Economic viewpoints on ecosystem services

H.J. Silvis and C.M. van der Heide

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WAGENINGEN UR
For quality of life

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Abstract

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The concept of ecosystem services has been introduced to help determine the different values of ecosystems. Ecosystem services are usually divided into four categories: provisioning services, regulating services, cultural services and habitat services (previously denoted as supporting services). This overview highlights economic theories about ecosystem services, distinguishing between pre-classical economics, classical economics, neoclassical economics and modern economics. In addition, specific attention is given to two special branches of economics: (i) natural resource and environmental economics and (ii) ecological economics. Natural resource and environmental economics basically deals with a welfare economics analysis of natural resource and environmental issues, such as pollution control, natural (i.e. renewable and non-renewable) resource exploitation, and global environmental problems such as climate change.

The more recent discipline of ecological economics was launched as a new paradigm with closer ties to the natural sciences. Whereas environmental economics focuses on value dimensions (i.e., utility and welfare in theory, and costs and benefits in practice), ecological economics – as a heterodox, non-coherent school of economics – is inclined to add ecological criteria to these dimensions, to cover aspects such as productivity, stability and resilience of ecosystems. Since a proper pricing system for many ecosystem services simply does not exist, various non-market valuation techniques have been developed to elicit the value of these services. Monetary valuation of ecosystem services remains problematic however, for one thing because of the hidden value of the ecosystem structure that supports the different ecosystem services (the 'glue value').

Finally, the issue of policy analysis and design is addressed. The rationale for regulation with regard to nature and ecosystem services is that adverse risks, such as over-exploitation, are not adequately priced in markets. Welfare economics tools for evaluating policies and projects include cost-benefit analysis and cost-effectiveness analysis. From an ecological economics standpoint, multicriteria analysis, the precautionary principle and the method of safe minimum standards are topical issues. The latter two policy tools suggest that we should err on the side of caution in the face of ecological uncertainty. The advancement of knowledge in this field requires further interdisciplinary cooperation between the natural and social sciences.

Key words: ecosystem services, history of economic thought, welfare theory, market failures, policy failures, economic valuation, cost-benefit analysis, cost-effectiveness analysis, multicriteria analysis, precautionary principle, safe minimum standards

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Preface

The term Ecosystem services – the economic and societal benefits derived from nature and the landscape – emerged in recent policy papers such as the UN Millennium Ecosystem Assessment in 2005 and the European Union project entitled The Economics of Ecosystem services and Biodiversity (TEEB) in 2010. It is advocated mainly by conservationists as a concept to underline the importance of nature as a source of welfare and well-being for mankind, apart from *and* in addition to its intrinsic value. A source that is allegedly at risk.

Economists tend to think of nature rather in terms of natural resources, and this has far more often been the subject of economic thought during the last few centuries than most people realise. The neoclassical preoccupation with efficient allocation of labour and man-made capital stems mainly from the period from the industrialisation era up to the late twentieth century. Before that period and in recent decades, however, the exploitation of natural resources, has never been far from the heart of the economic discipline.

Thus, natural resources have been much more on the economists' mind than most critics are aware of. The keyword here is *scarcity*. Efficient use of scarce means of production is the economists' main concern. In pre-industrial times this concerned fertile land. Labour was available in abundance and man-made capital played a minor role. This situation changed, especially as regards man-made capital and infrastructure, during the first period of industrialisation. Later on, labour, at first abundant because of migration from the countryside towards the industrial centres, and especially trained labour, became increasingly scarce. The issue of labour–capital substitution arose. These were the two production factors that mattered most; in the new territories on the American and Australian continents, the reserves of raw materials and natural resources were still vast. All this came to an end at the closing of the last century: due to population growth and unprecedented levels of wealth and consumption, technological progress became unable to offset the pressure on the Earth's carrying capacity. 'Planet'-like problems emerged, at first mainly framed in terms of pollution and environmental issues, but in the last two decades more often in terms of ecological sustainability. The focus of economics and economists changed in response to these changing scarcities, as Chapters 3–5 of this report illustrate.

This publication discusses these changes over time and their repercussions on economic thought, ending with the recent developments and practical methods of measurement, valuation and policy design. Though covering a long period, it is pleasantly concise, while still offering a sufficiently complete overview. It provides a useful introduction for scholars from other social sciences as well as for biologists/ecologists acting as policy advisors. In addition, researchers with a primarily business-economic background and an applied science attitude will find much that is of interest to them.

Also on behalf of the authors, I would like to thank prof. dr. Wim Heijman en dr. ir. Roel Jongeneel, both from Wageningen University, for their constructive reviews of the draft document.

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Coordinator Knowledge based research for the Statutory Research Tasks Unit for Nature and the Environment.

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Contents

Introduction

1

1.1 Context

The discipline of economics tries to find out how best to fulfil people's unlimited needs and aspirations under scarce resource constraints. Without scarcity – for example, the Garden of Eden, where all external goods are available in superabundance – there are no economic problems which force people to make choices among available alternatives (Sowell, 2007). But when ecological concerns are connected to or intertwined with economics, the challenge is to meet human needs without degrading the natural environment. Or as it was more eloquently phrased by Heal and Small (2002, p. 1352): "Ecosystem services are scarce, make material contributions to economic welfare, cannot be taken for granted, and can be affected by conscious choices. These features place them within the purview of economic analysis."

In economic terms, ecosystems may be regarded as a special form of capital assets. Like reproducible capital assets (roads, buildings and machinery), ecosystems depreciate if they are misused or are overused. But ecosystems differ from reproducible capital assets in several ways. Depreciation of natural capital may be irreversible, or the systems may take a long time to recover. Generally speaking, it is not possible to replace a depleted or degraded ecosystem by a new one. And ecosystems may collapse abruptly, without much prior warning (Dasgupta, 2008).

As ecosystems are threatened by human activities, it is important to take better account of long-term ecosystem health and its role in enabling human habitation and economic activity. It is in this context that the concept of ecosystem services has been put forward to assist in assigning economic values to the role of ecosystems and designing policies for sustainable development.

In recent years, there has been considerable development in the understanding of ecosystem goods and services, and interest has grown in refining the analyses and evaluation at various scales (for example, http://www.teebweb.org). Moreover, ecosystem services are emerging in national initiatives, such as the UK National Ecosystem Assessment – an advanced interdisciplinary assessment of ecosystems and their services (UK National Ecosystem Assessment, 2011). However, the concept has also been criticised as becoming just another environmental buzzword, just like the term biodiversity (Brown *et al*., 2007) or as a 'complexity blinder' (Norgaard, 2010).

1.2 Objectives

The aim of this report is to provide a clear understanding of the concept of ecosystem services and how this concept relates to economics and policy. Understanding ecosystem services requires various sources of knowledge, (i) about the ecological processes, components and functions that generate these services; and (ii) about the way in which these services translate into specific benefits (Barbier, 2007). This report focuses on the economic sources of knowledge, addressing issues of scarcity, provision, supply and demand, ownership, valuation and policy.

The report is not so much intended for professional economists as for ecosystem researchers and policy analysts. It focuses on the economic foundations of the analysis and the evaluation of ecosystem services. What do we know from the literature? How can we apply the results in policy-oriented studies for governments, business and civil society? How can we contribute to the policy debate on preserving the planet?

1.3 Method

In order to answer the above questions, we have borrowed heavily from, and will duly refer to, the existing literature on economic aspects of ecosystem services. There is a rapidly growing number of papers dealing with ecosystem services. Although the concept of ecosystem services was introduced several decades ago, it has received massive attention since the appearance of the Millennium Ecosystem Assessment in 2005 (see Chapter 2). For example, scientific journals such as *Ecological Economics* (2007), PNAS (2008), *Frontiers in Ecology and the Environment* (2009) and *Biodiversity and Conservation* (2010) have dedicated special issues and special sections to the topic. In July 2012, a new academic journal *Ecosystem Services* was launched by Elsevier. In addition, entire volumes have been published on ecosystem services (Naeem *et al*., 2009; TEEB, 2010).

1.4 Structure of the report

The concept of ecosystem services covers a wide variety of costs and benefits of ecosystems. The next chapter not only describes the modern classification of these services but also adds some critical viewpoints from economics and ecology. This is followed by three chapters on the history of economic thought. Chapter 3 is about general economics and Chapters 4 and 5 about the sub-disciplines of natural resource and environmental economics and ecological economics. Chapter 6 discusses the valuation of ecosystems and their services and Chapter 7 the analysis of trade-offs. Chapter 8 concludes the report by advocating interdisciplinary research.

The concept of ecosystem services

2

2.1 Introduction

Humans benefit from a multitude of resources and processes that are supplied by natural ecosystems, and are collectively labelled as ecosystem services. Box 2.1 explains the history of the concept.

The concept of ecosystem services was popularised and its definition formalised by the United Nations 2004 Millennium Ecosystem Assessment (MEA, 2005). This assessment focused on the contributions of ecosystems to human well-being (i.e. an anthropocentric point of view), while at the same time recognising the potential for non-anthropocentric sources of value. The Millennium Ecosystem Assessment (MEA) was called for by the United Nations Secretary-General Kofi Annan in 2000. It was carried out between 2001 and 2005 to assess the consequences of ecosystem change for human well-being, by attempting to bring the best available information and knowledge about ecosystems to bear on policy and management decisions.

The MEA established the scientific basis for action needed to enhance the conservation and sustainable use of ecosystems and their contribution to human well-being. The MEA was in part a global assessment, but to facilitate better decision making at all scale levels, 34 regional, national and local scale assessments (or sub-global assessments) were included as core project components. Since the release of the MEA, further sub-global assessments have started. Some of the main findings of the MEA are summarised in Box 2.2. The publication of the MEA has stimulated international debate about the importance of the links between ecosystems and human well-being, and there is now considerable interest in assessing ecosystem services at regional and national scales.

Box 2.1 History of the concept of 'ecosystem services'

One of the oldest records of the idea of ecosystem services is from Plato (c. 400 BC) who realised that deforestation could lead to soil erosion and the drying up of springs (Daily, 1997).

The modern ideas about ecosystem services probably began with Marsh (1864), who suggested that the Earth's natural resources were not unlimited by pointing to changes in soil fertility in the Mediterranean. His observations went largely unnoticed at the time, and it was not until the late 1940s that society's attention was again drawn to the idea. Several authors advocated the recognition of human dependence on the environment in combination with the idea of 'natural capital'.

The term 'environmental services' was introduced in a report from the Study of Critical Environmental Problems in 1970, which listed services such as insect pollination, fisheries, climate regulation and flood control. In subsequent years, variations on the term were used, but eventually 'ecosystem services' became the standard in the scientific literature (Ehrlich and Ehrlich, 1981).

The review by Vandewalle *et al*. (2008) of 208 articles discussing the concept of ecosystem services provides an overview of studies from the 1960s and 1970s dealing with the loss of services and its consequences, as well as the failure of 'human-made' substitutions.

Much of the current understanding of ecosystem services was developed during the 1990s, which saw an explosion of books and articles dealing with and expanding the concept.

Source: Huitric *et al*., 2009.

Box 2.2 Four main findings of the MEA

1. Over the past 50 years, humans have altered ecosystems more rapidly and more extensively than in any comparable period of human history, largely with the intention to meet rapidly growing demands for food, fresh water, timber, fibre and fuel. This has resulted in a substantial and largely irreversible loss of the diversity of life on Earth.

2. The changes that have been made to ecosystems have contributed to substantial net gains in human well-being and economic development, but these gains have been achieved at growing cost in the form of the degradation of many ecosystem services, increased risk of nonlinear changes, and the exacerbation of poverty for some groups of people. These problems, unless addressed, will substantially diminish the benefits that future generations obtain from ecosystems.

3. The degradation of ecosystem services could become significantly worse during the first half of this century and is a barrier to achieving the Millennium Development Goals.

4. The challenge of reversing the degradation of ecosystems while meeting increasing demands for their services can be partially met under some scenarios that the MEA has considered, but these involve significant changes in policies, institutions and practices that are not currently underway. Many options exist to conserve or enhance specific ecosystem services in ways that reduce negative trade-offs or that provide positive synergies with other ecosystem services.

Source: Perman *et al*., 2011.

The MEA was unable to provide adequate scientific information to answer a number of important policy questions relating to ecosystem services and human well-being. In many cases it is clear either that the data needed to answer the questions were unavailable or that the knowledge of the ecological or social system was inadequate (VandeWalle *et al*., 2008).

2.2 Classification of ecosystem services

The MEA categorised ecosystem services into four classes. These are:

- *Provisioning Services*, which are the products obtained from ecosystems, including food, fibre, fuel, genetic resources, ornamental resources, freshwater, biochemical products, natural medicines and pharmaceuticals.
- *Regulating Services*, which are the benefits obtained from the regulation of ecosystem processes including air quality regulation, climate regulation, water regulation, erosion regulation, water purification and waste treatment, disease regulation, pest regulation, pollination and natural hazard regulation.
- *Cultural Services*, which are the non-material benefits that people obtain from ecosystems through spiritual enrichment, cognitive development, reflection, recreation and aesthetic experiences, including cultural diversity, spiritual and religious values, knowledge systems, educational values, inspiration, aesthetic values, social relations, sense of place, cultural heritage values, recreation and ecotourism.
- *Supporting Services*, which are necessary for the production of all other ecosystem services. They differ from provisioning, regulating and cultural services in

that their impacts on people are often indirect or occur over a very long time, whereas changes in the other categories have relatively direct and short-term impacts on people. Some services, like erosion regulation, can be categorised as both a supporting and a regulating service, depending on the time-scale and the immediacy of their impact on people. These services include soil formation, photosynthesis, primary production and nutrient and water cycling.

In its recent attempt to synthesise work in this field, TEEB has revised the MEA definition to replace 'Supporting Services' with 'Habitat Services' (TEEB, 2010). See also Table 2.1.

Stocks and flows

It is important to distinguish between an ecological stock and an ecosystem service flow (Ash *et al*., 2010). Stocks are generally expressed in units of quantity (e.g., metric tons, m^2 or ha), while flows are expressed as quantities per unit of time (e.g., kg/year or m^3/s). Ecosystem services are usually flows, both on the supply side and the demand side. Stocks and flows need to balance: if consumption exceeds production over a given period,

Table 2.1 Typology of ecosystem services in TEEB.

Category	Main service types	
PROVISONING SERVICES	1	Food (e.g. fish, game, fruit)
	2	Water (e.g. for drinking, irrigation, cooling)
	3	Raw Materials (e.g. fibre, timber, fuel wood, fodder, fertiliser)
	4	Genetic resources (e.g. for crop improvement and medicinal purposes)
	5	Medicinal resources (e.g. biochemical products, models & test organisms)
	6	Ornamental resources (e.g. artisan work, decorative plants, pet animals, fashion)
REGULATING SERVICES	7	Air quality regulation (e.g. capturing (fine) dust, chemicals, etc)
	8	Climate regulation (incl. C-sequestration, influence of vegetation on rainfall, etc.)
	9	Moderation of extreme events (eg. storm protection and flood prevention)
	10	Regulation of water flows (e.g. natural drainage, irrigation and drought prevention)
	11	Waste treatment (especially water purification)
	12	Erosion prevention
	13	Maintenance of soil fertility (incl. soil formation)
	14	Pollination
	15	Biological control (e.g. seed dispersal, pest and disease control)
HABITAT SERVICES	16	Maintenance of life cycles of migratory species (incl. nursery service)
	17	Maintenance of genetic diversity (especially gene pool protection)
CULTURAL & AMENITY SERVICES	18	Aesthetic information
	19	Opportunities for recreation & tourism
	20	Inspiration for culture, art and design
	21	Spiritual experience
	22	Information for cognitive development

Source: TEEB, 2010.

the stock will be depleted by an equivalent amount. It is usually necessary to express ecosystem services in both flow and underlying stock terms. The significance of a particular flow is hard to judge unless the size of the stock is known (and for renewable resources, the maximum flow that could be extracted from it without depleting the stock). Similarly, a stock by itself seldom says anything useful about the ecosystem service flows that are actually, or potentially could be, derived from it.

'Ecosystem services' are not restricted to living or renewable resources. Non-renewable natural resources, such as ore bodies, fossil aquifers, and deposits of coal, oil or gas, can also be regarded as natural capital stocks delivering a flow of services that end up supporting human well-being.

Not all ecosystem services are 'consumed' when they are used. For instance, admiring a cultural landscape or a 'biodiversity icon' does not necessarily make it unavailable for admiration by someone else. Even water is not destroyed when it is used: it is typically converted to another form (e.g., somewhat polluted), which may be unsuitable for immediate reuse for the same purpose but may be useful for another purpose.

The flows of provisioning services can often be directly measured, such as a harvest yield over a period of time. Alternatively, they can sometimes be measured as a change in the stock over a given period. For instance, ecosystems may provide a climate regulating service by sequestering carbon from the atmosphere. It is possible to measure this flux of carbon dioxide (CO_2) from the atmosphere into the ecosystem directly, but the equipment required is expensive and difficult to use. Over time, the net flux will show up as a change in the stock of carbon in the biomass, soil, sediment or water body, and this is easier to measure.

Ecosystem services and ecosystem disservices

Ecosystems are a double-edged sword; there are ecosystem services but also disservices. An example is provided by wetlands. The service value of wetlands has long been established, as they provide a range of benefits from bird habitat to water purification. However, they are also a source of disease in many parts of the world; for example, malaria can be classed as a very serious ecosystem disservice. Regarding disservices for agriculture, Zhang *et al*. (2007) distinguishes between (i) pest damage, (ii) competition for water from other ecosystems and (iii) competition for pollination services.

Other examples include those set out by Lyytimaki *et al*. (2008) regarding ecosystem disservices in urban areas. Bats, rats and foxes in urban parks can cause nuisance or fear. People can also feel unsafe in poorly managed urban green spaces, especially at night. Disservices can be seemingly inconsequential, such as fallen leaves, but these may cause increased braking distances and traffic accidents. Foliage along roadsides can decrease visibility (at corners for example), also leading to more traffic accidents.

The examples here show that ecosystem disservices can occur in a wide range of contexts (from rural to urban) and affect a wide range of ecosystem services (from provisional to cultural). While much of this report focuses on services, any policy or project analysis must be aware of potential disservices. Many of the points that will be made regarding services are equally applicable to disservices.

2.3 Drivers and pressures for change

Ecosystem services are dynamic, so the British Department for Environment, Food and Rural Affairs (DEFRA) considers them in terms of the drivers and pressures for change and how these result in policy responses. Bringing ecosystem services into the policy sphere requires an integrated approach. It also requires recognising the nature of the evidence and the various stages shown in Figure 2.1.

This framework summarises the cycle that links human societies and their well-being with the environment, building on the framework used by the Millennium Ecosystem Assessment (MEA). The framework emphasises the role of ecosystems in providing services that benefit people. Ecosystem services are the outputs of ecosystems from which people derive benefits including goods and services (e.g. food and water purification, which can be valued economically) and other values (e.g. spiritual experiences, which have a non-economic value). The combination of these goods, services and values provides our overall human well-being (expressed in society as health, wealth and happiness). The values that people

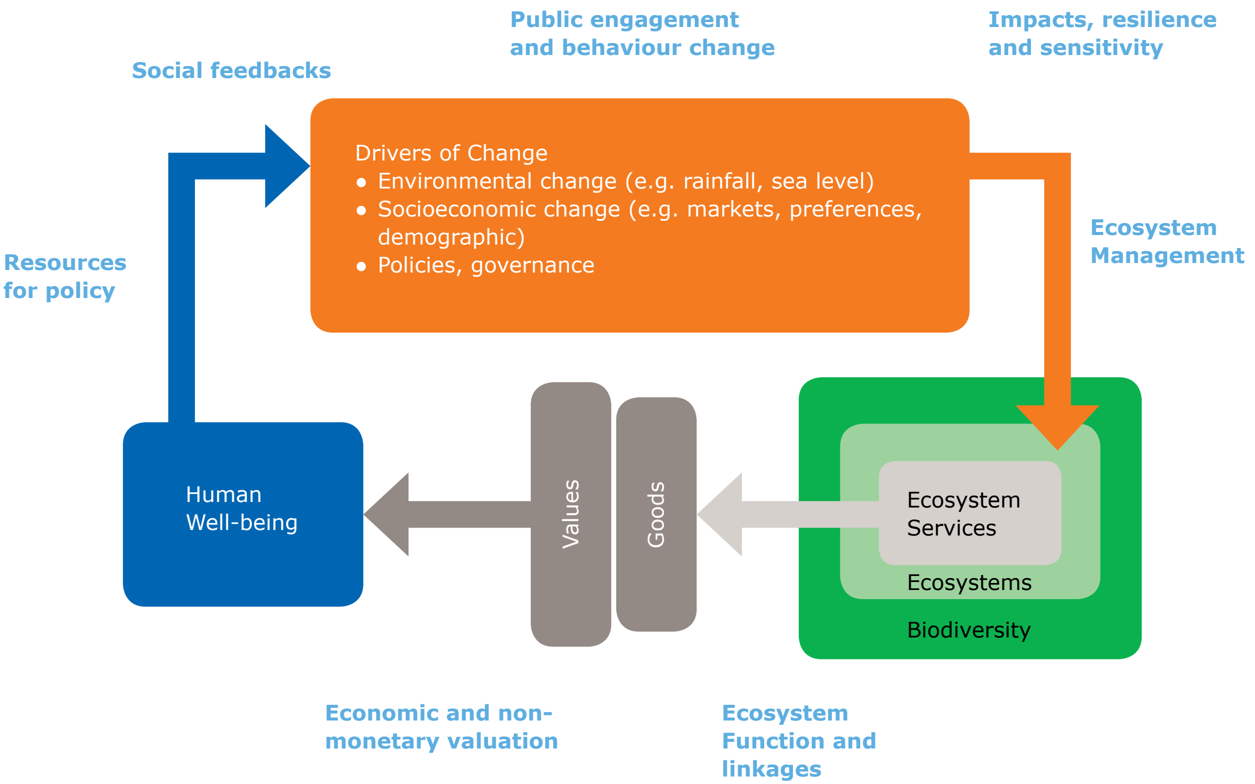

Figure 2.1 Ecosystem services in the policy sphere. Source: DEFRA, 2010.

derive from ecosystems may alter the way that they choose to use and manage the environment, which in turn leads to further changes in the environment. As a concept for better management and provision of ecosystems, the concept of ecosystem services relies on being incorporated into wider processes in order to have real-world effects.

Evidence is required at a variety of points (shown in bold in Figure 2.1) in order to enable the value of the ecosystem services to be taken into account in the policy/decision making process (DEFRA, 2010). This means that it is useful to gather and share knowledge between different disciplines and/or different evidence themes. Knowledge sharing also needs to occur across the variety of scales at which ecosystem services are provided and managed (e.g., between national and sub-national levels). Knowledge here refers to data and methods involved in providing evidence in the six themes (DEFRA, 2010).

2.4 Conceptual debate

There is substantial variation of opinion and much debate over the very definition of the term 'ecosystem services' (Dempsey and Robertson, 2012). To some economists and development planners, it is a useful heuristic covering a range of externalities: non-monetised elements of nature from which humans draw comfort and utility, but which should not necessarily be treated with a calculative approach assessing monetary value. To others, the term subjects nature to the strict logic of GDP (Gross Domestic Product) and cost-benefit analyses that feed into policy decisions.

The economic notion of a 'service' invokes the tertiary economy of non-consumptive exchange values and a certain distance from manufacturing and primary resource exploitation. Since the terminology suggests fungible commodities, one might expect ecosystem services to be defined with the same care and discrimination that apply to traditional service commodities such as medical service or administrative service: final products that are consumed directly to increase consumer utility.

However, there is a widespread tendency to use the term in a much broader sense (see Section 2.2). It seems that all features of the environment are called 'services' as long as they are connected in some way to an increase in human well-being. Critics have said that the definition has an 'everything but the kitchen sink' quality (Boyd and Banzhaf, 2007). To these economists, a great deal of what is counted as 'services' may in the strict sense actually be 'goods', 'benefits', or 'functions'. This worries them, predominantly because of the confusion it causes in accounting.

To be economically meaningful and to avoid double-counting, services must be final products – not processes – that input directly into a household production function. This results in a dramatically limited definition: 'Ecosystem services are components of nature, directly enjoyed, consumed, or used to yield human well-being' (Boyd and Banzhaf, 2007). In this definition there is no place for ecosystem processes. The definition is taken from welfare accounting, in which the 'distinction between end-products and intermediate product is fundamental.'

In response to this, Costanza (2008) has argued that the MEA definition of ecosystem services is 'appropriately broad and appropriately vague' (p. 350). For Costanza, the entire point of the ecosystem services approach is that the conventional economic approach is too narrow 'and tends to limit benefits only to those that people both perceive and are "willing to pay for" in some real or contingent sense' (p. 350). He recognises the potential for double-counting, but does not see it as a justification for a wholesale ejection of intermediate ecosystem processes from the domain of ES. He suggests that a single definition may be inappropriate, and that different definitions may be necessary for use in the different policy settings in which ecosystem services are commonly found, such as (1) the heuristic recognition and naming of externalities in nature; (2) the reform of governance and decision-making structures (e.g. national accounting, cost–benefit analysis) to recognise new kinds of assets; and (3) the formation of markets in new kinds of ecosystem commodities.

Another fundamental issue has been raised by Norgaard (2010), who is very concerned about the impact of the concept: "The ecosystem service metaphor now blinds us to the complexity of natural systems, the ecological knowledge available to work with that complexity, and the amount of effort, or transactions costs, necessary to seriously and effectively engage with ecosystem management." He explains that today's ecology does not have the predictive capacity to identify the sustainable use of any particular ecosystem service, to describe the trade-offs between uses of ecosystem services, or to be able to do this in the face of ecosystem change due to climate and other drivers (see Box 2.3). Norgaard: "The ecosystem services approach can be a part of a larger solution, but its dominance in our characterisation of our situation and the solution is blinding us to the complexity of the challenges we actually face."

Box 2.3 Limits of the stock-flow framework

Most of the ways in which ecologists think do not fit the stock–flow framework. Evolutionary and behavioural ecology, for example, provide insights into the nature and management of ecosystems, but these frameworks do not reduce to a stock-flow model. Indeed, to the extent that these other frameworks do provide insights, these insights are cautionary rather than complementary to the mechanistic prediction and control facilitated by stock-flow models. Very little ecological research has been conducted within an ecosystem service framework. Rather, ecologists think in terms of aspects such as population dynamics, food webs, energy flows, interactive behaviours, biogeochemical cycles, spatial organisation across landscapes and co-evolutionary processes. Furthermore, most ecological studies do not address human well-being. Similarly, most of the studies into human behaviour and social systems neither fit a stock-flow model nor connect to the ecosystem services or to the way social systems drive ecosystems. In short, the literatures representing our scientific understanding do not fit neatly into the ecosystem service framework.

Source: Norgaard, 2010.

From pre-classical economics to modern economics

3

3.1 Introduction

This chapter reviews the historic development of the conceptualisation of nature by economists and examines critical landmarks in economic theory and practice[1]. A distinction is made between four phases: pre-classical economics, classical economics, neoclassical economics and finally modern economics. Landmarks in the evolving conception of nature by economics are presented in Figure 3.1.

In pre-classical economics, the exploitation of land (nature) was conceived as the main source of wealth. In classical economics, land was superseded by labour as the main source of wealth, although the combination was still seen as crucial. In neoclassical economics, the source of wealth was conceptually decoupled from the physical world, while in modern economics, the environment came back as an issue of crucial importance for human well-being.

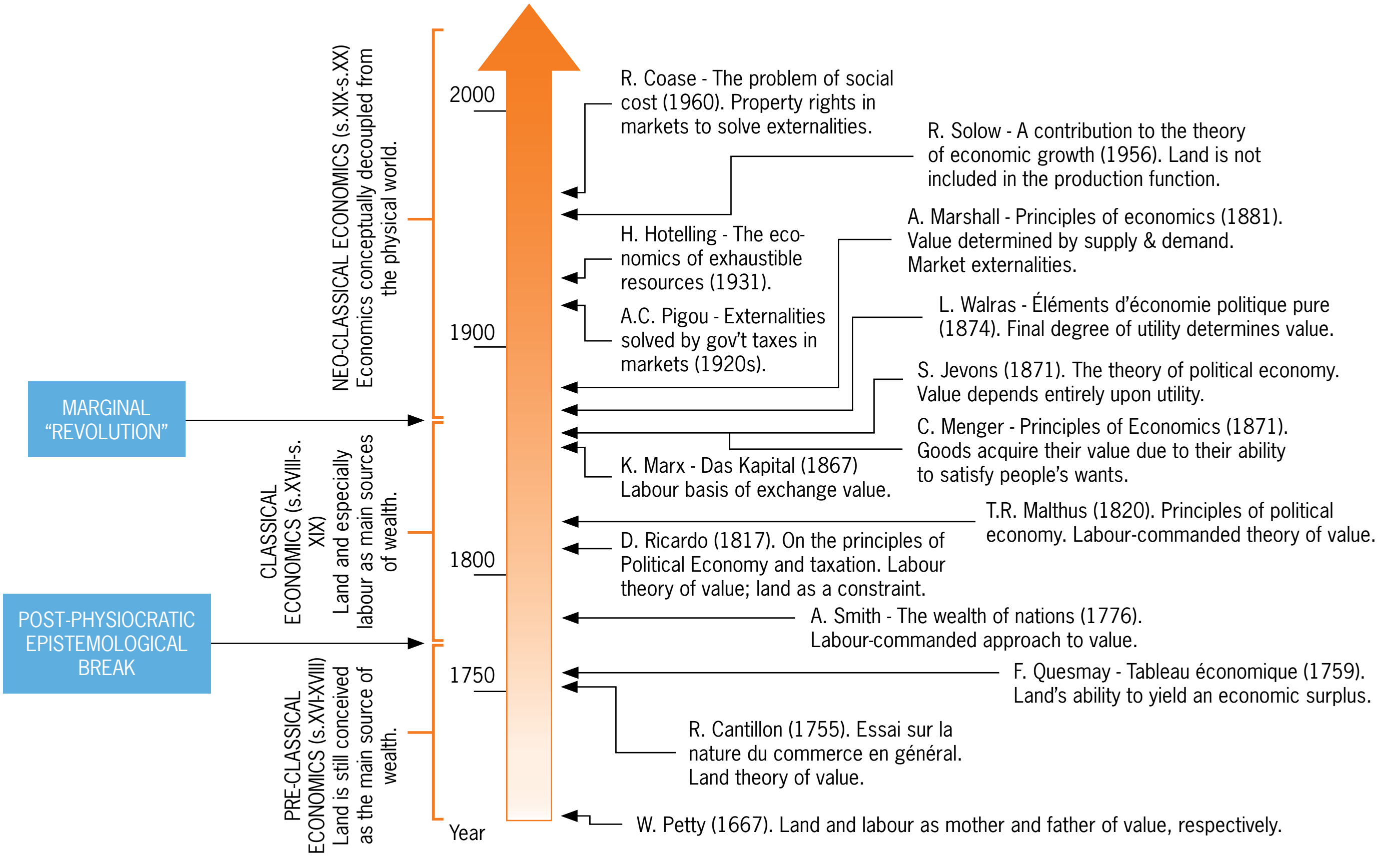

Figure 3.1 Landmarks in the evolving conception of nature by economics. Source: Gómez-Baggethun *et al*., 2009.

[1] This chapter draws heavily on the reviews by Hubacek and Van den Berg (2006) and by Gómez-Baggethun *et al*. (2009).

3.2 Pre-classical economics

From the 16th to the 18th century, economic philosophy and practice were led by mercantilism, the counterpart of political absolutism. It promoted governmental regulation of a nation's economy for the purpose of augmenting state power at the expense of rival national powers. According to mercantilism, wealth was mainly based on a large population providing a large labour supply and on the extraction of precious metals, such as gold and silver. If a nation did not possess mines or have access to them, precious metals were obtained by trade. Land was an important source of wealth, as it allowed feeding a growing population and served as a source of valuable materials. In addition, it functioned as the pivotal element in the feudal order, being the stable basis of the military, judicial, administrative and political systems.

In the 1750s, a school of economic thought developed in France which had as its first principle that natural resources, and fertile agricultural land in particular, were the source of material wealth. Physiocracy, meaning literally 'rule of nature,' is generally acknowledged as the first organised scientific school of economic thought. The Physiocrats maintained that the economic process could be understood by focusing on a single physical factor: the productivity of agriculture. The movement was particularly dominated by François Quesnay (1694–1774) and Anne-Robert-Jacques Turgot (1727–1781).

The most significant contribution of the Physiocrats was their emphasis on productive work as the source of national wealth. This contrasted with mercantilism, which focused on the ruler's wealth, accumulation of gold, or the balance of trade. Physiocrats viewed the production of goods and services as consumption of the agricultural surplus, since the main source of power was from human or animal muscle and all energy was derived from the surplus from agricultural production.

The perceptiveness of the Physiocrats' recognition of the key significance of land was reinforced in the following half-century, when fossil fuels had been harnessed through the use of steam power. Productivity increased manifold. Railways and steam-powered water supply and sanitation systems enabled the development of cities inhabited by several millions of people, with land values many times greater than those of agricultural land.

According to the Physiocrats, agriculture was the supreme occupation because it alone yielded a disposable surplus over cost. The agricultural labourers formed the 'productive' class, whereas the artisans and merchants were labelled the 'sterile' class. Juxtaposed between the two was the 'proprietary' class consisting of the land-owners, the king and the clergy, who received, in the form of rent, taxes and tithes, the dollar value of the net product produced by agriculture.

Kenneth E. Boulding has explained this view on the special role of land as a 'food chain theory': "The farmer produces ... more corn than the farmer and his family alone can eat. This results in a surplus. If this is fed to cattle it produces meat and milk, which improve human nutrition and perhaps enable the farmer to produce more food.... Food and leather 'fed' to miners produce iron ore. Food and iron ore 'fed' to a smelter produce iron. Food and iron 'fed' to a blacksmith produce tools or, 'fed' to a machinist, machines. The tools and machines 'fed' back to the farmer produce more food."

In the physiocratic model, economic rent was derived from unrecompensed work done by nature since in setting food prices, cultivators take into account their labour and expenses, as well as the surplus value contributed by the fertility of the soil. Quesnay measured and traced the value of the net flow of product between the three classes in his Tableau Economique, a model which represented for the first time, albeit in crude form, economic concepts such as general equilibrium and the Leontief input–output system, both of which became widely used economic models.

Influential for both the Physiocrats and later the classical economists was Cantillon's 'Equation de la Terre & du Travail'. Cantillon regarded land as the only truly original or primary input. The intrinsic values of commodities were reducible to the quantity of land directly and indirectly required for their production.

The influence of the Physiocratic School peaked in the 1760s and declined rapidly thereafter. For most economists, the Physiocrats represent a historical curiosity, though a few of their biophysical principles are evident in neoclassical or Marxist theory. However, their steadfast belief that nature was the source of wealth became a recurring theme throughout bio-physical economics.

3.3 Classical economics

Classical economics started in the early stages of the Industrial Revolution. This was the time of the rise of the industrialist class, and the decline of the importance of landlords. The main research agenda of classical economists was to derive the factors determining the wealth of nations and the distribution of income amongst the factors of production: land, labour and capital. The importance of technological progress and capital for productivity and thus for economic growth was recognised, but many classical authors retained the Physiocrats' special treatment of land.

In contrast to the Physiocrat belief that land was the primary source of value, classical economists began to emphasise labour as the major force backing the production of wealth. Many of the fundamental concepts and principles of classical economics were set forth in Adam Smith's An Inquiry into the Nature and Causes of the Wealth of Nations (1776). At the time when Smith wrote his treatise, only a small number of waterpower-driven industrial establishments existed and the Industrial Revolution had barely started. This helps to explain his conviction that agriculture, and not manufacturing, was the principal source of wealth. Smith considered the produce of the land as the principal source of the revenue and wealth of every country. For him, agriculture was more productive than manufacturing because it has two powers concurring in its production, land and labour, whereas manufacturing had only one (labour). Division of labour was the main element in productivity increase.

In Smith's theory of value, under competition, a costless item can never have a price. The services provided by land are costless in comparison to the capital invested in the land. The price paid for the use of land is, according to Smith, a monopoly rent. Smith's theory of rent anticipated later approaches to rent, which varied with different levels of fertility, the location and the transport system.

Classical economists found natural resources worthy of separate analytical treatment because the services they offer are free. Besides labour (and later also capital), land remained a separate factor in the production function. The fact that it was considered a nonsubstitutable production input explains to a degree the emphasis that some classical economists put on physical constraints on growth. This is reflected for instance in:

- *Ricardo's law of diminishing returns as applied to land*
  The law of diminishing marginal returns, propounded by David Ricardo, expresses a relationship between input and output, stating that adding units of any one input

(labour, capital, etc.) to fixed amounts of the others will yield successively smaller increments of output ('Diminishing Returns').

- *Malthus' concerns about population growth*
  Robert Malthus believed that natural rates of human reproduction, when unchecked, would lead to geometric increases in population: the population would grow at a rate of 2, 4, 8, 16, 32, 64 and so on. At the same time, he believed, food production increased only in arithmetic progression: 2, 4, 6, 8, 10. It seemed obvious to him that something had to keep the population in check to prevent wholesale starvation. He said that there were two general kinds of checks that limited population growth: preventative checks and positive checks. Preventative checks reduced the birth rate; positive checks increased the death rate.

Natural capital, in the form of land, which according to Malthus included 'the soil, mines, and fisheries of the habitable globe', thus retained a core position in classical economic analysis.

Whereas Malthus, Ricardo and others focused on different qualities of land, Johann Heinrich von Thuenen used distance as the central concept. Spatial economics and geography claim Von Thuenen as one of the founding fathers of their discipline. His concept of diminishing returns is also perceived as a precursor to the marginalist approach of neoclassical economics. Von Thuenen examined the pattern of agricultural production around the central town in an isolated state, in a homogenous featureless plain of uniform fertility. He tried to identify the principles that would determine the prices that farmers receive for their products, the rents that are earned and the patterns of land use that accompany such prices and rents. He developed a system of concentric circles, in which bulky or perishable goods are produced closer to the town and valuable or durable goods are imported from further away. The price of a product like grain in the central town is determined by the production and transport costs from the most distant farms whose produce is required to satisfy the town's demand. Since grain must sell at the same price irrespective of its location of production, ground rent is highest in the first concentric ring and decreases with distance. Von Thuenen arrived at similar conclusions as Ricardo in observing that differences in the quality of the soil will determine the ground rent in the same manner as its proximity to the central town.

In the 19th century, driving forces such as industrial growth, unprecedented technological development and the acceleration of capital accumulation triggered a series of changes in classical economic thinking in a direction that progressively resulted in nature losing the separate analytical treatment it had previously received. Three critical changes can be highlighted: a slow shift of the primary focus from land and labour towards labour and capital; a shift from physical to monetary analysis and a shift in the focus from use values to exchange values.

3.4 Neoclassical economics

The unifying approach of classical economists was their analysis of the values (land, labour and capital) embodied in a product to determine its price. Even though utility was seen as a precondition for goods to have value, classical economists were led by their orientation towards the longer term, where relative prices were only determined by costs of production. Hence their search for a labour or land content to establish values and prices. A very different orientation was adopted by the new neoclassical school, initiated by Jevons, Marshall, Menger and Walras, in their search for interdependencies between utilities in consumption and costs in production (Sandmo, 2011).

The marginalist revolution, which started in the 1870s, would have profound effects on the subsequent economic analysis of nature. The distinguishing characteristics of neoclassical economics were probably shaped by the longevity of the industrial revolution, the pace of technological developments, shifts from food- and fibre-based economies to mineral- and fuel-based economies, and economies in the industrialised world that seemed to be almost independent of extractive industries.

Towards the end of the classical economics period, some authors kept paying substantial attention to natural resources in physical terms. For instance, the 1865 book The coal question by William Stanley Jevons raised concerns about the depletion of coal stocks. The so-called Jevons paradox (recently 'rediscovered' as rebound effect) stated that gains in energy efficiency per unit of production could augment total energy consumption (Missemer, 2012).

After the marginalist revolution, neoclassical economics gradually restricted its analysis to the sphere of exchange values. Quite explicitly in this respect, Pigou (2006) wrote: "The one obvious instrument of measurement available in social life is money. Hence, the range of our inquiry becomes restricted to that part of social welfare that can be put directly or indirectly into relation with the measuring rod of money."

Neoclassical economic theory started to examine how technological innovation would allow for increased substitutability between production inputs such as land and capital, eventually consigning concerns about physical scarcity to oblivion (Georgescu-Roegen, 1971). Substitution was elevated to the status of a central principle which is used to explain both the price system and the production system. The neoclassical approach ignores the essential complementarity between different factors of production or different types of activities.

As such, neoclassical economists consider that different forms of capital (be they natural, man-made, social or financial) can substitute one another, which gives technology and innovation a more important role than natural capital and its ecosystem services. As a result of this view the problem of physical scarcity was reduced to a problem of scarcity of capital, considered as an abstract category that could be expressed in homogeneous monetary units. Scarcity of natural resources is then measured only in terms of the cost or price of a resource, not in any physical measure of its calculated reserve. Scarcity, in other words, is temporary and can be overcome by substitution driven by changes in relative prices. As such, economic production is seen as a self-contained circular flow process, without any connection to the anthropology, biology or physics (Gowdy and Ferreri Carbonell, 1999). As a result, the neoclassical approach led many economists away from nature. Or, in other words, nature has been ill-served by 20th-century mainstream economics (Dasgupta, 2008).

Thus, by the second half of the 20th century, land, or more generally speaking environmental resources, completely disappeared from the production function and the shift from land and other natural inputs to capital and labour alone, and from physical to monetary and more aggregated measures of capital, was completed. As Gowdy and Ferreri Carbonell put it (1999, p. 342): "The hermetic nature of production theory has resulted in the neglect of the scale of the impact of the economy on the natural world. Neoclassical utility theory is also hermetic in that it sees decisions made by individuals as independent of space, time, and the biophysical world. In the neoclassical theory of the consumer, only human preferences count. It does not matter where these preferences come from or what the consequences for the rest of the world are."

Likewise, it became common practice in international trade theory to exclude natural resource-intensive products from consideration. For example, the two primary factors of production in the factor proportions theory, which explains the pattern of comparative advantage by inter-country differences in their relative endowment with primary factors of production, are capital and labour.

The above overview of classical and neoclassical economic thinking on natural resources is summarised in Table 3.1. The economic conception of nature's benefits as use values in classical economics has given way to their conceptualisation in terms of exchange values in neoclassical economics.

3.5 Modern economics

The second half of the 20th century experienced a wave of environmentalism that the discipline of economics could not ignore. Rachel Carson's Silent Spring was first published in 1962, and in 1974 Lester Brown founded the World Watch Institute as an independent research institute devoted to global environmental concerns. Their work was quickly recognised by opinion leaders around the world for its foresight and accessible, fact-based analysis.

In economics, specialist sub-disciplines started to address shortcomings in standard economic thinking to analyse environmental problems. These sub-disciplines are the subject of the next chapter. Here we conclude the historic overview of economic thought with some general notions of the environmental problem.

The modern concern with the environment goes much beyond the perennial population problem that was addressed by Malthus. The new worries about ecology represented an awakening to a hitherto unknown state of human affairs. As Heilbroner (1980) explains: "It is that our abode is a vessel of limited capacity for the absorption

Table 3.1 Economic thinking on natural resources.

Period	Economics school	Conceptualisation of nature	Value–environment relationship
19th C.	Classical economics	Land as production factor generating rent (income)	Labour theory of (exchange) value Nature's benefits as use values
20th C.	Neoclassical economics	Land removed from the production function	Land as substitutable/producible by capital, and thus monetisable

Source: Gómez-Baggethun *et al.*, 2009.

of the noxious byproducts of production itself." In a word, we live on what Kenneth Boulding has aptly called Spaceship Earth. But far from conducting our affairs with the infinite care required of the inhabitants of such a vehicle of limited capacity, we continue to use up resources and to spew out the residues of productions as if the resources and the absorption capacity of the earth were infinite (Box 3.1). In Boulding's phrase, "we act as if we lived in a Cowboy Economy."

Economists like Boulding inspired thinking about the economic use of limited materials, energy and food supplies. This represented a shift from resource allocation in an economic system to the interdependency of ecological and economic systems. This view has been extended with the notion of 'hierarchies of systems', where the economic system is a subsystem of the social system, which is itself embedded in the ecosystem. Also new is the notion of co-evolving processes, which helps us understand how natural and social systems interconnect and change.

Box 3.1 From the open to the closed Earth

"We are now in the middle of a long process of transition in the nature of the image which man has of himself and his environment. Primitive men, and to a large extent also men of the early civilizations, imagined themselves to be living on a virtually illimitable plane. There was almost always somewhere beyond the known limits of human habitation, and over a very large part of the time that man has been on earth, there has been something like a frontier. That is, there was always some place else to go when things got too difficult, either by reason of the deterioration of the natural environment or a deterioration of the social structure in places where people happened to live. The image of the frontier is probably one of the oldest images of mankind, and it is not surprising that we find it hard to get rid of. (...) Economists in particular, for the most part, have failed to come to grips with the ultimate consequences of the transition from the open to the closed earth."

Source: Boulding, 1966.

Natural resource and environmental economics

4

4.1 Introduction

As such, economists have studied the original endowment of the Earth since the beginning of their discipline. By the second half of the 20th century, it became clear that the natural resource assets were subject to increasing pressures. This resulted in the emergence of natural resource and environmental economics as a distinct field (Crocker, 1999).

The first academic community that specialised in the field of environmental economics was associated with the Society of Environmental and Resource Economics, whose origins lie in the early 1960s (Turner *et al*., 1994). In those years, and due to increasing environmental problems and the emerging environmental policy agenda, the literature on the optimal use of renewable and non-renewable resources, common property problems, amenities associated with unspoiled natural environments, and pollution grew rapidly (Røpke, 2004). As mentioned above, one of the incentives was Rachel Carson's 1962 book Silent Spring, which explained how pesticides were causing serious pollution and killing many organisms.

Natural resource economics deals with the exploitation of resources, which can be classified into stock resources (renewables and non-renewables) and flow resources (solar radiation, wave and wind power). Environmental economics typically deals with problems of pollution (targets, instruments). Both branches of economics are based on neoclassical welfare economics. The scope of analysis of orthodox neoclassical economics is broadened by developing methods to value economic impacts on the environment and internalise them in decision making.

Traditional neoclassical economics largely neglected the economic contribution of nature by restricting its scope of analysis to those ecosystem goods and services that have a price-tag. After all, the perspective of neoclassical economics is that the market system is considered to be the preferred institution for allocating scarce resources. Ecosystem goods and services are, from this traditional perspective, considered to be of economic concern only to the extent that they are considered scarce, i.e. that demand exceeds supply at zero prices (Hussen, 2013). Hence, the systematic undervaluation of the ecological dimension in decision making would be partly explained by the fact that the services provided by natural capital are not adequately quantified compared with economic services and manufactured capital. From this perspective, non-marketed ecosystem services are viewed as positive externalities that, if valued in monetary terms, can be more explicitly incorporated in economic decision making.

Since natural assets (and thus also ecosystem goods and services) are scarce and increasingly exposed to the risk of irreversible degradation, it would be in the best interest of any society to optimise the management of its natural environment. This means that ecosystem goods and services should be considered when taking account of all the social costs and benefits. Whether this could be done through the regular operations of the market system requires a thorough understanding of certain complications associated with the assignment of ownership rights to ecosystem services (who reaps the benefits of nature?).

4.2 Welfare economics[2]

Welfare economics attempts to provide a framework in which normative judgements can be made about alternative configurations of economic activity. In this respect

[2] This section is based on the textbook by Perman *et al*. (2011).

the notion of allocative efficiency or Pareto optimality has been generally accepted in economics as a method to develop prescriptions for resource allocation. An allocation of resources is efficient if it is not possible to make one or more persons better off without making at least one other person worse off. A gain by one or more persons without anyone else suffering is a Pareto improvement. When all such gains have been made, the resulting allocation is Pareto optimal (or Pareto efficient).

The necessary conditions for markets to produce efficient allocations are the following:
- markets exist for all goods and services produced and consumed;
- all markets are perfectly competitive;
- all transactors have perfect information;
- private property rights are fully assigned in all resources and commodities;
- no externalities exist;
- all goods and services are private goods, that is, there are no public goods;
- all utility and production functions are 'well behaved';
- all agents are maximisers.

If there are goods and services for which no markets exist, then the market system cannot produce an efficient allocation, as this concept applies to all goods and services that are of interest to any agent. This fundamental condition necessitates the private property condition: a market in a resource or commodity can only exist where there are private property rights in that resource or commodity.

With respect to the provision of inputs to production, that is, natural resources, distinctions must be made between stock and flow resources, and for the latter, between renewables and non-renewables. There are generally no property rights in flow resources as such. There are no property rights, for example, in solar radiation. But economic agents can own land, and, hence capture the solar radiation falling on that land. Deposits of non-renewable natural resources are generally subject to private property rights. Often these reside ultimately with the government, but are sold or leased by it to individuals and/or corporations.

The non-existence of property rights is a much bigger issue in the renewable resource economics literature. Many biotic populations are not subject to private property rights. Ocean fishery is the standard example. For this 'open-access resource', systems of government regulation of 'common property' have been designed that promote behaviour consistent with efficiency on the part of the private agents exploiting the fishery.

4.3 Market failures

In the presence of market failures, economic pursuit on the basis of individual self-interest does not lead to what is best for society as a whole – privately optimal choices may deviate from economically efficient choices. There are different types of market failure. In the context of ecosystem services, externalities and public goods are particularly relevant.

Externalities
Externalities have been studied by economists ever since the days of Marshall and Pigou. Starting from the traditional neoclassical economic framework, the logical way to look at problems of environmental pollution is through the prism of externalities (Verhoef, 1999).

An externality occurs when the production or consumption decisions of one agent have an impact on the utility or profit of another agent in an unintended way, and when no compensation/payment is provided by the generator of the impact to the affected party (Perman *et al*., 2011). Consumption and production behaviour often do affect, in uncompensated/unpaid for ways, the utility gained by other consumers and the output produced, and profit realised, by other producers. Some authors omit from the definition of an externality the condition that the effect is not paid or compensated for, on the grounds that if there were payment or compensation then there would be no lack of intention involved, so that the lack of compen-sa-tion/payment part of the definition is redundant. The definition given here calls attention to the fact that lack of compensation/payment is a key feature of externality as a policy problem. Policy solutions to externality problems always involve introducing some kind of compensation/ payment thus removing the unintentionality, though the compensation/payment does not necessarily go to or come from the affected agent.

Externalities can be positive, i.e. benefiting others, or negative, i.e. harming others (Table 4.1). A positive externality exists when an individual or a firm making a decision does not receive the full benefit of the decision. In other words, the benefit to the individual or firm is less than the benefit to society. Thus, when a positive externality exists in an unregulated market, the marginal benefit curve (the demand curve) of the individual or the firm making the decision is less than the marginal benefit curve to society. As a result, positive externalities imply that less is produced and consumed than the socially optimal level.

Positive externalities from agricultural production include the conservation of agro-biodiversity and the benefits derived from scenic beauty generated by rural landscape and open space. Beekeepers can collect honey from their hives, but the bees will also pollinate the surrounding fields and thus aid farmers. But also: carefully maintaining your yard increases the value of your house and also increases the value of your neighbours' houses.

Consumers can be encouraged to consume more of a good that has a positive externality by means of a subsidy, which will increase the marginal benefit they receive when they consume the good. The subsidy can be paid for by all those who receive the external benefits.

A negative externality occurs when an individual or firm making a decision does not have to pay the full cost of the decision. If a good has a negative externality, then the cost to society is greater than what the consumer is paying for it. Since consumers make a decision based on the point where their marginal cost equals their marginal benefit, and since they do not take the cost of the negative externality into account, negative externalities result in market inefficiencies – unless proper action is taken.

When a negative externality exists in an unregulated market, producers do not take responsibility for the external costs. These costs are passed on to society. Thus, the producers have lower marginal costs than they would otherwise have, and the supply curve is effectively shifted down (to the right) of the supply curve faced by society. As the supply curve is shifted upwards, more of the product is bought than the efficient amount – that is, too much of the product is produced and sold. Since marginal

Table 4.1 Beneficial and harmful externalities.

Effect on others	Originating in consumption	Originating in production
Beneficial	Vaccination against an infectious disease	Pollination of flowers arising from proximity to apiary
Adverse	Noise pollution from radio playing in park	Chemical factory discharge of contaminated water into water systems

Source: Perman *et al*., 2011.

benefit is not equal to marginal cost, a deadweight welfare loss results.

A common and well-known example of a negative externality is pollution. For example, a steel producing firm might emit pollutants into the air. While the firm has to pay for electricity, materials etc., it is the individuals living around the factory who pay for the pollution, since it will cause them to have higher medical expenses, poorer quality of life, reduced aesthetic appeal of clear skies, etc. Thus the production of steel by the firm is associated with a negative cost to the people living near the factory – a cost that the steel firm in this example does not have to pay. The situation may be corrected by environmental regulations.

Negative externalities are a property rights problem. Who owns the air that is polluted by the steel mill? Ronald Coase proposed a solution which is known as the Coase Theorem (Perman *et al*., 2011). If there are negligible transactions costs, as long as someone owns the rights to the air around the steel mill, the efficient outcome will prevail. For example, if the steel mill owns the rights, then the people who live around the mill will be willing to pay the steel mill for not producing – up to the cost that they are incurring for health care, etc. The amount that they are willing to pay becomes an opportunity cost for the steel mill if they produce, so they will cut production to the optimal level. On the other hand, if the people own the air, then the steel mill will have to pay them the same amount for the right to produce. Thus the negative externality is directly added to the steel mill's marginal cost. Hence, according to the Coase Theorem, bargaining and market exchange may lead to an efficient outcome irrespective of how the property rights are distributed (Verhoef, 1999). In practice, however, obstacles to bargaining or poorly defined property rights can prevent Coasian bargaining.

Another way to solve the negative externality problem is to simply tax the producer to the amount of the negative externality. This adds to the producer's marginal cost and will cause them to reduce output. In an attempt to correct alleged market failures, the Environmental Economics literature has developed a range of methods to value external environmental costs and benefits (see Chapter 6).

Public goods and common pool resources

The concepts of externalities and public goods are often lumped together or used interchangeably. However, in their comprehensive handbook on the theory and policy implications of externalities, Cornes and Sandler (1996) describe and explain the relationship between these two concepts: externalities represent a variety of market failures, one of which is public goods.

Many natural assets, such as species and ecosystems, are characterised by the absence of fully defined property rights. Many of these assets are public or collective goods, or possess some features associated with such goods. As is summarised in Table 4.2, pure public goods have the characteristics of non-rivalry and non-exclusion (Slangen *et al*., 2008).

- *Non-rivalry*: Once the good is provided to a consumer, it can be made available to other consumers at no extra cost; that is, the marginal social cost of supplying the asset to an additional individual is zero. For example, wildlife areas protected by or for one agent will benefit everyone else who can access the area.
- *Non-exclusion*: one user cannot prevent consumption by others. Due to the non-exclusion attribute – that is, due to the fact that it is impossible or at least very costly to deny access to a natural asset – markets fail to efficiently allocate resources with public good characteristics. This may be understood by noting that prices do then not reflect the true scarcity of the asset.

Table 4.2 General classification of economic goods.

Excludability	Rivalry	
	Low/Absent	**High**
Easy	Toll or club goods (for example water storage, nature reserves)	Private goods (for example timber, minerals, food, fish)
Difficult	Pure public goods (for example sunsets, climate regulation mechanism of the Earth's atmosphere)	Common-pool resources (for example wild game for hunting, open access resources)*

Note: * Rivalry does not necessarily need to be high. In certain cases, such as rivers, large bodies of water or groundwater basins, rivalry is medium rather than high.

Source: Based on Moretto and Rosato (2002, p. 5, Table 1).

The existence of public goods is one of the reasons why there is a role for government in economic activity. Public goods are not supplied by markets, as follows from their non-excludability characteristic. Though many ecosystem goods and services differ from private goods in that they possess the characteristics of public goods, it must be stressed that many public goods are not pure public goods.

Most natural assets, such as a lake or ocean, a fishing ground, or a forest, are 'common-pool resources'. It is difficult or costly to exclude users from them or limit their access, but one person's consumption reduces the resource availability for others (Ostrom, 1999; Ostrom *et al*., 1999; Ostrom, 2002; 2003).

A unit of a common-pool resource harvested by one user is thus not available for others. As is shown in Table 4.1, this rivalry of resource units is shared with private goods. The difficulty of excluding users, however, is typically a public goods property. Table 4.1 also shows that the benefits of both toll goods and pure public goods are non-rival so that the consumption by one user does not necessarily detract from the benefit still available to other users. However, whereas a toll good is restricted to people who pay the producer or the holder of the good, the benefits of a pure public good are shared by all consumers, whether they paid for them or not.

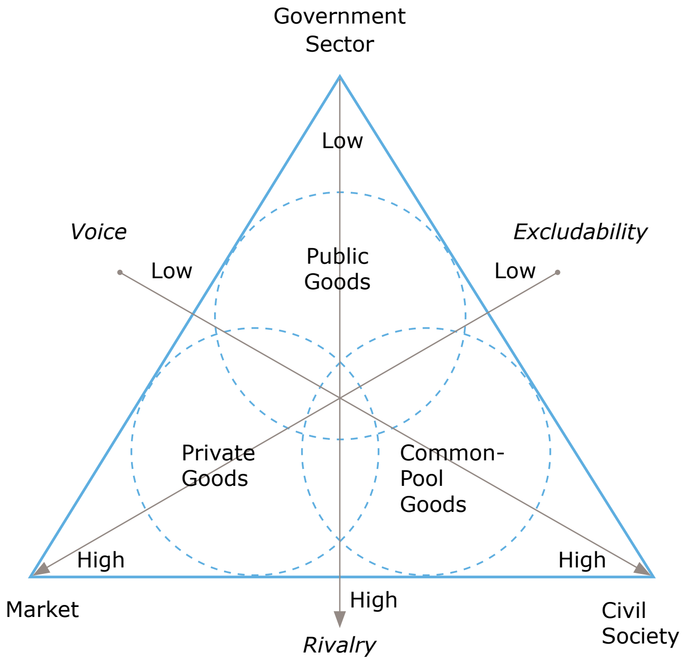

Figure 4.1 Institutions and the provision of goods and services. Source: Picciotto, 1995.

Particular institutions tend to be better suited to govern transactions related to particular types of products. Picciotto (1995) distinguishes three general types and then describes what type of transactions are best governed by these institutions. Each sector represents different individuals and has different incentives.

First, in the private sector, individuals and businesses owning property seek to maximise their return on asset investment (profit). The market sector tends to dominate whenever property rights can be assigned to make rival goods excludable. The property of exclusion allows private firms to sell at the marginal cost of production (i.e. the lower left-hand corner of the triangle in Figure 4.1).

Second, the government sector is best at producing public goods – the low excludability makes privatisation infeasible while the low voice component makes it difficult for the collective sector to organise (i.e. top of the triangle).

Third, the participation sector represents subsets of society with common interests who voluntarily join because they believe that benefits can be obtained by collective action. This sector is best at governing common-pool goods – these goods lack excludability, preventing them from becoming private goods, while the collective group will usually have more information that will enable them to more effectively manage the resource and capture the benefits (i.e. the lower right-hand corner of the triangle).

Free-rider problems

For both common-pool resources and public goods, the problem of excluding beneficiaries can lead to substantial free-riding; that is, trying to make individual gains without contributing to maintaining and improving the resource itself. Due to free-riding, overexploitation is a potential threat to common-pool resources, though it is absent in regard to pure public goods. This is because one's use of a pure public good does not subtract from the availability of that good to others.

The free-rider problem arises because there is no incentive for people to pay for the good. They can, in other words, consume it without paying for it. Since this will lead to no public good being provided, there will be social inefficiency. Thus there will be a need for the government to provide the public good out of general tax revenues.

Some goods can be public goods as well as private goods. An example is hedgerows. Farmers have a private incentive to maintain their hedgerows to reduce soil erosion and surface run-off. Moreover, hedgerows can play an important role in pest management. But hedgerows also increase the cultural, aesthetic and recreational quality of the landscape, thereby delivering public good values. In this respect, farmers who grow and maintain hedgerows essentially produce a private and a public good. This suggests that farmers can provide a certain amount of public good but only as far as it is privately optimal to do so. There may be a role for the government to further enhance the maintenance of hedgerows if it judges that private provision is below the social optimum.

4.4 Policy failures

An analysis of 'market failures' should not conclude that all government intervention in the functioning of a market economy is either desirable or effective. Government intervention offers the possibility of realising efficiency gains, but it does not always or necessarily realise such gains, and may even cause losses. Perman *et al*. (2011) list four reasons for such policy failures.

First, the removal of one cause of market failure does not necessarily result in a more efficient allocation of resources if other sources of market failure remain. In this case, the Second Best Theorem may be applied. If there are two or more sources of market failure, correcting just one of them will not necessarily improve matters in efficiency terms. In fact, it may make things worse. What is required is an analysis that takes account of multiple sources of market failure, and derives, as 'the second best policy', a package of government interventions that do the best that can be done given that not all sources of market failure can be corrected.

Second, government intervention may itself induce economic inefficiency. Poorly-designed tax and subsidy schemes may distort the allocation of resources in unintended ways. Any such distortions need to be offset against the intended efficiency gains when the value of intervention is being assessed.

Third, the chosen policy instruments may simply fail to achieve the desired outcomes. This is particularly likely in the case of instruments that take the form of quantity controls or direct regulation.

Fourth, actual government interventions are not always motivated by efficiency, or even equity, considerations. Adherents of the 'public choice' school of economics argue that the way government actually works in democracies can best be understood by applying to the political process the assumption of self-interested behaviour that economists use in analysing market processes.

Ecological economics

5

5.1 Introduction

A series of theoretical differences of opinion within the Society of Environmental and Resource Economics resulted in the emergence of a new transdisciplinary field: Ecological economics. It was institutionalised with the establishment of the International Society for Ecological Economics in 1988 and the launch of the first issue of the Ecological Economics journal in 1989 (Røpke, 2005). The scholarly journal was co-founded by Herman E. Daly, who is widely regarded as the founding father of ecological economics (Box 5.1).

Influenced by the work of researchers from systems ecology, biophysical economics, environmental and resource economics, agricultural economics, socio-economics, energy studies and general systems theory, the initiators aimed to address 'the relationship between ecosystems and economic systems in the broadest sense' (Costanza, 1989, p. 1). This aim was based on the view that the human economy and ecosystems are much more intertwined than is usually recognised. Ecological economists base their theorising on the economy's embeddedness in nature. They have a 'natural view' of the world, thereby emphasising natural laws, interdependencies between sectors and systems and limits to the material growth of the economy.

Whereas conventional environmental economics applies mainly neoclassical economic concepts to environmental and natural resource problems, ecological economics adopts a broadly 'diversified approach' (Venkatachalam, 2007, p. 550) and relies heavily on a range of relevant natural and social sciences. It integrates perspectives from a variety of fields, such as population biology, evolutionary biology, genetics and ecology, fisheries and wildlife management, as well as sociology and psychology. Moreover, as Baumgärtner *et al*. (2008, pp. 385, 386)

Box 5.1 Herman E. Daly

Herman E. Daly taught economics at Louisiana State University from 1968 to 1988. He then served as Senior Economist in the World Bank's Environmental Department until 1994, when he became a professor at the University of Maryland's School of Public Affairs. He studied under the economist Nicholas Georgescu-Roegen (1906-1994), whose book The Entropy Law and the Economic Process (1971) explained the decisive economic importance of the second law of thermodynamics (the entropy law) in a closed system: the availability of useful energy always declines. Daly maintains that the economy is a subset of an ecosystem which is finite, non-growing, and materially closed (i.e., no matter enters or leaves it), and that it uses the environment as a source of material inputs and as a sink for wastes. Unfortunately, he argues, the economy has become so large relative to the ecosystem that human activity is undermining the ecosystem's ability to support human life. Resource finitude and the entropy law make perpetual economic growth impossible. Accordingly, we must abandon growth (quantitative enlargement) in favour of development (qualitative improvement), and of a 'steady-state economy' which can be sustained long-term (though not forever), in which population and capital stocks are constant, and throughput (the flow of low-entropy matter and energy which is taken from the environment and transformed into high-entropy wastes) is minimised. Daly's first book-length state-ment of his ideas, Steady-State Economics: The Economics of Biophysical Equilibrium and Moral Growth (1977), attacks the ideology of economic growth and argues that biophysical limits to growth make shifting to a steady-state economy imperative.

Source: Attarian, 2003.

showed, a prominent feature of ecological economics is the inter- and transdisciplinary form of science, 'where interdisciplinarity is broadly understood as some kind of cooperation between scientific disciplines, and transdisciplinarity as some kind of interrelationship between science and society.'

The International Society for Ecological Economics (ISEE) and the Ecological Economics journal take the view that ecological economics is a 'transdisciplinary' field. It recognises that practical solutions to pressing social and environmental problems require new interdisciplinary approaches that focus on the links between economic, social and ecological systems. Neither the traditional practice of economics nor the natural sciences alone are held to be sufficient to address these issues. Nor can each alone explain the past history of the human–environment system.

In this view, the starting point and central organising principle of ecological economics is that the economy is embedded in and dependent upon the ecosphere – it is part of a larger system. Energy, material inputs, and environmental services are extracted from the natural environment and eventually return to the environment as waste heat, pollution or waste (Figure 5.1). Studies of this interconnected environment–economy system must take into account natural science principles from thermodynamics, ecology etc., as well as principles from psychology and other social sciences. Thus, ecological economics aims to integrate economics and various social and natural sciences (not just ecology).

In practice, the emerging field has attracted more economists than non-economists, so it is natural for some of these scholars to see ecological economics as a new paradigm in economics, alongside existing paradigms such as the mainstream neoclassical economics. They argue that ecological economists need to reject the neoclassical approach to economics, though there is no agreement on what to replace it with (see Section 5.3). But there are also natural scientists who believe that ecological economics can overturn and replace mainstream economics. Both these groups reject the core model of neoclassical economics – that economic theory should be primarily based on modelling the decision-making processes of individual consumers and firms with the default assumption that these agents maximise utility or profits. There have been ongoing tensions between mainstream and heterodox economists in ISEE (Røpke, 2005) as well as tensions between those who see ecological economics as an academic field and those who see it as a social movement or form of activism. By contrast, many mainstream environmental economists think of ecological economics as either a new field within mainstream economics that deals with the management of complex ecological systems or as a subfield within the field of environmental and resource economics (Røpke, 2005).

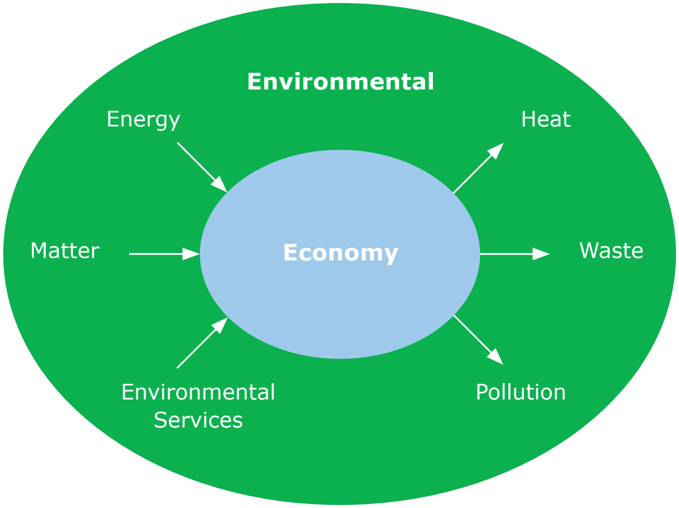

Figure 5.1 Economy and environment. Source: Stern, 2012.

5.2 Assumptions and approaches[3]

Principles of ecological economics

Ecological economists who see the field as going beyond a specialty within environmental economics share a common set of assumptions and approaches (Røpke, 2005):

- The economy is a subsystem of the larger human–environment system.
- Models of the economy have to comply with biophysical principles.
- There are limits to our ability to substitute human-made inputs and knowledge for natural resources and the environment in both production and consumption. These limits are due to several considerations:
  - Thermodynamics: there are minimum amounts of energy required to transform and move matter, which is the foundation of economic activity.
  - Basic human needs for food, shelter etc. that require some material and energy inputs, and perhaps a greater psychological need for contact with nature.
  - Essential 'natural capital' required to support life on our planet.
- Economic policy must consider the combined objectives of economic efficiency, equity and sustainability, instead of primarily emphasising efficiency. Ecological economics has been characterised as 'the science and management of sustainability'.

The first three principles imply that there are limits to the possible physical scale of the economy. Unlimited growth of the use of resources is not possible. Considering the third and fourth principles together has led many ecological economics to argue that sustainability requires minimum levels of natural capital or natural resources to be maintained, as human-made inputs have limited ability to substitute for them in the provision of human welfare.

This last idea is termed 'strong sustainability'. By contrast, many mainstream environmental economists assume that human-made inputs can substitute extensively for natural inputs. They argue that sustainability could be achieved as long as sufficient investment is made in human-produced capital. This is referred to as 'weak sustainability' (Neumayer, 2004).

Comparison with environmental economics

While environmental economics focuses on price, ecological economics focuses on quantity. Environmental economics focuses on market failures as the main determinant of environmental problems. Seen in terms of external costs, environmental economics regards the problem as one of incorrect prices and the solution as implementing the right prices. In many cases, these prices must be determined through research, hence the emphasis on valuation in environmental economics (see Chapter 6). Ecological economics sees environmental problems as being primarily problems of scale – that the scale of exploitation of natural resources and the production of wastes are both too large relative to the Earth's carrying capacity. Therefore, ecological economists are more likely to analyse economic–ecological systems in terms of quantities of flows of materials and energy. Tools of analysis include the ecological footprint, a quantity indicator. Ecological economics focuses primarily on sustainability – equitable distribution of resources over time – while environmental economics focuses on efficiency – ensuring that marginal costs and benefits of activities are equal.

Ecological economics can be associated with a sustained functioning of the combined ecological–economic system. For example, with respect to exploiting natural resources, ecological economists are particularly concerned with the

[3] This section is based on the review by Stern (2012) and on Van den Bergh (2001).

scale of exploitation relative to the dimensions of the ecosystems on which mankind depends. As a result, explicit attention for spatial scales is common in many studies (Gowdy and Ferreri Carbonell, 1999).

Whereas the core concept in environmental economics is market failures, ecological economics has sustainable development as its central concept (Van den Bergh, 2001). Whereas in economics, sustainable development is usually regarded as being identical to sustainable growth, ecological economics takes absolute physical limits to growth seriously and regards the problem of a 'maximum scale' of the economy as relevant. Differences between environmental economics and ecological economics are – somewhat simplified – summarised in Table 5.1.

Ecological economics is inclined to add ecological concepts to the 'pure' economic values. Examples of these concepts are 'life support functions', 'internal environmental system functions', 'ecosystem health' and 'resilience of ecosystems'.

In a recent paper Farley (2012) addressed not only sustainability and efficiency but also justice as a central issue concerning the economics of ecosystem services. Justice is about the allocation of resources among groups and individuals. In the case of ecosystems services, justice concerns entitlements to both the structural building blocks of ecosystems and the services they generate: "The two of course are frequently in conflict. If one individual has the right to the timber in a forest, this may conflict with the right of another individual to enjoy the water purification, flood regulation, climate regulation and other services provided by that forest."

Table 5.1 Differences between environmental economics and ecological economics.

	Environmental Economics	Ecological Economics
1	Optimal allocation and externalities	Optimal scale
2	Priority to efficiency	Priority to sustainability
3	Optimal welfare or Pareto efficiency	Needs fulfilled and equitable distribution
4	Sustainable growth in abstract models	Sustainable development, globally and North/South
5	Growth optimism and 'win-win' options	Growth pessimism and difficult choices
6	Deterministic optimisation of intertemporal welfare	Unpredictable co-evolution
7	Short- to medium-term focus	Long-term focus
8	Partial, monodisciplinary and analytical	Complete, integrative and descriptive
9	Abstract and general	Concrete and specific
10	Monetary indicators	Physical and biological indicators
11	External costs and economic valuation	Systems analysis
12	Cost-benefit analysis	Multidimensional evaluation
13	Applied general equilibrium models with external costs	Integrated models with cause-effect relationships
14	Maximisation of utility or profit	Bounded individual rationality and uncertainty
15	Global market and isolated individuals	Local communities
16	Utilitarianism and functionalism	Environmental ethics

Source: Van den Bergh (2001, p. 16, Table 1).

5.3 Heterodox viewpoint

Gowdy and Erickson (2005) believe that ecological economics offers viable alternatives to the theoretical foundations of neoclassical economics, and that it is therefore poised to play a leading role in recasting the scope and method of economic science. The authors admit that ecological economics has not yet coalesced into a coherent school of thought, but they nevertheless believe (p. 219) that it is "a leading contender among heterodox schools to become a comprehensive alternative to neoclassical orthodoxy."

However, the actual development of ecological economics has been criticised by Spash (2012), who rejects the methodological pluralism advocated by the Ecological Economics journal. He wants to "place the future of ecological economics firmly amongst heterodox economic schools of thought and in ideological opposition to those supporting the existing institutional structures perpetuating a false reality of the world's social, environmental and economic systems and their operation." Since orthodox economics shows substantive failures in addressing reality, Spash argues, the superficial transdisciplinary rhetoric needs to be replaced by serious interdisciplinary research. This section summarises some elements of his plea for realism and reasoned critique.

Spash observes that the first introductory book of ecological economics (Costanza *et al*., 1998) maintained an uneasy balance between requesting a new worldview, to address our social and environmental woes, and not ejecting the body of orthodox thinking. Another introductory text (Common and Stagl, 2005) explicitly falls back on standard orthodox economic theory and methodology. This includes using the same philosophy of science and ethical theory as that associated with neoclassical economics. Such a position seems to ally ecological economics closely with mainstream environmental and resource economics. It is not surprising then, an irritated Spash notes, that the Journal of Economic Literature classifies ecological economics under: 'Q5–Environmental Economics'. The more specific entry is 'Q57–Ecological Economics: Ecosystem Services; Biodiversity Conservation; Bioeconomics; Industrial Ecology'.

Mainstream economics appears prescriptive and restrictive in its ever increasing reliance on mathematical formalism as a monist methodology. Expressing all theory in terms of individual behaviour which can be captured in formal mathematics prevents a more realistic model from developing: "The decision as to where ecological economics should engage seems rather self-evident when given the choice between discourse with closed-minded formalists employing outdated behavioural psychology to defend an unrealistic position, and open-minded social psychologists or sociologists sharing common critiques."

Spash warns against close association with mainstream economic ideas and incorporation of economic formalism: "...these are obstacles for creating new knowledge about environmental and socio-economic problems." He finds the mix of mainstream economics and ecological economics confusing and contradictory. The continued support for mathematical formalism and quantification as providing the means to scientific rigour and validity is damaging to an alternative vision for ecological economics: "Ecological economics is, and should be in part, an empirically based subject, but the form of that empiricism needs development and should not be restricted to a narrow, dogmatic, anti-pluralist, prescriptive caricature, nor based upon appeals to the most popular methodology."

Valuation of ecosystem services

6

6.1 Introduction

The concept of ecosystem services has been introduced to determine the different values of ecosystems. However, this does not mean that values have to be explicitly presented in monetary terms. Depending on the reason for valuation or on the target audience, values can also be expressed in qualitative or quantitative terms.

There may be several reasons to value ecosystem services, each of which require a specific approach (Slootweg and Van Beukering, 2008, p. 18):

- *Advocacy*: economic valuation is often used to advocate the economic importance of the ecosystem services, with the purpose of encouraging sustainable development.
- *Decision making*: valuation can assist governments in allocating scarce resources to achieve economic, environmental and social goals. Economic valuation studies are critical to assist decision makers in making fair and transparent decisions.
- *Damage assessment*: valuation is increasingly used as a means of assessing damage inflicted on an ecosystem.
- *Sustainable financing*: valuation of ecosystem services can be used to set taxes or charges for the use of these goods and services at the most desirable level.

Whereas environmental economics focuses on value dimensions (i.e. utility and welfare in theory, and costs and benefits), ecological economics is inclined to add (ecological) criteria to these dimensions, to cover aspects such as productivity, stability and resilience of ecosystems. Ecological economics criticises the utilitarian approach used in environmental economics because they take no account, or insufficient account, of items such as internal environmental system functions and resilience (Van den Bergh, 2001).

Valuation of ecosystem services is controversial because of theoretical and empirical problems, and the potential effect of the resulting values on public opinion and policy decisions (Loomis *et al*., 2000). For example, biologists such as Ehrlich and Ehrlich (1992) argued that ecosystems are complex, indivisible entities that operate on time scales outside the range of human perception, and that they have values that are difficult or impossible to measure (see also Gowdy, 1997). Not only biologists, but also scholars from other disciplines (and even economists) may find monetary valuation a 'hopeless' exercise. Philosophers such as Sagoff (2000; 2008) and economists such as Bromley (Vatn and Bromley, 1994) dismiss monetary valuation as ethically insupportable and impracticable. Nunes and Van den Bergh (2001), on the other hand, claim that monetary valuation can make sense, although they point out that the various valuation methods should not be considered as universally applicable to all levels of biological diversity or to all types of biodiversity values or ecosystem services.

6.2 Categories of values

Most ecosystem services are characterised by the fact that they have no price tag because they are not fully captured in markets. There are some exceptions to this rule, which are particularly related to the provisioning services. Foodstuffs, for example, are generally traded in markets. Nevertheless, the fact that no market-based price tags exist for many ecosystem services does not imply that these services are of no value. Efforts to take these values into account may be assisted by a framework for distinguishing and grouping the various values of an ecosystem.

Total economic value consists of two main elements (Figure 6.1). One element is that of the services provided in the course of the actual use of an area in consumption and production activities. This is referred to as use value. By contrast, non-use values involve no tangible interaction between the area under consideration and the people who use it for production or consumption. Since non-use values are closely linked to ethical concerns and altruistic motives, they are more amenable to debate than use values – even though they can be substantial (Brown *et al*., 2007).

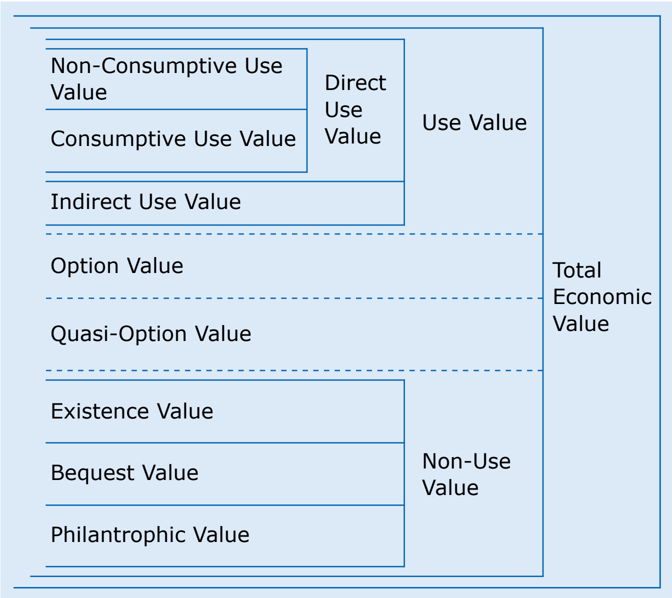

Figure 6.1 The concept of the total economic value (TEV) of an ecosystem. Source: Turner *et al*. (1998, p. 13, Figure 2).

As regards use values, a distinction is made between direct and indirect use values. Direct use values are concerned with the enjoyment or satisfaction received directly by consumers of the area, which involves both commercial and non-commercial activities. Direct uses include both consumptive uses (for example, agriculture, water use, hunting, fishing and the gas mining industry) and non-consumptive uses (for example, recreation, tourism, and in situ research and education). Consumptive use values are conceptually clear and offer the best chances of being measurable. After all, they can be marketed, resulting in market prices that signal the (true) scarcity of the asset. Non-consumptive use values, however, relate to assets that provide value without being traded in the market place and are therefore much more difficult to measure. Indirect use values indicate the indirect support to economic activity by natural assets and services, and as such they relate to life-support benefits. Examples of indirect use values include storm water containment and treatment, water purification, watershed protection, soil formation, and the decomposition and assimilation of wastes. As such, they are especially related to regulating and habitat ecosystem services.

While use values arise from the use of an area, or ecosystem service, non-use values are independent of current or potential use. Non-use values exist where the preferences of individuals who do not intend to make use of, say, the Amazon rain forest would nevertheless feel a ‘loss’ if the area was to disappear. Depending on exact definitions, non-use values may include all of the following: option values, quasi-option values, bequest values, philanthropic and existence values.

Option value relates to the amount that individuals would be willing to pay today to safeguard an ecosystem service for future direct and indirect use. In the economic literature it has been suggested that option value represents a difference between ex ante and ex post valuation, where the terms ‘ex ante’ and ‘ex post’ refer to the amount of information that is available. Ex ante relates to the situation where the state of the world is still unknown, while ex post refers to the situation after the state has been revealed. If there is uncertainty about the future value of an ecological function, and one has to await improved information before giving up the option to protect the asset, then quasi-option value may be derived from delaying economic activities. Quasi-option value is the expected benefit of awaiting improved information, the benefit deriving from delaying exploitation and conversion of an ecosystem service. It suggests a value attached to protection, given the expectation of the growth of knowledge. (Note that in Figure 6.1, both option value and quasi-option value are indicated by a dotted line, since possible double counting needs to be taken into account when adding up these values.)

Bequest value is a willingness to pay to keep ecosystem services intact for the benefit of one's descendants, or more generally, for the next and future generations. This value is associated with inter-generational equity.

Philanthropic value results from individuals placing a value on the conservation of ecosystem services for contemporaries of the current generation to use (Turner *et al*., 1998). As such, this value represents the satisfaction of knowing that other people in the current generation have access to nature's benefits (Bateman *et al*., 2002). The individual, in other words, is concerned with intra-generational equity.

Existence value involves a subjective valuation as it is based on the satisfaction that individuals experience from knowing that certain ecosystem services exist, for themselves and for others, without being used now or in the future.

The TEV framework is a tool for exploring the types of values that may be attached to each ecosystem service (Table 6.1). This helps determine the valuation methods required to capture these values. There are many dimensions of the value of ecosystem services that are already included in the System of National Accounts (SNA), for example those for provisioning services (e.g., timber) or when ecosystem services contribute input to the production of goods and services that are traded on the market (e.g., pollination services contributing to agricultural production). In many other cases, the value of ecosystem services is not included because they represent flows outside of the SNA production boundary (e.g. carbon storage or flood protection of wetlands). In both cases, explicitly measuring these flows can improve our ability to understand the links between the environment and the economy.

Table 6.1 Ecosystem services and Total Economic Value.

Ecosystem Services		Economic values			
Category	**Examples**	**Direct Use**	**Indirect Use**	**Option Value**	**Existence Value**
Provisioning	Food, fibre, fuel, biochemicals, natural medicines, pharmaceuticals, fresh water supply	X		X	
Regulating	Air quality regulation, climate regulation, water regulation, natural hazard regulation		X	X	
Cultural	Cultural heritage, recreation and tourism, aesthetic values	X		X	X
Habitat	Primary production, nutrient cycling, soil formation	These services are valued through the other categories of ecosystem services			

Source: DEFRA, 2007.

6.3 Monetary valuation techniques

Monetary valuation of ecosystem services and the development of appropriate valuation methods have aroused considerable interest in recent decades. It is noteworthy that publication rates on monetary valuation methods increased rapidly after the Exxon Valdez oil spill in March 1989.

Intuitively, the importance of ecosystem services to society is best represented by monetary values. However, reality is more complicated than common intuition suggests. There is an important difference between prices and values. Prices that arise from market transactions offer, in some sense, objective information, whereas many concepts of value are subjective (Heal, 2000). As a result, valuation cannot be dissociated from choice. Economic analysis provides several measurement techniques to assign a subjective value to the benefits of, or damage avoided through, changes in ecosystem services.

In environmental economics, values (costs and benefits) are generally measured in terms of:

- *Willingness to pay* (WTP), i.e. the amount individuals are prepared to pay for goods and services;
- *Consumer surplus* (CS), i.e. the benefit an individual receives from utilising a resource over and above what they have to pay for it;
- *Producer surplus* (PS), i.e. the profit that a producer makes from selling a product (i.e. the difference between the cost of producing the product and the market price).
- *Opportunity cost*, i.e. the value of something in its next best alternative use.

The economic benefit associated with using an environmental asset is known as economic surplus, which is a combination of CS and PS. However, where the costs of production are not known, some studies have adopted valuations based on gross revenues and consumer surplus.

Table 6.2 gives an overview of valuation techniques. Assigning a monetary value to ecosystem services can be done by different measurement techniques. These are based on market values (such as the replacement cost method or the production function approach), observed market behaviour (revealed preferences or indirect methods) or stated preferences (direct methods). Revealed preference techniques include, among others, travel cost and hedonic pricing methods. These techniques can, however, not measure non-use values of ecosystem services (i.e., values that are not associated with actual use, for example the existence of tropical forests). This tends to be the domain of stated preference techniques, such as contingent valuation and choice experiments. During the last two decades, both revealed and expressed preference techniques have enjoyed a steady increase in application. Nevertheless, the existing techniques suffer from various shortcomings, such as high costs (related to, for example, the administration of contingent valuation questionnaires) and length of time associated with conducting valuation studies, substantial data requirements, and potential biases. The high costs and length of time to undertake monetary valuation studies have led to an increased interest in benefits (or value) transfer. This method can be used to apply values estimated at one site to another site.

Book-length treatments of these measurement techniques include Bateman *et al*. (2002) and Hanley and Barbier (2009). None of these techniques is a panacea, however. Depending on the situation and objectives, some techniques tend to be better than others (Hussen, 2013).

Table 6.2 Environmental valuation techniques.

Category of technique	Name of Technique	Description of approach
Market price based	Market values	Value based on market prices (minus costs of production) and taking into account government intervention such as taxes and subsidies.
	Change in productivity	Value is based on the change in quality and/or quantity of a marketed good and the associated change in total net market value (e.g. measuring fishery support function).
	Damage costs avoided	Value of an asset is equivalent to the value of the economic activity or assets that it protects (e.g. the damage avoided by maintaining a coast protection function)
	Substitute/surrogate prices	Value of a non-marketed product is based on the market value of an alternative product providing the same or similar benefits.
	Expected values	Value is based on potential revenues (minus potential production costs) multiplied by probability of occurrence.
Cost based	Replacement cost	Value is based on the cost of replacing the environmental function.
Revealed preference or surrogate market	Travel cost method	Value can be inferred from the cost of travel to a site (i.e. expenses and value of time) using regression analysis.
(uses market-based information to infer a non-marketed value)	Hedonic price	Value of goods is based on the value of individual components (e.g. the landscape premium in property prices) which can be determined through regression analysis.
Stated preference or constructed market approach	Contingent valuation	Carefully constructed and analysed questionnaire survey technique asking a representative sample of individuals how much they are willing to pay to prevent loss of, or to enhance, an environmental good or service.
(questionnaire surveys to ask people's direct willingness to pay)	Choice experiments	As above, but involves asking respondents to select their preferred package of environmental goods at different prices and then inferring specific component values via econometric analysis.
Transfer of values	Benefits (value) transfer	The transfer of economic values estimated in one context and location to estimate values in a similar or different context and location.

Source: FAO.

Table 6.3 Suitability of valuation methods for the categories of ecosystem services.

Method	Ecosystem service category			
	Provisioning	Regulating	Cultural	Habitat
* Market prices	+	+/-	+/-	-
* Cost approaches	+	+	-	+/-
* Revealed preferences	+/-	-	+	+/-
* Stated preferences	-	-	+	+

Explanation: + = suitable; +/- = suitable in certain circumstances; - = not or hardly suitable. Source: Based on Pascual and Muridian (2010).

The most appropriate technique depends on the type of good or service (Table 6.3). The choice of valuation technique generally depends on the availability of resources, time and data for the study.

Valuation studies of natural assets have mainly been conducted for species preservation, recreation and water management. Essentially, monetary values placed on these assets and their services are rooted in people's values. Hence, although based on an appraisal of nature, monetary values are all fundamentally anthropocentric, meaning that they are based on the utility of nature to humans. The basis for monetary valuation is to enquire what is the most an individual is willing to pay (WTP) for incremental changes in the availability or quality of ecosystem services, or the minimum compensation (s)he is willing to accept (WTA) to forgo such a change (Hanley and Barbier, 2009).

Costanza *et al*. (1997) estimated the current economic value of 17 ecosystem services on a biosphere-wide basis at an average of US$ 33 trillion per year. The paper evoked a great deal of debate, and the estimates were heavily criticised for various reasons (see several commentaries in the special issue of Ecological Economics (Costanza, 1998)).

Box 6.1 Common errors in economic valuation

While many of the ways in which programmes and projects may be assessed are straightforward and make common sense, common errors should be avoided (source, Ash *et al*. (2010):

- *Marginal versus total values*. Economic value is determined by how much an additional amount of a thing is worth, not how much the thing is worth in total. If an ecosystem service is to be reduced but not eliminated, the loss to be estimated is the benefits forgone as a consequence of the reduction.
- *Substitutes*. If there are alternative ways to generate the goods or services of natural ecosystems, the value of such goods and services cannot be greater than the cost of the alternative.
- *Replacement costs*. It follows from the above observation that if there are cheaper ways of producing a good or service than replacing the system that currently provides it, the cost of replacement will overstate the value of the good or service.
- *Double-counting*. There are often many ways of estimating economic values. Calculating values by different methods is sometimes useful to check on against the other, but it is important not to count the same value twice.
- *Alternative metrics*. While embodied energy, ecological footprints, and other physical measures may be useful for some purposes, they generally cannot be used in economic valuation.

6.4 Valuation problems

The framework of total economic value does not imply that the total value of an ecosystem should or can be measured. Ecological processes, functions and structures, on which many natural assets depend, are extremely difficult to quantify in purely monetary terms (Nijkamp *et al*., 2008). As a result, nature, biodiversity and ecosystem services cannot be expressed completely in financial numbers.

There are several problems with economic valuation. Common errors in valuation are presented in Box 6.1, one of them being valuing a particular type of benefit more than once.

Errors are easily made because of more fundamental problems with the Total Economic Value framework. First of all, although the difference between use and non-use values might be conceptually clear, the distinction between the two categories of value is in practice rather ambiguous (Boardman *et al*., 2006). The distinction between use and non-use values implies a kind of reduction to separate items of ecosystem services, although they often do not exist in distinct units like market goods. Second, and related to the previous problem, many of the values that economists attribute to ecosystem services are ignored in private valuations, where the focus is generally on the consumptive, direct use values (Van Kooten, 2013).

In applying the Contingent Valuation Method (CVM), lack of experience can lead to situations in which individuals' responses are poor approximations of their true WTP. Moreover, the hypothetical character induces an impressive list of potential biases that result from using CVM. Well-documented biases include payment vehicle bias, starting point bias, part–whole bias, embedding bias and strategic bias (for an explanation see Bateman *et al*., 2002; OECD, 2002; Boardman *et al*., 2006). As a result, leading scholars, such as Sagoff (2000), have expressed doubts about the suitability of CVM. Proponents of CVM acknowledge that CV studies range from very good to very bad and that the technique suffers from various design problems that require effort and skill to resolve (Carson, 2000). However, they believe that extensive research and quality improvements (see, e.g. NOAA, 1993) have already increased the reliability and feasibility of the CVM.

A fundamental problem, as Admiraal *et al*. (2013) show, is that total economic value is inadequate to maintain sustainable use of ecosystem services, as it is based on a utilitarian and opulence perspective. That is, total economic value bases the value of ecosystems on the flow of human benefits from services of ecosystems. Ecologists have argued that some of the underlying structure and functions of ecological systems which precede the ecosystem services cannot be taken into account in terms of economic values (Turner, 2001). Total Economic Value will therefore underestimate the true value of ecosystems. This enabling value of the ecosystem structure has been called 'primary value' and consists of the system characteristics upon which all ecological functions depend. Their value arises in the sense that they produce functions which have value (secondary value). The secondary functions and values depend on the continued 'health', existence, operation, and maintenance of the ecosystem as a whole. The primary value notion is related to the fact that the system holds everything together (and is therefore also referred to as a 'glue value') and as such has, in principle, economic value. Thus the Total Value of the ecosystem exceeds the sum of the values of the individual functions. It can also be argued, according to Turner (2001), that a healthy ecosystem contains an ecological redundancy capacity and there is thus an 'insurance' value in maintaining the system at some 'critical' size in order to counteract stresses and shocks over time.

In their overview of the state of the natural environment in the United Kingdom, Watson and Albon (2011) rejected attempts to measure the total value of ecosystem services, as many of these services are essential to continued human existence, and claimed that total values are therefore underestimates of infinity.

Policy analysis and design

7

7.1 Introduction

If markets function fully effectively, then the outcome is efficient, and there would be little rationale for government involvement or regulation. However, as Chapter 4 demonstrated, the conditions for a fully competitive economy are not satisfied. In response to market inadequacies, economic, social and environmental regulations have been developed.

Economic regulations of various kinds have a long history, as countries have sought to deal with traditional types of market failure such as that associated with monopoly power. Increasingly over the last few decades, the emphasis of regulatory efforts has shifted from economic regulation to social and environmental regulation. Regulatory concerns dominating the policy agenda today involve issues such as emissions of greenhouse gases, consumer protection, the effect of pollution on health, and more generally environmental quality. Moreover, the development of a global economy has created new classes of regulatory problems, as policies to address climate change and the preservation of scarce natural resources assume larger dimensions.

The rationale for regulation in the areas of environmental quality and risk is different from that for economic regulation. Here the issue is that adverse risks are not adequately priced in markets. The fact that there is a market failure does not mean that regulation will be beneficial (see Section 4.3). Market failure simply creates a potential role for government action. If the government action is to be worthwhile, it must be shown that the regulatory policy enhances overall social welfare.

Government regulation takes many forms. Regulations that govern economic behaviour affect pollution decisions, transport rates, prices of different commodities and virtually every aspect of our lives. The regulatory decision is generally based on an assumption that there is some inadequacy in market operation. Nevertheless, economics may play a constructive role in indicating how to approach the choice of regulatory policy. What is the rationale for different kinds of intervention in regulatory contexts? What are the merits of different kinds of regulation? How to choose from among the different alternatives the one that is in society's best interest? This chapter addresses the role of economics in answering such questions.

7.2 Costs, benefits and risks[4]

In any policy context, whether it involves regulation or not, the government must specify the objectives it wishes to promote. At the basic level, these objectives are simply a list of concerns relevant to evaluating the desirability of a policy. Often there is no clear articulation of policy concerns. The advantage of developing a detailed specification of objectives is that one can be more confident that all concerns have been recognised and incorporated in the analytical and policy assessment process. Articulation of objectives is also important to highlight what trade-offs must be made in pursuit of these policy objectives. All policies involve competing concerns, not the least of which is that there are costs. Formulation of objectives and evaluation of a policy with respect to these objectives is useful even if one has adopted an analytical approach, such as cost assessment or risk analysis, which addresses only one component of the problem. Awareness that other important concerns at stake are being ignored may lead to a broader approach to policy.

[4] This section is primarily based on Kip Viscusi, 1997.

The purpose of obtaining an assessment of the merits of policies is to ensure that they have a sound foundation in reality. Most importantly, is society gaining sufficient benefits from these policies to justify the costs that are being imposed? Since these costs are frequently not budgetary costs but instead are borne by third parties, policy makers are usually less aware of these costs than if they were dealing with an expenditure programme. As the costs imposed by regulation continue to escalate, the need for more refined regulatory analyses will increase. Much of the impetus for the increased reliance on analytical judgements comes from the recognition that the costs of regulation are becoming substantial. Some mechanism must be found to ensure that society is gaining as much benefit as it can from these expenditures.

Table 7.1 provides a summary of a number of analytical techniques, ranging from comprehensive attempts to assess the costs and benefits of regulation to more limited techniques. None of these approaches is without limitations, but each of these techniques may illustrate the different dimensions of policy effects that should be considered and how they relate to criteria for sound regulatory policy.

Cost–benefit analysis

Cost–benefit analysis (CBA) is the conventional economic approach to quantifying and evaluating projects. The technique incorporates principles for assessing the net difference between the costs and benefits over the lifetime of an investment. The main criterion for project

Table 7.1 Alternative approaches to regulatory analysis.

Concept	Description	Advantages	Disadvantages
Cost–benefit analysis	Regulation is desirable if estimated benefits exceed the costs.	Reflects both favourable and adverse effects of a regulation and the need to ensure that, on balance, policies are in society's best interest.	Some important benefit components may not be quantified and consequently given less weight. Criterion is less compelling if those adversely affected by a policy are not compensated.
Cost–effectiveness analysis	Calculation of cost per unit of benefit achieved. Policies that can generate the same or greater benefits at no greater cost are preferred.	Eliminates any clearly inefficient policies from consideration and provides an index of the relative efficacy of policies in generating benefits.	Does not resolve the choice of the optimal level of benefits. Criterion is inconclusive when different benefit levels are generated and one policy does not produce greater benefits at less cost.
Cost assessment	Assessment of the costs of regulating businesses, consumers, and workers. May include attempt to ensure that cost levels are not too high.	Attempts to comprehensively determine the total price society is paying for the regulation and provides insight into its economic feasibility.	Does not address the benefits of the regulation or ascertain the extent to which particular levels of costs are warranted by the favourable effects of the regulation.
Risk analysis	Quantitative assessment of the magnitudes of the risk affected by the policy and their associated health consequences.	Provides decision makers with a sense of whether the policy will be effective in significantly reducing risks.	Risk impacts may be diverse and not commensurate. Does not address the costs of achieving risk reduction or assess policy impacts other than risks.

Source: Kip Viscusi, 1997.

appraisal is economic efficiency, which under certain conditions is assured by applying CBA. If applied properly, CBA can play an important role in legislative and regulatory policy debates on protecting nature and its ecosystem services. CBA provides a useful framework for consistently organising disparate information – without double-counting –, which improves the process and outcome of policy analysis. Traditionally, CBA has been defined in terms of the gains and losses to society, so the method helps decision makers evaluate public sector projects or projects with non-market environmental consequences.

Although cost-benefit analysis is a widely practised technique of project appraisal, a number of difficulties are posed by applying it to ecosystem services issues (Perman *et al*., 2011).

- *First*, as mentioned above, many ecosystem services possess the characteristics of public goods. As a result, there are inherent problems in measuring benefits in monetary terms.
- *Second*, determining society's discount rate appears to be extremely difficult, whereas the outcome is usually very sensitive to its precise value.
- *Third*, as CBA is an incremental procedure, it values small changes in ecosystem services.
- *Fourth*, CBA implies that the value of something is always relative to something else. Critics, however, argue that nature possesses intrinsic value. Its value cannot be measured relative to other things.
- *Fifth*, CBA does not consider differences between one person's valuation of nature and another's. The fact that each person's valuation is given the same weight is one of the main criticisms of CBA among ecologists.
- *Finally*, conducting a CBA of a policy having significant ecological implications requires detailed knowledge about ecosystem functioning and complexity as well as about the reversibility or irreversibility of ecological changes. Unfortunately, this knowledge is often incomplete and of a qualitative nature. Traditional CBA is not equipped to address issues of ecological irreversibility and foregone preservation benefits, so adjustments to the technique are required when performing an evaluation of major decisions regarding ecological and environmental issues (see Section 7.5).

Cost–effectiveness analysis

The purpose of cost–effectiveness analysis (CEA) is to create a basis for sound decisions about the allocation of scarce resources. CEA can take two forms. The first is the called the 'least cost method'. Where there are alternative options to achieve a specific target, CEA can be used to assess the cheapest way to achieve that target. The second method is known as the 'constant cost method'. It assumes a fixed budget and seeks the alternative that will result in the maximum effect on a specific target variable from the given resources. Developed in the military, CEA is nowadays widely used in the health and environmental sectors.

CEA is closely related to cost–benefit analysis (CBA). Both CEA and CBA are evaluative tools to compare the advantages (benefits) and disadvantages (costs) of the alternatives under consideration. However, whereas CBA is a decision-making technique used to select alternatives that maximise the economic value to society, CEA is usually preferred when policy makers are unable to measure the benefits. CEA is primarily used as an ex ante tool to evaluate competing alternatives on the basis of their costs and a single quantified objective. As benefits of natural goods and services often have no price tag, it is not surprising that CEA is increasingly used as an evaluation method in the field of environmental policies.

CEA can only give relative answers. Another obvious problem for CEA is that it compares apples and oranges if the benefits that result from alternative activities are not measured in comparable units. Furthermore, although the targets or objectives in some environmental applications of CEA can be simply measured in terms of a particular standard (for example, reduction in tonnes of CO_2 emissions), the objective in other cases, such as ecosystem restoration, may be extremely difficult to define because it is intangible.

Despite these limitations, CEA can serve as a useful guide for evaluating policy scenarios. As it links the outcomes of the ecological indices to their costs, CEA offers the potential – at the ex ante stage of policy making – of identifying financial resource savings. Hence, CEA can reveal useful insights as to how nature policy measures can be implemented efficiently.

Cost assessment
Another approach to policy analysis is to ignore benefits and to focus simply on costs. This is a partial approach that will not provide comprehensive guidance. Yet it does provide some indication of the extent to which society is committing resources to a particular effort. Indeed, it is usually the recognition that costs are potentially consequential and must be evaluated that forms the first step towards countries adopting more highly refined types of analysis. Costs of regulation may be borne by multiple parties. Costs may also be imposed on businesses and their shareholders, while consumers and workers may bear costs that are incorporated in the prices they pay for products and the wages they receive.

Risk assessment
A key element of any policy analysis of a regulation intended to reduce risks to human health or safety, or to the environment, is to determine the magnitude of the risk being addressed. Are the risks of consequence? By how much does the policy reduce the risk? Obtaining some assessment of the degree to which policy improves the health and safety of those whom it is trying to protect is of concern irrespective of the policy objective. Risk analysis focuses on only one aspect of policy effects – the risks that will be reduced. Unlike cost–benefit or cost–effectiveness analysis, there is no assessment of the costs incurred to achieve the risk reduction. Similarly, there is no requirement to calculate all benefit and cost components and to balance societal interests, as in cost–benefit analysis. Thus, risk analysis is more limited in scope than either of these other policy approaches. Nevertheless, risk analysis is important both as a component for more comprehensive policy evaluation and as a decision-making test in its own right.

Risk assessment (calculation of the probability of harm) must not be confused with risk management (strategies for reducing the risk). Risk assessment should be separate from the task of making policy decisions. A sound risk assessment is necessary irrespective of whether the ultimate objective of risk assessment is to incorporate it in the context of a cost–benefit analysis, a cost–effectiveness analysis, or simply an examination of the risk to see whether it is important given the mandate of the regulatory agency.

7.3 Multicriteria analysis (MCA)

The purpose of multicriteria analysis (MCA), or multi-criteria decision analysis (MCDA), is to indicate the best alternative that satisfies a pre-determined set of objectives. It can be used to identify a single most preferred option, to generate a ranking, or simply to distinguish acceptable from unacceptable alternatives. In contrast to CEA, MCA allows the comparison of projects that seek to meet different objectives. For the proposed set of objectives, the policy maker establishes measurable criteria to assess the extent to which the objectives have been achieved. MCA makes explicit the alternative options and their contributions to the different criteria. The technique usually provides an explicit relative weighting system for the different criteria.

The emphasis in MCA on the judgement of the policy maker in establishing objectives and criteria and in estimating the relative importance weights can be a matter of concern. After all, the outcome of MCA is, in principle, affected by the decision maker's own choices of objectives, criteria, weights and assessments in terms of achieving the objects. This is in contrast to cost–benefit analysis (CBA), which is based on the preferences of all the consumers on whose behalf the CBA is being undertaken.

MCA is an open and explicit evaluation technique; the choice of objectives, criteria, scores and weights can be amended if necessary. Besides the potential problem of subjectivity, another limitation of MCA is that it does not reveal whether the implementation of a project adds more to welfare than it detracts. In MCA, there is no necessity that benefits should exceed costs. In other words, unlike cost–benefit analysis (CBA), there is no explicit rationale for a Pareto improvement rule that benefits should exceed costs. Thus in MCA, as is also the case with CEA, the most preferred option can be inconsistent with improving welfare. However, whereas distributional considerations are absent from standard CBA, they can be included in MCA as one or more criteria.

As MCA can be regarded as a tool for analysing complex problems that are characterised by several – often conflicting and contradictory – points of view, it enables policy makers to work towards solving a decision problem where a mixture of monetary and non-monetary objectives must be taken into account. It needs to be stressed that there usually does not exist one scenario that is obviously best in terms of achieving all objectives, as some trade-off is evident amongst the objectives.

Unfortunately, these standardisation procedures do not lead to identical results: the final ranking of scenarios may be influenced by the type of standardisation applied. Nevertheless, despite the dependence on the standardisation procedure, the weighted summation method is a useful instrument for a full ranking of the scenarios and for providing information on the relative differences between them.

It is clear that the ordering of scenarios in MCA depends on (politically determined) weights for the successive criteria. The set of weights incorporates information about the relative importance of the criteria; that is, the weights describe quantitatively how important each criterion is with respect to the other criteria. Obviously, the criterion with the greatest importance is given the highest weight. Establishing subjective weights reflects the preferences of decision makers or interest groups. It is therefore expected that attaching different weights to the criteria will lead to different outcomes of the MCA.

7.4 Negotiations between stakeholders

The techniques described in Sections 7.2 and 7.3 can not only support governments in policy making, but can also guide stakeholders in dealing with potentially conflicting uses of natural resources. Giller *et al*. (Figure 7.1) describe an interdisciplinary and interactive approach for:
- the understanding of competing claims and stakeholder objectives;
- the identification of alternative resource use options, and
- scientific support to negotiation processes between stakeholders.

Central to the outlined approach is a shifted perspective on the role of scientific knowledge in society. When scientific knowledge is understood as entering societal arenas and as fundamentally negotiable, the role of the scientist becomes a more modest one, that of a contributor to on-going negotiation processes among stakeholders. Scientists can, therefore, no longer limit themselves to describing and explaining resource-use dynamics and competing claims, which are the drivers of conflicts. In doing so, they should actively contribute to negotiation processes between stakeholders operating at different scales (local, national, regional and global). Together with stakeholders, they explore alternatives in the field of opportunities that can contribute to more sustainable and equitable use of natural resources and, where possible, design new technical options and institutional arrangements.

7.5 The precautionary principle and the safe minimum standards approach

The ecological effects of economic activities are complex, incompletely understood and subject to variable external influences. This problem is magnified by the fact that the consequences of current losses of nature extend far into the future. Among other things, this leads to information problems, including ignorance not only of environmental processes but also of the identity and personal preferences of those who suffer from ecological losses in the future, as well as about future technologies and resource costs. If economic development projects jeopardise

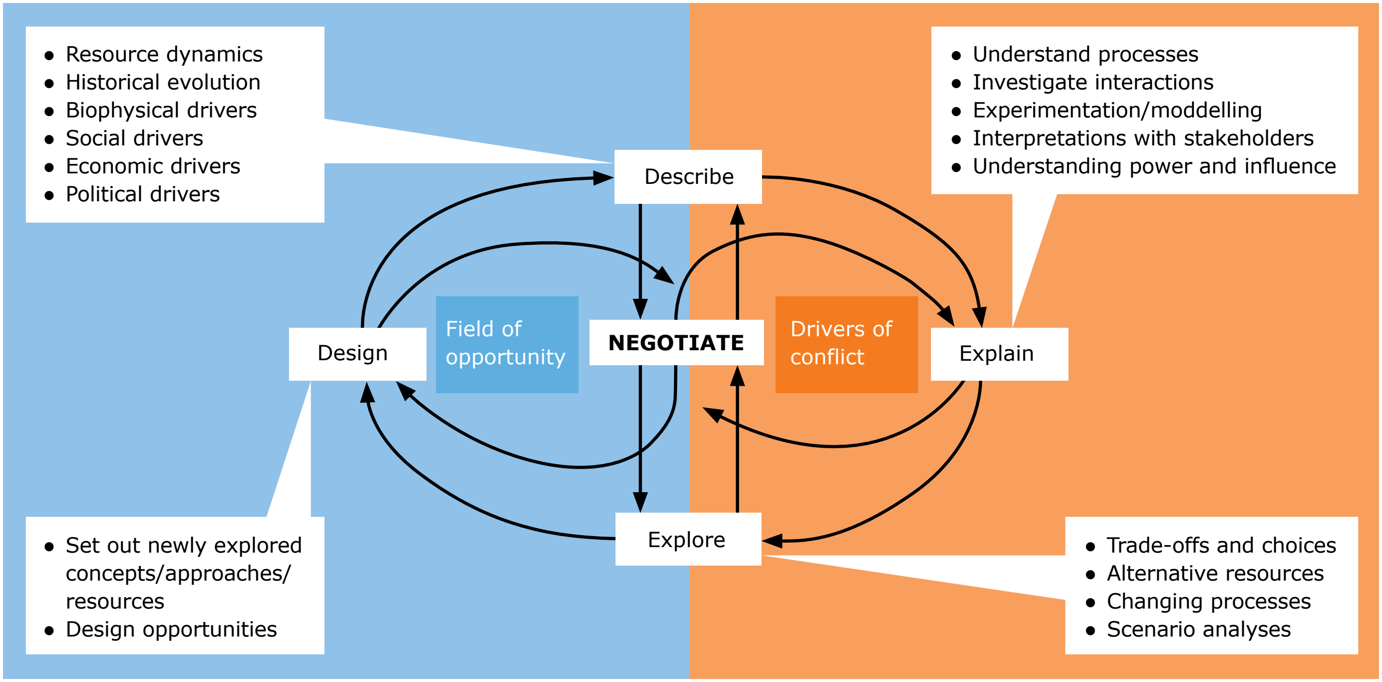

Figure 7.1 Competing claims on natural resources: an iterative cycle of stakeholder negotiated research phases (NE-DEED). Source: Giller, *et al.*, 2008.

ecological systems, economic efficiency is not an appropriate criterion for increasing social welfare.

In extreme cases of scientific uncertainty, the precautionary principle may be rational (Figure 7.2). This principle relates particularly to ecological uncertainty – for example, the evolution of ecosystems, global warming and loss of biodiversity – rather than to economic uncertainty – for example, business cycles and macroeconomic stability (Van den Bergh, 2001). The precautionary principle implies that where significant or irreversible ecological risks are involved, any lack of scientific evidence with respect to cause and effect should not be used as a reason for not taking appropriate action to prevent ecological degradation. For instance, a precautionary approach to biodiversity loss would involve measures to reduce habitat fragmentation, despite uncertainty about the exact extinction rates due to the fragmentation process, or about the (cumulative) effects of species loss on the benefits that human populations derive, directly or indirectly, from them.

The precautionary principle can also be thought of as proposing an approach to target setting. If the sustainability criterion is of overriding importance, then other criteria may be required to do as well as possible in terms of this measure. If this leaves open more than one option, then other desirability criteria can be employed to choose among the restricted options set.

Alternatively, a constraint approach could be adopted, determining the best policy using an efficiency criterion but subject to an overriding sustainability constraint. The difference is that the first approach entails maximising objectives sequentially, whereas constraints only need to be satisfied.

An example of a constraint approach is the safe minimum standard (SMS) of conservation. This has been proposed to prevent major irreversibilities as fully as possible. An SMS approach to nature conservation represents a decision-making principle which suggests that there should be a presumption in favour of not harming the natural environment unless the costs of that approach are intolerably high. Some argue that this concept, which was introduced by Ciriacy-Wantrup in the 1950s and adopted and revitalised by Bishop (1978) in the 1970s, bridges the gap between economists and ecologists (see Spash, 1999).

The SMS defines the level of preservation that ensures survival and implies a conservative approach to risk bearing. In effect, deciding to conserve today can be shown to be the risk-minimising way to proceed given the presence of uncertainty about the consequences of ecosystem losses. In a situation of scientific uncertainty about the consequences of using natural assets, an SMS approach shifts the burden of proof from those who wish to conserve to those who wish to develop (Norton and Toman, 1997). The SMS approach is related to the precautionary principle, but it may permit more scope for economic development. The barriers to economically rational actions that threaten the natural environment are lower under an SMS than when the precautionary principle is adopted (Van Kooten and Bulte, 2000).

The virtue of the SMS approach in circumstances of great uncertainty is that it places natural assets beyond the reach of routine trade-offs. However, the approach also has some problems. Perhaps the most serious limitation of SMS is that the priorities for nature conservation depend solely on the costs of conservation. That is, it disregards the scientific information available about benefits. Furthermore, in order to make the concept of SMS operational, two aspects require special attention: determining the principles that identify an SMS, and specifying what cost level is considered unacceptably high. Decisions regarding these two aspects, however, are arbitrary and thus political. As a result, the outcomes of an SMS approach depend very much on the societal or interest-group preferences of the persons who make these decisions (Turner, 1999). If the decision maker involved and his or her incentives remain the same, SMS is likely to lead to similar decisions as CBA. Both methods, after all, then rely on the same individual subjective preferences.

The precautionary principle represents a supplement to cost–benefit analysis and stresses the uncertainty of decisions about nature, that is, the difficulty of knowing what may be lost. It implies that natural assets have substantial value, although not measurable. The SMS approach emphasises the protection of nature by minimising possible losses to society wherever thresholds of irreversible damage are in danger of being transgressed and there is uncertainty regarding the benefits of nature protection.

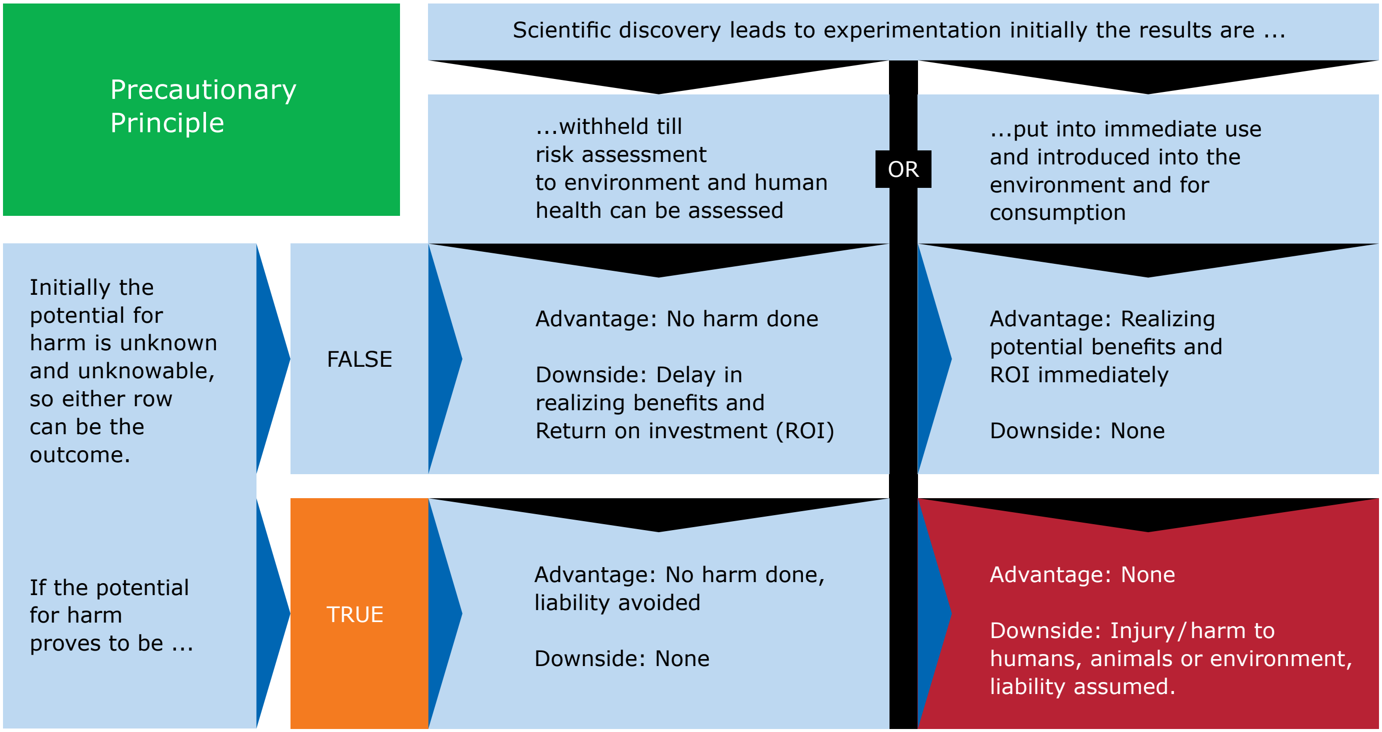

Figure 7.2 The precautionary principle.

Concluding remarks

8

Bridging concept
Economic and natural systems are deeply intertwined. Ecosystem services represent a bridging concept between ecological values and economic values, which has been introduced to determine the different values of ecosystems. TEEB distinguishes four categories: provisioning services, regulating services, cultural services and habitat services (previously denoted as supporting services). This report has highlighted landmarks in economic theories about ecosystems, distinguishing between pre-classical economics, classical economics, neoclassical economics and modern economics. In addition, specific attention has been given to two special branches of economics: (i) natural resource and environmental economics and (ii) ecological economics. These were then linked to economic valuation and to policy analysis and design. This chapter concludes the report with some remarks about the way forward.

Economic valuation
Valuation of ecosystem services is complicated for several reasons. Only some of the ecosystem services have explicit prices. The ecosystem services that are most likely to be managed as commodities traded in markets are provisioning services (e.g. fibre, crops, fish and water). The other categories of ecosystem services – regulating, cultural and habitat – are more difficult to quantify in monetary terms, because – with the exception of a few of them – they have not been integrated in markets. Since they are not fully captured in markets, there is no proper pricing system for many ecosystem services. As a result, much that Thomas Paine has been said about liberty – "what we obtain too cheap, we esteem too lightly" (the quote is from Sagoff, 2008, p. 87) – also holds for most ecosystem services: they may be esteemed too lightly and too superficially. Hence, although there are several reasons for undertaking a monetary valuation of ecosystem services, the raison d'être for this type of valuation is concern about things that are valuable, and in expressing this concern, money speaks louder than words.

Although the number of monetary valuation studies has grown rapidly in recent decades, it is clear that nature and its ecosystem services cannot exclusively be expressed in financial numbers. Ecological processes, functions and structures, on which many ecosystems depend, are impossible to quantify in purely monetary terms. Nevertheless, although economic valuation does not capture all types of value, it is much broader than is usually presumed. As a result, and especially after the controversial and highly debated work of Costanza *et al*. (1997), monetary valuation has brought to ecologists the ability to express the value of nature and biodiversity in metrics that appeal to the public and to policy makers.

Valuation of ecosystem services can contribute to nature policy decisions as it helps decision makers to make better informed decisions about such policies where there are alternative ways of allocating scarce financial and other resources. However, two comments are called for here. The first is that one should acknowledge the limits, risks and complexities involved in valuation. The challenge therefore is a well-considered deployment of research efforts and resources and the search for a wide consensus among scientists as well as decision makers about the valuation method to be used. In other words: valuation is an interdisciplinary and (when applied to actual policy making) transdisciplinary undertaking linking natural science with social science, and as such requires a full range of perspectives on human behaviour. As a result, one has to be aware that valuation studies are usually extremely time-consuming and expensive to undertake. The quantities of data required to facilitate the assessment of costs and benefits are huge, and the complexities of natural systems, particularly in the case of uncertainty and lag effects, may hamper the use of basic economic tools. Understandably, policy makers, who are frequently under pressure to meet short-term goals and therefore tend to focus on short-term decisions, are seldom keen to await the outcomes of such lengthy studies, let alone to finance them. The second comment is that placing 'correct' values on the benefits of nature does not guarantee its protection. After all, the other side of the coin is that the concept of benefits is counterbalanced by the concept of costs, including opportunity costs. Under cost-benefit analysis, it is always possible that the costs of protection are higher than its benefits.

Policy analysis and design

The rationale for regulation with regard to nature and ecosystem services is that adverse risks, such as over-exploitation, are not adequately priced in markets. The standard welfare economics tool for evaluating policies and projects is cost-benefit analysis (CBA). There are, however, a number of difficulties in applying CBA to ecological and nature-related issues. These difficulties are not restricted to the problematic issue of reducing the value of ecosystem services to commensurable monetary units. Another problem is the fact that CBA is an incremental procedure, valuing small and marginal changes, whereas the limits of marginal analysis are apparent in the case of ecosystem service valuation. As an alternative to CBA, cost-effectiveness analysis (CEA) can be applied. This approach usually expresses the effects (or benefits) in physical terms (e.g. the number of species) rather than in monetary terms, while the costs are expressed in monetary units. The objective of CEA is to choose the policy measure that supplies a unit of effect at the lowest cost, or the policy measure that achieves the most effect per unit of cost. Ecological economics has advocated multicriteria analysis (MCA). MCA allows for the multiple dimensions that characterise many decision-making problems related to nature and ecosystem services. In MCA, there is no necessity for benefits to exceed costs. The precautionary principle and the method of safe minimum standards (SMS) suggest that we should err on the side of caution in the face of ecological uncertainty. SMS emphasises the protection of ecosystem services by minimising maximum possible losses to society wherever thresholds of irreversible damage are in danger of being exceeded, and there is uncertainty regarding the benefits of nature protection.

The way forward

The issues and problems of ecosystem services cannot be explained or solved by ecologists or economists alone. Nonetheless, scholars when dealing with ecosystem services tend to speak variously of ecological explanations, economic explanations and other explanations appropriate to the perspective of individual disciplines. Ecological economics might provide a platform for building a unified knowledge base with which problems of maintaining and analysing ecosystems and ecosystem services can be thoroughly addressed and solved. In any case, we are firmly convinced that the advancement of knowledge in this field requires further interdisciplinary cooperation between the natural and social sciences.

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