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Accounting for the hidden costs of agrifood systems in data-scarce contexts

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Abstract

A number of studies have estimated the hidden environmental, social and health costs associated with global agrifood systems. The methods used, as well as the data required, vary considerably from category to category. However, all studies are based on the true cost accounting approach, and demonstrate that hidden costs of agrifood systems are considerable and that action is needed at global, national and local levels.

True cost accounting can facilitate improved decision-making by policymakers, businesses, farmers, investors and consumers. To apply the approach at country level, however, the methods developed must be downscaled and the data limitations overcome. This review goes through each cost category – environmental, social and health – and proposes approaches to deal with them.

Where data are not available or time is limited, methods combining secondary data are suggested. In some cases, the suggestion is to collaborate with research centres, especially those working on health impacts at national level.

In appraising policies and measures, account must be taken of impacts in all categories. This review identifies these and indicates which ones need special attention. A key potential trade-off is that of the increase in the cost of food. However, there are combinations of policies that can avoid, or at least limit, these consequences.

As true cost accounting expands, knowledge of what works and what does not will grow and the advice given on conducting future policy assessments will improve.

Keywords: True cost accounting, hidden costs, agrifood systems, decision-making, trade-offs, food prices

JEL codes: C81, M41, M48, O57, Q18.

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1 Introduction to true cost accounting

1.1 Definitions

As the Food and Agriculture Organization of the United Nations (FAO) says in Chapter 1 of *The State of Food and Agriculture 2023*, true cost accounting (TCA) can be defined as: "a holistic and systemic approach to measure and value the positive and negative environmental, social, health and economic costs and benefits generated by agrifood systems to facilitate improved decisions by policymakers, businesses, farmers, investors and consumers" (UNEP *et al.*, 2021).

It is worth emphasizing that this approach aims to use TCA to improve decision-making, so the way in which costs and benefits are measured and reported should serve that purpose. It is also worth noting that the uses of the approach vary depending on which costs and benefits are covered, how they are valued (in monetary or non-monetary terms) and how the information is used for policy advice.

The approach incorporates several important economic concepts. The first is externalities, defined as: "a positive or negative consequence of an economic activity or transaction that affects other parties without this being reflected in the price of the goods or services transacted" (TEEB, 2018).

Examples of externalities in agrifood systems include, among other things, releases of pesticides from farming which contaminate land and water bodies, emissions of greenhouse gas (GHG) into the atmosphere and emissions of local pollutants from the burning of agricultural waste.

TCA also includes the impacts of food production and consumption that are not strictly externalities under the above definition, but which nevertheless result in health and social costs.[1] The consumption of a diet high in fat and meat would be an example. It involves transactions that do not affect other parties, but which damage the health of the consumer and come at a cost to public health systems. Such goods or services are referred to in the literature as "demerit goods" (Musgrave, 1987).

In policymaking terms, governments can undertake various measures to discourage them, ranging from awareness campaigns to, in some recent cases, taxes. The distinction between externalities and demerit goods is important, as the policy prescriptions for the two can vary; there is less agreement on regulatory or fiscal actions to limit the consumption of demerit goods than there is for externalities.

The policy implications of externalities depend to a significant extent on how many other parties are affected by the economic activity or transaction. If the number is small, bargaining between the parties can resolve the externality, so no government action is necessary beyond ensuring that legal frameworks are respected (Coase, 1960). When, however, the number of parties affected is large (for example, as in the case of air pollution), such negotiations are not possible on a private basis and government action is needed. The term often used to refer to an externality that affects a large number of parties is a "public bad", while reducing it is a "public

[1] Some analysts extend the definition of externalities to include all demerit goods and social costs. The Food and Land Use Coalition (FOLU, 2019) study, for example, does this and acknowledges that it is using a wider definition than is strictly correct. Given the aim to link the TCA structure to the policy framework, it is better to keep them separate.

good”. The extreme example of a public bad is GHG. In this case, international policy action is required, as the effects of emissions on any one country will not be enough for it to justify unilateral action.

The term used in *The State of Food and Agriculture* report to describe the costs arising from externalities, demerit goods and public “bads”, as well as other policy and institutional failures, is “hidden costs”. Note that these costs are “hidden” in the sense that they do not appear in the accounts of the provider of the good or service that created them. They are, however borne by other parties in society.

2 Estimates of the hidden costs of agrifood systems

A number of studies have estimated various components of hidden costs associated with agrifood systems (FOLU, 2019; Hendriks *et al.*, 2023). Table 1 gives the figures from three major global estimates under three category headings: environmental, health and economic/social.

The Food and Land Use Coalition (FOLU) comes up with a total annual cost of USD 11.8 trillion, of which 26 percent is environmental, 56 percent is health and 18 percent is economic/social. The United Nations Food Systems Summit (UNFSS) study group estimate is USD 19 trillion, made up of 37 percent environmental costs, 61 percent health costs and 4 percent economic/social costs (Hendriks *et al.*, 2023). The latter also provides uncertainty estimates, suggesting that the total could be between USD 7.2 trillion and USD 51.8 trillion. The major source of uncertainty is health costs, where the lower bound is one-tenth of the upper bound. Both sources acknowledge that not all cost items are covered; interestingly, a number of items missing in one study are covered by the other, and vice versa.

The third and most recent study is one undertaken for FAO by the Oxford Environmental Change Institute (Lord, 2023). It has a lower estimate of USD 6.3 trillion, of which 43 percent is environmental, 33 percent is health-related, and 23 percent is economic/social. Uncertainty ranges corresponding to 5 percent and 95 percent of the distribution are also provided; they are greatest for the environmental category and smallest for the social category. The study covers fewer sources of impact than the other two studies (especially FOLU). It does, however, provide estimates at national level for 153 countries, of great help when starting the process of designing policy responses.

One category that is not addressed in the three studies is plastics. The true cost estimate of this source of pollution is put at USD 75 billion a year, with food companies and restaurants accounting for 24 percent of that (UNEP, 2014).

Table 1. Global annual hidden costs of agrifood systems

Study	Item	Mean	Range		As % of market prices	Coverage	Not covered
		(USD billion)			(%)		
Hendriks *et al.* (2023)	Environmental	7 000	4 000	11 000	122	GHG, water pollution from N and P, air pollution NH_3, land degradation	Soil degradation, land use other than cropland, other air pollutants
	Health	11 000	3 000	39 000	433	Mortality from unhealthy diets (undernutrition and obesity)	Antibiotic resistance, zoonoses, undernutrition, productivity losses due to disease
	Economic/ social	1 000	200	1 800	20	Morbidity from unhealthy diets	Rural poverty, food waste, fertilizer leakage
FOLU (2019)	Environmental	3 100	Not given		31	GHG, land degradation, water scarcity, loss of pollinators, overfishing	Air pollution (NH_3)
	Health	6 600	Not given		66	Obesity, undernutrition, air pollution, pesticide exposure, antimicrobial resistance (AMR)	Zooneses, productivity losses
	Economic/ social	2 100	Not given		21	Rural poverty, food waste, fertilizer leakage	Morbidity from unhealthy diets, environmental effects of fertilizer leakage
Lord (2023)	Environmental	3 470	1 550	6 680	66	GHG, water scarcity/quality, conversion of forest/grassland to cropland and pasture, nitrogen (NH_3)	Ambient air pollution, loss of pollinators, overfishing, AMR
	Health	2 660	2 360	3 000	50	Undernutrition, poor diets	Morbidity costs, zooneses
	Economic/ social	1 880	1 760	2 000	36	Rural poverty	Food waste, fertilizer leakage, morbidity from unhealthy diets

Sources: Hendriks, S., de Groot Ruiz, A., Herrero Acosta, M., Baumers, H., Galgani, P., Mason-D'Croz, D. *et al.* 2023. The True Cost of Food: A Preliminary Assessment. In: J. von Braun, K. Afsana, L.O. Fresco & M.H.A. Hassan, eds. *Science and Innovations for Food Systems Transformation*. Cham, Switzerland, Springer. https://doi.org/10.1007/978-3-031-15703-5_32; FOLU. 2019*. Growing Better: Ten Critical Transitions to Transform Food and Land Use*. London. https://www.foodandlandusecoalition.org/global-report; Lord, S. 2023. *Trends in external costs of the global food system from 2016 to 2023 – Background paper for The State of Food and Agriculture 2023.* FAO Agricultural Development Economics Technical Study, No. 31. Rome, FAO.

2.1 Methods for estimating hidden costs

The methods used to calculate the hidden costs of agrifood systems, as well as the data required, vary considerably from category to category. Methods for estimating the different categories (in monetary terms) – as set out in the TEEB agrifood report – can be described as follows (TEEB, 2018):

- For **environmental** costs, the different environmental burdens are first estimated in physical terms, then the losses they generate to different ecosystem services are valued.
- For **health** costs, the physical impacts of agrifood systems on mortality and morbidity are first estimated, then the physical impacts are valued in monetary terms.
- **Social** and **produced** costs are derived from the market value of losses (food waste, fertilizer leakage and productivity losses due to the prevalence of undernourishment), as well as the expenditures required to close the poverty gap.

The TEEB report and the TCA Agrifood Handbook list the main estimating techniques as (TEEB, 2018): [2]

a. **Market-based methods**, where the environmental impacts cause a measurable loss of agricultural output or socioeconomic impacts cause a loss of commodities with a market value. All the economic components of costs such as food waste, fertilizer leakage and productivity losses are estimated using market-based methods.

b. **Cost-based methods**, which estimate the additional costs imposed by the damage to ecosystems on agrifood systems, the costs of treating the effects of poor diets, or the effects of air pollution on health. These are used as a proxy for actual damage. However, to evaluate whether the abatement is justified or not, a comparison of abatement costs and actual damages avoided is needed.

c. **Revealed preference methods**, where data on the value of services across different landscapes are compared to derive an estimate of how much the value varies according to landscape characteristics. In this way, the degradation of a characteristic such as soil quality can be valued. For example, data on the price of land with different characteristics, including soil quality, can be analysed using statistical methods to determine how much price varies with this factor (other things being held constant). These methods can also be used to estimate costs arising from the risks posed by environmental pollution (for example, expenditure to protect against environmental impacts).

d. **Stated preference methods**, where individuals who have suffered losses of service from ecosystems are asked to estimate the loss in value they have experienced.

While each method has advantages and limitations, most analysts would contend that estimates based on market data (or, failing that, cost data) are those mostly widely accepted by governments for policy purposes. This is because they are seen as being based on real transactions as opposed to modelling approaches. However, it is not always possible to use a market- or cost-based approach and, in some cases, information that is at least as good can be obtained from revealed preference methods or stated preference methods. Examples

[2] Trucost, a company that estimates the hidden costs of companies' unsustainable use of natural resources, uses methods set out in TEEB's TCA framework (available at https://www.spglobal.com/esg/trucost). Trucost's estimates of hidden costs are discussed for water scarcity and plastics.

include estimates of the costs to water quality of the recreational use of water bodies and estimates of the costs of changing diets by altering behaviour. Table 2 shows the different methods and links them to the categories of impact that correspond to the four capitals: natural, human, social and produced.

Table 3 shows possible measures for obtaining estimates for each cost category. It also presents the advantages and limitations of the methods used. Methods for estimating the hidden costs and benefits of the different categories are discussed in later sections, with a focus on country-level estimates and analysis, especially in data-scarce countries.

Table 2. Methods for estimating the hidden costs of agrifood systems

Method	Area of impact		
	Environment	**Health/human**	**Produced**
Market-based	Air pollution on crops Land degradation Loss of pollinators Social cost of carbon for GHG Lost productivity due to antimicrobial resistance (AMR)	Value of disability-adjusted life years (DALYs) for poor diets and undernutrition Value of premature mortality for air pollution, pesticide poisoning, AMR Productivity losses from poor diets	Value of food wasted Value of fertilizer leakage
Cost-based	Costs of restoring fisheries Costs of GHG abatement Costs of clean-up of water pollution Costs of replacing sources of land productivity Cost of replacing natural pollination	Morbidity costs of air pollution Morbidity costs of poor diets Morbidity costs of pesticides Morbidity costs of AMR	Expenditure needed to close poverty gap
Revealed preference **Stated preference**	Damage from water scarcity to ecosystem services Fertilizer/pesticide releases on water quality Impacts of water quality on recreational use of water bodies Disamenity from plastic pollution Costs of plastic pollution to marine life	Value of premature mortality air pollution, pesticide poisoning, AMR	

Source: Author's own elaboration.

Table 3. Estimation of hidden costs: methods, advantages and limitations

Source of hidden cost	Methods used	Advantages	Limitations
GHG emissions	Average costs of abating emissions over period or social cost of carbon (SCC)	Abatement cost data are easy to obtain; SCC is an available global estimate	Not clear which costs to take; choice depends on policy being considered
Water pollution	Revealed and stated preferences or costs of clean up	Clean-up costs are easy to obtain; revealed and stated preference needed for loss of recreational benefits	Clean-up costs do not allow evaluation of whether clean-up is appropriate; other studies are difficult to conduct
Air pollution effects on crops	Market value of loss of yields	Methods give widely accepted results	Estimating loss of yields using production functions is complex
Land degradation	Market value of loss of yields; cost of replacing inputs and revealed and stated preference	Data on costs of replacing inputs are easy to obtain; revealed and stated preference needed for wider loss of ecosystem services	Estimating loss of yields is complex; revealed and stated preference are difficult to conduct; replacement costs cannot evaluate whether replacement is appropriate
Loss of pollinators	Market value of loss of output; replacement cost	Replacement cost method is relatively easy to apply	Does not allow evaluation of decision on whether replacement is appropriate; market value loss estimation difficult to conduct
Overfishing	Market value of future losses due to unsustainable fishing or costs of restoring stocks to sustainable levels	Costs of restoring stocks to sustainable levels are relatively easy to estimate	Estimation of sustainable fish yields and future losses is complex
Water scarcity	Losses to ecosystems due to withdrawal for agriculture using revealed and stated preference	Data have been calibrated to make the estimates based on extent of scarcity	Suitability of calibration for specific situations is not clear
Unhealthy diets	Loss of disability-adjusted life years (DALYs) and costs of treatment; DALYs valued using a mix of market-based and other methods	Historical estimates of losses and value of DALYs available by country from IHME and other researchers	Future projections will require complex modelling
Undernutrition **Air pollution effects on health**	Loss of DALYs and excess mortality valued using a mix of market-based and other methods	Historical estimates available by country from IHME and other researchers	Future projections will require complex modelling; some figures available from the World Health Organization (WHO)
Antimicrobial resistance (AMR)	Loss of productivity, excess mortality and morbidity costs	Methods for loss of productivity established	Estimation of other costs requires further modelling, which is complex

Source of hidden cost	Methods used	Advantages	Limitations
Rural poverty	Revenues needed to bring all agricultural workers above the poverty line	Simple to calculate	Rural poverty covers more than this group
Food waste	Value of food lost; changes in GHG	Simple to calculate	Average values could be misleading
Fertilizer leakage	Value of fertilizers leaked; loss of ecosystem services as a result	Value of lost fertilizers simple to calculate; loss of ecosystem services needs revealed and stated preference	Calculating loss of ecosystem services is complex

Source: Author's own elaboration.

A more detailed discussion of the component costs is given in Annex 1, which covers the following 14 cost sources: GHG emissions. water pollution, land degradation, loss of pollinators, overfishing, water scarcity, unhealthy diets, undernutrition, air pollution and health, antimicrobial resistance (AMR), pesticide pollution, plastic pollution, food waste and fertilizer leakage. It also discusses issues arising in the estimation of each and examines cross-category factors in the treatment of uncertainty and the use of tools to pick up cross-sectoral impacts and spillovers.

Section 3 explores the policy implications of the gap between actual and true costs, discusses how the gap might be estimated in data-scarce contexts and looks at what policy options are available to close the gap.

3 Policy implications of a gap between the actual and true costs of agrifood systems

3.1 Implications for the price of food

Does the gap between the actual and true cost of agrifood systems imply that the price of food should rise to close the gap? Not necessarily. It depends on what instruments are used to close the gap. Broadly, there are two principles for addressing externalities: the polluter pays principle (PPP) and the beneficiary pays principle (BPP). In addition, if the programme to close the gap is undertaken at the same time as measures are introduced to increase efficiency in agriculture and raise productivity (for which there is considerable potential) (Damania *et al.*, 2022), prices increases can be avoided.

Under the PPP, the costs of achieving the desired environmental outcomes should be borne by those responsible for creating the environmental burdens (OECD, 1975). This can be done using administrative regulations that stipulate less environmentally harmful farming practices or market-based instruments, such a tax or charge on the polluter. It can also be achieved through the creation of markets for rights to pollute or to gain access to open-access resources, such as fisheries.

There are many cases where governments enforce regulations in the agricultural sector. There are fewer examples of the use of market-based instruments in this sector, but there are some. A number of Organisation for Economic Co-operation and Development (OECD) countries have imposed levies on pesticides and fertilizers and taxes on emissions of pesticides to water (Barbier and Markandya, 2012). In developing countries, examples include fishing licences sold in accordance with sustainable catch limits in countries such as Uganda, Namibia and the United Republic of Tanzania; taxes on organic discharges of biological oxygen demand and total suspended solids in Colombia; charges on water effluent in China; and a fee on palm-oil effluent in water in Malaysia. The introduction of administrative or market-based measures, when applied to the agrifood system, will normally raise the price of food unless is it accompanied by actions to support farmers, such as advice on better management practices.

The alternative to applying the PPP is to place the burden of ensuring that the true cost of an agrifood activity is covered by the beneficiaries. In such cases, the policies should not result in an increase in the price of food. The beneficiary is usually the public, which benefits from the closure of the gap, but it can also be specific groups who bear a higher cost for their activities on account of the pollution generated by farming and related activities. A simple application of the principle would be to support and even subsidize the adoption of cleaner and less polluting practices. Support through agricultural extensions services is widespread, but does not always focus on reducing external costs. Subsidies for reduced environmental harm in OECD countries include tax discounts when investing in pollution abatement, subsidies for nature conservation, subsidies for environmental elements in agriculture and subsidies for water-saving devices (OECD, 2023).

In principle, there is no reason why such schemes should not be adopted. They are not inferior to those undertaken on the PPP, although some subsidy schemes can be subject to misuse and result in fiscal costs that are unaffordable. The difference between the two is largely a matter of equity, which can favour either the PPP or the BPP, depending on who the polluters and beneficiaries are. In OECD countries, the presumption is that PPP will be applied, but there are exceptions, and the use of environmental subsidies to promote land-use changes that lower the hidden costs of land use is part of the toolbox. In developing countries, PPP is less widely applied for fear of raising prices. When choosing a policy instrument to reduce hidden costs, governments need to analyse carefully what the distributional implications will be. They must also consider the fact that subsidy-based schemes place a burden on scarce fiscal resources, so the BPP should be used with this in mind.

An important set of measures that puts the cost of reducing the environmental cost of agrifood and other land-based activities on the BPP is payments for environmental services (PES). In contrast to the PPP, the idea here is that the beneficiary pays the parties whose activities are damaging the environment to modify their behaviour. A simple example would be a river basin, where the downstream area is highly urbanized and relatively wealthy, and the upstream area is rural and relatively poor. Farming practices upstream damage the source of water supply downstream and both parties can gain if the upstream farmers are paid to adopt less polluting agricultural methods.

In principle, there is no reason why such schemes should not be adopted, and they are in no way inferior to those adopted on the PPP. The difference between the two is a matter of equity, which can favour either PES or the PPP, depending on who the polluters and the beneficiaries are. The main PES schemes relevant to agrifood systems are watershed protection, biodiversity conservation, carbon sequestration, and landscape and beauty services. Surveys of those that have been implemented indicate that while they can work successfully, difficulties arise when schemes are driven more by government aims and objectives and less by local needs (Pagiola *et al.*, 2004; Pagiola, Arcenas and Platais, 2005). In such cases, payments often do not guarantee environmental improvements despite large outlays. This can be avoided by making sure that schemes are based on the full participation of all relevant parties and proper account is taken of how providers will respond to the incentives offered.

One set of policies that can involve a mixture of PPP and BPP is the repurposing of agricultural subsidies. The removal of some output-based subsidies for production that is environmentally harmful might be seen as a move towards the PPP, while the introduction of a new environmentally friendly subsidy could be considered an application of the BPP.

As a recent World Resources Institute report notes, current agricultural subsidies are provided in a way that often rewards unsustainable land use and production (Ding *et al.*, 2021). Globally, governments spent more than USD 708 billion (USD 619 billion in net transfers) a year on agricultural subsidies from 2017 to 2019. However, the costs of deforestation and land degradation could be nine times that, at USD 6.3 trillion a year, in terms of lost ecosystem services. These include, but are not limited to, agricultural productivity, the provision of clean air and freshwater, and the regulation of the climate.

The WRI report shows that restoration practices can improve soil health and lead to a global average increase in crop yields of 2 percent by 2050 compared with a baseline scenario, with a significant rise in agricultural productivity. Thus, by shifting underperforming agricultural subsidies to protecting and restoring degraded farmland, governments can better support local communities and help achieve their countries' climate, biodiversity and rural development goals. It is unclear, however, to what extent the costs of such policies fall on current polluters (who lose their subsidies) or on beneficiaries (those who benefit from the gains in biodiversity, rural development and on). Some case studies in the WRI study suggest that the repurposing can be designed in a such way as to avoid losses to small landholders.

To conclude, eliminating hidden costs without raising the price of food is possible, but it requires the careful application of the BPP, rather than the PPP. It will also be easier to achieve if the programme of moving to a true cost agrifood system is conducted in parallel with measures to increase efficiency and productivity in farming, bearing in mind those true costs.

3.2 Compromises and trade-offs: closing the gap to other policy objectives

The estimation of the true costs of agrifood systems has focused on individual sources of cost. Yet, when policies to reduce such costs are being considered, they need to take into account their impacts on a range of categories. Trade-offs are most frequent between the environmental and economic categories of cost. For example, measures to reduce the environmental footprint of production and distribution (such as adopting alternate wetting and drying for rice cultivation) could increase the cost of food initially and have an impact on poverty and health. A subsidy for fertilizers in poor areas with very low applications, meanwhile, would increase productivity, but could cause environmental harm.

Another case of trade-offs between environmental categories is the adoption of irrigation schemes to address water scarcity, which increase GHG emissions. A more complex example is a subsidy for livestock that lowers production costs. This is considered harmful to the environment, as it increases meat production and, hence, GHG emissions, as well as other environmental pressures. If, however, it was coupled with support for livestock production subject to environmental conditions, such as a maximum animal stocking rate per hectare, or orientated to maintain grazing and pasturing activity, it could generate environmental benefits compared with the counterfactual.

3.2.1 Life-cycle assessment for analysing trade-offs

Trade-offs such as these require the use of more complex tools, such as life-cycle assessment (LCA), to analyse their impacts on different areas over the lifetime of the emissions, as well as economic tools to track changes in behaviour resulting from the policy and their economic effects. The economic toolbox includes partial and general equilibrium models. Data-scarce countries may be limited in their capacity to use such models to assess trade-offs, but they can usually look at the effects of policies on one sector (for example, restrictions in some agricultural practices, such as residue burning) and on the environment (for example, air quality) and assess what the trade-offs will be.

In general, where there are a number of policy objectives, there may be compromise, but the extent of the compromise will be minimized if there are at least as many policy instruments as there are objectives (sometimes referred to as the Tinbergen Rule) (Schaeffer, 2019). Hence, a policy package that allows the different objectives to be addressed is desirable. So, for example, if a country seeks to restore fish stocks, but also to address rural poverty, imposing a blanket restriction on catch alone could create an increase in poverty in the artisanal fisher community. Introducing a second instrument, however, such as income support or alternative employment opportunities (or an exemption for small fishers), could allow both objectives to be met.

3.3 Cost-benefit analysis for better decision-making in agrifood systems

Policies to address the gap between the actual and true cost of agrifood systems involve a consideration of the costs involved, as well as the gains made by implementing the policies.

The general framework for doing this is cost-benefit analysis (CBA), which is mandated by many governments and institutions (either directly or as a partial input to a regulatory impact analysis) before a programme or policy can be improved (OECD, 2020).

The aim of the CBA is to account for all costs and benefits of any investment or policy over the foreseeable future, so that an assessment can be made as to where the benefits exceed the costs. Guidelines on the application of CBA to cover environmental, health and social policies are available (HM Treasury, 2022).

It is worth bearing several points in mind when applying CBA to policies related to agrifood systems. First, while the cost figures presented in the global studies are useful, they should not be used as a target for cost reductions. This is because the costs of totally closing the gap between market costs and true costs will be prohibitively high. By way of example, the amount of food waste globally is estimated at 1.6 billion tonnes. No one imagines that actions can be taken to reduce that to zero. From an economic perspective, the aim would be to make reductions to the point where the costs of another small reduction were equal to the additional costs involved.

The problem, of course, is identifying such a level. To this end, a programme can be designed to make a reduction of, say, 20 percent in environmental or other burdens and estimate the costs associated with that reduction. The benefits of a 20 percent reduction, however, will not necessarily be 20 percent of the total cost estimated. For the economic costs of rural poverty or food waste, a proportional cut may apply, but where reductions in emissions to air and water are concerned, some modelling will be required to estimate the cost. Another example is land remediation, where the gains from remediating a percentage of the degraded land will give greater or smaller benefits than that percentage, depending on which parcels of land are selected. The same holds for the aforementioned health impacts.

The models required will vary according to impact and are discussed further below. As noted, they will need to take into account spillovers from the implementation of the policy, so changes in emissions and social and health impacts generated by the policy will need to be tracked. Not all of these spillovers can be measured in monetary terms, however. They will need to be recorded alongside the monetary costs and benefits, resulting in a review of the policy that covers more than the estimated monetized information.

An important area where information on non-monetary indicators complements that on the costs and benefits of interventions is thresholds for the functioning of ecosystems (for example, a water body that becomes unsafe for contact recreational use, or a rate of extraction of non-timber forest products that is unsustainable). If a particular measure exceeds the threshold value for such indicators, this should be recorded and taken into account when choosing what action to take. Other physical indicators that policymakers may wish to consider are the number of jobs created or the number of lives saved by the measure. These should be provided along with information on the monetary values of costs and benefits.

Lastly, when appraising policies in the area in question, it may not always be appropriate to use CBA. An alternative could be to look at the cost-effectiveness of a range of measures. While CBA compares the benefits and costs of different interventions, a cost-effectiveness approach compares the costs of meeting a given physical improvement in emissions or services provided. This is particularly relevant when examining options for reducing GHG from agrifood systems. The direct costs of the options can be relatively easily established, as can the reductions in GHG achieved. In some cases, there may be indirect costs or co-benefits (such as reductions in local air pollutants). These can be subtracted from the direct costs (in the case of co-benefits) or added (in the case of indirect costs) to give a net cost figure. Governments will then select the options with the lowest cost per tonne of GHG abated for implementation.

4 Applications of true cost accounting in practice

Estimates of the true cost of agrifood systems at global level need to be scaled down to national level. This section discusses options for doing so. Each of the areas covered at global level is reviewed to see what methods can be used, especially when data are scarce, and how they are linked to the methods set out in Figure 1 and Table 2. In the upcoming sections, a decision tree is proposed for each category of impact and for different policy issues that need to be addressed.

4.1 Measuring hidden costs in data-scarce contexts: the role of benefit transfer

Before going into detail on the options available for undertaking studies, it is useful to consider the role of benefit transfer (BT), also referred to in the literature as value transfer (VT). BT is defined as the use of research results from pre-existing studies at one or more sites for a range of time periods to predict value estimates for other sites and time periods. The transferred results can come from any of the four methods described in Section 2.1. Studies of the values of ecosystem or health benefits are location and time specific and usually do not cover all locations in a country. In cases lacking the time or resources to conduct primary valuation, it is common to use BT methods.

BT can be based on a single unit transfer – taking the estimate from one site or country and applying it to another – or on some adjustment to the value being transferred to account for differences between sites (for example, in per capita income, as discussed below). A more sophisticated form of BT is meta-analysis, in which data from several value studies are used to estimate a function that determines how much the value depends on site and population characteristics (Walton, Boyd and Markandya, 2014). All methods have been used to form estimates of the external costs of pollution. While it cannot be said with certainty that one method is better than another, in general, the more information that can be used to adjust the transferred value for differences between the value site and the site where it is being applied, the better.

Two common adjustments that are made in a transfer involve accounting for differences in the population exposed and the per capita income of the two populations. In the first case, if the estimates are not in per capita terms, it is relatively simple to adjust for the difference by taking the ratio of the population in the value site to that of the application site. Adjustments for per capita income can be done in a similar way, multiplying the value site estimate by the ratio of income per capita in the application site to that of the value site. A variant of this has been developed by the OECD particularly for estimates of health costs (OECD, 2012):

$$V_P = V_S\,(Y_P/Y_S)^e$$

where Y is purchasing power-adjusted income per capita, V is the value estimate for a given service and *e* is income elasticity. *P* is the site where the estimate is being transferred and *S* is the site where the estimate has been made. For costs of mortality, the OECD recommends an elasticity of 0.8.

When discussing options for estimating true costs, BT is frequently invoked as one of the less costly alternatives.

4.2 A decision tree for guiding policymaking at national level

This section presents a decision tree to assist policymakers in reducing the hidden costs of agrifood systems (Figure 1). Countries wishing to reduce the hidden costs of agrifood systems can best start by looking at the costs associated with the different sources of hidden costs. The data sources can be divided up as follows:

- **Global:** i) the Oxford Environmental Change Institute, which provides data for 153 countries on the environmental costs of GHG, land and water use, and nitrogen; health costs of dietary risks; and economic costs associated with rural poverty and undernourishment; ii) the Institute for Health Metrics and Evaluation (IHME), which provides data on health costs from air pollution;[3] and data from iii) FAO and the World Bank on fish stocks and associated costs of overfishing. All of these sources can be complemented by (sub)national data, where available.
- **Mix of global and local**: i) ministries of health and the World Health Organization (WHO) on the morbidity costs of poor diets, air pollution, pesticide poisoning and AMR; and information from ii) FAO and ministries of agriculture on food waste.
- **Local**: i) ministries of environment on plastic pollution and fertilizer leakage; and ii) ministries of economy/finance on rural poverty.

Not all data will be in monetary terms, but the ensemble will provide a useful picture of what the major challenges are with regard to TCA. The first part of Figure 1 lays out the data assembly stage. It goes on to refine the more aggregate estimates using national and subnational data.

The second stage is to establish the measures that can be taken to address the challenges in each category. More than one measure will normally be considered in each category, but governments may also take the view that, based on the initial screening, some sources of hidden costs do not need special attention at this stage. This may be because the data show that the costs are not that large, or because there simply is not enough information on the likely benefits to justify taking action to address them.

The third stage is to evaluate the effectiveness of the measures and make a selection. This involves estimating the costs of the measures, as well as their impacts in terms of reducing hidden costs. A combination of CBA and cost-effectiveness analysis is proposed in the discussion of the measures. Here, it is also important to recognize that actions taken with a focus on one category will often have impacts on others, and these impacts (positive or negative) have to be taken into account. Figure 2 (which complements Figure 1) depicts a matrix showing where the links are likely to be strongest. These spillover effects need to be considered in the evaluation. The links identified in the cells are based on the discussions in this report. Some merit further discussion:

  - Most measures to reduce GHG in the agrifood sector will not lower agricultural productivity and may even increase it, so the undernutrition impacts are unclear. However, some measures that focus specifically on increasing food production as a

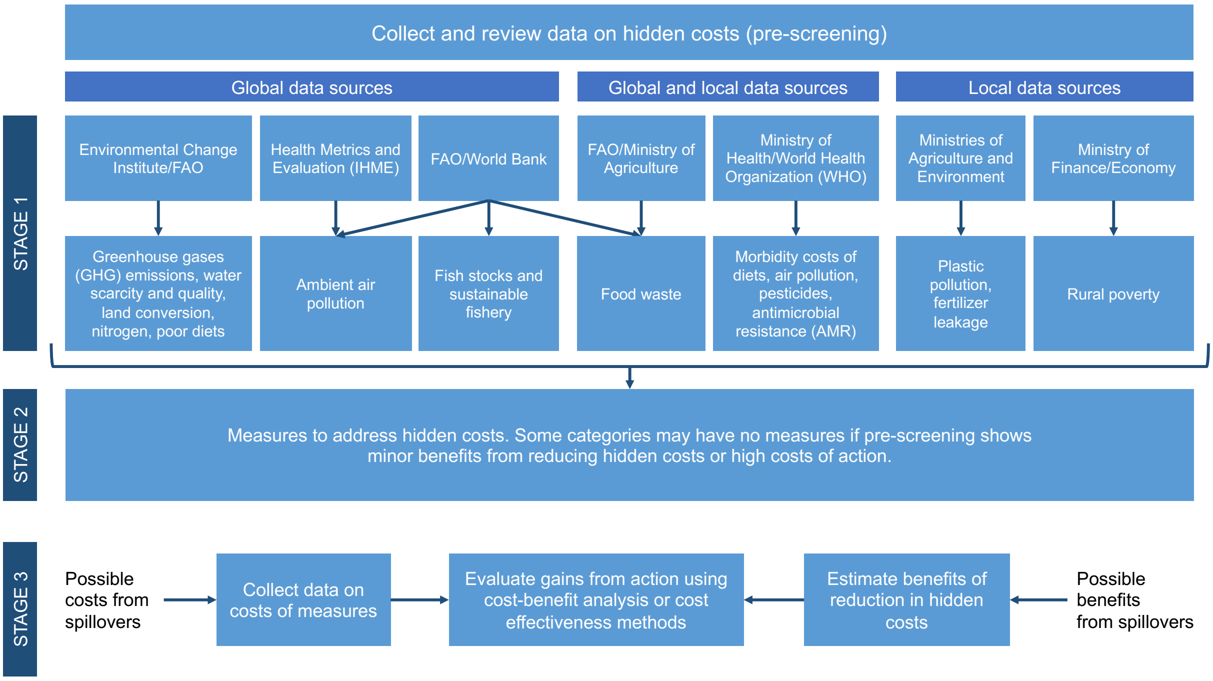
[3] With the share attributable to agrifood systems roughly calculated from their share of particulate matter and ozone emissions provided by ministries of agriculture and environment.

means of lowering undernutrition may increase GHG, as well as have an impact on costs from land, water and ammonia (NH_3).

- Measures to address poor diets will tend to reduce GHG, as they involve less meat, but measures focused on reducing GHG will not generally impact diets.

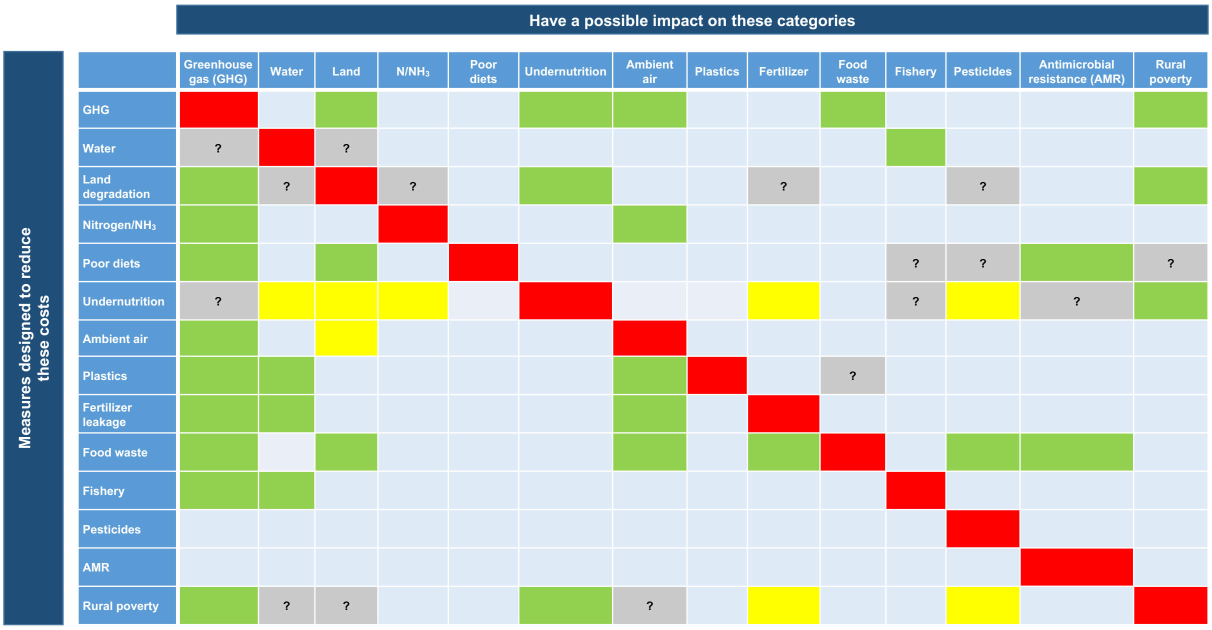
The remaining discussion looks at possible measures and ways in which data for each cost category can be collected in data-scarce countries.

Figure 1. Decision process for actions to address hidden costs in agrifood systems



Source: Author's own elaboration.

Figure 2. Spillover effects of measures taken to address one category of hidden costs on other categories



Notes: Legend is the following: red = main impact; yellow = likely negative impact unless specifically addressed; green = likely positive impact; ? = impact unclear. Cells in light blue indicate that no significant direct impact is expected from a reduction in the costs listed in the rows. Impacts are not always symmetrical. For example, lowering undernutrition may require an increase in GHG from expanding food production, but measures to lower GHG need not involve a decline in agricultural productivity.

Source: Author's own elaboration.

4.2.1 Greenhouse gas emissions

Costing GHG will arise in decisions about measures to be taken to reduce emissions as part of a national programme under the intended nationally determined contribution or as a cost saving from measures taken to address other pollutants, particularly through land management, air pollution, food waste, rural development and plastic pollution.

Actions targeted directly at reducing GHG normally calculate the cost of the reduction per tonne abated and compare this cost across different actions, some of which will be in the agrifood sector and some in other sectors. Governments then choose those that meet their target commitment at least cost. For this purpose, no direct estimate of the damage caused by emissions of GHG is required. There is a wide range of measures that can be taken to reduce GHG from agrifood systems. Most of them are part of the implementation of climate smart agriculture, which seeks the "triple" wins of higher productivity (and thereby higher incomes and less rural poverty), increased resilience to climate change and lower GHG (Sova *et al.*, 2018). A World Bank review finds very few cases where technologies that support adaptation to climate change and raise productivity do not also reduce GHG emissions (World Bank, 2015). The one area where action to reduce GHG could have negative impacts on land is incentives for producing biofuels. These could result in an increase in land under cultivation, possibly replacing land allocated for food, with negative impacts on nutrition and poverty.

Analysis of an action that reduces GHG can be reported in two ways. The first is to estimate all benefits and costs of the action in monetary terms, but to estimate the reduction in GHG in physical units. This includes the direct costs of the measure, but also any indirect costs or co-benefits. If there is an indirect cost, it should be added to the direct cost, and if there is a co-benefit, it should be subtracted.

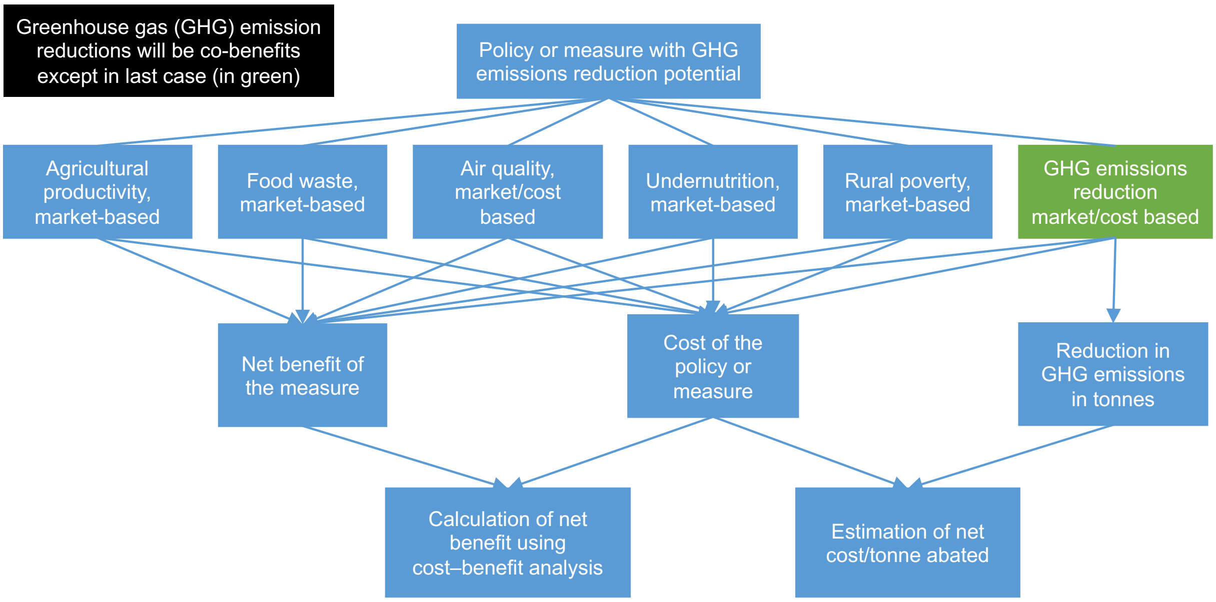
Figure 2 indicates that there will generally be co-benefits. These could include improved land productivity, less undernutrition and poverty, and (in the case of measures focused on waste) less food waste. An additional potential co-benefit of the measure is a reduction in local pollutants, such as $PM_{2.5}$. Thus, a value must be estimated for these reductions, based on methods described for each. Calculating the change in net costs of the action and dividing by the reduction in GHG will give the cost per tonne abated.

The second way is to include the benefits of the GHG reduction in estimating the net benefits of the action. These can be valued in terms of the social cost of carbon, its marginal abatement cost or a market price for the GHG where one is applicable for local reductions. All approaches have been used and the choice depends on what data are available. Abatement cost information is available from national agencies (ministries of environment or energy/climate change) dealing with climate change.[4] Market prices of GHG in the voluntary market are available from records of transactions if the country has engaged in them, otherwise, recent price data are available from the World Bank (World Bank, 2022b).

The two approaches are shown in Figure 3. Policies with potential reductions in GHG are divided into: (a) those where the reduction is a co-benefit of actions designed to address other hidden costs and (b) those where the focus is on GHG reduction itself. In Figure 3, all the blue cells in the second row relate to case (a) and the green cells to case (b). For policies involving case (a), the GHG emissions are included as a co-benefit and valued using the methods described above. For policies focused on GHG mitigation, the reductions are not valued, but any benefits the actions generate in terms of air quality and so on are credited to the action, and a unit net cost per tonne abated is calculated. The estimates from both methods will inform decisions on the set of abatement actions, as well as the selection of programmes as part of a response to increasing agricultural productivity and making agrifood systems more climate resilient. Figure 3 sets out the steps involved in the decision process.

[4] True Cost Initiative (2022) recommends the use of abatement cost, but UNEP (2014) report on plastics uses the social cost of carbon. The latter is an exception in the literature.

Figure 3. Decisions relating to greenhouse gas emission costs in policy



Source: Author's own elaboration.

4.2.2 Water pollution

Measures on water pollution can be the focus of attention when current quality results in high costs of treatment for water utilities or when the quality indicators do not meet standards set by the government. There can also be pressure to improve water quality at a watershed or beach to allow activities that are not currently feasible, such as boating or swimming. Co-benefits include possible benefits to fisheries, estimated in terms of increased market value (for commercial fisheries) and stated and revealed preferences (in the case of recreational fisheries). Impacts on GHG and terrestrial ecosystems are unclear and need site-specific investigation.

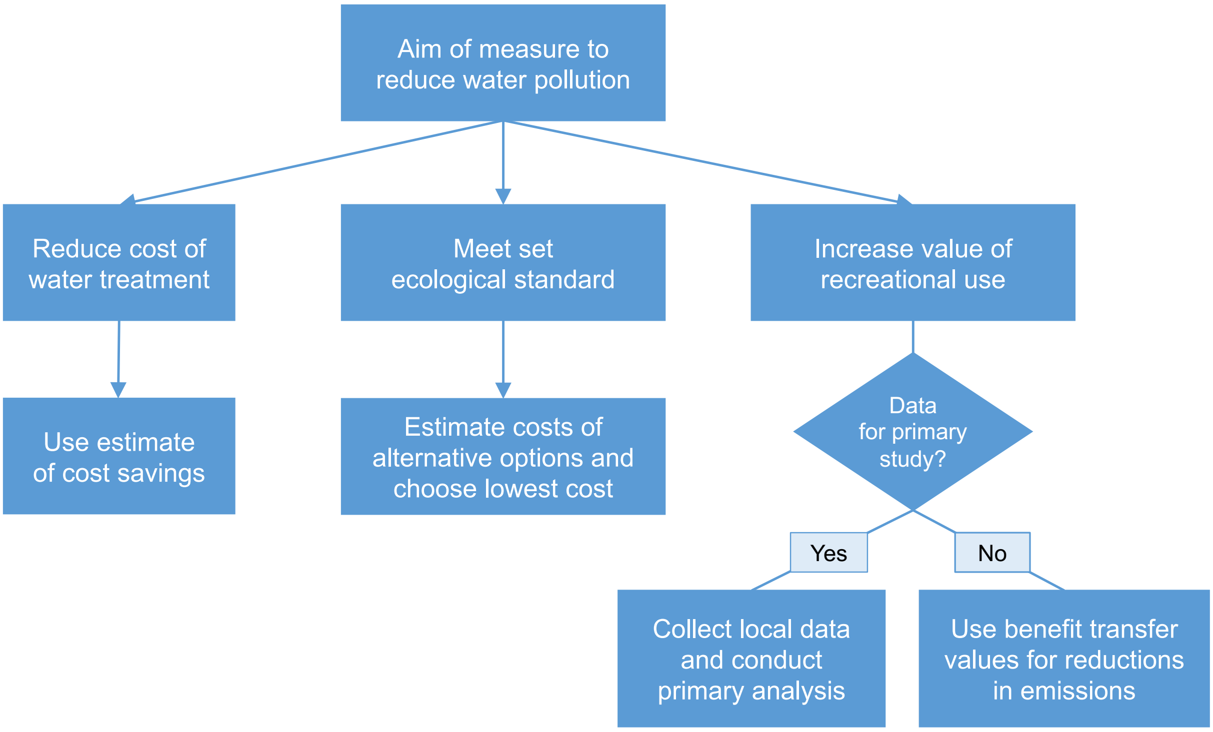
Where the focus is on higher treatment costs for water, the benefits of any improvement in quality are the savings generated by other parties using the water, such as water utilities. Estimates of these savings can be made by water engineers once the improvement in the quality of the water received is known.

Where wider quality issues are a concern, a watershed may be deemed highly polluted and below water-quality standards. In such a situation, farms will be required to take mitigation measures, but the measures to be taken, and the locations where they are most urgently needed will depend on the contribution of different sources to the quality indicators. This problem can be tackled by modelling the impacts of these contributions and introducing treatment standards that meet the quality indicators at least cost.

In the second case, the objective may be to decide whether an improvement that allows higher-value recreational use is justified. This requires an estimation of the benefits of increased recreational use. The recreational benefits are local and site specific. As one cannot conduct a study for every watershed, a "value function" has been derived to give the value as a function of water quality and other variables (in Annex 1, the case of the UK National Ecosystem Assessment is cited as an example). Unfortunately, such a function is not available for all countries. The first thing is to check whether national studies of the benefits of different uses of a water body are available. Some countries have such studies and can use them in conjunction with data for the watershed to obtain estimates relevant to that location (Markandya, 2006).

If that is not feasible due to data constraints, some wider BT may be possible. A Trucost plastics study used a meta-analysis BT function from an European Union study, which used a formula to estimate the monetary cost per kilogram in Europe of toxic substances deposited in natural ecosystems as a function of population density, type of ecosystem and size of ecosystem, as well as an estimate of the ecosystem damage potential (CE Delft, 2018; World Bank, 2021).This approach gives the values of different emissions to land and water in EUR/kg. These could be applied at a more local level, adjusting for differences between the country and the European Union by using the BT formula described above. Figure 4 sets out the steps involved in the decision-making process.

Figure 4. Decision-making in relation to water pollution



Source: Author's own elaboration.

The first decision is whether the water quality problem relates to costs of water treatment, to issues of not meeting ecological standards or to loss of recreational use value.[5] In the first case, estimates of the cost savings of treating drinking water by reducing water effluent will give the hidden cost. In the second case, alternative methods of achieving the required biophysical standards should be compared and the one with the lowest cost selected. In the last case, the benefits of increased use need to be compared with the costs of achieving it. If local data on benefits and costs are available, these are the preferred option. In the absence of such data, BT values, as described in Annex 1, can be used.

[5] There may be situations in which two or three of these apply. In such cases, the hidden costs will be the sum of those proposed, taking into account any cross benefits of action on one criterion on the others.

4.2.3 Land degradation

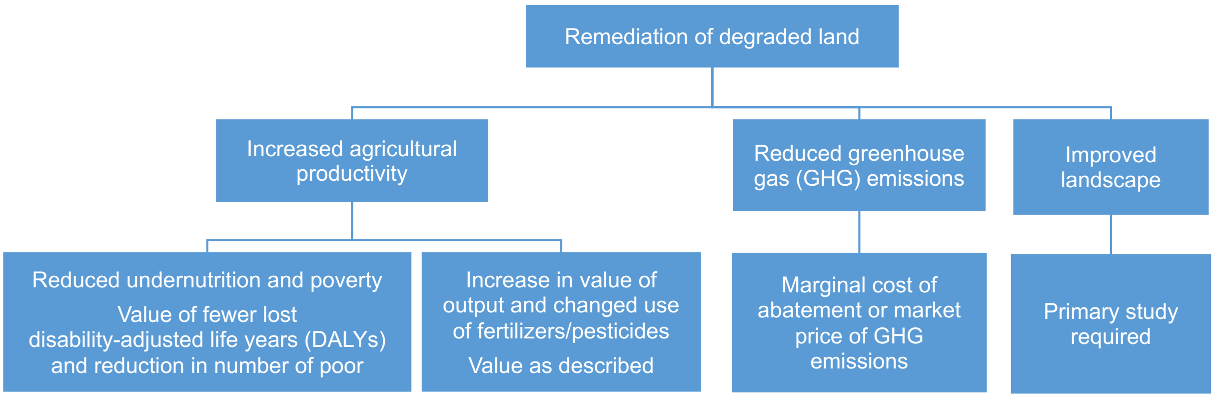
The main benefit of land remediation programmes will be an increase in sustainable agricultural productivity. Co-benefits may include reductions in GHG emissions and amenity benefits from an improved landscape. There is also the possibility that greater productivity will reduce undernutrition and poverty. Impacts via changes in inputs such as fertilizers and pesticides are uncertain. The main benefits are best estimated by modelling the changes in productivity or obtaining them from similar sites that have been remediated.

The co-benefits of reduced GHG emissions are costed in terms of marginal abatement costs or the market price of traded GHG, as outlined above. Others are valued as described in the individual categories.

In the case of landscape benefits, data on the local use of the landscape will be required. This must be combined with data on willingness to pay for the improved landscape, derived using stated preference methods) and/or estimates of increased visitation rates (derived from travel cost models). Both approaches are data intensive, but there is no alternative to getting an estimate of these benefits.

Figure 5 sets out the decision-making process. Programmes for remediation are tracked first and foremost for sustainable increases in agricultural productivity. They are also screened for changes in GHG emissions or improved landscapes. The gains in productivity are valued in terms of the sustained increase in value added (this may come after a certain period once remediation measures have been introduced and allowance must be made for this). In addition, there may be benefits in the form of reduced undernutrition and poverty, which should be recorded and included in the evaluation. Other co-benefits are valued as indicated. All benefits are compared against the costs of the measures using CBA, with supplementary indicators for impacts that cannot be valued in monetary terms.

Figure 5. Decisions relating to land degradation



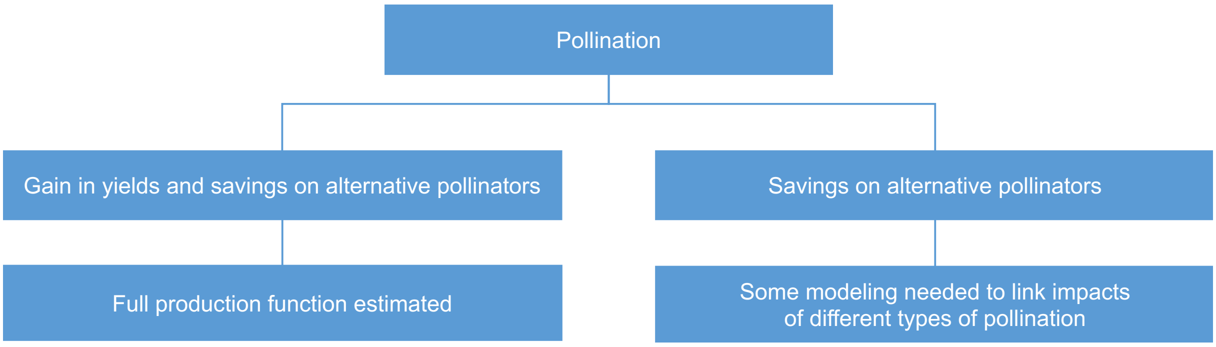
Source: Author's own elaboration.

4.2.4 Pollination

The review of the pollination discussion in Annex 1 identifies two methods of assessing the benefits of improved pollination. The first is a production function approach, which estimates the gain in yields as a result of the increased natural pollination along with the increase in revenue. Added to this are the savings in reducing alternative placement methods. The second, simpler calculation computes the savings on replacement pollinators, such as pollen dusting, hand pollination and managed beehives, which are currently used as a substitute.

Figure 6 shows the decision-making process as it relates to pollination. Measures to increase pollination will result in gains in yields and savings on alternative pollinators. The first pathway is taken if both impacts can be valued. If gains in yields are too difficult to establish, a lower bound will be obtained by estimating the savings on alternative pollinators. These benefits are compared with costs in a cost-benefit framework, taking into account the time profile of both costs and benefits.

Figure 6. Decision-making process relating to pollination



Source: Author's own elaboration.

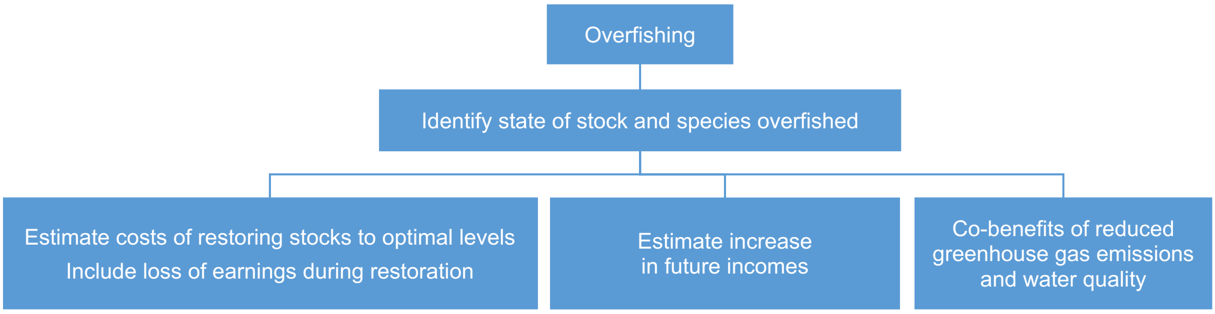
4.2.5 Overfishing

The aim of policies in this area is to restore stocks of species to levels at which they can provide a sustainable yield. The bioeconomic model used to derive global estimates for a single species is not refined enough to form the basis of a national-level appraisal of costs and benefits. The starting point would be to identify a species as being overfished or at risk of being overfished. An assessment then has to be made of actions needed to restore stocks. The costs would comprise income lost to fishers as a result of restricting catch for long enough to allow populations to reach desired levels. In addition, for some fish resources, management plans would need to be strengthened. This would include regulations such as the size of fish that can be caught and plans to increase the abundance of spawning stock through better protection of juvenile fish. Together, these costs would provide an approximation for the costs of overfishing, which are then compared with the increased future income from a higher yield at a lower level of effort.

Possible co-benefits include reduced GHG (from the reduced use of fuels for vessels) and better water quality (where measures include improvements in quality).

Figure 7 sets out the decision-making process for overfishing. The first step is to establish the stock of fish by species and identify those that are overfished. The next step is to decide on the actions needed to restore numbers to desired levels and maintain them. These actions will entail a number of costs, including loss of earnings during restoration. At the same time, the restoration of stocks and measures to keep harvests below rates of growth in the stock will increase future incomes. There may also be co-benefits of reduced GHG and improved water quality. All the benefits are added up and their present values compared with the present values of the costs to evaluate the programme.

Figure 7. Decision-making process for overfishing



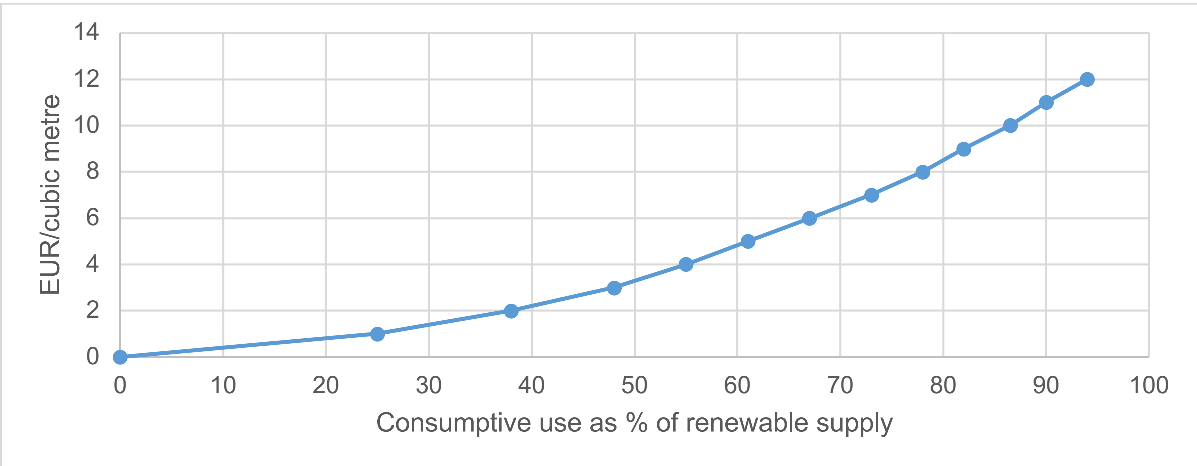
Source: Author's own elaboration.

4.2.6 Water scarcity

As water scarcity varies greatly from one watershed to another across a country, the ideal approach is to estimate the impact of water withdrawal for agriculture in a watershed on other ecosystem services. These could include the use of water bodies for recreation, fishing and other amenities. Methods for doing so are discussed in the previous section. As mentioned, they require local data on water use, as well as studies that estimate the benefits of different uses.

The alternative is to use the Trucost methodology, which relies on an estimated value function. This function is derived from United States of America's studies and gives the indirect cost per cubic metre withdrawn as a function of the consumptive use of water in a watershed as a percentage of renewable supply. This is reproduced in Figure 8 from the environmental accounts of PUMA, which used the Trucost methodology. The Trucost study took a global average of consumptive use and applied the function accordingly, but noted that costs would vary greatly between watersheds. It is possible to apply the same function for different levels of use and to adjust the cost from the one used in the global report to allow for differences between per capita income in the country and global average per capita income (as explained in the discussion on BT in Section 3.3) (FAO, 2014). This would provide a cost per cubic metre which, combined with the change in total withdrawals, would give an estimate of the benefits of a reduction in withdrawals to address scarcity.

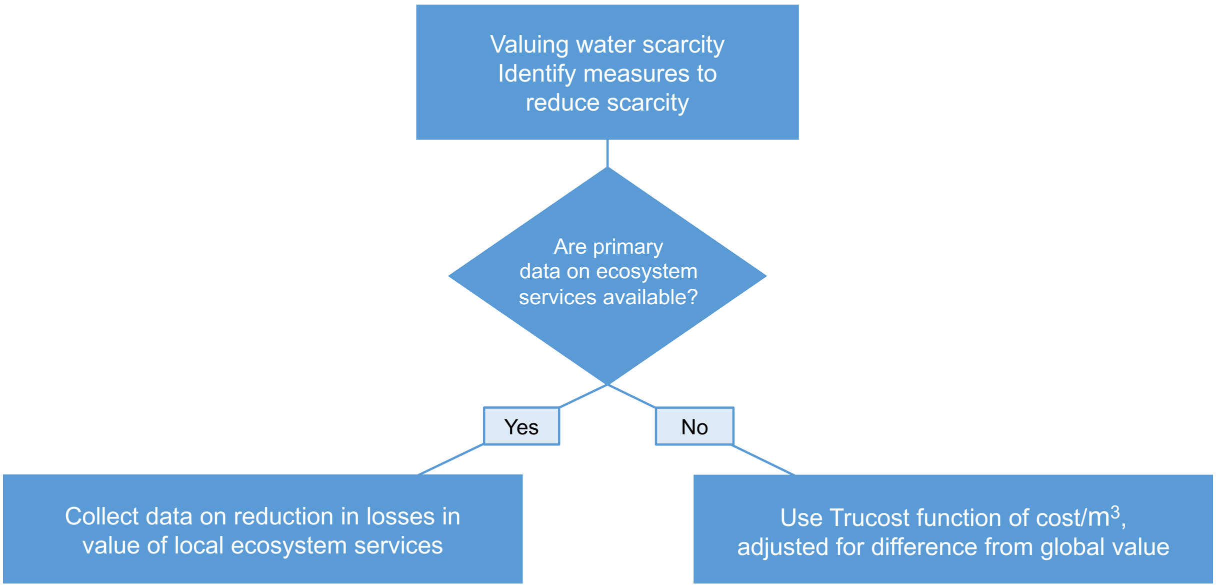
Figure 8. Indirect costs of water withdrawal



Source: PUMA. 2012. *Environmental Profit and Loss Account 2010. Herzogenaurach*, Germany. http://danielsotelsek.com/wp-content/uploads/2013/10/Puma-EPL.pdf

Figure 9 shows the decision-making options for water scarcity. Begin by identifying the measures that could reduce water withdrawal from water bodies in the region and collecting data on the costs of the measures. There are two alternatives for estimating the benefits of the measures. The first is where data are available on the benefits of reduced extraction. In this case, estimate those benefits in monetary terms and compare them with the costs. If such data are not available, value the reduction in extraction using the Trucost function in Figure 8, adjusted for local values.

Figure 9. Decision-making process for water scarcity



Source: Author's own elaboration.

4.2.7 Unhealthy diets

The global studies listed in Table 1 give the loss in disability-adjusted life years (DALYs) resulting from unhealthy diets. The source of the figures is the Global Burden of Disease programme, managed by IHME, which also gives estimates by country. The valuation of each DALY was put at average global per capita income. Others take a different view and estimate values for the European Union that are much higher than the region's per capita income (Holland, M, 2014; Rabl, Spadaro and Holland, 2014).Thus, applying the BT described earlier to obtain a global average, one would get a higher value than global per capita income. In any event, policymakers can decide on a value based on the literature and the arguments made. Co-benefits include lower GHG (where less meat is consumed), less land conversion for agriculture and less antimicrobial use. These are valued as indicated in the sections discussing those impacts.

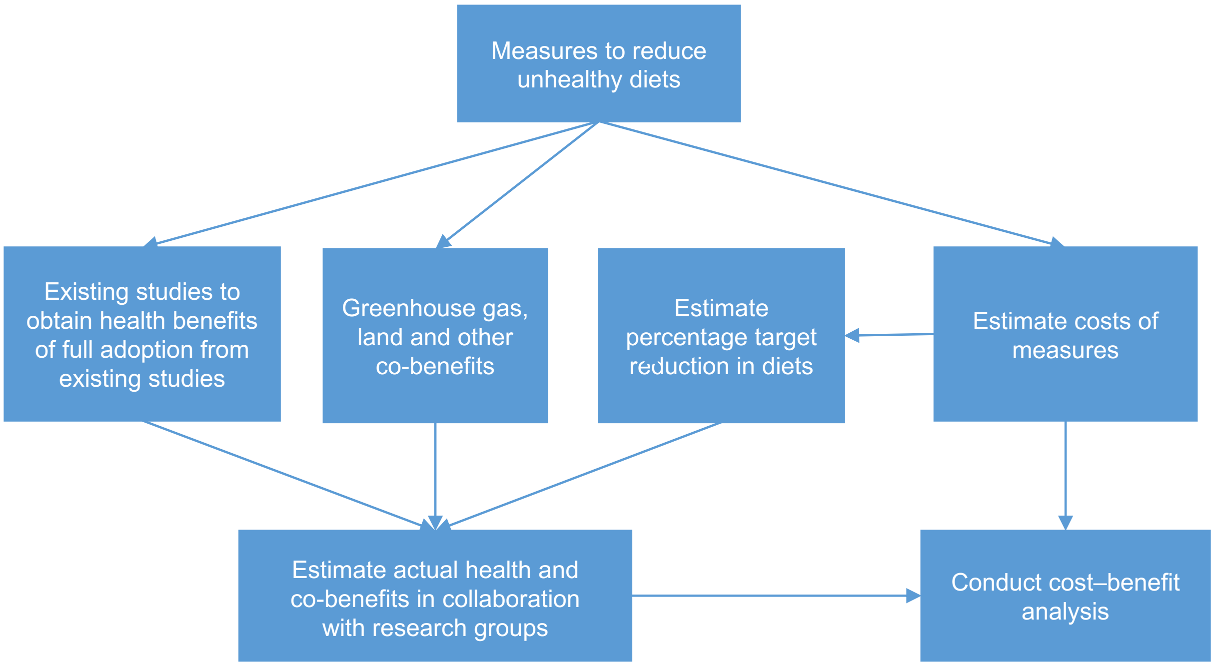
In a policy context, it is necessary to look at different actions and estimate the change in diet they cause, and then the benefits of the change in terms of reduced loss of DALYs or lower mortality. Studies have been undertaken recently that do this. They calculate the gains from adopting diets in line with global dietary guidelines on the consumption of red meat, sugar, fruits and vegetables, and total energy intake, as well as more plant-based (flexitarian) diets that more comprehensively reflect current evidence on healthy eating by including lower amounts of red and other meats and greater amounts of fruits, vegetables and nuts (Springmann *et al.*, 2018). They also look at the costs of a wide range of diets (pescatarian, vegan, flexitarian) relative to current diets (Springmann *et al.*, 2021). The benefits range from fewer environmental impacts (including lower GHG) and lower health costs. They do not, however, value DALYs or lives saved (preventable deaths). The analysis is conducted on a regional level.

The health benefits of diets that are in line with global dietary guidelines are substantial. If the range of diets described above that are broadly in line with guidelines had been implemented in 2017, it would have resulted in an 87 percent to 93 percent reduction in diet-related health-care costs globally, as well as across regions in that year.[33] Similar figures apply to projections for 2050 (although there is a wider range of estimates for that year). In terms of the cost of the diets, the results are more mixed. Compared with the cost of current diets, the cost of the healthy and sustainable dietary patterns were, depending on the pattern, up to 22–34 percent lower in upper-middle-income to high-income countries on average, but at least 18–29 percent more expensive in lower-middle-income to low-income countries (Springmann *et al.*, 2021).

This kind of analysis is highly data intensive and requires considerable time and skill to undertake. At a country level, the best course of action is to estimate the costs of the different diets. The benefits can be obtained from ongoing research studies (regional estimates can be broken down to country level on request). The more difficult, but necessary accompanying task is to investigate measures that can be introduced to bring about the changes in diet. This will involve behavioural modelling of incentives (including some fiscal ones) and the costs must then be compared with the gains cited. As the full adoption of any combination of these diets is highly unlikely, the modelling should also be extended to consider partial adoption.

Figure 10 shows the decision-making process for unhealthy diets, combining the health benefits and co-benefits in evaluating actions to change diets. For each measure to reduce unhealthy diets, the direct benefits in terms of reduced loss of DALYs can be obtained from existing studies, assuming full adoption of the measures. Co-benefits can be estimated in parallel from data collected on reduced GHG and other impacts. Likewise, the costs of the measures can be estimated; they arise from the actions needed to change the diets. At the same time, an estimate must be made of what percentage of the target reduction is likely to be achieved, based on experience with similar programmes in many countries. This, combined with the estimated benefits of full adoption, will give an estimate of the benefits of actual adoption, which can be compared with the costs.

Figure 10. Decision-making process for unhealthy diets



Source: Author's own elaboration.

4.2.8 Undernutrition

The IHME database provides estimates of the DALYs lost on account of undernutrition for almost all countries. Projections of future levels have also been made by the WHO under different climate change scenarios (WHO, 2014).[6] Actions may have negative impacts, such as conversion of land to agriculture, an increase in GHG emissions, greater demand for water and land, and higher demand for inputs such as fertilizers and pesticides. These are costed as described in the sections dealing with those impacts.

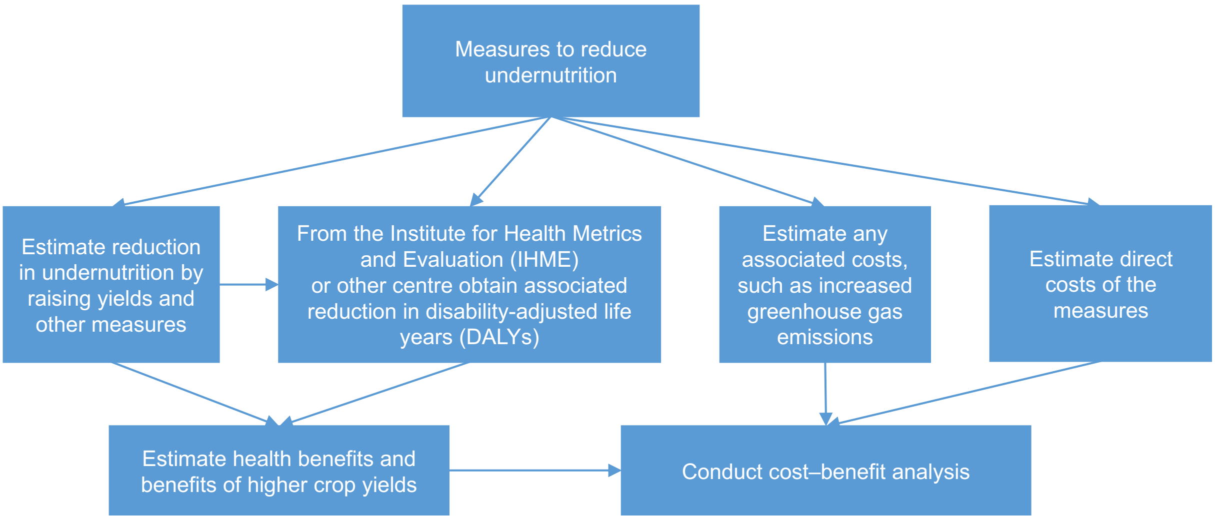
Measures to reduce undernutrition include adaptation through climate smart agriculture. While such measures have been analysed extensively for the benefits obtained (World Bank, 2015), quantitative estimates of changes in undernutrition have not been found in the literature review conducted for this study.

Countries wishing to undertake such an analysis will have to model the impacts of interventions to boost agricultural productivity and resilience to obtain an estimate of the changes they bring about in undernutrition, as defined by the Global Burden of Disease/IHME framework (Kumanyika *et al.*, 2020). This can be combined, through collaboration with IHME or other institutions working in this area, to determine estimates of lost DALYs or reductions in mortality. These reductions can then be valued by the methods reviewed earlier and fed into a CBA of the intervention.

[6] The modelling is complex. It combines climate models, crop models, trade models and national calorie availability with in-country calorie distribution to derive undernutrition in children under the age of five. From that, epidemiological modelling is used to obtain estimates of stunting and associated mortality. The underlying crop and trade model is from the International Food Policy Research Institute, and it also provides separate projections of hunger with and without climate change.

Figure 11 shows the decision process for undernutrition. Measures to reduce undernutrition are first assessed for increased production (if any), as that will be part of the overall assessment, along with changes in the level of undernutrition. This will require evaluating the changes in food production and its availability to groups at risk of undernutrition. The information on the changes in undernutrition is then passed on to a collaborating institution, which can estimate the savings in DALYs. These are valued using the methods of BT described in Section 4. The third component is possible changes in GHG or other environmental impacts resulting from the measures, which are costed as indicated in the sections dealing with those impacts. The data from all three sources are combined to estimate the flow of benefits from the measure and compare these with the costs.

Figure 11. Decision-making process for undernutrition



Source: Author's own elaboration.

4.2.9 Air pollution

Ambient and household air pollution are problems caused by many sources, not just agriculture. Measures to address the costs arising will involve actions to address transport, the burning of agricultural waste and access to clean energy for households, especially in rural areas.

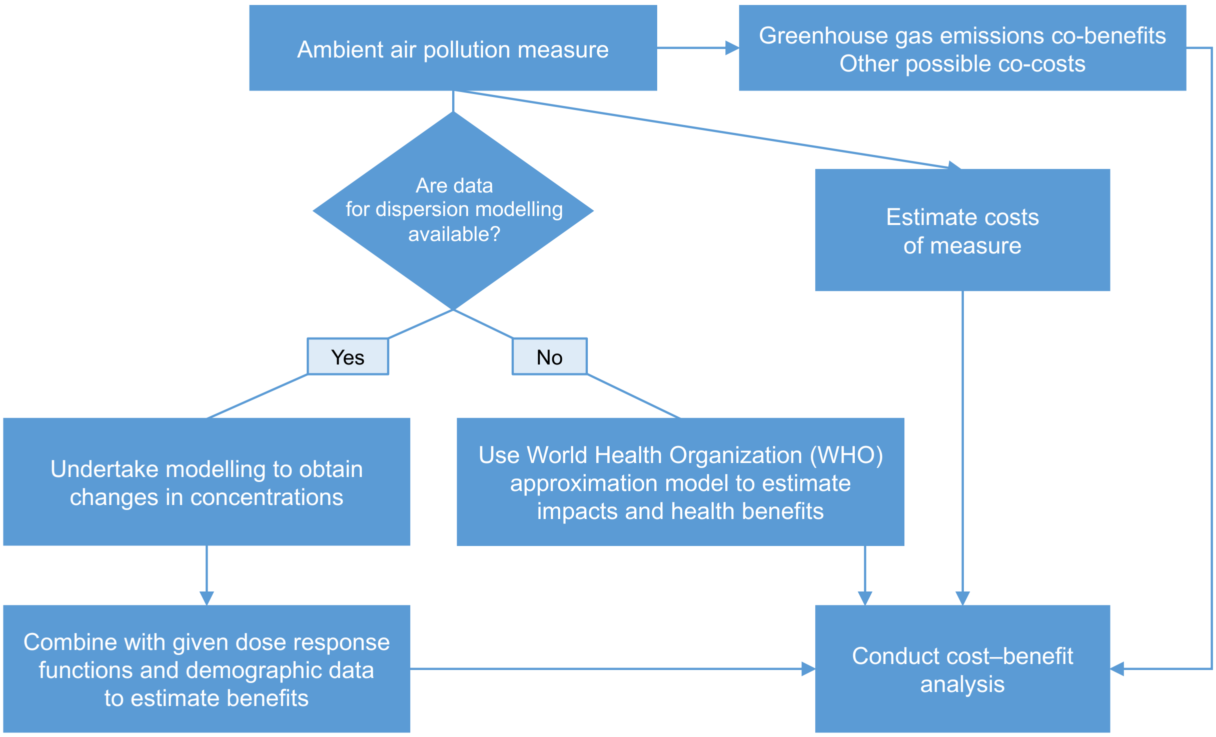
Estimating the benefits of measures to reduce ambient air pollution requires modelling the changes in concentrations resulting from the measures. This, in turn, is based on measuring stations that record concentrations. This is a demanding process, but an important one, as the sources of high concentrations of pollutants can be far from the areas where the impacts are greatest, and modelling must be done at the air-shed level.

Most countries have some monitoring stations to facilitate concentration modelling, which can be used to evaluate changes arising from specific measures. Where available, these should be used. In addition, WHO is undertaking a review of methods for tracking policies that address both ambient and local air pollution. It has also developed a tool to enable estimates to be made of interventions without full dispersion modelling (that is, without a model that takes data on emissions from all sources and calculates the concentrations across the country) (WHO, 2023). If an estimate can be made of the change in emissions consequent to a measure, the model has default value for most countries, enabling them to calculate the health benefits from reduced mortality. Associated positive impacts of the measures to reduce emissions are lower GHG, while possible negative impacts could include increased use of land (in the case of biofuels).

As far as indoor air pollution is concerned, the estimation of benefits and costs is easier. An evaluation has been conducted of the methods that can be used and the results obtained, covering most countries (Hutton, Rehfuess and Tediosi, 2007). Countries can take estimates directly from this study or, in collaboration with the authors, make more detailed national-level estimates of interventions, such as replacing traditional fuels with liquefied petroleum gas and using better stoves.

Figure 12 shows the decision-making process for air pollution. Measures to improve ambient air quality are assessed for any changes in associated GHG emissions. Those changes in emissions, valued as explained in the section on GHG, are included directly in the CBA. For the changes in local pollutants, the first questions are whether data for dispersion modelling are available and whether there is modelling capacity. If so, the measures will give data on changes in emissions at different locations, which can be combined with meteorological data to estimate changes in concentrations at different locations. This information is combined with demographic data and dose response functions that link changes in concentrations to health impacts to obtain the health benefits, expressed in terms of reduced premature mortality and reduced DALYs. The valuation of these proceeds uses methods of BT, as set out in Section 4. If dispersion modelling is not possible, a direct link between the change in emissions and health impacts can be made using approximation models of the kind that WHO (2023) has developed.

Figure 12. Decision-making process for ambient air pollution



Source: Author’s own elaboration.

4.2.10 Antimicrobial resistance

While there are estimates of the costs of AMR, there is very little on the reduction in these costs due to preventative measures. This is especially the case when it comes to the use of antibiotics in agriculture. The FAO Action Plan on AMR identifies action in four areas: awareness, surveillance and monitoring, governance and the promotion of good practices (FAO, 2016). Further research is needed to develop estimates of the quantitative gains from these actions. The OECD is currently updating its earlier work, which should provide some guidance on methods and results. Where a reduction in the use of antibiotics can be estimated as part of the measures taken (for example, a reduction in livestock), this should be noted as a potential benefit, but not monetized.

No decision process is set out for this category.

4.2.11 Pesticide pollution

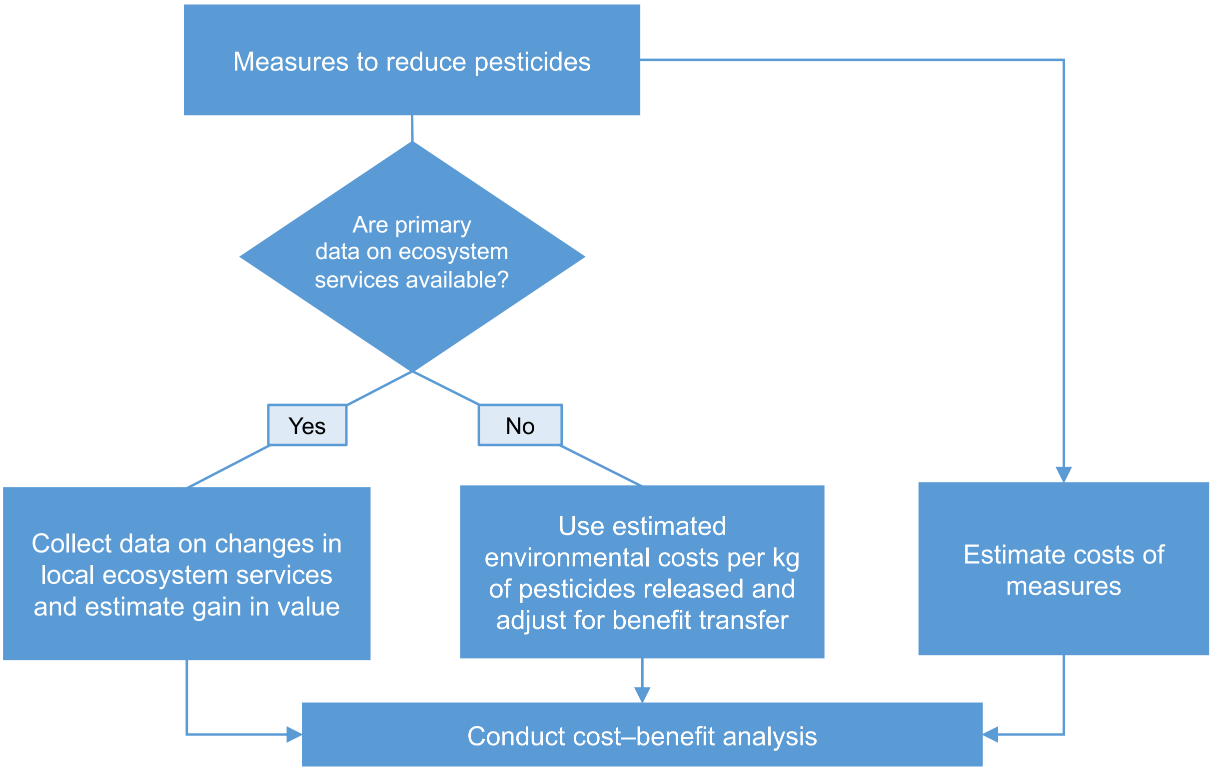
Countries can undertake location-specific studies of the benefits and costs of pesticide pollution. The example of Nepal describes the data that has to be collected and the analysis that needs to be undertaken (Atreya, 2008).If this can be done in Nepal, which is not a data-rich country, it should be possible in other developing countries. The methods involved, however, are demanding and require time and resources.

Where pesticide releases are part of a broader set of issues that are being tackled and a specific study on this aspect is not possible, the global value used in FOLU (2019) of an estimated loss of 0.02 DALYs/kg can be taken. The DALYs can be valued using the costs of a DALY, as discussed in Section 4.2.7 on unhealthy diets.

Alternatively, as mentioned, a set of values has been compiled for the costs of damage caused by the release of different pesticides in Europe, reported in euros/kilogram (CE Delft, 2018). A BT can be made, using a simple ratio of the per capita income in the EU-28 in 2018 to that of the country to which benefit is being applied. The formula for doing so has been discussed in Section 4.1.

Figure 13 gives the decision-making process for pesticides. In evaluating measures that reduce pesticides, the preferred option is to use primary data on the impacts of pesticides on health and ecosystems. If these are not available, estimated damages per kilogram of pesticides released can be used from global data. These estimates can be adjusted for local conditions, as explained in Section 4. The benefits in terms of reduced damages are compared with the costs of the measures, which are estimated separately.

Figure 13. Decision-making process for pesticides



Source: Author's own elaboration.

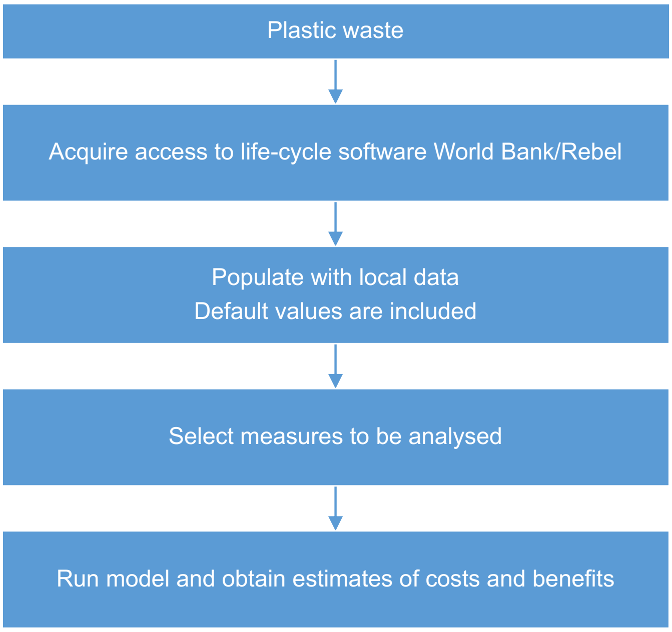
4.2.12 Plastic pollution

A Trucost study for UN Environment has shown that the impacts of plastic pollution cut across many environmental areas and that the analysis of these is complex (UNEP, 2014). A complementary study, also based on the Trucost methodology, is being developed jointly by the World Bank and Rebel, a consulting group (World Bank, 2021). It involves an LCA tool that can be applied to any country. It also calculates the costs of plastic waste in that country from all the sources identified in the Trucost study and was recently applied to Viet Nam (World Bank, 2022a). Availability of the software to other countries is subject to the agreement of the World Bank, but the aim is to make it widely available, so this should not be an issue.

The most effective way to investigate the impacts of policies on plastic pollution would be to use the tool developed by the World Bank and Rebel. The issues that could be evaluated are material to substitutes for plastics, the impacts of greater collection and clean-up from public places, higher rates of reuse and so on.

Figure 14 shows the decision-making process for plastic pollution. As mentioned, the best approach would be to use the software described to evaluate measures that have been selected locally.

Figure 14. Decision-making process for plastic pollution



Source: Author's own elaboration.

4.2.13 Rural poverty

Programmes for rural development and climate smart agriculture track the changes they bring about in the incidence of rural poverty. The benefits are measured in terms of the gain in income to rural households. In addition, any elimination of poverty is noted as a non-monetary benefit. The methods developed to estimate hidden costs in the Trucost approach do not provide any guidance on how such programmes or policies are to be evaluated. As the WRI study cited earlier shows, many existing subsidies benefit the larger farmers (Ding *et al.*, 2021); repurposing them through ecological transfers can generate both environmental benefits and gains in terms of poverty reduction.

The decision process is simply to conduct a CBA of the projects, focusing on rural development or climate smart agriculture using the guidelines available for such assessments. Assessments should also note any change in rural poverty and any additional information to the CBA.

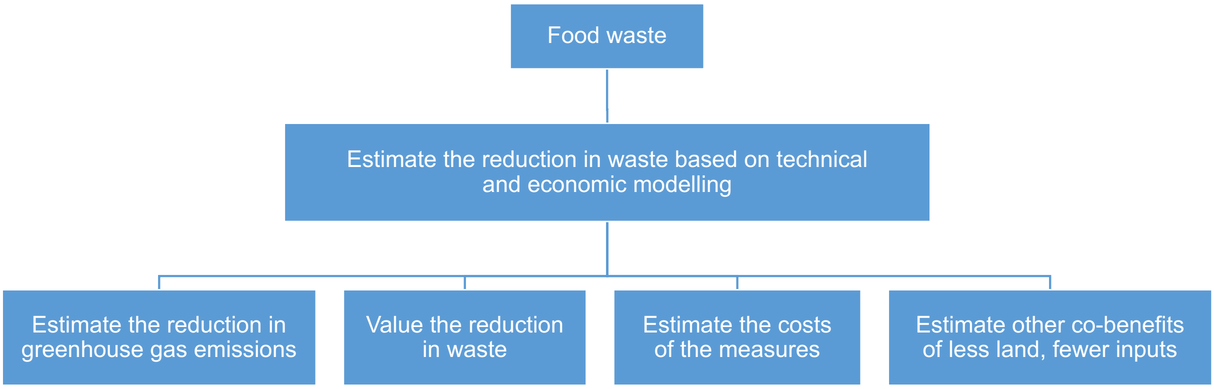
4.2.14 Food loss and waste

Actions to reduce food loss and waste are being considered in many countries. Moreover, the costs of some measures are low compared with the benefits. FAO studies in India showed that 75 percent of emissions from rice lost through poor management and storage could be eliminated at a low cost of between USD 0.1 and USD 9.3/tonne of CO_2 (FAO, 2018). Co-benefits of reduced food waste include lower GHG, less land for cultivation, lower ambient air pollution, less fertilizer and pesticide use, and less AMR. Estimating these benefits follows the decision-trees provided.

Measures to reduce wastage by other means have not been investigated as fully. The main issues arise from estimating with accuracy the success rate of the actions. Pilots of programmes to raise awareness and various incentives can be undertaken to generate data that can be used for scaling up. The estimation of benefits, once the quantities of waste reduced are known, does not raise any major problems.

Figure 15 shows the decision-making process for food waste. For each selected measure, the first task is to estimate the reduction in waste that is achieved. This will begin slowly and take some time to achieve its full potential. Once this is known, the value of the savings will be a key benefit. Others include lower GHG, and co-benefits resulting from less land needed for cultivation. These benefits are added up and compared with the costs of the measures.

Figure 15. Decision-making process for food waste



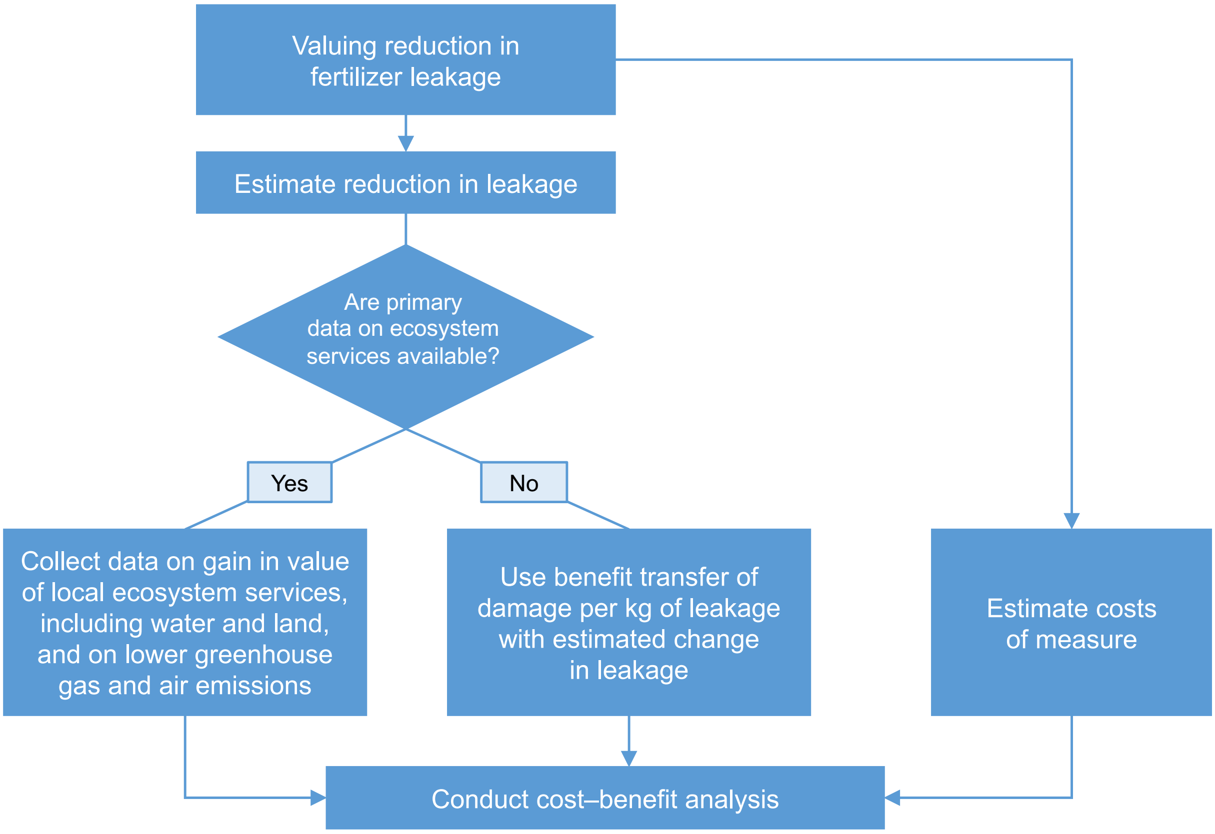
Source: Author's own elaboration.

4.2.15 Fertilizer leakage

The FOLU (2019) study simply costs the leakage in terms of the value of the fertilizer lost. It is possible to get a similar estimate at country level. Leakage rates in the FOLU study can be used if local estimates are not available. An environmental loss can also be added. Unless local studies of costs arising from leakages are available (which is, of course, the preferred option), a BT can be made from the European Union study of costs for different leakages (CE Delft, 2018). These can be adjusted for per capita income differences using the formula proposed earlier (see Section 4.1), to give an approximate cost of the environmental damages. The analysis would be applied where agricultural run-off is causing damage to ecosystems and measures to reduce such run-off need to be evaluated.

Figure 16 shows the decision-making process for fertilizer leakage. The first task is to estimate the reduction in leakage brought about by the measure. Then, if data are available on the impacts of reduced leakage on ecosystem services, this should be used to derive estimates of the gains in services. In the absence of such data, a BT estimate can be made based on data from the CE Delft Handbook referenced above, adjusted for local conditions using the per capita income method described in Section 4.

Figure 16. Decision-making process for fertilizer leakage



Source: Author's own elaboration.

5 Conclusions and recommendations

The community working on the Trucost approach has done a great service in demonstrating the huge hidden costs associated with agrifood systems. The approach covers environmental, health and economics areas, with health being the largest. The team has also made estimates of the benefits of adopting less damaging production methods and dietary lifestyles and the investment costs of achieving these changes, but does not account for all of the expenditures associated with public policy reforms and private-sector changes. Nevertheless, the indications are that reforms pay off at the global level by bringing greater benefits than costs.

To apply the method at country level, the methods developed must be downscaled and account has to be taken of data limitations. This review has gone through each category of cost and proposed approaches to deal with them. The aim has been to choose the most accurate method, but this can involve a large amount of data and expertise. Where these are not available or time is limited, methods combining secondary data have been suggested. These make use of the BT approach and will be less accurate but, for many purposes, useful as an input into appraising alternative measures. In some cases, the suggestion is to collaborate with research centres, especially those working on health impacts at national level. This should also be possible for the use of software developed to conduct sophisticated LCA, for example, for plastic waste.

In appraising policies and measures, account must be taken of impacts in all categories. The analysis has identified these and indicated which ones need special attention. A key potential cross-impact of introducing measures is that they will raise the cost of food for the consumer and increase poverty and food insecurity. There are combinations of policies that can avoid, or at least limit, these consequences.

The estimates obtained from all methods are by no means certain and are even higher when secondary data are used. It is important to give an indication of the magnitude of this uncertainty and to reflect it in the reported CBA.

As the body of studies expands, knowledge of what works and what does not will grow and the advice given on conducting future policy assessments will improve.

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Annex 1. Literature review of hidden costs of agrifood systems

This annex provides an in-depth review of the estimates made for the different sources of hidden costs of agrifood systems, including cross-sectoral issues, such as the treatment of uncertainty and the use of tools for estimating different impacts.

A. Greenhouse gas emissions

As the main cause of climate change, the damages caused by GHG to human well-being have been estimated in detail. They include losses in agriculture, forestry and fisheries, impacts on health and damage from extreme events. Drawing on a number of models, the Government of the United States of America has reviewed studies of global damages resulting from a marginal increase in emissions over the time into the atmosphere and arrived at a range of estimates (US Government Interagency Working Group on Social Cost of Carbon, 2013). These estimates of the expected damages, referred to as the social cost of carbon, is the present value of such damages and consequently depends on the discount rate: the higher the chosen rate, the lower the present value will be. Based on a review of the different assessment models, the document gives a range of USD 14.9–USD 80.5/tonne CO_2 in 2020, rising to USD 19.8–USD 94.1/tonne CO_2 in 2030 (in USD 2019). The mean values of these ranges are USD 47.7/tonne CO_2 and USD 56.9/tonne CO_2 for 2010 and 2030, respectively. Even this wide range does not encompass all of the figures in the literature – a more recent study, for example, suggests much higher values (Moore and Diaz, 2015).

The social cost of carbon is an important variable in determining climate policy and targets for the reduction of GHG at global level. Given the large uncertainties, however, the True Cost Initiative recommends the use of an abatement cost for the TCA assessment (True Cost Initiative, 2022). The example given is the cost of replacing coal with wind or other renewables for electricity generation. The most recent United Nations System of Environmental Economic Accounting (SEEA) guidelines for valuing environmental costs make a distinction between valuing carbon retention and carbon sequestration (NCAVES and MAIA, 2022).[7] The two use different methods of cost estimation. For carbon retention (or its converse, carbon release), the use of the social cost of carbon is recommended, as this aligns with the framing of avoided damages. For carbon sequestration, where a value is placed on carbon that is removed from the system, the guidelines recommend using the abatement cost, or even the current market price for carbon that is traded. In the values reported in Table 1, details are only available for FOLU, which used a figure of USD 100/tonne CO_2e as an average marginal abatement cost from 2020–2050 in the literature (CPLC, 2017).

For applications of TCA in individual countries, the approach taken will depend on what policy is being considered. The most common one related to agrifood systems is to select measures to reduce such emissions. The abatement costs of each measure are compared across

[7] The carbon retention component consists of i) estimating carbon stocks of relevant carbon pools retained at the beginning of the accounting period; ii) multiplying this by a suitable carbon price; and iii) turning this into an annual service flow by multiplying this value by a suitable rate of return (to create an annuity). The carbon sequestration component is measured by the net ecosystem carbon balance, which takes all changes in carbon stocks (for example, respiration, timber harvest, forest fires) into account. Carbon sequestration has a value to society, as it reflects the removal of carbon from the atmosphere, mitigating the effects of climate change.

measures and the ones with the lowest cost per tonne abated are selected. In this case, the social cost of carbon is not relevant, but a careful estimation of the abatement cost of each measure needs to be made. The social cost of carbon is an input in deciding on global emission reductions and in selecting dates for achieving the target reduction. These issues are discussed further in the next section.

B. Water pollution

Releases of pesticides and excess fertilizers into water bodies result in losses of fisheries, eutrophication and other negative impacts. These, in turn, cause a decline in the value of recreational and commercial use of the water bodies, as well as costs to water utilities. Estimates of these losses make use of a range of techniques.

Cost-based methods are used when the presence of agricultural chemicals raises the cost of treating drinking water. The increase in such costs is based on the additional treatment costs the water utility incurs. The question of who pays these costs is a separate matter (see discussion in Section 3.1). The SEEA report recommends this approach (NCAVES and MAIA, 2022).

Another cost-based approach is to estimate the cost of treating farm waste, such as animal manure, prior to its release into the environment and to use that as an estimate of the monetary costs associated with the manure. The True Cost Initiative recommends this approach, but the method does not enable a judgment to be made on whether such treatment is justified (Dasgupta, 2021). For that, it is necessary to estimate the damages caused be the release of the untreated effluent and compare it with the cost of treatment. If the latter is greater than the former, the treatment is justified.

The techniques of revealed and stated preference also provide information on the costs of pollution at local level. An example of revealed preference methods is the hedonic method, where the effects of pollution, resulting in odours or visually unattractive plant growth, on property values are estimated using econometric techniques. Le Goffe has estimated the impacts of agricultural activities on the rental prices of rural self-catering cottages, or gîtes, in Brittany. He found that intensive livestock farming caused the rental prices of gîtes to decrease, whereas permanent grassland had the opposite effect (Le Goffe, 2000).

Stated preferences can be used to obtain the loss of value associated with a decline in the quality of water in a water body. People visiting such areas are asked about their willingness to pay for avoiding such a decline or their willingness to accept compensation for the decline. Another application of stated preferences would be to determine what compensation farmers might need to move towards production methods that involve lower releases of pollution. While they have an incentive to overstate the amount they need, there are methods available to reduce such biases. One approach could be to use a sealed bid contracting system.[8] This could be effective if farmers were told that contracts would be awarded according to the combination of pollution reduction and cost.

The wider recreational benefits of land and water that can be affected by agricultural practices require an estimate of the value of services using these revealed and stated preference methods. The methods themselves have improved greatly in recent years. They are, however,

[8] As the term indicates, this would involve farmers proposing an amount of reduction and a required unit payment in a sealed bid. The ones accepted will be based on the lowest cost combination of offers.

demanding, both technically and in terms of data collection (OECD, 2018a). An example of how they might be applied and what can be obtained is the UK National Ecosystem Assessment, which estimated the rural recreation benefits arising from a change of land use from conventional farming towards multipurpose, open-access, woodland (OECD, 2018a). This involved the use of a “recreational value function”, which combined: (a) the value of a visit to a site based on its characteristics and (b) the number of visits to a site. The value function drew on an extensive body of literature that estimated such values based on revealed and stated preference methods. The visits generation function was estimated for the UK from data on the number of visits to each site from a defined set of locations, based on travel time to the site, the characteristics of the site and availability of substitutes to the site (Sen *et al.*, 2003). The results showed where benefits from a change of land use were significant. This could be compared with the losses from agricultural output from the conversion, so that a decision could be made on where to implement the changes.

The choice of which method to use will clearly depend on the policy questions being addressed. For issues relating to drinking water, a cost-of-treatment approach (see Table 3) is the easiest to deploy. For pollutants that affect sensitive watersheds or rivers, information on the costs of alternative methods of treatment of the farm waste before release will be critical. In determining the locations where the problem is most serious, however, some estimates of the loss of value to recreational uses is important. Lastly, where a programme to improve water quality is being considered at the national or watershed level, estimates of the benefits from reduced water pollution for all affected parties will be required.

C. Land degradation

The loss of agricultural productivity from land as a result of misuse and unsustainable practices is measured either in terms of the market value of the lost output or in terms of the cost of restoring the lost productivity. A particular source of loss, such as soil erosion resulting from the removal of tree cover, can be estimated in by calculating the decline in yield and multiplying it by the net income derived from that yield (market-based method). Alternatively, it can be estimated by calculating the cost of additional fertilizer and other inputs needed to restore the net income to the level it was before the erosion (cost-based method). The True Cost Initiative recommends both methods (Dasgupta, 2021). It is important to recognize that both approaches need information on the physical relationship between the type of degradation and the production of output (crops or livestock products). The degradation is measured in terms of an increase in an input into the production process for a given output or a decline in output if inputs are constant. The relationship between the inputs and outputs is referred to as the production function. Estimates of the production function are part of the literature on crop and livestock productivity and can be used if they have been estimated in countries or regions similar to the one where the estimation is being made.

An example of the use of a crop production function to estimate the loss of productivity can be found in a study on the role of soil quality and soil investments, along with other inputs, on crop yields in Kenya (Ekbom and Sterner, 2008). Here, the farmer is assumed to produce a given output based on a specific choice of traditional economic factors – labour, fertilizers, manure and agricultural land – and other variables – soil conservation investments, access to public infrastructure and tree capital, and soil capital – represented by the soil properties. These factors are, in turn, dependent on others such as household characteristics (for example, the number of members of the household), crops planted and their mix, and extension activities provided to the farmers that affect quality. The responsiveness of output to change in various inputs is captured through elasticities, which give the percentage change in output for a 1 percent change in any given input.

The study showed that soil quality and soil quality improvements have a positive and significant influence on output (elasticity = 0.20), with nitrogen (elasticity = 0.27) and potassium (elasticity = 0.35) increasing the output significantly. High levels of phosphorous (elasticity = –0.22), meanwhile, are actually detrimental to output, thus drawing attention to the need to adapt fertilizer policies to local biophysical conditions. Investments in soil capital by minimizing soil disturbance, maintaining four-season soil coverage by using cover crops and diversified crop rotation, and enhancing soil quality with compost and other natural soil amendments have an important role to play in agricultural output. Thus, measures to arrest soil erosion can help farmers increase food production and reduce food insecurity.

While the main impacts of land degradation will be the loss of agricultural productivity, there could also be some losses of amenity and other services provided by the ecosystem. The National Ecosystem Assessment for the UK, mentioned earlier, showed how recreational values of sites depended on the characteristics of the site. Where agricultural land is also used for walking, hiking and so on, any change in those values as a result of the degradation of the land should also be taken into account. Examples in the TEEB report include the loss in value of ranch open space in Arizona because of the conversion of some of that land for development (Rosenberger and Loomis, 1999), and the impacts of green land cover on property values in St. Louis County, Missouri (Walls, Kousky and Chu, 2015). The Arizona study uses a combination of stated preference and information on travel costs, while the Missouri study uses the hedonic method to derive estimates of the amount and condition of agricultural land based on property values.

Other services that are lost as a result of degradation include carbon retention (to be valued as discussed in Section A1.1) and possibly flood prevention (to be valued as discussed in Section A.1.6).

D. Loss of pollinators

Loss of pollinators is a major issue in many countries. As a result of habitat conversion for agricultural production, the use of some agricultural inputs and an increase in invasive species, there has been a decrease in pollinator diversity in most global regions. Moreover, it is expected to continue (Dasgupta, 2021). It is estimated that, globally, by 2050, as many as 5 billion people could face insufficient pollination for their crops (Chaplin-Kramer *et al.*, 2019).

The main costs associated with the loss of pollination services are those arising from lower crop yields. This can be derived using a market-based method that combines a direct estimate of the loss in physical units and by multiplying it by the net income per unit (see Table 3). The method requires an estimate of the amount of pollination a particular crop requires (this varies considerably from crop to crop) and an estimate of the availability of pollinators for the crop in question. The analysis must be spatially disaggregated, given the spread of crops and pollinators across the landscape. The United Nations SEEA Guidelines recommend this as a first option (NCAVES and MAIA, 2022).

A second method is the cost-based method of using cultivated bees and other pollinators in place of wild pollinators to replace the lost pollinator services. TEEB and the True Cost Initiative recommend this approach, as it is less data demanding than the market-based approach (Dasgupta, 2021; TEEB, 2018). In many countries, there are well developed markets in the supply of cultivated pollination services, although there is concern that these contribute to the extinction of wild bees (Iwasaki and Hogendoorn, 2022).

An example of the use of the second method is given in the TEEB report and pertains to South Africa (TEEB, 2018). When wild insect pollinators do not provide pollination services, alternatives include pollen dusting, hand pollination and managed beehives (domesticated bees). Using the Western Cape deciduous fruit industry in South Africa as a case study because of its dependence on managed honeybees, an estimate was made of the value of both wild and managed pollination services. Two scenarios were considered: i) no insects (wild or managed) remain for crop pollination; and ii) managed pollination is not commercially viable or possible, leaving only wild pollination services. The decline in value in the case of a fall in all pollinators was found to be very large (Allsopp, Lange and Veldtman, 2008).

The use of a replacement cost approach is the preferred approach for estimating losses in pollination in data-scarce countries. Even then, however, the data demands are quite significant in terms of obtaining, first, the losses suffered as a result of reduced pollination and, second, the amount by which alternative methods can restore that loss.

E. Overfishing

The costs of overfishing are calculated as the loss in net fisheries revenue due to unsustainable fishing practices over a number of years. Data show that by 2011, 90 percent of fish stocks were fully fished, overfished, depleted or recovering (World Bank, 2017). This is based on stocks that maximize sustainable net benefits or maximum economic yield (MEY).[9] The estimated increase in net income if fisheries had been operating under MEY in 2012 was USD 83 billion. The figure indicates that the world's currently unsustainable fisheries management practices have led to globally depleted fish stocks that produce USD 83 billion less in annual net benefits than would otherwise be the case. The biggest losses are in Asia (65 percent), followed by Europe (15 percent) and Africa (12 percent). To reverse the losses caused by unsustainable practices would, of course, take several years: the estimated recovery time frame was estimated to be by 2040. The FOLU study used this figure as its estimate of the hidden costs of overfishing (FOLU, 2019).

The transition to a sustainable level of fishing would involve significant costs and changes in policy, but the benefits are estimated to be well in excess of costs. The single largest source of economic gain from moving to a sustainable level of fishing would be the reduction in fishing costs as level of effort per catch declines (52 percent), followed by higher prices of landed catch brought onshore (33 percent) and higher harvests as a result of a higher stock (15 percent).

To estimate the costs of overfishing at national level would require a cost-based approach. It would estimate the costs of restoring stocks of species that were currently being fished unsustainably in its territorial waters.

[9] In FAO statistics, fish stocks are defined as fully or overfished if their biomass is at or below the level that supports maximum sustainable yield (MSY). The maximum economic yield (MEY) is greater than MSY.

F. Water scarcity

Water extracted for agricultural purposes from rivers and lakes or from groundwater sources can result in less water being available for other purposes, such as ecosystem maintenance and nutrient recycling. The uses of water that are compromised are referred to as "indirect use values", while withdrawal for agriculture or other purposes is referred to as "consumptive use". The degree to which indirect uses are compromised (and, hence, suffer a loss) will depend on how great consumptive use is relative to renewable supply to the system from which is it withdrawn. The FOLU study takes its estimates of the costs of water scarcity from FAO (FAO, 2014). This, in turn, is based on a Trucost study that estimates a function linking indirect losses to consumptive uses as a percent of renewable supply (PUMA, 2012). This function is derived from a sample of 18 studies from the United States that span a wide range of local environments, from the arid to the relatively water abundant. Based on this, and adjusting for differences in value in different countries, a global average cost due to water scarcity was estimated at USD 1.15/m^3 (based on 2012 costs). However, the variation of water scarcity estimates between countries was found to be huge, ranging from USD 0.02/m^3 to USD 18.8/m^3. The FOLU estimate of the hidden costs of water scarcity uses the global average and applies it to the 25 percent of withdrawals for agriculture that are considered unsustainable or at risk of becoming unsustainable (FOLU, 2019). The 25 percent figure is taken from the Global Biosphere Management Model (GLOBIOM) model,[10] which forms the basis of much of FOLU's estimation of trends in the use of natural resources and their impacts and sustainability (IBF and IIASA, 2023).

At the country level, estimates of water scarcity will vary greatly. One approach could be to take the estimated relationship between consumptive use as a percentage of renewable supply in the PUMA Trucost study and the cost per cubic metre extracted. This could be applied to each watershed in the country.

G. Unhealthy diets

The FOLU report estimated the costs of obesity by calculating the loss of productive life measured in DALYs caused by over-consumption (148 million DALYs) and multiplying it by the global average per capita income (USD 17 971 in 2018) (FOLU, 2019). The valuation of DALYs in this case is based on market-based methods, linking them to per capita income. This gives a total annual cost of USD 2.7 trillion. The estimation of the loss of DALYs is from the Global Burden of Disease programme tracked by the IHME (IHME, 2023).[11]

Hendricks *et al.* (2023) focused on premature mortality as a result of unhealthy diets, drawing on studies that looked at increased incidence of cardiovascular diseases, diabetes and cancers. Each premature death was valued by the global mean value of a statistical life (VSL),

[10] The model (available at https://iiasa.github.io/GLOBIOM) represents the mainland-use sectors, including agriculture and forestry, with a high spatial resolution. It covers major GHG emissions from agriculture, forestry and other land use. GLOBIOM was initially developed to assess the impact of climate change mitigation policies in land-based sectors, including biofuels, but is increasingly being implemented for a wide range of sustainable development goals, including those related to water.

[11] The Global Burden of Disease programme is a tool that provides a comprehensive picture of mortality and disability across countries, time, age and sex. It quantifies health loss from hundreds of diseases, injuries and risk factors, so that health systems can be improved and disparities eliminated. It is managed by IHME. No uncertainty ranges are given for the estimates cited. A request has been made to IHME to obtain these ranges.

based on the method recommended by the OECD (OECD, 2012). This study came up with an average estimate that was much larger than that of FOLU. The total figure was USD 11 trillion, with a range from USD 3 trillion to USD 39 trillion.

A key factor explaining the difference between the two studies was the use of VSL as opposed to DALYs to value the health impacts. VSL is an estimate of the loss in monetary terms of a premature death, whereas a DALY is the loss of a life year due to health impairment. The average VSL value used in the UNFSS study was around USD 1 million. This would be equal to 56 DALYs using the DALY value in the FOLU report. The relationship between mortality and DALYs is complex, but the number normally associated with a death is much lower than 56 DALYs. As an example, the European Environment Agency uses 10 years of lost DALYs for one $PM_{2.5}$ death (EEA, 2020), and one of the main papers behind the UNFSS report estimates 23 DALYs for each death (Afshin *et al.*, 2019). This explains why the VSL-based estimate yields far higher costs than an estimate based on DALYs.

The uncertainty behind the UNFSS study is mainly the result of a wide range for the value of VSL and not so much uncertainty about the total number of premature deaths, which is relatively narrow.

At country level, estimates of the loss of DALYs for dietary reasons can be obtained from the Global Burden of Disease programme for years up to 2019. The valuation of a DALY is subject to some debate (see next section), but a range can be obtained based on accepted methods.

The policy issues arise in estimating: i) the change in diet as a result of the policy and ii) the benefits of the change in terms of reduced loss of DALYs or lower mortality. There is now work that gives information on the benefits and costs of different diets at national level.

H. Undernutrition

The costs of undernutrition were only estimated in FOLU (2019). DALYs caused by undernutrition were estimated at 101 million a year. Multiplying that by the global average per capita income (USD 17 971 in 2018) gave a total annual cost of USD 1.8 trillion. As with over-consumption and obesity, the estimation of the loss of DALYs was from the Global Burden of Disease programme tracked by IHME (2023).

National-level estimates of the loss of DALYs on account of undernutrition are available from IHME. The policy analysis carried out by FOLU assumes that these losses can be eliminated by 2030 through universal food security. This will, however, have to take place under a deteriorating scenario in terms of agricultural output in some places due to climate change. WHO (2014) has made estimates of the additional mortality due to undernutrition resulting from climate change under different climate scenarios for 2030 and 2050. Measures to meet a target of eliminating undernutrition would have to consider these increases, as well as any changes resulting from demographic or other factors.

I. Air pollution and its effects on health

The costs of air pollution resulting from agriculture-related activities was estimated by FOLU (2019) at USD 372 billion. This was arrived at in two parts: the first dealing with ambient air pollution and the second with household cooking fuels. For ambient air pollution, the figure was USD 386 billion, comprised of multiplying the estimated loss of DALYs due to ambient air pollution (90 million) by the global average per capita income (USD 17 971 in 2018). Of this, 23 percent was attributed to emissions from food and land use systems – the 23 percent figure being the share of GHG emissions coming from these sources. This must be considered a rough approximation, as the correspondence between emissions of GHG and concentrations of ambient air pollutants (principally $PM_{2.5}$, ozone and nitrous oxides) responsible for negative health impacts is not one to one. For the costs of the combustion of household cooking fuels, the number of lost DALYs was estimated at 60 million. This was multiplied by global average per capita income to get a total costs figure. Of this total, 90 percent was attributed to solid cooking fuels from biomass (including agricultural residues, biomass, charcoal, dung and wood). The resulting estimate was USD 970 billion. Together, the two sources represented a cost of USD 1.3 trillion.

Estimates of the costs of both ambient and domestic air pollution are available from IHME for most countries. The difficulties arise in estimating the decline in DALYs or premature morality as a result of different measures, particularly those related to ambient air quality, such as cleaner transport systems and reduced burning of agricultural wastes. The reduction in emissions of key pollutants combines with atmospheric and chemical factors to determine the changes in concentrations of such pollutants across the national land area (and beyond). To derive estimates of changes in concentrations normally requires dispersion modelling, but some approximation tools are being developed by WHO. For the domestic combustion of fuels, it is relatively easy to establish a link between reductions in the use of biomass for cooking and heating and health benefits.

J. Antimicrobial resistance

FOLU calculated the costs of microbial resistance from two global studies. The first, a study by the Research and Development (RAND) Corporation, estimated the loss of global gross domestic product as a result of AMR from increases in HIV, tuberculosis, malaria and infections from E. coli, S. aureus and K. pneumonia at USD 1.3 trillion (RAND Europe, 2014).

This annual cost was taken from the projected present value of losses from 2010 to 2050, combined with the share accounted for by food systems in the United States, which was estimated at 23 percent (Centers for Disease Control, 2013). The 23 percent figure was applied globally to give a total cost estimate of USD 300 billion.

The losses included are only those of lost productivity and do not cover mortality or morbidity costs. In this respect, they are an underestimate compared with the costs of unhealthy diets, air pollution and under nutrition. Another estimate indicates that health service costs could add about 57 percent to the lost productivity cost (Smith and Coast, 2013). The OECD estimates that treating AMR complications could cost about USD 3.5 billion a year in the 33 high-income countries is covers (OECD, 2018b). In RAND Europe (2014), there are, in fact, seven scenarios reflecting different rates of increase in resistance over time, giving a range of annual costs. The figure in FOLU (2019) is middle of the range, which spans USD 437 billion (no increase in rate of resistance) to USD 6 trillion (absolute resistance). Estimates are also broken down into five world regions (high income, Eurasia, Middle East and North Africa, Latin America and the Caribbean, and sub-Saharan Africa).

Estimates at national level of the costs of AMR are a research area, as is estimating the effectiveness of measures to reduce AMR. As far as agriculture is concerned, the OECD notes that extensive use of antimicrobials in livestock production makes agriculture a critical sector in the fight against AMR. At the global level, antimicrobial consumption in livestock production is predicted to increase by 70 percent by 2030 due to increased demand for meat and changes in livestock production, particularly in low- and middle-income countries.

A review by OECD (2019) indicates that in the countries it covers, three out of four deaths could be averted by spending just USD 2 per person a year on measures as simple as handwashing and more prudent prescription of antibiotics. The same report also notes that policies in other areas, notably to promote the prudent use of antimicrobials in agriculture and livestock production, also play a critical role in combating AMR as part of a “One Health” approach. FAO (2016) published an action plan on AMR in the food and the agricultural sector, which identifies four key pillars: awareness, surveillance and monitoring, governance and the promotion of good practices. It does not, however, evaluate the costs and benefits of different actions to reduce AMR in the sector.

K. Pesticide pollution

A global estimate of the costs of pesticide pollution was based on a loss of 0.02 DALYs per kilogram of insecticide, herbicide, fungicide and bactericide applied and a total application of these chemicals of 4 million tonnes in 2016 (Fantke and Jolliet, 2016). This gives a total loss of 80 000 DALYs, each of which is valued in FOLU (2019) at USD 17 971, resulting in a total cost of USD 1.4 trillion.

Rather than use the global estimates, it is also possible to obtain them from a country-specific study, such as that of Nepal (Atreya, 2008; TEEB, 2018). The study estimated the health costs associated with pesticide exposure in rural central Nepal based on data collected from 291 households from January to June 2005, taking into account household demography, personal characteristics, farm size and characteristics, history of pesticide use, history of chronic illness and property of the households. The costs of illness combined with the costs of averting action were used to estimate the cost of pesticide use. Households bear an annual health cost of NPR 287 (USD 4) as a result of pesticide exposure (10 per cent of annual household expenditure on health care and services). These costs vary with fungicide exposure. A 10 percent increase in hours of exposure increases costs by about 24 per cent. Taking into account the abatement costs, the total annual economic cost of pesticide use for the population of the Panchakhal and Baluwa Village Development Committees was estimated to be NPR 1 105 782 (USD 15 797) per year in the study area, equivalent to 55 percent of the annual development and administrative budgets the two village development committees receive from the Government of Nepal.

L. Plastic pollution

The UN Environment Trucost study quantified the environmental impacts associated with plastic use by using LCA techniques and then valued them using a mixture of methods, discussed below for each component (UNEP, 2014). Impacts include GHG emissions, water abstraction, air, water and land pollutants from the extraction of natural resources to their conversion into plastic feedstock. They also include the end-of-life impact of chemical additives in plastic leaching into the environment, the loss of amenity caused by litter, the economic cost of litter to the marine industries and the ecological cost associated with the loss of species. The study is a good example of combining LCA with cost estimation of the different impacts. The cost methods used were as follows:

- GHG: A social cost of carbon of USD 113/tonne was applied based on estimates for the United Kingdom of Great Britain and Northern Ireland (market-based method).

- Air pollution: Based on a literature review of damages, estimates per tonne of emissions in different regions were derived. Figures are global averages (mainly market-based methods).
- Disamenity: The localized impacts of landfill and littering activity that generate negative reactions were derived from hedonic studies of the loss of value to sites, depending on the presence of these activities. Figures are region specific (stated and revealed preference methods).
- Water scarcity: This was valued using the water consumption model described above. Figures are region specific (stated and revealed preference methods).
- Land and water pollution: The study used LCA models that quantify the health and ecosystem damage of different pollutants released to land and water. Health impacts are calculated in DALYs and ecosystem damages in ecosystem damage potential (EDP). Based on secondary literature, Trucost derived a value for DALYs and EDP based on societal willingness to pay. Figures are global averages (methods for health impacts are market based; methods for ecosystem damages are revealed and stated preference based).
- Plastics in marine environments: This covered the economic impact on fisheries and aquaculture, tourism and the opportunity cost of volunteer time, as well as the entanglement and ingestion impact on marine species. The UN Environment Trucost study used secondary literature on the economic impact of plastic and on the quantity of marine species impacted by plastic entanglement and ingestion. Willingness-to-pay studies were used to assess the value that society places on marine species. Figures are global averages (methods are stated preference based).

The overall cost of USD 75 billion a year is broken down by source, so regional estimates can be obtained. The study is an interesting example of how data can be taken from a range of sources to obtain estimates of costs that are region or even country specific. The methods, however, are subject to considerable uncertainty and the study does not provide ranges. The use of such methods is referred to as “benefit transfer” and its application for country-level studies is discussed further in Section 4.1 in the main report.

M. Rural poverty

The costs of rural poverty were measured by FOLU (2019) as the amount by which the income of the rural poor employed in agriculture fell below the poverty line of USD 5.5/day. The average gap between this line and actual income for this group was estimated at USD 803/year and the number of people in this category was about 1 billion, giving a total cost of USD 0.8 trillion.

Each country will have its own, more accurate estimates of the poverty gap and the number of people engaged in agriculture (and their dependents) who fall into this gap. The evaluation of policies to reduce this gap is a major part of all countries’ rural development programmes and include the other areas covered in the true cost assessment, such as environmental gains, health benefits and so on.

N. Food waste

Food waste is a major issue, with implications for GHG and environmental damage. Globally, around one-third of all food, or 1.6 billion tonnes, is wasted, ending up in landfill or clogging drainage systems and harming the environment as a whole. Lowering waste would reduce the amount of food that needed to be produced and transported, thus reducing the amount of land under agriculture (28 percent of the world's agricultural area is used to produce food that is wasted), GHG and local air pollutants (FAO, 2013).

FOLU (2019) estimated the costs of food waste in a simple way. It took the value of total food production at the end-use level as USD 3.7 trillion and applied a 32 percent loss rate to arrive at a loss in cost terms of USD 1.2 trillion. The figures used are from FAO sources.

FAO (2018) has undertaken some interesting studies of the costs and benefits of measures to reduce the amount of waste, focusing on losses in storage and through inadequate methods of transportation, notably in India. They show a significant reduction in GHG (especially from better rice management) at a modest cost when the value of the saved food produce is taken into account. What are still lacking are comprehensive studies of the benefits and costs of programmes to reduce waste at the household and retail level through a range of behavioural interventions.

O. Fertilizer leakage

Fertilizer leakage is a combination of over-application and inadequate management of run-off from the land. The FOLU report took the total global amounts of nitrates (110 million tons) and phosphates (48 million tons) and applied average leakage rates of 44 percent for nitrates and 50 percent for phosphates (Johnston and Roberts, 2015; Yara International, 2015). To the losses, the research team applied the average retail price for phosphates in 2019 of USD 135 per tonne and World Bank data of USD 74 per tonne to arrive at a value of USD 8 billion.

The cost of fertilizer leakage is underestimated by this method, as no account is taken of the losses to ecosystem services resulting from the leakages. These include eutrophication from nitrate releases, impacts on marine life and releases of nitrous oxides, which are a GHG. These costs need to be included in the estimation of the true cost of fertilizers.

P. Uncertainties and limitations of the costs

The review of different categories of cost demonstrates considerable uncertainties in the estimates that have been made. Unfortunately, not all studies report these, which makes them less useful. In this review, an attempt has been made to obtain ranges for the estimates, and some efforts are ongoing (sources have been contacted). Where ranges are presented, as in Hendricks *et al.* (2023), the largest uncertainties are for health costs, with the lower bound 13 times the upper bound. The equivalent figure for economic costs is ten times and for environmental costs is three times.

These figures indicate that the use of the data needs to be connected to the policy purposes for which it is intended. It is also important to bear in mind that not all cost components are measured in money terms, so are not included in the aggregate. In Table 1, it was noted that a number of impacts were not covered in the global studies, notably, costs arising from zoonotic diseases. Animal-to-human transmission is the source of 75 percent of infectious diseases, and livestock rearing and wildlife trade are both significant drivers of global biodiversity loss (Taylor, Latham and Woolhouse, 2001). The harvesting, transport and trade of wild meat and the intensive rearing of livestock have both been linked to the emergence and spread of zoonotic diseases (UNEP, 2020). Restrictions on the hunting and consumption of wild meat are being used in several countries as a measure to prevent future episodes, but they have implications for agriculture and deforestation. For example, it has been estimated that replacing wild meat in the Congo Basin with livestock such as cattle would mean converting 25 million hectares of forest into pastureland (University of Oxford, 2020).

Another set of impacts that has not been well covered relates to the implications of current agrifood practices for biodiversity. It has been shown that in habitats that have faced pressure from combinations of plantation, cropland, pasture, infrastructure and urban land expansion, species richness was reduced by an average 77 percent and total abundance by 40 percent (Newbold *et al.*, 2015).The costs associated with these changes, however, have not been comprehensively evaluated.

Q. Tools used to obtain the estimates

A very wide range of tools has been used to estimate the impacts of agrifood systems examined here and to cost them. On the physical side, they involve estimating the direct and indirect effects in a particular environmental, health or economic category from all activities relating to that category. For example, with GHG, the objective is to track all emissions associated with the production, distribution and consumption of food. Thus, in 2020, emissions of GHG from forestry and other land-use change accounted for about 9.5 Gt CO_2e, but agrifood systems as a whole were responsible for around 18 Gt CO_2e – or around one-third of all emissions that year when all links to supply chains were accounted for (UNEP, 2022). Similarly, in the case of pesticide pollution, the health effects were traced through the people directly affected by the pesticide at their place of work, as well as those exposed to it in the wider environment.

A key tool in making the widest estimation of impacts is LCA. It is defined as “a systematic set of procedures for compiling and examining the inputs and outputs of materials and energy and the associated environmental impacts directly attributable to the functioning of a product or service system throughout its life cycle” (IOS, 2006). LCA examines physical impacts across the value chain. For each of these steps, an inventory is made of the use of material and energy and the emissions to the environment, creating an environmental profile that allows identification of the weak points in the life cycle of the system studied. These weak points are then made the focus for improving the system from an environmental point of view. In most cases, the impacts are only reported in physical units and not converted into monetary terms. There are several cases, however, where a value can be attached to the physical units. Examples, include emissions of GHG or the loss of DALYs per kilogram of pesticide. Use of LCA for plastics pollution has also been developed to obtain the monetary values of damages.

Other tools that can used in the assessment of the physical units include production functions for crops and livestock, epidemiological models and economy-wide models. Epidemiological models have been developed to estimate mortality and morbidity rates in a given population as a function of age, ethnicity, socioeconomic characteristics and exposure to environmental pollutants. From this are recovered the impacts of different factors on disease and mortality. Such functions have been used to calculate the aforementioned mortality and morbidity relationships for obesity, undernutrition and exposure to air pollution. As they are highly specialized exercises, the results are taken from databases that have conducted such studies.

Economy-wide models (also referred to as computable general equilibrium models, or CGEs) are used when a policy is introduced with ripple effects across many sectors, even though the focus is on one sector (such as incentives for biofuels), or when an external change is expected to have wide-ranging repercussions (such as climate change). The models cover all the major sectors of the economy and estimate the consequences of the change not only in the sector in question, but more broadly. Examples include the effects of climate change and water scarcity on crops and livestock, as well as on the income of poor groups in society (Skoufias, Rabassa and Olivieri, 2011); the European Commission's MAGNET model, used to assess the impacts of agriculture, land-use and biofuel policies on the global economy (Boulanger *et al.*, 2016); and the assessment of socioeconomic impacts of improving agriculture water use efficiency (Liu *et al.*, 2017). CGEs can be important tools for agrifood policy analysis, but are not at the core of the TCA approach, which seeks to estimate the gap between the market costs and the true costs in the economy as it currently functions. The True Cost Handbook, for example, does not refer to such tools or to similar ones, referred to as system dynamics (TEEB, 2018).

The use of these tools in data-scarce contexts will always be problematic. This makes using a number of applications of the methods set out in Figure 1 and Table 2 difficult. LCA is the basis of estimating impacts in physical terms for market- and cost-based approaches, but it needs emissions data across the value chain. This is often the limiting factor in conducting local estimates, more so than the economic data. Similarly, production functions and epidemiological models are data intensive and can act as a constraint on applying the methods described here. In the long run, countries need to build up the databases that permit the use of these tools. When they are not available, alternative methods are proposed in Sections 3.3 and 3.4.

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