CHAPTER 7

TEEBAGRIFOOD METHODOLOGY: AN OVERVIEW OF EVALUATION AND VALUATION METHODS AND TOOLS

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SUMMARY

Chapter 7 presents an overview of available evaluation and valuation methods and tools relevant to the analysis of dependence and impacts of various agricultural and food systems on human wellbeing. The market and non-market valuation tools and methods address to varying degrees the positive and negative externalities along the value chain of eco-agri-food systems. However, challenges emerge from the complexity of the systems, stemming from the temporal and spatial dimensions and management practices and value attribution across multiple ecosystem services. As decision making requires integration of economic values with other social and economic dimensions, the chapter presents an integrated systems approach, which helps in incorporating various dimensions together to evaluate the impact of various policies on the human wellbeing.

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7.0 KEY MESSAGES

- This chapter presents an overview of available evaluation and valuation methods and tools relevant for the analysis of dependence and impacts of various agricultural and food (eco-agri-food) systems on human wellbeing.
- The eco-agri-food system has undergone deep economic and technological transformation. As a result there have been a number of intended and unintended impacts on human well-being. These necessitate a careful evaluation of the associated external effects and the social, economic and environmental impacts.
- Several market and non-market valuation tools and methods can take into account the externalities along the value chain from the farm gate to the food plate of the eco-agri-food system. However, no single tool or model addresses all the needs of the stakeholders and effectively takes account of the complexity of the system analysed.
- Valuation methods can provide credible numbers but to do so they require a lot of data as well as information on the context, purpose and the assumptions behind the values.
- The challenges of valuation of agricultural and food systems arise from their spatial dependence, scale of occurrence of ecosystem services, temporal dimensions, management practices and attribution of values across multiple services.
- The transferability of values from one context to another is possible but requires extensive socio-economic and environmental information about the site where they were estimated and the site where they will be applied.
- Decision making does not depend only on economic values but also included wider dimensions. There are tools that can integrate the economic values into wider dimensions of policy making.
- The external impact of the eco-agri-food value chain is dynamically linked to economic and social impacts through positive and negative feedback loops. Thus the system has to be analysed and integrated as a whole, taking account of these dynamic factors.
- Use of a systems approach can support the integration of knowledge across fields and complement existing work by generating an assessment of the social, economic and environmental impacts of production and consumption, and by estimating strategy/policy impacts for a specific project/policy and for society.
- The scenarios of the systems approach can help simplify and understand the complexity of the eco-agri-food system, and evaluate the short vs. longer-term advantages and disadvantages of the analysed interventions.

CHAPTER 7

TEEBAGRIFOOD METHODOLOGY: AN OVERVIEW OF EVALUATION AND VALUATION METHODS AND TOOLS

7.1 INTRODUCTION

This chapter presents an overview of evaluation and valuation methods and tools to assess the dependence and impacts of agricultural and food (agri-food) production, processing, distribution and consumption activities on supporting ecosystems and their services, and on human wellbeing. These ecosystems are an essential part of the asset base of a country or region, which includes produced, natural, human and social capital, as discussed in the previous chapter.

Whereas Chapter 6 described the TEEB Evaluation Framework and established *what* should be evaluated regarding the social, economic, and environmental elements as well inputs and outputs across the value chain, this chapter explores *how* to carry out the evaluation, making the distinction between (and presenting examples of) methods for the economic *valuation* of ecosystem services and disservices in both monetary and non-monetary terms. It also covers evaluation methods and *modelling* tools and techniques. The distinction between valuation and evaluation is explained in the next section. Evaluation and valuation methods can help in addressing for instance, questions such as:

1. To what extent can food security be improved through agricultural intensification, as opposed to expanding the area devoted to agricultural production, and in both cases, what are the external costs and benefits?

2. Organic farming and low external input agriculture are presented as alternatives to conventional farm management systems, which proponents claim will better protect the health of soils, plants and wildlife. What are the impacts of these practices on society?

3. Food production has multiple environmental impacts and ecological dependencies. What farm management systems and practices can ensure food security while reducing adverse environmental impacts? What are the synergies and trade-offs involved?

The chapter is structured as follows: the rest of this section explores the issues that need to be investigated. We introduce the concept of external costs in the context of agricultural systems. Section 7.2 explains the distinction between *valuing* the impacts of eco-agri-food systems and a wider *evaluation* of the systems as well as policies to make them more effective. Section 7.3 describes the different valuation methods relevant to the sector and discusses their strengths and weaknesses. Section 7.4 does the same for various evaluation methodologies. Section 7.5 discusses how different modelling tools can inform the evaluation process, while Section 7.6 introduces the use of integrated modelling. Finally, Section 7.7 provides a summary and concluding remarks.

7.1.1 Key issues and factors in the selection of evaluation and valuation methods and criteria

Complexities in agriculture and food systems and the feedback with ecosystem services

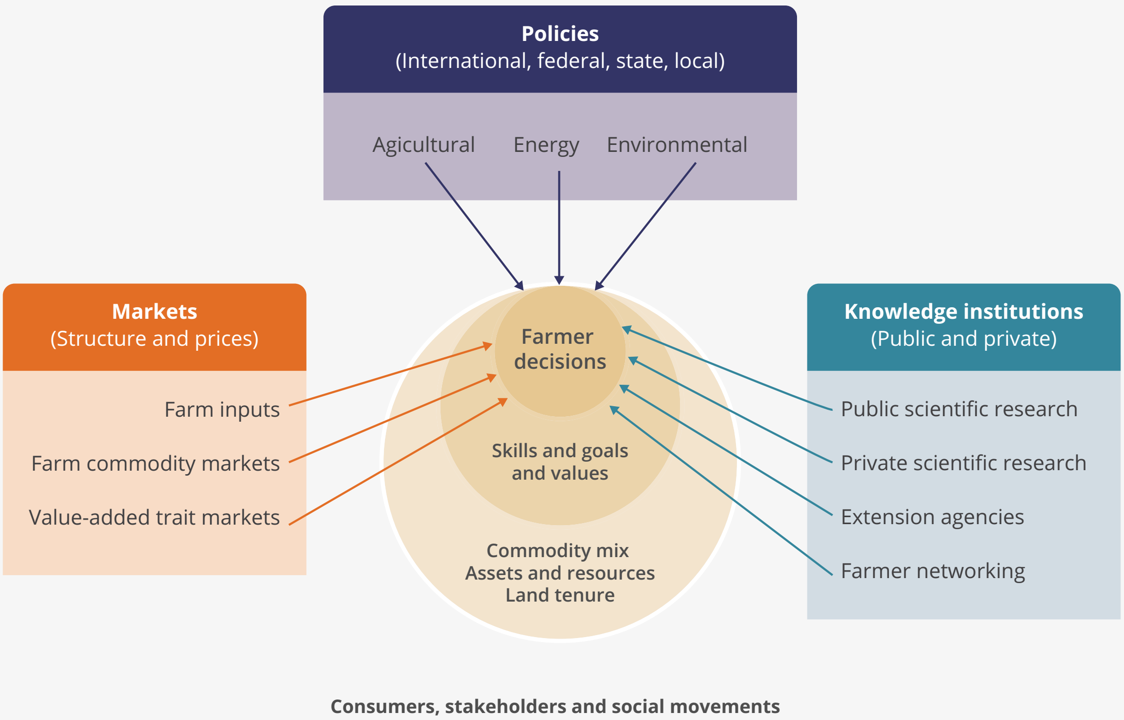
Agricultural systems, though managed to provide food, fibre and fuel, are unique in receiving and providing ecosystem services as well as generating disservices to other ecosystems (Swinton *et al.* 2007). Producers rely on ecosystem service inputs, which they combine with land, seeds, labour and technology to produce a range of valuable products, along with other ecosystem services and disservices, which vary in their effects on human well-being. For example, the quality of soil including the quantity of soil carbon is one of the key inputs necessary to generate a good yield but it is impacted by soil tillage, crop rotation practices, the level of organic inputs and erosion. The services from these ecosystems can also be seen as a return to the stock of natural capital. Changes in the expected flow of services arising from non-sustainable use, for example, will be reflected in a decline in the value of natural capital, which can act as a guide to the dangers of some eco-agri-food practices.

According to OECD (2000), the following risks are common to the agriculture sector: production risks (weather conditions, pests, diseases and technological change), ecological risks (climate change, management of natural resources such as water), market risks (output and input price variability, relationships with the food chain with respect to quality, new products) and regulatory or institutional risks (agricultural policies, food safety and environmental regulations).

Farms are managed ecosystems and their final output depends on the choices that the farmer or farm manager takes, and are linked to the farm's external environment, which depends on a range of natural, technological, social, economic and political factors (see **Figure 7.1**). Farm output not only depends on a farmer's own decisions but also on the actions of other farmers and consumers, policy-makers, general conditions of trade, etc. For example, if a farmer decides to plant eucalyptus trees on her land to sequester carbon for the offset market, this might lower the water table more widely. If a farm suffers from a sudden infestation of pests, a neighbouring farm is also at risk. The introduction of alien species or invasive plants can have detrimental effects on some native pollinators but in certain cases may support other native pollinators.

Decisions made by farmers, like those involving crop diversity, fertilizer and pesticide use etc., impact on the environmental quality of their land and beyond (Tilman 2002). These impacts from the agricultural production systems are transmitted by biological, chemical or physical processes and the external costs (and benefits) are not reflected in the price of goods in this sector. Usually the impacts are borne (or enjoyed) by society more widely and by people who may not be actually producing these impacts, which raises both efficiency and equity concerns. Pretty *et al.* (2000) describe five features of externalities from agriculture: i) markets neglect many external costs and benefits; ii) they often occur with a time lag; iii) they affect groups whose interests are not always represented in decisions; iv) the identity of the producer of the externality is often not known; and v) externalities can result in suboptimal economic and policy outcomes, including more output and higher levels of pollution (the efficiency concern). In many countries, farming has evolved to a state where it is often in conflict with environmental protection. The costs of agricultural externalities can be substantial, as shown by estimates made for Germany (Waibel and Fleischer 1998), Netherlands (Bos *et al.* 2013), UK (Pretty *et al.* 2000; 2005) and the USA (Tegtmeier and Duffy 2004). For losses of ecosystem services due to modernization of agriculture in Sweden, see Björklund *et al.* (1999). A more detailed breakdown of the external costs in the UK from Pretty *et al.* (2000) is given in Section 7.4, where methods of valuation are discussed.

Figure 7.1 Drivers and constraints that affect farmers' decisions (Source: adapted from Reganold 2011)



7.2 THE NEED FOR VALUATION AND EVALUATION OF ECO-AGRI-FOOD SYSTEMS

As mentioned above, many of the ecosystem service dependencies and impacts of the eco-agri-food system are not fully captured in markets. Economic valuation tools can be helpful to quantify dependencies and impacts in monetary terms and make them more comparable to other things we value.

However, valuation alone cannot provide a complete picture; we need additional evaluation techniques to understand the relative merits of different actions, strategies, and policies. Different policies (e.g. subsidies or taxes, agricultural policies), resource allocations (e.g. how much water to use for irrigation) and production decisions (e.g. what type of crop rotation to implement) made by different stakeholders (farmers, policy makers, consumers) involve trade-offs for the economy, the environment and various stakeholders. Economic valuation methods can provide the data needed to evaluate such trade-offs. Evaluation techniques are then used to understand whether the benefits are worth the costs not only to society as a whole but also to groups of producers and consumers, while also assessing the wider social (particularly distributional), economic and environmental impacts of decisions.

Agriculture depends on ecosystem services as inputs as well as providing many ecosystem services (see **Table 7.1**). Food produced by farmers goes through stages, from land clearance and preparation, to planting, growing, harvesting, preparing products for the consumer market, consumption and final disposal of any wastes. At each stage, a number of economic impacts are generated, in the form of incomes to producers, wages to employees, tax revenues to the government or subsidies from the government, possible imports of inputs and exports of outputs and so on. Some of these impacts are captured through market transactions or flows of financial resources from one agent in society to another, while several other intended (positive) and unintended (negative) impacts on the economy and well-being are not captured. Some modern industrial food systems also pose health hazards for consumers, which are not appropriately valued.

For example, modern farming practices have improved livestock feed efficiency through the use of antibiotics. Less time is needed to bring animals to slaughter, reducing costs to the producer, improving profits and decreasing consumer costs. Similarly, antimicrobial products have improved prevention, control and treatment of infectious diseases in animals. Van Lunen (2003) reports that in the U.S., 52 per cent of total antimicrobials were used for the treatment of infectious diseases in animals, and 25-70 per cent of cattle received the drugs through feed. However, both of these technologies can pose significant health risks to humans and some countries have banned the use of antimicrobials for livestock production (Barug *et al.* 2006). These hazards were discussed in greater depth in earlier chapters.

It is important to consider eco-agri-food systems as a whole if effective strategies to internalize the externalities from ec0-agrif00d systems are to be designed and implemented. In much of the literature, each stage of the value chain is analysed separately. Partial exceptions include the work of Pretty *et al.* (2005; 2015), some life cycle assessments (Shonfield and Dumelin 2005, discussed in Section 7.5.2), and the propensity scoring method (Setboonsarng and Markandya 2015, discussed in Section 7.5.4).

There are positive and negative feedback loops across the whole value chain of eco-agri-food processes (FAO 2014). Changes have both backward and forward linkages with economic, environmental and social outcomes in other stages of the value chain. For example, a change in consumer preferences for organic food can affect the earlier food production and processing stages and create environmental and social consequences. Likewise, an increase in crop yields will have social and environmental impacts at the production stage as well as on levels of profits, prices, nutrition and consumption. Changes outside the eco-agri-food sector, such as an increase in the demand for biofuels, for example, may raise the price of land and increase crop prices. This in turn will have impacts on poverty and malnutrition at the production and consumption stages (IFPRI 2008; Gerasimchuk *et al.* 2012).

Some of the health hazards of eco-agri-food systems do not qualify as conventional externalities, particularly in the consumption stages of the process, such as over-consumption of products high in sugar and fats: consumers pay for the products and make a conscious decision to consume them without being obliged to do so. Nevertheless, such consumption is a social concern because of harmful effects on consumers, which impact publicly funded health services (Green *et al.* 2014). The term used to refer to such goods or activities is *demerit goods* or activities[1]. A demerit good is defined as a good which can have a negative impact on the consumer and society, but these damaging effects may be unknown or ignored by the consumer. There is a debate as to how much the government should control the availability of harmful products and what form such interventions should take. The opposite of a demerit good or service is a merit good or service – one whose consumption has wider social benefits (e.g. vaccinations, education). The notion of merit and demerit goods thus extends the concept of externalities.

1 For a definition of merit goods and demerit goods see Musgrave, 1987. Strictly speaking demerit goods are not externalities in the sense that their consumption harms a third party (e.g. if I smoke in my home with no one else around I am not generating an externality in the conventional sense, but I am consuming a demerit good insofar as overall social welfare is diminished by such consumption).

Table 7.1 Classification of ecosystem services from agriculture (Source: EEA 2018)

<table>
<tr><th>Section</th><th>Division</th><th>Group</th><th>Class</th></tr>
<tr><td rowspan="6">Provisioning</td><td rowspan="2">Nutrition</td><td>Biomass</td><td>Cultivated crops
Reared animals and their outputs
Wild plants, algae and their outputs
Wild animals and their outputs
Plants and algae from in-situ aquaculture
Animals from in-situ aquaculture</td></tr>
<tr><td>Water</td><td>Surface water for drinking
Groundwater for drinking</td></tr>
<tr><td rowspan="2">Materials</td><td>Biomass</td><td>Fibre and other materials from plants
Plants, algae, animal materials for agriculture
Genetic materials from all biota</td></tr>
<tr><td>Water</td><td>Surface water for non-drinking purposes
Groundwater for non-drinking purposes</td></tr>
<tr><td rowspan="2">Energy</td><td>Biomass based</td><td>Plant-based resources
Animal-based resources</td></tr>
<tr><td>Mechanical based</td><td>Animal-based energy</td></tr>
<tr><td rowspan="10">Regulation and Maintenance</td><td rowspan="2">Mediation of waste, toxics and other nuisances</td><td>By biota</td><td>Bioremediation by micro-organisms etc.
Filtration/sequestration/storage/ accumulation by micro-organisms etc.</td></tr>
<tr><td>By ecosystems</td><td>Filtration/sequestration/storage/accumulation
Dilution by atmosphere, freshwater, marine ecosystems
Mediation of smell, noise, visual impacts</td></tr>
<tr><td rowspan="3">Mediation of flows</td><td>Mass flows</td><td>Stabilisation & control of erosion rates
Buffering & attenuation of mass flows</td></tr>
<tr><td>Liquid flows</td><td>Hydrological cycle & water flow maintenance
Flood protection</td></tr>
<tr><td>Air flows</td><td>Storm protection, ventilation and transpiration</td></tr>
<tr><td rowspan="5">Maintenance of physical, chemical, biological conditions</td><td>Habitat and gene pool protection</td><td>Pollination & seed dispersal
Maintaining nursery populations & habitats</td></tr>
<tr><td>Pest & disease control</td><td>Pest control
Disease control</td></tr>
<tr><td>Soil formation & Composition</td><td>Weathering processes
Decomposition and fixing processes</td></tr>
<tr><td>Water conditions</td><td>Chemical condition of fresh & salt waters</td></tr>
<tr><td>Atmosphere & Climate regulation</td><td>Global climate regulation by reducing GHGs
Micro & region climate regulation</td></tr>
<tr><td rowspan="4">Cultural</td><td rowspan="2">Physical & intellectual interactions with biota/ ecosystems</td><td>Physical & experiential</td><td>Experiential use of plants, animal landscapes
Physical use of land/ seascapes in different ways</td></tr>
<tr><td>Intellectual & representative interactions</td><td>Scientific, educational, heritage/cultural, entertainment and aesthetic interactions</td></tr>
<tr><td rowspan="2">Spiritual, symbolic interactions with biota/ ecosystems</td><td>Spiritual and/ or emblematic</td><td>Symbolic
Sacred and/or religious</td></tr>
<tr><td>Other cultural</td><td>Existence
Bequest</td></tr>
</table>

A comprehensive assessment of agricultural and food system complexes taking into account all externalities from farm gate to the food plate, as well as impacts that are not strictly speaking externalities but constitute effects of social concern, requires market and non-market valuation of the dependencies, services and disservices provided by agriculture and food systems. Without valuation, we cannot understand the net benefits or net costs of an intervention. For example, a decision to ban neonicotinoid pesticides in the EU could lead to decline in agricultural yield, but is this good or bad (see Goulson [2013] for evaluation of this case study)? It may be good for insects and the pollination services (a public good/ public benefit) they provide to farming (not to mention their role in ecological health) but bad for yield and thus private profits (private costs). The question arises, is this ban worth the cost? Valuation tools allow for assessment of the impacts of a ban on production (negative) and the contribution to pollination (positive).

Section 7.4 reviews various methods and models that have been used to evaluate the agri-food system. No one model can address all the needs of different stakeholders and effectively account for the full complexity of the system, but using a systems analysis approach can support the integration of knowledge from across disciplines and shed light on the diverse social, economic and environmental impacts of production and consumption. In Section 7.5, Systems Dynamic modelling is presented as a methodology that allows analysts to identify and anticipate the emergence of potential side effects, leading to the formulation of complementary policy interventions for improved resilience and sustainability. First, however, we review the various valuation methods available to assess the eco-agri-food system.

7.3 PRACTICAL METHODS FOR THE ECONOMIC VALUATION OF ECOSYSTEM SERVICES, DISSERVICES AND DEPENDENCIES IN ECO-AGRI-FOOD SYSTEMS

7.3.1 Economic Valuation

Farmers' dependencies on ecosystem services, their provisioning of ecosystem services and the impacts of agricultural practices on the wellbeing of people both on and off-farm follow several pathways. Some of these dependencies, outputs and impacts involve market transactions and can be quantified and valued in money terms while other dependencies do not involve such transactions and need different methods of valuation. This section reviews methods for valuing these non-market impacts and dependencies of the eco-agri-food system.

As noted, many ecosystem services are intangible and their role can only be inferred. For example, the nutrient cycling service of soil microbes cannot be directly experienced but food producers, through their experience, know that certain practices lead to better nutrient exchange and enhanced crop output. Similarly, some ecosystem services are more local in nature while others are global. For example, nutrient cycling is experienced only on farm, while aesthetic values are often regional, and carbon regulation is a global service. Ecosystem services most relevant to farmers, local communities and society at large may differ (Swinton *et al.* 2015). A key feature of many ecosystem services or disservices is that consumers/producers need not pay to benefit from the service, nor can they necessarily be excluded from consuming the output (e.g. if at a reasonable distance a farmer manages beehives for pollination, other farmers cannot be excluded from consuming the service provided by travelling pollinators).

The fundamental basis for valuing any goods and services – marketed or non-marketed – is the individual willingness to pay for them. The techniques discussed in this section utilize that base concept, although some methods may depart from the ideal due to lack of data[2].

Many studies have been undertaken to value the flow of services from ecosystems[3], much of which was summarized in TEEB (2010).

The methods used to elicit estimates of ecosystem services cover the whole range of valuation techniques used in environmental economics. **Table 7.2** summarizes different techniques used in a comprehensive review of valuation studies by de Groot *et al.* (2012). One main method used is direct market valuation, notably direct market pricing. Direct market valuation methods include market pricing, market based payments for environmental services, factor income/production function methods and the cost based approaches. Where data from actual markets are available, direct market, valuation approaches are preferred. They are most often deployed for valuing

2 One example of an approach that deviates from willingness to pay is surveys of happiness, which seek to measure wellbeing using a subjective happiness scale. This approach has been used in recent years to track progress in a number of areas, but there are few cases relating to ecosystem services. For a recent example of the happiness approach see Tsurumi and Managi (2017).

3 See www.es-partnership.org for access to a wide range of databases linking to such studies, as well as the Environmental Valuation Reference Inventory (EVRI 1997), Cost of Policy Inaction Valuation Database (Braat *et al.* 2008), ENValue (2004)ValueBaseSwe (Sundberg and Söderqvist 2004), and work done by de Groot *et al.* (2012), McVittie and Hussain (2013) and Costanza *et al.* (2014).

provisioning services but are also frequently used for habitat services and cultural services. Cost-based valuation methods include: avoided cost, restoration cost, and replacement cost[4]. They are most often used to value regulating services (water regulation, erosion control, air quality regulation, human disease regulation). However, only a sub-set of ecosystem services can be valued using direct market valuation methods.

However, in several cases, direct market data is not easily available or markets do not exist. In such cases, the revealed preference or stated preference methods are used. The revealed preference methods consist of hedonic pricing and travel cost methods where individuals reveal their preference through their observed behaviour in the surrogate markets (e.g. through travel costs to visit agricultural landscapes, paying a premium price for buying a property with good views etc.); these are used mainly for valuing cultural services (recreational or amenity values). Finally, stated preference methods consist of contingent valuation, conjoint choice and group valuation and uses hypothetical (or simulated markets) to elicit values through willingness to pay to obtain the ecosystem service or willingness to accept as compensation for losing access to an ecosystem service.

The approach is typically used for valuing habitat and cultural services (Pearce *et al.* 2006). Stated preference techniques are the only way to value some ecosystem services (like biodiversity) when the ecosystem services cannot be valued through markets or surrogate markets.
The categories given in **Table 7.2** cover a wide range of services with different methods of elicitation of values. Some might question whether the services valued using stated preferences or indirect valuation methods of revealed preferences are as "real" (i.e. since they are not based on actual transactions, do they represent the true underlying preferences of the respondents) as those obtained using market methods. Evidence shows that non-market methods for valuation, when used with care and following the best available techniques, do provide credible numbers that can be compared to those obtained from market transactions.

When choosing the economic valuation technique appropriate to a given application, the following considerations should be noted:

i) There is spatial variation in the ecosystem services provided by (or to) agriculture, which depend not only on farm management practices but also on the landscape attributes (e.g. agricultural land next to a tropical forest is different from farm land adjacent to grasslands). The valuation of agricultural and food systems is challenging due to this spatial dependence.

ii) The level of ecosystem services/disservices provided by (or to) agriculture is also dependent on the management practices adopted by producers, which in turn depend on prices of other inputs. Thus, it is difficult to generalize or transfer values from one site to another without complete information.

iii) The scale at which particular changes in ecosystem services occur is very important. While changes in soil carbon affect farm output and occur at the level of farm and have implications for profitability for the farmer, soil erosion can also have impacts downstream and affect people more broadly. Thus the value of a particular ecosystem service to the farm and to society need not be the same.

iv) There is a temporal dimension as well, owing to time lags in both provision of ecosystem services as well their impacts.

v) There is a risk of double counting. For example, grassland diversity improves crop yield due to increased abundance of insect pollinators (leading to increased food production). In this case the grassland diversity results in improved pollination services leading to higher crop yields. Here pollination is an intermediate service. Thus ecosystem services from grasslands and ecosystem services from agriculture cannot be added separately. Not all categories of regulating benefits, however, constitute double counting. Care is needed when assembling total values and it should be noted that the total figures may contain some double counting).

7.3.2 Direct market value approaches (primary market based approaches)

Market value approaches to measuring agricultural output rely on the value of ecosystem services that are directly sold in markets. For example, the provisioning services from agriculture, such as food, fuel and fibre, can be relatively easily quantified based on market prices (although price distortions arise due to uncompetitive markets or taxes or subsidies). The benefit from any project (say soil conservation through terracing), if it results in increased yield, can be measured in terms of the increase in consumer surplus or producer surplus realized through the output sold in the market[5].

4 Replacement cost is not a desirable standalone method of valuation as it is not necessarily based on the willingness to pay for the service. In many instances it is used as a first approximation and so has been included here.

5 The consumer surplus is the difference between what a person is willing to pay for something and what she actually pays. The producer surplus is the difference between the revenue a producer receives and the cost of producing the good or service.

Table 7.2 Methods used to value ecosystem services (per cent (%) of studies that use different values for a given ecosystem service) (Source: adapted from de Groot *et al.* 2012)

Ecosystem Services	Direct Market Values	Cost Based Methods	Revealed Preference	Stated Preference
Provisioning	84%	8%	0%	3%
Regulating	18%	66%	0%	5%
Habitat	32%	6%	0%	47%
Cultural	39%	0%	19%	36%

Note percentages sum from left-to-right. Where they do not sum to 100 per cent methods were not stated clearly

Thus the value of soil conservation can be estimated in terms of the reduced costs of production (e.g. reduction in fertilizer costs). Some of the methods of ecosystem service valuation that fall under direct market value approaches include measurements of Production Functions and Dose Response Functions, analysis of Averting or Defensive Expenditure, Residual Imputation methods, and various cost-based techniques (Replacement/Restoration/Cost Savings). The rest of this section describes each of these approaches in turn, including their uses and limitations.In the section below, the different methods of valuation are described further and their potential application to eco-agri-food systems is discussed.

Production function

Measuring the value of an ecosystem service involves measuring several independent inputs, which are combined and transformed to produce a single commodity or multiple agricultural commodities. As several of these inputs are biophysical and do not have market values, a way to estimate the value of these inputs is to use the production function method. The production function is, by definition, the technical relationship between outputs and technically feasible inputs. The farmer combines a range of inputs including land, labour, seeds, capital, soil, technology, fertilizers, pesticides, water, pollination services and other environmental variables to produce output. Different combinations of inputs are possible to produce a given level of output (some are fixed inputs and others are variable). Some of these inputs are complementary and some can be substituted (consider fertilizer and soil quality: if soil is of good quality, one can use less fertilizer). The production function gives us the maximum attainable output from a given combination of inputs under efficient management. Inefficient management reduces output from what is technologically possible.

The first step in estimating a specific production function for the inputs and outputs associated with a farm or set of farms for example involves the choice of relevant inputs such as labour, capital, purchased inputs (fertilizers, pesticides), environment inputs (quality of soil, water, climate etc.), management practices, and socio-economic factors that represent the farmer's knowledge, ability and attitude towards producing output. For inputs that are substitutable, several combinations might give the same level of output. Substitutability depends on elasticity, which is estimated from the parameters in the production function. The second step involves choosing the algebraic form of the production function linking inputs to outputs. The appropriate production function chosen depends on the nature of inputs, their substitutability and their relation to output[6]. The third step involves choosing an appropriate econometric technique for estimating the coefficients of the production function that quantify for example, the relationship between each input and the output. The production function gives the relative contribution of each input to the output. Any changes in the inputs leads to changes in crop yields, and maintaining the output at a constant level requires corresponding changes in the quality of input as well.

This approach is very useful in understanding the value of agricultural resource investments (or of their absence), the economic impact of land degradation (soil erosion, for instance) or measuring the value of conservation practices (terracing) etc. See **Box 7.1** for an illustration of how the production function can be applied.

6 Commonly used production functions are the Cobb-Douglas production function, linear production function, Fixed-proportion production function, Constant Elasticity of Substitution (CES) production function. In the linear-production function, the inputs are perfect substitutes. In fixed-proportion production function, the inputs must be combined in a constant ratio to one another (the inputs are complements). The Cobb-Douglas is intermediate between linear and fixed proportion production function (assumes unitary elasticity of substitution) and is most commonly used. The linear production function, fixed proportions production function and Cobb-Douglas are special cases of CES production function.

Box 7.1 Production function analysis of soil properties and soil conservation investments in tropical agriculture

Biophysical and socio-economic factors jointly contribute to agricultural productivity. Including these factors together is very important. The production function approach has the ability to combine these two factors together in a single equation. In an example, soil is a key asset in agricultural production and soil erosion significantly depreciates the soil capital and reduces crop yields along with increasing societal costs. Ekbom and Sterner (2008) examined the role of soil quality and soil investments along with other inputs on crop yield in Kenya using production function approach. Here the farmer is assumed to produce a given output by a specific choice of traditional economic factors – labour, fertilizers, manure and agricultural land, other variables – soil conservation investments, access to public infrastructure and tree capital, and soil capital – represented by the soil properties; these factors are in turn dependent on others like household characteristics (e.g. number of members of the household), soil investments, crops planted and their mix and extension activities provided to the farmers which affect quality. The responsiveness of output to change in various inputs is captured through elasticities. The study showed that soil quality along with soil quality improvements has a positive and significant role on output (elasticity = 0.20) with nitrogen (elasticity = 0.27) and potassium (elasticity = 0.35) increasing the output significantly while high levels of phosphorous (elasticity = -0.22) are actually detrimental to output, thus drawing attention to the need for adapting fertilizer policies to local biophysical conditions. Investments in soil capital have an important role in agricultural output, and thus measures to arrest soil erosion can help farmers increase food production and reduce food insecurity.

Another application of the production function approach study was used by ELD Initiative and UNEP (2015), where they applied a two stage production function approach. In the first stage, it developed econometric model for estimating soil nutrient depletion as a function of biophysical and socioeconomic drivers. In the second stage, it estimated aggregate cereal crop yield as a function of soil nutrient depletion (as proxy of erosion induced land degradation, which is a predicted result from the first stage equation), fertilizer, land, and labour and controlling for unobserved factor. The study also further applied Cost Benefit Analysis as an evaluation tool.

Limitations

The production function method is data intensive and requires observations over a period of time and across farms to get a clearer understanding of the changes in various inputs on output. As some of the investments can impact output with a time lag, use of observations over time and space can better capture these impacts but lack of such data is often a limiting factor. Environmental variables are not easily measurable – thus limiting the use of such variables to one or two. Often several factors that contribute to the output are not considered as they are not easily measured, resulting in biased estimation.

Dose Response Function

The dose response method is similar to the production function approach and investigates the impact of the changes in environmental quality on the desired output (productivity, health etc.). For example, clear dose-response relations can be established in case of pesticide use and disappearance of the house sparrow, pesticide use and farmer's health, water quality improvements and increase in commercial fisheries catch etc. Here the dependent variable is the outcome (agricultural productivity, health etc.) and the independent variables are the exposure variables (levels of various ecosystem inputs, environmental quality input etc.). The method can be quite data intensive.

One common application of dose-response function analysis is the impact of air quality (ozone, global warming) on agricultural production. Dose-response function approaches require the relationship between input (dose) responsible for damage (response) to be well identified along with other variables that influence the relationship. Once the physical relationship between the dose and response are established, monetary values are derived by multiplying the change in output (or the change in a physical indicator of damage) with the price or value of the output or the object that is damaged. Again, note that the prices here should be efficient prices (i.e. prices generated by free markets in the absence of market power or discrimination or other interventions). The method is very useful in obtaining the marginal values (the impact of addition dose).

The approach can give reasonable approximation of the economic value of the resource. The main limitation of dose-response functions is that they require explicit modelling of the relationship between the input changes and the output, which is possible but data intensive. Additional complications can arise in case of interactions between several inputs. For example, the impact of consuming sugary food on health depends on individual genetic make-up, life style etc. Shea (2003) argues that children are at high risk of developing infections with drug resistant organisms linked directly to the agricultural use of anti-microbials. In such cases it may be too complicated to establish such a direct causal relationship. The dose-response technique can be further complicated if in response to the reduction or loss in ecosystem service, consumers and producers change their behavioural response, thereby impacting the producer and consumer surplus. Dose-response functions, if correctly estimated, are theoretically rigorous and thus very useful. They are best applied when external factors such as prices of inputs and outputs are not changed by the measures (see **Box 7.2** and **Box 7.3** for examples).

Averting expenditures / Defensive expenditures

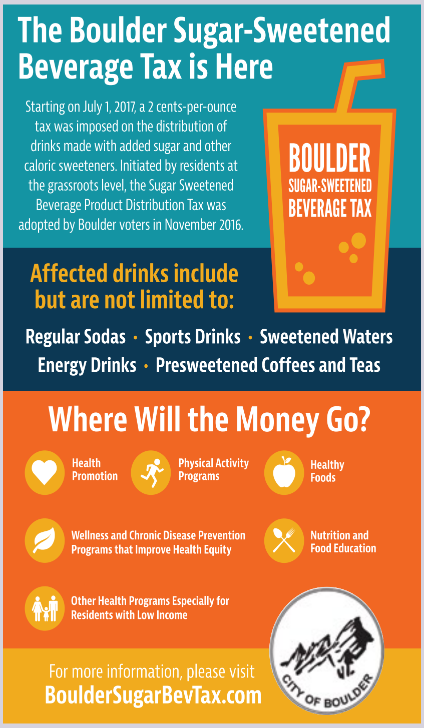
Agents (individuals, firms or governments), exposed to a degradation in quality of an environmental factor, incur defensive expenditures or avert costs in order to avoid a poor outcome (e.g. loss in productivity, poor health, deposition of silt). All the expenses incurred as a result of this averting behaviour - direct expenses for self-protection (e.g. masks for spraying pesticides, pills to prevent malaria) and indirect costs (including the time costs or the leisure foregone) are considered as averting expenditures.

One example of such expenditures is the cost incurred by individuals, firms, and governments to shift from contaminated drinking water (polluted due to agricultural pollution) to safe sources. Users make a decision on which averting actions to take. Choices available in this case can be purchasing bottled water, installing a water filtration system at home, shifting to uncontaminated source (in case where such a choice is available) and boiling water. For example, Harrington *et al.* (1987) assessed the economic losses of water borne disease outbreak in United States. Each of these cases requires households to change their behaviour and incur out-of-pocket expenditures, which would have been otherwise not necessary in case of non-deterioration of environmental quality.

Box 7.2 Sugar – Not so sweet?

Taxes on sugar-sweetened beverages (SSBs) are being levied (in Colorado, US, for example, as illustrated in **Figure 7.2**) and proposed in several countries and cities, due to the association of SSBs with poor health and obesity. Unhealthy diets and high body mass index are key risk factors that contribute to the burden of disease; implementation of SSB taxes are thought to help address this issue. An SSB is defined as a non-alcoholic drink with added sugar, including carbonated soft drinks and flavoured mineral waters. Fruit juices and drinks, energy drinks, milk-based drinks, and cordials are generally excluded.

Figure 7.2 Poster of Sugar-Sweetened Beverage Tax in Boulder, Colorado, US (Source: bouldercolorado.gov)



In Australia, Veerman *et al.* (2016) using epidemiological modelling, found that imposition of a 20 per cent ad valorem tax, assumed to apply in addition to the existing Goods and Services Tax (GST), would result in a decrease in demand for SSBs (i.e. the 'dose'), thereby the Bo and thus the average Body Mass Index (BMI). The study modelled the impact of the tax on nine obesity related diseases and found the proposed 20 per cent tax was estimated to lower the incidence of Type II diabetes by approximately 800 cases per year. The estimated benefit for 20–24 year old males is the equivalent of about 7.6 days in full health per year, of which 4.9 days of in life extension and 2.7 days of improved quality of life. For their female peers the model predicts 3.7 health-adjusted days gained, of which 2.2 from increased longevity. This translates to a substantial gain of 112,000 health adjusted life years for men and 56,000 life years for women (using the Disability Adjusted Life Years approach) over the lifetime of the Australian adult population in 2010. The tax would also generate revenue of around AUD 400 million each year, while the costs to the government to implement the tax was estimated at AUD 27.6 million. The overall health care expenditure over the lifetime of the 2010 population aged > = 20 was estimated to be reduced by AUD 609 million (95 per cent Uncertainty interval (UI): 368 million– 870 million) as a result of this intervention. The annual health care savings rise over the first 20 years and then stabilize at around AUD29 million per year. In other words, the costs of legislation and enforcement of the tax would be paid back 14 times over, in the form of reduced health care expenditure.

While using an averting expenditures approach, care should be taken to ensure that only costs incurred specifically to avoid the undesirable outcome are considered. Sometimes the expenditures are incurred off-site. For instance, soil erosion can increase the cost of dredging or reduce the capacity of reservoirs. To avoid this, governments may protect forests in catchment areas, which requires additional expenditures. Similarly this approach can also be used to quantify the benefit of food safety regulations.

This approach can be used in the following situations:

i) if the welfare losses due to changes in the condition of the resource can be established/anticipated and appropriate actions can be taken to mitigate this loss:

ii) The relation between the change in ecosystem quality and the averting action chosen to mitigate the impact can be established and the averting good exhibits no 'joint-ness' in production (i.e. it cannot be an input into two different production functions simultaneously).

Another important consideration is to ensure that the expenditures were incurred mainly due to changes in environmental quality, rather than for other reasons. See **Box 7.4** for an illustration of this approach.

Limitations

The method can estimate only those values that individuals can directly perceive or connect with (e.g. soil conservation, water quality, air quality etc.). In some cases, the individuals may incur multiple averting expenditures and this also depends on risk averseness of the individuals and their income. There is a possibility that the actual risk is different from the perceived risk, which depends on individual's perceptions, attitudes, incomes and other socio-economic factors; thus the averting expenditures may be biased on either side. The values so obtained are only a small proportion of the benefits and thus should be used as lower bound.

Residual imputation approaches

Profitability is a central concern in the farming sector and the rate of return on different farm assets, farm land, labour and management are important factors. The residual imputation approach is most commonly used to judge the productivity of a resource that is not easily measured in direct terms (e.g. impact of management practice, good quality land, use of particular farming technology, value of irrigation water). Using this approach, the total returns to production are divided into shares based on their marginal productivity until the total product is completely exhausted. Using this approach, which can be seen as a simplified version of a production function, the incremental contribution of each input in a production process can be computed. If prices (or estimated shadow prices) can be assigned to all inputs (other than the particular resource whose value is to be estimated), the value of the residual inputs (e.g. water) is the remainder obtained by subtracting the total value of all factors and inputs from the total value of product. This includes, however, any scarcity rents to other fixed factors not included in the assigned valuations (land could be an example) and has to be seen as the value of all residual inputs.

The residual value represents the maximum amount the producer is willing to pay for a resource for which she does actually make a payment (e.g. land, well-drained soil or water) and still cover all other factors or input costs (land, labour, technical inputs etc.). Turner (2004) states the following conditions under which this approach is valid: i) factors other than the resource considered are rewarded an amount equal to exactly the value of their contribution to net revenue in the contribution they make to production; ii) all other factors of production employ productive inputs to the point at which the marginal product is equal to the opportunity cost; iii) the surplus over and above the cost of production is attributable to the remaining factors in production. As this approach is extremely sensitive to the variations in the nature of production or prices, it is most suitable where the residual input contributes significantly to the output (e.g. well-drained soils, irrigated lands). This approach can be used to compare the per acre returns for different practices. It can also help in the analysis of management practices, e.g. the use of inorganic vs. organic fertilizers etc. A further application would be its use in obtaining the value of input that substantially adds to gross value added but one that is an intermediate good for which well-established markets do not exist (e.g. pollen services in fruit production). The additional returns represent the maximum amount the producer would be willing to pay for use of the resource, after accounting for any other factors that may have been excluded from the list of measured variables in the analysis. In **Box 7.4** an example is provided to illustrate this approach.

Limitations

The method is valid as long as the requirement of the competitive model is satisfied. If the factor inputs are not employed at the level to where their unit prices are equal to the value of the input in terms of what it contributes to production (known as the marginal value product in economics), this method gives erroneous results.

Replacement / restoration costs / cost savings technique

Replacement cost / restoration cost techniques approximate the benefits of environmental quality by estimating the costs that would be incurred by replacing/ restoring ecosystem services using artificial technologies. It can be applied only if replacement is indeed possible and cost-effective. The technique differs from averting cost, which infers value from actual behaviour (revealed preference). In this case, the substitute that replaces the ecosystem asset should provide a service similar to the original ecosystem asset.

Box 7.3 Health costs from exposure to pesticides in Nepal

Use of pesticides has significant negative impact on farmer's health including headaches, dizziness, muscular twitching, skin irritation and respiratory discomfort in addition to ecosystem health. Based on data collected from January to June 2005 from 291 households in Central Nepal, taking into account household demography, personal characteristics, farm size and characteristics, history of pesticide use, history of chronic illness and property of the households, Atreya (2008) estimated the health costs associated with pesticide exposure in rural Central Nepal. The cost of illness and averting action approach was used to estimate the cost of pesticide use.

In the first step, the probability of falling sick was measured by a set of acute symptoms during or within the 48 hours of pesticide application, and the possibility of taking averting action (i.e. costs associated with precautions taken to reduce direct exposure to pesticides, such as masks, long sleeved shirts or pants sprayers, etc.) was modelled on a set of socio-economic, environmental and individual characteristics. The dose response and averting actions are specified as a function of insect and fungicide doses applied (defined as concentration multiplied by spray duration), average weekly temperature, education levels, training in pest management, and farmers' body mass index. Greater exposure is expected to lead to greater averting action.

The cost of illness (COI) and averting actions are used for valuing health damages due to pesticide exposure. The health care costs considered are the costs of consultations, hospitalizations, laboratory tests, medications, transport to and from clinics, time spent travelling, dietary expenses resulting from illness, work efficiency loss, work-days lost, and time spent by family members in assisting or seeking treatments for the victim. The health care costs (annualized with the expected life spans) are predicted for users and non-users of pesticide respectively as the sum of weighted average annual treatment costs (and productivity losses) and average costs of averting actions for users and non-users, with the probabilities of falling sick due to pesticide exposure for users and non-users used as weights respectively. The actual health costs for an individual due to exposure to pesticides is calculated as the difference between the costs for the two groups. The predicted probability of falling sick from pesticide-related symptoms is 133 per cent higher among individuals who apply pesticides compared to individuals in the same household who are not directly exposed. Households bear an annual health cost of NPR 287 ($4) as a result of pesticide exposure (10 per cent of annual household expenditure on health care and services). These costs vary with fungicide exposure. A ten per cent increase in hours of exposure increases costs by about twenty-four per cent. Taking into account the averting costs, the total annual economic cost of pesticide use for the population of Panchakhal and Baluwa Village Development Committees is estimated to be NPR 1,105,782 (US$ 15,797) per year in the study area, which is equivalent to 55 per cent of the annual development and administrative budgets that the two village development committees receive from the Government of Nepal.

Box 7.4 Value of irrigated water in agriculture using residual imputation method

The value of water can be estimated through both observed market behaviour (water rights, value of land, etc.) methods, direct techniques which elicit information (demand for water as final good, e.g. water markets) and indirect techniques inferring economic value (where water is an intermediate good). The most commonly used methods to value water as an intermediate good are the production function approach and residual imputation method. Most often in developing countries water is not priced efficiently or is underpriced. In Jordan farmers pay a very negligible price for water and actual market behavior is not relevant. Water is subsidized and farmers view this as free gift. Hence any technique that relies on asking farmers to state their willingness to pay does not yield good estimates.

Using the Residual Imputation method, the value of irrigation water has been estimated by Al-Karabelih *et al.* (2012) in Jordan. The average value has been estimated to be JD 0.51/m^3 at the country level (approx. USD0.72/m^3), which amounts to a significant share of total value. Other factors include labor, machinery, fertilizer etc. The study revealed a high level of variability in irrigation water values. It was shown that the differences in water values can be mainly attributed to two factors that can be relevant for policy makers and extension services: i) the characteristics of irrigation system and ii) the type of crop grown. The aggregate average water value for field crops was 0.44 JD/m^3 (0.62 USD/m^3) for the vegetable crops in this study it was 1.23 JD/m^3 (1.73 USD) and for fruit trees is 0.23 JD/m^3 (0.32 USD). The aggregate average water value for horticulture is 0.51 JD/m^3 (0.69 USD/m3).

This technique has been widely used to estimate the value of soil conservation – micronutrients, soil carbon – but also irrigation, pollination services, water retention capacity etc. Deforestation, shifting cultivation and poor agricultural practices can accelerate soil erosion with both on-farm and off-site. The key on-site impact is a decline in productivity due to loss of topsoil and nutrients, organic matter and water retention capacity of the soil. Improper irrigation practices can also reduce the quality of soil due to salinization. In both cases, the replacement cost technique has been commonly used, as it is relatively easy to observe actual expenditures made and engineering estimates are widely available. An important assumption of this method is that the individuals affected by the change in ecosystem service would be willing to incur the costs needed to replace the services provided by the original asset. The approach can provide reliable estimates only if we have reason to believe that the replacement costs incurred are less than aggregate Willingness to Pay (WTP) (Bockstael *et al.* 2000) for the benefits of the original asset that is replaced or restored. In this case, when correctly used, the technique can provide a lower bound of value.

The replacement cost method, although very popular, can be used to estimate only a few ecosystem service values (for which the substitutes or the engineered substitute can provide the same quality and level of service – for e.g. pollination, micronutrients, irrigation, water retention capacity etc.). The cost savings method estimates the value, in terms of savings relative to the use of the next best marketed economic alternative, and this approach has same limitations as that of the replacement cost method.

However, not all inputs can be bought or are substitutable. In this case a closer proxy is used. For example, the only way to substitute for the lost micronutrients from soil erosion is to add more fertilizer. In this case the impact of change in soil quality or environmental capital is estimated by valuing the increased cost of the substitute fertilizer[7]. As this input has a market price, the additional cost of that input represents the value of the lost micronutrients. Caution should also be taken in the use of market prices – these prices must be 'efficient prices' i.e. they should include any externalities (arising due to market imperfections and policy failures) generated in the production of the fertilizer or associated with damage from runoff. **Box 7.5** provides an example of application of this approach.

Limitations

Replacement cost uses costs as a proxy for benefits, which is not accurate in all situations and thus could provide a lower bound to the true cost only if used accurately. The main assumption here is that the environmental service being replaced is of comparable quality and magnitude and the least costly alternative is chosen among the set of alternatives available to provide a similar level of service. If the substitute chosen is not the least costly alternative, the replacement cost estimates can be overstated and thus misleading. The second assumption is that the cost of replacing or restoring the environmental service does not over- or underestimate the loss in service, which is often not the case (for e.g. in case of soil erosion, some soil may be deposited on-farm and some off-farm and thus may not be completely lost). The method can be applied only when the benefits from the ecosystem services are larger than the cost of producing the services through substitute means. Several resources cannot easily be restored or replaced (e.g. climate, water, species extinction). This method can only capture use values but not non-use values. Furthermore, the approach cannot provide marginal values. Despite its limitations, it is widely used owing to the ready availability of market data but a great deal of care is needed while using this technique.

7 In estimating such an impact, it is important to have an estimate of the productivity of the micronutrients in the production function. If this is not measured the estimate of the amount of fertilizer needed will be biased.

Box 7.5 Valuing insect pollination services with cost of replacement

Insect pollination is a key input for approximately 84 per cent of the 300 commercial crops grown worldwide. What options do farmers have if wild insect pollinators do not provide this service? Existing alternatives include pollen dusting, hand pollination and managed beehives (domesticated bees). Using the Western Cape Deciduous fruit industry in South Africa as a case study, due to its dependence on managed honeybees, Allsopp *et al.* (2008) estimated the value of both wild and managed pollination services. Two scenarios were considered: i) no insects (wild or managed) remain for crop pollination, ii) managed pollination is not commercially viable or possible, leaving only wild pollination services.

Possible options for the replacement of pollination services are limited: i) the use of managed non-honeybee pollinators, which is not considered feasible in the Western Cape, ii) producing fruit without fertilization, which is not a practical short term solution, iii) pollination by mechanical means, which requires pollen to be collected from appropriate cross-pollinating cultivars, and then applied either by hand or mechanical means (e.g. pollen dusting). Pollen dusting may be done by aircraft and helicopters (efficacy unverified) or with hand operated pollen blowers. Hand pollination entails the manual application of pollen to the stigmas of individual flowers by means of a paintbrush or similar tool. Three hand pollination methods were considered. The output of fruits resulting from pollen dusting is estimated to be 73.5 per cent less as compared to insect pollination. Fruit weight from pollen dusting is estimated to be 42 per cent less when compared to insect pollination. By contrast, hand pollination of flowers is expected to deliver equal or more fruit output than insect pollination and as big or bigger fruit. Depending on which of the four value estimation methods were used, replacement values varied significantly due to differences in pollination efficiencies and the costs of different replacement methods, ranging between 0.23–1.30 of proportional production estimates. However, irrespective of the choice of replacement method, the value of wild pollination services has been underestimated in the past.

Caution: It must be noted that the estimated replacement cost may not reflect actual producer behaviour.

Table 7.3 Pollination service values using different approaches (to the Western Cape deciduous fruit industry), US $ millions, 2005 (Source: Allsopp *et al.* 2008)

Valuation method	All insect pollinators	Managed pollinators	Wild pollinators	Ratio of wild to managed value
"Traditional"				
Total production value approach	501.0	378.3	122.7	0.32
Proportional (dependence) production value approach	358.5	312.2	46.3	0.15
Revised service value estimates based on experimental evidence				
Proportional (dependence) production value approach	338.3	119.8	218.5	1.82
Production value derived from pollination services	333.9	118.0	215.9	1.83
Cost of pollination (hive rental)				
Current direct cost	-	1.8	-	-
Estimated direct cost assuming managed honeybee substitution	4.3	1.8	2.6	1.44
Pollination service replacement value (income lost)				
Pollen-dusting	292.9	107.8	185.2	1.72
Hand pollination (method 1)	161.2	44.9	116.3	2.59
Hand pollination (method 2)	433.8	122.8	310.9	2.53
Hand pollination (method 3)	77.0	28.0	49.1	1.75

7.3.3 Revealed Preference Approaches

Revealed preference approaches draw statistical inferences from observations based on actual choices made by people in markets. The travel cost method and the hedonic price method, discussed below, fall into this category. For example, individuals value different environmental attributes (for example, clean air, landscape, etc.) and reveal their preference for these attributes through the market price they pay to buy a property. Similarly, individuals reveal the value they hold for a particular ecosystem by their travel choices and the costs they incur to visit that location. By estimating a relationship between the observable choice variable, individual specific variables and the price they pay to obtain it, we can estimate the value of marginal changes in the choice variable (say, an environmental attribute) under consideration.

Hedonic Pricing techniques

The hedonic pricing method became popular after Rosen (1974), showed how a homogeneous good (house, land, job, etc.) can be regressed on its characteristics or services and the unique implicit price of each attribute can be estimated if the markets are in equilibrium. The method can be applied to commodities, products or services with clearly differentiated attributes (e.g. organic vs. inorganic products). The method has also been used to establish the relationship between wages and job attributes (for example, exposure to harmful chemicals).

Productivity of agricultural land depends on various attributes (agronomic variables, neighbourhood, environmental and policy variables) and the land prices indicate the value that consumers or producers are willing to pay for these attributes. Two different pieces of land may look identical but their characteristics and environmental attributes (e.g. soil quality, biodiversity) may be different, and thus they may fetch different prices.

The price differential between the lands due to difference in one such characteristic can be used as a measure of the marginal value of the characteristic. This is called the "Hedonic Price method". The technique has been widely used to measure various characteristics such as the implicit price for soil (Miranowski and Hammes 1984), the impact of soil erosion (Gardner and Barrows 1985; Ervin and Mill 1985), the value of erosion control (Palmquist and Danielson 1989), impact of climate on agricultural productivity (Mendelsohn *et al.* 1994, Dinar *et al.* 1998, Maddison 2009), the recreational and amenity benefit from agricultural open space or the dis-amenity from intensive animal production to adjoining properties.

Using the hedonic price method requires two steps. In the first step, the value of agricultural land per unit (hectare, acre) is estimated as a function of the quality of land, neighbourhood and environmental characteristics. Once this function is estimated (which is the hedonic price function), the implicit price (change in price/value of land due to change in any of the attributes) for each of the statistically significant attributes can be computed (which could include ecosystem services). This price is the first derivative of the implicit price function with respect to the attribute/service considered. In the second step, the implicit price is regressed on the quantity of the characteristic as well as the socio-economic characteristics of the farmers to estimate the changes in welfare due to changes in the particular environmental or ecosystem service attribute (see **Box 7.6** for illustration).

Key advantages of this approach include: i) the method allows compressing the attributes of the composite good into one dimension, ii) the approach can be used to reflect the marginal trade-offs between different attributes through examining the difference in prices for change in different attributes (Rosen 1974).

Limitations

The method can only be deployed to estimate use values. The key assumption of this technique is that information on the land and its attributes is readily available to the farmer, who can then factor this into a decision on how much to pay for the land. Another limitation of this approach is that agricultural markets are rarely as dynamic as housing markets. The data requirements, as is the case with several other methods, can be quite intensive. The method works well if markets can pick up quality differentials, which may not be the case for agricultural land, due to the non-observability of some attributes (e.g. some bio-physical features).

Travel Cost Method

The travel cost method, first used by Hotelling (1947) can estimate the value of recreational sites, which may be public or quasi-public goods[8] (e.g. recreational value of agricultural landscapes). The model uses actual expenditures and other costs (including the value of time) incurred by individuals in visiting a specific recreational site to estimate the value of the benefits obtained from the site. Primary data are collected from a sample of tourists visiting the recreational site. The survey includes information on the place of origin of the tourist, the expenditure they incurred, their mode of transport, the time spent on site, along with various socio-economic characteristics. A demand curve is generated with the visitation rate (number of visits per period) as the dependent variable and distance, cost

8 Quasi -public goods have characteristics of both private and public goods and are partially excludable (i.e. the party responsible for managing the good can prevent others from using it), partially rival/congestible (i.e. if one person benefits from the good, others cannot fully benefit from it).

per trip, presence of substitute sites, socio economic conditions as explanatory variables (Garrod and Willis 1999). From the resulting demand curve, the consumer surplus can be estimated. The underlying assumption is that people will visit a site only if the marginal benefit of recreation is at least as large as the marginal cost (see **Box 7.7** for the illustration).

Limitations

One of the assumptions of the travel cost method is that there is a clear perceived relationship between the environmental attribute in question and visitors' travel patterns, which may not be true. In many cases, visitors know the quality of a site only after they visit; it can therefore be difficult to value changes in recreational or environmental quality. In addition, the method is quite data intensive and can be complicated if the tourist visits multiple sites on a single trip. The method can only be used to obtain use values. The method does not give reliable results if the site or travel zones are very close to each other or if there is not enough variation in the explanatory variables. The method is also very sensitive to the type of statistical analysis chosen and to how the opportunity cost of time is measured.

7.3.4 Stated Preference Methods

Stated preference approaches are based on eliciting values directly from a set of the affected population. There are two broad methods: contingent valuation and choice experiment.

Contingent Valuation Method

The contingent valuation method has been extensively used for the valuation of non-marketed environmental resources (see **Table 7.4**). The approach requires eliciting individual preferences directly through individual surveys (a stated preference approach) through simulating hypothetical markets. The survey aims to understand the preferences of individuals by describing a scenario (i.e. describing the good, provision of the good, existing state of the environment), and how the provision would change under different management responses or hypothetical alternatives. The scenario also mentions who would provide the good and how. Respondents are then asked to state their willingness-to-pay (WTP) to avoid or willingness to accept (WTA) this change using different elicitation methods and the payment vehicle (taxes, user fee, one time payments etc.). It should be noted that the WTP and WTA may be different. Along with this, some information on the socio-economic background of the individuals, their knowledge on environmental issues, their attitudes towards environmental good under consideration as well as the preferences for general environment is also elicited. The demand for the environmental good is then estimated through different econometric approaches.

Box 7.6 The value of natural landscapes: application of the Hedonic Price Method

Living in close proximity to nature provides positive welfare benefits through improved health and well-being. These cultural services provided by agricultural landscapes can be estimated through stated and revealed preference techniques. Hedonic price studies have been commonly used to investigate the effect of environmental amenities on property prices (for instance the impact of water quality, or proximity to protected areas such as wetlands, forests, beaches, scenic views, or open spaces on property prices).

Walls *et al.* (2015), using property sales data from the St, Louis Country, Missouri, Revenue department for the years 1998 through 2011, estimated the value of home's sale price as a function of the percentage of its view that encompasses various 'green' land covers – forests, farm land and grassy recreational lands, as well as proximity to such green spaces. Data was also collected on structural characteristics of relevant buildings, such as number of stories, square footage, number of bedrooms, and lot size among other attributes. The hedonic price function has been mapped with georeferenced parcel boundaries. The property price (adjusted for inflation) has been estimated as a function of building age, the share of property in natural land cover, the diversity of the view of the property, and the year of sale, using a fixed effect panel data model. The results from the model suggest that proximity to all three kinds of open space has positive value to home buyers, but the effects of views are more mixed. The larger the forest view from a property the lower the property price (because people valued a more mixed landscape rather than a single monotonous view in this particular case), all else being equal. However, the farmland and grassy land have positive effects, with farmland coefficient being statistically significant. A 10 per cent increase in the amount of farmland in a home's 'view shed' leads to an increase in almost 2 per cent of its price. The reason for significant positive value of farmland on home sale prices is due to the scarcity of the farmland due to their increased conversion for property development.

Limitations

Contingent valuation has been widely used in the environmental valuation literature and in several circumstances remains the only method available to estimate the non-use values. However, the method is complex, data intensive, costly to implement and requires carefully designed surveys to gather unbiased information. The estimates are dependent on the respondent's knowledge, ability to understand and visualize the circumstance of the good or service being considered. Respondents may understate or overstate their WTP/WTA depending on their beliefs and other factors not related to valuation.

Choice Experiments

In choice experiments, rather than presenting a single scenario respondents face a sequence of choice sets. These present different environmental attributes of varying quantity and quality including the cost to provision the good or the price the consumer or user may have to pay to obtain the good. The respondents' preferred option, in response to the change in attribute levels, are modelled to determine the people's WTP or WTA for the changes in different levels or quality of attribute under consideration. Thus it is possible to see how people trade one attribute or preference against the other and the welfare changes can be calculated (see **Box 7.9** for application of choice experiment technique).

Limitations

The choice experiment method is based on the notion that attributes of the good being considered can be used to understand the trade-offs. However, the success of the method depends on selection of the appropriate attributes and levels. Unfamiliar trade-offs, too few alternatives or too many alternatives may give incorrect estimates as the respondents or may end up choosing the alternatives that are simpler.

Box 7.7 Value of ranch open space in Arizona

Agriculture provides positive externalities, but the land market may not be working efficiently to capture the value of such externalities. Rosenberger and Loomis (1999) measured the benefits to tourists associated with ranch open space in the resort town of Steamboat Springs in Routt County, Colorado. The traditional ranch practices in Yampa River valley have preserved open space, with more than 10,000 acres of privately owned ranch land in the area. However, the area near Steamboat Springs lost approximately 20 per cent of its ranch land to development uses between 1990 and 1995. Thus research seeks to answer the questions: "Do people choose to visit the area, in part, because of the existing ranch landscape, and how much does it contribute to the enjoyment of a Steamboat Springs summer visit?' and "How would visitation rates change with additional subdivision of valley ranch land?"

Survey data was collected through in-person interviews of 403 adult visitors on stratified random days. Information on the characteristics of summer visitors to the resort area, including state of residence, mode of travel, type of lodging, choice of recreation activities, spending patterns and attitudes towards services provided in the area were collected. The observed behaviour data collected included total number of trips and total number of days the individual expected to spend in the Steamboat Springs area during the summer season and the distance between their home and another resort area with comparable ranch open space. The contingent behaviour questions asked whether they would increase, decrease or not change their current visitation rates if all the current ranch open space were converted to urban and tourism development uses. If they stated they would change their rate of visitation, they were asked to state the number of days. The change in the number of days was computed by first estimating the average number of days per trip spent onsite based on observed levels and then adjusting the number of trips spent onsite based on observed levels, and then adjusting the current number of trips by the ratio of days per trip based on the contingent number of days. The model was estimated using panel data Poisson technique. The average consumer surplus (a measure of welfare) per group trip is estimated by estimating the area under the estimated travel cost demand function (or integrating under the demand curve), which plots the number of trips on the horizontal axis and the cost per trip on the vertical axis. The integration is carried out between the average travel cost per trip and the maximum price at which no trips are made. This is done both under the current conditions and hypothetical condition without ranch space. The average consumer surplus received per group trip was $1,132 with existing ranch open space. This value was used to value the changes in number of visits when open space was altered and thus to compare benefits from visitors with benefits from ranching.

Box 7.8 Consumers attitudes towards to Genetically Modified Organisms in the UK: Application of choice modelling

Gene technologies, while significantly benefitting society, can pose potential risks to humans. While the benefits, such as higher productivity, are immediately realized, the risks of affecting other plants and species are often not immediately visible, and thus countries have regulations enforced to protect the health and safely of people and to safeguard the environment. For example, the EU has placed restrictions on the import of genetically modified soya, and the UK food and drink manufacturer and retailers agreed to label foodstuffs containing GM soya or maize protein.

Burton *et al.* (2001) set out to identify consumer WTP to avoid these products in order to help in identifying the appropriate level of policy response. Choice modelling approaches require presenting different attributes to users or consumers, and in this case of GM crops, the consumer was presented two attributes in each option in the form of technology used to produce food (traditional or GM) and the level of the weekly food bill for the individual. In selecting between these two, the respondent was asked to compare the reduced food bill with the change in technology. Option 1 is chosen if the welfare from its level of attributes is preferred to that generated by Option 2. Three production technology levels were identified: traditional, plants modified by plant genes and plants modified by both plant and animal genes.

The survey was administered over the summer of 2000 in Manchester UK using drop-off and collect approach. A total of 228 complete surveys were obtained over a six-week period, seeking to answer how much consumers would be willing to pay to avoid GM technology, computed as change in food bill. The inclusion of food bills acts as payment vehicle. Personal characteristics were included in the analysis, interacting with attribute levels to explain the choices. The study found a univocal aversion to GM food across all users – infrequent, occasional and organic food users. The infrequent group was prepared to pay 13 per cent more on food bills to achieve a 10 per cent reduction in GM use.

7.3.5 Risk, uncertainty and quasi-option values

The discussion so far has been on methods for eliciting certain values of eco-agri-food systems that are normally unaccounted for in decision-making processes. In this subsection we consider certain categories of value that are important in decision making for this sector. They may be elicited through a variety of techniques; what is critical to understand where they come into play in the decision-making process.

The agriculture and food industry is subject to significant risks and uncertainty, which adds a considerable degree of complexity to decision making. Imperfect knowledge about the future is referred to as risk, if the likelihood of consequences is known and probabilities used. If the likelihood is not known, the lack of knowledge is referred to as uncertainty. Broadly speaking the risks in agriculture arise from the variability in market prices, exchange rate fluctuations, government policies; uncertainty arises due to the natural variability in the production of crops, weather, incidence of pests and diseases (e.g. foot and mouth disease, incidence of E.Coli), food quality and safety, catastrophes and climate change.

Despite risk and uncertainty, decisions have to be made regarding the allocation of resources. The nature of the decision depends on whether the individuals or businesses are risk averse, risk neutral or risk loving. Risk averse farmers, for example, adopt diversified farming systems, buy crop insurance (drought or flood insurance) or undertake actions to adapt to risk and uncertainty (such as supplemental irrigation measures to offset the risk of insufficient rainfall or constructing dams and levees to control flooding). Accessibility of information plays a crucial role in decision-making, especially considering the irreversibility of certain decisions, and thus it is important to value the information. For example, biotechnology increases crop yields, reduces pesticide costs and enhances crop adaptation. However, there are potential risks to human and animal health and irreversible risks to the environment. While the benefits are known with some certainty the costs (the risks) are uncertain. As a result, some countries have adopted a precautionary approach, an example of value of information by delaying the action, which is the quasi-option value.

The quasi option value is the value gained by waiting for additional information before making an irreversible investment (Arrow and Fisher 1974). **Box 7.9** illustrates the example of quasi-option value from delayed input use from Magnan *et al.* (2011). Drought is a major risk factor where farmers can have three alternatives to choose from – farming in locations known to have lower risks, investing in irrigation structures, or choosing crops, technologies or seeds that are drought resistant and/or adjusting input use in growing seasons (Magnan *et al.* 2011). Farmers who are flexible in adjusting their input use can choose between no till (NT) agriculture and conventional tilling (CT). However, the inflexible farmers do not factor the stochastic rainfall in their decisions in period 1 and thus cannot change the decisions later on. The difference in the profits between CT and NT gives the quasi-option value.

Box 7.9 Quasi option value from delayed input use

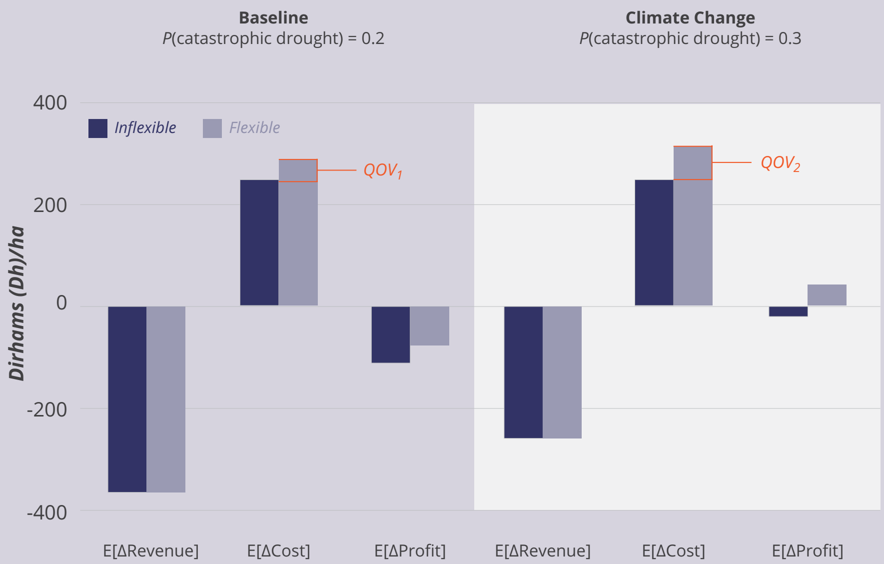
No-till agriculture (NT) allows farmers to forgo plowing by seeding directly through the stubble of previous years' crops, which the farmer is required to leave on the field. The benefits of no till agriculture are: lower planting costs (leaving more resources to replant), improvement in soil quality, efficiency in water use, higher yields in years of mild drought and many environmental benefits in the form of lowered emissions, reduced erosion and increase soil organic carbon. In addition, NT changes the input timing so that relatively fewer costs are incurred early in the growing season (lower pre-planting costs but higher costs during the growing season) compared with CT. However, the risk is that cost savings may be offset by the increased crop protection costs and higher fertilizer use at a later stage to maintain the same yield. The flexible farmers (willing to adopt NT) may get lower yields than the conventional farmers due to greater experience with CT. Farmers may perceive additional risk with NT than CT. The decision to opt for till or no-till has to be taken at the beginning of the planting season when he does not have information whether there would be normal rainfall or drought. Based on surveying 197 rainfed wheat farmers in Morocco, Magnan et al (2011) estimate the quasi-option value. Two scenarios are assumed – base case (catastrophic droughts occur with 0.2 probability) and climate change (which increases the probability catastrophic droughts to 0.3). The decision-making matrix of the farmers is based on the following payoff matrix:

Table 7.4 Expected benefits and costs of decision making under two management scenarios

Time	Action	Base case (probability of catastrophic disaster = 0.2)		Climate change (probability of disaster = 0.3)	
		Costs	Expected benefits	Costs	Expected benefits
t = 1	No Tillage (NT)	1380 Dh/ha	EB(CT) - 356	1380 Dh/ha	EB (CT) - 262
t = 2		1785 Dh/ha		1785 Dh/ha	
t = 1	Conventional Tillage (CT)	1830 Dh/ha	EB(NT) + 356	1380 Dh/ha	EB(NT) + 262
t = 2		1585 Dh/ha		1785 Dh/ha	

Under the baseline scenario, the expected net revenue from No Till (NT) is 356 Moroccan Dirhams/ha ($45/ha) lower than under CT. The inflexible farmer (adopting CT) saves 250 Dh/ha on production costs. Under climate change the expected net revenues of the farmer under NT is assumed to be 262 Dh/ha less than under CT. The cost savings are still the same 250 Dh/ha under this scenario as well. The inflexible farmer in the case of base case scenario saves 250 Dh/ha on production costs. But the flexible farmer receives 40 Dh/ha more of quasi option value. However, in the case of climate change, the quasi option value of delayed input use is 60 Dh/ha, which increases total expected savings to 310 Dh/ha (24 per cent increase) and the total expected benefit of adoption increases to 48 Dh/ha for the flexible farmer (see **Figure 7.3**).

Figure 7.3 Changes in expected revenues, costs and profits from adopting no-tillage (Source: adapted from Magnan *et al.* 2011)



7.3.6 Using valuation to derive aggregate estimates of external costs and dependencies

The previous section reviewed the methods available to value farmers' dependencies on ecosystem services and the externalities related to eco-agri-food systems (positive and negative). **Table 7.5** summarizes the findings from that review. Each method has strengths and weaknesses. Despite the limitations, if used with care these valuation methods can generate reliable estimates of the external costs of different combinations of agricultural practices.

One limitation that merits special mention is that these valuation methods do not directly deal with the question of *who* gains and who loses from a change in ecosystem services. The focus is on aggregate gains and losses and, while these are made up of gains and losses to individuals or particular groups, the breakdown is not generally presented in the reporting of results. Such distributional aspects are of course important, as they bear on the social capital of a community or society. They emerge as issues to be considered in any wider evaluation of the changes under consideration. It is important to note, however, that data relevant to such an evaluation can often be found in the detailed assessment of the values of ecosystem services (ESS).

Table 7.5 Methods for Valuation of Ecosystem Services (Source: authors)

Method	Data Required	Best Suited For	Main Limitations
Market Values	Prices and quantities of the inputs and outputs	All cases where market data are available	Cannot be used to value those services that have no market value
Production Function	Quantities of inputs and outputs in physical units. Prices of key outputs and inputs	Cases where data on a wide set of inputs and outputs is available	Gives biased estimates when data is missing on key inputs and when prices change.
Dose Response functions	Input in question and outputs that are affected	Cases with clear links – e.g. air pollution, weather/ climate	By itself does not take account of more complex responses to changes in dose on production across sectors.
Averting expenditures	Expenditure to avoid a negative externality and magnitude of the externality	Cases where strong averting behaviour is observed	Complex responses that may include an element of averting behaviour are difficult to model and need a lot more data
Residual Imputation Approaches	Data on all inputs and other outputs except the one of interest.	Estimation of the residual value of one ESS	It is rare to get all other data so values for the residual will contain more than just the value of the input of interest.
Replacement/ Restoration	Data on amount of ESS los and cost of replacement	Where one ESS is reduced and it is reasonable to assume you will want to find a replacement	Not based on willingness to pay. Costs are used as a proxy for benefits, which is not always the case
Hedonic Prices	Price and quantity of the good or service and quantities of all related attributes	Cases where values of land are strongly affected by some ESS	Extensive data requirements and assumption of efficient markets
Travel Cost	Data on number of visitors, cost of travel, attributes of visitors and attributes of sites.	Largely cultural sites and other recreational uses of land	Extensive data requirements. Estimation of opportunity cost of time
Contingent valuation/ Choice experiment	Survey data on money values of individuals given hypothetical information about a situation	Cases where individuals are able to express clear preferences Non-use values	Biases in answers possible but can be limited be design. Data requirements are extensive

In **Box 7.10** that follows, some examples of the application of valuation methods to estimate aggregate (national or regional) external costs are presented. While there are many gaps that need to be addressed, the applications described in **Box 7.11** show the power of valuation methods for estimating the effects of externalities related to agriculture on the sector and on society at large. The studies summarized here have been included to give an idea of the total value of external costs that emerge from the literature and what they tell us about where the externalities arise.

The research by Pretty *et al.* (2005) showed an external cost for the UK in 2005 of around 0.1 per cent of GDP, which may seem a small figure but includes potentially significant costs for human health and emissions to the atmosphere. Interestingly, the costs were estimated to be considerably lower (75 per cent less) if all production were to go organic. Other studies also show that significant reductions in external costs can be achieved at the national or regional scale if measures for conservation are introduced. These studies are not without criticism, but they are important in showing what could be done using the methods described here. Further improvements in estimates can be expected once approaches described in this report are put in practice.

Box 7.10 Application of externality valuation to estimate the aggregate impacts of agricultural practices

A value-based approach was taken by Pretty *et al.* (2005) who undertook an economic analysis of the costs imposed by the UK food system. The external costs of the current agricultural system were compared with those that would arise were the whole of the UK to be farmed with organic production systems (see **Table 7.6**). They used standard organic protocols to estimate the contribution that would be made to the total costs by each of the ten sectors listed in the table. The study assessed the full cost of the UK weekly food basket by estimating the environmental costs to the farm gate for 12 food commodities, and the additional environmental costs of transporting food to retail outlets, and then to consumers' homes, and the cost of waste disposal (shown in **Table 7.6**). The methods used in these studies were largely cost-based rather than demand-based, and involved the use of replacement costs (e.g. hedgerows, wetlands), substitute goods (e.g. bottled water), loss of earnings (e.g. due to ill health), and clean-up costs (e.g. removal of pesticides and nitrate from drinking water). The results show a considerable reduction in costs from a switch to organic production. The present costs are also measured relative to the amount paid and found to be about 12 per cent of that figure. No attempt was made to assess the savings in external cost relative to the higher cost of shifting to organic production. The valuation methods have improved considerably since this study was done, but it is still one of the few studies that values the external costs in a way that covers the full value chain as set out in **Figure 7.4.**

Other studies that measure the loss of ESS related to agriculture include Pimentel *et al.* (1995) and Gascoigne *et al.* (2011). Pimentel *et al.* (1995) estimated damages caused by soil erosion in the US and compared them against the costs of avoiding erosion. Erosion was valued in terms of additional energy, nutrients and water needed to maintain a given level of production, as well as the costs of siltation and damage caused by soil particles entering streams and rivers and harming habitats. Total damages amounted to about USD 100 ha^{-1} yr^{-1}. Costs of conservation through methods such as ridge planting, no-till cultivation, contour planting, cover crops and windbreaks were estimated at around USD 45 ha^{-1} yr^{-1}, thus providing a healthy net benefit in overall terms. Valuation methods did not, however, include the recent work on damages from pesticides and fertilizers on streams and rivers.

Gascoigne *et al.* (2011) compared the societal values of agricultural products and ecosystem services produced under policy-relevant land-use change scenarios and explored the effectiveness of mitigating loss with conservation programs in the native prairie pothole regions of Dakota. Crops were valued using market data. ESS of carbon sequestration, sedimentation and waterfowl production were estimated by biophysical models and valued by benefit transfer. The authors evaluated four scenarios for a 20-year period ranging from aggressive conservation to extensive conversion for agriculture, in terms of changes in market and non-market ESS and including any costs incurred in implementing these scenarios. In benefit cost terms, the scenarios where native prairie loss was minimized and Conservation Reserve and Wetland Reserve lands were increased provided the most societal benefit. This included taking account of the value of land lost to production.

Table 7.6 The negative externalities of UK agriculture, 2000 (Source: adapted from Pretty *et al.* 2005)

Sources of adverse effects	Actual costs from current agriculture (£ M yr^{-1})	Scenario: costs as if whole of UK was organic (£ M yr^{-1})
Pesticides in water	143.2	0
Nitrate, phosphate, soil and Cryptosporidium in water	112.1	53.7
Eutrophication of surface water	79.1	19.8
Monitoring of water systems and advice	13.1	13.1
Methane, nitrous oxide, ammonia emissions to atmosphere	421.1	172.7
Direct and indirect carbon dioxide emissions to atmosphere	102.7	32.0
Off-site soils erosion and organic matter losses from soils	59.0	24.0
Losses of biodiversity and landscape values	150.3	19.3
Adverse effects to human health from pesticides	1.2	0
Adverse effects to human health from micro-organisms and BSE	432.6	50.4
Totals	£1,514.4	£384.9

7.4 OVERVIEW OF EVALUATION METHODOLOGIES

The previous section focused on the use of specific valuation techniques that generate monetary estimates of the external costs and benefits of eco-agri-food systems and their dependencies on ecosystems. These estimates are of great value to both public policy makers and private investors, but questions of equity, education and awareness in promoting health practices and contributing more widely to the Sustainable Development Goals (SDGs) should also be considered in food production. In addition, links across the economy, between the eco-agri-food system and other sectors, as well as the contribution of the sector to employment and economic growth will always be important considerations. Evaluation methodologies that help us understand how eco-agri-food systems function in light of these wider goals include:

i) Cost Benefit Analysis `

ii) Life cycle assessment

iii) Evaluating the role of merit goods

iv) Integrated approaches that evaluate several goals

v) Multi-Criteria Analysis and Cost-effectiveness Analysis

Not all the evaluation methods listed above use monetary valuation, although many do. Some non-monetary methods such as life cycle analysis provide data that can be used for monetary approaches, as well as being of direct use in their own right. Other methods, such as multi-criteria analysis, incorporate and extend some of the methods described above.

This section and the next show how these methods can help us better understand and evaluate the performance of eco-agri-food systems across the economic, environmental and social dimensions of the value chain. This analysis could help address issues such as:

- How the development of organic food products affects the incomes of farmers, as well as the sustainability of farming systems

- How the development of 'fair' trade schemes affects the incomes of growers, land use and biodiversity

- How changes in technology that reduce production costs and increase yields affect incomes and consumption habits but may increase external costs

- Increased demand for biofuels and its effects on deforestation, food prices, income of farmers and farming practices

- Effects of trade liberalization on farm incomes across different farm sizes as well as on deforestation and biodiversity

7.4.1 Cost Benefit Analysis

Cost Benefit Analysis (CBA) is a systematic process for calculating and comparing benefits and costs of a given policy or project, based on assigning a monetary value to all the activities associated with the project (either as input or output). CBA techniques are commonly used to evaluate the feasibility and profitability of business strategies and private and public projects, as well as public policy interventions. This approach generally compares the total investment and other costs required for the implementation of the project (which might include investment in fixed assets, labour and training costs, as well as the time utilized for training or implementation) against its potential returns (e.g. increased revenues).

CBA helps make clear the total costs of an intervention, as well as the benefits generated. Additional indicators include the payback period (the time needed for the investment to pay for itself); net present value (NPV, a comparison of the discounted present value of all costs and benefits); rate of return (the percentage return on investment, equal to the discount rate that makes the NPV equal to zero); and benefit to cost ratio, which is the ratio of the present value of benefits to costs (a ratio greater than one would be necessary but not sufficient for a project to be selected). A key feature of CBA is the aggregation of costs and benefits in different periods to a single value using a discount rate. To get one number for the costs of a project and one for the benefits, the analysts add together the costs and benefits in different periods but give lower weight to costs and benefits further into the future. These weights are based on a discount rate. Box 12 below describes the role of the discount rate in valuations, especially CBA.

An early example of the application of CBA methods to eco-agri-food systems was a study by Pimentel *et al.* (1995), referred to in Box 7.10, where the costs of preventing soil erosion in the USA were compared to the benefits from reducing soil erosion. The study has been criticized as a simplistic scenario but it remains useful as a guide to the method. A more recent example, also referred to in **Box 7.10**, is shown by Gascoigne *et al.* (2011), which compares the societal values of agricultural products and ecosystem services produced under policy-relevant land-use change scenarios and explores the effectiveness of mitigating environmental losses with conservation programs.

Cost benefit analysis a powerful tool but one with limitations. Most importantly it does not address the distributional question of who gains and who loses. It also gives no importance to non-valued costs and benefits. For both these reasons it is a major input to any evaluation process but is never sufficient to determine the outcome of the evaluation.

7.4.2 Life Cycle Assessment

Life Cycle Assessment (LCA) is defined as: "*a systematic set of procedures for compiling and examining the inputs and outputs of materials and energy and the associated environmental impacts directly attributable to the functioning of a product or service system throughout its life cycle*" (IOS 2016). LCA examines physical impacts across the value chain; it can also be viewed as "*a tool for the assessment of environmental loadings of entire life cycle processes related to a production system, covering all the processes, activities and resources used*" (Mogensen *et al.* 2012). For each of these steps an inventory is made of the use of material and energy and the emissions to the environment, creating an environmental profile that allows identification of the weak points in the lifecycle of the system studied. These weak points are then made into the focus for improving the system from an environmental point of view. In most cases the impacts are only reported in physical units and not converted into money terms. An example of LCA applied to food products is Shonfield and Dumelin (2005), who examine the LCA for different kinds of margarine, as laid out in **Figure 7.4.**

Emissions for different kinds of margarine are measured in terms of energy use, acidification, eutrophication, global warming and photochemical smog. In principle it is possible to value these impacts, although such measurements will be subject to considerable error bounds. It should also be noted that not all categories of impacts are negative externalities in the sense that they are damaging to the environment -–for example energy use may not be, if derived from renewable sources. Nevertheless, the LCA can be useful for policy makers and those looking for stages in the lifecycle with significant environmental impacts.

One area that LCA needs to take into account is the indirect land use implications of a policy in the eco-agri-food sphere. Biofuel policies in Europe, for example, are well known to have impacts on land use in developing countries that convert forests to grow palm oil (AETS 2013). However, the problem is more widespread and policies for land set-aside in Europe or other developed regions can also have implications for land conversion in the developing world (i.e. setting-aside reduces production and raises prices, which can impact prices and production in developing countries). For this reason it is important to distinguish between LCA accounting methods that stop at national boundaries and those that include international dimensions in a more global accounting context. The more extensive the coverage the more complete the assessment will be.

LCA is a useful complement to other data sources and can feed into other evaluation tools. It can also provide direct input into tools such as CBA or MCA. The main limitations are difficulties in tracking spillovers from one sector to another and the fact that values are rarely attached to biophysical flows (although in many cases they can be added).

Box 7.11 Discount rates and discounting

The discount rate is a parameter used to compare economic effects that occur at different points in time. Societies and individuals prefer, for different reasons, to have something now rather than to have the same thing in the future. Hence future benefits and future costs have a lower value associated with them than present day benefits and costs. If a benefit or cost has a value of $1 in the present period and the same benefit is given a value of $0.95 in one year's time then the discount rate is said to be approximately 5 per cent.

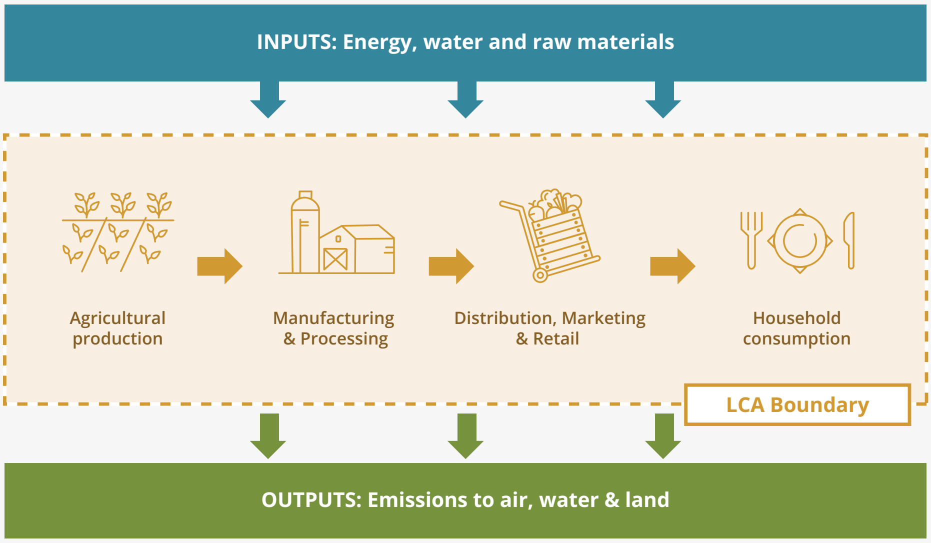
The major question is what discount rate to use when carrying out a CBA. A high discount rate makes it difficult for projects with high upfront costs (but benefits that come in small amounts over a long period of time) to have a benefit-to-cost ratio greater than one. This can make it hard to justify investments in, for example, reforestation or adaptation to climate change. A low discount rate, on the other hand can result in many projects passing the benefit-to-cost ratio test and can often imply large infrastructure projects such as dams being approved, which can also have negative environmental consequences.

Discount rates also matter when valuing natural capital. A World Bank (2006) study valued natural capital in terms of the discounted present value of the services provided by different biomes. One of these is grasslands, often used for agricultural production, where values were based on the current rental rate (i.e. the percentage of the price that is net income) combined with current prices. In the future, both of these were expected to be constant and discounted at 4 per cent. The areas of grassland depended on expected conversion to other uses and rates of degradation. Sensitivity to various parameters was examined. While the choice of discount rates mattered it did so less than assumptions about future prices.

The choice of discount rate varies according to whether it is based on private considerations or social ones. Private sector decisions that involve benefits and costs over time are usually decided on a relatively high rate – 10per cent and more, depending on the risks associated with the project. The public sector rate, however, is lower and can be in the range of 3-5 per cent in most cases. Recently, a case has also been made for adopting different rates in the public sector, according to the length of time for project or program under consideration. In this case benefits and costs are discounted at a higher rate for the earlier years and at lower rates for later years. The governments of the UK and France have adopted declining rates for public sector projects. In the UK for example, costs and benefits for the first 30 years are discounted at 3.5per cent, those for years 31-75 at 3 per cent, years 76-125 at 2.5p er cent, years 126-200 at 2per cent and so on.

How does one reconcile these two different rates? Governments apply the social rate for investments and capital valuations in the public sector and leave the private sector to apply whatever rate it considers appropriate for its decisions. This is a workable solution in most circumstances, except that some private decisions involve investments in and valuations of natural capital, which entail some use of natural capital that is not private. An example would be private investment that may degrade an ecosystem, with loss of services over many years leading to unsustainable outcomes. Creating regulations requiring such assets to be protected during any development by the private sector, based on values using low discount rates, is clearly a possibility.

Figure 7.4 Life Cycle Assessment (LCA) boundaries (Source: adapted from Shonfield and Dumelin 2005)



7.4.3 Analysis involving merit goods

Examining the effect of certain dietary choices on GHG emissions and on the health of the consuming population provides an opportunity to analyse the concept of merit goods. A study by Markandya *et al.* (2016) looks at what it would cost in terms of loss of ex ante personal welfare for the adult diet in Spain to be modified in order to meet the World Health Organization (WHO) dietary guidelines in terms of calories, fats, sugars etc. The changes in diet are brought about through a model that evaluates a 'bonus-malus' program in which foods that take the diet closer to the guidelines are subsidized while those that take it away from the guideline value are taxed. At the same time the diets are evaluated in terms of their life cycle GHG emissions. The modelling, which consists of looking at demand systems, shows that taxes and subsidies required to achieve the full transition to a healthy diet are too high to be politically acceptable (based on the authors judgment). On the other hand, with taxes and subsidies limited to between 30 and 40 per cent of the current price, an improvement in the region of 20-25 per cent in the diet is feasible (measured in terms of the reduction between the desired diet and the actual diet), while also making a reduction in GHGs that is significant. The dietary changes that the bonus-malus program brings about are a reduction in the consumption of red meat and other high GHG foods and an increase in the consumption of vegetables and low fat foods.

Measures to reduce red meat consumption through awareness programs can be evaluated in terms of the reduction in GHGs (depending on whether other goods were substituting for the lowered meat consumption), as well as expected improvements in health indicators. Both of these can, in principle, be valued in money terms but the methods of analysis generally require looking at more than just the monetary impacts and draws on wider economic analysis than is normal for most externality studies.

Such modelling is valuable in understanding the complexities involved in introducing a policy with a goal that appears to be clear and simple but in reality is not. The difficulty in using it is the problem of obtaining the model parameters and the baseline data.

7.4.4 Integrated approaches that evaluate several goals

The above review of different analysis of economic, environmental and social impacts of eco-agro-food policies shows a focus on individual impacts, as well as in combinations, notably environmental/economic and economic/social. Rarely, however, has the whole value chain been analysed as an entity. Setboonsarng and Markandya (2015) have attempted to so by addressing a policy of the adoption of organic farming by poor farmers in Thailand, Laos, Cambodia, and Sri Lanka. The methodology used, referred to as the Propensity Score Matching Method, consisted of comparing farmers who had adopted organic farming with another group that was as similar as possible but that had not adopted organic farming. Data was collected on indicators like farm inputs, outputs, income, health status, and education of children. for both groups and the results compared. **Box 7.12** summarizes the findings of a quantitative analysis that looked at the economic, health, gender and environmental impacts of a given policy.

Box 7.12 Evaluating the impacts of organic agriculture in South East Asia

In a quantitative evaluation of the pathways and magnitude of impacts of organic agriculture on the MDGs, Setboonsarng and Markandya (2015) study analyzed 11 datasets from smallholder organic farmers in marginal areas in six countries: Thailand (rice), China (tea), Sri Lanka (tea), Cambodia (Nieng Malis rice), Laos (Japanese rice), and Bhutan (lemongrass). In all but one case, household surveys were conducted on organic and conventional farmers of the same socioeconomic group and agro-ecosystem. The main findings were as follows:

1. Organic farmers earned higher profits than conventional farmers on account of lower production costs and price premiums. As organic agriculture required lower cash inputs, there was less need for credit. Organic agriculture was also pro-smallholder, as small plot size with utilization of family labour often produces better yields. As organic agriculture was more labor-intensive, it absorbed surplus rural labour. This showed especially in the practice of tea growing in China, where use of family labour was as much as 35 per cent higher in organic than in conventional agriculture.
2. In terms of MDG 4 (reduce child mortality), 5 (improve maternal health), and 6 (combat HIV/AIDS, malaria and other diseases), organic agriculture positively affected the health of farmers by reducing exposure to toxic agrochemicals as reflected in their lower medical spending.

With respect to MDG 7 (ensure environmental sustainability), organic agriculture utilized resources with less harm to the environment. The benefits of organic agriculture ranged from increasing biodiversity of farming systems to reducing GHGs in the atmosphere. As revealed in the case studies, organic farmers observed increases in the number and kinds of animal and plant species in their fields. This natural environment, which is not so negatively affected by organic practices, showed how organically farmed land can act as a gene bank that contributes to long-term food security.

Value Chain Analysis

One multi-dimensional approach currently being developed to help better determine linkages across the eco-agri-food value chain is 'value chain analysis'. The approach seeks to represent the linkages across social, economic and environmental indicators for each stage of the value chain in terms of the stocks and flows of produced, social, human and natural capital. The intention is to assess the strength and dominance of feedback loops over time, for indicators of performance that are key to many types of economic actors, as well as for society. The steps involved in applying such an analysis are suggested in **Box 7.13** below.

7.4.5 Cost-Effectiveness Analysis

Cost-effectiveness analysis (CEA) compares the relative costs and outcomes (non-monetary effects) of two or more courses of action. It is narrower than a CBA and excludes any valuation of benefits, focusing instead on the costs of attaining a given target. An example of a CEA would be looking at the cost of different options to restore a given amount of degraded land. Once the area of land and other desired outcomes are defined, the CEA method can help identify the least costly option for achieving that goal. An example is the restoration of coastal areas in Louisiana (Caffey, 2014), where dredge-based "marsh creation" (involving essentially the establishment of a wetland) and diversion-based coastal restoration (where built capital was used to restore and protect coastal areas) projects were compared. A cost effectiveness analysis showed that the marsh creation approach provided similar benefits at lower cost.

The ultimate aim is to assess all three areas of impact (social, economic and environmental), where feedback loops across value chain stages are identified and assessed to capture the vulnerability and risks of the eco-agri-food value chain, as well as risks for society. The TEEBAgriFood Evaluation Framework is laid out as the direction for future work. The intention is that, based on this report and future pilot studies, case studies can be developed.

The tool is widely used in many sectors, including agriculture. It has the advantage of not needing explicit benefit estimates, but the corresponding limitation that it is based on the assumption of a given physical goal as desirable. Once a social decision has been taken to make a certain investment (e.g. protect land from the consequences of a 1:100 year flood) the method is frequently used to compare different methods to achieving that goal. Complications arise, however, when the goal has broader social consequences, some of which have benefits and others may have costs. These have to be taken into account for the method to be effective but that comes down to valuing some of the benefits associated with the action, which was what the method was designed to avoid.

Box 7.13 Steps involved in evaluating eco-agri-food systems ('value chain analysis')

1. Set out the different stages of the value chain to be analyzed.

2. For each stage, identify the key social, economic and environmental indicators of performance.

   - Based on these indicators, identify economic impacts as well as the externalities and those relating to merit or demerit goods. The economic impact assessment not only serves to get the value-added at each stage but also includes who benefits from the production and where the costs are incurred.

   - Identify the social and environmental impacts that are desirable and that emerge as side effects, being both direct, indirect and induced impacts of economic activities. Estimate, when possible, economic values for these impacts.

Assess how these key indicators of performance are interconnected with each other. This can be done by developing a Causal Loop Diagram (CLD) (see, for example, Figure X), or a map of the system analyzed. In addition to the causal relations, space is important. As a result, the location of impacts is crucial (e.g. the proximity of economic activities to a river, and how the local population relies on such water are critical elements).

Carry out an assessment of the impact of economic activities, under various scenarios of policy interventions and practices utilized. This comprises the preparation of an assessment that considers simultaneously the social, economic and environmental impacts of economic activities, and the economic valuation of social and environmental externalities.

7.4.6 Multi-Criteria Analysis

Multi-Criteria Analysis (MCA) expands the boundaries of the analysis beyond cost benefit or cost effectiveness results and allows the assessment of projects against a variety of criteria, including quantitative and qualitative indicators. In contrast to CBAs and CEAs, MCAs can be conducted in cases where multiple objectives and decision criteria exist (e.g. economic growth, employment creation and emission reduction). An example of the use of MCA related to agriculture was done by UNEP (2011) where a series of studies were conducted to evaluate adaptation options to deal with climate change. In the case of agriculture, the method took into account climate change impact as well as other factors[9]. Options considered were classified under the following categories: market-based financial instruments (21), public investment programs (18), regulatory instruments (11), information based instruments (16) and international cooperation programs (7). Each of the 73 individual options was evaluated with respect to criteria grouped in the following sets: public financing needs, implementation barriers, climate related benefits, economic benefits, environmental benefits, social benefits and political and institutional benefits. Using these to generate 19 criteria, each option is scored, using both objective and subjective scoring systems, and the scores are weighted and added to arrive at an overall score. The method was applied to a case study in Yemen. Governments around the world have used MCA to assist in evaluating projects and policies that have complex socio-economic and environmental impacts that are often hard to measure in monetary terms.

The main limitations to MCA relate to selecting which criteria to include and what weights to give to the different criteria; both can greatly impact the results of the exercise. It can also be difficult to convince policy makers of rankings based on MCA, which they may see as having a major subjective component.

In practice, all decisions relating to projects or policies involve policy makers taking account of multiple criteria, of which the benefits and costs as reported under a CBA would be one. They do not often employ formal MCA methods, however, and the process of arriving at a decision remains a political one. Almost always, policy makers will want CBA as part of their information set and in recent years we have seen the boundaries of CBAs expand, reaching closer to those of MCAs. This is the case of integrated or extended CBA (UNEP 2016), where externalities (social and environmental, as well as indirect and induced project outcomes, such as employment and income creation) are monetized and included in the assessment of the financial viability of projects[10]. The CBA method has also been used to include distributional considerations through the use of "weights" so transfers to a poor person are given a higher weight than the same transfer to a rich person or where employment has a

9 The case study is available at www.mca4climate.info.

10 See for instance: www.iisd.org/project/SAVi-sustainable-asset-valuation-tool

direct additional benefit, thus reducing the labour cost of the project. If a project is being evaluated by developers or investors some factors such as distributional weights would not generally be used, but if it is being analysed by someone in the public sector, evaluating the options on behalf of society, then such weights would be relevant, as would all the externalities.

7.5 MODELLING TOOLS AND TECHNIQUES

Chapter 2 of this volume presents the rationale for using a systems approach to analyse the eco-agri-food system. In this section, several modelling techniques that can be used to carry out such systemic analysis are reviewed and discussed. These models can make use of the valuation techniques presented in Section 7.3 and can also be used to support the evaluation methods described in Section 7.4. For instance, simulation models can be utilized to estimate the total investment required to implement a project or reach a stated policy target, and to forecast the impact of such interventions on various indicators of interest, such as land cover. Subsequently, these results can be used to assess the economic viability of the investment (i.e. Cost Benefit Analysis). Specifically: i) the investment amount can be used as a direct input for the CBA; ii) the impact on land cover can be used to determine the extent to which ecosystem services are gained or lost, and also to determine the economic value of resulting change in ecosystem services. The latter value can be used as input to the CBA, as a potential avoided cost. An example is provided below.

The list of models reviewed here is not exhaustive. There is a large and growing literature on complex systems, and on the use of modelling approaches to analyse specific geographical contexts. Emerging approaches include Agent Based Models, which assess the ways in which economic agents (e.g. farmers, or economic actors in the eco-agri-food value chain) behave under various scenarios. With this in mind, we believe that our framework can help identify what should be included in more comprehensive modelling approaches and how the results from different approaches should be interpreted.

7.5.1 Land use and biophysical models

Biophysical models help planners decide how to manage the land and draw long-term plans for development, including the location of different activities and their impact on land, ecosystems and people. Such models can be a key input into the valuation of ecosystem services related to agriculture (see Section 7.4.1) and, in the case of land use models, spatial data are sometimes used as an input for the estimation and economic valuation of present and future ecosystem services. Products are often highly visual (e.g. maps, graphs, diagrams, and charts) but considerations of social and economic variables are in most cases qualitative.

Biophysical models require several types of data, often spatially explicit. Examples include data on land cover and on physical flows, both regarding inputs and outputs to production or other natural processes. For instance, in the context of water-related studies, data are required to estimate the supply of water (e.g. precipitation, evapotranspiration, percolation) and its consumption (e.g. land cover by type and by crop, specific daily or monthly water requirements by crop, population and resulting water consumption for sanitation). Estimating ecosystem services requires additional information, depending on the assessment. Examples include maps on soil and vegetation types, multipliers for carbon sequestration, by land cover and vegetation type. The availability of data for biophysical models is improving, especially from international databases (e.g. Group on Earth Observations, EXIOBASE[11]). On the other hand, issues often arise in relation to the (low) resolution of maps and the validation of data on the ground (required to ensure the accuracy of the data extracted from the map). As a result, local validation is required, or customization of the model should be performed to better capture the local context.

A few examples are provided, on spatial planning, water supply and water requirements, and on the estimation of a variety of ecosystem services.

Spatial planning tools

Marxan and IDRISI Land Change Modeler are land use models, and are used to plot out optimal physical placement of economic activities, human settlements and other land uses. Practically, through the identification of trends (e.g. for population) and/or the use of assumptions for future land use change (e.g. land use per person), these models generate future land cover maps that optimize placement in space (e.g. with population being located close to urban centres or to infrastructure, or with agriculture land being in located in the most productive areas depending on soil types and water availability, or with the minimization of forest loss, and hence decline in carbon sequestration capacity and biodiversity loss). These models allow users to modify a specific set of parameters (e.g. hectares of land cover by type, or their determinants, such as population growth), but often do not include consideration to what the assumed/forecasted land use change means for socioeconomic effects or monetary valuation of loss/gain in natural capital assets.

11 For more information see www.geoportal.org/ and www.exiobase.eu/

Water supply and water requirements (CROPWAT and SWAT)

CROPWAT is a decision support tool developed by the Land and Water Development Division of FAO[12]. It facilitates the calculation of crop water requirements and irrigation requirements based on soil, climate and crop data. Concerning its application, CROPWAT informs the development of irrigation schedules for different management conditions and the calculation of required water supply for varying crop patterns. An example of the application of CROPWAT in Africa is done by Bouraima (2015) in Benin, where they estimated the crop reference and actual evapotranspiration, and the irrigation water requirement of Oryza sativa in the sub-basin of Niger River of West Africa.

The Soil and Water Assessment Tool (SWAT) is a river basin scale model developed to quantify the impact of land management practices in large, complex watersheds. SWAT is a continuous time model that operates on a daily time step at basin scale (Texas A&M University 2015). SWAT was developed to predict the impact of land management practices on water, sediment and agricultural chemical yields in large complex watersheds with varying soils, land use and management conditions over long periods. It can be used to simulate at the basin scale water and nutrients cycle in landscapes whose dominant land use is agriculture. It can also help in assessing the environmental efficiency of best management practices and alternative management policies.

Integrated Valuation of Environmental Services and Trade Offs (InVEST)

The Integrated Valuation of Environmental Services and Trade Offs (InVEST)[13] is a family of models developed by the Natural Capital Project that quantifies and maps environmental services and supports (if required) their economic valuation using the techniques described above. InVEST is designed to help local, regional and national decision-makers incorporate ecosystem services into a range of policy and planning contexts for terrestrial, freshwater and marine ecosystems, including spatial planning, strategic environmental assessments and environmental impact assessments. There is also some discussion about applying InVEST to corporate level activities.

Artificial Intelligence for Ecosystem Services (ARIES)

ARIES is a web-based model that assists rapid ecosystem service assessment and valuation (ESAV)[14]. ARIES helps users discover, understand, and quantify environmental assets and the factors influencing their values, for specific geographic areas and based on user needs and priorities. ARIES encodes relevant ecological and socioeconomic knowledge to map ecosystem service provision, use, and benefit flows.

Multi-scale Integrated Models of Ecosystem Services (MIMES)

Scientists at the University of Vermont's Gund Institute developed the Multi-scale Integrated Model of Ecosystem Services (MIMES) for Ecological Economics[15]. MIMES uses a systems approach (in that it considers entire ecological systems, but not social and economic dynamics) to model changes in ecosystem services across a spatially explicit environment. The model quantifies the effects of land and sea use change on ecosystem services and can be run at global, regional, and local levels.

Strengths and limitations

There are several advantages to using biophysical models (see **Table 7.7**). First, they allow the analyst to estimate, and fully consider, the characteristics of a landscape, region or country and its carrying capacity. Second, the use of spatially explicit datasets and the generation of maps, allows visualization of past and future trends, and better estimates of the value of the ecosystem services that may be gained or lost.

Among the limitations is the lack of social and economic dimensions to the analysis, for which spatial data are generally less available and thus impact can only be inferred and not estimated directly. Furthermore, the analysis of land use changes and the resulting need for inputs to production (e.g. water) does not normally include the analysis of endogenous feedback loops, rendering the analysis comparatively static. In other words, the analysis does not consider that the expansion of agricultural land may lead to an increase in population, which may result in water consumption being higher than expected, and hence affect irrigation requirements and land productivity. As a result, the use of biophysical and spatially explicit models is primarily for scenario analysis rather than for supporting policy formulation and evaluation, where the anticipation of side effects is crucial. Finally, many of the parameters of the models are unknown and educated guesses have to be made about their values. This often makes the results they generate lacking in empirical data, a factor that highlights the strength of these models in policy formulation (where possible targets are set), rather than in policy assessment (where specific provisions are identified, and where a more in-depth assessment of local dynamics is required).

12 For more information, see: www.fao.org/land-water/databases-and-software/cropwat/en/

13 For more information, see: www.naturalcapitalproject.org/InVEST.html

14 For more information, see: aries.integratedmodelling.org/

15 For more information, see: www.afordablefutures.com/services/mimes

Table 7.7 Potential contribution of biophysical models to the assessment of the sustainability of the agri-food system (Source: authors)

Capital Base stocks	Produced capital	
	Human capital	
	Social Capital	
	Natural capital	Fully includes various types of natural capital stocks (e.g. soils, water resources, biodiversity)
Flows through the value chain	Capital input flows	Includes the estimation of ecosystem services (e.g. water provisioning) that could be used as input to production
	Ag and food goods and services flows	Estimates the output of agricultural activities (e.g. crop production)
	Residual flows	Estimates residual flows, such as ecosystem services affected by production (e.g. N&P and water quality)
Outcomes	Economic	
	Health	
	Social	
	Environment	Estimates changes to natural capital (e.g. deforestation)
Value chain impacts		
Spatial disaggregation		Spatially disaggregated, at the level of using GIS maps

7.5.2 Partial equilibrium models

At their simplest level, Partial Equilibrium (PE) models can be conceptualized as the interaction of supply and demand in a single market. PE models are a family of models that cover a single sector, generally at a high level of detail when compared to economy-wide models (e.g. CGE models). They range from single-sector single-company, or up to country models or single-sector multi-country models (FAO 2006). PE models typically use a "bottom-up" approach, placing emphasis on specific policy interventions (e.g. fiscal policies) or technology adoption. In both cases, PE models estimate the impact of such interventions on demand and production in a given sector.

Based on the new situation (policy scenario) and specific formulations and parameters explaining the strength of the relationship between demand and supply (i.e. elasticities), the PE model calculates a new equilibrium for the sector and provides output on a range of indicators (FAO 2006). With this background, several studies have expanded the boundaries of PE models to consider the indirect and induced impacts of production, with the goal to support policy and investment impact assessment. As an example, Callaway and McCarl (1996) compared the fiscal and welfare costs of achieving specific carbon targets through afforestation, and examined the welfare, fiscal, and carbon consequences of replacing existing farm subsidies, wholly or in part, with payments for carbon.

In addition to the detailed presentation of variables in the sector analysed, coverage of environmental, economic and social indicators can also be found in PE models. An example involving both economic and environmental aspects would be the application of pesticides. Estimating the damage done by different products is undertaken, often as part of a risk assessment, in which the risks are traded off against the benefits from the application. Certain products considered as highly toxic (e.g. endocrine disruptors) may be banned in certain locations if impacts are found to be present. In other cases, products may be permitted but with limitations on quantity, season etc. A review of the economic issues is given in Fernandez-Cornejo *et al.* (1998).

Partial equilibrium models generally require detailed information on a given sector, including: i) economic accounting for revenues and costs of production, ii) knowledge of production inputs (e.g. employment and labour cost, energy consumption and related expenditure, capital and material inputs and required investment), iii) information on key determinants of demand and supply (e.g. the responsiveness of demand to price changes) and iv) knowledge of the cost of interventions (e.g. technology investments) and their effectiveness. In the case of eco-agri-food system models, information for the estimation of revenues would be required on agriculture land, yield and prices, and concerning costs on infrastructure (e.g. mechanization and irrigation), labour, water and other inputs (e.g. energy, fertilizers and pesticides). When considering the value chain, additional data would be required on transport costs and the capacity to process food, including the revenues and costs (and their main determinants) of food processing. Given their high degree of customization, PE models, when data are available, can include a high degree of detail for the sector analysed.

Strengths and limitations

The advantage of PE models, which represent a piecemeal approach (in that these models focus only on part of the whole eco-agro-food process) is that the model can be highly customized and that the analysis is comparatively transparent, being tractable and relatively easy to carry out (see **Table 7.8** for their potential contribution to agri-food systems). In fact, detail can be added more easily than with macroeconomic (e.g. CGE) models. Further, data requirements are normally not extensive, and the model can be structured according to the availability of data. Conversely, the estimation of economic impacts across the whole value chain can be complex, spanning across several economic activities and disciplines of research, and data are not easy to obtain, interpret and use. As a result, if the item of interest is a particular activity (e.g. farm-related non-point pollution) it may be reasonable to focus on that component only.

The main limitation of PE models regards its sectoral and primarily economic focus, and whether assessing the impacts of policies and investments in isolation from other stages of the value chain (or in isolation from the sector and the economy as a whole) is reasonable, accurate and realistic. For instance, a technological breakthrough that lowers the cost of sugar production from cane may increase production and result in land clearance and other environmental impacts, which would be analysed as part of that process. But the lower costs of sugar production would also lower the costs of sugar as an input in the eco-agro-food process, making high sugar products cheaper and increasing problems of obesity and type II diabetes. This would normally not be considered in a partial equilibrium analysis that focuses on sugar production. This is because a PE analysis does not consider feedback effects, from the macro to the sectoral level. Similarly, given their limitation in addressing system-wide dynamics, PE models are not the best option to assess social equity concerns. While these models allow for the estimation of aggregate employment and income-related impacts, they generally fail to describe detailed distributional impacts of policy interventions and investments.

Table 7.8 Potential contribution of Partial Equilibrium models to the assessment of the sustainability of the eco-agri-food system (Source: authors)

Capital Base stocks	Produced capital	Includes capital stocks (e.g. assets), both in physical and monetary terms
	Human capital	
	Social Capital	
	Natural capital	May include certain types of natural capital stocks (e.g. land)
Flows through the value chain	Capital input flows	Generally includes infrastructure, labour inputs and certain ecosystem services
	Ag and food goods and services flows	Considers both inputs and outputs
	Residual flows	Can estimate both waste and other residuals
Outcomes	Economic	Estimates value added, taxes, subsidies and possibly wages, also considering trade dynamics
	Health	
	Social	
	Environment	Can estimate changes to natural capital (e.g. deforestation, affecting land cover)
Value chain impacts		It can include various stages of the value chain
Spatial disaggregation		

7.5.3 Computable General Equilibrium (CGE) models

A general equilibrium approach models supply and demand across all sectors in an economy. Analysis is typically conducted using computable general equilibrium (CGE) models (see, for instance, Lofgren and Diaz-Bonilla [2010]). CGE models are a standard tool of analysis and are widely used to analyse the aggregate welfare and distributional impacts of policies whose effects may be transmitted through multiple markets, or contain menus of different tax, subsidy, quota or transfer instruments (Wing 2004).

CGE models utilize input-output tables (Leontief, 1951), which can also be utilized as standalone models for more static analysis, and which represent inputs and outputs of several economic activities (e.g. the amount of labour, energy and material input required to produce a unit of production output). Equations are estimated that explain the relationship between inputs and outputs of a given process, or sector (e.g. how much energy is required for a unit of output, given the use of a specific technology in the production process). In other words, the model uses productivity multipliers that serve for the calculation of the output values given a specific set and quantity of inputs, or it estimates the required inputs for a given value of output (Tcheremnykh 2003). While being most often primarily focused on economic flows, CGE models have in several cases been extended to include environmental impacts of production and consumption on water, land and air. As a result, these models can assess the impacts of changes such as climate or trade liberalisation on outputs and prices across all sectors as well as on the incomes of different groups in society.

There are numerous applications focusing on the agricultural sector that use such models, for instance, the effect of climate change and water scarcity on crops and livestock, as well as on the income of poor groups in society. See for example Skoufakis *et al.* (2011), or the MAGNET model of the European Commission, which has been used to assess the impacts of agriculture, land-use and biofuel policies on the global economy (Boulanger *et al.* 2016). Other applications for the agriculture sector include the assessment of socio-economic impacts of improving agriculture water use efficiency (Liu *et al.* 2017), analyzing climate change related impacts on water availability and agriculture production (Ponce *et al.* 2016), and the estimation of the outcomes of public investments in irrigation infrastructure and training agriculture labour (Mitik and Engida 2013).

CGE models optimize utility for economic actors, and the three conditions of market clearance, zero profit and income balance are employed to solve simultaneously for the set of prices and the allocation of goods and factors that support general equilibrium. Practically, this means that CGE models assume that the demand and supply for a product and service always match, through the identification of a price that satisfies both consumers and producers. As opposed to partial equilibrium models, CGEs are in general 'top-down', meaning that variables such as food production are determined by parameterised equations (e.g. balancing demand and supply through prices), rather than considering individual technologies. The underlying assumption is that if there is demand (e.g. through consumption), there will be production as well. Bottom up models estimate instead what production level is feasible and at what costs, depending on the technology available and utilized.

CGE models require a large amount of detailed data on across all economic sectors, including factors of production and international trade. Traditional data inputs for CGE models are the Social Accounting Matrices (SAMs), and the System of National Accounts (SNA).

Strengths and limitations

The main advantages of CGE models include the estimation of direct and indirect impacts of policy interventions and investments, and the use of an economy-wide approach. As a result, interdependences across sectors, and countries, are taken into account. The variables included in CGE models are, among others, sectoral consumption and production, wages, household income and inflation, as well as trade. Nowadays most agricultural sector analysis involving taxes or subsidies or changes in trade regimes would make use of CGE models. This results in CGE models being used very often to assess equity impacts, especially in terms of income distribution across income classes and employment groups. On the other hand, CGE models do not generally support the assessment of non-monetary dimensions of equity, such as access to services and resources. CGE models are useful in examining the relationship between climate change and agriculture, where increases in temperature and precipitation are expected to lower yields for some crops by significant amounts. The size of the effect varies from one region to another and with trade the implications for price and welfare in different regions will vary. Among the key factors are the relevance of the sector in the economy (e.g. production and contribution to GDP, as well as employment), its reliance on trade and exposure to changing weather conditions, the extent to which support is provided through subsidies (Randhir and Hertel 2000), and the relevance of a given food product in household consumption (Hertel *et al.* 2010). **Table 7.9** lists the potential contribution of CGE models to the assessment of the sustainability of food systems.

CGEs have significant limitations. First the modelling is complex and depends on a number of parameters whose values are uncertain. This emerges for instance when data are not available, but also when the underlying input-output tables and the Social Accounting Matrix, which are often generated every five or ten years, are outdated (e.g. when policy analysis is required for the period 2018-2025, but the underlying data are from the year 2012). Hence the results have a high level of uncertainty. Second, the level of detail of CGE models is often not adequate to support the analysis of sectoral dynamics in detail. Third, CGE models often suffer from the lack of supply-side constraints (especially physical ones), in that they assume that extra output can be achieved and that scarcity is not a concern (Gretton 2013). In reality the boundaries of the analysis should be expanded to account not only for the availability of labour and capital, but for natural resources as well. Practically, CGE models lack the explicit representation of biophysical stocks and flows and rely on underlying assumptions on equilibrium and the maximization of welfare that may not represent reality.

Table 7.9 Potential contribution of CGE models to the assessment of the sustainability of the eco-agri-food system (Source: authors)

Capital Base stocks	Produced capital	Includes capital stocks (in monetary terms)
	Human capital	Includes labour productivity
	Social Capital	
	Natural capital	Models for agriculture would include land cover
Flows through the value chain	Capital input flows	Includes capital and labour, models focused on agriculture may include certain ecosystem services
	Ag and food goods and services flows	Considers both inputs and outputs, generally with less detail than PE models
	Residual flows	Could include GHG emissions
Outcomes	Economic	Estimates value added, prices, taxes, subsidies and wages, also considering trade dynamics
	Health	
	Social	Estimates impacts on consumption and income for various household groups
	Environment	
Value chain impacts		It can include various stages of the value chain
Spatial disaggregation		Spatial disaggregation is found for multi-country models, at the national level

7.5.4 System Dynamics (SD)

Systems Thinking (ST) is a methodology for "seeing systems" and assessing policy outcomes across sectors and actors, as well as over time (Meadows 1980; Randers 1980; Richardson and Pugh 1981; Forrester 2002). ST can help to assess how different variables in a system interact with each other to shape trends (historical and future). While Systems Thinking is qualitative, System Dynamics is a quantitative methodology. In fact, it aims to define causal relations, feedback loops, delays and non-linearity to represent the complex nature of systems (Sterman 2000). It does so by running differential equations over time (i.e. representing time explicitly, with days and months). In contrast to CGE and PE models, System Dynamics models do not optimize the system (i.e. they do not estimate the best possible setup of the system to reach a stated goal). Instead, these are causal-descriptive models used to run "what if" simulations.

Created by Jay W. Forrester in the late 1950s, System Dynamics (SD) allows a modeler to integrate social, economic and environmental indicators in a single framework of analysis. SD models are based on the assumption that structure drives model behaviour and uses causal relationships to link variables. By way of further explanation, SD models include feedback loops (a series of variables and equations connected in a circular fashion). The feedback loops generate non-linear trends that ultimately determine the trends forecasted. This is what is meant by saying "structure" (i.e. the variables and, more importantly, the feedback loops in the model) determine "behaviour" (i.e. the trends forecasted over time). In all other modelling approaches that are linear (i.e. with no feedback loops), the "behaviour" is primarily driven by the data used (not by the equations, or the structure of the model).

SD approaches provide a more explicit representation of the factors driving demand (e.g. population divided by age cohorts, income divided by household group, and prices) and supply (for agriculture production these factors include land productivity as affected by soil quality, mechanization, labour, production inputs, water availability and weather conditions), merging biophysical and economic indicators as stocks and flows. The complexity of a system is represented using Causal Loop Diagrams (CLD) and models can be customized to analyse the socioeconomic implications of different actions across sectors (social, economic and environmental) and actors (e.g. households, private sector and the government), within and across countries.

A CLD can be used to explore and represent the interconnections between key indicators in the sector or system of interest (Probst and Bassi 2014). Examples are shown in **Figure 7.5** as well as **Figure 2.6** in Chapter 2. John Sterman states, "A causal diagram consists of variables connected by arrows denoting the causal influences among the variables. The important feedback loops are also identified in the diagram. Variables are related by causal links, shown by arrows. Link polarities [a plus or minus sign indicating the positive or negative causality between two variables] describe the structure of the system. They do not describe the behaviour of the variables. That is, they describe what would happen if there were a change. They do not describe what actually happens. Rather, it tells you what would happen if the variable were to change" (Sterman 2000). The creation of a CLD has several purposes: first, it combines the team's ideas, knowledge, and opinions; second, it highlights the boundaries of the analysis; third, it allows all the stakeholders to achieve basic-to-advanced knowledge of the dynamics underlying the sector or system analyzed.

The pillars of SD models are feedback, delays and non-linearity.

- 'Feedback is a process whereby an initial cause ripples through a chain of causation ultimately to re-affect itself' (Roberts *et al.* 1983). Feedbacks (also called feedback loops in systems modelling) can be classified as positive or negative. Positive (or reinforcing) feedback loops amplify change, while negative (or balancing) counter and reduce change.

- Delays are characterized as "a phenomenon where the effect of one variable on another does not occur immediately" (Forrester 2002). Sometimes becomes difficult to attribute certain effects to specific causes, as cause and (perceived) effect are distant in time. For example, when there is an increase in the use of fertilizers, it takes time for nitrogen and phosphorous to reach water bodies and negatively impact the ecological integrity of a bay or river basin.

- Non-linear relationships cause feedback loops to vary in strength, depending on the state of the system (Meadows 1980), and determine how structure defines behaviour. For instance, with agriculture yield being influenced simultaneously by the type of seeds used, nutrients, climate, and land use practices, each embedded in a variety of feedback loops, non-linear behaviour emerges from the model.

SD models inform policy formulation and assessment, and also monitoring and evaluation. By running "what if" scenarios, SD can inform policy measures that may improve several indicators at once (e.g. providing affordable food supply while generating employment and reducing forest loss), rather than estimating the optimal policy package. Turner *et al.* (2016) conclude that SD provides a useful framework for assessing and designing sustainable strategies for agriculture production systems. Typical applications include the analysis of systemic challenges for smallholder farmers and conservation agriculture in South Africa (Von Loeper *et al.* 2016), and the assessment of policy interventions in the context of national Green Economy Strategies (Deenapanray & Bassi 2014; Musango *et al.* 2014; UNEP 2015).

SD models typically need data on socioeconomic and environmental variables, depending on the boundaries of the model. Practically, more data across social, economic and environmental indicators are required than in the case of other modelling approaches, but the level of depth and disaggregation of the data is lower than what is normally required by biophysical, partial and general equilibrium models. These data are sourced from multiple disciplines and databases and checked for consistency (or harmonized) for inclusion in the integrated model. Further, it is worth noting that SD models start simulating in the past (e.g. year 2000) and, unlike other methodologies (e.g. econometric modelling), rely on historical data only for the parameterization of the simulation model, not for the creation of forecasts. In other words, while econometric models investigate the correlation among historical time series to determine how future trends may be shaped, correlation factors in SD models are not an input for simulations; instead, these emerge from the simulation of endogenous feedback loops (based on causality) and exogenous parameters (Sterman 2000).

Strengths and limitations

The main strengths of SD include the ability to estimate strategy and policy impacts for a specific project or policy and for society, and how these impacts unfold dynamically over time. In fact, the simulation of scenarios with quantitative systems models allows decision-makers to evaluate the impact of selected interventions within and across sectors as well as economic actors, using social, economic and environmental performance indicators (both stocks and flows). Second, the simulation of causal descriptive models helps to simplify the complexity of the eco-agri-food system (because it more transparently shows all the relationships existing across modelled variables, and how changes in one variable are reflected in all the others), and can evaluate the short vs. longer-term advantages and disadvantages of the analysed interventions. In other words, it reduces complexity. Third, a causal descriptive model can capture new and emerging trends (or patterns of behaviour) emerging from the strengthening (or weakening) of certain feedback loops, and help identify potential side effects and additional synergies. This is particularly useful in assessing physical and economic impacts, and how these are interconnected (such as in the case of access to resources and services). In other words, SD models can estimate the strength of a feedback loop and forecast changes that may emerge in the future. For instance, the price of a limited resource may be low when such resource is abundant. As a result, the balancing feedback loop that leads to resource efficiency would be weak (i.e. the resource is so cheap that investments that improve resource efficiency may not be bankable). On the other hand, as consumption increases in the future and the stock of such resource declines, its price would increase. In this situation the balancing feedback loop of resource efficiency would become stronger, because a higher price justifies investments that reduce resource consumption. Practically, SD models can forecast whether feedback loops that were weak in the past may gain strength in the future, and whether feedback loops that were strong in the past may become weak in the future.

There are also limitations to the use of SD models. First, the effectiveness of a CLD and SD model is directly related to the quality of the work and the knowledge that goes into developing them. Two aspects need to be considered: the source of the knowledge embedded in the model, and the skills of the modelling team. On the former, multi-stakeholder perspectives should be incorporated and cross-sectoral knowledge is essential to correctly identify the causes of the problem and design effective interventions. In addition, the selection of relevant variables and the way in which they are mapped (most often in a group model building exercise) is crucial. On the skills of the modelling team, building valid SD models requires extensive experience to develop a sufficiently detailed and representative description of the system (i.e. the dynamic hypothesis). The lack of experience increases the difficulty to correctly identify and estimate the underlying feedback structure of the system. A second limitation of SD models is the correct identification of boundaries of the system, not an easy task. Errors in identifying the boundaries of the model (i.e. what variables and feedback loops to include/exclude) may lead to biased assessments of policy outcomes, overstating or underestimating some of the impacts across sectors and actors. Third, SD models are highly customized, and are better suited for use in a specific geographical context. In other words, this is not an ideal approach for assessing trade dynamics among several countries; it is an approach better suited to analysing national dynamics, and possibly linkages between two or three countries. It is not well suited to carry out assessments on trade that involve five or more countries. Finally, concerning implementation, the development of a SD model requires a substantial amount of interdisciplinary knowledge. The data needs depend on the level of detail being modelled and increase with every new subsystem that is added. As a result, SD models are generally focused on horizontal integration (i.e. across sectors) rather than vertical integration (i.e. adding sectoral detail). As a result, SD models are weaker than CGE models in the analysis of the distributional impacts of policy intervention, generally including less detail on economic activity, household and income groups.

Table 7.10 Potential contribution of System Dynamics models to the assessment of the sustainability of the eco-agri-food system (Source: authors)

Capital Base stocks	Produced capital	Includes capital stocks (e.g. assets), both in physical and monetary terms
	Human capital	Includes labour productivity
	Social Capital	Can include qualitative indicators representing governance and accountability
	Natural capital	Can include several stocks of natural capital
Flows through the value chain	Capital input flows	Includes capital and labour, as well as ecosystem services
	Ag and food goods and services flows	Considers both inputs and outputs, generally with less detail than PE models
	Residual flows	Can estimate both waste and other residuals
Outcomes	Economic	Estimates value added, taxes, subsidies and wages, within a specific geographical context (e.g. trade dynamics across countries are normally not captured)
	Health	Can include nutrition and diseases
	Social	Can estimate impacts on consumption and income, and access to ecosystem services, but with less detail than CGE models
	Environment	Can estimate changes to natural capital (e.g. deforestation, affecting land cover)
Value chain impacts		Possible, but with a lower degree of disaggregation when compared to PE and CGE models
Spatial disaggregation		Spatial disaggregation is found, mostly at sub-national level (e.g. provinces)

Table 7.10 summarizes the key contribution of the methodologies and models reviewed to the analysis of the sustainability of the eco-agri-food system. The rows of the table are elements of the evaluation framework presented in Chapter 6. More details for each technique follow, with an overview of their strengths and weaknesses and applicability to the eco-agri-food system.

Table 7.11 links the analytical tools used in the evaluation of eco-agri-food systems to the systemic approach presented in Chapter 2, and the capital accounting framework laid out in Chapter 6 and developed by the UN in its Inclusive Wealth Report (UNU-IHDP and UNEP 2014). The models use, in different ways, data on the stocks of produced human, social and natural capital as well as data on changes in these stocks through flows. Policies and actions then estimate the outcomes that track changes in economic, health, social and environmental indicators.

Table 7.11 Overview of the main characteristics of the modelling techniques reviewed, in relation to the evaluation framework (Source: authors)

		Land use and biophysical models	Partial Equilibrium	Computable General Equilibrium (CGE)	System Dynamics (SD)
Capital Base stocks	Produced capital		Includes capital stocks (e.g. assets), both in physical and monetary terms	Includes capital stocks (in monetary terms)	Includes capital stocks (e.g. assets), both in physical and monetary terms
	Human capital			Includes labour productivity	Includes labour productivity
	Social Capital				Can include qualitative indicators representing governance and accountability
	Natural capital	Includes various types of natural capital (e.g. soils, water resources, biodiversity)	May include certain natural capital stocks (e.g. land)	Models for agriculture would include land cover	Can include several stocks of natural capital
Flows through the value chain	Capital input flows	Includes the estimation of ecosystem services (e.g. water provisioning) that could be used as input to production	Generally includes infrastructure, labour inputs and certain ecosystem services	Includes capital and labour, models focused on agriculture may include certain ecosystem services	Includes capital and labour, as well as ecosystem services
	Ag and food goods and services flows	Estimates the output of agricultural activities (e.g. crop production)	Considers both inputs and outputs	Considers both inputs and outputs, generally with less detail than PE models	Considers both inputs and outputs, generally with less detail than PE models
	Residual flows	Estimates residual flows, such as ecosystem services affected by production (e.g. N&P and water quality)	Can estimate both waste and other residuals	Could include GHG emissions	Can estimate both waste and other residuals

Outcomes	Economic		Estimates value added, taxes, subsidies and possibly wages, also considering trade dynamics	Estimates value added, prices, taxes, subsidies and wages, also considering trade dynamics	Estimates value added, taxes, subsidies and wages, within a specific geographical context
	Health				Can include nutrition and diseases
	Social			Estimates impacts on consumption and income for various household groups	Can estimate impacts on consumption and income, and access to ecosystem services, but with less detail than CGE models
	Environment	Estimates changes to natural capital (e.g. deforestation)	Can estimate changes to natural capital (e.g. deforestation, affecting land cover)		Can estimate changes to natural capital (e.g. deforestation, affecting land cover)
Value chain impacts			It can include various stages of the value chain	It can include various stages of the value chain	Possible, but with a lower degree of disaggregation when compared to PE and CGE models
Spatial disaggregation		Spatially disaggregated, at the level of using GIS maps		Spatial disaggregation is found for multi-country models, at the national level	Spatial disaggregation is found, mostly at sub-national level (e.g. provinces)

7.6 AN INTEGRATED MODELLING APPROACH FOR THE ECO-AGRI-FOOD SYSTEM

In order to carry out an assessment of the social, economic and environmental impacts of production and consumption in the eco-agri-food system, knowledge integration is required. No single model can address all the needs of various stakeholders, some of which are concerned with macroeconomic trends (e.g. employment creation at the national level) while others are more preoccupied with localized impacts (e.g. nutrition and water quality). The TEEB approach proposes a modelling framework that integrates several modelling approaches. In other words, it makes use of the main strengths of each approach, and by linking them it removes some of their weaknesses.

There are several gaps that need to be addressed in the way quantitative assessments are being carried out. Specifically, more systemic analyses are required in order to assess policy outcomes across sectors and actors (considering all capitals and their interdependencies), as well as over time. Such analyses would allow the analyst to anticipate the emergence of side effects, leading to the formulation of complementary policy intervention, and ultimately resulting in improved resilience and sustainability of the eco-agri-food system.

Mainstream modelling approaches are typically designed to answer a specific policy question, and, in order to excel in one task; these models simplify the complexity of the system. In the context of TEEBAgriFood, this highlights a disconnect between our 'systemic' thinking and available models. To ensure that the wider evaluations support the decision-making process for sustainable eco-agri-food systems effectively, emphasis should therefore now be put on the development and use of models that allow for a fuller representation of the complexity of the eco-agri-food system, including the many causes and mechanisms responsible for the emergence of problems as well as for the success (or failure) of proposed solutions.

Considering the various methods and models available to analyse the eco-agri-food system and its parts, several opportunities for using a complementary approach emerge. System Dynamics could be utilized as a knowledge integrator, incorporating the key features of various evaluation methods, and providing a systemic and dynamic view of the problem under consideration and its possible solutions. Practically, a SD model could make use of inputs from biophysical models, and integrate these with those received from economic models, possibly allowing for a spatially explicit analysis. This modelling approach would then complement the analysis carried out with input-output, partial equilibrium and general equilibrium models, providing information on both capital base stocks, flows through the value chain and outcomes. Specifically, this modelling approach can make use of the higher level of detail included in partial equilibrium models as well as of the larger detail on economic activities included in CGE models; coupling these with the explicit spatial representation of biophysical models provides an integrated assessment that includes social and environmental indicators and related dynamics. This analysis would capture feedbacks existing across social, economic and environmental indicators, better assessing policy impacts in highly interconnected and rapidly changing environments.

A high degree of customization is required to create this type of model. This is to account for: i) local circumstances, ii) the tacit and explicit local knowledge, and iii) the identification and understanding of the priorities of local decision makers. Specifically, it is crucial to use local knowledge sources in the identification of causal relations and feedback loops. Further, the analysis must provide information on indicators that decision makers deem important to increase policy impact[16] (Rouwette and Franco 2014). **Box 7.14** illustrates an application of integrated modelling to the eco-agro-food system with an example from Tanzania.

16 "Local knowledge refers to information and understanding about the state of the bio-physical and social environments that has been acquired by the people of a community which hosts (or will host) a particular project or programme." (Baines et al. 2000).

Box 7.14 Illustration of integrated modelling for the eco-agri-food system, Kilombero, Tanzania

In 2010, the Government of Tanzania launched the Southern Agricultural Growth Corridor of Tanzania (SAGCOT) initiative as a public-private partnership dedicated to ensuring food security, reducing poverty and spurring economic development in Tanzania's Southern Corridor (SAGCOT Centre 2013). TEEB launched a study to create and compare alternative quantitative scenarios for land management of the Rufiji River Basin in Tanzania, using a systems approach.

The TEEB project for Tanzania combined: i) spatial planning tools, ii) biophysical ecosystem service models, iii) socioeconomic models based on System Dynamics, and iv) nonmarket environmental valuation methods. Together, these tools and methods have been used to carry out a holistic analysis of development impacts and land-use change (planned or otherwise) and the socioeconomic implications of such change and translated these into spatial outputs. Practically, four modelling methods and tools were combined and incorporated in an integrated model.

Figure 7.5 Causal Loop Diagram (CLD) of the study area, emphasizing the impacts of implementing the SAGCOT agriculture intensification plan (Source: authors)

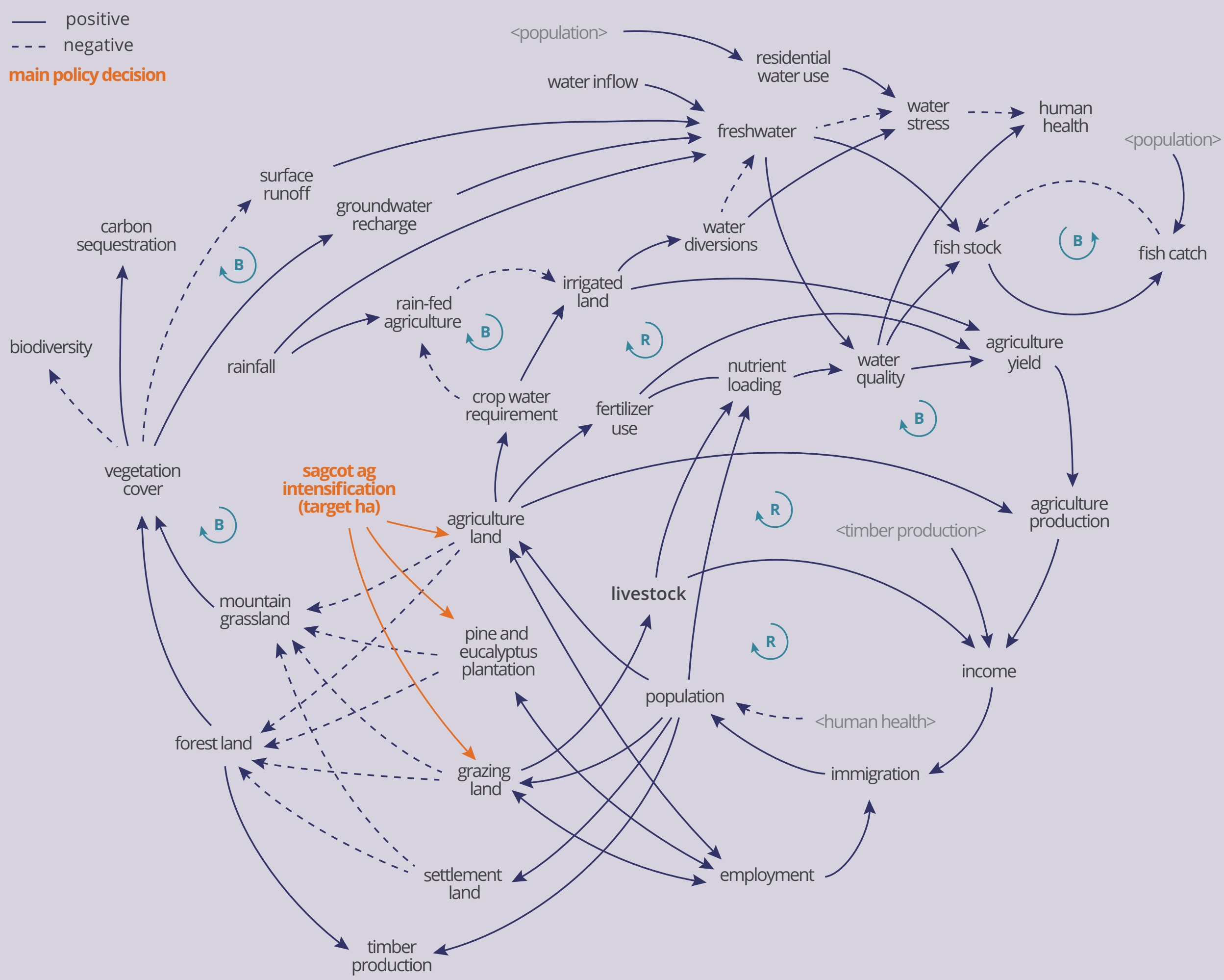


Table 7.12 Performance comparison of policy scenarios on key indicators, relative to expectations on the implementation of SAGCOT (Source: authors)

	land use	water stress	carbon sequestration	production	employment
SAGCOT	↑	↑	↓	↑	↑
water constraints	↓	↑	↑	↓	↓
water efficiency (30 per cent)	=	↓	=	↓	↓
intensification (50 per cent)	↓	=	↑	=	↓
Combination	↓	↓	↑	=	=

Given that water availability is a key enabler of agriculture production and one of the main drivers of well-being, CROPWAT was used to estimate irrigation requirements and SWAT was used to estimate water yield and runoff. In order to fully account for the potential impact of upcoming investment strategies, socio-economic analysis is also required that complements the work done with CROPWAT and SWAT. This is because population dynamics and policy responses (e.g. deforestation) can greatly affect the effectiveness of national policy. Finally, in order to inform this policy discourse, the economic valuation of ecosystem services was carried out. This is to identify and estimate the potential loss of natural capital under the baseline scenario, and as well as what could be gained under alternative scenarios.

Figure 7.5 presents the CLD that was created through a group model building exercise for representing the main drivers of change in the Kilombero basin. There are four main feedback loops that underlie the dynamics of the area studied. The first causes the expansion of agriculture land, the second loop is represented by the increase in employment that is caused by the expansion of agriculture land under policy scenarios, such as in the case of SAGCOT, the third loop highlights the relevance of vegetation (which increases groundwater recharge and lowers surface water and runoff) and the fourth shows the importance of the type of crops planted and their respective water requirements.

The analysis carried out with this suite of models, integrating biophysical and socio-economic tools, indicates that the combination of fostering cluster development, intensifying and diversifying agriculture production, and improving water efficiency allows for maintaining the positive outcomes on employment, income and production that are expected from SAGCOT, by avoiding the negative consequences related to water availability, social issues and ecosystem integrity would have in the BAU scenario (see **Table 7.12**). Coupled with sustainable agriculture practices, which would limit the use of chemical fertilizers, and thereby avoiding water pollution, this strategy would maximise the performance of the system across social, economic and environmental indicators, ensuring long term social and environmental sustainability and economic viability for the agriculture sector in the Kilombero valley[17]

17 Quantitative results are provided in the project factsheet: Managing Ecosystem Services In Rufiji River Basin: Biophysical Modeling And Economic Valuation, available at www.teebweb.org/areas-of-work/teeb-country-studies/tanzania

7.7 SUMMARY AND CONCLUSIONS

The eco-agri-food sector is of great economic and social importance. It has been subject to many changes over recent years, often with negative impacts on the environment and on vulnerable groups. At the same there have been policy initiatives to address these negative impacts and to make the system more consistent with the goals of sustainable development.

This chapter has been devoted to presenting the toolbox at our disposal to review the impacts of the functioning of the eco-agri-food sector and to enable policy makers to compare different policies and measures, especially when faced with evidence of inadequate performance of some parts of the system.

The complexity of the system must be acknowledged; agriculture not only involves the growing of crops and husbandry of livestock, but is also part of a configuration in which the activities of production, processing, distribution, consumption and waste disposal are all key components. In the past these linkages have tended to be ignored when formulating and appraising agricultural policies. The chapter shows the importance of the linkages and feedbacks between these activities and why they need to be seen as an integrated framework.

On the environmental side there is an important link between agriculture and food production and the ecosystems in which such activities are embedded. These ecosystems provide key services to the agri-food system and in turn the way in which the latter works has an effect on the ecosystems. Consequently it is important to understand these linkages, which requires an appreciation of the different ecosystem services and their relation to food production, as well as the subsequent steps in the agri-food system.

As far as the tools are concerned a distinction is made between the valuation, in monetary terms, of impacts of the agri-food system and of policies that target that sector; and a wider evaluation of the system that takes account of other factors of importance, such as equity, human health and sustainability. The monetary valuation of impacts is organized around the idea of externalities, which are made up of impacts of the eco-agri-food system that are not accounted for in market transactions. The chapter gives several examples of such externalities and ways of estimating the costs they generate on society. There are several tools at our disposal for undertaking these estimations; each has its strengths and weaknesses and each is best suited to the valuation of particular externalities.

The data collected from the estimation of externalities can be used to appraise a policy option in conjunction with tools such as cost benefit analysis, cost effectiveness analysis, partial equilibrium modelling and general equilibrium modelling. With such tools the costs of the policy and the costs associated with the externalities are combined to obtain an economic measure of the net impacts of the policy compared to the case of no policy or an alternative policy.

For the wider evaluation of the functioning of the eco-agri-food system and of different policies a number of other tools are presented. These include life cycle analysis, propensity scoring methods, value chain analysis, multi-criteria analysis, merit good assessments and system dynamics. In these cases the analyst obtains information on a range of physical impacts of a given eco-agri-food system under a given set of regulations and compares these with the impacts under an alternative set of regulations or other changes in the eco-agri-food system. Each tool has its strengths and weaknesses and is best suited to specific problems, which are discussed in the chapter.

With all the tools discussed there is a key role for the biophysical modelling of the links between different parts of the eco-agri-food system and of the ways in which these parts respond to different regulatory instruments, such as taxes or charges, subsidies, prohibitions etc. Some tools use the modelling to obtain the physical indicators that are their end product, while others use the modelling as the basis for physical values that are then valued in monetary terms. In both cases the end product is only as reliable or as effective as the underlying biophysical modelling, which is often quite weak and uncertain.

There is considerable work to be done to undertake comprehensive evaluations of different policies and measures related to the functioning of eco-agri-food systems. Ideally one should be able to say with some confidence what are the externalities associated with each euro or dollar spent on a given kind of food, produced, distributed and disposed of in a given way. We are making progress toward that goal and with the changes in practices proposed in this chapter, which lays the foundations for future work in this area and provides the analyst with an overview of the toolbox at her disposal, we may be more successful.

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