Food wastage footprint
Full-cost accounting

Final Report

Food and Agriculture Organization
of the United Nations

ISBN 978-92-5-108512-7 (print)
E-ISBN 978-92-5-108513-4 (PDF)

About this document

The Food Wastage Footprint (FWF) is a project led by Nadia El-Hage Scialabba, Climate, Energy and Tenure Division. Phase I of the FWF project modeled the impacts of food loss and waste on climate, land, water and biodiversity. Phase II of the project, commissioned to the Research Institute for Organic Farming (FiBL), Switzerland, expanded the project to include modules on full-cost accounting of societal externalities of food wastage. This report is part of a series of publications produced by FAO to raise awareness of the serious impacts of food wastage: (i) Food Wastage Footprint: Impacts on Natural Resources (FAO 2013); (ii) Toolkit: Reducing the Food Wastage Footprint (FAO 2013); and Mitigation of Food Wastage: Societal Costs and Benefits (FAO 2014). With this volume, FAO aims to establish the basis for natural resources accounting in the food and agriculture sector, including the cost of natural resources degradation and its impact on social well-being.

Acknowledgements

FAO wishes to thank FiBL staff Adrian Muller, Christian Schader, Uta Schmidt and Patricia Schwegler for the modeling work, as well as Daniel Fujiwara, London School of Economics and SImetrica (UK) for the social well-being work. Thanks go to FAO colleagues Devin Bartley, Jan Breithaupt, Barbara Herren, Mathilde Iweins, Jippe Hoojeveen, Soren Moller, Francesco Tubiello, Ronald Vargas, Harry van der Wulp, as well as Nelson Sabogal, UNEP. Francesca Lucci is thanked for the design of all products of the FWF project, including videos and publications.The FWF project was undertaken with the generous financial support of the Federal Republic of Germany.

The FWF project products are available at: www.fao.org/nr/sustainability/food-loss-and-waste

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FOOD WASTAGE FOOTPRINT

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Acronyms

BHPS	British Household Panel Survey
CBA	Cost-Benefit Analysis
CBD	Convention on Biological Diversity
CS	Compensating Surplus
DCM	Damage Cost Method
EPA	Environmental Protection Agency
ES	Equivalent Surplus
FAO	Food and Agriculture Organization of the UN
FCA	Full-Cost Accounting
FUND	Framework for Uncertainty, Negotiation and Distribution (Climate)
GAMS	General Algebraic Modelling System
GDP	Gross Domestic Product
GHG	Greenhouse Gas
HLPE	High-level Panel of Experts on Food Security and Nutrition
IFGB	Institute for Development of Agricultural Economics
IV	Instrumental Variable
IPCC	Intergovernmental Panel on Climate Change
IWMI	International Water Management Institute
NBI	National Biodiversity Index
OECD	Organisation for Economic Co-operation and Development
OLS	Ordinary Least Squares
OMAFRA	Ontario Ministry of Agriculture and Food
PAGE	Policy Analysis for the Greenhouse Effect
P	Phosphorus
RP	Revealed Preference
SCC	Social Cost of Carbon
SOL-m	Sustainability and Organic Livestock model
SP	Stated Preference
SWB	Subjective Well-Being
TEEB	The Economics of Ecosystems and Biodiversity
TEV	Total Economic Value
WBCSD	World Business Council for Sustainable Development
WTA	Willingness to Accept
WTP	Willingness to Pay
WV	Well-being Valuation
WVS	World Values Survey

Executive summary

Approximately one-third of all food produced for human consumption is lost or wasted. The economic costs of this food wastage are substantial and amount to about USD 1 trillion each year. However, the hidden costs of food wastage extend much further. Food that is produced, but never consumed, still causes environmental impacts to the atmosphere, water, land and biodiversity. These environmental costs must be paid by society and future generations. Furthermore, by contributing to environmental degradation and increasing the scarcity of natural resources, food wastage is associated with wider social costs that affect people's well-being and livelihoods. Quantifying the full costs of food wastage improves our understanding of the global food system and enables action to address supply chain weaknesses and disruptions that are likely to threaten the viability of future food systems, food security and sustainable development.

This document introduces a methodology that enables the full-cost accounting (FCA) of the food wastage footprint. Based on the best knowledge and techniques available, FCA measures and values in monetary terms the externality costs associated with the environmental impacts of food wastage. The FCA framework incorporates several elements: market-based valuation of the direct financial costs, non-market valuation of lost ecosystems goods and services, and well-being valuation to assess the social costs associated with natural resource degradation.

To demonstrate the proposed FCA methodology, this study undertakes a preliminary assessment of the full costs of food wastage on a global scale. In addition to the USD 1 trillion of economic costs per year, environmental costs reach around USD 700 billion and social costs around USD 900 billion. Particularly salient environmental and social costs of food wastage include:

- 3.5 Gt CO2e of greenhouse gas emissions. Based on the social cost of carbon, these are estimated to cause USD 394 billion of damages per year.
- Increased water scarcity, particularly for dry regions and seasons. Globally, this is estimated to cost USD 164 billion per year.
- Soil erosion due to water is estimated to cost USD 35 billion per year through nutrient loss, lower yields, biological losses and off-site damages. The cost of wind erosion may be of a similar magnitude.
- Risks to biodiversity including the impacts of pesticide use, nitrate and phosphorus eutrophication, pollinator losses and fisheries overexploitation are estimated to cost USD 32 billion per year.
- Increased risk of conflict due to soil erosion, estimated to cost USD 396 billion per year.
- Loss of livelihoods due to soil erosion, estimated to cost USD 333 billion per year.
- Adverse health effects due to pesticide exposure, estimated to cost USD 153 billion per year.

FCA gives an indication of the true magnitude of the economic, environmental and social costs of food wastage: USD 2.6 trillion annually, roughly equivalent to the GDP of France, or approximately twice total annual food expenditure in the USA. However, these results must be treated with a degree of caution as the calculation of non-market environmental and social costs of food wastage on a global scale requires a number of strong assumptions. The total environmental and social costs that have been calculated in this study are most likely to represent an informed underestimate as many impacts could not be included because of a lack of data or appropriate methodologies.

Further research should focus on specific contexts, at national or supply chain level. The FCA framework can serve as a template for more targeted research to inform mitigation policies. To assess the optimum level of food waste reduction for societies, it will be important to incorporate economic equilibrium analysis to simulate the interactions between food supply, prices, income and welfare in a dynamic economy. A further priority is to improve aspects of the social cost estimates. For instance, it is difficult to determine the exact impact of environmental conditions on individual well-being; many of the environmental variables associated with food wastage are highly correlated while others may not accurately measure the effect on well-being that is intended. We focus on three pathways to value environmental impacts on conflict, health and livelihoods, but there are likely to be many more. While our preliminary estimates are based on the best methods and data that are currently available, future work may be able to add missing pieces of the puzzle to further refine current estimates.

By unveiling the hidden environmental and social costs of food wastage, FCA provides an illustration of the market distortions in the global food system. These costs are real and they demand action. Despite the uncertainties that remain, it is apparent that food waste mitigation makes sense from economic, environmental and social perspectives. For future population scenarios, food wastage mitigation could play a crucial role in assuring food availability while respecting critical planetary boundaries.

Introduction

This document presents the FAO methodological approach for full-cost accounting (FCA) of food loss and food waste, the combination of which is referred to hereafter as "food wastage". While (Gustavsson, Cederberg *et al.* 2011) quantified food wastage volumes and the (FAO 2013a) quantified the environmental impacts of wastage, this study provides a first quantification of some of the costs due to these impacts. It also contributes to the ongoing intensive debate on food wastage, its causes, impacts and mitigation measures – a debate that involves a cross-section of stakeholders, from grassroots organizations to governments (FAO 2013d, FAO 2013b, HLPE 2013, HLPE 2014).

The FCA of a project, action or situation aims at accounting for all of the priced and unpriced costs and benefits that come with it. For food wastage, FCA can monetize the inputs of unpriced natural resources to food supply chains, as well as the welfare costs related to loss of natural resources and ecosystem services. FCA can give a more realistic picture of the apparent profitability of unsustainable production and consumption by indicating which costs are not internalised and informing about the risks and opportunities associated with depleting natural resources and ecosystems. By providing estimates of those external costs, this paper raises awareness of the full societal costs of food wastage – costs that lay well beyond the direct market price of the lost produce. Once those costs are known, it is possible to understand the true gains that may come from mitigation of that food wastage.

Through the concrete food wastage example, this FCA methodology and its preliminary results point to areas for expert discussion and further research. It takes an economic approach to making a valuation of the environment in order to assess, to the extent possible, the total economic value of air, water, land, ecosystems, biodiversity and other resources lost, polluted or consumed due to food wastage. In order to account for social costs, which is particularly challenging, this FCA version integrates a non-market valuation approach to social well-being related to the environmental externalities of the food and agriculture sector that result from food wastage.

This report starts by presenting the methodological framework and general approach taken to monetize societal externalities of food wastage. It then presents the detailed methodologies used for estimating the costs of environmental and social impacts. This is followed by the results of the full-cost accounting of food wastage. Finally, it identifies issues that require further research and discussion.

1. Full-Cost Accounting Framework

1.1 Introduction

Approximately one-third of all food produced for human consumption is lost or wasted. Important steps have been taken to quantify food wastage volumes differentiated by regions and commodity groups (Gustavsson, Cederberg *et al.* 2011), and to quantify resulting environmental impact of that wastage (Kummu, de Moel *et al.* 2012, FAO 2013a). This work has determined that the monetary value of the actual food wasted amounts to USD 936 billion[1], yet this does not account for environmental and social costs of the wastage that are borne by society at large. Until now, due to lack of understanding of the full magnitude of the costs of food wastage, it could seem more profitable to let food rot, at both post-harvest and distribution levels, than to take steps to mitigate the wastage. Understanding the big picture of the impact of food wastage should prove to be what is needed to promote investments in food wastage reduction measures, including supporting financial, policy and other incentives to reduce barriers to effective food wastage reduction which have largely been lacking until now.

A large part of the environmental impact of food wastage occurs, and can be measured, at the agricultural production level (Kummu, de Moel *et al.* 2012, FAO 2013a), but there are also effects at later levels in the value chain. For example, fossil fuel used for storage, processing and distribution of food needs to be taken into account for any food wasted at consumption level. Food wastage also has environmental impact at the "end-of-life" level, such as from methane gas emissions in landfills.

In addition to these direct effects of wasted or lost food volumes, it is critical to recognize that food wastage has more complex interactions in the food system – interactions that affect food prices and availability, production patterns and input use. It is at this more complex level that the connection between food wastage and hunger or reduced livelihoods (e.g. due to reduced access to natural resources) must be assessed. A comprehensive framework for the full-cost accounting of food wastage is therefore needed for informed decision-making.

1 The published figure in (FAO 2013a) is USD 750 billion and referred to 2009 producer prices. This figure is hereby corrected by using average import/export market prices (instead of producer prices) from 2005-2009 for the valuation of post-production wastage, using producer prices for the pre- and post-harvest stage only, resulting in an estimate of 846 billion, which is then transferred to year 2012 dollars.

1.2 Framework for analysis

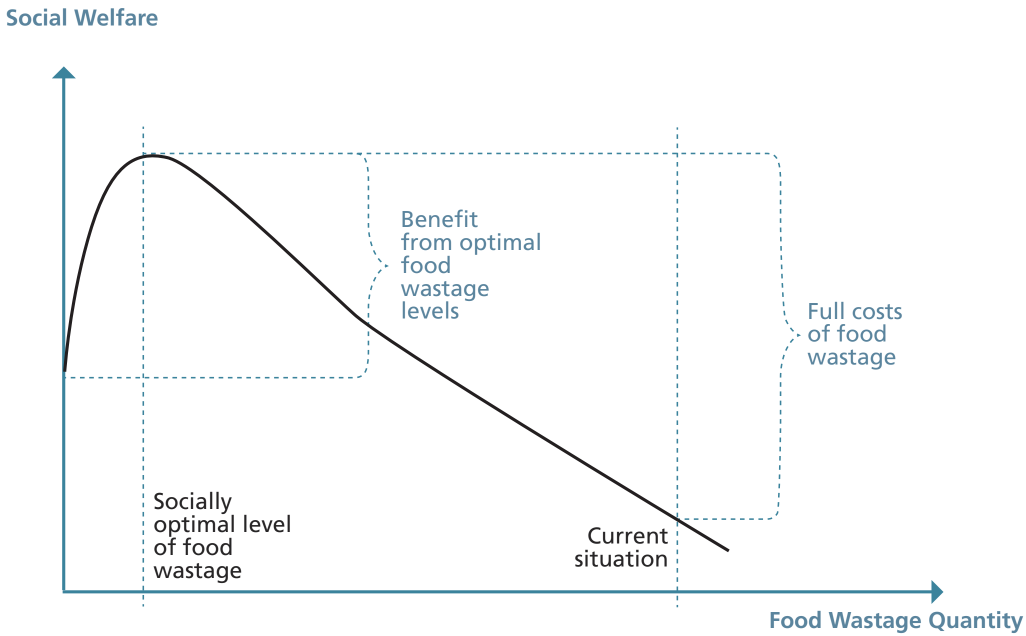
1.2.1 Economic equilibrium analysis

Ideally, assessment of the costs of food wastage to society would be based on the complete scenario of supply and demand for agricultural commodities, the inputs needed for their production, and the resulting totals of commodity and input volumes and prices. In an ideal case with full information and no external costs, this would describe a general market equilibrium. However, in a non-ideal situation with external costs and lack of information, the general market equilibrium provides a framework for capturing effects of changes in quantities on prices, scarcities, supply and demand, and all impacts and their costs and how they relate to each other. Adopting an approach inspired by economic equilibrium analysis means that the system investigated is organized around production volumes and their prices (i.e. supply and demand for the food commodities concerned). But it also computes the quantities and prices of inputs needed for this production and the quantities and costs of external effects (i.e. "bad outputs") it may cause. Volumes and prices are linked via "elasticities" that describe how demand and supply for a product change with its price. In principle, this approach not only covers environmental costs, it also covers certain "social" costs as well as benefits, which can include effects on income due to price changes, health damage due to pesticide use and resulting lost labour productivity or changed labour demand. This is possible only as long as the various elements can be captured by economic concepts and included via prices or costs. The necessity to adopt such an equilibrium framework or the underlying ideas for food wastage cost accounting as emphasized in publications of the High-level Panel of Experts on Food Security and Nutrition (HLPE 2013) and by the Government of the USA (HLPE 2014). (Rutten 2013) also discussed the necessity such a framework, detailing which insights can be expected from it and how one may proceed to achieve it.

Through adopting this general equilibrium framework, the full costs of food wastage can then be defined as the difference between the aggregate net welfare in society (i.e. total benefits minus total costs) derived from the current food system (i.e. with food wastage) and the aggregate net welfare from a hypothetical food system with less food wastage. The food wastage level that would be optimal is when the welfare difference is maximal between the current and hypothetical food systems. This accounts for the fact that a zero-food-wastage world is not socially optimal in economic terms, while a lower but positive level of food wastage is (see Figure 1).

Economically speaking, the optimal level of food wastage is reached when the costs of additional reduction of food wastage become higher than the gains from such additional reduction. An example would be the cost of additional fossil fuel and greenhouse gas emission required for faster transportation of some food commodities as compared with the cost reductions due to reduced food wastage.

Figure 1: Economic approach to total welfare in relation to food wastage quantities



But there are also non-economic reasons for wastage, as consumers gain utility from increased choice; thus, food waste is one consequence of the utility derived from choice (de Gorter 2014). But, it also should be noted that a growing body of evidence from the psychological sciences suggests that too much choice can impact negatively on decision-making performance (Iyengar 2010).

Using this economic equilibrium approach to total welfare with current and reduced levels of food wastage would identify the net costs of food wastage volumes beyond the optimal level of food wastage. An alternative would be to compare current welfare to the welfare in a situation without food wastage. This would then estimate the net costs of food wastage in relation to a zero-waste case.

This second estimate will be lower than the first, as some food wastage bears net benefits as well as costs, namely in situations where further reductions would cost more than related gains. This is the case, for example, if ensuring zero post-harvest losses would result in huge costs to ensure safe storage even under rare but extreme weather conditions, such as exceptionally prolonged humid periods. In any case, a crucial part in this exercise is the estimation of total welfare and changes thereof.

1.2.2 Opportunities and challenges of the general equilibrium approach

Ideally, the full-cost accounting of food wastage should be implemented in a computable general equilibrium model. However, data available on food wastage volumes and related environmental effects (Gustavsson, Cederberg *et al.* 2011, FAO 2013a) have been derived from outside an economic equilibrium context, because crucial information for full implementation of an equilibrium model is lacking[2]. In order to assign concrete values for certain cost categories of food wastage, considerable restrictions, simplifications and approximations have been undertaken. Hence, the linear approach taken in this document, and enumerated in Chapters 2 and 3, offers linear approximation to parts of the full general equilibrium framework.

1. System boundaries

The system boundaries for the analysis need to be made explicit. Thus, they include all parts of the food system where wastage may occur, meaning they incorporate the following:

- ***The whole supply chain.*** This goes from agricultural production, storage, food trade, transport, distribution, consumption and the final destination of any food wastage, such as the landfill.

- ***All inputs to these supply chain steps.*** This includes inputs to agricultural production, such as land, fertilizers or pesticides, or inputs to refrigeration storage or transportation, such as electricity and fossil fuels. The necessary quantities of these inputs and the related impacts and environmental costs from their production directly relate to the volumes of agricultural production and, thus, to food wastage volumes.

- ***All outputs.*** This includes "bad" outputs, such as pollution, and all places where impacts and costs of agricultural production and, thus, of food wastage may occur. Outputs also encompass ecosystems, the climate system, local air volumes (to account for pollution from biomass residues burning), water bodies (to account for impacts on water quality) as well as social contexts, such as households (due to changes in food prices).

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2 See Rutten 2013 for further details on this economic approach.

2. Production structure

When assessing the costs of food wastage by comparing the welfare in a situation with current wastage levels with one that has reduced wastage levels (as shown in Figure 1), it is assumed in this methodology that the agricultural production structure does not change[3]. The characteristics of agricultural production are thus the same in both cases (e.g. regarding per hectare levels of irrigation and fertilizer use). In particular, similar yields and intensities in agricultural production are assumed with full and with reduced food wastage. This results in an assessment of the difference between costs of a food system with current wastage volumes in relation to one that is as similar as possible to this original one, besides exhibiting reduced wastage volumes. The full general equilibrium approach would allow for changes in the production structure, but in the approximations adopted in this document, this is not possible. If changes, for example, in irrigation efficiency were incorporated as well, the effect of food wastage reduction on costs would be mixed with the effects from changes in the efficiency and sustainability of the agricultural production structure (even though these are independent of whether food wastage volumes change or not).

3. Production quantities and prices

Equilibrium effects of changes in production quantities and prices are not included in this study. The most important effect of food wastage is that reduced wastage volumes lead directly to reduced demand for agricultural production. In the food wastage context, this means more food needs to be grown during the agricultural production phase to supply a given level of consumption compared with scenario of a context with less food wastage. Assuming similar production patterns results in assuming no changes in production intensity, while land occupation would change with food wastage reduction. In a situation with the current food wastage level, there is increased land use in relation to a context with less food wastage, which leads to an increase in the demand for inputs and, correspondingly, to increased impacts and costs from their use, as compared with the situation with less food wastage. Thus, food wastage leads to increased natural resource depletion (e.g. water, energy, forest), capital use (e.g. machinery, buildings, fertilizer, pesticides) and pollution (e.g. nitrate, greenhouse gases) which contributes to climate change, land, water and biodiversity loss and the degradation of ecosystem services. These environmental impacts have both environmental and social costs.

Arguing from the supply side, reduced food wastage at the producer level would lead to larger supply and correspondingly lower unit prices, which tend to go along with increased demand. Thus, price effects of food wastage reduction at the producer level could even lead to increased food wastage at the consumer level, as food becomes cheaper. Without a full equilibrium framework, it is impossible to capture these various interlinked and opposite effects of price and quantity changes due to food wastage reduction. Figure 2 illustrates the linkages between food wastage, environmental impacts and societal costs. The subsequent section 1.2.3 then describes how this full picture may be approximated to arrive at concrete cost estimates.

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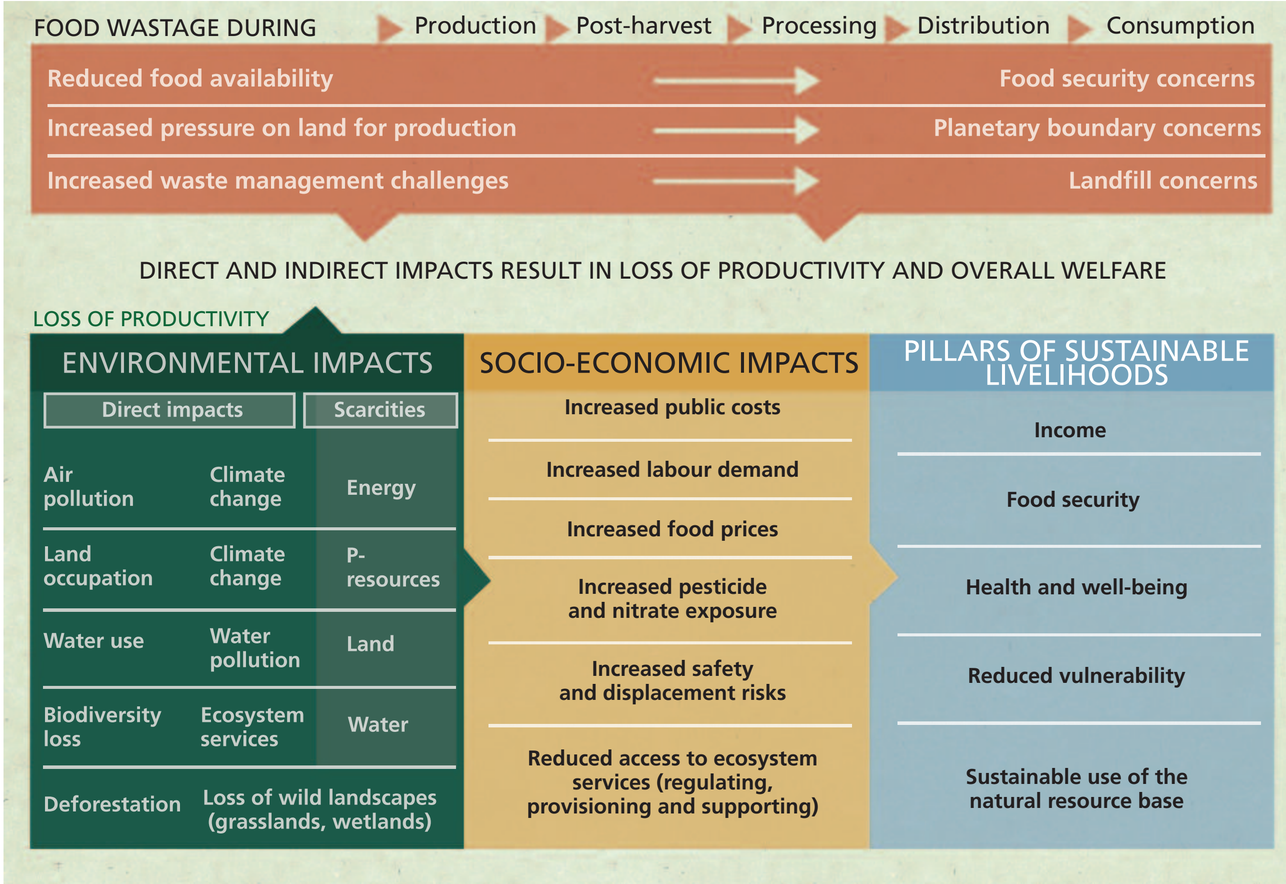
3 Excluded, for example, are variations induced by reducing food wastage resulting in reduced demand, which in turn reduce pressure to keep extraordinary high yields.

Figure 2: Full landscape of the impacts of food wastage on the environment, society and livelihoods



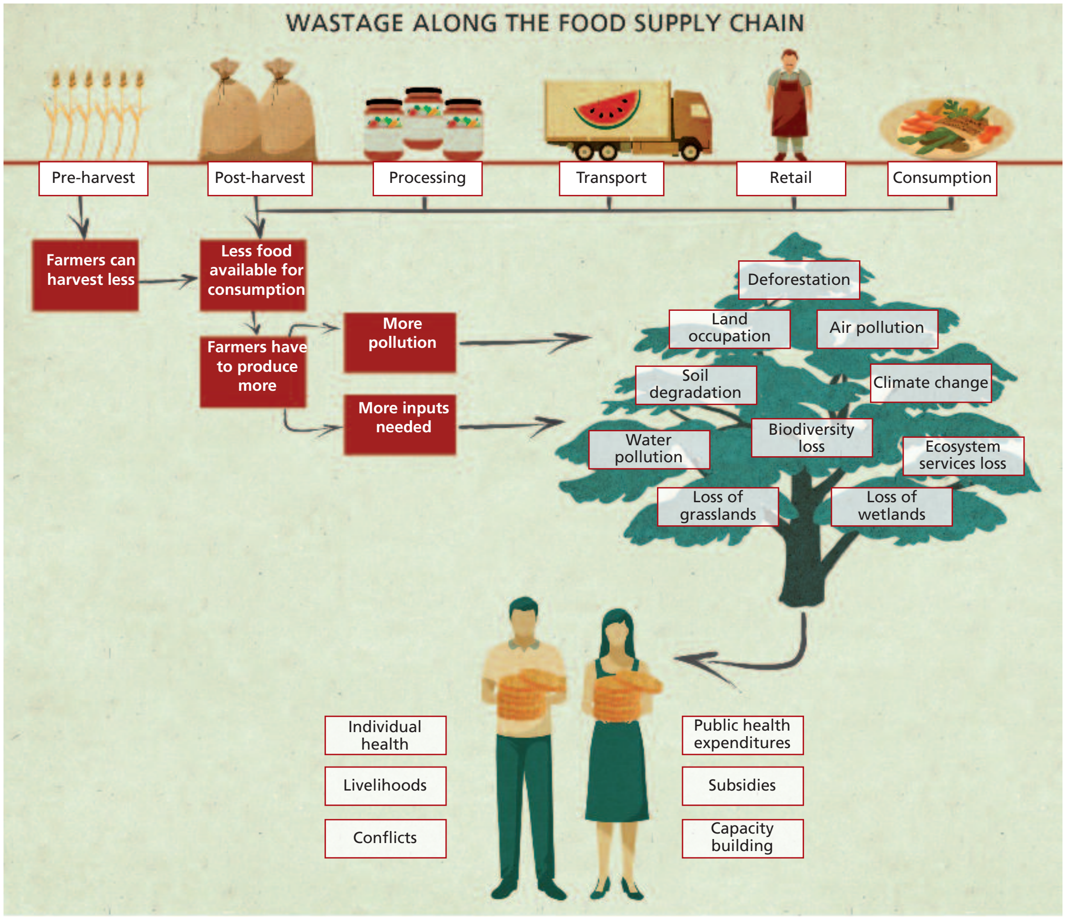
1.2.3 Three levels of approximations

1.2.3.1 Wastage quantities and impacts and costs per unit

Looking beyond the interaction between quantities and prices, the impacts and costs linked to food wastage volumes directly correspond to the benefits from reducing those. The relationship can be assessed via input demand and pollutant emissions and the related costs per unit of wastage.

This first basic approximation of the full costs of food wastage, displayed in Figure 3, corresponds to the direct internal and external costs of food production for food that is eventually lost or wasted at each stage of the value chain. These are the absolute costs of food wastage that can be directly linked to quantities of food lost or wasted.

Figure 3: First approximation of direct impacts of food wastage



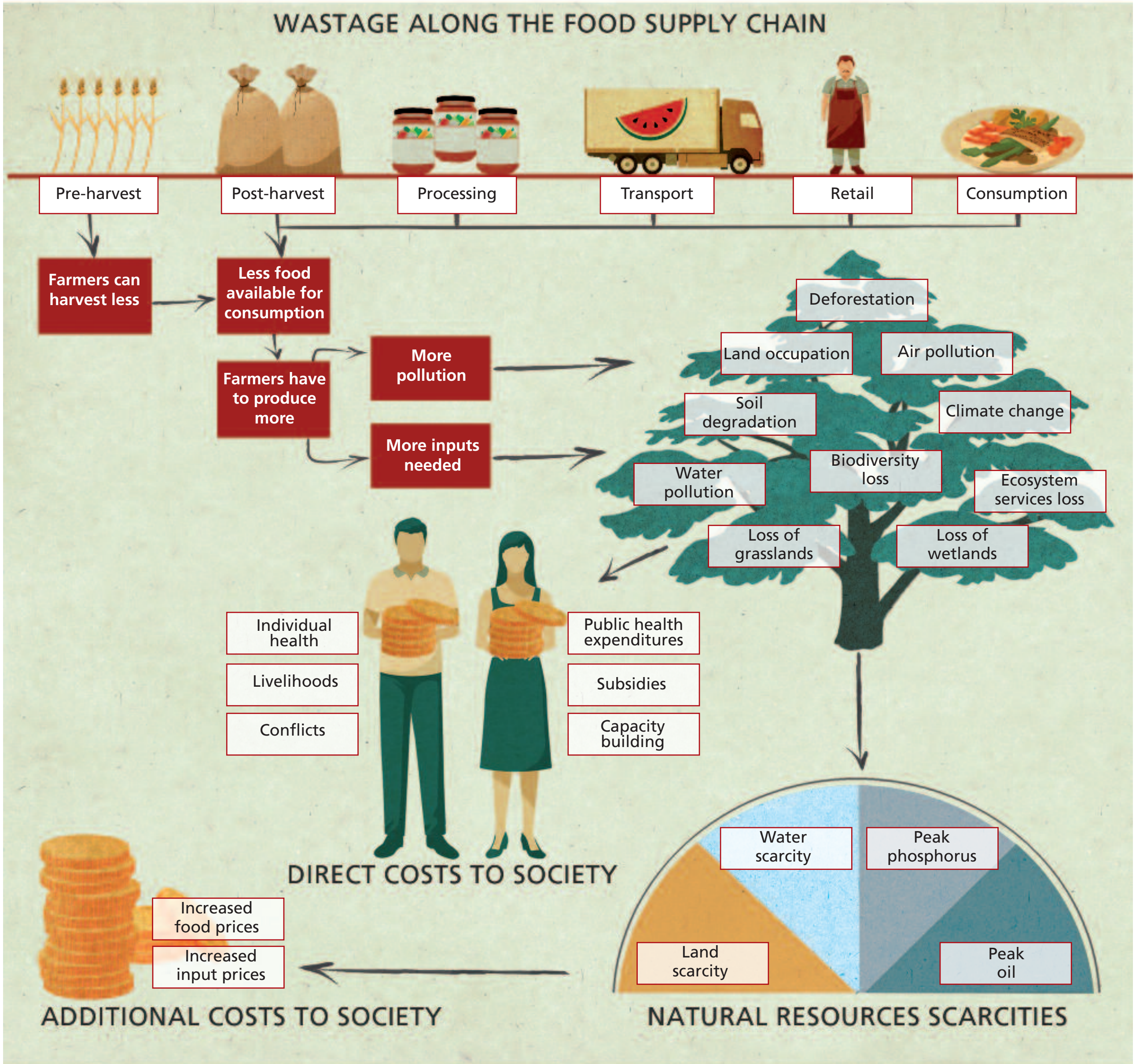
1.2.3.2 Wastage quantities and resource scarcities

A food system that is inefficient in terms of food wastage needs to produce more to supply a given level of consumption. In addition to the absolute costs of food wastage described in section 1.2.3.1., this also places increased pressure on natural resources in total and leads to costs that relate to available resource stocks, not just to quantities used. These relative costs are more complicated to assess as they depend on an assessment of the imminent scarcity of the resources. For example, a relatively large quantity of water wasted where water is abundant will have a smaller cost in terms of increasing scarcity than a relatively small quantity of water wasted in dry regions and seasons. Those costs arise because supply of the resources that become scarcer becomes correspondingly more expensive, thus increasing the costs of agricultural production that depends on these resources and also the costs of their alternative uses, such as drinking water.

These scarcity costs are not covered in the first approximation which refers to direct impacts only. However, they are part of the effects that changes in food wastage volumes have on prices of products and inputs, and how those feedback to production. They also serve as an illustration for the effects at work in the general equilibrium framework. In fact, change in input costs would affect production volumes with corresponding changes in related input use and outputs, impacts and costs. For example, reduction in food wastage reduces water scarcity and, thus, leads to reduced costs per unit of irrigation water. This would make the production of irrigated crops relatively less expensive but would lead to increased supply of crops that have correspondingly higher irrigation demand, which again would affect water scarcity. Even more, not only does resource use affect crop quantities and prices, it also impacts pollution, climate change and the degradation of land and ecosystem services that in turn affect the agricultural production itself in a feedback loop by generally reducing productivity with corresponding consequences on quantities, prices and input use. As with the previous example on prices and quantities, such full equilibrium effects related to quantities and input use are not covered in this document, with the exception of water scarcity. These scarcity costs are depicted in Figure 4.

Thus, fully accounting for scarcity effects is only possible in a full equilibrium model, as linkages between changes in quantities and prices would be included. However, some approximation to some of these effects is possible by relating those aspects to wastage quantities, and treating them in a similar way to the direct costs described in section 1.2.3.1. There are, for example, some estimates on the average costs that a tonne of water used generates with regards to scarcity in a certain context. Multiplying the amount of water wasted due to food wastage with this scarcity cost estimate per tonne of water provides some linear approximation to the water scarcity effects of food wastage.

Figure 4: Direct impacts of food wastage and additional scarcity costs



1.2.3.3 Wastage quantities and stakeholder linkages

A range of other important aspects that could only be covered in a full equilibrium approach have been left aside in this study, due to lack of data for making suitable approximations. This includes, for example, how price changes due to food wastage or food wastage reduction affect household incomes and how this affects household consumption. Another example is the fact that food wastage will affect costs and benefits, depending on how it affects different stakeholder groups and where in the supply chain it occurs.

While increased fertilizer and pesticide use impose negative external costs, agricultural expansion may also provide positive external benefits through the provision of ecosystems services and cultural values related to agricultural landscape. Likewise, the social impacts of food wastage can be positive or negative for various stakeholder groups. Increasing prices for agricultural produce, for example, may affect farmers positively but consumers negatively. Also, the presence of some food wastage may contribute to food security, as part of it could be eaten without any adverse health effects in the case of some societal shocks, for example food that is wasted due to its non-compliance with aesthetic and ease-of-processing quality requirements that do not reflect the safety of the food.

1.2.4 The full-cost accounting framework

This section describes how the general food wastage framework presented in sections 1.2.1 to 1.2.3 has been applied and made operational to arrive at a FCA framework for food wastage. A framework linking environmental externalities of food wastage to the full range of possible impacts, including livelihood aspects, was discussed in an e-forum held in November 2013 (FAO 2013b) and adapted subsequently. Participants in this e-forum also emphasized the importance of factors such as charity donations, obesity, disruption of traditional lifestyles and social unrest in social cost-accounting for the FCA of food wastage. It is debatable whether such aspects can be analyzed rationally and monetized in an economic framework (Fine 2002). An alternative approach calls for refraining from monetization of certain aspects that are outside the economic equilibrium framework, as they would target aspects beyond the economic sphere.

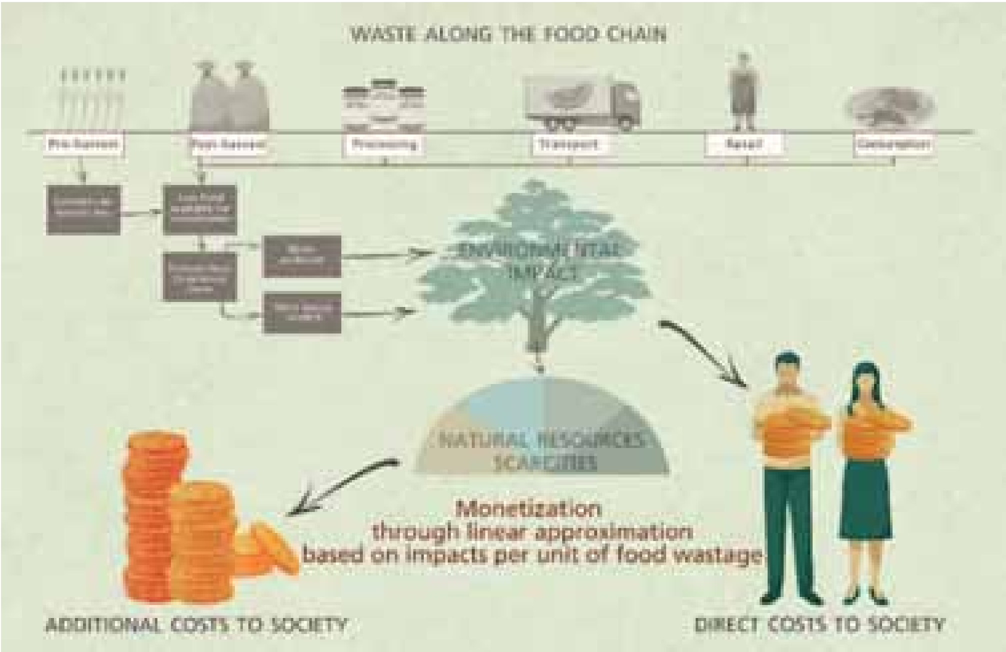
As the aim is to attempt monetization of all possible effects to make them visible and comparable by capturing them with the same monetary metric, different valuation methods were chosen for different aspects. While the valuation of traded goods can be based on the prices paid, such a situation is rarely encountered in the context of environmental goods and services, for which no markets exist and that have no price (e.g. free clean air). This is when alternative valuation methods are needed. Economists offer two main methods of valuation for such non-market outcomes:

- preference valuation methods – values based on people’s revealed or stated preferences;
- well-being valuation approach – values based on observed changes in well-being due environmental changes).

Each method comes with its own relative pros and cons, which can be technical or normative in nature. For example, one particularly contested normative issue that is inherent in all valuation methods is the assumption of substitutability of monetary income and the non-market good, which may include such complex issues as conflicts or social relations (Freeman III 2003, Nussbaum 2010). The different valuation methods and some conceptual background are discussed in detail in sections 1.2.5 and 1.2.6.

The food wastage cost-estimates provided in Chapter 4 are derived by food wastage quantities and cost estimates per unit food wastage. This provides a linear first order approximation to the full equilibrium framework described in sections 1.2.1 and 1.2.2. It also assesses food wastage costs for total wastage quantities, that is, in relation to a zero waste situation. It does not account for the fact that a socially optimal food wastage level based on economic considerations will be larger than zero. Figure 5 provides a graphical presentation of the framework for FCA of food wastage, where the direct impacts of food wastage and the effects of scarcity each lead to costs to society that are monetized by valuation methods based on the impacts per unit of food wastage. In addition, social impacts such as health, livelihoods and conflicts are also monetized.

Figure 5: Valuation method of food wastage costs to society



1.2.5 General concepts behind economic valuations

As described in sections 1.2.1 to 1.2.4, the equilibrium approach provides a theoretical framework for comparing a situation with reduced food wastage to the baseline in order to derive the full costs of food wastage, and offers approximations for implementing this approach in the context of incomplete information. The first order approximation to the equilibrium framework is a linearization of its constituents. This means that valuation is done via multiplication of food wastage volumes related to environmental impact levels, with costs per unit of food wastage volume or environmental impact. This is done without addressing any feedback of changes in prices due to internalization of external costs, and without addressing the labour and other sectoral effects of such impacts (such as in the health sector). The key challenge is to provide reliable estimates of those unit costs.

This is best discussed in a broader context of valuation where the ultimate goal is to assess human welfare (or well-being) and how this is affected by food wastage or its reduction. Theoretically, this is captured in economic concepts of compensating and equivalent surplus.

- Compensating surplus (CS) – the amount of money, paid or received, that will leave the agents in their initial welfare position following changes in their environment.
- Equivalent surplus (ES) – the amount of money, to be paid or received, that will leave the agents in their subsequent welfare position in absence of a change in their environment (Bockstael and McConnell 1980).

Therefore, changes in the environment can be a change in the amount of forests in a country (deforestation), a change in a person's job or a change in total food wastage with related changes in environmental impacts and their effects. Thus, CS and ES are the theoretical concepts behind valuation of the effects of food wastage and its mitigation. And in fact, the total value of food wastage can be more formally defined as the aggregate of compensating measures of benefits and compensating measures of costs. This is akin to the Kaldor version of the compensation test in cost benefit analysis (CBA) (OECD 2006) and, equally, we could measure the values in equivalent measures. In the FCA framework, therefore, valuations must ultimately reflect the impacts of food wastage on human well-being.

In an ideal market context (i.e. in the general equilibrium framework outlined in section 1.2.1) where all goods have a price and consumers hold well-informed rational preferences for a complete set of goods, CS and ES can, in a first approximation, be estimated from consumer demand curves and information on quantities consumed or used. Under the assumption that opportunity costs equal marginal price unit costs, estimates could, in turn, be used as approximations for market values, if such data are available. However, even if they are available, they may not capture the full costs. For example, market values for irrigation water tend to be distorted by government subsidies. In addition, in many cases, market values are not available and other approaches are required.

1.2.6 Valuation methods

The traditional approach to measuring economic value is through data on people's stated preferences. Under this approach, an individual's preferences provide a measure of his/her welfare (termed utility by economists), because "what would be best for someone is what would best fulfil his desires" (Parfitt 1984, p 4). A basic assumption, when using preference data in valuation, is that it is possible to map choices over a number of binary options onto a well-defined utility function and this is the case if preferences are rational (i.e. that they conform to a set of behavioural criteria that assumes transitivity and completeness). If these assumptions are met, then people will behave as if they are maximising some utility function. In addition, for the purposes of valuation, there is need to add a non-satiation assumption (i.e. that preferences are never fully satiated) such that the individual always places a positive value on more consumption. Also, policy-makers may require the assumption that preferences be well-informed, if they are to be used in valuation and policy decisions – although from a purist point of view, economists tend not to make any substantive claims regarding level of information.

Using preference data, compensating and equivalent measures of value (CS and ES) can be estimated for non-market goods in relation to people's willingness to pay (WTP) or willingness to accept (WTA) in actual or hypothetical markets. Table 1 describes the relationship between CS, ES and the preference measures WTP and WTA.

Table 1: Relationship between preference-based valuation measures

	Compensating surplus (CS)	Equivalent surplus (ES)
Welfare gain	WTP for the positive change	WTA to forego the positive change
Welfare loss	WTA the negative change	WTP to avoid the negative change

The following discusses the valuation methods used in the FCA of food wastage in more detail. The methods are presented in two broad categories – preference valuation methods and well-being valuation methods.

1.2.6.1 Preference-based valuation

1.2.6.1.1 Revealed preferences

Generally speaking, where proxy markets exist, the favoured approach to valuation is to estimate WTP or WTA from people's market behaviour using revealed preference (RP) methods. RP methods uncover estimates of the value of non-market goods by using evidence of how people behave in the face of real choices. The basic premise is that non-market goods affect the price of market goods in other well-functioning markets and price differentials in these markets can provide estimates of WTP and WTA.

Hedonic pricing. The most commonly employed method, hedonic pricing, involves examining people's purchasing decisions in markets related to the non-market good. It has commonly been applied using data from housing and labour markets. In the former, the intuition is that the price differential between otherwise identical houses that differ in their exposure levels from non-market goods and bads, such as good schools, pollution and crime, reveals information regarding individuals' WTP/WTA for such goods. Labour market applications follow a similar logic, though the focus is typically on the compensating wage differentials that are paid in relation to job characteristics, such as health and safety risks or job security.

RP methods may also use behaviour observed through the actions people take to insulate themselves from things that lower their welfare, or the amount of money people lose or spend to remedy negative outcomes. Respectively, these are known as the defensive expenditure and the damage cost methods.

Defensive expenditure. The defensive expenditure approach assumes that a rational individual will take defensive measures as long as the damage avoided exceeds the costs of the defensive action (Dickie 2003). Therefore, the defensive costs usually depict the least amount of money a person would be willing to pay to avoid the bad outcome. For example, expenditures made by water companies to remove pesticides and nitrates from drinking water comprise a lower bound indicator of the real cost of water pollution since it shows the amount that society is at least prepared to pay to purify water. The same type of argument is applicable, for example, to defensive costs incurred to protect biodiversity.

Damage cost method (DCM). The damage cost method is related to defensive expenditure methods, except that DCM is not designed to estimate theoretically consistent measures of economic value (i.e. compensating measures such as WTP and WTA), whereas defensive expenditure methods are (Dickie 2003). In sum, the DCM "attempts to measure the resource cost associated with environmental changes, rather than WTP" (Dickie 2003, p 430). The fundamental challenge is that the DCM does not provide a measure of value associated with welfare change (Dickie 2003). This is illustrated in Figure 6 which shows the generic relationship between value, price and costs for a well-functioning market.

Figure 6: Value, price and cost relationship

Price is an entity that lies somewhere between the cost of producing the good and the value that consumers place on the good, where value is defined as a compensating measure, such as CS and, hence, relates back to individual welfare. DCM can be used to measure the costs associated with health conditions and loss of environmental resources. Values in the DCM are usually based on the total cost of lost environmental resources and of health – an area where DCM is regularly employed – with values usually representing costs associated with treatment. Generally speaking, one would expect costs to lie beneath value. The evidence suggests that WTP to avoid health conditions generally exceeds damage costs for the same health condition by a factor of 2 to 21 (Agee and Crocker 1996, Krupnick and Cropper 1992, Berger *et al.* 1986, Chestnut 1985). In this respect, DCM values will provide lower bounds for compensating measures, such as WTP and WTA.

Another problem directed at DCM is that while the normative basis of the preference-based valuation is individuals' welfare (embodied in the extent to which their preferences are satisfied), the decision to incur health expenditures is not made by the individual alone, but by policy-makers, governments and taxpayers. "This can introduce uncertainties about what the (DCM) approach is actually measuring. When the focus is expenditure made by the individual, one can be (reasonably) confident that these expenditure decisions reflect the preferences of the individual for reduced negative impacts. However, expenditure decisions made by social administrators, politicians and so on might reflect other considerations, including politics and ethics" (OECD 2006). This nevertheless could be a potentially valid estimate of the value of some situation for society as a whole, but only if those decision-making institutions are representative of the individuals' preferences in a society.

Despite these issues, there are some good reasons why damage costs are used for valuation. The first is related to the issue of costs as lower bounds of value. For the purposes of CBA, this is useful information because if the project passes the cost-benefit test even when benefits are measured through the DCM, one can be confident that the project has net positive effects on society since in reality, benefits are understated using the DCM. Second, in practical terms, costs are much easier to estimate than CS and ES because they are simply measured by the market prices of inputs, or of foregone goods and services.

Although the DCM is a recognised method in valuation, it is important that the caveats presented in the previous paragraphs be kept in mind when employing a damage cost approach. In general, one can expect DCM value estimates to understate values.

1.2.6.1.2 Stated preferences

Very often, proxy markets do not exist for the non-market good in question and, instead, there is a need to ask people directly about their WTP or WTA. Stated preference methods (SPs) use surveys to ask people about the value they place on a good, or on some attributes of a good.

Contingent valuation methods. Contingent valuation methods construct and present hypothetical markets to survey respondents. The survey includes a detailed description of the good and how it will be provided, and information on the method and frequency of payment, which is usually manifest in the form of an increase in taxes. Following this, respondents are asked to state their maximum WTP for the good or their minimum WTA for the bad.

Choice modelling methods. Non-market goods can be described by their attributes. Choice modelling methods present respondents with a series of alternative descriptions of a good. The alternative descriptions are constructed by varying the levels of the good's attributes. For these methods, as long as cost or price is included as an attribute, statistical techniques can be used to recover WTP estimates for the attributes of the good.

These stated preference methods face a range of problems, detailed in the following paragraphs, that have been increasingly highlighted in the economics literature.

Context dependency. At a fundamental level, preferences have been found to be highly context-dependent in many situations. A large and growing literature in the decision sciences (see Slovic and Lichtenstein 2006) has shown that preferences can often be biased by irrelevant factors, which means that what people want may not always align well with what is best for them.

Prediction inaccuracy. Numerous experiments have shown that people are unable to accurately predict the pleasure or benefits they will get from different goods and services – this is true even for everyday goods such as yogurt, music and ice cream (Kahneman and Snell 1992, Wilson and Gilbert 2003). One of the drivers of this phenomenon is that people are unable to predict how much they will adapt to different things and circumstances. Asking people about how something will affect their lives, or about their preferences for different states of the world often leads to a focussing illusion (Schkade and Kahneman 1998, Kahneman, Krueger *et al.* 2006). This means that, at the time of preference elicitation, people tend to focus only on the salient aspects of the condition which may not reflect how they would actually experience these conditions or states in real life. The fundamental problem is that what the focus is on in a preference question is often not what the focus of attention is on in the actual experiences of lives, where many other phenomena alter attention and people may adapt to certain things (Dolan and Kahneman 2008).

Irrelevant values. People tend to systematically anchor their values for non-market goods on irrelevant numbers or cues that appear in the environment at the time (Ariely *et al.* 2003). This applies to both real market scenarios and to SPs. For example, real estate agents are influenced by random house listing price anchors when valuing a property (Northcraft and Neale 1987).

Preference reversals. People may reverse preferences when the same information about the good is presented in slightly different ways. Preference reversals violate the rationality assumptions set on preferences for valuation, which makes it difficult to judge which state of the world ultimately makes the individual better off. Famous examples of preference reversals include Lichtenstein and Slovic's (1971) experiments on preferences over different gambles, where people show an inconsistency between choice and price or value over probability bets (those with the highest probabilities) and money bets (those with highest payout), and Hsee's (2000) studies on separate evaluation vs joint evaluation, where people use different aspects of the same information set when jointly evaluating a good (say an organic and conventional version of the same food item) and can end-up stating or placing different values on the same good dependent on whether it was evaluated on its own or in comparison against another good. These types of preference reversals have been observed in SP survey responses as well (Irwin, Slovic *et al.* 1993).

Survey-related biases. A set of survey-related biases is inherent in SP methods. Problems labelled as embedding effects include: i) sequencing effects, whereby the stated WTP for a good depends on the order in which it is presented against other goods, and ii) insensitivity to scope, which is when WTP for a non-market good is insensitive to the size of that good. For instance, Desvousges *et al.* (1992) found no significant difference in the mean levels of WTP to save 2 000, 20 000 or 200 000 migrating birds from death.

Incentive incompatibility. These include: i) hypothetical bias, which is when stated WTP is higher than actual WTP, as revealed in real market decisions; ii) strategic bias which means people may strategize to affect policy by, for example, stating an extremely high value in order to encourage policy-makers to provide the good); iii) protest values, which is when people highly value the good but they state a zero WTP out of protest because they don't believe the government should be intervening in the particular issue, or are put off by the thought of being asked to place a monetary value on the good.

Personal aversions. There have been concerns expressed about the acceptability of asking people for their willingness to pay for goods and services, such as health, that they may have an aversion to expressing in monetary terms.

1.2.6.2 Well-being valuation

Subjective well-being (SWB). Measures of subjective well-being data, such as life satisfaction, happiness and purpose in life, offer another complementary platform for estimating economic values. Rather than relying on real or hypothetical market data, SWB data are used to assess the impacts of different life events and externalities on people's self-reported well-being using large national datasets and econometric methods such as regression analysis, matching and difference-in-difference estimators. Essentially, the method calls for estimating the impact of policies, non-market goods and economic events directly on measures of human welfare, rather than trying to assess price responses or stated preferences. Economists have used SWB data to assess the impacts of labour market interventions, climate change, pollution, inflation, unemployment rates, health, war, natural disasters and many other policy-relevant areas (Fujiwara and Campbell 2011).

Well-being valuation (WV). To attach monetary values to non-market goods using SWB data, the marginal rates of substitution between the non-market good and income is assessed, which is a measure that allows deriving estimates of CS and ES. The well-being valuation approach estimates the impact of the good or service and income on people's SWB and uses these estimates to calculate the exact amount of money that would produce the equivalent impact on SWB. Usually, life satisfaction is used as the measure of SWB, but other measures such as happiness can also be used. This approach is based on the critical assumption of full substitutability of income with the impacts of the respective policies, non-market goods and events of interest.

For example, assessing the cost of conflict due to resource scarcity uses a two-stage statistical analysis.

- *Stage one.* Data on life satisfaction is used to estimate the (negative) impact that the conflict due to resource scarcity has on the well-being of individuals. It has been found, for example, that conflicts lead to a 5 percent decrease in people's life satisfaction.
- *Stage two.* The exact amount of money that would compensate the 5 percent reduction in life satisfaction is calculated using the same type of statistical methods. For example, the analysis may find that USD 12 000 per year in extra income would also induce a 5 percent change in life satisfaction, enabling a conclusion that the cost of conflicts due to resource scarcity is, on average, USD 12 000 per person per year for the sample considered. This is an exact measure of monetary value that aligns with welfare economic theory and resembles a WTA or compensation value. Large national datasets that contain data on SWB, such as the World Values Survey (WVS) and the British Household Panel Survey (BHPS), can be used for such estimates.

The WV approach uses data on people's actual experiences by looking at how experiencing certain outcomes impacts SWB. Doing so gets around many of the problems encountered with traditional preference-based methods. In well-being valuation, there is no need to ask people how much they value something, which means there are no issues related to whether they have good information about the outcomes, there are no survey-related biases and it is impossible for people to influence the valuation results in any way. Most importantly, it is possible to estimate the value of different goods and outcomes as people experience their lives, rather than from data about their hypothetical preferences, which are tainted by people's focussing illusions. In sum, one can value outcomes such as improved health and cost of conflicts in terms of how people experience these things in real-life.

On the flip-side, however, there are a number of problems related to WV that should be considered when undertaking this analysis. First, a single metric outcome such as life satisfaction may not pick up everything related to quality of life. There is evidence to suggest that life satisfaction is a reasonable measure, e.g. it correlates with health and suicide rates in the expected direction and with areas of the brain associated with pleasure and well-being under neural imaging (Fujiwara and Campbell 2011). However, it has been shown that life satisfaction can be disproportionately influenced by minor events that should have little impact on one's overall quality of life, such as the weather right now, the actions and behaviour of person interviewing you, or the order of the questions in the survey (Schwarz and Strack 1999).

Second, whereas preferences are formed based on predictions about future feelings and opinions, SWB ratings are based on retrospective assessments of one's life. It goes without saying that memory is not a perfect instrument, but evidence suggests that there may be systematic biases involved too. For example, Kahneman *et al.* (1993) found that people's memories of their experiences were based solely on the peak and end emotions of the activity, and the duration of the activity was neglected. Thus, when thinking about the past in forming their well-being scores, people may not remember how the events were actually experienced at the time.

Third, the econometric methodology should be robust and is reliant on estimating unbiased causal effects for the outcome of interest (e.g. conflict) and money. This is problematic in large observational datasets where treatment has not been assigned randomly. We are reliant on statistical techniques (e.g. multivariate regression and matching estimators) that control for observable differences across intervention and control groups, but there is always the risk that some important unobserved factors are missing from the model, which would bias our estimates. This is especially problematic for the income variable that is found to be significantly under-biased in regression, because of measurement errors and because income is endogenously determined.

Fourth, it is not possible to pick up non-use values in WV since the outcome of interest needs to be "experienced" directly by the survey respondent.

WV is an evolving methodology but it features in the UK HM Treasury Green Book Guidance on policy appraisal, and has been used by a number of UK government departments (e.g. Department for Business, Innovation and Skills, Department for Work and Pensions, Department for Culture, Media and Sport, and the Cabinet Office), and is used by the OECD. It has also been used a number of times for assessing the costs of environmental factors and pollution and has featured in a number of high-profile journals (e.g. Levinson 2009, Luechinger and Raschky 2009). The WV approach and technical details for the related valuations undertaken in this work are described in more detail in Chapter 3.

In sum, there are relative advantages and disadvantages associated with preference and well-being-based valuation methods, and the discussion and caveats presented in this and the preceding sections should be taken into consideration when interpreting and using the results. In the case of estimating global costs and damages over a number of countries, it could be argued that the well-being valuation offers the most feasible method at this scale because stated preference methods would be very costly and time consuming, and revealed preference methodology could not be used to assess many types of social costs due to lack of proxy markets. In addition, health and conflict are extremely difficult outcomes or concepts for people to place a value on when asked, such as problems related to health valuation (Fujiwara and Dolan 2014), which may mean that for these outcomes, well-being valuation may well represent the only feasible valuation approach.

1.3 Modelling full costs of food wastage

The environmental impacts of food wastage have been monetized according to cost and value estimates applied to the linear approximation of the equilibrium approach described in section 1.2.1 – 1.2.3. That is, all costs are estimated via the wastage quantities and unit costs of the related environmental (and some social) impacts. This also applies to the categories that are assessed on the basis of per-area cost data, as the area numbers related to food wastage are again in the end linked to the food wastage quantities. The FCA of food wastage thus represents a "production function approach" to economic valuation, whereby a set of functions link food wastage to environmental and socio-economic impacts and those impacts are valued separately. This indirect approach – which differs from a direct approach that would value or cost food wastage directly – is a standard and accepted approach to measuring economic values, e.g. for environmental goods (OECD 2006).

1.3.1 General approach

As described in section 1.2, an encompassing approach, adopted for the valuation of the costs of food wastage, ultimately seeks to measure costs in terms of impacts on human welfare. It acknowledges that the environment can engender numerous types of value to society. In addition, since FCA addresses all of these value types, there will be no restriction of the estimated values. The broadest number of outcome values (given the data available) is thus estimated to the fullest extent possible.

Most of the relevant environmental impacts relate to the agricultural production phase, with only greenhouse gas emissions occurring along the entire food chain[4] (FAO 2013c). The various external cost categories for agricultural production (e.g. Pretty, Brett *et al.* 2000b, Pretty, Brett *et al.* 2001, Tegtmeier and Duffy 2004) offer assessments of the external costs of agriculture in the UK or the USA. A similar structure has been adopted for this FCA framework for the categories: a) air/atmosphere, b) water, c) soil, d) biodiversity and landscape/ecosystems, e) human health, f) economic value, and g) individual well-being, expressed for livelihood, health and conflict. In assessing food wastage, the external costs of the impacts from production are not the only issue. It is also important to consider costs associated with resources wasted, such as resource use per se and the lost value of wasted production in addition to externalities. Those cost estimates mainly use damage costs and defensive expenditure valuation approaches, because those are the only areas where data are available. Details are given in Chapter 2.

The well-being valuation approach was applied to measure the costs of loss of livelihoods, health conditions and national conflicts due to environmental outcomes of food wastage. The impact of soil erosion was used to derive livelihood loss and conflict, and pesticide use was used for deriving health damages. Details are given in Chapter 3.

4 It has been noted that the water used during processing is minimal, as compared to the production phase.

With an assessment such as this one, double counting is a particular challenge. For example, using social costs of carbon numbers to assess the costs of GHG emissions, may already be covered as partial costs of nitrogen impacts on ecosystems (via N_2O). Double counting is also an issue for the production value estimates which are based on food prices (e.g. farm gate). This means they cover all internalized costs of input use, including costs for irrigation water, labour and land rental. Although they can be estimated separately for illustration of their relative importance in the total cost estimate, but they must not be added to the total.

Another important point is the fact that the cost categories a) to f) relate to costs that are determined with a clear focus on a societal perspective, i.e. they are related to the costs as determined by society as a whole, while the well-being costs in category g) are determined with a clear focus on the single individual's valuation.

1.3.2 Benefit transfer

The cost estimates presented in this document are based on values from literature reviews, which most often exist only for individual countries. Generalization to the global level is an undertaking fraught with difficulties. For this methodology, the generalization is achieved by translating the results from specific countries to other regions or globally by means of the benefit transfer method (Ready, Navrud *et al.* 2004). This is often the only viable approach in situations where estimates of externalities are not available for all countries and further primary data collection is too expensive (Pearce, Atkinson *et al.* 2006).

International benefit transfer allows accounting for some of the relevant differences between countries. Thus, in addition to determining differences in income and purchasing power, it is also important to take exchange rates and inflation into account. Cultural values and traditions that may influence valuation of environmental goods and the costs of adverse effects on those also can be assessed with benefit transfer but, in most cases, not enough data are available to reliably implement it (Ready, Navrud *et al.* 2004) and we do not employ any corrections to account for this.

There is a considerable body of literature criticizing benefit transfer. (Ready, Navrud *et al.* 2004) stated that it should only be applied if valuation errors of +/-50 percent would not alter decisions. (Kaul, Boyle *et al.* 2013) identified a similar error range of about 40 percent in a recent meta-analysis of benefit transfer studies from which they derived several recommendations, the most important of which were: "… (3) transfers describing environmental quantity generate lower transfer errors than transfers describing quality changes; (4) geographic site similarity is important for value transfers; […] and (6) combining data from multiple studies tends to reduce transfer errors." In this case, examples of quality indicators were human health, erosion, farming practices, air or water pollution, and examples of quantity indicators were fish catch rates, water supply or access to recreation sites. Thus, the estimates, presented in Chapters 2 - 4, mainly refer to quality indicators. However, values are used from different studies for a range of cost categories, which also allows for benefit transfer, but only within regions of more similar countries. An error bar of 50 percent is less a problem, given the huge uncertainties involved in the cost estimates used, also before applying benefit transfer.

In the cost estimates undertaken in the following Chapters 2 - 4 benefit transfer is always applied by using values for one or several countries, translating them into US dollars for the year 2012, duly corrected for inflation and exchange rates. Those values are then used in the other countries after application of benefit transfer, based on purchase power parity corrected per capita GDP values.

The well-being valuation estimates for costs of livelihood loss, conflict and health damages are based on research using the World Values Survey that relates well-being levels in a wide range of countries to pollution and conflict. The values are derived from data from a sample of 55 countries across the world and thus, cost estimates represent global-level values without recourse to benefit transfer techniques.

2. Monetization of Environmental Costs

Full-cost accounting of food wastage was performed by using the SOL-model developed for another FAO project (i.e. Sustainability and Organic Livestock), as it is physical mass balance model that can be applied to the entire food system. The model is programmed in general algebraic modelling language (GAMS) and designed as an optimization model. SOL-m uses FAOSTAT data, covering 215 primary activities, including 180 crops as grown on the field and 35 activities from 22 different livestock types, with 229 single countries and territories as geographic reference units. This dataset provides the most comprehensive overview of the current global food system available.

For the analysis of the "current situation", SOL-m used arithmetic mean values for the years 2005–2009, in order to smooth the yearly fluctuations in production, yields, trade and prices of agricultural products. It also used the most recent data available that is compatible with other data sets.

Regional wastage volumes for different commodity groups were taken from the detailed data used in the Summary Report of the Food Wastage Footprint – Impacts on Natural Resources (FAO 2013a). However, FCA calculations require values on single country and commodity levels. Therefore, based on the wastage shares of the commodity groups and regions (FAO 2013a), wastage shares were derived for all single commodities and for all single countries within SOL-m. Multiplying those shares with the production volumes provided by SOL-m then produced the wastage volumes from the production and post-harvest phases for each commodity in each country. Multiplying the wastage shares by the domestically available quantities provided the wastage volumes at the post-production level for each commodity and country. SOL-m was then used to determine areas and animal numbers related to the commodities wasted.

Environmental effects of the food wastage volumes during the production phase were derived via the environmental effects per tonne, hectare or animal, as provided by SOL-m and the quantities, areas and animal numbers related to food wastage. Environmental impacts of wastage volumes at the post-production phase were taken from the detailed data set used in the Summary Report of the Food Wastage Footprint (FAO 2013a), linked to the respective quantities, areas and animal numbers via SOL-m. Results of the environmental impacts at the production level were cross-checked with (FAO 2013a) for consistency. The costs of the impacts were then derived in SOL-m according to the first order approximation to the general equilibrium effects described in section 1.2, i.e. based on cost information per unit of environmental impact (e.g. tonne CO_2e or tonne N leaked) multiplied with the impact level related to the food wastage quantities. The information on cost was usually only available for one or a few countries, so benefit transfer was employed to derive cost information for the other countries.

Table 2 presents a compilation of the cost categories monetized in this FCA of food wastage and the valuation methods used, and also provides further details of the data used and the calculations performed for each of the various cost categories. Details on the cost estimates undertaken with the well-being approach, including for the categories "health", "livelihoods" and "conflicts", are provided in Chapter 3.

As shown in Table 2, the cost estimates provided here cover only a small part of the full costs of food wastage. Due to attempting such first approximation as described in section 1.2 without general equilibrium feedback effects, there are several gaps that need to be recognized:

- long-term societal costs and chronic effects of pesticide poisoning are missing;
- water use costs are based on water prices that are heavily subsidized and do not account for true infrastructure and provisioning costs;
- loss of services from grasslands, wetlands and biomes other than forests are not covered;
- well-being losses are estimated for adults only;
- no data on land values and opportunity costs from lost alternative uses were available and the corresponding costs are not estimated.

The following section details valuation approaches taken for these cost estimates.

Table 2: Cost estimates for the FCA of food wastage

Impact category	Valuation method	Unit value used (USD 2012)
Atmosphere		
GHG emissions (including deforestation and managed organic soils)	Social cost of carbon (based on a range of approaches, most importantly damage costs/defensive expenditure)	113 $/t$CO_2$e (globally, no benefit transfer needed)
Ammonia emissions	WTP to avoid	5.36 $/ha (derived from USD 103 million for total ammonia emissions costs from UK agriculture with BT to other countries with correction for N inputs and agricultural areas)
Water		
Water quality (nitrate and pesticide contamination of drinking water, N/P eutrophication)	Defensive expenditures (costs of pesticide, N, P removal from drinking water), damage costs, WTP to avoid	16.33$/ha for N eutrophication (based on 0.286$/kgN leached in UK, correction for N input and output levels and agricultural areas in each country, and BT) 64.15$/ha for P eutrophication (based on 12.32$/kg P leached, correction for P input and output levels and agricultural areas in each country and BT) 1.83$/ha for nitrate contamination (derived from USD 35.2 million, total nitrate pollution costs from agriculture in the UK, BT to other countries with correction for N inputs and agricultural area) 40.42$/ha (UK) and 0.78$/ha (Thailand) for pesticide contamination (total 264 million in UK, 14.6 million Thailand, corrected for toxicity levels, area, BT)
Water use	Damage costs (value lost)	0.1$/m^3 (UK) plus BT
Water scarcity	Damage costs/defensive expenditure	0-18.8$/m^3 (based on the scarcity function from USA and national water scarcity levels)
Soil		
Soil erosion (due to water and wind)	Damage costs (on- site and off-site)	21.54$/ton soil lost from water erosion, 27.38$/t for wind erosion (US values plus BT, plus per ha soil erosion levels from 48 countries and regional averages derived from them; corrected for soil erosion potential of different cultures)
Land occupation (only via loss of ecosystem services from deforestation)	Damage costs due to the linkage of land occupation to deforestation	Average 1 611$/ha forest lost (based on 14 country estimates and regional BT)
Biodiversity		
Biodiversity loss from pollutants (pesticides, N/P eutrophication)	Damage costs, defensive expenditure	5.46$/ha for N eutrophication (based on 0.024$/kgN applied in UK, correction for N inputs, area and BT) 4.76$/ha for P eutrophication (based on 0.26$/kgP applied in UK, correction for P inputs, area and BT) 4.21$/ha (UK) and 1.89$/ha (Thailand) for pesticide impacts on biodiversity (total 27.5 million in UK, 35.5 million Thailand, corrected for toxicity levels, area, BT)
Fisheries overexploitation	Damage costs (cost of loss of fishing effort linked to low fish populations)	Global estimates for the total fishery sector from the literature, scaled by wastage shares
Pollinator losses	Damage costs (loss in pollination services)	Global estimates from the literature, scaled by wastage shares
Social		
Loss of livelihood (for adults of age 18+ only)	Well-being valuation (based on well-being loss due to environmental degradation; proxy: soil erosion from water)	8.54*10-8 (OECD) and 1.25*10-7 (Non-OECD) $/cap/y/t soil lost from water erosion (no BT needed)
Individual health damage (for adults of age 18+ only)	Well-being valuation (based on well-being loss due to toxicity levels)	9.67*10-8 (OECD) and 9.93*10-8 (Non-OECD) $/cap/y/unit toxicity level (no BT needed)
Pesticide poisoning	Damage costs (acute treatment costs)	0.34$/ha (UK) and 22.7$/ha (Thailand) for pesticide contamination (total 2.2 million in UK, 426 million Thailand, corrected for toxicity levels, area, BT)
Conflict (for adults of age 18+ only)	Well-being valuation (based on well-being loss due to conflicts induced by environmental degradation (proxy: soil erosion from water))	3.21*10-7$/cap/y/t soil lost from water erosion (based on the 10 conflict countries in the period 2005-8, no BT needed)
Economic costs		
Wasted food	Damage costs (lost economic value)	Country and crop-wise producer prices for production level wastage, gross trade prices for post-production
Subsidies (OECD only)	Damage costs (subsidies wasted)	Total subsidies for single OECD countries (Europe as EU-27 only), divided by areas (ha)

Note: Benefit transfer (BT) is done as region-wide as possible. Where values for the UK and Thailand are given, UK numbers are used for developed country BT and Thailand numbers are used for developing country BT.

2.1 Atmosphere

2.1.1 Greenhouse gas emissions

GHG emissions from food wastage amount to about 2.7 Gt CO_2e (without emissions from deforestation and organic soils), which is less than the 3.3 Gt calculated in the Food Wastage Footprint (FAO 2013a). This difference is due to different calculation methods for the production phase – the SOL model employs a full life-cycle analysis for each commodity, while the Foot Wastage Footprint is based on literature values per unit produced. SOL-m also employs detailed herd structure models for cattle, pigs and chickens to differentiate the feed requirements and total emissions from animals at various ages and production levels. It also covers some additional commodities that were not previously covered (FAO 2013a), including sugar, coffee and alcoholic beverages. Given the huge uncertainties related to these calculations, the two estimates are largely consistent. This is at the same time a consistency check for the two different approaches, as they do not differ by more than 20 percent. Additional consistency checks of the GHG calculations within the SOL model, undertaken by comparing the SOL-m results with results from FAOSTAT which uses different methods (Tubiello, Salvatore *et al.* 2013), found SOL-m and FAOSTAT yielded largely the same results.

Food wastage-related emissions from deforestation and organic soils added 0.64 Gt CO_2e and 0.15 Gt CO_2e, respectively. These amounts were determined by relating national deforestation values, corresponding emissions from deforestation and emissions from managed organic soils from FAOSTAT (Tubiello *et al.* 2013) to agricultural areas for each crop according to its share in total area. Doing so provided deforestation effects and emissions from managed organic soils per hectare of agricultural land for each crop. Areas related to food wastage quantities were then multiplied with these per hectare deforestation and organic soil emissions to arrive at the estimates for the effects of food wastage from the production phase. For the post-production phase, global average values per hectare were used to account for the fact that the origin of the traded products that end up as food wastage is unknown.

Valuation of these emissions used the Stern Review (2007) estimates for the social cost of carbon (SCC), based on the total costs of a tonne of emitted CO_2e. The SCC is the estimated cost of the global damage caused by an additional tonne of GHG emitted today and over its lifetime in the atmosphere (100 years or longer). This approach reflects two specific characteristics of climate change. First, as a global pollutant, GHG emissions from each country contribute to damages everywhere, not just the source country. Second, GHGs emitted today continue to cause damage into the future, and the marginal cost of these damages increases at higher atmospheric concentrations of GHGs.

The SCC represents the marginal cost of CO_2. These costs are estimated using market data from existing or surrogate markets. For example, the effect of climate change on crop yield is estimated from the market price of the loss of agricultural productivity, which is a direct existing market value. The impacts on health can be measured through benefit transfer techniques using studies of the valuation of mortality risks from other (non-environment) contexts. This is a surrogate or proxy market approach (Tol 2011).

The SCC reflects society's WTP to avoid future damages caused by carbon emissions. This is reflected in the marginal costs because, as a society, we should be willing to incur costs to reduce emissions up to, and no more than, the damage we expect the emissions to cause (Price *et al.* 2007). Interestingly, a comparison of the SCC with SP studies that asked people directly about their WTP to reduce carbon emissions, found that WTP estimates from these surveys are in line with the SCC (Tol 2011).

SCC estimates vary widely, depending on the choice of certain parameters and the coverage of climate impacts and economic effects that are included. Some of the key parameters with substantial impact on the results are discussed in the following sections. The Stern Review (2007) assumed a range and distribution for most of those parameters and then derived a distribution of cost estimates based on Monte Carlo Simulations, i.e. running the calculations thousands of times, each time with different randomly drawn values from those distributions for each parameter.

Parameter 1: cost coverage. The coverage of climate impacts and economic costs differs between SCC estimates. In practice, any SCC is an estimate based on a partial subset of the full costs of climate change only, as many impacts are unknown or uncertain and others cannot be quantified in monetary terms. Figure 7 presents a matrix of climate change impacts and costs.

Most SCC studies only cover direct climate change impacts (associated mainly with temperature rise) and direct market costs (light blue zone of Figure 7). Some more recent studies, such as (Waldhoff, Anthoff *et al.* 2011) included a wider range of impacts and costs that are more difficult to calculate (medium blue zone). (Stern 2007) also modelled possible systems changes and surprises (dark blue zone). In the bottom right corner of the matrix, “socially contingent” effects of climate change (grey zone) include major catastrophes such as conflict, famine and poverty. Arguably, the large-scale loss of life and impacts on societies and economies are impossible to calculate; they involve ethical and equity dimensions that cannot be valued in monetary terms (Ekins 2005).

Figure 7: Social cost of carbon risk matrix, adapted from (Watkiss 2005)

Uncertainty in predicting climate change ↓ / Uncertainty in valuation →	Market	Non-market	Socially contingent
Projection (sea level rise)	Coastal protection Loss of dryland Energy (heating/cooling)	Heat stress Loss of wetland	Regional costs Investment
Bounded risks (droughts, floods, storms)	Agriculture Water Climate variability	Ecosystem change Biodiversity Loss of life Secondary social effects	Comparative advantage and market structures
System change and surprises (major events)	Significant loss of land and resources Non-marginal effects	Higher order social effects Regional collapse Irreversible losses	Regional collapse

Illustrates the gradient of difficulty (from light blue to grey) in taking different climate effects categories into consideration.

Parameter 2: discount rate. The choice of the discount rate is crucial. Discount rates are based on the observation that people would prefer to have something valuable today rather than in the future. Because the compliance costs of climate change are incurred in the short-term and benefits of mitigation are mostly realized in the long-term, the choice of the discount rate has a significant influence in the analysis of climate impacts. It is important to emphasize that the choice of the discount rate involves a normative judgement, reflecting the present value we assign to future generations' welfare. The Stern Review (2007) uses a discount rate of about 1.4 percent which is low in comparison to the values used in other calculations for the social costs of climate change (Sterner and Persson 2008).

Parameter 3: equity weighting. The concept of equity weighting is based on the theoretical and empirical observation of diminishing marginal utility of wealth. This means that the same amount of additional money has more utility to a poorer person than a richer one. In the context of climate change modelling, equity weighting implies that damages that occur in poorer countries/regions are weighted more heavily.

Box 1: What is, and is not, included in climate change cost estimates

Watkiss *et al.* (2005) summarized the impacts and costs that are generally included/excluded at differing degrees of uncertainty for sea level rise, energy use, agriculture, water supply, health and mortality, ecosystems and biodiversity, extreme weather events, catastrophic events and major climate discontinuities.

Waldhoff *et al.* (2011) included agriculture, forestry, sea-level rise, cardiovascular and respiratory disorders related to cold and heat stress, malaria, dengue fever, schistosomiasis, energy consumption, water resources, unmanaged ecosystems, diarrhoea, and tropical and extra tropical storms. They differentiated between the three greenhouse gases, CO_2, CH_4 and N_2O, according to their lifetimes in the atmosphere and the fact that CO_2 has a positive CO_2-fertilizing effect (although the amount is highly uncertain). CO_2-fertilization is the yield-increasing effect of higher atmospheric CO_2 concentrations. Accounting for this is important for agricultural GHG emissions that largely consist of N_2O and CH_4 (Smith *et al.* 2007).

Stern (2007) used the Policy Analysis for the Greenhouse Effect (PAGE 2002) model (Hope 2003) that is based on studies that estimate market impacts in the various sectors of the economy, in particular due to sea-level rise. It also is based on agriculture and health, as well as some non-market damages to human health, amenities and the environment (Nordhaus and Boyer 2000, Mendelssohn *et al.* 2000, Tol 1999 cited in IPCC 2001, Working Group II, p 940). The values from Tol (1999) were based on an earlier version of the Climate Framework for Uncertainty, Negotiation and Distribution (FUND) model employed in Waldhoff *et al.* (2011) and covered the same impacts as Waldhoff *et al.* (2011) did. Stern (2007) also includes a simplified modelling of the risk and costs of a catastrophic climate event occurring as temperatures increase.

Parameter 4: climate sensitivity. Climate sensitivity captures the magnitude of the temperature increase associated with a doubling of atmospheric CO_2e concentrations. For example, (Waldhoff, Anthoff *et al.* 2011) determined that if climate sensitivities of 2.0°C or 4.5°C were used instead of the 3°C, the social costs fell from their central estimate of USD 8/t CO_2e by more than 50 percent to USD 3/t CO_2e and rise by more than 100 percent to USD 18/t CO_2e, respectively. The Stern Review (2007) worked with climate sensitivities between 2°C to 5°C, with their likelihood distributed as derived by (Meinshausen 2006).

Parameter 5: emissions profile. Having a total emissions profile over time is crucial for computing the social costs of carbon. This is due to the relationship between marginal damage costs and the GHG stock in the atmosphere, as marginal damage costs tend to increase with GHG concentrations. If emissions increase sharply, marginal damage costs will also rise. The Stern Review (2007) used the emission profile from the Intergovernmental Panel on Climate Change (IPCC) Third Assessment Report calculations, specifically its A2 scenario, which captures developments without particular focus on mitigation actions and correspondingly rather higher emissions (IPCC 2001).

To illustrate the approach to carbon monetization, SCC values from (Stern 2007) and (Waldhoff, Anthoff *et al.* 2011) were compared for this study. Both are recent studies from opposite ends of the spectrum of SCC estimates. The estimate of Stern (2007) was further refined by (Weitzmann 2007). Each model was subject to critical review: Ackerman and Stanton (2010) critiqued the model used in (Waldhoff, Anthoff *et al.* 2011) and, likewise, the Stern Report (2007) triggered ample criticism (e.g. Nordhaus 2007), mainly due to the low discount rate used that did not reflect expected market developments. However, refinements of the calculations were possible and even with high discount rates, accounting for relative scarcities of different resources with climate change and correspondingly changing relative prices allowed for SSC to arrive at values similar to Stern (Sterner and Persson 2008).

(Waldhoff, Anthoff *et al.* 2011) reported a central value of USD 8/t CO_2e (range: 2–240) for CO_2, USD 10 (2–160)/t CO_2e for CH_4, and USD 20 (4–330)/t CO_2e for N_2O. Interestingly, in a more recent application of the same model, central estimates arrived at a global level of about USD 180/t CO_2e, which (Anthoff and Tol 2013) reported as USD 50/t C in an article containing a detailed assessment of the importance of changes in various parameters on the results. (Stern 2007) proposed a central estimate of USD 113/t CO_2e in 2012, compared to USD 85 in 2000. The wide ranges were due to the uncertainties described in the previous paragraphs. Differences in discount rate and equity weights were particularly significant, as each can lead to estimates that differ by two orders of magnitude (e.g. when the discount rate varies from 0.1 percent to 3 percent). The combination of several of these uncertainties results in an even wider range of values.

Final valuation of the costs of food wastage due to greenhouse gas emissions was done by means of the cost estimate presented by (Stern 2007). This was due to the wide acceptance of such higher levels of SCC, which also was reflected in the fines of GHG emissions trading schemes such as those the EU set at Euro 100/t CO_2e (EU 2013). There is no formal update of the estimates given in (Stern 2007) that would have similar widespread reception globally, but the order of magnitude (USD 113 or Euro 100) may serve as a good cost level to work with. Using the USD 113 value, the final cost estimate of food wastage impact on GHG is about USD 394 billion. Although Stern (2007) did not directly provide a range for the SCC estimate, we derived a range based on numbers given from 15 percent to 5 times the central value, i.e. USD 59-1972 billion (Stern 2007, p 287).

2.1.2 Ammonia emissions

These estimates are based on the total costs of ammonia emissions for the UK – as no other data were available (Pretty, Brett *et al.* 2000b).

Ammonia emissions contribute to eutrophication and acidification. The data were based on WTP estimates for ammonia pollution reductions, combined with exposure and health impact levels. Similarly to the nitrate calculations, per hectare and kilogram, N input values were derived by dividing the total crop and grassland area by the total N inputs. Benefit transfer then provided the corresponding values for other countries, and multiplying with the N-inputs and areas provided the total estimates. As ammonia emissions are mainly from nitrogen in manure and less from other fertilizers, this approach tends to overestimate emissions and related costs.

2.2 Water

2.2.1 Pesticides in sources of drinking water

These estimates are based on the total costs of pesticide in sources of drinking water for the UK, USD 264 million (Pretty, Brett *et al.* 2000b); the USA, USD 142 million (Tegtmeier and Duffy 2004); and Thailand, USD 15 million (Praneetvatakul, Schreinemachers *et al.* 2013) – as no other data were available.

The US values are much lower than the UK values, in particular when accounting for the differences in active ingredient quantities applied (these references report 22.5 million kg applied in the UK and 447 million kg applied in USA). Most likely, this difference is largely driven by the difference in contamination limits applied in the UK and USA, which are up to a factor of 100 higher in the US (Pretty, Brett *et al.* 2000a, USEPA 2000, USEPA 2014). The UK data were based on the annual capital expenditures of water companies for pesticide removal and the share of pesticide loads stemming from agriculture, and the USA data were based on treatment facility expenditure estimates from the USA Environmental Protection Agency (EPA) with 30 percent of harmful chemicals being pesticides. The authors of the USA study point-out that unregulated pesticides are not covered. The estimates in this paper use the UK values referring to the stricter EU contamination limits.

(Praneetvatakul, Schreinemachers *et al.* 2013) is also based on (Pretty, Brett *et al.* 2000b) and benefit transfer. It specifically accounted for characteristics of Thailand, most importantly for the much higher exposure of agricultural workers to pesticides due to the higher share of the workforce active in agriculture, but also collected additional data on specific key crops not covered in (Pretty, Brett *et al.* 2000a), such as rice. For this reason, we retained the values of (Praneetvatakul, Schreinemachers *et al.* 2013) for developing countries and did not base everything directly on (Pretty, Brett *et al.* 2000a). Those national values were translated to per-hectare values by dividing by the total cropland area of the UK and Thailand, respectively, and then weighted with a national indicator of pesticide use intensity and regulation based on expert judgements[5] on the pesticide intensity of the various crops, the general pesticide use level in the country, and the stringency of national pesticide use regulations. Each of these three aspects was assigned a value of 1 to 3 (in 0.5 steps) and then multiplied to provide a general indicator for pesticide use intensity and regulation. Thereby, the stringency of regulations was coded with low values signifying high stringency. Those per hectare values were then transferred to other countries via benefit transfer. These calculations were based on UK numbers for the developed countries and on the numbers from Thailand for the developing countries. Total costs of food wastage were then derived by multiplying with the areas that corresponded to food wastage quantities and weighting with the value of the indicator for pesticide use intensity and regulation in each target country of the benefit transfer.

This approach can be criticized for many reasons, in particular because the drinking water provision and pesticide contamination is highly dependent on the local situation, which cannot be captured in benefit transfer. Nevertheless, such an estimate is the best possible and can provide some indication of the size of the related costs.

2.2.2 Nitrate in sources of drinking water

These estimates are based on the total costs of nitrate in drinking water for the UK – as no other data were available (Pretty, Brett *et al.* 2000b).

The data were based on the annual capital expenditures of water companies for nitrate removal and the share of nitrate loads stemming from agriculture. Per hectare and kilogram N input values were derived by dividing the total crop and grassland area by the total N inputs. Benefit transfer then provided the corresponding values for other countries and multiplying with the N-inputs and areas provided the total estimates. Clearly, the criticism provided for the pesticide costs in section 2.2.1 applies here as well.

2.2.3 Water use

These irrigation water use volumes ("blue water") were estimated with data based on AQUASTAT irrigation volumes per hectare irrigated area (differentiated by countries and crops) (AQUASTAT 2013), in combination with the shares of irrigated areas in total arable areas in each country provided by (FAOSTAT 2013).

This allowed estimation of average irrigation volumes per tonne for each crop and country. Thereby, no differentiation for irrigation intensities of different crops was undertaken and irrigated areas were allocated to the different crops according to their area shares in total arable areas. These irrigation volumes per tonne of produce were then combined with the wastage volumes to arrive at total irrigation water volumes lost due to food wastage. Country-wise (i.e. national) irrigation intensities were used for food wastage from the production phase, while global averages were used for food wastage from post-production value chain levels. The total irrigation water volume lost due to wastage amounted to about 300 km^3. Estimates were also made of irrigation water directly used as drinking water uptake of animals. A gross estimation of this amounted to 5 km^3. This calculation is based on average values for water uptake for different animal types (OMAFRA 2007), e.g. 50 l/day/head for cattle, reducing the reported numbers from Ontario according to lower yields in most countries) and the number of animals related to food wastage volumes. It does not account for differences in water requirements due to climatic conditions, animal age or production levels. So, with drinking water uptake in the range of about 2 percent of the irrigation volumes lost, the two types add up to about 305 km^3.

5 Jan Breithaupt (FAO), colleagues at the Research Institute of Organic Agriculture FiBL, Switzerland, staff of the Federal Department for the Environment, Switzerland, and several FAO country experts provided first estimates and subsequent cross-checks and consolidation for this indicator.

The Food Wastage Footprint project initially estimated that the irrigation water quantity lost due to food wastage amounted to 250 km^3 per year. The different value derived here (i.e. 300 km^3 for irrigation without animal drinking water) is due to inclusion of water consumption from additional commodities not previously covered, including sugar, coffee and alcoholic beverages, and to the use of a different data base (AQUA-STAT instead of Water Footprint Network) for water use (Hoekstra, Chapagain *et al.* 2011). Given that both these data sets exhibit major uncertainties, those values can be judged to be largely consistent.

Using the 305 km^3 water volume for irrigation and animals, and taking a range of USD 0.013 to 0.63 m^3 (in 2012 values, expanded to global scale via benefit transfer as described in section 1.3.2), from available per m^3 irrigation cost estimates resulted in a range of costs from USD 1 billion to USD 50 billion. The irrigation cost estimates were taken from (FAO 2004, Garrido, Martinez-Santos *et al.* 2005, Qureshi, Connor *et al.* 2007, Ghazouani, Molle *et al.* 2012) and (Solbes 2003). Four outliers that were much higher than the rest, as well as two that were much lower were dropped (the full range is USD 0.003 to 7.7/m^3). The central value in this range is a water price of USD 0.1/m^3 for the UK (year 2012), after benefit transfer resulting in total global costs of USD 7.7 billion (2012). This central value corresponds to expert judgments of irrigation costs (Jippe Hoogeveen, personal communication) which also suggest a narrower range of USD 0.05-0.2/m^3, resulting in total costs of USD 4–17 billion.

For the cost estimates needed, several challenges have to be emphasized. There is major spatial variation both between and within countries, and seasonal variation in water prices. Groundwater schemes (in Spain at least and probably as a general rule) are largely driven by market forces. Private entrepreneurs take the risk to invest in infrastructure and costs are borne by farmers. In contrast, traditional surface water schemes are heavily subsidized by governments. This spatial and temporal variance and the institutional differences cannot be captured by the average prices used. In addition, the cost estimates provided are lower end estimates, as they are based on a part of the full irrigation costs only. Cost recovery is usually restricted to operating and maintenance costs and rarely includes a small portion of initial capital costs. Furthermore, collection efficiency is not accounted for (for some studies, values on this are available). Also, formal charges do not capture the full water payments made by farmers through extra-legal payments, contribution of labour and additional on-farm costs. On the other hand, not all officially estimated costs represent real costs, due to, for example, possible overstaffing, poor management and corruption. Water fees are generally insufficient to cover operation and maintenance expenses in developing countries. Many countries also face difficulties in collection efficiency. OECD countries are more likely to cover 100 percent of operating and maintenance costs. We emphasize that for our assessment, we assumed the same costs for drinking water for animals as for irrigation.

This estimate of water-use costs is illustrative for this specific cost category, but due to the potential for double counting, it must not be summed to a total cost estimate. In other words, these water-use costs are estimated via data on per m^3 irrigation costs, which are already covered in producer prices and unit values and are thus part of the economic costs reported separately.

In addition to reported irrigation costs, non-market-based cost estimates that are not included in market prices could be used, such as those based on opportunity cost estimates (Dachraoui and Harchaoui 2004, Garrido, Martinez-Santos *et al.* 2005, Samarawickrema and Kulshreshtha 2008, Martinez-Paz and Perni 2011). While this approach is not further pursued here, an assessment of the results from those studies shows that they are also in the cost range reported here. A range of cost estimates, some of which are related to agriculture, can also be found in (WBCSD 2012).

Finally, it has to be highlighted that there are two different views regarding valuation of water wastage. While this study estimates water use from food wastage (305 km^3/year) based on consumptive water use (i.e. incremental evapotranspiration due to irrigation), other studies consider the whole volume of water withdrawn. Estimates based on the volume of water withdrawn/allocated are much higher than the ones based on consumptive use, such as the Comprehensive Assessment of Water Management in Agriculture (CA) (IWMI 2007). An argument supporting estimates based on water withdrawn is that dams and reservoirs are designed in function of water withdrawn, so investments in infrastructure and management refer to withdrawal. Using water withdrawals to calculate water wastage rather than using consumptive use always implies an over-estimation of the amount of water wasted. At the same time, using consumptive use to calculate water wastage implies an under estimation of the water wasted, because some of the return flow (water withdrawn for irrigation but not evaporated) may not be recoverable.

2.2.4 Water scarcity

The water cost values presented in (PUMA 2012) are based on the costs of water scarcity, depending on where the water use occurs. While the global average value of the costs due to water scarcity is USD 1.15/m^3 (based on 2012 costs), the variation of water scarcity estimates between countries is huge, ranging from USD 0.02/m^3 to USD 18.8/m^3. This is based on a water scarcity function developed by Trucost, based on USA data, extended to global coverage via benefit transfer and calibration with studies from some additional countries, and applied to the country-wise water scarcity level as reported in AQUASTAT (based on the annual water withdrawal in relation to the total renewable annual water supply) (PUMA 2012). Total country-wise water scarcity costs use are then estimated through multiplying the water scarcity per-m^3 estimates by the water quantities consumed. Some outliers in water scarcity values such as Saudi Arabia were removed, and values were correspondently adapted downwards to avoid disrupted results. Introducing an upper maximal scarcity level of 80 percent which corresponds to the upper level for which the scarcity function as derived from the US makes sense. The formula for the scarcity function is:
scarcity costs = (scarcity level)2*8.5/0.64

Generally, the literature provides an extremely wide range of values for additional, non-consumptive values of water such as recreational use, and of non-use values such as existence value. These uncertainties reflect the dependence on a range of strong assumptions that have to be made for determining such values, in relation to the local and regional character of such estimates (e.g. depending on the people surveyed for a contingent valuation study). These values are particularly difficult to generalize, and even with help of benefit transfer and the interpretation of results, it has to be done very cautiously.

2.3 Soil

2.3.1 Soil erosion

The cost of soil lost from erosion linked to production of food which ends up being wasted has been estimated on the basis of a wide range of country-specific values for per hectare soil erosion levels all over the world (see Annex). For the countries missing from the soil erosion dataset displayed in the Annex, we used the average soil erosion rates from the countries with data in the same subregion. This covered water erosion only, as data for wind erosion is too scarce to be used in such an assessment. However, the few existing values for wind erosion (USDA 2007, Sidochuk *et al.* 2006, Darmendrail *et al.* 2004, Lal *et al.* 1989) point to additional effects of the same order of magnitude as from water erosion. We thus inserted a similar value of USD 35 billion for this.

Several studies provide values for the total costs of soil erosion for single countries (e.g. FAO 1994, Pimentel, Harvey *et al.* 1995b, Pretty, Brett *et al.* 2000b, Stocking 2001, Berry, Olson *et al.* 2003, Hein 2007). They are based on the different on- and off-site damages incurred due to soil erosion. Values from (Pimentel, Harvey *et al.* 1995a) from the USA, one of the most detailed, are used here. Table 3 presents what is covered as on-site and off-site damages in this study, and the Annex details erosion rates for the 48 countries considered, with relevant references.

Table 3: On- and off-site damage categories from water and wind erosion

On-site damages	Off-site damages
Nutrient loss	Sedimentation
Lost yield	Flooding
Drop in land values	Water treatment
Biological losses	Electricity Power Generation
	Repairing public & private property (roads, cars, etc.)
	Global warming
	Health
	Cost to business
	Cost to irrigation and conservation districts
	Biological impacts
	Navigation
Source: adapted from (Telles, Dechen *et al.* 2013), and (Pimentel, Harvey *et al.* 1995a)	

The resulting central value for the cost estimate is about USD 34.6 billion. We also have included differentiated numbers for water erosion from grasslands for a range of countries and accounted for differing erosion potential of different crops based on expert views and the literature (Stone and Hilborn 2012). Erosion costs were assigned to wastage quantities via the related areas cropped in vain due to food wastage.

We derived a range for this central value from the information on soil erosion rates that is often provided as a minimum and maximum value, without indication of a central value. The ranges for individual countries vary widely, from factors of 2 to more than 50 between lower and upper estimates. Most ranges do not vary by more than a factor of ten and for most of the few cases where lower, upper and central values are given, the latter lies at about a third to half of the upper estimate. We thus decided to use a range from one-fifth to double the central estimate (USD 7–70 billion) and applied the same rule for wind erosion.

2.3.2 Land occupation

Land occupation includes the costs of converting forest or wetlands to cropland or managed grassland and of converting wild grassland to cropland. This can be assessed via the difference between the value of total ecosystem services from forests, grasslands and wetlands that are converted, and the services from agricultural land established on these converted sites. For this estimate, the values reported in the Economics of Ecosystems and Biodiversity (TEEB) ecosystem valuation data base (Van der Ploeg and de Groot 2010) for forest and cropland were combined with the deforestation rates derived from FAOSTAT (Tubiello, Salvatore *et al.* 2013). It was decided to report only the lost ecosystem values from deforestation, as the TEEB database offered estimates that encompassed a wide and complete or nearly complete set of forest ecosystem services for a large number of countries, thus allowing for regional benefit transfer (see Table 4). We attempted to compare this to respective estimates for cropland values, but estimates for those were too scarce and incomplete to derive useful global estimates. The forest values were reported on a per hectare basis and thus combined with deforestation rates per country. If several values per country were available, average values were used. Deforestation rates were derived from FAOSTAT deforestation numbers in relation to change in agricultural land. This allowed assigning an area deforested per hectare to an increase in average agricultural land area and, in case of an area decrease, it allowed assigning the avoided deforestation due to a decrease in this area. If no changes in agricultural areas were reported, we related deforestation rates to the total agricultural area, thus deriving a value for area deforested per hectare of agricultural land occupation. Such valuation of avoided deforestation due to reduced food wastage levels actually over-estimated the value gains due to the absence of estimates for the ecosystem services from croplands, which are not zero. Due to lack of data, it was not possible to valuate other land-use changes such as from grassland to cropland.

Table 4: Countries for which total or almost total forest ecosystem services valuations are provided in the TEEB database and ecosystem services (not all services are covered in all countries)

Countries	Ecosystem services covered (in at least one of the countries)
Australia Brazil Cameroon Canada China India Kazakhstan Laos Malaysia Mexico Samoa Spain Tanzania United States of America World (global estimate)	Air quality regulation [unspecified] Attractive landscapes Biochemicals Biodiversity protection Biological control [unspecified] Bioprospecting Capturing fine dust Climate regulation [unspecified] C-sequestration Cultural use Cultural values [unspecified] Deposition of nutrients Drinking water Education Erosion prevention Fire Prevention Flood prevention Fodder Food [unspecified] Fuel wood and charcoal Gas regulation Genetic resources [unspecified] Hydro-electricity Maintenance of soil structure Microclimate regulation Nutrient cycling Pest control Pollination Prevention of extreme events [unspecified] Raw materials [unspecified] Recreation Refugia for migratory and resident species Science / Research Seed dispersal Soil detoxification Soil formation Timber Total economic value (TEV) Tourism Water purification Water regulation [unspecified]

2.4 Biodiversity

2.4.1 Biodiversity impacts of pesticide use

Cost estimates of biodiversity on-site and off-site impacts category, based on national aggregate values, reported USD 35.5 million for Thailand (Praneetvatakul, Schreinemachers *et al.* 2013), USD 27.5 million for the UK (Glendining, Dailey *et al.* 2009) and USD 1 458 million for the USA (Tegtmeier and Duffy 2004). Those values were assigned per ha of cropland under different crops, as for the pesticide contamination of drinking water described in section 2.2.1. Per hectare values were then expanded to both developed and developing countries, with benefit transfer and accounting for the different national biodiversity levels via the National Biodiversity Index (NBI) (CBD 2014) – dividing by the NBI of the data source country and multiplying by the NBI of the benefit transfer target country. Total costs were then derived by multiplying this with the areas underlying the food wastage quantities. Thereby, we used UK or USA values for developed countries, and both versions resulted in similar estimates. Impacts on bees are kept separately, so there is no double counting of this cost category with pollinator losses, which are addressed separately in section 2.4.4).

2.4.2 Biodiversity impacts of nitrate and phosphorous eutrophication

(Glendining, Dailey *et al.* 2009) provided values for costs of biodiversity impacts per kg N and P input in UK agriculture and per kg P and N leached from use (Table 5). The values were transferred to other countries with benefit transfer and accounting for national biodiversity level as described in section 1.3.2. Total cost estimates were then arrived at by multiplying total N and P inputs or the N and P balances (i.e. inputs minus runoff) with the share of these values that corresponds to areas underlying food wastage quantities. Some double counting with ammonia emissions' costs may arise for N, but given the low absolute value of those, this is not further pursued here.

Table 5: Costs of biodiversity impacts from N and P use in agriculture

Category	Costs per kg N or P (in USD at 2012 rate)
N input	0.0242
P input	0.264
N leakage	0.286
P leakage	12.32

The comparably large value for P leakage (and also P inputs) is due to the similar total costs from N and P leakage as explained in the supplementary material of (Glendining, Dailey *et al.* 2009) and the relative molecular weights of the reference substances N and PO_4 and P.

2.4.3 Fisheries overexploitation

(World Bank 2009) evaluated the loss of economic benefits due to over fishing at around USD 50 billion a year due to overexploitation of fisheries and their resulting under performance. (FAO 2013a) estimated the global fish wastage to be about 20 percent, which put the gross estimate for the loss of economic benefits due to fish wastage contributing to fisheries overexploitation at USD 10 billion/year. This estimate did not include losses due to recreational fisheries, marine tourism or illegal fishing, nor did it consider the economic contribution of dependent activities such as fish processing, distribution and consumption, or the value of biodiversity losses and any compromise to the ocean carbon cycle. This suggests that the annual losses to the global economy from unsustainable exploitation of living marine resources actually would exceed USD 50 billion quite substantially. At the same time, this number also could be overestimating the true losses, as it did not consider the market effects of extra landings (if the fishing potential were fully realized) and the value of this additional catch was calculated by the price realised for the actual quantities caught. However, an increase of fish quantities might lead to price decreases.

2.4.4 Pollinator losses

Data on the costs of bee colony and other pollinator losses due to agriculture is scarce. (Pretty, Brett *et al.* 2000a) assigned a gross estimate of USD 2.2 million based on the value of pollinator services and loss of bee colonies over past decades in the UK, and roughly assigned half of those losses to agriculture – half of which was due to pesticide use and half due to habitat losses. This allowed assigning estimates of food wastage costs due to pesticides analogously to the estimate of pesticide use in other areas, where national numbers were reported (e.g. on drinking water). (Praneetvatakul, Schreinemachers *et al.* 2013) reported the corresponding values for Thailand and arrived at roughly one-fourth the value as for pesticides in drinking water (again assuming half of the effect on bee colonies was due to pesticides). Employing the same calculations enabled estimating the global cost of bee colony losses due to pesticide use at about USD 1 billion. The TEEB database (Van der Ploeg and de Groot 2010) contains only 9 values for pollination services from specific ecosystems in 7 countries, without a clear option on how values may be related to areas of cropland or production output in such a way as to reliably derive global values. It was thus decided not to attempt a global estimate based on this.

In a second approach to valuing pollinator losses, (Bauer and Wing 2010) determined total global pollinator losses would lead to economic costs of about USD 330 billion. This study is closely related to (Gallai, Salles *et al.* 2009), who reported Euro 190–310 in 2005 values for this, i.e. year-2012 USD 280–340 billion, which is of a similar size. This number cannot be further refined for regions and commodities, so we thus put the global share of food wastage in agricultural production at a third. Furthermore, the actual global situation indicates only a decrease, not a total pollinator loss. (Garibaldi, Steffan-Dewenter *et al.* 2013) estimated that overall, agricultural systems are managed in such a way that pollination services are about 50 percent of the optimal level, resulting in 24 percent lower yields than would be possible for pollinator dependent crops. Thus, this share of costs could be used for a first gross cost estimate. If pollinator loss were fully due to agriculture, food wastage would then be responsible for about USD 25–30 billion (i.e. 330*0.24*1/3, assuming a world average for food wastage of a third of production). Given that there are other drivers, but that agriculture is the most important one, we may assume a value of USD 20–25 billion. Clearly, this is a very gross estimate that has to be further refined, in particular on regional scale, and regarding the contribution of agriculture. In addition, the model from (Bauer and Wing 2010) which is behind the data used here looked at pollinators' complete extinction. Cost estimates for the loss of only a fraction of the pollinators thus needs to be addressed in more detail, in particular as the relationship between pollinator losses and economic impacts likely is not linear, assumingly facing increasing marginal losses with increasing extinction levels.

This second estimate is higher than the first, but changing some assumptions may bring them closer together. For example, the first estimate might be a underestimation as it calculates how much the lost bees could have produced but not how much more production could have happened if there had been as many bees as optimally needed, as this optimal situation might necessitate even higher pollinator populations than if all the dead bees were kept alive. We thus decided to keep a central value of USD 15 billion with a range from USD 1-25 billion.

3. Well-being Valuation of Social Costs due to Environmental Damage

3.1 Background

In addition to economic costs and environmental costs, food wastage potentially presents a broad range of social costs. The social costs of carbon (described in section 2.2.1 refer only to a part of social costs. General resource depletion and pollution due to agriculture leads to additional costs, such as the individual and societal health costs due to various pollutants (e.g. chronic well-being losses due to health impact of pesticide exposure) as well as food security risks, loss of livelihoods, likelihood of civil conflict and increases in crime due to resource depletion, or loss of well-being and societal value due to loss of habitat and landscape amenities or species extinction with related existence value losses. As laid out in the FCA framework presented in section 1.2 part of these costs are monetized with the well-being valuation approach and complement the environmental costs discussed in the previous section. Due to data availability, the well-being valuation only covers a small part of those and focuses on livelihood loss, individual health and conflicts (cf. sections 3.2-3.5).

These social costs should be measured in terms of losses to human welfare or quality of life in line with microeconomic theory. Broadly speaking, there are two components of social cost:
i) primary costs – felt by the individual in terms of direct impacts on quality of life or well-being;
ii) secondary costs – felt more widely by society as a whole, such as increased health expenditures (medical services, medication, etc.) due to adverse health effects.

Primary and secondary effects and costs (or benefits) are equally important components of cost-benefit analysis and full-cost accounting. All three types of cost – economic, environmental and social – will have both primary and secondary costs. The well-being valuation approach focuses on the primary social costs, while the environmental and economic costs described in section 2 are mainly secondary.

The well-being valuation (WV) approach is used to derive social costs associated with livelihood loss, health damages and conflict due to food wastage. As explained in section 1.2.6, there are a number of ways to estimate values and costs. Social costs are estimated by first assessing the extent to which environmental damage from food wastage impacts negatively on livelihoods, health and conflict. In turn, the impact of livelihood loss, health damages and conflict on people's self-reported well-being (that is, their subjective well-being) is estimated. These represent the primary costs associated with livelihood loss, health damages and conflict due to food wastage.

3.2 Well-being valuation: statistical methodology

The following approach has been adapted from Fujiwara (2013) and Jujiwara and Dolan (2014) to estimate the costs associated with livelihood loss, health damages and conflict due to food wastage.

3.2.1 Model 1: estimating livelihood, health and conflict impacts on well-being

First, the impact of livelihood loss, health damages and conflict on life satisfaction is estimated for a broad number of countries across the world using the following regression model:

$$LS_i = \alpha+\beta_H H_i+\beta_C C_c+\beta_L L_i+\beta_X X_i+\varepsilon_i \qquad (1)$$

where LS_i = life satisfaction of individual i (measured on a 1-10 point scale in response to the question "All things considered, how satisfied are you with your life as a whole these days?"); H_i = health of individual i; C_c = whether a conflict has occurred in individual i's country C; L_i = livelihood of individual i; X_i = a vector of other determinants of life satisfaction and ε_i = the error term under the standard assumptions. A full description of the variables can be found in Table 1 below.

The model is run using ordinary least squares (OLS) regression, as is standard practice in much of the SWB literature[6].

3.2.2 Model 2: estimating environmental impacts on livelihood, health and conflicts

In order to link livelihoods, health and conflict to food wastage, models are estimated to derive the impacts of the environmental conditions associated with food wastage on these three outcomes:

$$D = \alpha+\beta_Z Z + \varepsilon_i \qquad (2)$$

6 Ferrer-i-Carbonell and Frijters (2004) found that it makes little difference in well-being models whether one assumes cardinality or ordinality in the well-being variable and, hence, OLS is used for ease of interpretation.

where D represents the three domains we are interested in (livelihoods, health and conflict) and Z is a set of explanatory variables that includes environmental conditions associated with food wastage. Equation (2) is a simplified notation of the models and the equation is run three times – once for each outcome. Livelihoods and health models are estimated at the individual level, including country-level variables on environmental damage. The conflict model, estimated at the country level and used to predict conflict from environmental damage, uses a logit[7] model. The livelihoods and health models are estimated using OLS and, hence, assume cardinality in these variables. From the SOL model, there is data on the following environmental variables that potentially can be used for such analysis:

- climate change (tonnes CO_2e) (three measures in total);
- land use (ha) (three measures in total);
- water erosion (tonnes soil lost/year);
- deforestation (ha/year);
- pesticide use;
- water use (m^3);
- non-renewable energy demand;
- N-surplus (two measures in total);
- P-surplus (two measures in total).

Many of these variables are highly correlated with each other and some may not measure what is intended. For example, factors that are used as part of income-generating agricultural activities may show up positively in regression analysis. Thus, a set of environmental variables was determined, based on the following criteria: i) environmental variables that are unambiguously bad, ii) environmental variables that could be expected to impact on the outcomes, in a theoretical or empirical sense, and iii) environmental variables that have good data for linkage back to food wastage data.

Land use, non-renewable energy demand, deforestation, N-, P- and water use were ruled-out on the basis that land, energy, wood and water provide resources for production and, hence, may have positive outcomes. These variables are likely to show up positively from an individual perspective as they are associated with resource inputs in production processes, whereas from a societal perspective they would show up as negatives. Of course, factors such as deforestation will also show up negatively at an individual level for indigenous communities that lose their traditional forest living areas, but these communities will not be represented in the WVS. Hence deforestation, on average, will show up as positive for individual well-being but would probably show-up as negative overall if indigenous people living in the forests and very rural communities were included in the surveys. Since the well-being valuation approach is based on well-being data and analysis at the individual level, we find that these types of environmental variables tend to show up positively. Thus, increases in these variables are likely to be correlated with less conflict, improved livelihoods, and maybe even better health. It could also be argued that pesticides have this characteristic, but a large body of evidence

7 A logit model is a regression model estimated for binary outcomes rather than outcomes on a continuous scale.

from the health sciences shows that pesticide use has a negative impact on health. Climate change is excluded on the basis that the social cost of carbon is included elsewhere in the model.

The models therefore use soil erosion for livelihoods and conflict, and pesticide usage for health damages. These variables are arguably unambiguously bad for the outcomes of interest and can be expected to have direct impact on the three domain areas, in a theoretical or empirical sense. In fact, soil erosion leads to problems such as desertification and decreases in agricultural productivity due to land degradation, which will have clear implications for livelihoods and resource-based conflict. Also, there is a large body of evidence on the link between pesticide use and health damages.

The partial derivatives from equations (1) and (2) are used to estimate the impact of environmental damage on life satisfaction via livelihood loss, health damages and conflict[8]:

- β_H x β_Z = the impact of pesticide use on life satisfaction via health.
- β_L x β_Z = the impact of water erosion on life satisfaction via livelihood loss.
- β_C x β_Z = the impact of water erosion on life satisfaction via conflict (where here β_Z has been converted from an impact on the log odds ratio to a probability impact).

These results can be used to attribute the impact on life satisfaction due to food wastage since environmental damages (water erosion and pesticide use) can be related back to food wastage levels in SOL-m.

3.2.3 Model 3: estimating well-being costs related to food wastage

Finally, WV is used to estimate the costs associated with livelihood loss, health damages and conflict due to food wastage. This is achieved by assessing the marginal rate of substitution between the non-market outcome and income. The WV model estimates the amount of income required to compensate people for the negative impacts of natural resource degradation caused by food wastage on livelihood, health and conflict.

3.2.4 The Well-being Valuation approach

Formally, Compensating Surplus (CS) is estimated as follows in the WV approach:

$$\boldsymbol{v}(p^0, Q^0, M^0) = \boldsymbol{v}(p^1, Q^1, M^1 + CS) \qquad (3)$$

where $\boldsymbol{v}(\cdot)$ is the indirect utility function; M = income; Q = the good being valued; p = prices. The *0* superscript signifies the state before Q is consumed (or without the good) and the *1 superscript* signifies the state after consumption (or with the good). Here Q is a non-market "bad" (such as health damages) in that it impacts negatively on utility $(\partial v/\partial Q < 0)$.

8 The relationship between environmental conditions (water erosion and pesticide use) and the three outcomes (livelihoods, health and conflict) is assumed to be linear in the models. This is a simplifying assumption but it is unlikely to make a large difference using non-linear parameters, given that the effect sizes are very small/marginal.

In practice, well-being valuation works with an "observable" measure of welfare (i.e. self-reported well-being rather than preferences) and it is possible to estimate the marginal rate of substitution between M and Q to measure CS using the *direct utility function* $u(\cdot)$:

$$u(Q, M, X) \qquad (4)$$

where X is a vector of other determinants of welfare (u). Empirically what is measured is:

$$LS(Q, M, X) \qquad (5)$$

where LS = life satisfaction. This is a short-hand version of equation (1) (where Q = livelihood loss, health and conflict):

$$LS_i = \alpha+\beta_H H_i+\beta_C C_c+\beta_L L_i+\beta_X X_i+\varepsilon_i \qquad (1)$$

Now equation (3) can be substituted into (1) (substituting Q for livelihood loss, health and conflict, and separately showing the income variable (M) which was previously included in the vector X in equation (1):

$$LS_i\,(\alpha+\beta_M M_i^0+\beta_Q Q_i^0+\beta_X X_i^0+\varepsilon_i) = (\alpha+\beta_M(M_i^1+CS) + \beta_Q Q_i^0+\beta_X X_i^1+\varepsilon_i) \qquad (6)$$

And solve for CS:

$$CS = e\left[\frac{-\beta Q}{\beta M} + \ln(M^0)\right] - M^0 \qquad (7)$$

In equation (7), β_M = the impact of income on life satisfaction and β_Q = the impact livelihood loss, health or conflict. As it stands, equation (7) will simply place values on livelihood loss, health or conflict overall. In order to estimate the costs associated with livelihood loss, health and conflict due to environmental damage, the cross-products of the impacts of environmental damage and livelihood loss, health and conflict can be used such that CS is estimable for each outcome as follows.

1. Cost of health damages due to an additional unit of pesticide use:

$$CS = e\left[\frac{-\beta H \times \beta Z}{\beta M} + \ln(M^0)\right] - M^0 \qquad (8)$$

2. Cost of livelihood loss due to an additional unit of water erosion:

$$CS = e\left[\frac{-\beta L \times \beta Z}{\beta M} + \ln(M^0)\right] - M^0 \qquad (9)$$

3. Cost of conflict due to an additional unit of water erosion:

$$CS = e\left[\frac{-\beta C \times \beta z}{\beta M} + \ln(M^0)\right] - M^0 \qquad (10)$$

The term $e[\cdot]$ accounts for the logarithmic form of the income variable in the income model and M^0 = sample average income. This means that value or cost estimates depend positively on level of income. For a given effect size on life satisfaction (e.g. $\beta c \times \beta z$), the richer the sample is, the higher the compensation value will be, because it takes more money to compensate people with a higher marginal utility of income.

A key technical issue involved in estimating monetary values in WV is that we have a robust estimate of the *causal effect* of income and the non-market good on life satisfaction. This issue is especially problematic for income. The income variable in life satisfaction models suffers from endogeneity due to reverse causality and selection effects and measurement error, which all tend to lead to *downward* bias in the income coefficient. Since the income coefficient acts as the denominator in the calculation of value in equations (8) to (10), this leads to an upward bias in the value of non-market goods using the WV method. For example, Clark and Oswald (2002) estimated the value of employment to be about £20,000 ($30,000 in 2002[9]) per month in addition to wage income. The evidence tends to suggest that happier people may be more likely to earn less or that there are important unobservable (to the econometrician) factors that cause people to earn less, while also helping them to be happy anyway. This adds to the downward bias created by measurement error in the income variable, which will lead to an underestimate of the impact of income on SWB[10].

Building on Apouey and Clark (2009), Gardner and Oswald (2007) and Fujiwara (2013) to deal with the issue of causality for the income variable in well-being models, an Instrumental Variable (IV) approach is used with lottery winners, which eliminates the correlation between the error term and the income variable due to measurement error and/or endogeneity. By law, lottery wins are random among lottery players and, by comparing small versus mid-sized lottery winners, the exclusion restriction also holds for IV. Extensive data on lottery playing is available in the British Household Panel Survey (BHPS) and hence, the BHPS dataset is used to estimate the causal impact of income on life satisfaction (β_M) in equation (6). The BHPS is used because there is no IV available for income in the WVS. Thus, following Fujiwara's (2013) Three Stage Well-being Valuation approach, the estimate of β_L x β_Z and β_H x β_Z is derived from the WVS data, while β_M is derived from the BHPS. Fujiwara (2013) finds that β_M = 1.1. This suggests that about a 120 percent increase in income leads to a 1.1 index point increase in life satisfaction measured on a scale of 1 to 7.

3.2.5 Benefit transfer

Note that β_C x β_Z, β_L x β_Z and β_H x β_Z are global estimates based on the full set of countries in the WVS, but that comes from a UK sample population using the BHPS. A benefit transfer technique can be applied to the valuation methodology here by adjusting M^0 in equations (8) to (10).

Benefit transfer takes a different approach in WV, as compared to preference-based valuation. In preference methods, the determinants of WTP/WTA are modelled and values are "transferred" to other country contexts. In WV in this current context, we "transfer" values by adjusting the impact of income on life satisfaction (β_M) by the average levels of income in the sample. This aligns with the mainstream economic theory on diminishing marginal utility of income. The estimates of the impact of the non-market goods/outcomes have already been modelled for our global sample of countries and, hence, do not need transferring.

9 Converted in to US$ using June 2002 exchange rates (£1 = $1.50) (XE Currency Convertor http://www.xe.com/currencyconverter).
10 This is inferred from the fact that studies that have used instrumental variables for income in SWB models to solve for endogeneity and measurement error problems have tended to consistently find that the income coefficient increases (see Pischke 2010 and Fujiwara and Campbell 2011).

3.3 What costs are captured in the well-being valuation approach?

There will be exclusions of certain types of social cost in two dimensions. First, there are exclusions in coverage (i.e. in scope or width) as the WVS data do not allow an analysis of all types of social impact and some are not covered in the estimates, such as impacts on crime and education of children. Second, there are exclusions in depth (i.e. in how detailed these costs are assessed within each cost type), as the cost estimates for livelihood loss, health and conflict do not capture all costs related to these issues, due to data restrictions. The aspects covered in the well-being valuation approach are summarised in Table 6.

3.3.1 Conflict

Primary costs: Conflict impacts negatively on quality of life for victims (those who are injured or killed) and for indirect victims (those who do not personally suffer any harm but suffer emotional consequences). The cost of conflict captures costs associated with injured victims and indirect victims. We therefore do not capture the costs of loss of life for people who are killed.

Secondary costs: Conflict also has economic and environmental costs. These include loss in natural and human capital, loss in national productivity and GDP, and environmental damage due to conflict. The WV method picks up the costs at an individual level, focusing on impacts on individual quality of life and hence does not include (and for accounting purposes are additional to) these economic and environmental costs.

3.3.2 Health

Primary costs: Adverse health conditions impact directly on the quality of life of the individual and indirectly on the quality of life family members. The well-being valuation method picks up the health costs at an individual level and is not able to capture third party (i.e. family) related costs in the WVS data.

Secondary costs: Health will have economic costs in the form of loss in human capital and GDP and increases in health care expenditures. These costs are not captured in the well-being valuation approach, meaning that the well-being values are additional to these economic costs.

3.3.3 Livelihood loss

Primary costs: Livelihood loss impacts negatively on people's quality of life. Food wastage may lead to increased food security risk and loss of income. The well-being valuation approach assesses the cost of livelihood loss due to these types of factors. That is, the costs associated with livelihood loss can be assumed to depend on factors such as food insecurity and income loss, even though the exact channels cannot be tested in the data).

Secondary costs: Livelihood loss may have some economic and environmental impacts but the main impact is likely to fall on individuals' quality of life and will be captured by the well-being valuation approach.

Table 6: Social costs related to conflict, health damages and livelihood loss that are captured in the well-being valuation model

Well-being factor	Primary social costs included	Primary social costs excluded	Secondary social costs excluded	Environmental costs excluded
Health due to pesticide exposure	Quality of life of affected individuals	Quality of life of relatives	Loss of human capital and GDP	
Conflict due to soil erosion	Quality of life of injured and indirect victims	Deceased victims	Loss of human capital and GDP	Environmental damage due to conflict
Livelihoods due to soil erosion	Income and food security of individuals	None	None	

3.4 Data

Models 1 and 2 use the fifth World Values Survey (WVS) (2005–2008), with a sample size of just under 83 000 individuals from 55 countries across all continents. The WVS is the largest global dataset in the world that contains data on subjective well-being. The list of the countries can be found in Table 7.

Livelihoods are measured as self-reported satisfaction with the financial situation of the household. This is based on the assumption that any threat to a household's livelihood and "financial health" will show up in people's satisfaction rankings. This will include many types of threats, such as loss of income, increases in consumer prices and resource depletion. Food security will show up in the livelihood measure because heightened food risks translate into increases in food prices (due to supply constraints). Where people are affected by increased food prices, they will report a decrease in satisfaction with the financial situation of the household. Thus, food security will be an element of the livelihoods measure in addition to loss of income and other potential factors.

Table 7: Countries used in the data analysis

Andorra	France	Mexico	Spain
Argentina	Georgia	Moldova	Sweden
Australia	Germany	Netherlands	Switzerland
Brazil	Ghana	New Zealand	Thailand
Bulgaria	Guatemala	Norway	Trinidad and Tobago
Burkina Faso	India	Peru	Turkey
Canada	Indonesia	Poland	Ukraine
Chile	Iran	Romania	Uganda
China	Iraq	Russia	UK
Colombia	Italy	Rwanda	USA
Cyprus	Japan	Serbia	Uruguay
Egypt	Jordan	Slovenia	Vietnam
Ethiopia	Malaysia	South Africa	Zambia
Finland	Mali	South Korea	

Health is measured, as broadly as possible, as self-reported overall health using responses to the following question, "All in all, how would you describe your state of health these days? (1='Very good'; 5= 'Very poor')."

Conflict measure is based on data from Uppsala University's Conflict Data Programme[11]. Any country listed as being in conflict and which had 25+ deaths in a single year between 2005 and 2008 is defined as a conflict country in the analysis. The list of conflict countries during this period can be found in Table 8. The life satisfaction question in the WVS is set on a scale of 1–10 (1='Dissatisfied'; 10='Satisfied').

Table 8: Conflict countries during the period 2005–2008

Colombia	Iraq
Ethiopia	Mali
India	Russia
Iran	Thailand

11 The Uppsala Conflict Data Program has recorded ongoing violent conflicts since the 1970s. The data provided is one of the most accurate and well-used data-sources on global conflicts. http://www.pcr.uu.se/research/UCDP/

The income model for Model 3 is estimated using the BHPS, which is a nationally representative sample of over 10 000 adult individuals conducted between September and December of each year, from 1991 to present. Respondents are interviewed in successive waves, and all adult members of a household are interviewed. The life satisfaction question was added to the BHPS in 1997. Individuals are asked "How dissatisfied or satisfied are you with your life overall?" and then asked to rate their level of satisfaction on a scale of 1 (not satisfied at all) to 7 (completely satisfied). Information on the lottery data and estimation methodology for the instrumental variable can be found in Fujiwara (2013).

Note that the reporting scale for the life satisfaction variable differs across the BHPS and WVS datasets. Life satisfaction impact estimates are normalized in the WVS on a 1–7 scale, so that the results are directly comparable to the life satisfaction responses in the BHPS.

The variables used in the WVS analysis are presented in Table 9.

Table 9: World Values Survey variable descriptions

Variable Name	Question	Scale
Life satisfaction	Satisfaction with your life	1 to 10
Livelihood	Satisfaction with financial situation of household	1 to 10
Age	Age in Years	Continuous
Age^2	Age squared	Continuous
Religion	If belong to a religious denomination	Binary
Male	Male gender	Binary
Married	If currently married	Binary
Children	If has any children	Binary
Education	Has the respondent completed secondary education or above	Binary
Unemployment	Employment status: unemployed	Binary
Health	State of heath (subjective)	1 to 5
Social	If see themselves as a member of their local community	Binary
Conflict	Whether the respondent lives in a country that is in conflict	Binary
Member of an environmental organization	Belong to a conservation, environment, ecology or animal rights group	Binary
Water erosion	Water erosion in respondent's country (tonnes in soil per year)	Continuous
Pesticide	Pesticide use (dimensionless: per ha)	Continuous
Income	GDP per capita ($) in the respondent's country	Continuous

3.5 Results

3.5.1 Model 1. Life satisfaction, livelihood loss, health damages and conflict

Livelihood loss and conflict have negative effects on life satisfaction and health has a positive effect. All three variables are significant at the 5 percent level[13] (see Table 10).

Table 10: Subjective well-being model (life satisfaction)

Dep Var: Life Satisfaction	Coefficients	Standard Errors
Livelihood	0.437[a]	0.022
Age	-0.007	0.004
Age squared	0.0001[b]	0.0005
Religion	-0.078	0.069
Male	-0.104[a]	0.019
Married	0.046	0.057
Kids	0.099[b]	0.039
Education	0.069	0.076
Unemployed	-0.233[a]	0.077
Health	0.576[a]	0.037
Social	0.170[a]	0.052
Conflict	-0.427[b]	0.189
Member environmental organization	0.058	0.062
Constant	1.871[a]	0.186
Sample size	55,931	
R-sq	0.34	

Notes: OLS regression. [a] significance at 1%; [b] significance at 5%

In an analysis not shown here (but available on request), the BHPS and Understanding Society data from the UK were used to assess the relationship between self-reported subjective measures of health (on a scale of 1 to 5) and actual health conditions[14]. Every one of the 16 health conditions in the data (ranging from arthritis and asthma to stroke and diabetes) were significantly negatively associated with self-reported health. Thus, one can be confident that single-dimension self-reported health scales are a good representation of individuals' health status.

12 Binary indicates that this variable takes on the value of "1" if the individual responds affirmatively to the question or "0" otherwise.
13 For an introduction to significance testing, see www.law.uchicago.edu/files/files/20.Sykes_.Regression.pdf
14 Including: asthma, arthritis, congestive heart failure, heart disease, angina, heart attack, stroke, emphysema, hyperthyroidism, bronchitis, liver condition, cancer, diabetes, epilepsy, high blood pressure, clinical depression.

3.5.2 Model 2. Impact of environmental damages on livelihoods, health and conflict

Table 11: Impact of water erosion on financial satisfaction (livelihoods)

Dep Var: Life Satisfaction	Coefficients	Standard Errors
Age	0.019[a]	0.001
Male	-0.053[a]	0.018
Married	0.248[a]	0.021
Kids	-0.415[a]	0.026
Education	0.512[a]	0.018
Health	0.835[a]	0.011
Water erosion	$-3.11*10^{-11}$[a]	$1.04*10^{-11}$
Constant	1.703[a]	0.054
N	72,118	
R-sq	0.10	

Notes: OLS regression. [a] significance at 1%; [b] significance at 5%

After controlling for other variables that may impact on livelihoods, water erosion was negatively associated (<1%) with household financial satisfaction. Since livelihood has a positive impact on well-being (Table 11), this inferred that water erosion has a small but negative effect on well-being through loss of financial security or livelihood. Note that the regression in Table 11 controls for employment status so the negative effect on financial satisfaction can be seen as the effect on people's perceived livelihood loss, in addition to any loss of income due to unemployment.

The magnitude of this indirect impact can be estimated using the product of the partial derivatives.

Impact of water erosion on life satisfaction through reduced livelihoods = $0.437*-3.11*10^{-11} = -1.36*10^{-11}$.

Table 12: Impact of pesticide usage on health

Dep Var: Life Satisfaction	Coefficients	Standard Errors
Age	-0.013[a]	0.0002
Male	0.084[a]	0.006
Married	0.059[a]	0.007
Kids	-0.036[a]	0.009
Education	0.187[a]	0.006
Pesticide use	$-1.96*10^{-11}$[a]	$5.80*10^{-12}$
Constant	4.243[a]	0.01
Sample size	73,006	
R-sq	0.09	

Notes: OLS regression. [a] significance at 1%; [b] significance at 5%

After controlling for other variables that may impact health, pesticide usage was negatively associated (<1%) with self-reported general health. Since health has a positive impact on well-being (Table 12), this inferred that pesticide usage has a small but negative effect on well-being through its adverse effects on health. The magnitude of this indirect impact can be estimated using the product of the partial derivatives.

Impact of pesticide use on life satisfaction through adverse effects on health = $0.576 * -1.96*10^{-11} = -1.13*10^{-11}$.

Table 13: Impact of water erosion on conflict (national level)

Dep Var: Life Satisfaction	Coefficients	Standard Errors
Water erosion	$8.10*10^{-10c}$	$-4.83*10^{-10}$
GDP per capita	0.178	-0.42
Constant	-2.997	-2.035
N	53	
R-sq	0.09	

Notes: Logit regression. [a] significance at 1%; [b] significance at 5%; [c] significance at 10%. R-sq is the Pseudo R-sq.

After controlling for average income levels, water erosion was negatively associated (<10%) with increased probability of national conflict (a percentage change increase of $8.57*10^{-11}$%). Since national conflicts have a negative impact on well-being (Table 13), this inferred that water erosion has a small but negative effect on well-being through adverse effects on conflict probability. Therefore, the significance level of <10% is rather low. The magnitude of this indirect impact can be estimated using the product of the partial derivatives.

Impact of water erosion on life satisfaction through increased probability of conflict = $-0.43*8.57*10^{-11} = -3.68*10^{-11}$.

As discussed in section 3.3.1, the estimated impact of conflict (due to water erosion) on well-being does not include the costs of the lives lost in conflict. It is the cost for people who are affected by conflict but who are still alive.

Note that in all these three regressions of well-being determinants on environmental damages, the coefficient of determination, i.e. the variance explained (R-square) is at about 10 percent and thus, rather low. This means that, besides the variables included in these regressions, other variables should be added to explain a bigger part of the variance observed in the data – however, this is not possible, due to lack of data. Rather low values for R-square are expected and not alarming in such contexts of regressions for human behaviour and well-being, but nevertheless, they should be accounted for when deriving conclusions. For example, it should be emphasized that observed variance in the data is only partially explained by the explanatory variables used, and that other potentially important influences play a role. This is particularly important for consequences of reducing food wastage: in a context of low R-square levels, a reduction of food wastage will result in the reduction of the corresponding impacts on average over a large number of countries only. For each single country, the level of health, livelihoods and conflicts after food wastage reduction can change into any direction and can only be predicted with high uncertainty (as only 10 percent of the level is due to the influence of food wastage, while 90 percent is due to other influence factors).

3.5.3 Residual effects

Testing was also done to determine whether there are any residual effects of water erosion and pesticide use on life satisfaction over and above any impact on health, conflict and livelihoods, in order to check whether there were any further costs that should be measured and included. This was done by adding water erosion and pesticide use to the overall well-being model in equation (1).

After controlling for health, conflict and livelihoods, it was found that water erosion and pesticide use do not have a direct effect on life satisfaction. This suggests that the main effects are indirectly captured through health, conflict and livelihoods (Table 14).

Table 14: Subjective well-being model with water erosion and pesticide use

Dep Var: Life Satisfaction	Coefficients	Standard Errors
Livelihood	0.432[a]	-0.022
Age	-0.005	-0.004
Age^2	0.000[b]	0
Religion	-0.047	-0.068
Male	-0.104[a]	-0.019
Married	0.046	-0.059
Kids	0.085[b]	-0.039
Education	0.043	-0.071
Unemployed	-0.248[a]	-0.075
Health	0.594[a]	-0.035
Social	0.141[a]	-0.048
Conflict	-0.410[c]	-0.215
Water Erosion	$-9.54*10^{-11}$	$-1.26*10^{-10}$
Pesticide	$1.60*10^{-10}$	$-1.94*10^{-10}$
Constant	1.857[a]	-0.196
N	55,796	
R-sq	0.34	

Notes: OLS regression. [a] significance at 1%; [b] significance at 5%; [c] significance at 10%

3.5.4 Valuation

Mean annual income (GDP per capita) of the sample countries is USD 13 689. Using the results from Tables 10 to 13, cost estimates were derived using well-being valuation for livelihood loss, health damages and conflict due to environmental damage (Table 15).

Cost of health damages due to an additional unit of pesticide use:

$$CS = e\left[\frac{-\beta H \times \beta Z}{\beta M} + \ln(M^0)\right] - M^0 = e\left[\frac{-1.13e\text{-}11}{1.1} + \ln(13{,}689)\right] -13{,}689 = \$1.18*10^{-7}$$

Cost of livelihood loss due to an additional unit of water erosion:

$$CS = e\left[\frac{-\beta L \times \beta Z}{\beta M} + \ln(M^0)\right] - M^0 = e\left[\frac{-1.36e\text{-}11}{1.1} + \ln(13{,}689)\right] -13{,}689 = \$9.83*10^{-8}$$

Cost of conflict due to an additional unit of water erosion:

$$CS = e\left[\frac{-\beta C \times \beta Z}{\beta M} + \ln(M^0)\right] - M^0 = e\left[\frac{-3.68e\text{-}11}{1.1} + \ln(13{,}689)\right] -13{,}689 = \$3.21*10^{-7}$$

Table 15: Costs derived from well-being valuation

Impact	Environmental factor	Coefficient (product)[1]	USD cost per unit[2]
Livelihoods	Water erosion (tonne soil lost)	$-1.36*10^{-11}$	$\$1.18*10^{-7}$
Health	Pesticide use	$-1.13*10^{-11}$	$\$9.83*10^{-8}$
Conflict	Water erosion (tonne soil lost)	$-3.68*10^{-11}$	$\$3.21*10^{-7}$

Notes:
[1] Indirect effect of environmental factor on life satisfaction.
[2] Average amount of individual-level monetary compensation required to offset a one unit increase in the environmental factor (annual costs per person per one unit increase).

3.5.5 Acute health impacts of pesticide use

As there is some data available on acute health treatment costs due to pesticide use, the estimate for the costs of pesticide use based on the well-being approach just derived is complemented with such acute health costs of pesticide use. While the costs based on the well-being approach refer to losses of individual well-being from pesticide use impacts (i.e. primary costs), those costs refer to societal costs, i.e. secondary costs. These costs – USD 2.2 million – were estimated with benefit transfer for developed countries from UK numbers as given in (Pretty, Brett *et al.* 2000a). They are also similar to Tegtmeier and Duffy (2004) results of USD 1 281 million reached when using USA numbers, and to Praneetvatakul *et al.* (2013) results of USD 426 million reached for developing countries with values from Thailand. As for the costs of pesticides in drinking water, country-specific pesticide use intensities were taken into account. For further methodological details, see section 2.2.1 on the costs of pesticides in drinking water.

It should be emphasized that this cost category mainly covers medical treatment costs of acute pesticide poisoning events. It does not cover costs from chronic health effects due to pesticide exposure nor the costs of individual well-being losses due to impoverished health from pesticide exposure. The latter aspect is covered in the well-being estimates for health effects given in the previous section. Thus, the health costs reported here and the well-being estimates related to health cover different cost categories and do not result in double counting. It is also important to note that these estimates are based on a rather gross and qualitative indicator for pesticide use and exposure intensity in single countries. This cannot account for the huge range of different pesticides currently in use that have different effects and behaviours in the environment, on biodiversity and on people, for example due to different decay time or effects on organisms metabolism.

Data on aggregate health effects of pesticide use is very rare, but for the estimates attempted here, effects of single pesticides would be too narrow. In addition to the studies used above, a range of publications is available from the Pesticides Policy Project of the Institute for Development and Agricultural Economics (IFGB) at the University of Hannover (IFGB 2014). Screening the reports revealed that few countries have national estimates of pesticide use health impacts, and many estimates are only for specific crops (e.g. Bt-cotton or coffee) and not for total agriculture. Only two dated studies, one from Mali (Ajayi, Camara *et al.* 2002) and one from Thailand (Jungbluth 1996) report aggregate health costs. While the Thailand study may be too old to be useful, the Mali study shows, for example, that direct health costs as reported here are only a small part of total pesticide health costs (in the case of Mali, only 7.5 percent). We refrained from using the Mali data for the FCA calculations, as there is no similarly encompassing study from a developed country to cover the developed countries via benefit transfer. In addition, the well-being approach undertaken in the previous sub-sections covers a relevant part of those impacts and is thus taken as an indication of these total costs. Just for illustration, scaling the acute health costs with this factor of 7.5 percent also results in an estimate on an order of magnitude (USD 112 billion) similar to the estimate based on the well-being approach (USD 145 billion).

3.5.6 Double counting

Impacts on livelihood and conflicts are both derived from water erosion impacts on soil, which increases the risk of double counting the negative effects. The impact of water erosion on conflict is measured after controlling for impacts on income (which is a proxy for livelihoods) and therefore, the conflict costs represent the cost as well as impacts on livelihoods. This means the two cost estimates for livelihood loss and conflicts can be added without any risk of double-counting.

It is also not the case that the costs estimated for livelihood loss, conflict and health damages based on the well-being approach overlap with costs estimated elsewhere in the FCA framework. Conflict and livelihood are only estimated with the well-being approach. With regard to health values, they are measured as societal damage costs related to pesticide use on the one hand, and as costs from individual well-being losses on the other. Thus, there is no risk of double-counting health impacts.

3.5.7 Economic benefits and costs

The values derived using the WV approach are costs per individual associated with the negative impacts of food wastage on livelihood loss, health and probability of conflicts related to natural resources degradation. They represent what is often called a "social cost" and are different from financial (or economic) costs. As de Goerter (2014) explained, there are large financial costs related to food wastage, but we should also acknowledge that there are some benefits to food wastage. It may be rational for final consumers (or even companies) to waste food because of the (often high) economic transaction costs involved in correctly matching food supply and demand. In other words, to some extent, the opportunity cost of resources involved in calculating exact future consumption or demand will outweigh the cost of the lost food, making it rational for some food to be wasted. This, of course, relies wholly on the economic definition of rationality and may conflict with other ethical standpoints that may put moral weight on food wastage per se (i.e. food wastage may be viewed as unethical under any circumstance). As already mentioned in section 1.2.1, there are also potential non-economic benefits of food wastage, such as those related to utility gains derived from choice.

In addition to this, there will be some positive labour market effects in that the harvesting and production of wasted food supports a large number of producers and workers across the world, and many of those who benefit economically are in developing countries. The utilitarian foundations of cost-benefit analysis and the full-cost accounting make no distinction between which utility-increasing activities can and cannot be included, but cost-benefit analysis generally ignores utility derived from illegal sources (e.g. drug use and crime) (Boardman *et al.* 2011). However, food wastage clearly does not fit into this category and other areas of economic activity are not discounted in national accounts where wastage is concerned.

Costs included in the FCA framework are not currently offset by gains in the labour market and, thus, it should be noted that net global costs will be over-estimates in this regard, unless an assumption is made that food wastage is ethically wrong regardless of the economic benefits that it can confer on some people.

3.6 Well-being valuation of global social costs of food wastage

The results from section 3.5 provide the per-unit well-being costs per person of soil erosion from water (i.e. tonnes soil lost) and pesticide use (dimensionless index), averaged on a global level. Those per-unit costs were then multiplied by the reported units lost due to food wastage and with the population in each country, in order to arrive at the country-wise well-being cost estimates due to food wastage in the three categories analysed. For consistency, it only considered the population of individuals aged 18+ years, as they were the only ones addressed by the well-being questionnaire. Clearly, there is a well-being loss for younger people and children as well, meaning that the values reported for adults only will be lower estimates of the true costs. For comparison, results for the full population including children are reported in some cases as well.

3.7 Regional differentiation

Finally, some gross regional differentiation of livelihood and health impacts is provided, based on splitting the data into OECD and non-OECD countries and estimating those two sets separately. This allowed capturing some regional differences while not reducing sample size too much. Further regional differentiation would only be possible with additional data. Due to the small sample size, no such regional analysis is possible for the conflict impacts. For this regional estimation, the approach described in this section has been followed, restricted to the following sets of countries covered in the World Values Survey (WVS) listed in Tables 16 and 17.

Table 16: OECD countries in the World Values Survey

Australia	Finland	Japan	Slovakia
Britain	France	Mexico	South Korea
Canada	Germany	Netherlands	Sweden
Chile	Hungary	New Zealand	Switzerland
Czech Republic	Israel	Norway	Turkey
Estonia	Italy	Poland	USA

Table 17: Non-OECD countries in World Values Survey

Andorra	Colombia	India	Morocco	Slovenia	Uruguay
Argentina	Egypt	Indonesia	Peru	South Africa	Vietnam
Brazil	Ethiopia	Iran	Romania	Taiwan Province of China	Zambia
Bulgaria	Georgia	Iraq	Russia	Thailand	
Burkina Faso	Ghana	Jordan	Rwanda	Trinidad & Tobago	
China	Guatemala	Malaysia	Serbia	Ukraine	

The results of this regional estimation of the impact of water erosion on life satisfaction through adverse effects on livelihoods and of the impact of pesticide use on life satisfaction through adverse effects on health are shown in Table 18.

Table 18: Individual costs derived from well-being valuation

Impact	Environmental factor	Coefficient (product)	USD cost per unit[15]
OECD			
Livelihoods	Water erosion (tonnes soil lost)	$-9.80*10^{-12}$	$8.54*10^{-08}$
Health	Pesticide use (toxicity level)	$-1.11*10^{-11}$	$9.67*10^{-08}$
Non-OECD			
Livelihoods	Water erosion (tonnes soil lost)	$-1.44*10^{-11}$	$1.25*10^{-07}$
Health	Pesticide use (toxicity level)	$-1.14*10^{-11}$	$9.93*10^{-08}$

4. Full Costs of Food Wastage: environmental, social and economic

4.1 Full costs of food wastage: global results

Table 19 shows the global results from the calculations described in Chapters 2 and 3 for the different impact categories. In total these amount to USD 2.6 trillion annually. This is roughly equivalent to the GDP of France or approximately twice total annual food expenditure in the USA[15]. It should be noted that many and very strong assumptions, simplifications and approximations are involved in the quantification of the impacts of food wastage and the related costs, in particular for the benefit transfer at a global scale. Thus, these results are only indicative of the order of magnitude of these costs and should be treated with caution. It is also emphasized that the following cost estimates cover only part of the full costs of food wastage, as certain impacts are not covered at all (see Chapters 2 and 3) and the impact costs covered capture only a fraction of the full costs. Therefore, the costs of the categories covered were determined by data availability and the possibility of establishing a linkage to food wastage. Also, these costs do not incorporate any of the potential benefits of food wastage discussed above. Chapters 2 and 3 contain further details and caveats for the calculations of each of the cost categories.

The full cost estimates, the environmental and social costs have been amended by basing economic costs on economic value and subsidies lost. Estimating the value lost due to food wastage incorporates the economic value of the wastage volumes, employs producer price data from FAOSTAT for wastage occurring at the production phase, and bases unit values on export/import prices from the trade data in FAOSTAT for the quantities lost and wasted at post-production value chain levels. Producer prices show considerably data gaps that were filled with the unit values, where those were available.

The FCA of food wastage also includes indirect economic losses due to public funds used to subsidize production of food that ultimately gets wasted. Data on this is available for OECD countries only and, thereby, for the EU-27 (as an aggregate only) (OECD 2012). These national subsidy figures can be related to agricultural land areas to determine average per hectare subsidy amounts. Combining those with the areas corresponding to food wastage quantities then leads to an estimate of subsidy loss due to food wastage. Therefore, only wastage at production level is included, as wastage at later supply chain levels may be imported from other OECD countries or from outside the OECD, and no information on this is available. Thus, it is not possible to derive the quantity of subsidies related to post-production waste. For OECD countries only, subsidies lost due to food wastage amount to USD 119 billion (2012).

15 Total food expenditure as calculated by the United States Department of Agriculture including purchases by consumers, governments, businesses and non-profit organizations. See: www.ers.usda.gov/data-products/food-expenditures.aspx

Table 19: Estimated costs of food wastage

Cost categories	Costs (billion USD, 2012)	Cost range (billion USD, 2012)[c]
Atmosphere		
Greenhouse gas emissions (without deforestation/organic soils)	305	45-1500
GHG from deforestation	72	10-350
GHG from managed organic soils	17	3-90
Ammonia emissions	1	
Water		
Pesticides in sources of drinking water	3	
Nitrate in sources of drinking water	1	
Pollution impacts of N eutrophication	3	
Pollution impacts of P eutrophication	17	
Water use (irrigation water)[a]	8	4-17
Water scarcity	164	
Soil		
Erosion (water)	35	7-70
Erosion (wind, very uncertain)	35	7-70
Land occupation (deforestation)	3	
Biodiversity		
Biodiversity impacts of pesticide use	1	
Biodiversity impacts of nitrate eutrophication	3	
Biodiversity impacts of phosphorus eutrophication	3	
Pollinator losses	15	1-25
Fisheries overexploitation	10	
Social[b]		
Livelihood loss	333	
Health damages (well-being loss)	145	
Acute health effects of pesticides	8	
Risk of conflict	396	
Economic		
Value of products lost and wasted	936	
Subsidies (OECD only)	119	
Sub-total environmental costs	696	
Sub-total social costs	882	
Sub-total economic costs	1055	
Total costs (all categories)	**2625**	

[a] The cost of irrigation water is included in the sub-total environmental costs as a proxy for water use; it is excluded from the total costs to prevent double counting as irrigation costs are already covered in the product value.
[b] When excluding children in the population numbers (as the well-being estimates are based on a sample of adults only), the total social costs sum to USD 579 billion (USD 229 billion livelihoods, 101 billion health, 249 billion conflicts). Those numbers more clearly underestimate these costs (as they neglect well-being losses from children) but are more accurate for the sample covered (i.e. for adults).
[c] Where no range is indicated, the numbers are point estimates indicating mid-values.

4.2 Full costs of food wastage: differentiation by regions and commodity groups

Assessments presented in this section, differentiated by regions and commodity groups, include only the most relevant cost categories where such differentiation is sensibly possible, given the data sources. These include the costs of greenhouse gas emissions, water scarcity, water pollution, soil erosion (from water), biodiversity/ecosystem impacts and lost production value. The differentiated assessment is presented separately by region and commodity groups, then in combination. The key physical impacts also are reported separately according to region and commodity group, as this helps to understand the patterns observed.

Due to the small sample size for the well-being estimates, values for OECD and non-OECD countries are presented with no further differentiation of the corresponding results. Table 20 presents the corresponding results.

Table 20: Well-being loss due to environmental impacts of food wastage for OECD and non-OECD countries (USD billion for 2012)

Costs	Global	OECD countries	non-OECD countries
Livelihood (adults)	229	8	231
Individual health (adults)	101	3	99
Conflict (adults)	249	n.a.	n.a.
Conflict (all population)	396	n.a.	n.a.

Note: The difference between OECD and non-OECD numbers for livelihood and individual health is due to basing calculations on per capita and annual costs of one unit of environmental impact (soil erosion/toxicity), and to the fact that these incidence levels are about 6 times higher in non-OECD countries than OECD, and that population in non-OECD countries is also about 6 times that of OECD. The costs from conflict referring to the full population (including children) are reported only for illustration. As already indicated in section 3.6, this is a biased estimate because the basis for those costs is derived from a sample of adult people only.

As a general caveat towards the results provided in the sections 4.2.1 – 4.2.8, it should also be noted that reported impacts and costs may not happen only in the regions reported but also may be partly in areas from which imports to these regions are sourced. Therefore, such regional analysis of costs can be somewhat misleading as to where they arise. This applies to wastage quantities at the post-production level and to feed inputs to animal production only, and is thus more relevant for industrialized regions than non-industrialized. Where this has some relevance, the countries and populations in the respective regions do not bear the totality of the impacts and costs reported. Data and modelling do not allow tracing of imports and exports in detail. However, also in such an assessment, the information provided correctly reports which impacts and costs can be saved if regional wastage is reduced but, again, it cannot be determined how much of those impacts and costs are saved within the region).

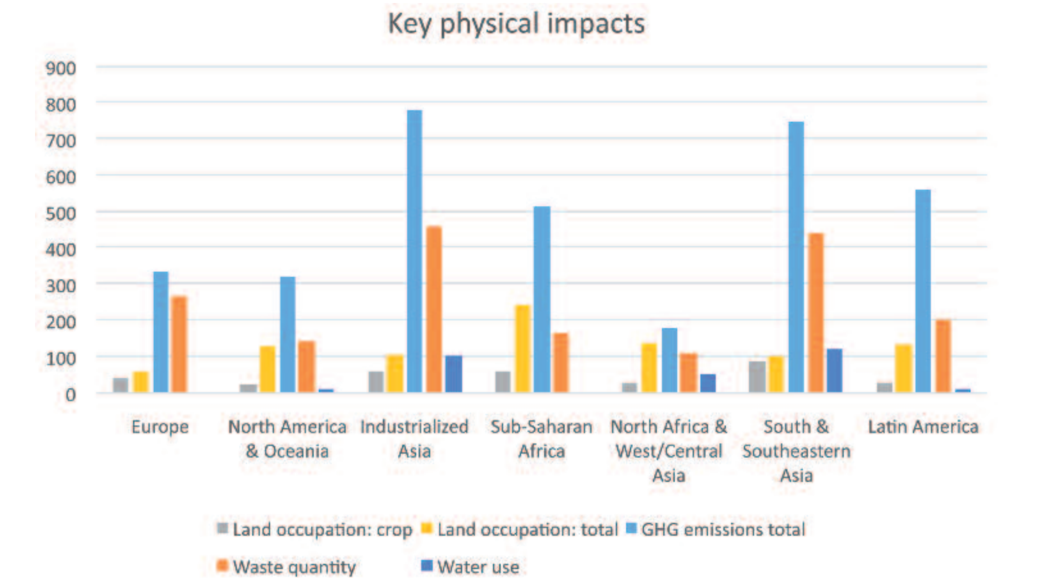
Non-traceability is particularly the case for greenhouse gas emissions costs, albeit due to other reasons. In this case, the data used (social costs of carbon) report global average costs of greenhouse gases emitted without any information on where the impacts of climate change and corresponding costs materialize and which countries and regions thus incur those costs. Thus, the values reported indicate the contribution of wastage from the respective regions and commodity groups to the corresponding global aggregate costs and not which costs may arise in a certain region due to wastage's contribution to climate change.

4.2.1 Global key impacts and costs by regions

The following separates the impacts and costs for regions on an all-commodity level, and for commodity groups on global level in sections 4.2.1 and 4.2.2, respectively. Presenting them separately helps to recognize the patterns emerging in the combined analysis of regions and commodity groups, as seen in sections 4.2.3 through 4.2.8.

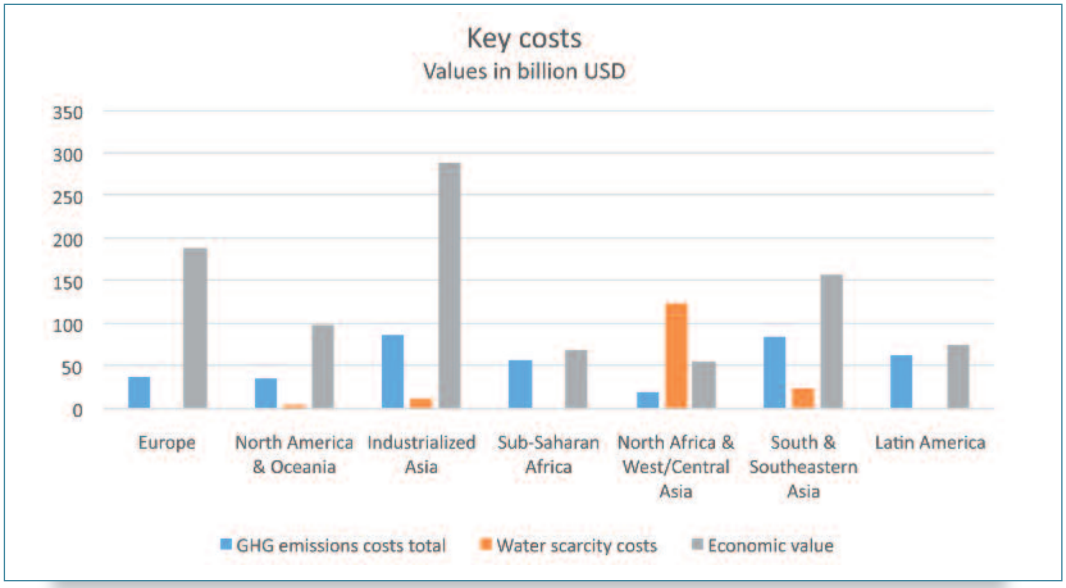
Figure 8 gives the regional overview on food wastage quantities and its physical impacts, reporting the numbers for the most important impact categories.

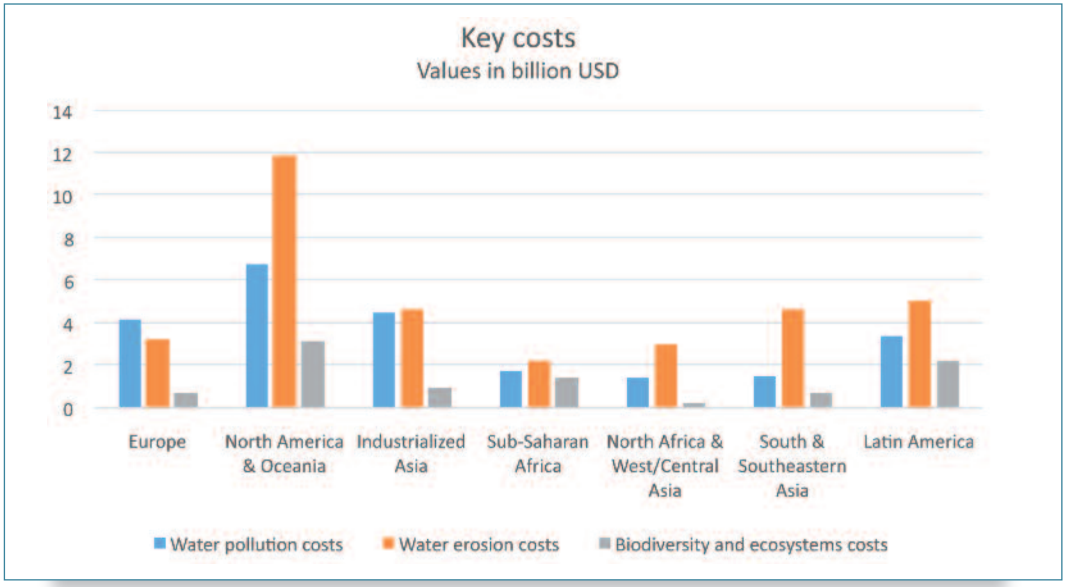
Figure 8: Key global environmental impacts of food wastage by regions



Values in million tonnes wastage, millions ha land occupation, million tonnes GHG emissions, and km^3 water use, all on the same axis.

Figure 9a and 9b: Key global costs of food wastage by regions (in billion USD)



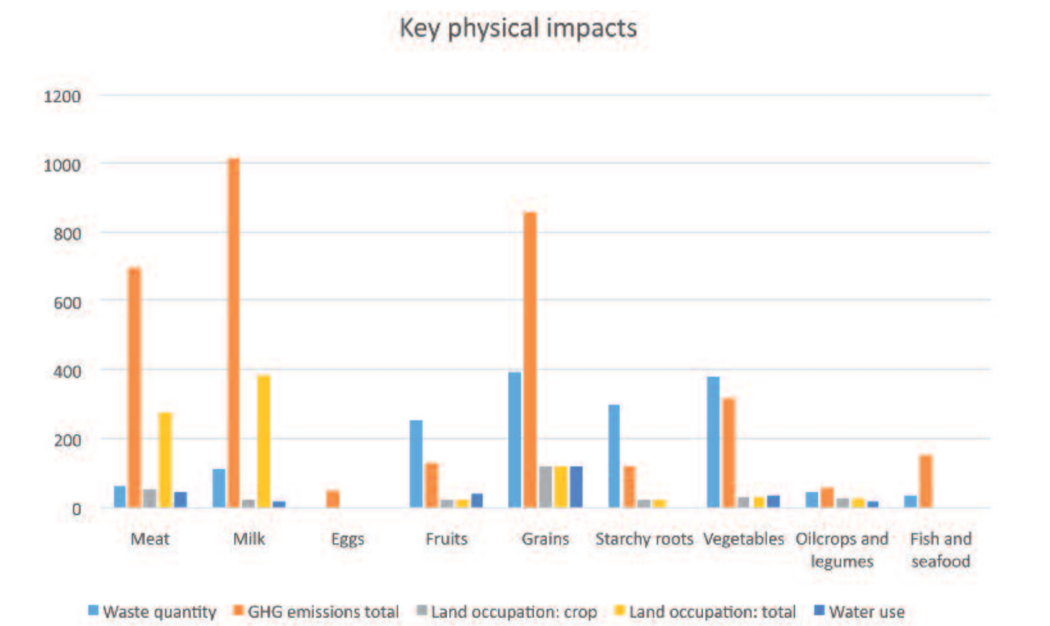


Figures 9a and 9b show the regional costs related to those physical impacts. Most important are the considerably high economic costs from lost produce value in most regions, the high water scarcity costs reported for North Africa and West and Central Asia, and the high soil erosion costs reported for North America and Oceania (the latter covering, among others, Australia and New Zealand).

4.2.2 Global key impacts and costs by commodity groups

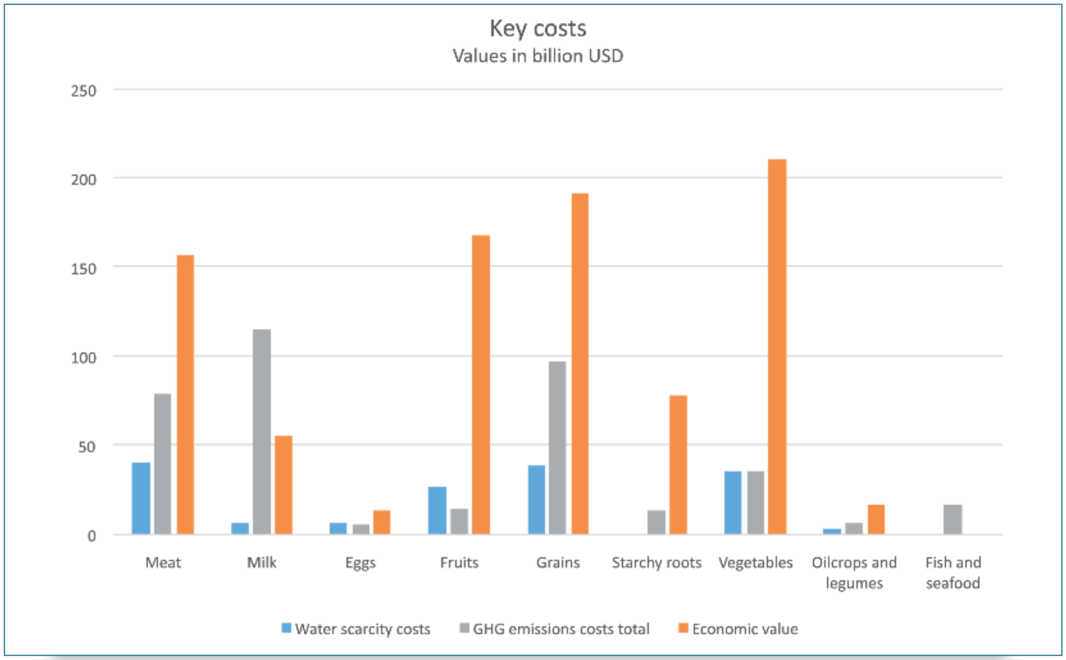
For the regional analysis, physical impacts of food wastage are presented first, followed by costs. Further details are provided in the combined regional and commodity group analysis in sections 4.2.3 – 4.2.8. For different commodity groups, observed land occupation (e.g. high for meat, milk and grains) and greenhouse gas emissions patterns (high for meat, milk and grains) are particularly relevant (as shown in Figure 10) and also largely explain some of the results observed in the combined analysis of regions and commodity groups in sections 4.2.3 -4.2.8. For the commodity group "fish and seafood", most data are lacking and only values for wastage quantities and related GHG emissions and their costs can be displayed.

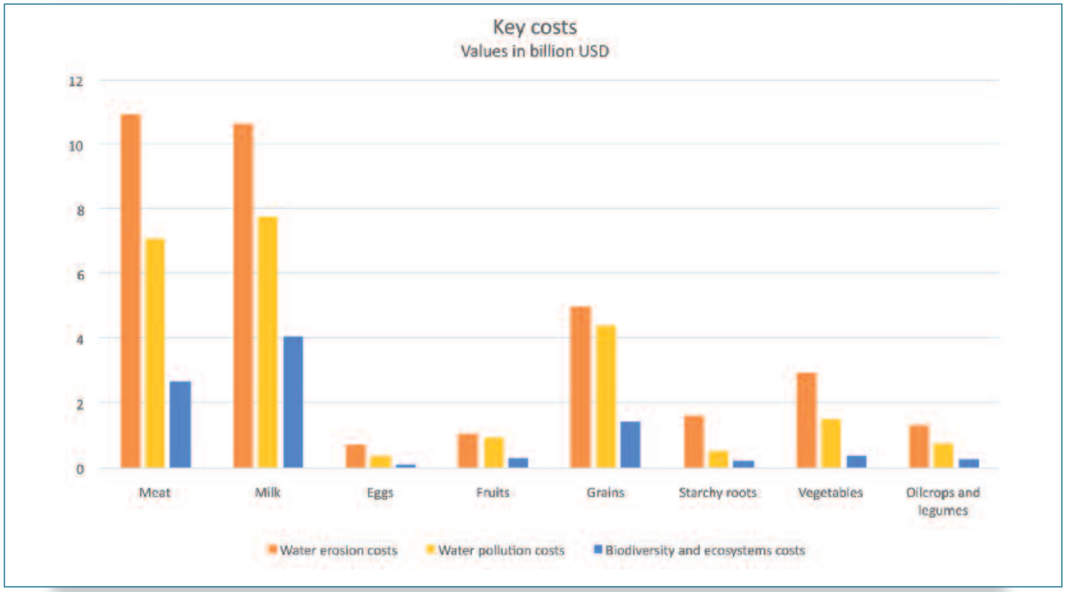
Figure 10: Key global environmental impacts of food wastage by commodity groups



Values in million tonnes wastage, millions ha land occupation, million tonnes GHG emissions, and km^3 water use, all on the same axis.

Figure 11a and 11b: Key global costs of food wastage by commodity groups (billion USD)





Figures 11a and 11b present key costs per commodity group as they correspond to key physical impacts, i.e. to land occupation (water erosion and water pollution) and GHG emissions.

4.2.3 Greenhouse gas emissions costs

This covers the costs of greenhouse gas emissions from food wastage along the whole value chain, including emissions from deforestation and utilized organic soils. Figure 12 illustrates that, corresponding to the emissions levels, meat, milk and grains are the most important categories.

Figure 12: Greenhouse gas emission costs by region and commodity group (billion USD)

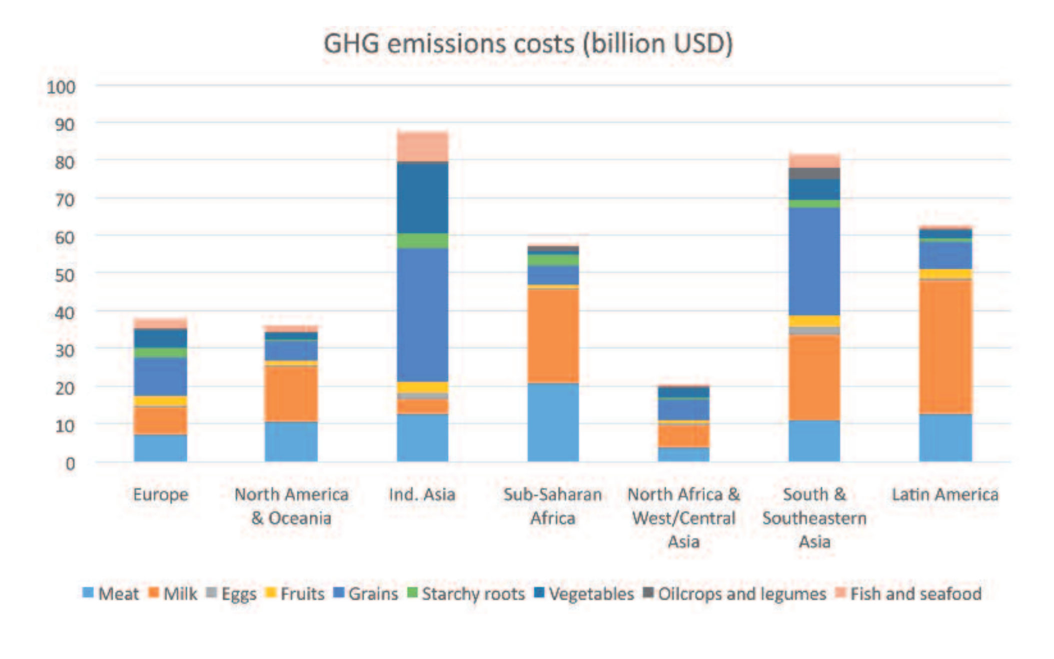
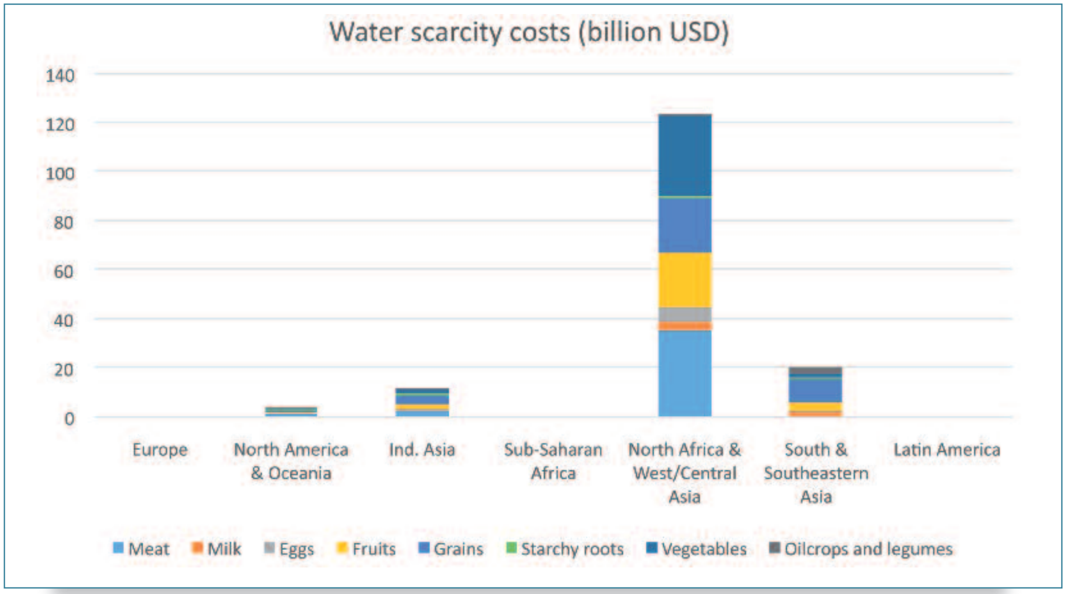
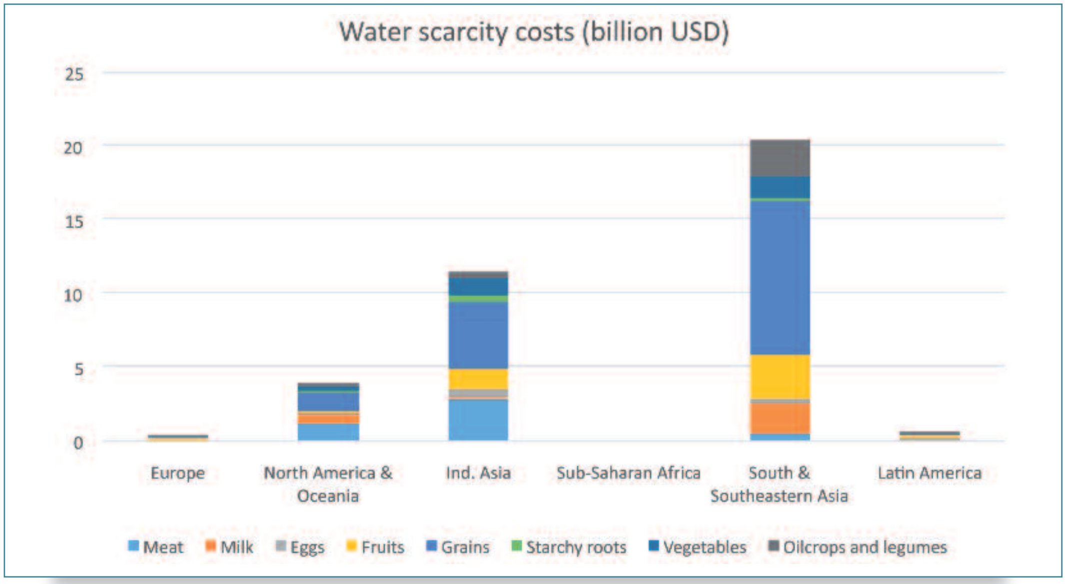


Figure 13a/b: Water scarcity costs per region and commodity group (billion USD)





Detail from the previous Figure 13a not displaying the dominant region "North Africa & West/Central Asia as this allows to provide details on the other regions that are not visible in Figure 13a, where this dominant region is included.

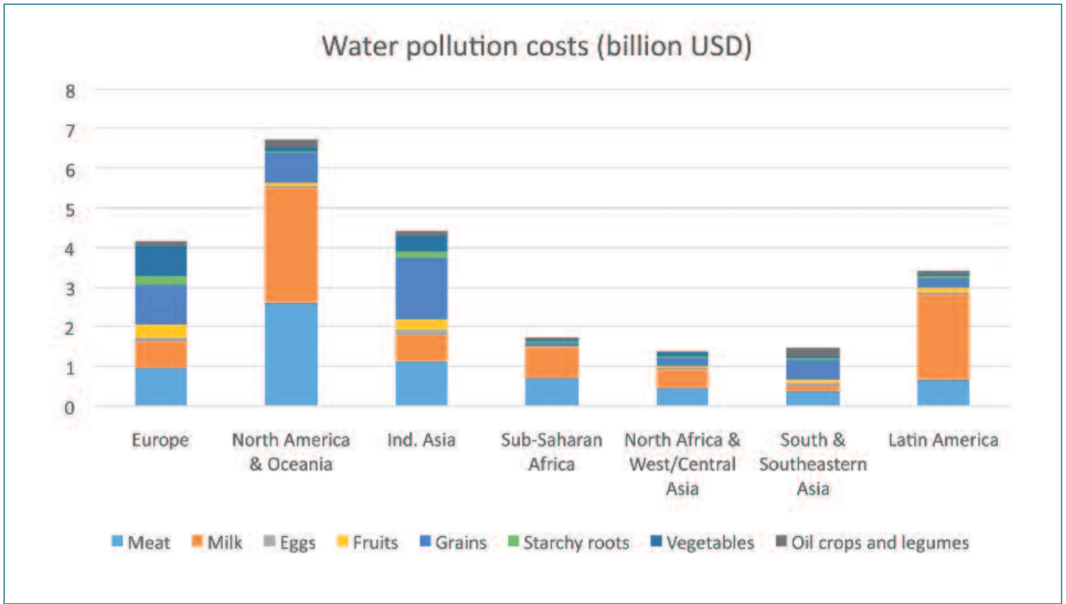
4.2.4 Water scarcity

Water scarcity is most prevalent in North Africa and West and Central Asia, with correspondingly high costs in those regions; these costs are also driven by the amount of irrigated area. However, water scarcity costs due to food wastage appear low in sub-Saharan Africa because: irrigated areas are low in this region, though the highly irrigated areas under grains are reflected in the results; the calculation includes a considerable data shortage on water scarcity values for several countries (Figures 13a and 13b).

4.2.5 Water pollution costs

Figure 14 covers the categories N/P eutrophication, and nitrate and pesticide pollution of drinking water. Due to the data used and the model calculations, these costs strongly relate to land occupation, which is reflected in the results, where meat, milk and grains dominate.

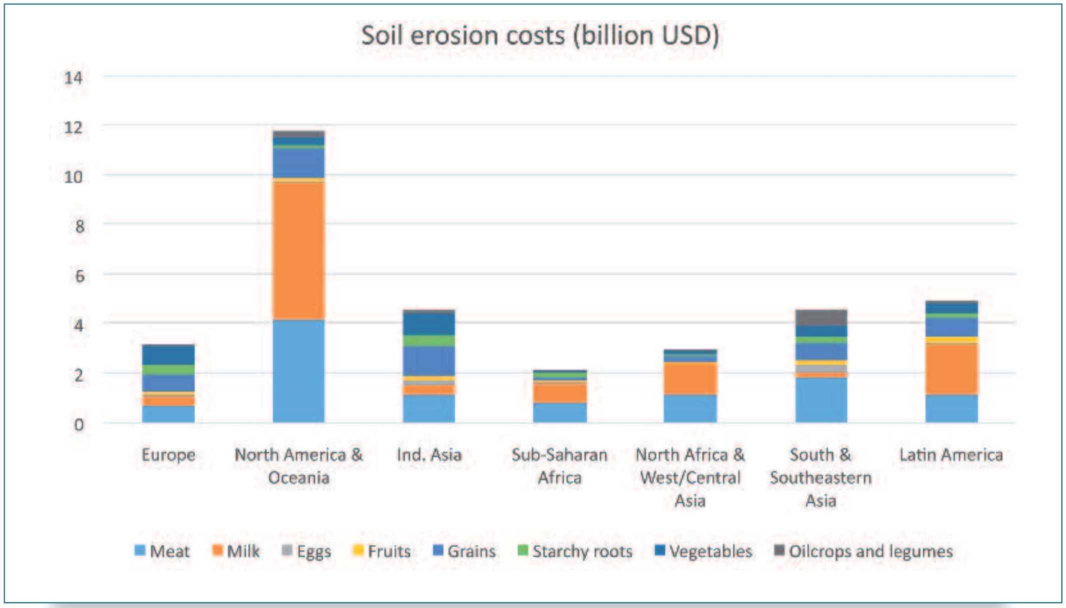
Figure 14: Costs of water pollution differentiated by region and commodity group (billion USD)



4.2.6 Soil erosion costs

Figure 15 covers the costs of soil erosion due to water. It does not address wind erosion because the few values available would enable a global estimate but further regional differentiation would not be possible. These results correlate to land occupation, making milk, meat and grains dominant categories. Total soil erosion levels being high in North America and Oceania (including Australia and New Zealand) are also reflected in this differentiation of corresponding costs.

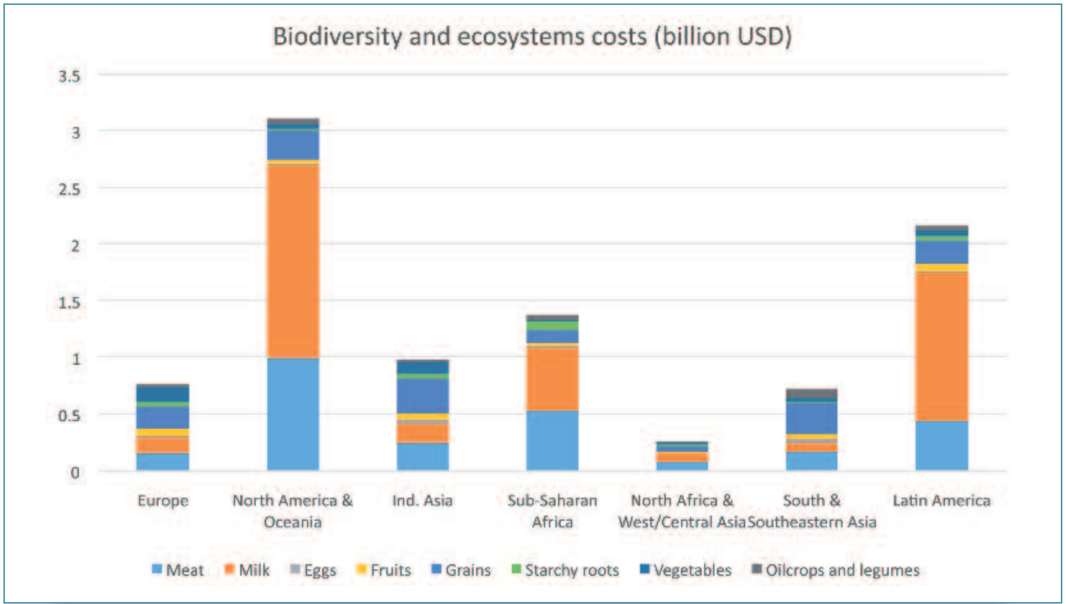
Figure 15: Costs of soil erosion from water (billion USD)



4.2.7 Biodiversity and ecosystems costs

This category covers the impacts of pesticide and N/P on biodiversity plus the costs from deforestation. The latter is added in this aggregate, as it is compiled from the values of a range of ecosystem services. It does not cover the estimates for pollinator loss, as those are available at global level only. Results correlate to land occupation and the P eutrophication impact on biodiversity is the dominant cost component (Figure 16).

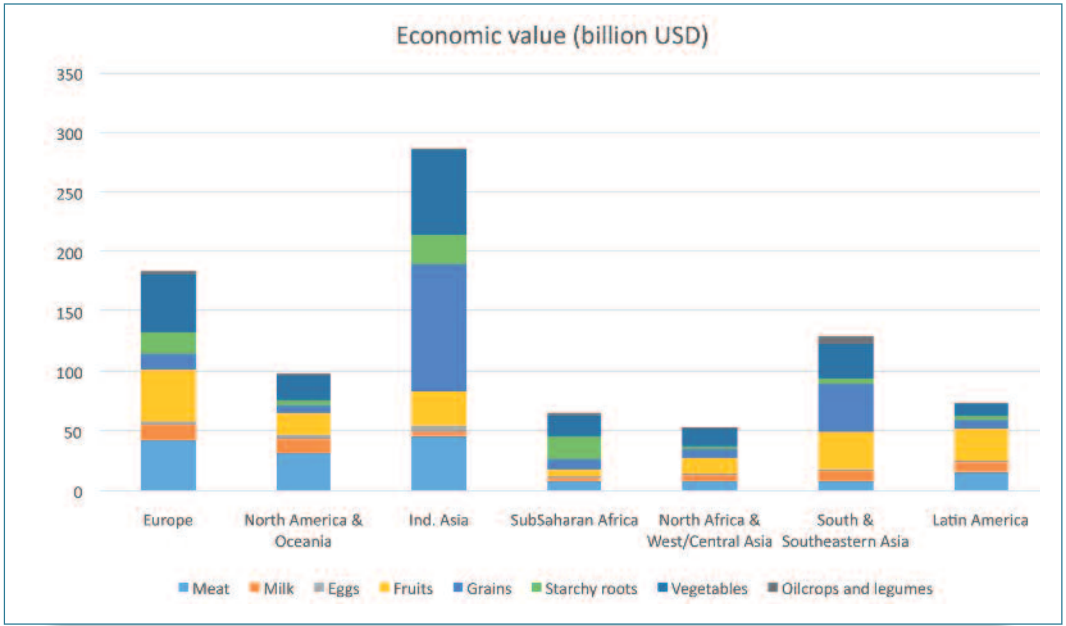
Figure 16: Costs of impacts on biodiversity and costs of ecosystem services lost from deforestation (billion USD)



4.2.8 Economic value

This category looks at economic value lost due to wastage, but does not include lost subsidies, as those are only available for OECD countries (EU-27 on aggregate only), and as there is no possibility to differentiate the subsidies by commodity group (Figure 17).

Figure 17: Economic value lost, differentiated by region and commodity group (billion USD)



5. Areas for Future Research

As this report has shown, existing data are not sufficient for accurate assessments of full costs of food wastage. For example, land values, which are potentially crucial inputs for prioritizing action, are not available. In addition, using data from just one or only a few countries to derive estimates for the other countries via benefit transfer is far from being an exact science. Considering data uncertainties, any cost estimate that will be generated thus provides a gross indication of the size of true full costs only. Nevertheless, the work presented here is a key step towards more encompassing full-cost accounting of food wastage and given the data available and the resources in this project, it provides the most robust estimates possible for the time being.

In addition, hidden costs of food wastage are huge and monetization, with all methodological and data uncertainties, gives a sense of the market distortions due to external costs in the global food system. As a gross summary of results, the societal costs of food wastage estimated here amount to about USD 2.6 trillion, of which USD 700 billion are societal costs of environmental impacts, USD 1 trillion are costs from economic losses of wasted and lost production, and USD 900 billion are costs due to individual well-being losses.

This report not only informs on the extent of food wastage, it raises awareness of societal costs which triple the financial value of the wastage. This knowledge cannot but trigger behaviour change, including mitigation investments (as it informs return on investment). It must also be noted that these are only first approximations of these costs. Future estimates should be able to complete the picture by adding missing aspects, such as the additional hardship on people created by natural resource scarcity (e.g. walking longer distance to fetch water or fuel) or linkages between labour input and food waste. So far, such full equilibrium effects have not been captured and social externalities must be further explored.

This work also defines the five areas listed below as those where research will be needed going forward, in order to have an even more complete full-cost accounting of food wastage.

- **Develop further and refine available data bases.** This means adding more detailed national or regional data, if available, from a more extensive review of the literature, including grey literature such as governmental and NGO reports, including those in national languages. For example, data on the health costs of pesticide use could be collected in this way. Additional national estimates would then allow refining and improving the benefit transfer to arrive at more complete and credible global estimates.

- **Develop the valuation framework in line with the well-being approach for all cost categories to capture societal costs more realistically.** This means moving away from damage cost estimates in order to value and cost outcomes in line with impacts on human welfare (i.e. compensating and equivalent monetary measures). Using revealed preference, stated preference and well-being valuation would rely less on cost estimates based on physical damages and, thus, would improve capture of the costs of food wastage impacts as valued by society as a community of individuals. This will require deeper reviews of the valuation literature and, if possible, primary data collection from affected stakeholders.

- **Assess the value or benefits of food wastage and determine a normative framework for handling these benefits in the cost-benefit analysis.** The CBA does assess costs and benefits, but it focuses on economic assessment and does not address whether or not costs and benefits and their relative relation may be "legitimate" in some ethical sense. This research would require branching out into the philosophical field of normative ethics, which has driven a large number of critiques and developments of the CBA in the past. When relying purely on the normative framework set out in neo-classical economic theory, for example, then some food wastage is normatively permissible and hence, so would be the related benefits. This may contradict the UN and FAO's mission statements regarding zero food waste and may receive fierce criticism from some philosophical approaches that aim at avoiding wastage on moral grounds.

- **Further develop the incorporation of food wastage into equilibrium models.** This will enable improved assessments of costs and benefits of food wastage in the context of all sectors of an economy, including trade. This would need a major data collection effort, in particular on costs of mitigation measures and on price elasticities of food items and agricultural inputs. Part of this information is available, but scattered in many different studies, but for many commodities and inputs such data is lacking.

- **Integrate valuation techniques into geographic information systems.** This will further ensure spatially explicit analysis and, thus, a more site-specific and relevant valuation for water, land, biodiversity and crucial ecosystem services such as global warming potential, erosion regulation, freshwater regulation and water purification. Combining tools is more useful for decision-makers and investors, as the system boundary and administrative jurisdiction can be matched, resulting in spatially and effectively targeted interventions.

The need for research to improve the quality and quantity of data will always exist. However, as we have shown, taking effective action on food wastage is key and the need for more research is no excuse for inaction. In fact, despite the huge data and knowledge gaps, enough impacts have been made evident to justify taking action on mitigating food wastage. Further efforts should focus on specific contexts, at national or supply chain level. The current FCA framework provides the basis for more targeted research.

Finally, assigning a monetary value to the impact of food wastage on the environment and society is key to engaging decision-makers in risk mitigation and securing sustainability of resource use. Moving ahead, it must be noted that benefit transfer may be cost effective and sufficient for global valuation, but it still presents significant data and reliability challenges that can only be avoided with local studies. Although uncertainties are inherent in current valuation estimates and these are not absolute figures, they are fit for relative use and can be used to indicate the huge implications of the problem.

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Compilation of values of soil erosion by water
(in tonnes of soil lost/ha)

Food and Agriculture Organization
of the United Nations

Annex: Values of soil erosion by water (tonnes soil lost/ha)

Country	Min	Agricultural Land/ Cropland Mean	Max	Min	Grassland/ Pasture Mean	Max	Forest	Orchard	Shrubs	Vineyard	Source
Albania	0.78		1.86								Grazhdani 2006
Argentina	0.20	18.80	38.00		0.00						Buck 1993 *in* Pimentel 1993, Lal 1989 (average)
Austria	0.50	8.93	39.00								Darmendrail 2004 (average), Strauss & Klaghofer 2006 (min,max)
Belgium	2.80	8.50	17.60								Darmendrail 2004 (average), Verstraeten *et al.* 2006
Benin	17.00		28.00								Lal 1989
Brazil		18.80									Lal 1989
Bulgaria	0.27	4.76	5.15	0.03	2.69	6.00		12.65		12.65	Rousseva *et al.* 2006
Burkina Faso	5.00		35.00								Pimentel 1993
China	10.00		251.00								Lal 1989 (water), Hoffmann *et al.* 2011 (wind)
Colombia		22.00									Lal 1989
Côte d'Ivoire	60.00		570.00								Lal 1989
Czech Republic	0.00	2.27	13.89								Dostal *et al.* 2006
Denmark	0.26	0.64	12.79		0.03						Darmendrail 2004, Veihe & Hasholt 2006
Ecuador	210.00		564.00								Lal 1989
Ethiopia	8.00		117.70	2.00		29.40					Hellden 1987 *in* Taddese 2001
Finland	0.10		2.35								Tattari & Rekolainen 2006
France		2.03			0.01					11.09	Darmendrail 2004
Germany		1.32			0.14		0.00		0.13	33.23	Darmendrail 2004, Auerswald 2006 (grassland)
Ghana	5.00		10.00								Pimentel 1993
Greece		0.58						0.05	1.17	0.41	Darmendrail 2004
Guatemala	5.00		35.00								Lal 1989
Guinea	17.90		24.50								Lal 1989
India		25.00									Ismail 2008
Indonesia (Java)	50.80		144.30								Magrath 1989
Italy		1.33			0.28		0.20		0.06	54.86	Darmendrail 2004
Jamaica		90.00									Lal 1989
Kenya	25.00		45.00		5.00		1.00		7.50		Cohen 2006
Lesotho		20.00									Bojö 1991 *in* Bojö1996
Lithuania	2.50	19.38	32.20		0.01						Darmendrail 2004 (average), Jankauskas & Fullen 2006 (range)
Malawi		20.00									World Bank 1992 *in* Bojö 1996
Mali		6.50									Bishop & Allen 1989 *in* Bojö 1996
Mexico	10.00		15.00								Margulis 1992
Nepal		40.00									Lal 1989
Netherlands		6.76									Darmendrail 2004
Nicaragua		11.00									Alfsen 1996
Niger	35.00		70.00								Lal 1989 (water), Bielders *et al.* 2000 (wind)
Nigeria		14.40									Lal 1989
Norway	0.20		3.50	0.10		2.60					Oygarden *et al.* 2006
Papua New Guinea	6.00		320.00								Lal 1989
Paraguay		18.80									Lal 1989
Peru		15.00									Lal 1989
Portugal		0.59			0.04				0.40		Darmendrail 2004
Romania	0.70		44.80								Ionita *et al.* 2006
Russian Federation	0.50	4.80	20.00								Sidorchuk *et al.* 2006
Rwanda	35.00		246.00								Berry 2003
Senegal	5.00		30.00								Pimentel 1993
Slovakia		20.00									Stankoviansy *et al.* 2006
Slovenia	2.39		10.94	0.04		1.89		4.77		22.12	Hrvatin *et al.* 2006
South Africa		5.00									McKenzie 1994 *in* Bojö 1996
Spain		0.30			0.84		0.00		0.52		Darmendrail 2004
Switzerland		0.67									Darmendrail 2004 (bare soil), Prasuhn 2004 (cropland)
The former Yugoslav Republic of Macedonia	0.04		4.77								Blinkov & Trendafilov 2006
Turkey		2.42									Demirci 2012
Uganda		5.10					0.10				Isabirye 2005
United Kingdom	0.59	2.09	5.60		0.01						Darmendrail 2004, Boardman & Evans 2006
United Republic of Tanzania	1.10		92.80								Lal 1986 *in* Pimentel 1993, Lal 1989
United States of America		6.68									USDA 2007
Zimbabwe		43.00									Bojö 1996

Source: Schwegler (2014).

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Job number I3992E/1/08.14

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ISBN 978-92-5-108512-7

9 789251 085127

I3991E/1/08.14