

On the Economics of Ecosystem Services

Draft 2

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1. Introduction

1.1 Objectives

In economic terms ecosystems must 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 take a long time to recover. Generally speaking, it isn't possible to replace a depleted or degraded ecosystem by a new one. And ecosystems may collapse abruptly, without much prior warning (Dasgupta, 2008).

Because ecosystems are threatened by human activities, it is important to better consider long-term ecosystem health and its role in enabling human habitation and economic activity. To help inform decision-makers, many ecosystem services are being assigned economic values, often based on the cost of replacing such services with anthropogenic alternatives. The on-going challenge of prescribing economic value to nature is prompting shifts in how we recognize and manage the environment, social responsibility, business opportunities, and humanity's future.

History of the concept of "ecosystem services"

One of the first 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 of ecosystem services probably began with Marsh (1864) suggesting that Earth's natural resources were not unlimited by pointing to changes in soil fertility in the Mediterranean. His observations passed largely unnoticed at the time and it was not until the late 1940s that society's attention was again caught by the idea. Several authors promoted 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 of the Study of Critical Environmental Problems in 1970, which listed services including insect pollination, fisheries, climate regulation and flood control. In succeeding years, variations of the term were applied 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 that considered the concept of ecosystem services provides a good 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, M. et al., 2009.

Some cynics claim that the term 'ecosystem services', in addition to the term biodiversity, starts to become another environmental buzzword (Brown et al., 2007) or complexity blinder (Norgaard, 2010). Nevertheless, during the last years, a considerable intellectual development in the understanding of ecosystem goods and services has taken place and interest has grown in refining the analysis and evaluation at various scales (for example, TEEB, 2009; 2010a; 2010b). Moreover, ecosystem services emerge in many national initiatives, such as the UK National Ecosystem Assessment – an advanced interdisciplinary assessment of ecosystems and their services (Watson and Albon, 2010; Bateman et al., 2011).

This report aims at researchers and policy analysts who can benefit from a clear understanding of how ecosystem services relate to economics and policy. It focuses on the economic foundations of the analysis and evaluation of ecosystem services. What do we know from the literature? How can we apply the results in policy oriented studies for government, business and civil society? How to contribute to the policy debate on the provision of ecosystem services? In order to answer these questions, we will borrow heavily from and duly refer to the existing literature on (economic aspects) of ecosystem services.

There is a fast growing number of papers dealing with those services. Although the modern concept of ecosystem services was pioneered several decades ago – it was used in the late 1970s to explain societal dependence on nature – it has received significant attention since the appearance of the Millennium Ecosystem Assessment in 2005 (see also 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) dedicated special issues and special sections to the topic. In July 2012, a new academic journal dedicated to ecosystem services was launched by Elsevier (Braat, 2012). Also entire volumes have been written on ecosystem services (Naeem et al., 2009; ten Brink, 2011).

1.2 Context

Ecosystem services are dynamic. It is useful to consider 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 recognizing the nature of the evidence and the various stages shown in Figure 1.1.

The conceptual 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 provide our overall human well-being (expressed in society as health, wealth and happiness). The values that people receive from ecosystems may alter the way that they choose to use and manage the environment. This in turn leads to further changes in the environment.

Valuing ecosystem services typically requires two sources of knowledge: (i) on the ecological processes, components, and functions that generate these services; and (ii) on the way in which these services translate into specific benefits (Barbier, 2007). In this report, we particularly focus on the second sources of knowledge, which is mainly economic in nature by dealing with issues of scarcity, supply and demand, ownership, and preferences. Moreover, the specific design of the valuation exercise for ecosystem services should depend on its purpose or the role that it will play in the policy process. In the words of Sloomweg and Van Beukering (2008, p. 18) “there are four reasons to value ecosystem services:

- **Advocacy:** economic valuation is often used to advocate the economic importance of the ecosystem services, with the ultimate purpose of encouraging sustainable development. (...)
- **Decision making:** valuation can assist in the government to allocate 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 those goods and services. (...) valuation results can be used to set taxes or charges at the most desirable level.”

Although the economic foundations of the analysis and valuation of ecosystem services are important for each of the four reasons, we focus in this report especially on the goal of decision making. As a concept for better management and provision of ecosystems, ecosystem services rely on the concept being incorporated into wider processes in order to have real-world effects. Evidence is required at a variety of a points (shown in bold in the diagram below) in order to take account of the value of the ecosystem services in the policy / decision making process (DEFRA, 2010).

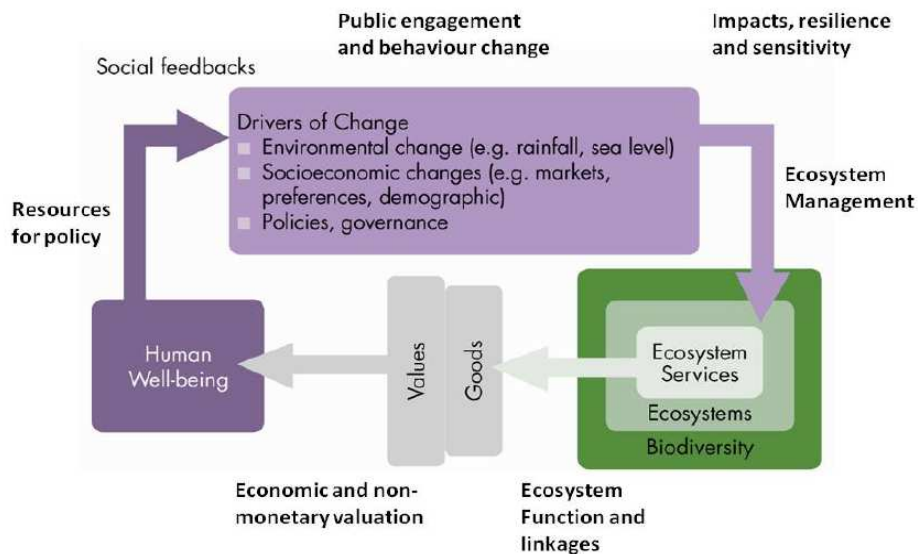


Figure 1.1 Ecosystem services in the policy sphere. Source: DEFRA, 2010

This means that knowledge gathering and sharing between different disciplines and/or different evidence themes is beneficial. 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 refers here to data and methods involved in providing evidence in the six themes. Knowledge sharing will increase the value of the evidence within the following themes (DEFRA, 2010).

- Understanding the value of ecosystems and the goods and services they provide
- Resources to enable others to embed an ecosystems approach in policy and decision making
- Public engagement and behaviour change to positively impact ecosystem services
- Better understanding the science regarding impacts on ecosystems, their resilience and sensitivity
- Management of ecosystems and the practice services they provide
- Examining the linkages/interactions between ecosystems and the services they provide.

Evidence is a prerequisite for effecting change in ecosystem services through policies and projects. This analysis of the evidence requirements and how to achieve them shows how, in practice, ecosystem services is a concept which embodies interdisciplinary working.

In dealing with the economics of the ecosystem services, this report focuses both on the economic valuation, as well as on the resources for policy.

1.3 Structure of this report

The concept of ecosystem services covers a wide variety of costs and benefits of ecosystems. The next chapter describes the modern classification of these services. This is followed by chapters on the history of economic thought. Chapter 3 is about general economics and chapter 4 about the subdisciplines of environmental and ecological economics. Chapter 5 discusses the valuation of ecosystems and their services and chapter 6 the analysis of trade-offs (chapter 6).

2. Classification of ecosystem services

2.1 Introduction

Humans benefit from a multitude of resources and processes that are supplied by natural ecosystems. Collectively these benefits are known as ecosystem services. While scientists and environmentalists have discussed ecosystem services for decades, these services were popularized and their definitions formalized by the United Nations 2004 Millennium Ecosystem Assessment (MEA, 2005). This assessment focused on the contributions of ecosystems to human well-being (an anthropocentric point of view, thus), while at the same time recognizing 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 in his report to the UN General Assembly, *We the Peoples: The Role of the United Nations in the 21st Century*. The MEA was carried out between 2001 and 2005 to assess the consequence of ecosystem change for human well-being, by attempting to bring the best available information and knowledge on ecosystem services 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 scales, 34 regional, national and local scale assessments (or sub-global assessments) were included as core project components. Since the release of the MEA, further subglobal assessments have started.

The MEA categorises ecosystem services into four different classes. These are:

- Provisioning Services which are the products obtained from ecosystems, including food, fibre, fuel, genetic resources, ornamental resources, freshwater, biochemical, 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 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 immediacy of their impact on people. These services include soil formation, photosynthesis, primary production, and nutrient and water cycling.

The publication of the MEA has stimulated widespread, international debate about the importance of the links between ecosystems and human well-being. The MEA found that at global scales, 60% of the ecosystem services on which people depend were being damaged through human action or mismanagement. As a result there is now considerable interest in assessing ecosystem services at regional and national scales. The MEA was unable however to provide adequate scientific information to answer a number of important policy questions related to ecosystem services and human well-being. In some cases, the scientific information may well exist already but the process used and time frame available prevented either access to the needed information or its assessment. In many cases it is clear that either the data needed to answer the questions were unavailable or the knowledge of the ecological or social system was inadequate (VandeWalle et al., 2008).

2.2 Typology of ecosystem services

There are various definitions of ecosystems services in the literature. The most recent revision by TEEB to synthesize work in this field and prevent double counting in ecosystem services audits, has revised the MEA definition to replace "Supporting Services" with "Habitat Services" (TEEB, 2010).

Table 2.1 Typology of ecosystem services in TEEB

Category	Main service types
PROVISIONING SERVICES	1 Food (e.g. fish, game, fruit)
	2 Water (e.g. for drinking, irrigation, cooling)
	3 Raw Materials (e.g. fiber, timber, fuel wood, fodder, fertilizer)
	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 in 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.

Stocks and flows

It is important to distinguish clearly between an ecological stock or an ecosystem service flow (Ash, N. et al., 2010). In general, stocks are expressed in units of quantity (e.g., metric tons, m², or ha), while flows are expressed as quantities per unit time (e.g., kg/year or m³/second). Usually, ecosystem services are flows, both on the supply side and the demand side. Stocks and flows need to balance: if the consumption exceeds the production over a given period, 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. Nonrenewable 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 to be admired by someone else. Even water is not destroyed when it is used: it is typically converted to another form (e.g., somewhat polluted) that 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, 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 (CO₂) from the atmosphere into the ecosystem directly, but the equipment needed 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.

2.3 Ecosystem services and ecosystem dis-services

Ecosystems are a two-edged sword; there exist ecosystem services and dis-services. An example is given by wetlands. The service value of wetlands has long been established, as they provide a suite of benefits from bird habitat to water purification. However, in many parts of the world they are also a source of disease; for example, malaria can easily be classed as a very serious ecosystem dis-service. As dis-services for agriculture Zhang (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 (2008) regarding ecosystem dis-services in urban areas. Bats, rats and foxes in urban parks can cause fear or anxiety. People can also feel unsafe in poorly managed urban green space, especially at night. Disservices can be seemingly inconsequential such as fallen leaves causing increased braking distances and traffic accidents. Foliage along roadsides can decrease visibility (at corners for example), also leading to increased 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 will focus on services, any policy or project analysis must be aware of potential dis-services. Many of the points that will be made regarding services are equally applicable to dis-services.

3. From pre-classical economics to modern economics

3.1 Introduction

This chapter reviews the historic development of the conceptualization of nature and examines critical landmarks in economic theory and practice. A distinction is made between four phases: the pre-classical economics, the classical economics, the neo-classical economics and finally the modern economics.

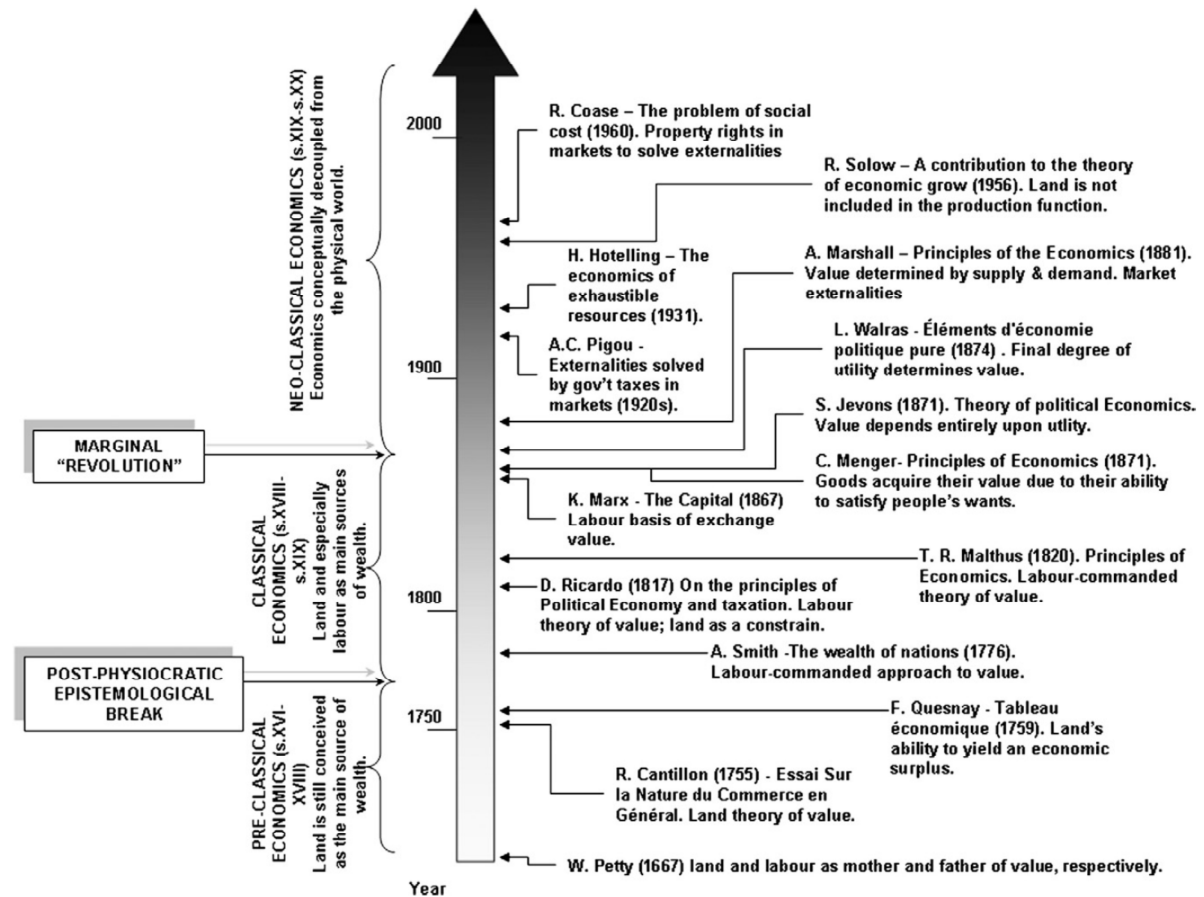


Figure 3.1 Landmarks in the evolving conception of nature by economics

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

In the pre-classical economics the exploitation of land (nature) was conceived as the main source of wealth. In the classical economics land was surpassed by labour as the main source of wealth, although the combination was still seen as crucial. In the neo-classical economics the source of wealth was conceptually decoupled from the physical world (Gómez-Baggethun, E., et al., 2009). In modern economics the environment would come back as an issue of crucial importance for human well-being.

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 that provided a large labor 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 precious 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 (Hubacek and Van den Berg, 2006).

In the 1750s there developed in France a school of economic thought 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 organized 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 is in contrast to 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, made possible cities of several millions, with land values many times greater than agricultural land.

According to the Physiocrats, agriculture was the supreme occupation because it alone yielded a disposable surplus over cost. The agricultural laborers 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 landowners, 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' (Hubacek and Van den Bergh, 2006): "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 in account their labor and expenses as well as the surplus value contributed by the fertility of the soil. Quesnay measured and traced the value of the flow of net 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 (Hubacek and Van den Bergh, 2006).

The influence of the Physiocratic School peaked in the 1760s and declined rapidly thereafter. For most economists, the Physiocrats represent a historical curiosity and 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 biophysical economics.

3.3 Classical economics

Classical economics started at the beginnings 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 for the wealth of nations and the distribution of income amongst the factors of production: land, labor, and capital. The importance of technological progress and capital for productivity and thus economic growth was recognized,

but many classical authors retained from the Physiocrats their special treatment of land (Hubacek and Van den Bergh, 2006).

In contrast to the Physiocrat belief that land was the primary source of value, Classical economists began to emphasize labour as the major force backing the production of wealth. Many of the fundamental concepts and principles of classical economics were set forth in Smith's *An Inquiry into the Nature and Causes of the Wealth of Nations* (1776). When Adam Smith wrote his treatise, only a small number of water-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 Smith agriculture was more productive than manufacturing because it has two powers concurring in its production, land and labor, whereas manufacturing has only one (labor). Division of labor was the main element of productivity increase (Hubacek and Van den Bergh, 2006).

In Adam Smith's theory of value, under competition, a costless item can never have a price. The services of 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, its location, and the transport system.

Classical economists found natural resources worthy of distinct analytical treatment because the services they offer are free. Besides labor (and later also capital), land remained as a separate factor in the production function. Its consideration as a nonsubstitutable production input explains to a degree the emphasis of some Classical economists on physical constraints to growth. This is reflected for instance in (Gómez-Baggethun, E., et al., 2009):

- Ricardo's law on diminishing returns on 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 (labor, capital, etc.) to fixed amounts of the others will yield successively smaller increments of output ("Diminishing Returns").

- Malthus' concerns on population growth;

Robert Malthus believed that natural rates of human reproduction, when unchecked, would lead to geometric increases in population: population would grow in a ratio of 2, 4, 8, 16, 32, 64 and so on. However, he believed that 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 maintained a core position in Classical economic analysis.

Whereas Malthus, Ricardo and others focused on different qualities of land, Johann Heinrich von Thunen used distance as the central concept. Spatial economics and geography claim von Thunen as one of their fathers of their discipline. His concept of diminishing returns is also perceived as a precursor to the marginalist approach of neoclassical economics (Hubacek and van den Bergh, 2006). Von Thunen was interested in the pattern of agricultural production around the central town in an isolated state, in a homogenous featureless plain of equal fertility. He sought 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 rent. He developed a system of concentric circles, in which bulky or perishable goods are produced closer to the city and valuable or durable goods are imported from further distance. In this central town the price of a product like grain is determined by the production and transportation 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 Thunen arrived at similar conclusions as Ricardo in observing that differences in the quality of 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 led nature to lose the distinct analytical treatment it had previously received. Three critical changes can be highlighted (Gómez-Baggethun, E., et al., 2009): a slow move of the primary focus on land and labor towards the factors labor and capital; a move from physical to monetary analysis; and, a move in the focus from use values to exchange values.

3.4 Neo-classical economics

The unifying approach of classical economists was their analysis of values (land, labor and capital) embodied in the 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 long-run, where relative prices were only determined by costs of production. Hence their search for a labor or land content establishing values and prices. A very different orientation was adopted by the new neoclassical school, triggered by Jevons, Marshall, Menger, and Walras, in their search for interdependencies between utilities in consumption and costs in production.

The marginalist revolution, started in the 1870s would have deep effects in 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 fiber- based economies to mineral and fuel-based economies, and economies in the industrialized world that seemed to be almost independent of extractive industries (Hubacek and Van den Bergh, 2006).

By the fall of the Classical economics period some authors kept paying substantial attention to natural resources in physical terms. For instance, in his 1865 book *The coal question*, 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.

Since the accomplishment of the marginalist revolution, Neoclassical economics gradually restricted its analysis to the sphere of exchange values. Quite explicitly in this respect, Pigou (cited by Gomez, ...) 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 elaborate on how technological innovation would allow for increased substitutability between production inputs such as land and capital, eventually consigning concerns on physical scarcity to oblivion (Georgescu-Roegen, 1975). Substitution was elevated to the central principle on the basis of which both the price system and the production system are explained. The neoclassical approach ignores the essential complementarity between different factors of production or different types of activities.

As such, Neo-classical economists consider that different forms of capital (be they natural, man-made, social or financial) are substitutable with one another, which gives technology and innovation an important role as natural capital and its ecosystem services decline.¹ Due to this view, and as stated by Naredo (2003. p. 250): “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 of the world (Gowdy and Ferreri Carbonell, 1999). The result was that the Neo-classical approach has led many economists away from nature. Or, in other words, nature has been ill-served by 20th century mainstream economics (Dasgupta, 2008).

¹ This is the so-called ‘weak sustainability’ approach: contributions of the natural environment to economic activities can be replaced by human made substitutes. In contrast to that approach is the ‘strong sustainability’ approach which says that the existing stock of natural capital must be maintained and enhanced because the functions it performs cannot be duplicated by human made substitutes.

So, by the second half of the 20th century land or more generally environmental resources, completely disappeared from the production function and the shift from land and other natural inputs to capital and labor 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 a common practice in international trade theory to exclude natural resource-intensive products from consideration. For example in the factor proportions theory, which explains the pattern of comparative advantage by inter-country differences in the relative endowment of primary factors of production, the two primary factors of production were capital and labor (Hubacek and van den Bergh, 2006).

Table 3.1 Economic thinking on natural resources

Period	Economic School	Conceptualization of nature	Value-environment relationship
19 th C.	Classical economics	Land as production factor generating rent (income)	Labor theory of (exchange) value Nature's benefits as use values
20 th C.	Neo-classical economics	Land removed from the production function	Land as substitutable/ producible by capital, and thus monetizable

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

This overview of classical and neo-classical economic thinking on natural resources is summarized in table 3.1. The economic conception of nature's benefits as use values in Classical economics has given way to their conceptualization 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 economic discipline could not ignore. Rachel Carson's *Silent Spring* was first published in 1962. In 1974 Lester Brown founded the World Watch Institute as an independent research institute devoted to global environmental concerns. This was quickly recognized by opinion leaders around the world for its foresight and accessible, fact-based analysis.

In economics specialized 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 represent 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 of the noxious byproducts of production itself. In a world, 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. In Boulding's phrase, we act as if we lived in a Cowboy Economy.*”

“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

Economists like Boulding inspired thinking about the economic use of limited material, 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 itself is embedded in the ecosystem. Also, new is the notion of co-evolving processes, which help us understand how natural and social systems interconnect and change (Hubacek and Van den Bergh, 2006).

4. Insights from Environmental and Ecological Economics

4.1 Introduction

In continuing the explanation of the last section about the environmental problem, this chapter presents insights from modern sub-disciplines of economics. First the birth and reasoning of Environmental Economics is sketched. This is followed by a discussion of market failures with respect to the environment. The basic idea is that economic decisions have impacts on the environment influencing social welfare and that these effects should be taken into account properly. Next, the chapter presents the critique on the underlying world view of Environmental Economics by Ecological Economics, arguing that the economy is fundamentally embedded in the natural environment. According to this sub-discipline different ecosystem service values cannot be reduced to a single measuring rod in cost-benefit analysis for decision-making. Instead a framework of multidimensional evaluation analysis is called for.

4.2 Reasoning of Environmental Economics

The discipline of economics tries to analyse how best to fulfil people's unlimited needs and aspirations under scarce resource constraints. As such, economists have studied the original endowment of the earth since the beginning of their discipline. Nevertheless, by the second half of the 20th century, it became clear that the natural resource assets were under increasing pressures. This resulted in the emergence of Environmental and Natural Resource Economics as a distinct field (Crocker, 1999).

The first academic community that specialized in the field of environmental economics gathered around 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 earlier, one of the kick-starters was Rachel Carson's book *Silent Spring* in 1962 that explained how pesticides were causing serious pollution and killing many organisms.

Broadly put, Environmental and Natural Resource Economics (we use hereafter the imprecise but at least brief term Environmental Economics) expands the scope of analysis of orthodox Neoclassical Economics by developing methods to value and internalize economic impacts on the environment into decision making.²

The objective of Environmental Economics is to find the optimal level of an externality (or external effect, or external cost), which follows from striving towards optimal social welfare, or Pareto efficiency (Van den Bergh, 2001).³ As such, traditional Environmental Economics is based on neoclassical welfare theory and microeconomics, in particular on the assumption of rational individual behaviour (utility or profit maximisation).

Environmental Economics is an applied, policy-oriented field of inquiry that "... would not exist as an identifiable sub-discipline of economics if unfettered competitive markets achieved, on their own, economically efficient and socially acceptable allocations of natural resources to economic production and consumption *plus* a clean and healthy environment." (Bergstrom and Randall, 2010, p. vii). So, within Environmental Economics, welfare economics took on the study of the environment, and focused thereby in particular on: (i) the background of the economic system's allocation system failures (where 'failure' means that the system fails to exhaust all potential economic surpluses), (ii) the measurement of the surplus foregone due to these failures, and (iii) the design of allocation systems capable of realising the foregone surplus (Crocker, 1999).

² Although they have theorems in common, and they share the welfare economic framework and methodology, textbooks and scientific articles tend to make distinction between environmental economics and resource economics (Røpke, 2004).

³ Pareto efficiency is defined as a situation in which an improvement in the welfare of any individual cannot be achieved without making at least one individual worse off.

The basic argument underlying Environmental Economics can be summarized as follows: traditional Neoclassical Economics largely neglects the economic contribution of nature by restricting its scope of analysis to those ecosystem goods and services that bear a price. 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. 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 in terms comparable with economic services and manufactured capital (Costanza et al., 1997). 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.

Stated crudely, the core of Environmental Economics is the theory of externalities. Research in the field of Environmental Economics shows that externalities are not exceptional cases, as they are often considered in the economic literature, but pervasive and persistent. Moreover, externalities become progressively more important, due to population and production growth.

Because 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 manage its natural environment optimally. This means that ecosystem goods and services should be considered taking account of all the social costs and benefits. Whether this could be done through the regular operations of the market system, requires a clear and thorough understanding of certain complications associated with the assignment of ownership rights to ecosystem services (who reaps the benefits of nature?). This will be discussed further in the following subsections.

4.3 Market failures

In the presence of externalities, 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. These deviations are described as market failures. These occur when there is an inefficient allocation of resources in a free market due to the lack of a mechanism to account for externalities. There are many different types of market failure. In the context of ecosystem services, especially relevant are externalities and public goods.

Externalities

Using the words of Boardman et al. (2006, pp84-85), an externality can be defined as “an effect that production or consumption has on third parties – people not involved in the production or consumption of the good. It is a by-product of production or consumption for which there is no market.” 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).

Externalities can be positive, i.e. they benefit others, or negative, i.e. they harm others. A positive externality exists when an individual who or a firm that makes 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, with positive externalities, 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 surrounding fields and thus aid farmers. But also: Keeping your yard well maintained helps your house's value and also helps the value of your neighbors' homes.

In order to get consumers to consume more of a good that has a positive externality, a subsidy can be provided. The subsidy 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 the cost the consumer is paying for it. Since consumers make a decision based on where their marginal cost equals their marginal benefit, and since they do not take into account the cost of the negative externality, 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 external costs that exist. These costs are passed on to society. Thus 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 that society faces. Because the supply curve is increased, more of the product is bought than the efficient amount – that is, too much of the product is produced and sold. Since marginal 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 pump pollutants into the air. While the firm has to pay for electricity, materials, etc., the individuals living around the factory will pay for the pollution since it will cause them to have higher medical expenses, poorer quality of life, reduced aesthetic appeal of the air, etc. Thus the production of steel by the firm has a negative cost to the people surrounding the factory – a cost that the steel firm does not have to pay.

Negative externalities are a property rights problem. Who owns the air that the steel mill pollutes? Ronald Coase put forth a solution which is known as the Coase Theorem. 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 individuals that live around the mill will be willing to pay the steel mill not to produce – up to the cost that they are incurring from health care, reduced aesthetic appeal of the air, etc. This amount that they are willing to pay becomes an opportunity cost for the steel mill if they produce. Thus they will cut production to the optimal level. On the other hand, if the people own the air, then the steel mill would have to pay them that same amount for the right to produce. Thus the negative externality is directly added to the steel mill's marginal cost. So, according to Coase Theorem, bargaining and market exchange will lead to an efficient outcome irrespective of how the property rights are distributed (Rose and Kverndokk, 1999; Verhoef, 1999).⁴

Another way to solve the negative externality problem is to simply tax the producer the amount of the negative externality. This adds to the producers marginal cost and will cause them to reduce output. With the aim to correct alleged market failures, the Environmental Economics literature has developed a range of methods to value external environmental costs and benefits. These are dealt with in the next chapter.

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. To summarize briefly, externalities represent a variety of market failures of which public goods form a member. Also Abler (2004, p. 9) shows that externalities and public goods cannot be seen as synonymous. He points out that agriculture produces positive externalities “that do not rise to the level of a pure public good.” As an example, Abler mentions open space, which may increase property values on adjacent parcels of land.

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 4.1, pure public goods have the characteristics of non-rivalry and non-exclusion (for example, Jongeneel and Slangen, 2004; Sandberg, 2007; 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, nature areas protected by or for one agent will benefit everyone else who can access the area (Proost, 1999).

⁴ In practice, however, obstacles to bargaining or poorly defined property rights can prevent Coasian bargaining.

- 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 allocate resources with public good characteristics efficiently. This may be understood by noting that prices do then not signal the true scarcity of the asset (Hanley et al., 1997).

Table 4.1 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 ground)*

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

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

Though many ecosystem goods and services differ from private goods because they possess the characteristics of public goods, it needs to 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 or limit users from these, while one person's consumption reduces resource availability for others (Ostrom, 1999; Ostrom et al., 1999; Steins and Edwards, 1999; Ostrom, 2002; 2003; Berkes, 2008). An 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 to exclude 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 the toll good is restricted to people who pay the producer or the holder of the good, the benefits of a pure public goods are shared by all consumers, whether they paid for them or not.

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, but absent in regard to pure public goods. The reason for the absence of overexploitation in pure public good situations is that one's use of a pure public good, such as climate, 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.

- However this will lead to there being no public good being provided.
- Therefore there will be social inefficiency.
- And thus there will be a need for the government to provide it directly out of general taxation.

Some goods can be public goods as well as private goods. A good 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. So considered, 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 provision of hedges if it is judged that private provision is below the social optimum.

4.4 Criticism from ecological economics

A series of theoretical divergences within the society of Environmental and Resource Economics resulted in the emergence of a new transdisciplinary field: Ecological Economics. It was

institutionalized with the establishment of the International Society for Ecological Economics in 1988 and the launch of the first issue of the journal *Ecological Economics* in 1989 (Røpke, 2005).⁵

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. So, whereas neoclassical economists view nature as a subsystem of the economy, ecological economists base their theorising on the economy's embeddedness in nature (see Figure 4.1). Compared to neoclassical economists, ecological economists have a much more 'natural view' of the world, thereby emphasizing natural laws, interdependencies between sectors and systems and limits to the material growth of the economy.⁶

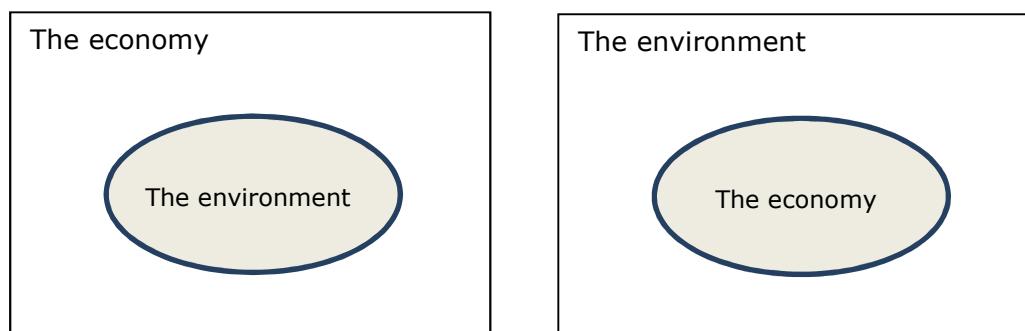


Figure 4.1 Contrasting world views of Environmental (left) and Ecological Economics (right)

Whereas conventional environmental economics applies mainly neo-classical economics 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, sociology, fisheries and wildlife management, and psychology. Moreover, as Baumgärtner et al. (2008, pp. 385, 386) show, 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.' Short overviews of ecological economics are offered by Turner et al. (1997), Spash (1999), van den Bergh (2001), and Røpke (2004; 2005). In addition, Costanza (1991), Costanza et al. (1997a), and Dovers et al. (2003) provide interesting and helpful textbook surveys of the (inter)discipline of ecological economics.

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 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).

In order to take account of interrelations between ecological and economic systems, ecological economics pays special attention to a wider view of values and conflicts; that is, it is characterised by a pluralistic approach to environmental research. This is reflected by the absence of consensus on the type of model to frame the interactions between economic and ecological systems.

⁵ The economists K.E. Boulding, H.E. Daly and N. Georgescu-Roegen – their interests in ecology and economics dates back to the 1960s – and the ecologists C.S. Holling and H.T. Odum are usually regarded as the intellectual founders and antecedents of Ecological Economics. The journal *Ecological Economics* was founded by R. Costanza and H.E. Daly (Van den Bergh, 2001).

⁶ Interestingly, Faber (2008) shows that not only neoclassical economics fail to conceptualise nature in an adequate manner, but that in many other modern sciences (e.g. philosophy) nature has been also neglected.

Neoclassical economists believe that, in principle, infinite growth is possible. Production and consumption can grow forever, because man-made capital can substitute for natural capital. Well-functioning markets are a crucial condition for this idea of perfect substitution between man-made capital and natural resources because they will signal the impending shortage of natural capital, stimulating technological progress to invent a substitute (Solow, 1974).

Classical and Neoclassical Economics theories were developed in a time when man-made capital was considered to be the resource of greatest relevance to the analysis of scarcity – and economics is the discipline that addresses problems of scarcity. Labour and natural resources were relatively plentiful in the early development of Classical and Neoclassical Economics. When theoretical refinement proceeded, raw materials used as inputs in the production process and any other goods and services provided by ecological systems remained omitted from consideration altogether (Williams and McNeill, 2005).

Although the differences between Environmental Economics and Ecological Economics remain controversial (Turner, 1999), they overlap in the use of specific techniques to measure sustainability, evaluate policies and assist decision-making (Gómez-Baggethun et al., 2010). However, the core in Environmental Economics is the theory of (negative) externalities or external costs is, whereas Ecological Economics has sustainable development as its central concept (Van den Bergh, 2001).

"Within Environmental and Resource Economics, sustainable development is usually regarded as being identical to sustainable growth, which is studied with general and abstract models that avoid any reference to historical and spatial aspects, as well as specific characteristics of countries. Environmental and Resource Economics does not seem to take absolute physical limits to growth as seriously as Ecological Economics, and regards the problem of a 'maximum scale' of the economy as irrelevant." (Van den Bergh, 2001, p. 15)

Differences between Environmental Economics and Ecological Economics are – somewhat simplified – summarized in Table 4.2.

Table 4.2 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	System 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).

Environmental Economics operates mainly within the axiomatic framework of Neoclassical economics, formulated in terms of maximization of utility in general, or of profits in particular. It is based on neoclassical assumptions and conditions, such as the theory of consumer choice, perfect and complete information, and marginal productivity theory of distribution (Van den Bergh et al., 2000, Gómez-Baggethun et al., 2010).

With respect to (monetary) valuation, Environmental Economics focuses on value dimensions, in particular on utility and welfare in theory, and costs and benefits in practice. This is quite different

in Ecological Economics, where – unlike Neoclassical economics – a total value of changes in nature or biodiversity is not regarded as the sum of private values. Rather, 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' (Van den Bergh, 2001).

So, in order to take account of interrelations between ecological and economic systems, Ecological Economics pays special attention to a wider view of values and conflicts; that is, it is characterised by a pluralistic approach to environmental research. This is reflected by the fact that there is no consensus on one particular type of model to frame the interactions between economic and ecological systems.

Some ecological economists have a critical attitude towards cost-benefit analysis. Their criticism and opposition is fundamentally based on the fact that environmental decision-making is faced with conflicting valuation languages that may not be commensurable in monetary terms (Martínez-Alier, 2002). Incommensurability signifies the idea that different types of value may not be expressed in a common measurement unit (Gómez-Baggethun et al., 2010) and relies on the philosophical foundation of weak comparability of values (Martínez-Alier et al., 1998). Ecological Economists maintain therefore critical standpoints towards environmental decision making tools that reduce ecosystem service values to a single measuring rod – e.g., cost-benefit analysis. They rather prefer a framework of multidimensional evaluation analysis, such as multi-criteria analysis (which together with Ecological Economics has gained much popularity).

Moreover, if economic development projects jeopardise ecological systems, the Neoclassical criterion of economic efficiency may be regarded as inappropriate and cost-benefit analysis becomes irrelevant and other concepts and methods of analysis are opened up. Especially in circumstances of great scientific uncertainty, policy makers should err on the side of caution. That is, when uncertainty is extreme, safe minimum standards (SMS) and the precautionary principle are rational, according ecological economists.

5. Valuation of ecosystem services

5.1 Introduction

The valuation of ecosystem services and the development of appropriate valuation methods have aroused considerable interest and controversy in the past decades. Much progress has been made in recent years on this subject, which is considered to be essential for sound policy making on ecosystem services. Adamowicz (2004) found a dramatic increase over the past 40 years in the number of publications using environmental valuation methods. It is noteworthy to mention that the trends in publication rates increased rapidly shortly after the Exxon Valdez oil spill in March 1989.

Decision making by policy makers and nature conservation organisations regarding ecosystem services can benefit from the provision of monetary information on the expected virtues, as it improves the rationale for spending on such policies. That is, the main reason for undertaking monetary valuation of these services is to facilitate cost-benefit analysis and thus to gain insight into the playing field between nature conservation and economic development.

Valuation studies of natural assets have mainly been conducted for species preservation, recreation and water management. Assigning a monetary value to the benefits of, or avoided damages due to, ecosystem services can be done by a number of measurement techniques, based on either observed market behaviour (revealed preferences or indirect methods) or stated preferences (direct methods). Book-length treatments of these measurement techniques include Garrod and Willis (1999), Louviere et al. (2000), Bateman et al. (2002), Mäler and Vincent (2005), and Hanley and Barbier (2009). 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.

Essentially, monetary values placed on ecosystem services are rooted in people's values (Banzhaf, 2010). The contingent valuation method – which is a stated preference approach – has been (most) frequently used for measuring the economic value of natural assets and ecosystem services. A main disadvantage though is that this method will fail for those value categories that the general public has no experience with or knowledge of (Nunes and van den Bergh, 2001).

Representative overviews of important valuation studies are provided by Brouwer et al. (1999), Smith (2000), Nunes and van den Bergh (2001), Nunes et al. (2001), Woodward and Wui (2001), Brander et al. (2006), and Ghermandi et al. (2008). McComb et al. (2006) review a number of databases which contain a myriad of primary valuation studies and that are accessible to the general public through Internet sites.

5.2 Categories of values and the concept of total economic value

Most ecosystem services are characterised by the fact that they have no price tag because they are not fully captured in markets. There are, however, some exceptions to this rule, which are especially related to the provisioning services. Foodstuffs, for example, are generally traded in markets. Nevertheless, the fact that for many ecosystem services no market-based price tags exist does not imply that these services are of no value. In order to take these values properly into account, a framework is required for distinguishing and grouping the various values of an ecosystem. The concept of total economic value is such a framework that has been widely used by economists to quantify the full value of different ecosystem goods and services (in an area). This concept consists of two main elements (Figure 5.1). One element contains 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. In addition, non-use values involve no tangible interaction between the area under consideration and the people who use it for production or consumption. Because non-use values are closely linked to ethical concerns and altruistic motives, they are more amenable to debate than use values.

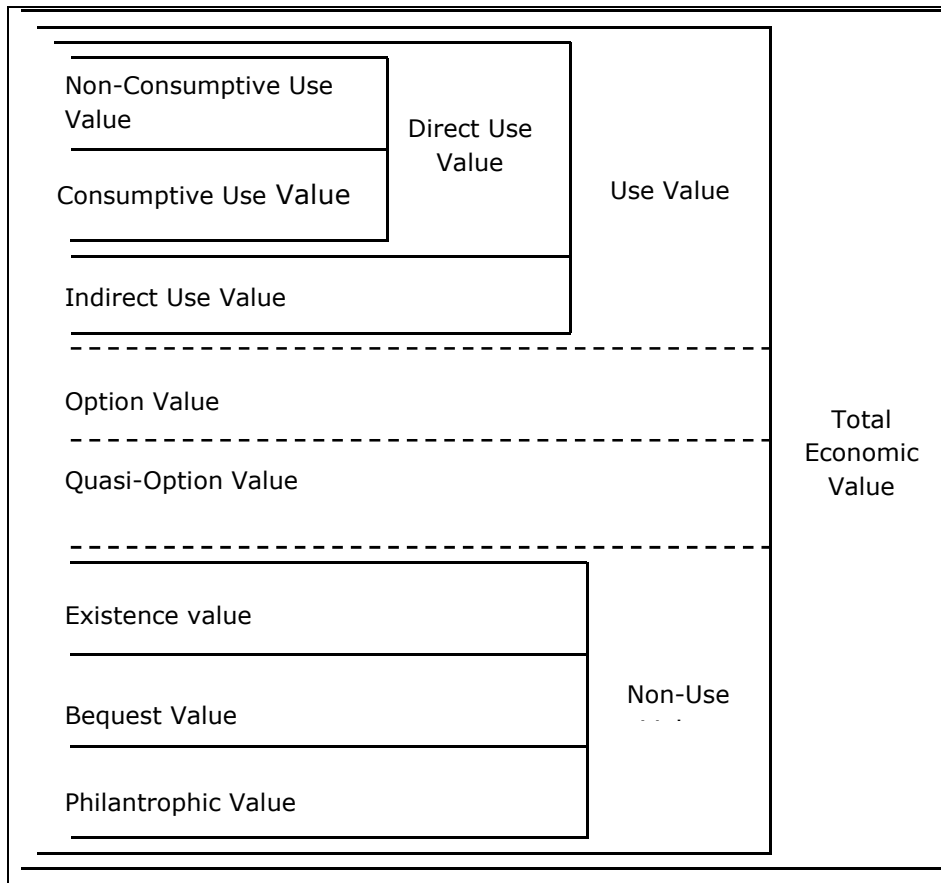


Figure 5.1 The concept of total economic value (TEV) of an ecosystem

Source: Turner et al. (1998, p. 13, Figure 2).

For use values, a separation 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) (Barbier, 1994). Consumptive use values are conceptually clear and offer the best chance 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 they relate, as such, to life-support benefits. Examples of indirect use values include stormwater 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 supporting 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 (Perrings 1995; Moran and Pearce 1997). 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, a complex and ambiguous concept originally introduced by Weisbrod (1964) as relevant for assets that might be difficult to reproduce, has been the subject of considerable debate (Smith 1983). It 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 (Bishop 1982; Smith 1983; Freeman 1984; Ready 1995). If there is uncertainty about the future value of an ecological function, and awaits improved information before giving up the option to protect the asset, then there may be quasi-option value derived from delaying economic activities.

Quasi-option value, a concept forwarded by Arrow and Fisher (1974), is thus simply the expected benefit of awaiting improved information derived from delaying exploitation and conversion of an ecosystem service today. It suggests a value attached to protection given the expectation of the growth of knowledge (Henry 1974; see also Graham-Tomasi 1995). (Note that in Figure 3, both option value and quasi-option value are indicated with a dotted line, since when adding these values possible double counting needs to be taken into account.)

Bequest value is a willingness to pay to keep ecosystem services intact for the benefit of one's descendants, or more generally, future generations.

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).

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 (Barbier 1995; Wills 1997). Empirical estimates, obtained through questionnaires, suggest that existence value can constitute a substantial component of total economic value (Moran and Pearce 1997; Alexander 2000).

The 'Total Value of Ecosystem Services' approach is problematic (Bateman):

- Of no policy use
- Wrong (total value of ecosystem = infinity)

The approach takes the value of a single unit (the 'marginal value') of ecosystem service and multiplies by the total number of units. But as the number of units decreases so their marginal value increases.

Limitations of economic methodology

- If feasible changes are substantial enough then the marginal values will begin to alter
- This links the value of service 'flows' to the size of the 'stock' of ecosystem assets they are provided by
- Need to adjust marginal values for this (tricky)
- Alternative: Asset check
- While most ecosystem services can be valued in economic terms, some are problematic
- While in principle we can estimate the use value of biodiversity (e.g. pollination services, bird-watching, etc.), the non-use existence value of conservation is more difficult to assess.
- Efficient use of resources is always necessary, but especially so in times of austerity
- Integrated environmental economic analysis can target policy so as to optimise efficiency and make the most of scarce resources.
- When Cost Benefit Analysis of environmental policy is not feasible, Cost Effectiveness can help target policy
- Advice: an ecological limits approach with economics confined to identifying cost-effective methods of ensuring safe minimum standards (which may be above present levels).

5.3 Monetary valuation ecosystem services

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), argue 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) 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. However, 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.

Research on the monetary value of ecosystem services can be motivated by the desire to better understand the importance of biodiversity (for a much more elaborate treatment of the need for valuation, see, for example, Simpson 2007). Intuitively, the importance of these services to society is best represented by monetary values. However, reality is more complicated than the common intuition suggests. Economics is more concerned with prices than with values or importance. An important difference between prices and values, which is particularly relevant for this study, is that 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 very measurement techniques to assign a (subjective) value to the benefits of, or avoided damage due to, changes in ecosystem services (Annex 1)

Assigning value to nature by putting a price tag on it is not an end in itself. In fact, it is just a small cog in the much larger and dynamic political machinery that is influenced by a variety of (economic) factors and prevailing cultural and social values. Economists, when dealing with nature and biodiversity, generally adopt an anthropocentric view that focus on instrumental values (in the sense that these values provide information that supports and informs policy making). According to this human-centred view, nature has not value in and of itself. Although economists are aware of the ecocentric perspective – which defines the value of nature intrinsically as possessing value independently of human judgement – they typically concentrate on the question of what nature is worth to humans. Also, monetary values are seen and best conceptualised as subjective and individual-based, because it is the individuals' preferences that define value rather than that these values are objectively given or defined (Ansink et al., 2008).

According to the neoclassical value theory, value is a marginal concept. It refers to the impact of small changes in the state of the world and not the state of the world itself (Nunes and van den Bergh, 2001; Baumgärtner, 2006). Marginal value applies best to a small change in quantity or quality. Therefore, monetary valuation has a greater chance of providing an accurate and reliable estimate of value if the change in nature that is being assessed is small relative to the total level of nature in the geographical area of interest.

A controversial study that does not seem to use marginal values has been published by Costanza *et al.* (1997b), who estimate the current economic value of 17 ecosystem services on a biosphere-wide basis at between US\$ 16 – 54 trillion (10^{12}) per year with an average of US\$ 33 trillion per year. According to the authors, US\$ 33 trillion is 1.8 times global GNP, which they report as US\$ 18 trillion in 1994. Although their paper has stimulated a great deal of debate and generated rich methodological discussions, Costanza *et al.*'s estimates are not supportable for various reasons (for a detailed critique, see Pearce (1998) and several commentaries in a special issue of *Ecological Economics* (Costanza, 1998).

Common errors in economic valuation

While many of the ways in which programs and projects may be assessed are straightforward and make common sense, common errors should be avoided (source: Ash, N. et al. Ecosystems and human well-being : a manual for assessment practitioners):

- 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.

Table 5.2 Suitability of valuation methods for the categories of ecosystem services.

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

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

6. Policy analysis and design

6.1 Tools

Because of the inescapable reality of scarcity, but also because resources – be they financial, human or natural – have alternative uses, the efficient use of these resources requires both consumers and producers to weigh alternatives (that is, to make trade-offs). Economics can provide guidance on how to approach these trade-offs. Environmental economics aims to elucidate the trade-offs between ecological, economic and social systems.

There are a number of tools for the evaluation of alternatives: cost assessments, cost-effectiveness analysis, cost-benefit analysis, social cost-benefit analysis and multi-criteria analysis.

Cost assessment (CA): CA assesses the costs of an alternative on businesses, consumers, and workers. Here benefits are ignored. The assessment may include an attempt to ensure that cost levels are not too high. Advantage is that the approach attempts to comprehensively determine the total price and provides insight into its economic feasibility. The approach is not partial and does not provide comprehensive guidance as benefits are ignored.

Cost-effectiveness analysis (CEA): CEA calculates the cost per unit benefit but does not assign monetary value to objectives. This approach eliminates the difficulty of attempting to value all benefits explicitly, at the same time providing comparisons of the costs of different ways of achieving a particular objective. It is particularly useful in weeding out alternatives that are clearly inferior, as it provides an index of the relative efficacy of policies in generating benefits. Disadvantage is that the approach takes as a given the desirability of achieving a particular benefit, and therefore it does not resolve the choice of the optimal level of benefits.

Cost-benefit analysis (CBA): CBA calculates the total benefits and total costs and if the benefits exceed the costs the alternative should be chosen. The advantage of the approach is that it reflects both favourable and adverse effects. The disadvantage is that some important benefit components may not be quantified and consequently given less weight.

Social Cost-Benefit Analysis (SCBA). SCBA is an instrument facilitating the weighing up of all current and future social advantages and disadvantages of various alternatives. The word 'social' indicates that costs and benefits are analysed and valued from the point of view of society as a whole. The focus is not only on the costs and benefits that can be expressed in monetary terms, but also on the costs and benefits which have not (or not yet) been expressed in monetary terms, relating to all kinds of other matters valued by society, such as the environment.

Multi-Criteria Analysis (MCA): MCA is based on various preferences of criteria (i.e. importance of indicators) that are used for the choice of an alternative. The analysis shows the contribution of the alternatives to the criteria, based on the weights (preferences) that are given.

6.2 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 of achieving 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 those given resources. Developed in the military, CEA is nowadays widely used in the health and environmental sectors (for some environmental-economic examples, see Turner *et al.*, 1994; Macmillan *et al.*, 1998; Ison *et al.* 2002).

CEA is very much related to cost-benefit analysis (CBA). Both CEA and CBA are evaluative tools of comparing the advantages (benefits) and disadvantages (costs) of the alternatives under consideration (van Huylenbroeck, 1988). However, whereas CBA is a decision-making technique used for selecting projects that maximise the economic value to society, CEA is usually preferred when policy makers are unable to monetise the project's benefits. CEA is primarily used as an *ex ante* tool for evaluating competing project 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 fields of environmental and ecological economics.

Because of the lack of a cut-and-dried estimate of the monetary value of the target variable, CEA can be criticised for the limitation that it does not provide an unambiguous answer to whether a project is worth undertaking. In other words, 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. Therefore, in order to make direct comparison of the indices possible, we will determine for each ecological index the effect per euro. Furthermore, in some environmental applications of CEA, the targets or objectives can be simply measured in terms of a certain standard (for example, target reduction in tonnes of CO₂ emissions). However, in other cases, such as ecosystem restoration, the objective is extremely difficult to define because it is intangible (Macmillan *et al.*, 1998).

Despite these limitations, CEA can serve as a useful guide for evaluating policy scenarios. As it links the outcomes of the ecological indices with 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.

6.3 Cost-Benefit Analysis (CBA)

Cost-benefit analysis (CBA) is the conventional neoclassical economic approach to quantifying and evaluating projects (Moran *et al.*, 1996; Drèze and Stern, 1987). The technique incorporates clear principles for assessing the net difference between the net monetary investment costs and benefits over the lifetime of an investment. The main criterion for project appraisal is economic efficiency, which under certain conditions is assured by applying CBA. Nevertheless, 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 greatly improves the process and outcome of policy analysis (Arrow *et al.*, 1996). Traditionally, CBA has been defined in terms of what the gains and losses are to society, and therefore the method offers an aid to decision makers in evaluating public sector projects or projects with non-market environmental consequences (see Hanley and Spash, 1993). It should be realised, however, that CBA cannot replace political judgement.

Although cost-benefit analysis is a widely practised technique of project appraisal, there are a number of difficulties posed by applying it to ecosystem services issues (see Hanley and Spash, 1993; Perman *et al.*, 1996; Pindyck, 2000).

- First, as already mentioned, many ecosystem services possess the characteristics of public goods. As a result, there are inherent problems in measuring benefits and costs in monetary terms.
- Second, determination of society's discount rate appears to be extremely difficult, whereas the outcome is usually very sensitive to its precise value (see, for example, Porter, 1982; Turner *et al.*, 1994).
- 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. Their value cannot be measured relative to other things (OECD, 2002).
- Fifth, CBA does not consider differences between one person's valuation of nature and another's. The fact that each person's valuation receives the same weight is among ecologists one of the most important criticisms of CBA (Goulder and Kennedy, 1997).
- Finally, conducting a cost-benefit analysis of a policy having significant ecological implications requires detailed knowledge of ecosystem functioning and complexity as well as (ir)reversibility of ecological changes. Unfortunately, this knowledge is often incomplete and qualitative in nature (Prato, 1999; Turner *et al.*, 2000). Traditional CBA is not equipped to address issues of ecological irreversibility and foregone preservation benefits, and therefore, adaptations of the technique are required in performing an evaluation of major decisions regarding ecological and environmental issues.

6.4 Multi Criteria Analysis (MCA)

The purpose of multicriteria analysis (MCA), or multicriteria decision analysis (MCDA), is to indicate the best alternative that satisfies a pre-determined set of objectives (van Huylenbroeck, 1988).⁷ 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 reach different objectives. For the set of objectives, the policy maker has established measurable criteria to assess the extent to which the objectives have been achieved. MCA makes the alternative options and their contribution to the different criteria explicit. The technique usually provides an explicit relative weighting system for the different criteria (Janssen and Munda, 1999). The emphasis of MCA on the judgement of the policy maker in establishing objectives and criteria and in estimating 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 of achieving the objects.⁸ However, 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.⁹ Extensive reviews of MCA are given in van Huylenbroeck (1988) and Janssen (1991); for a technical introduction to MCDA, see Vincke (1992).

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 a nature policy maker to advance in solving a decision problem where a mixture of monetary and non-monetary objectives must be taken into account.¹⁰ It needs stressing, however, that there does not exist, in general, one scenario that will be obviously best in achieving all objectives, because 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 complete ranking of the scenarios and for providing information on the relative differences between them.

It is clear that in MCA the ordering of scenarios is dependent 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 has the highest weight. Establishing subjective weights reflects decision makers' or interest groups' preferences. It is therefore expected that different weights on the criteria will lead to different outcomes of the MCA.

OEII method (Ministry of Transport and Water Management / Ministry of Finance, 2000, Evaluation of infrastructure projects, Manual for cost-benefits analysis, Research programme Economic Consequences of Infrastructure). This does not permit the financial valuation of many landscape functions.

Witteveen en Bos has developed the 'Kentallenboek' (Ministry LNV, 2006, Kentallen waardering Natuur, Water, Bodem en Landschap).

⁷ Although the terms 'multicriteria analysis' and 'multi-attribute analysis' are often used synonymously in the literature, we prefer to use the former. The terms 'multicriteria analysis' (MCA) and 'multicriteria decision analysis' (MCDA) do not have a very distinct meaning. They are frequently used interchangeably.

⁸ 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 (Rietveld, 2002).

⁹ Another important distinction between MCA and CBA is that the risk of double counting is much smaller in CBA than in MCA (Rietveld, 2002). Moreover, whereas distributional considerations are absent in standard CBA, they can be included in MCA as one or more criteria.

¹⁰ In order to deal with effective decision making on complex problems, such as environmental problems, various computer programmes that are easy to use have been developed to assist the technical aspects of MCA (see, e.g., Janssen *et al.*, 2000).

6.5 Safe minimum standards approach and the precautionary principle

The fundamental policy goal for which economic values of natural assets needs to be considered carefully is priority setting (OECD, 2002). Saving all biological resources is impossible, if only for a lack of funding. Setting protection priorities is thus inevitable, leading to the question how most species can be supported or how a given amount of biodiversity can be protected against least cost. It is often emphasised that a great deal of effort and many financial resources should be directed at the scarcest or most threatened species (see, for example, Myers *et al.*, 2000).

However, overemphasis on the degree of threat of extinction ignores the reason why the biodiversity is severely threatened in the first place. Hence, if policy measures do not address the fundamental causes of extinction, the allocation of resources to conservation is likely to be wasteful. Economists therefore maintain that a more sound conservation strategy would focus on the largest amount of conservation for a given level of expenditure; that is, they suggest an approach based on cost-effectiveness rather than scarcity (Moran *et al.*, 1997). Given the fact that financial resources are limited, the concept of opportunity cost occupies the central place. As any practical decision-making criterion has to account for the benefits that are sacrificed by nature conservation, it is a task of economists to assist in setting priorities among alternative nature policy and management options. This involves a formal procedure, notably cost-benefit analysis (OECD, 2002).

Cost-benefit analysis (CBA) is the conventional neoclassical economic approach to quantifying and evaluating projects (Moran *et al.*, 1996; Drèze and Stern, 1987). The technique incorporates clear principles for assessing the net difference between the net monetary investment costs and benefits over the lifetime of an investment. The main criterion for project appraisal is economic efficiency, which under certain conditions is assured by applying CBA. Nevertheless, if applied properly, CBA can play an important role in legislative and regulatory policy debates on protecting nature and biodiversity. CBA provides a useful framework for consistently organising disparate information – without double-counting –, which greatly improves the process and outcome of policy analysis (Arrow *et al.*, 1996). Traditionally, CBA has been defined in terms of what the gains and losses are to society, and therefore the method offers an aid to decision makers in evaluating public sector projects or projects with non-market environmental consequences (see Hanley and Spash, 1993). It should be realised, however, that CBA cannot replace political judgement.

Although cost-benefit analysis is a widely practised technique of project appraisal, there are a number of difficulties posed by applying it to ecological and nature issues (see Hanley and Spash, 1993; Perman *et al.*, 1996; Pindyck, 2000). First, as already mentioned, many of the natural assets possess the characteristics of public goods. As a result, there are inherent problems in measuring benefits and costs in monetary terms. Second, determination of society's discount rate appears to be extremely difficult, whereas the outcome is usually very sensitive to its precise value (see, for example, Porter, 1982; Turner *et al.*, 1994). Third, as CBA is an incremental procedure, it values small changes in the stock of biological diversity. Fourth, CBA implies that the value of something is always relative to something else. Critics, however, argue that nature possesses intrinsic value. Their value cannot be measured relative to other things (OECD, 2002). Fifth, CBA does not consider differences between one person's valuation of nature and another's. The fact that each person's valuation receives the same weight is among ecologists one of the most important criticisms of CBA (Goulder and Kennedy, 1997). Finally, conducting a cost-benefit analysis of a policy having significant ecological implications requires detailed knowledge of ecosystem functioning and complexity as well as (ir)reversibility of ecological changes. Unfortunately, this knowledge is often incomplete and qualitative in nature (Prato, 1999; Turner *et al.*, 2000). Traditional CBA is not equipped to address issues of ecological irreversibility and foregone preservation benefits, and therefore, adaptations of the technique are required in performing an evaluation of major decisions regarding ecological and environmental issues.

Krutilla and Fisher (1975) proposed an approach to handle the irreversible effects that many economic development projects have. Consider, for example, a project to build an industrial estate in a nature area. The net benefits of development are then usually calculated by the gross benefit from the project less the costs associated with the project. All projects with a the net present value (NPV) greater than zero adds to the welfare of society and should, in principle, be undertaken. The traditional calculation of this net present value of the project at time t is given by

$$NPV(D) = -C + \int_{t=0}^{\infty} D e^{-it} dt = -C + \frac{D}{i} \quad (6.1)$$

where C is the initial costs of the project, i is the social rate of time preference and D is an infinite, constant series of benefits D per annum. However, the development of an industrial estate destroys any benefit society might have derived from the continued nature area in an undisturbed state. Therefore, the foregone benefits of destruction of the environment as a natural asset should also be included. So, the present value calculation must then include the foregone flow of benefits of protection, P . Now, the NPV becomes

$$NPV(D) = -C + \frac{D}{i} - \int_{t=0}^{\infty} P e^{-it} dt = -C + \frac{D}{i} - \frac{P}{i} \quad (6.2)$$

Rearranging equation (3.2), the development project is only admissible if

$$\frac{D - P}{i} > C, \text{ or } \frac{D - P}{C} > i \quad (6.3)$$

Benefits of protection are likely to increase over time as the natural environment decreases in quantity, so that its 'scarcity rent' increases. In addition, the demand for nature can grow, for example, due to an increase in income and expenditures, higher education levels, and improved knowledge and understanding of the importance of ecosystems and biological diversity. The benefits of protection may thus also grow, e.g. at a positive annual rate, r . Benefits of economic development will neither be perpetual. Especially the force of continuous technological progress will reduce the costs of existing, or even introduce entirely new, competitive technologies, thus eroding the benefits of the current economic project. This is represented by the annual rate of decline in development benefits, g .

Thus, assuming that g is negative and r is positive and including these two annual rates in equation (3.3), the net present value is

$$NPV(D) = -C + \int_{t=0}^{\infty} D e^{-(i+g)t} dt - \int_{t=0}^{\infty} P e^{-(i-r)t} dt = -C + \frac{D}{(i+g)} - \frac{P}{(i-r)} \quad (6.4)$$

Equation (3.4) is a statement of the Krutilla-Fisher model of irreversible development. The Krutilla-Fisher approach ensures that the benefits of nature protection are correctly included in the basic cost-benefit equation. As Porter (1982) shows, for given values of P , D , g and r , the sign of the present value depends on the discount rate i that is used. Economic development will only be profitable if certain discount rates are adopted. High discount rates simply reduce the value of D , i.e. the benefits the development project yields are too heavily discounted to offset the initial cost of the project. Low rates result in preservation benefits to dominate.

The Krutilla-Fisher model is an interesting approach to irreversible economic development. Another line of economic thinking on irreversible changes has been followed by Arrow and Fisher (1974), who developed the concept of a 'quasi-option value', the expected value of the information derived from delaying exploitation (see also Henry, 1974). As such, it is based on the uncertain future benefits, rather than on discounted net benefits. The essence of quasi-option value is that it attaches a value to nature conservation given the expectation of the growth of knowledge (Graham-Tomasi, 1995). Even when more informative research reveals that the maintenance of a nature area is less valuable than initially thought, the mere prospect of getting better information in the future can lead to greater conservation of the area. It should be realised that the option to postpone the economic exploitation has value only because decision makers are assumed to learn about future benefits of conservation by waiting. Moreover, decision makers are assumed to have the flexibility to postpone the exploitation so that they can progress towards fuller information.

The quasi-option value in the environmental literature coincides with the option value of the theory of investment under uncertainty by Dixit and Pindyck (1994) (see Fisher, 2000a). It is dynamic, not dependent on risk aversion and nonnegative. Graham-Tomasi (1995) reviews some of the

literature on quasi-option value and concludes that, although the concept is fundamental to problems of resources use, the difficulty lies in empirical treatments.

There are a number of problems with the Krutilla-Fisher algorithm (see Hanley and Spash, 1993). First, although it is common in empirical studies to approximate the growth rate of preservation benefits by the growth rate in real per capita income, and the rate of depreciation of development benefits by the rate of technological progress, these two growth rates may be different and generally unpredictable. Second, the initial benefits of especially nature conservation are not easily quantifiable and therefore difficult to estimate. Third, there is uncertainty about the preferences of the individual in the future. That is, foregone conservation benefits are measured using the preferences of current individuals, which may not reflect the preferences of future generations. Finally, the Krutilla-Fisher algorithm leads to a rather conservation-oriented rule, one which – due to its CBA basis – is arrived at entirely on the grounds of economic efficiency. Most nature policy programmes, however, also deal with other goals, such as equity and sustainability, and the trade-offs between them.

All in all, the Krutilla-Fisher algorithm is modest in its way of ecological-economic integration. Further adjustments can be undertaken by adding ecological constraints to the approach, reflecting ecological thresholds and current knowledge of the relationship between ecosystem stability and biodiversity. In modelling terms this would then give rise to a constrained dynamic optimisation approach.

The safe minimum standards approach

If economic development projects jeopardise ecological systems, the criterion of economic efficiency may be regarded as inappropriate. The reason for this is that efficiency supposes a level of accuracy of analysis, policy setting, policy implementation, and enforcement that is unrealistic. In the context of nature policy this is particularly the case as the ecological effects of current economic activities are complex, incompletely understood and subject to variable external influences outside the control of humans – think of climatic dynamics. This problem is magnified because the consequences of current losses of nature extend far into the future. Among others, this leads to information problems, which cover ignorance about not only environmental processes but also the identity and personal preferences of those who suffer from nature loss in the future as well as about future technologies and resource costs (Wills, 1997). If striving for efficiency is no longer realistic, CBA becomes irrelevant and other concepts and methods of analysis are opened up.

When scientific uncertainty is extreme, a precautionary principle is rational (Gollier *et al.*, 2000).¹¹ This precautionary principle particularly relates to ecological uncertainty – for example, 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; Pindyck, 2000). 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 avoiding taking appropriate action to prevent ecological degradation. For instance, a precautionary approach to biodiversity loss would involve, for example, 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.

In the context of nature policy the safe minimum standard (SMS) of conservation has been proposed to prevent as best as possible major irreversibilities (Perrings *et al.*, 1992; Randall and Farmer, 1995; Crowards, 1998). An SMS approach to nature conservation represents a decision-making principle which suggests that there be a presumption in favour of not harming the natural environment unless the costs of that action are intolerably high (Randall and Farmer, 1995; OECD, 1999). Some argue 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 (Randall, 1988). In effect, deciding to conserve today can be shown to be the risk-minimising way to proceed given the presence of uncertainty about the consequence of nature loss (Tisdell, 1991; Hanley *et al.*, 1997). Due to scientific

¹¹ Various contributions to ecological uncertainty are provided in a special issue of *Resource and Energy Economics* on irreversibilities (Fisher, 2000b).

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 (Randall and Farmer, 1995; Norton and Toman, 1997). The SMS approach is related to the precautionary principle, but it permits more scope for economic development. The barriers to economically rational actions that threaten the natural environment are under an SMS lower than when the precautionary principle is adopted (Wills, 1997; van Kooten and Bulte, 2000).

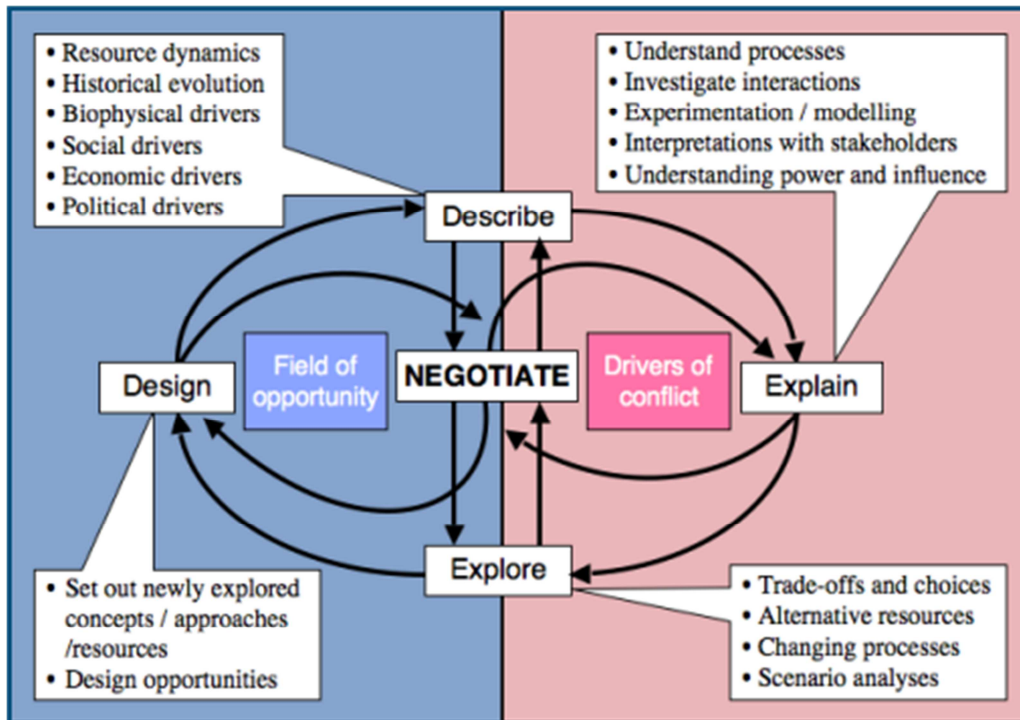
The virtue of the SMS approach in circumstances of great uncertainty is that it places natural assets beyond the reach of routine trade-offs. Unfortunately, the approach also has some problems. Perhaps the most serious limitation of safe minimum standards is that the priorities for nature conservation depend solely on the costs of conservation. That is, it disregards the scientific information available about benefits (Wills, 1997). Furthermore, in order to make the concept of SMS operational, two aspects require special attention: the determination of the principles that identify a safe minimum standard, and the specification of what cost level is considered unacceptably high. Decisions regarding these two aspects, however, are arbitrary and thus political. As a result, the outcomes of a safe minimum standards approach depend very much on the societal or interest-group preferences of the persons who make these decisions (Turner *et al.*, 1999). If the responsible decision maker and his or her incentives remain the same, SMS is likely to lead to similar decisions to CBA. Both methods, after all, then rely on the same individual subjective preferences.

Cost-benefit analysis and the monetary valuation of costs and benefits are controversial. Nevertheless, economists commonly use cost-benefit analysis in their identification and evaluation of public and private projects, and even government programmes, arguing that money should only be spent on a particular item only if the total benefits exceed the total costs. One of the fundamental criteria in cost-benefit analysis is economic efficiency, measured as the difference between benefits and costs. Because society has limited financial resources to spend on nature policy, cost-benefit analysis can inform decisions about how scarce resources can be put to the greatest social good. Such an analysis thus aids in the selection of projects by illuminating the inevitable trade-offs involved in making different kinds of investments. It needs stressing that cost-benefit analysis leaves the final decision to the political process. In this regard, the politician needs not to choose the project or program that is most efficient as calculated by economists.

The Krutilla-Fisher algorithm adds a few ecological considerations to the cost-benefit analysis as it explicitly recognises asymmetric growth rates in economic development and nature protection benefits. The fundamental point of the Krutilla-Fisher algorithm is its rejection of the view that the profitability of a project is an adequate criterion for acceptability when it destroys ecological values. The safe minimum standards approach and the precautionary principle represent other supplements to cost-benefit analysis and stress the uncertainty of decisions about nature; that is, the difficulty of knowing what may be being lost. They both imply that natural assets have substantial value, even if that value is not measurable. The SMS, which allows more scope for economic development than the strict precautionary principle, emphasises the protection of nature by minimising maximum possible losses to society wherever thresholds of irreversible damage are threatened and uncertainty over the benefits of nature protection exists.

6.6 Design

A major challenge for science and policy is to guide stakeholders in dealing with potentially conflicting uses of natural resources. Giller *et al.* describe an interdisciplinary and interactive approach for: (i) the understanding of competing claims and stakeholder objectives; (ii) the identification of alternative resource use options, and (iii) the scientific support to negotiation processes between stakeholders. Central to the outlined approach is a shifted perspective on the role of scientific knowledge in society. Understanding scientific knowledge as entering societal arenas and as fundamentally negotiated, the role of the scientist becomes a more modest one, a contributor to ongoing negotiation processes among stakeholders. Scientists can, therefore, not merely describe and explain resource-use dynamics and competing claims, but 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 that can contribute to more sustainable and equitable use of natural resources and, where possible, design new technical options and institutional arrangements.



The analysis of competing claims on natural resources: an iterative cycle of stakeholder negotiated research phases (NE-DEED).

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Annex 1

Methods for valuing ecosystem services

Non-demand approaches

Following Bateman and Turner (1993), two basic approaches are distinguished: those which value an ecosystem service via a demand curve (see the following subsection), and those which do not. In general, non-demand approaches are based on the principle assumption that if people incur costs to avoid damages caused by lost services (such as the pollination of agricultural crops by bee species) or to replace ecosystem services, then the value of those services are worth at least the amount of money that people have paid to replace them. There are a number of non-demand approaches, such as the opportunity cost approach, the dose response approach, the preventative expenditure approach, the averting behaviour approach, and the replacement cost approach (see, for example, Turner et al. 1994, and Garrod and Willis 1999).

Demand approaches: revealed preference and stated preference methods

Demand curve approaches are broadly divided into revealed preference and stated preference methods. The first type of approach seeks to elicit preferences (for ecosystem services) by examining the purchases of related goods or services in the private market place. In other words, revealed preference methods use market information (such as travel costs or the price of housing) as a proxy to estimate the benefits from a certain (nature) area.

Stated preference methods consider a change in the quantity or quality of ecosystem services and, by using survey techniques, they seek to directly measure the value of these changes (Perrings 1995; Garrod and Willis 1999; van der Heide et al. 2003). As such, these methods derive a value by simulating, or constructing a hypothetical market for the service in question. Although economists are generally much more comfortable with valuations based on actual transactions and observed behaviour in markets than those given in response to survey questionnaires (Heal 2000), for some ecosystem services there are simply no accurate means of inferring preferences from observations. In such circumstances, there are no viable alternatives to asking individuals how much they would be willing to pay for a ecosystem service or accept in compensation for its loss in a hypothetical situation of payment.

Travel cost method

The travel cost method (TCM), a revealed preference method, is one of the oldest valuation methods employed by environmental economists (Clawson and Knetsch 1966). It is especially useful for assessing the value of outdoor recreation in natural parks, and to this end, it has been widely used in the USA and to a lesser extent the UK (Perrings 1995). The underlying assumption of the TCM is that the incurred costs of visiting a national park, nature reserve, open space or any other recreational site are directly related to the benefits individuals derive from the amenities within the area, such as hiking, camping, fishing, bird watching and, swimming. The method involves using the value of time spent in travelling, the cost of travel (e.g., petrol costs) and entrance and other site fees as a proxy for computing the demand price of the ecosystem service. TCM is primarily concerned with recreation and tourism values and application for valuing anything other than recreational values is rather limited. That is, the method is well suited to assessing values related to leisure and recreation – i.e. cultural services – but it is incapable of assessing other ecosystem services, such as flood control or the contribution to climatic stability. Although TCM is uncontroversial and widely used by government agencies in the United States, it has its own limitations, which have been addressed extensively in the literature (for example Bateman 1993; Hanley and Spash 1993; Turner et al. 1994; Smith 1997; OECD 2002). For example, the method is only capable of estimating use values, not non-use values, as these are – by definition – not associated with any measurable activity. It should be observed that the TCM results in a perverse outcome if travelling to and entering area becomes so expensive that no one decides to go there. The method would reveal that the value of the area, or the price the public would be willing to pay to secure this form of land use, is zero (Perrings 1995, Shechter 2000). For an extensive discussion of the TCM, particularly its history and scope, the underlying demand and benefits theory, design principles and administration of surveys, measurement of variables, data management and analysis and various applications we refer to Ward and Beal (2000).

Hedonic pricing method

The hedonic pricing method (HPM) derives the value of environmental amenities from actual market prices of some private goods. Just like the travel cost method, the HPM is based on observed behaviour. By far the most common application of HPM is to the real estate market. House prices are affected by

many factors, not only by house characteristics like the number of rooms and the size of the garden, but also by the environmental quality of the surroundings, including proximity to natural areas and the quality and uniqueness of such areas. If the non-environmental factors can be controlled for, then the remaining differences in real estate prices are expected to be the result of environmental differences (Turner et al. 1994). In principal, HPM is suitable for the estimation of changes in ecosystem services, but is especially appropriate for assessing noise and air pollution (Spash and Carter 2002). A number of problems (including statistical problems such as multicollinearity) are associated with hedonic pricing. These are discussed in detail in Bateman (1993), Hanley and Spash (1993), and Garrod and Willis (1999).

Contingent valuation method

TCM and HPM are both revealed preference methods. A common problem with these methods is that in the absence of appropriate data or interdependent market goods, assessing the value of ecosystem services is either not possible or will lead to spurious results. Stated preference methods bypass the need to refer to market prices by asking people directly what their willingness to pay (WTP) for a change in ecosystem services is. This requires the presentation of a change scenario, for example the conversion of agricultural land into a wetland that filters and regulates water levels, or the establishment of new forests to enhance carbon sequestration.

The contingent valuation method (CVM) is the most used stated preference method: there are now thousands of papers and studies dealing with the topic (Carson 2000; for an overview of fifty years of CVM, see Smith 2004). CVM has been used extensively in the valuation of biological resources, including rare and endangered species, habitats and landscapes, and ecosystem services, although it should be recognised that this method may fail for those value categories that the general public has no experience with (Nunes and van den Bergh 2001). It invokes a framework of a contingent (or hypothetical) market, used to indicate what individuals are willing to pay for a beneficial change or what they are willing to accept by way of compensation to tolerate an undesirable change (Garrod and Willis 1999; Carson 2000; Carson and Hanemann 2005; Boardman et al. 2006).

The main advantage of CVM is that, in theory and to varying degrees, it is capable of capturing most of the value categories related to the functions nature areas. The hypothetical nature of CVM offers flexibility in application, and as a result, the method is quite versatile. Nevertheless, results from CVM studies are heavily dependent on the choice of the particular format used to elicit information about the respondent's preferences. The success CVM has experienced relates directly to the energy, time and money spend on the design state of the contingent valuation survey as well as to the question wording, the question sequencing and the individual interviewers (Diamond and Hausman 1993). Respondents are assumed to be rational and knowledgeable and the best judge of their own interests. Moreover, CVM has many acknowledged problems. The hypothetical character induces the occurrences of an impressive list of potential biases that result from using CVM. They include strategic bias, embedding bias, part-whole bias, starting point bias, and payment vehicle bias (see, for example, OECD 2002; de Blaeij 2003; Boardman et al. 2006).

So although CVM has become one of the most widely used non-market valuation technique, debate persists over the reliability of CVM. That is, opinion among scientists is divided. Many economists have expressed 'discomfort' with using the estimates from contingent valuation to measure consumers' WTP for changes in non-market goods, such as regulating and cultural services (e.g. landscape scenery) and supporting services. Almost twenty years ago, Diamond and Hausman offered the most dogmatic rejection of the method. They (1994, p. 62) tell us that when using a CVM "... one does not estimate what its proponents claim to be estimating." and "people simply do not have preferences over non-use values" (see also Vatn 2004). Also other scholars, such as Kahneman et al. (1990) and Sagoff (2000) express their doubts with respect to the suitability of CVM for various reasons. Nonetheless, the opinions of these (resolute) detractors have not slowed research in this area – on the contrary.

Although 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 (Smith 1992; Hanemann 1994; Carson 2000), they believe that extensive methodological research and quality improvements have already increased the reliability and feasibility of the contingent valuation approach. An important stimulus to the use of CVM was the recommendations of the National Oceanic and Atmospheric Administration's, or NOAA, panel on contingent valuation (see NOAA 1993). This panel of social scientists, co-chaired by two Nobel Prize-winning economists, Kenneth Arrow and Robert Solow, specified an extensive set of guidelines for CVM survey development, administration, and analysis. It found that CVM, if appropriately conducted, could convey useful information. The NOAA panel's recommendations are now being considered as possible standards for employing CVM. As a result, the CV technique is continuing to play a role in monetary valuation, including non-use values, of ecosystem services.

Choice modeling

Choice modelling (CM) is, like CVM, a stated preference method that is capable of measuring the total economic value of a good, and not just the 'use part' of this value. In the field of monetary valuation, CM is being increasingly applied as an alternative to CVM (Adamowicz et al. 1994; 1998). Hanley et al. (2001, p. 436) define CM as "... a family of survey-based methodologies for modelling preferences for goods, where goods are described in terms of their attributes and of the levels that these take." As such, the term choice modelling is somewhat broader in coverage than the method of choice experiments; or, to put it more precisely, choice experiments are a derivative of CM.¹²

CM is capable of measuring consumer acceptance of multi-attribute commodities. Unlike CVM, which tends to provide a single value for an expected (spatial or environmental) quality change, a CM enables estimation of the value of the change as a whole as well as the implicit values of its (spatial) attributes. Generally, in a CM, respondents are confronted with a number of commodity descriptions, or situations, that differ according to the attributes described. Respondents are then asked to rank or rate the bundles of attributes, or select the most-preferred one from the set. The basic premise underlying this method is that commodities, such as wetlands, have a value because of their attributes. In order to decide which commodity they want, people make trade-offs between the attributes (de Blaeij 2003).

The theory underlying the method of CM is Lancaster's model of consumer choice, which hypothesizes that consumers derive satisfaction not from goods themselves, but from the attributes they provide (Lancaster 1966). In addition, CM are consistent with random utility theory which "... is based on probabilistic choice, where individuals are assumed to choose a single alternative which maximizes their utility (welfare) from a set of available alternatives." (Horne et al. 2005, p. 191). A third key element in the method of CM – besides Lancaster's theory of consumer choice and random utility theory – is the experimental design; i.e. the construct used to develop and implement an empirical data framework within which choices can be studied and contextualized (see Garrod and Willis 1999; for thorough and critical descriptions of CM, see Roe et al. 1996; Farber and Griner 2000; Louviere et al. 2000).

CM is designed to determine the structure of preferences that underlie the judgement of multi-attribute commodities and to that end, it measures the rates at which people are willing to make such trade-offs (Shechter 2000). The inclusion of at least one monetary attribute, such as the cost of provision or the WTP for conserving an area, allows for the derivation of implicit prices for each of the other attributes. Of course, it is possible to conduct a CM without the inclusion of a monetary attribute. However, if we want to calculate the welfare measures of a change in ecosystem services, it is necessary to include a monetary attribute such as price or a cost. Speaking broadly and generally, if one of the attributes is measured in monetary terms, CM can be used to estimate the welfare implications of a specific spatial policy. By deriving empirical values of the willingness to pay (for some benefit) or the willingness to accept (compensation for some harm or damage), changes in consumer surplus can be identified as an indicator of changes in welfare resulting from a spatial policy action.

Unlike CVM, CM does not directly ask for a 'willingness-to-pay', but offers the opportunity to rank all of the alternatives from highest (best) to lowest (worst). Defenders of CM therefore argue that the technique outperforms CVM on the point of strategic behaviour. The indirect way of questioning in the CM allows respondents to explicitly determine trade-offs between different attributes of the ecosystem service under valuation. Because CM is designed to rank multi-attribute alternatives, it seems better suitable to value ecosystems, which provide a multitude of joint goods and services, than the typical one-dimensional CVM (Farber and Griner 2000). It allows obtaining values for specific characteristics of scenarios rather than one specific scenario.

¹² The term choice modelling is often used interchangeably with the term conjoint analysis (e.g. Hanley et al. 2001).