCAs directly determine taxes or subsidies, or whether or not firms are deemed to have met environmental standards. As Rajagopal et al. (2017) note, this use of LCA requires greater accuracy and verifiability of claims relative to the other uses.

The *OECD Recommendation of the Council on Regulatory Policy and Governance* (OECD, 2012) recommends that countries integrate Regulatory Impact Assessment into the early stages of the policy process. Such *ex ante* assessments are particularly important for food systems, given the multitude of possible synergies and trade-offs that may occur for any given policy intervention (OECD, 2021). LCA can play an important role as a methodology in these impact assessments (Sala et al., 2016).

Despite their growing popularity, researchers have noted several obstacles to a more widespread adoption of LCA in policy (Rajagopal et al., 2017). In addition to the methodological challenges noted earlier, some implementation challenges include a lack of confidence among policy makers in the robustness of life cycle assessments; the difficulty of holding specific producers responsible for indirect economy-wide effects (i.e. those studied in consequential LCAs, as opposed to attributional LCAs); a sometimes inadequate choice of alternatives considered in LCAs; a lack of communication and engagement between analysts, policy makers and stakeholders; and the cost of conducting detailed LCAs. Successful examples of the use of LCA in public policy were characterised by high levels of stakeholder involvement from the beginning of the study, which led to a better understanding and increased trust. Seidel (2016) concludes that the failure of LCAs to contribute more strongly to public policy development is not due to technical problems with LCA, but rather to the process within which the LCA is incorporated.

3.9. Strengths and limitations

LCAs are an essential tool. Consumers, producers, and policy makers all require reliable and comparable information on environmental impacts of different products, production processes, or supply chain stages as inputs in their decision-making. These questions call for life cycle thinking, and a major strength of LCA is that it presents an increasingly standardised and harmonised approach to conducting the necessary assessments. LCA practitioners can rely on international standards (ISO 14040/14044), detailed handbooks (Hauschild et al., 2018a; Matthews et al., 2014), and extensive databases and models (Bjørn et al., 2018b; Rosenbaum et al. 2018b), and benefit from ongoing efforts to harmonise and further develop LCA approaches, for example through the Life Cycle Initiative.

LCAs have been increasingly used in the context of food systems, and their potential is illustrated well by the synthesis of food LCAs in Poore and Nemecek (2018). Their work confirms earlier findings that environmental impacts along food supply chains are often concentrated in the agricultural production stage (including land use change); that the environmental performance of food products shows a large degree of heterogeneity; but that it is nonetheless clear that food products differ systematically in their environmental footprint, with notably worse environmental performance for ruminant meat.

The intrinsic heterogeneity of food production suggests that food LCAs should use detailed information wherever feasible, rather than relying on extrapolation or broad averages (Notarnicola et al., 2017; Lathuillère et al., 2021). In particular, the importance of geospatial heterogeneity, and of heterogeneity within the same production region, suggests that for some purposes it may be necessary to trace products back to specific producers or sub-national regions in order to correctly assess their environmental impact. As discussed below, several recent approaches have emerged which try to map environmental impacts of food production through international trade to final consumption, including approaches to map sub-national supply chains.

Major policy interventions are likely to affect markets, creating substitution and other feedback effects. Understanding the consequences of a proposed policy would thus require analysts to go beyond attributional LCAs towards consequential LCAs that take into account those market adjustments. However, consequential LCAs require many additional assumptions and can therefore come with large uncertainty ranges. Some consequential approaches are also discussed in the section on trade-based approaches.

Because food systems exert pressures on a wide range of environmental issues, it is important that food LCAs cover as many environmental impacts as possible, not just climate change. Many food LCAs have indeed done so, or enable the estimation of these multifaceted impacts using characterization models (Poore and Nemecek, 2018), although important gaps remain in terms of products, impacts, geographies, and supply chain stages covered. It has also been suggested that food LCAs should take into account not just environmental impacts but also social/economic impacts, as well as health or nutrition effects (e.g. Soussana, 2014). Having these broader impacts included in a single assessment would help with clarifying the trade-offs and synergies involved.

While LCAs generally report impacts across different impact categories, some approaches have proposed placing monetary values on these impacts in order to facilitate comparison. Such approaches aim to account for the “True Cost of Food”; life cycle assessment can be an important input in such analyses (Box 1).

Box 1. Life cycle assessment and true cost accounting

In recent years, there has been growing interest in quantifying the “true cost of food” by taking into account various externalities associated with the production and consumption of food, including not only positive or negative environmental externalities, but potentially also health externalities and social and economic aspects (Hendriks et al., 2021; de Adelhart Toorop et al., 2021; Sandhu et al., 2021). The idea behind true cost accounting (TCA) is that systematic measurement and valuation of the wide range of impacts is an important first step in facilitating more sustainable choices by governments and stakeholders.

True cost accounting can be seen as a further development of cost-benefit analysis (OECD, 2018; Merrigan, 2021). Several methodologies have been developed (see de Adelhart Toorop et al. (2021) for a comparison). One well-known approach is The Economics of Ecosystems and Biodiversity (TEEB) initiative, which has been active since 2007 and has developed a framework for evaluating the diverse impacts of food systems on human well-being in terms of environmental, economic, health, and social impacts. The evaluation framework distinguishes between stocks (of natural capital, produced capital, human capital, and social capital), flows (through food supply chains), outcomes (interpreted as changes in the four stocks), and impacts (translating these changes into effects on human well-being). A specific evaluation framework for food systems (TEEBAgriFood) has been developed as well (TEEB, 2018). Another approach is the Food System Impact Valuation Initiative (FoodSIVI) (Lord, 2019); various non-sector-specific TCA approaches exist as well (de Adelhart Toorop et al., 2021).

Life Cycle Assessments can be used as the starting point for a TCA calculation. As noted earlier, LCA can lead to an estimate of impacts along a number of environmental dimensions. These findings can be translated into a “true cost” assessment by applying estimates of the social cost of the different impacts (e.g. the social cost of carbon). Following a similar logic, approaches have also been developed to quantify social and nutritional impacts (WBCSD, 2018).

Any evidence gaps that affect LCA in food systems also carry over to TCA estimates. Hendriks et al. (2021) note that there is substantial uncertainty due to (1) incomplete coverage of the various impacts of food systems, (2) major uncertainties in primary data, (3) uncertainties in trade data, (4) uncertainties in the modelling of impact pathways, and (5) uncertainty in how to place a monetary value on impacts. Biodiversity and pollution in particular have high uncertainty. Incorporating non-environmental factors (e.g. health impacts of diets, impacts of undernutrition, social aspects such as child labour) also poses important conceptual and data challenges.

Nonetheless, there is clearly a growing interest in TCA among researchers, policy makers, and other stakeholders. Many frameworks and approaches have been developed in the last few years (d Adelhart Toorop et al., 2021; Baker et al., 2020; Gemmill-Herren et al., 2021), and private sector stakeholders (including major accountancy firms such as KPMG or Ernst & Young) are increasingly involved in operationalizing TCA principles to facilitate reporting by businesses (de Adelhart Toorop et al., 2021; WBCSD, 2018).

4. Trade-based approaches

The various approaches reviewed in this section use information on economic flows (input-output tables, bilateral trade data, customs data) to try to map flows of products, and hence to link environmental impacts from where production takes place to where the final products are consumed. Much of this research has taken place in the last few years, and the literature, methods and data are rapidly evolving. This section first discusses several attributional approaches, before briefly considering consequential methods which take into account market responses.

4.1. Input-output analysis

Life cycle assessments are traditionally conducted using a process-based approach, as described in the previous section. Such approaches have often been complemented with input-output analysis, which relies on economy-wide input-output tables (Mattila, 2018). Such tables show how the outputs of one sector in the economy are used as inputs in other sectors. For example, the production of trucks depends on the production of metals, machines, and ore mining (among other inputs), but the production of these intermediate inputs itself involves many feedback loops: metal production itself requires ore, machines, and trucks, while machine production requires metal and trucks. If the goal is to understand, for example, the amount of metal needed to produce a truck, these indirect effects need to be taken into account. Ideally, an input-output table would capture these flows between sectors, which then makes it possible to use linear algebra to calculate the total (direct and indirect) amount of inputs required to produce a specific output. In the context of a life cycle assessment, this estimate of the required inputs can then be combined with information on environmental flows directly related to the production of the inputs to arrive at total "embodied" environmental impacts associated with the final product.

The approach of using input-output tables for assessing environmental impacts is known as Environmentally Extended Input-Output Analysis (EEIO). One example is recent OECD work to quantify the CO_2 emissions from fuel combustion that are embodied in international trade and in final demand using the OECD Inter-Country Input-Output (ICIO) tables (Yamano and Guilhoto, 2020), and work on consumption-based GHG emissions indicators for the Agriculture, Forestry, and Land Use (AFOLU) sector (Garsous, 2021; Hong et al., 2022).

Two strengths of the EEIO approach are its speed (since for some applications data is readily available) and comprehensiveness (as it in theory captures all inputs, whereas a process-based life cycle assessment necessarily involves determining a system boundary). An important limitation is that input-output tables describe the economy in terms of a higher level of aggregation compared with process-based LCA. In addition, creating economy-wide input-output tables takes time, so available input-output tables are usually several years old. Another limitation of EEIO is that the data usually only covers a limited number of environmental impact categories, compared with the potentially much more detailed information found in a process-based LCA. For this reason, some analyses combine the two approaches, using a process-based LCA for the main system being studied, while using EEIO for other elements not explicitly studied. Finally, most input-output tables show linkages between sectors in monetary terms, not in physical terms, although there is ongoing work developing datasets on physical flows.

Commonly used databases

In a global economy, the relevant input-output table includes international linkages. Several multi-region input-output tables (MRIOs) with environmental impact data exist, such as the World Input-Output

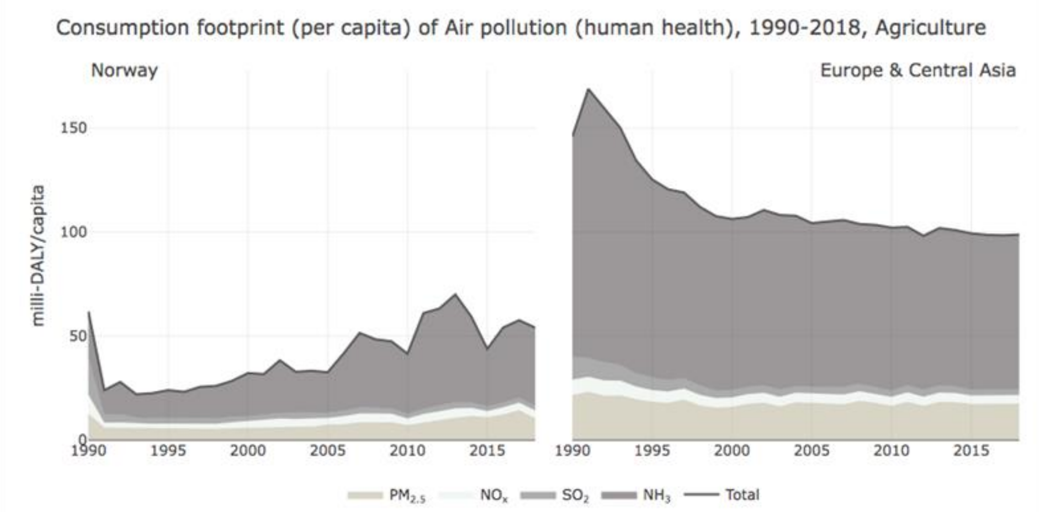
Database (www.wiod.org), Exiobase (www.exiobase.eu), and EORA (www.worldmrio.com) (see Mattila, 2018 for a discussion).[17] The OECD ICIO database has also been extended for environmental impact assessment (Yamano and Guilhoto, 2020; Garsous, 2021). Some analyses also rely on the GTAP database (https://www.gtap.agecon.purdue.edu/).[18] Further developments in environmental-economic accounting could help improve the quality of MRIO approaches over time (Box 2).

Sustainable Consumption and Production Hotspot Analysis Tool

The Sustainable Consumption and Production Hotspot Analysis Tool (SCP-HAT, http://scp-hat.lifecycleinitiative.org/) is an example of environmentally extended input-output analysis taking into account international trade flows.[19] SCP-HAT combines the EORA multi-region input-output database with additional datasets on environmental pressures, environmental impacts, and socio-economic information. This makes it possible to allocate environmental effects not only to the location where production takes place but also to the location where the final goods are consumed.

Figure 5 shows one type of analysis made possible by the SCP-HAT database – an estimate of human health impacts caused by air pollution related to agricultural products consumed in Norway, as well as a benchmarking of this impact to a regional average.

Figure 5. Norway's consumption footprint (per capita) of human health impacts caused by agricultural air pollution, 1990-2018



Note: The first panel shows the human health impact (in millions of disability-adjusted life years per capita) related to the consumption of agricultural products in Norway. The second panel shows the same statistic for Europe and Central Asia.
Source: SCP-HAT version 2, http://scp-hat.lifecycleinitiative.org/module-2-scp-hotspots/ (consulted 7 Jan 2022).

[17] The Exiobase3 dataset is presented in Stadler et al. (2018).

[18] However, as the online GTAP documentation points out, it is not recommended to use GTAP as an MRIO, as the database was not designed for this purpose.

[19] SCP-HAT is a joint initiative of the Life Cycle Initiative, the One Planet Network, and the International Resource Panel.

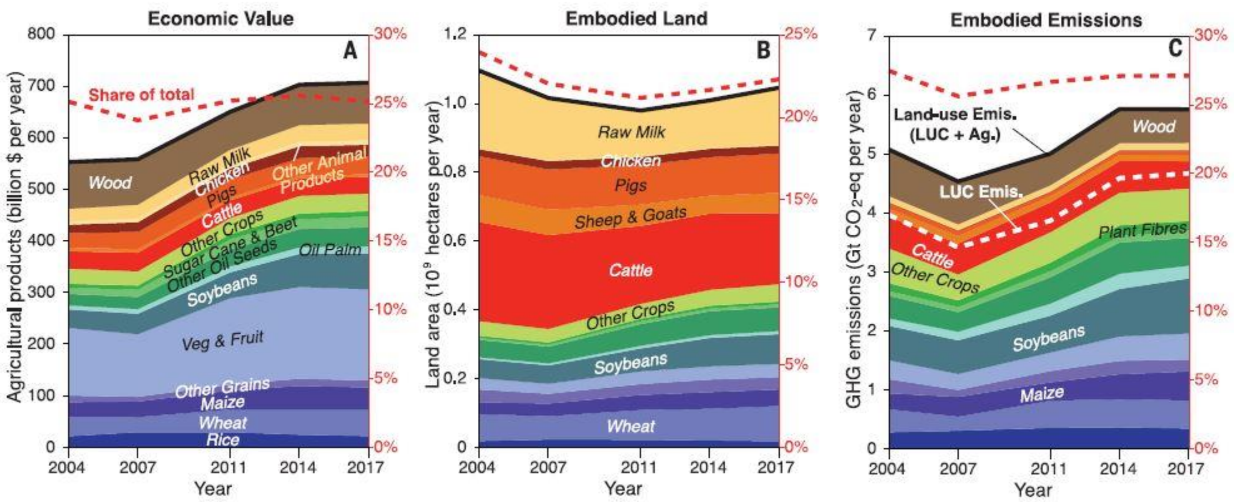
In its first version, SCP-HAT distinguished 26 sectors, which meant the level of detail on different food supply chains was limited (e.g. only agriculture, fishing, and food and beverages were distinguished). The second version, launched in September 2021, provides greater sectoral coverage (for example, agriculture is now separated into 20 distinct activities such as wheat production or poultry production). At present, however, SCP-HAT does not yet provide a view of environmental impacts along different stages of food supply chains.

Hong et al.'s estimates of land use emissions embodied in trade

A recent study by Hong et al. (2022) presents new estimates of land use emissions (i.e. direct emissions from agriculture as well as emissions from land use change) embodied in international trade flows. The trade analysis is based on the GTAP database. Main results are shown in Figure 6.

Several interesting patterns emerge from this analysis. First, 25% of the economic value of global agricultural production, 22% of agricultural land, and 27% of emissions are embodied in international trade. This indicates that international trade accounts for a significant share of the environmental impacts of food systems, although the majority of impacts are in fact related to purely domestic flows. Second, the relative share of products varies depending on the indicator studied. Vegetables and fruit are roughly a quarter of traded products by value, but account for less than 8% of embodied land and emissions. Animal products account for more than half of embodied land, but less than 20% of embodied emissions. Hong et al. (2022) also provide further decompositions and analysis.

Figure 6. Value, land use, and land use emissions of agri-food products embodied in trade



Note: Chart shows the economic value (panel A), land use (panel B) and emissions (panel C) of agricultural products (including wood) embodied in international trade. The solid black line and coloured areas show absolute values. In addition, the dotted red line (measured on the right axis) shows what share of agricultural production value, land, and emissions are embodied in international trade.
Source: Hong et al. (2022).

Box 2. The System of Environmental Economic Accounting (SEEA)

The System of Environmental Economic Accounting (SEEA), developed jointly by the United Nations, the European Commission, FAO, the OECD, the IMF, and the World Bank Group, provides an international standard for environmental-economic accounting, analogous to the System of National Accounts. The SEEA establishes concepts, definitions, classifications, accounting rules and tables to organise statistics on the environment and its relationship with the economy in an internationally comparable way. Moreover, the SEEA framework is explicitly designed to be consistent with the System of National Accounts. As with the System of National Accounts, the SEEA can be used to generate a wide range of statistics, accounts, and indicators for a variety of analytical applications.

The Central Framework of the SEEA was adopted as the first international standard for environmental-economic accounting in 2012. It covers "environmental assets" (water resources, energy resources, forests, fisheries, etc.), their economic use, and returns back to the environment (waste, and air and water emissions). As of 2020, more than 90 countries had compiled SEEA accounts while many others are planning to do so. Several international organisations (including UN agencies, Eurostat, and OECD) are collaborating to explore the possibility of creating global databases.

Complementing the Central Framework is SEEA Ecosystem Accounting, which focuses on habitats and landscapes, and ecosystem services. The United Nations Statistical Commission adopted the SEEA Ecosystem Accounting in 2021. In contrast with the Central Framework, the Ecosystem Accounting takes a spatially explicit approach, defining a framework which could be used to compile accounts that provide information on the location and size of ecosystem assets as well as the location of households, businesses and governments benefiting from ecosystem services. For this reason, ecosystem accounts can be represented using maps in addition to tables, and can be compiled at different spatial scales, e.g. at the river basin level, for a specific protected area, etc., in addition to the more traditional national or sub-national administrative levels. A growing number of ecosystem accounts are being developed (e.g. the National River Ecosystem Accounts for South Africa), and ecosystem accounts are expected to play an important role in monitoring progress towards the Sustainable Development Goals.

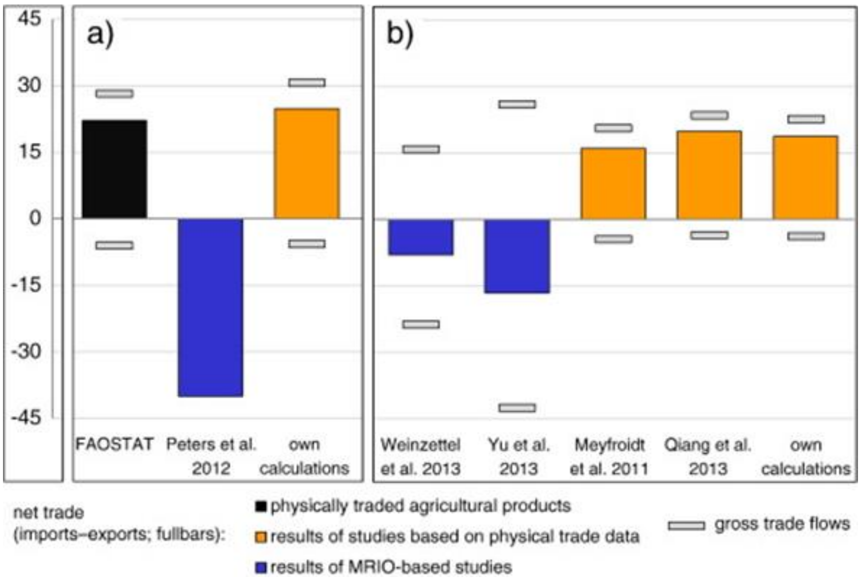
Source: United Nations Department of Economic and Social Affairs, "System of Environmental-Economic Accounting," https://seea.un.org/content/homepage (consulted 14 Oct 2021).

4.2. Physical trade flows

Input-output tables are typically expressed in monetary values rather than physical quantities. The EEIO approach therefore implicitly assumes that environmental impacts are embodied in product flows proportionally to their monetary value. In terms of the LCA methodology discussed in the previous section, most EEIO approaches therefore by construction use an economic allocation method. This approximation may give misleading results. For example, Kastner et al. (2014a) demonstrate that data on physical product flows suggest that China was a major net importer of cropland embodied in products in 2004, whereas an approach based on monetary input-output tables implausibly suggests that China was a major net exporter (Figure 7).

As Kastner et al. (2014a) note, one possible explanation is that price differences and product heterogeneity skew the results. For example, the price of a higher-quality version of a product may easily be several times higher than the price of a lower-quality version, even if the actual amount of physical resources embodied in the product is fairly similar.

Figure 7. Comparison of physical and value-based estimates of Chinese net trade in agricultural products and embodied land



Note: Estimates of Chinese net trade of (a) crop products (in million tonnes of carbon per year) and (b) cropland embodied in trade flows (in million hectares per year), 2004.
Source: Kastner et al. (2014a).

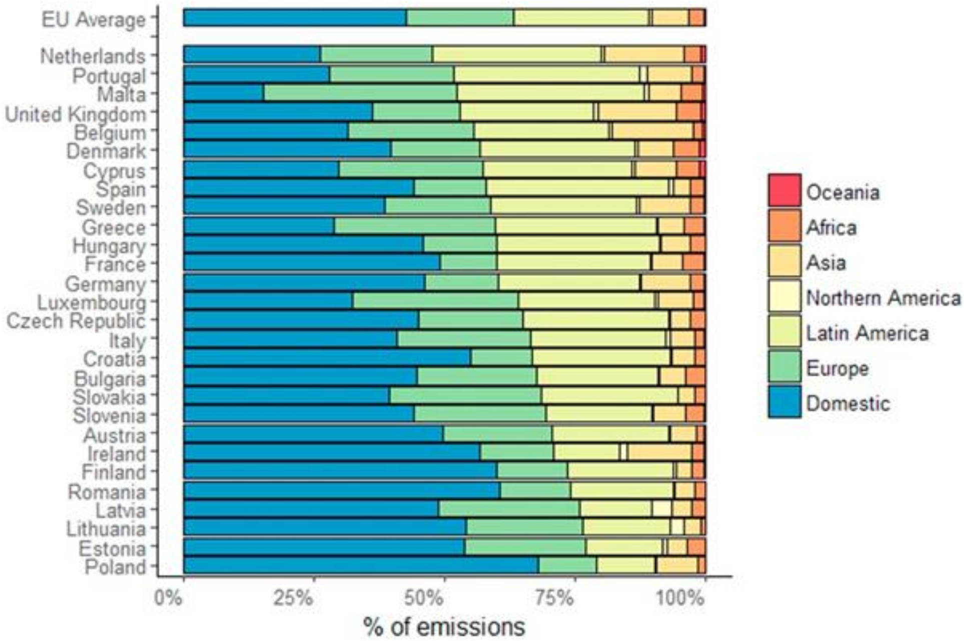
Kastner et al.'s approach

For this reason, physical data on bilateral trade flows have been used as an alternative approach to assess the embodied environmental impacts of food (Kastner et al., 2011; Kastner et al., 2014b).[20]

Sandström et al. (2018) used such an approach based on physical trade to quantify GHG emissions of average diets in EU countries. Their method starts with country-level food supply statistics (from FAO's Food Balance Sheets), to which an estimate of animal feed embodied in animal product consumption is added. These figures are then converted into primary product equivalents. Following Kastner et al. (2011) and Kastner et al. (2014b), data on bilateral physical trade in food and agricultural products are then used to link these consumption figures back to the original producing country (taking into account intermediate re-exporting countries). Various data sources are used to calculate GHG emissions, including those associated with transport and with land use change. The results show the importance of accounting for international trade: consumption-based emissions are 40% higher than production-based estimates of EU food-related emissions, and about one-third of total emissions related to the EU diet are incurred outside of the EU (notably in Latin America) (Figure 8).

[20] For related work on physical flows of energy, see Guilhoto et al. (2021).

Figure 8. Emissions of EU diets, by production region



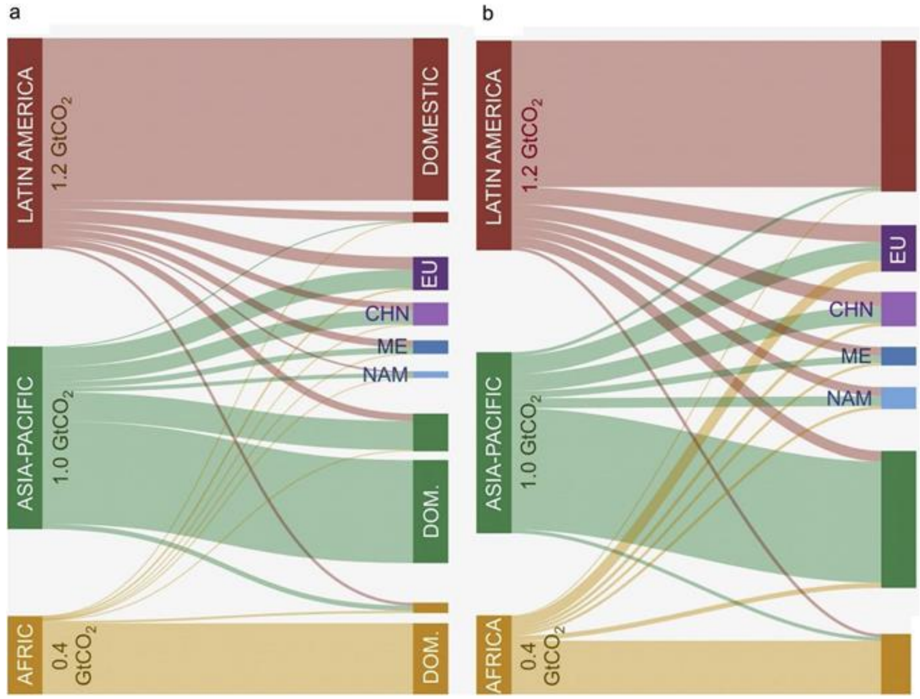
Note: Figure shows the share of different production regions in the dietary GHG emissions of EU-28 countries in 2010. ("EU Average" refers to EU-28).
Source: Sandström et al. (2018).

As Sandström et al. (2018) note, their approach does not capture full life cycle emissions of EU diets, as the methodology does not account for emissions from energy use, the production of inputs (e.g. fertilisers), food processing and manufacturing, transportation emissions within the consumption country, or emissions related to retail, storage, cooking, or waste processing. Moreover, the emissions factors used are national averages of the producing countries, which does not capture heterogeneity within those countries, e.g. in different sub-national regions or across different producers or production systems within the same region.

A similar approach based on physical trade data was used by Pendrill et al. (2019) to assess the extent to which consumers in high-income countries contribute to deforestation abroad. In a first step, observed deforestation is attributed to the expansion of cropland, pasture, or plantations (e.g. oil palm). In a next step, the expansion of cropland is attributed to different crops. Using the methodology of Kastner et al. (2011) and Kastner et al. (2014b), physical data on bilateral trade in agricultural and food products is then used to track products from where they are produced to where they are consumed (again, taking into account intermediate re-exporting countries). The authors also use an MRIO model as comparison. Figure 9 shows the estimated flows: using the physical trade model, an estimated 26% of deforestation-related emissions are embodied in international trade; this share rises to 39% when using the MRIO model (which considers more indirect links between sectors).[21]

[21] In the original paper the trade share using the physical trade model was erroneously reported as 29%; see Pendrill et al. 2020 for a clarification.

Figure 9. Trade flows of embodied emissions from deforestation, 2010-2014



Note: Trade flows of embodied GHG emissions from region of production (left) to region of consumption (right), based on (a) a physical trade model and (b) an MRIO model. Region abbreviations: EU – Europe, CHN – China, ME – Middle East, NAM – North America. For panel (a), "Domestic" or "Dom." denotes embodied emissions consumed in the same country; for panel (b), due to regional aggregation, it is not possible to distinguish domestic and intra-regional flows. Figures denote average annual emissions due to deforestation.
Source: Pendrill et al. (2019)

Taherzadeh et al. (2019) similarly use a physical trade approach to calculate embodied water and land use in international soybean trade. They find that about one-third of global soybean-related water and land use is driven by international trade. Sporchia et al. (2021) use a similar approach to assess the environmental impacts of cocoa, coffee, tea and tobacco supply chains.

The Food and Agriculture Biomass Input–Output Model (FABIO)

To facilitate analyses in terms of physical trade flows, Bruckner et al. (2019) introduced the Food and Agriculture Biomass Input–Output Model (FABIO), a set of multiregional supply, use and input–output tables documenting physical flows of agricultural and food products. The database relies on FAOSTAT data on crop production, trade, and utilization, and covers 191 countries and 130 agriculture, food, and forestry products from 1986 to 2013.

Helander et al. (2021) use the FABIO model to compare the relative effectiveness of dietary shifts (towards more plant-based diets) and reductions in food waste, using Germany as a case study. They use FABIO to capture the biomass, cropland, and water footprints embedded in German diets. Their results show that dietary changes are more effective than reductions in food waste, except for water footprints. The results also show that a dietary shift might increase food waste due to a greater consumption of products with higher food waste shares. Combining different approaches seems to hold the greatest potential.

4.3. Spatially Explicit Information on Production to Consumption Systems (SEI-PCS)

As the examples above show, the availability of physical bilateral trade data for agricultural and food products makes possible a more fine-grained analysis than what could be achieved with multi-region input-output tables. Yet one important limitation of input-output models also applies to the approaches reviewed here: their use of national averages of environmental impacts in the producing countries. Studies typically assume that these average impacts can be proportionally allocated between domestic consumption and exports, and between different importing countries. For example, the methodology of Pendrill et al. (2019) assumes that if X% of the soy produced in country A is consumed in country B, then X% of the soy-related deforestation in country A is allocated to country B. Similarly, Taherzadeh et al. (2019) use national averages for land and water use in soybean production.

Such approximations ignore important heterogeneity, as different sub-national regions, different producers, or different production systems may differ in their environmental impacts. This would be true, for example, if exporting firms and domestic firms source from different regions, or if exporting firms use stricter standards than firms oriented towards the domestic market. Moreover, importing countries do not all source their products from the same exporting firms, which means the allocation of environmental impacts across importing countries is not necessarily proportional to total quantities imported.

Fortunately, in the last few years important progress has been made on providing more fine-grained assessments. Much of this work has built on the Spatially Explicit Information on Production to Consumption Systems approach (SEI-PCS) pioneered by Godar et al. (2015) and subsequently developed by the Trase initiative (www.trase.earth) (Box 3).

The core insight underlying the SEI-PCS approach is that various datasets can be combined to link export volumes to specific trading firms; to link these firm-specific export volumes back to production regions at a subnational scale (e.g. municipalities); and to follow these export flows “forward” towards importing countries. By linking production and export volumes back to subnational production regions, it becomes possible to better account for heterogeneity in environmental (and social) impacts, and to trace how these impacts flow towards importing countries.

Box 3. The Trase initiative

Trase (www.trase.earth) is a partnership between the Stockholm Environment Institute and Global Canopy to create greater transparency around supply chains of commodities with high deforestation risk. The goal of Trase is to allow companies, investors, governments and other stakeholders to identify opportunities for improving environmental and social sustainability of commodity supply chains.

Researchers affiliated with Trase build on the Spatially Explicit Information on Production to Consumption Systems approach pioneered by Godar et al. (2015). By using a variety of different data sources (e.g. customs, shipping, tax, or logistics data) this approach provides a detailed view of supply chains linking importing countries with the places where production takes place. The approach also identifies the firms involved in the supply chain, as well as their investors. This information is freely available on the Trase website, as are several interactive tools to explore the data.

Trase currently provides detailed information on soy, beef, coffee, cocoa, cotton, corn, pork and chicken from Brazil, soy and beef from Paraguay, soy from Argentina, palm oil from Indonesia, shrimp from Ecuador, and coffee from Colombia. Related work is mapping the ownership and investment patterns of firms active in soy and beef in Brazil, and palm oil in Indonesia.

Source: www.trase.earth (consulted 27 September 2021).

Because the availability and granularity of data differs by country, SEI-PCS is not a single methodology but rather an overarching approach to achieving a fine-grained subnational mapping of commodity supply chains. The mapping typically relies on data such as customs records, maritime shipping contracts, tax registration data, ownership and capacity data of logistics hubs, and subnational production data. Where detailed data is not available, modelling approaches can be used to allocate export volumes to production regions (e.g. using assumptions around transport costs) (Trase, 2018). The Trase initiative, set up to further develop and apply the SEI-PCS approach, aims to cover all major commodities linked to deforestation risks.

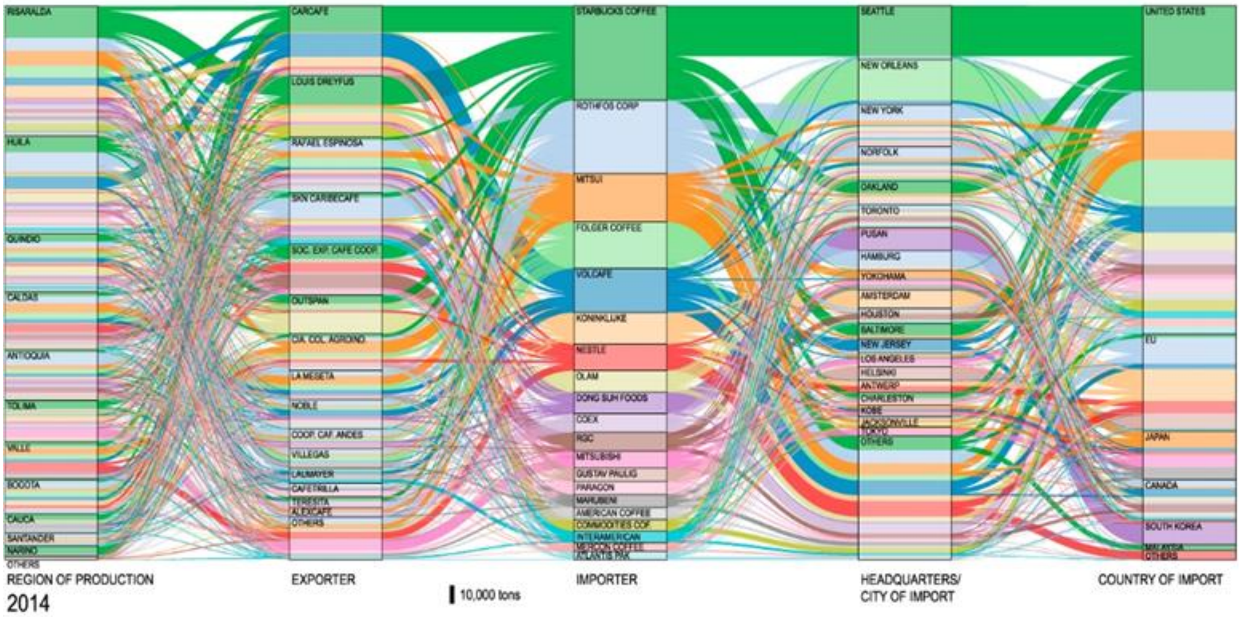
As Godar et al. (2016) explain, this approach can be seen as a “middle ground” between macro-level trade-based approaches and micro-level detailed life cycle analyses or fine-scale traceability systems.[22] Godar et al. (2016) provide insights on several commodity supply chains for which the SEI-PCS approach has been implemented. For example, in 2012 Chinese consumption of Brazilian soybeans was more concentrated in the south and southeast of Brazil and in the western Cerrado; EU consumption by contrast depended more on soy from the northern Cerrado (which has seen rapid deforestation), the Cerrado–Amazon frontier in Mato Grosso, and the southern Amazon. The EU also consumes most of the soy produced in the central and eastern Amazon. Moreover, the EU sourced more Brazilian soy from “critical deforestation municipalities” than any other major consumer. In the case of Indonesian palm oil production, the SEI-PCS approach was even able to identify whether plantations were managed by smallholders or private firms. The analysis also showed that about half of all palm oil exports in 2010 were shipped through just three ports, which could be useful information in implementing a targeted monitoring system. In the case of Colombian coffee exports (Figure 10), the SEI-PCS approach showed that Starbucks had become the main importer of Colombian coffee beans by 2014, sourcing mainly from Carcafe and Louis Dreyfus, and with a large share of its imports coming from the Risaralda department. Such information could be useful in identifying relevant stakeholders who could work together to improve environmental outcomes.

A number of academic studies have used the SEI-PCS approach, usually with a focus on soy or beef supply chains.

Escobar et al. (2020) use the disaggregated information on Brazilian soy supply chains to derive life cycle carbon footprint estimates for Brazilian soy exports between 2010-2015. Their analysis differentiates between roughly 90 000 different life cycles from the municipalities where production takes place to the different countries of origin. This differentiation enables differences in land use pressures, agricultural yields, input use levels, modes of domestic transport, processing, and international shipping to be taken into account. The results show large spatial variability in carbon footprints across Brazil (Figure 11). Across municipalities, carbon footprints range from 0.1 to 29 tonnes of CO_2 equivalent (t CO_2eq) per tonne of soybeans produced, with a range between the 10th and 90th percentile from 0.28 to 0.75. The variability of carbon footprints is mostly explained by different land use pressures.

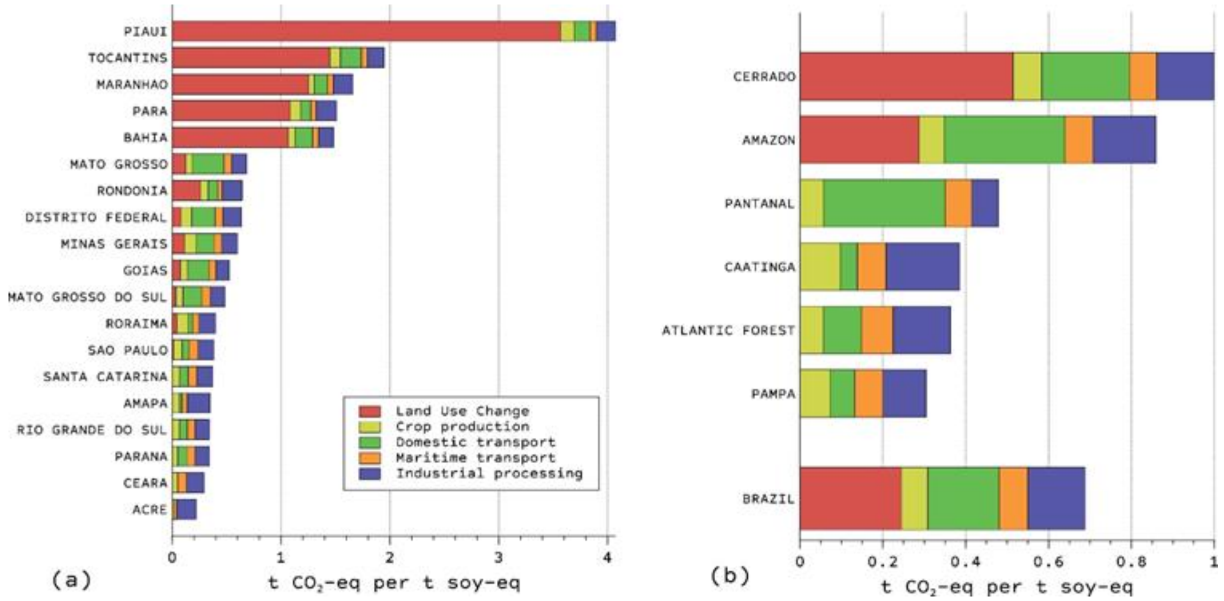
[22] While the SEI-PCS approach is spatially explicit by construction, it is also possible to integrate spatial information into other approaches such as input-output models. Sun et al. (2020) combine the Exiobase input-output model with FAOSTAT data on crop and livestock production, and estimates of the spatial distribution of crops and livestock around the world. To allocate domestic production to exports versus domestic use, these spatial estimates are combined with data on global road coverage: areas with greater road density or quality are assumed to be more likely to export their production. The resulting estimates make it possible to show the spatial distribution of products consumed in specific countries, and their role in specific ‘hotspot’ areas. However, the underlying trade structure in the Exiobase model is not very detailed, and is for example not able to differentiate palm oil, soybean, and rapeseed trade flows. For a review of spatial information in input-output approaches, see Sun et al. (2019).

Figure 10. Mapping of Colombian coffee exports using the SEI-PCS approach



Note: The diagram shows a simplified overview of the supply chains for the 20 largest importers of Colombian coffee in 2014. Colours correspond to different importers.
Source: Godar et al. (2016).

Figure 11. Spatial variability in carbon footprints of Brazilian soy



Note: First panel shows carbon footprint for different exporting states; second panel shows carbon footprint for different biomes and for Brazil as a whole. All figures refer to the period 2010–2015.
Source: Escobar et al. (2020).

Zu Ermgassen et al. (2020) have used the same approach to map the supply chain of the Brazilian beef export sector.[23] By combining geospatial information on deforestation and pastures with municipal-level data on cattle production, they calculate deforestation risks of cattle production by municipality of origin. This results in an estimated cattle-associated deforestation risk of about 500 000 hectares per year, of which about 74 000 hectares per year were linked to exports. As with the findings for soy, the environmental risks of beef exports vary considerably by municipality.

The findings also highlight the importance of domestic consumption in understanding deforestation risks. More than 80% of Brazilian beef production is destined for domestic consumption rather than exports. In addition, export-oriented supply chains have lower relative deforestation risks: despite accounting for 17-18% of production, exports were linked to only about 13-14% of deforestation. Compared to supply chains oriented towards the domestic market, export-oriented production was more likely to originate from more established municipalities with higher agricultural GDP and less recent deforestation. However, as with all attributional studies, interpreting these findings causally is difficult. For example, between 2010 and 2017 exports grew by 30%, which may have caused a displacement of domestic-oriented production towards frontier regions. Similarly, over time soy and sugar cane production have expanded onto pasture, which in turn has expanded into forest.

While most of the publications so far have focused on deforestation and GHG emissions, other environmental impacts can also be considered. For example, Flach et al. (2016) apply the SEI-PCS approach to estimate virtual water flows embedded in Brazilian sugarcane and soy production, tracing these from municipalities in Brazil to countries of consumption, while Green et al. (2019) use the SEI-PCS approach to study the impact of international trade on biodiversity loss in the Brazilian Cerrado region. Their analysis of trade flows also shows the centrality of the Netherlands as a major importer of soybeans and exporter of animal feed, suggesting that the country could play an important role in limiting deforestation and reducing biodiversity losses in the Cerrado. Information on such hubs in global value chains could thus be a valuable addition to production- and consumption-based indicators.

Data and analysis provided by the Trase initiative cover a growing number of commodities and producing countries. Recent work also aims to identify the investors holding debt or equity in firms with high deforestation risks (Box 3).

Across the different commodities studied by Trase, a recurring finding is the heterogeneity of environmental impacts within a country: a minority of production regions is usually responsible for a large share of total environmental impacts (Trase, 2020). More than half of the deforestation risk associated with soy exports in Brazil in 2018 is concentrated in less than 1% of the more than 2 300 municipalities producing soy; for beef in Brazil, more than half of deforestation risks are found in 2% of the 2 800 municipalities where cattle are raised. For soy in Argentina, 2% of soy-producing departments similarly account for half the deforestation risk, while in Indonesian palm oil 6% of producer districts account for half of the deforestation.

4.4. Hybrid approaches

Several researchers have creatively combined the strengths of different approaches. For example, the IOTA (Input-Output Trade Analysis) approach developed by the Stockholm Environment Institute combines monetary input-output analysis with physical production and trade data on specific commodities (Croft et al., 2018). The approach starts from physical production data of individual commodities (e.g. using FAO

[23] The analysis traces cattle from more than 2 800 municipalities, via 152 slaughterhouses, 204 exporting firms, 3 400 importing firms, and 152 importing countries between 2015 and 2017. Exports accounted for about 18-19% of Brazilian cattle production during this period, although the export share varies by region. Out of the several hundred companies involved in the sector, three firms (JBS, Minerva, and Marfrig) account for more than 70% of exports; exporting firms also typically own and operate their own slaughterhouses.

data, as in the physical trade flows approaches of Kastner et al. (2014b) and the FABIO model, or using SEI-PCS/Trase data), and follows these as far as possible along the supply chain before using sector-level data from input-output tables to complete the consumption end of the supply chain. Croft et al. (2018) illustrate the approach by combining the SEI-PCS/Trase and input-output approaches (using the GTAP database) to link subnational information on Brazilian soy production to EU importing countries and to specific sectors within those countries. Croft et al. (2021) have applied a similar hybrid approach to develop an experimental statistic to measure overseas environmental impacts of UK consumption of key commodities in the context of the UK Government's 25 Year Environment Plan. Other approaches have also been explored to integrate spatial information in input-output analysis (Sun et al. 2019; 2020).

4.5. Consequential approaches

As with the Life Cycle Assessments discussed earlier, a distinction can be made between attributional and consequential trade-based approaches. All of the examples provided so far (input-output models, physical trade flows, and the SEI-PCS approach) are essentially capturing only patterns as they existed at some point in time; they do not necessarily provide a good answer to the question of how actions by producers, consumers, or governments will affect the outcomes. Interventions in one region, or with one set of buyers, may simply divert products to other buyers or may displace environmental impacts (Godar et al., 2016). Attributional approaches will provide a good approximation if there is "stickiness" in supply chains (dos Reis et al., 2020); but to the extent that supply chains can adapt, the conclusions from attributional approaches may be misleading. Consequential approaches (which take into account how actors adapt their behaviour) are then needed.

An example of a consequential approach is Yao et al. (2018)'s analysis of the economic and environmental consequences of the Brazilian soybean boom. While some authors have used the GTAP database as a multi-region input-output table, Yao et al. (2018) use the economic model of GTAP, which takes into account consumer and producer responses to changes in prices, as well as other economic mechanisms. This allows for a more accurate decomposition of the relative role of different drivers.

While the analysis of Yao et al. (2018) is focused on explaining the historical importance of various drivers, the same approach can be used for counterfactual simulations, e.g. to assess the impact of interventions. Busch et al. (2022) study how an EU import ban on high-deforestation palm oil from Indonesia would have affected deforestation in the country. Their approach combines a global trade model with a spatially explicit model of land use change in Indonesia to assess what would have happened if an import ban had been in place between 2000 and 2015. The results show that such a ban would have had only a limited effect on deforestation in Indonesia (a 1.6% reduction compared to actual deforestation over the same period). The impacts are small in part because more than half of European imports of high-deforestation palm oil would shift to other countries, further confirming the importance of distinguishing between attributional and consequential approaches. Another example of a consequential approach is the analysis by Valin et al. (2015) on land use changes caused by different scenarios for EU biofuels policies.

Cantele et al. (2021) review equilibrium models which try to capture linkages between economic and environmental systems.[24] Their analysis found 438 studies in the past decade, of which half had an economic focus, one quarter a focus on agriculture, forestry and other land use (AFOLU), and the remaining quarter focused on energy. Geographic coverage is often uneven: while e.g. China or the United States are included in many studies, there are notable gaps in Central America, Africa, the Middle East,

[24] Consequential approaches are typically *ex ante* and model-based. In principle it might be possible to also use *ex post* approaches, e.g. using careful econometric analysis to assess the net effects of past interventions. Given the complex feedback effects in international trade flows, it might be challenging to correctly identify the causal effects. Even where causality can be established, evidence on past interventions may be of limited use in assessing potential new interventions.

and Central Asia. Models also tend to rely on aggregation of countries into regions and of various production inputs and outputs into aggregates.

One particular challenge in modelling is that food systems are simultaneously shaped by global processes (e.g. economic growth and international trade, climate change) as well as highly local conditions (e.g. soil conditions, socio-economic context). While global drivers affect local outcomes, the reverse is true as well: local outcomes may in turn feed back to regional, national or global processes. Recent advances in modelling try to capture such "global-local-global" linkages. Hertel et al. (2019) review this emerging literature with a focus on land-use/land cover change.

As with attributional approaches, different modelling assumptions and underlying datasets may lead to different results in consequential approaches. Comparing different approaches can be useful, as it may highlight relative strengths and weaknesses of various models and assumptions. For example, Hertel et al. (2016) review predictions on long-term food demand, cropland use, and prices from different models, and trace the different outcomes back to the major areas of disagreement or uncertainty. Similarly, the Agricultural Model Intercomparison and Improvement Project (https://agmip.org/) is a global collaboration of scientists working to compare and improve different models related to agriculture and food systems. Modelling studies in this tradition rarely take a full life cycle view; integrating their insights into LCA approaches can thus be valuable.

4.6. Findings from trade-based approaches

The approaches discussed in this section can generate a wealth of insights. Information on environmental impacts embodied in trade flows can be used to construct consumption-based indicators, showing which countries are, for example, net importers or exporters of environmental impacts. The same data could also be used to assess the contribution of different products. Spatially explicit approaches provide an even greater level of detail. This wealth of information makes it difficult to distill simple lessons from trade-based approaches. However, two key findings stand out:

- First, international trade accounts for a significant share of environmental impacts related to food, although a majority of environmental impacts are caused by domestic flows. Hong et al. (2022) estimate that 27% of agricultural (direct and land use change) emissions are embodied in international trade. Pendrill et al. (2019) estimate that between 26% and 39% of deforestation-related emissions are embodied in international trade. Similarly, Zu Ermgassen et al. (2020) show that domestic demand for beef and cattle in Brazil accounts for more than 80% of production, and an even greater share of cattle-related deforestation.
- Second, spatially explicit approaches confirm the findings of Poore and Nemecek (2018) regarding heterogeneity among producers at a subnational level. This was clear, for example, in Escobar et al. (2020)'s analysis of GHG emissions for Brazilian soy, and is echoed in several other studies by the Trase project. One implication is that impacts are highly skewed, with a minority of total production often accounting for a large share of total impacts.

4.7. Strengths and limitations

Trade-based approaches try to map environmental impacts along food supply chains from where production takes place to where consumption takes place. It is notable that most of these approaches have been developed in just the last few years, with many interesting conceptual and analytical developments taking place.

An advantage of input-output approaches is their wide coverage, which in theory makes it possible to trace effects throughout all sectors of the economy. Some drawbacks include the relatively high level of aggregation, and the fact that input-output tables tend to be expressed in monetary values rather than physical quantities, which means the approach implicitly assumes that environmental damage is proportional to the monetary value of product flows. To overcome this, other approaches have used physical data on bilateral trade flows. The recent development of the Food and Agriculture Biomass Input-Output Model (FABIO) is an example of this approach.

Both the traditional monetary input-output approach and more recent physical versions tend to use national averages of environmental impacts and trace these along bilateral trade flows from production to consumption. By contrast, spatially explicit approaches following the SEI-PCS example “zoom in” on subnational heterogeneity. The SEI-PCS approach provides an interesting example of how disparate evidence sources can be combined. Studies often rely on a mix of subnational production statistics (obtained from government statistics offices), satellite data (e.g. on deforestation), and privately held information (e.g. detailed shipment records). Where public statistics are used, the approach usually combines data held by different agencies (e.g. customs data, information on slaughterhouses). This allows the SEI-PCS approach to link across different spatial scales and processes, and helps to uncover patterns of subnational heterogeneity, as well as key actors. Although the approach currently covers a limited number of commodities and countries, it is clear that the method can potentially be used more widely.

However, these three approaches are all attributional rather than consequential: they can provide an accounting of observed effects but they are not necessarily a good guide to predicting what will happen after an intervention. The fourth set of approaches try to model explicitly how consumption and production decisions and the resulting trade flows would change following an intervention. Recent developments here have tried to better link local and global processes.

Despite promising trends, there are currently two other limitations common to most trade-based approaches. First, most trade-based approaches (with the exception of MRIO) offer less information on domestic flows than on international flows. This is problematic because, as noted, trade-based approaches reveal that much of the environmental damage of food supply chains is related to domestic demand. Even in countries where trade accounts for most of production, domestic demand usually plays an important role. Yet while international trade leaves a documentary trail of customs and shipment records, domestic flows are harder to trace. Further developments of the SEI-PCS approach (e.g. using data on sanitary and disease controls) might be able to close this gap.

A second limitation is that even the most detailed trade-based approaches are still relatively coarse, and rely on some level of aggregation across producers. For example, the SEI-PCS approach as applied to Brazil allocates environmental impacts at the municipality level equally to all production sourced in that municipality. This is a major advance over earlier approaches which used national averages, and allows for important insights. However, this is still relatively coarse from the point of view of policy design. If several producers are active in the same municipality, an individual producer would have little incentive to improve the environmental impact of its own activities if all production from the municipality will be treated alike. Moreover, producers who did make an effort to improve their environmental impact could object to unfair treatment. Depending on the policy objective, a greater degree of producer-specific information may be warranted.

5. Conclusion

The downscaled estimates, Life Cycle Assessments, and trade-based approaches provide distinct but complementary lenses to study environmental impacts along food supply chains. Looking at the wide range of studies reviewed here, a few conclusions can be drawn regarding the environmental impacts themselves, regarding evidence gaps, and regarding strengths and weaknesses of the different methods.

5.1. Findings on environmental impacts along food supply chains

In terms of supply chain *stages*, it is clear that the vast majority of environmental impacts occur either through land use change or at the agricultural production stage. This is the case, for example, for greenhouse gas emissions and deforestation, water use, and acidification and eutrophication (Poore and Nemecek, 2018), although some recent work suggests a growing role for other supply chain stages in greenhouse gas emissions (Tubiello et al., 2021).

In terms of *products*, LCAs generally show a greater environmental footprint for ruminant products than for other livestock products, and a greater footprint for livestock products than plant-based products. This is true for greenhouse gas emissions, but generally also for other impact categories such as land use, scarcity-weighted water use, acidification, and eutrophication.

However, a third important finding is that there is a large *heterogeneity* among producers. Some methods (notably downscaling and traditional input-output analysis) are unable to capture this, but other studies show the importance of accounting for heterogeneity: Poore and Nemecek (2018) report that impacts are highly skewed, with 25% of global production often responsible for more than half of all environmental impacts, across a range of products and impact categories. Such skewed impacts are also found within countries; the Trase project reports that more than half of soy- and beef-related deforestation risk in Brazil is concentrated in 1-2% of municipalities, with similar findings for soy in Argentina, and 6% of Indonesian producer districts accounting for half of palm oil-related deforestation.

A fourth finding regards the role of *international trade*. The evidence shows that the share of food systems' environmental impacts linked to international trade is significant, but considerably less than half. Hong et al. (2022) show that 27% of all agriculture-related emissions (including direct emissions and land use change) are embodied in international trade; Pendrill et al. (2019) had earlier calculated that 26-39% of deforestation-related emissions are embodied in trade.

5.2. Evidence gaps

Not all *impacts* have been equally well studied. This is particularly the case for biodiversity and soil carbon, both of which are important for a proper assessment of the environmental impacts of food products. Greenhouse gas emissions, land use, and freshwater use are generally the most studied impacts.

In addition, it is clear that there remain major gaps in terms of coverage of *products*. Livestock and major commercial crops are generally well studied in LCAs, with other products (e.g. inland fisheries, goat, cassava, sorghum, millet) receiving much less attention. Approaches other than LCA can sometimes shed light on these other food products, but Halpern et al. (2019) nonetheless find that "underassessed" foods account for more than half of animal production in 76 countries, and more than 25% of total food in 40 countries.

This is reinforced by *geographic* blind spots, as there are notable gaps for LCAs conducted in Africa and Central Asia. Some of the trade-based approaches in principle use a global lens, although e.g. input-output analysis tends to use a fairly high level of aggregation for products, and sometimes geographies.

Moreover, given the evidence of heterogeneity, it is clear that existing information sources are not yet sufficiently *granular*. For example, downscaled estimates do not provide sufficient detail to understand producer-level performance, and input-output analysis traditionally relies on national averages of environmental impacts. Similarly, the French Agribalyse database, which brings together LCA estimates for 2 500 food products, currently only reports a single “generic” estimate per product, and hence cannot yet capture differences between producers.

5.3. Strengths and limitations of different approaches

The approaches reviewed here are complementary: each has strengths and weaknesses, and each is suitable for some purposes but not others.

The strength of the *downscaling* approach is that it provides relative magnitudes of environmental impacts (e.g. GHG emissions) for different stages of food supply chains, which is useful in prioritising efforts. By construction, however, the approach cannot illuminate impacts of products or individual producers.

LCAs are an essential tool, in part because they can rely on an increasingly harmonised approach codified in ISO standards and handbooks, complemented by widely-used databases and models. On the other hand, LCAs take a “bottom-up” approach, often providing in-depth assessments of a single product only. To understand system-wide impacts thus requires a creative re-analysis (as in Poore and Nemecek, 2018) and complementary approaches to answer questions which are harder to answer through LCA, such as to what extent imports of one country are linked to environmental impacts in another.

Several *trade-based* approaches map environmental impacts of food production through international trade to final consumption, with recent progress in modelling physical (as opposed to monetary) flows, and in shedding light on the structure of sub-national supply chains. Some studies explicitly model how consumers and producers adjust their behaviour in response to policy interventions and changing market conditions. These approaches make it possible to develop *consequential* assessments which can capture substitution and other feedback effects which would occur if policies are implemented on a large scale. This contrasts with the *attributional* (“snapshot”) assessments by other approaches. However, consequential approaches typically require many additional assumptions.

The coming years may see new developments combining the strengths of the different approaches outlined in this paper. The growing availability of earth observation data and other detailed spatial data suggests that future studies could include a much broader range of environmental (and social) impacts (Godar et al., 2016; Sun et al., 2019; Moran et al., 2020). As an example of the growing opportunities offered by earth observation data, on 2 May 2022 the environmental data company GHGSat announced that it had for the first time successfully measured methane emissions from cows using satellite data (GHGSat, 2022). In addition to earth observation data, data obtained directly from producers may be able to provide even more granular insights (Poore and Nemecek, 2018). Governments can also play an important role in enabling better analysis, e.g. by publishing detailed customs data and subnational production statistics (Moran et al., 2020).

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