SUSTAINABLE BIODIVERSITY: EVALUATION LESSONS FROM PAST ECONOMIC RESEARCH

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Abstract:

Biodiversity has received much attention in environmental research and public policy in recent years. There is a world-wide interest in its relevance for the carrying capacity of rich but fragile ecosystems. Voices suggesting building up proper protection mechanisms for unique and scarce diversity become louder. The question emerges whether – and which combination of – ecological and economic insights can help us to identify meaningful policy options to map out proper roads towards a sustainable future. This paper surveys and highlights the potential and limitations of an ecological-economics perspective on biodiversity. Such a perspective on complex biodiversity issues, if firmly supported by modern ecological insights, can help to clarify the processes, functions and values associated with biodiversity. This study aims to offer a historical review of key ecological and economic concepts that are essential in building bridges between ecology and economics, and discusses ways to integrate them. In addition to such issues as biodiversity indices or ecosystem management principles, particular attention is given to various monetary valuation approaches and methods from the perspective of preservation and sustainable use of biodiversity. Furthermore, the use of ecological and value indicators in integrated economic-ecological modelling and analysis is addressed as well. Throughout the paper, several illustrative applications are presented to demonstrate the usefulness of the various approaches discussed. Finally, the paper offers principles for public decision-making regarding biodiversity protection.

Keywords: biodiversity; biological resources; ecosystems; millennium ecosystem assessment; monetary valuation; species values; integrated model assessment; certification

1. The Ecological Paradigm

Biological diversity has in the past decades become a source of concern for both policy makers and scientists, as well as for the world community at large. The recent study on *The Economics on Ecosystems & Biodiversity* (TEEB 2010) expresses this concern as follows: *"Biodiversity loss and ecosystem degradation continue, despite the fact that policy makers, administrators, NGOs and businesses around the world have been seeking ways to stem the tide. There are many reasons for this, but perverse economic drivers as well as failures in markets, information and policy are significant factors. Markets tend not to assign economic values to the largely public benefits of conservation, while assigning value to the private goods and services the production of which may result in ecosystem damage"*(p. 27). In the same vein, Terborgh (1999) calls for serious action in a study on *Requiem for Nature*. The question addressed in the present article concerns the role of economics in valuing ecosystems' functions and biodiversity in particular.

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The global interest in environmental-economic matters is partly caused by the increased pressure that mounting population and increased production and consumption exert on the earth's natural resource base. In addition, as personal incomes rise and leisure time becomes more freely available in the developed world, concern for more immediate human needs has been accompanied by interest in nature preservation and conservation for future generations. Although resource and environmental issues manifest themselves on local or regional scales, they are part of a globally interwoven ecosystem. Consequently, the 'new scarcity' has spatial and temporal horizons that extend far beyond the current level of thinking and acting (Carraro et al. 2009).

In the past decades, the concept of sustainable development has gained much popularity. It was officially proposed as a policy goal in the publication *Our Common Future* by the World Commission on Environment and Development (Brundtland 1987). The Commission called attention to the need to consider our planet as an integrated social, economic, ecological and political system that requires collective initiatives in order to ensure continuity under changing conditions. The report suggested that economic growth and environmental protection can go together and are not necessarily mutually antagonistic forces. The idea of compatibility between growth and the environment was also critically investigated by Duchin et al. (1994). Using a multisector-multiregion model for the world, they found that economic growth and environmental quality are in conflict, given certain expectations about technological change and innovation. This complex relationship has induced heated debates, but has also prompted much theoretical and conceptual research, while it also has led many empirical applications, in particular in the field of the so-called 'green Kuznets curve' (see for an extensive review De Bruyn 1999). There is an abundance of literature on the question whether a re-orientation in economic thinking is needed to pave the road towards a sustainable future (McKibben 2007).

A prominent issue in recent discussions about sustainable development is the worry about the loss of biological diversity (or biodiversity). Biodiversity requires our attention for two reasons. First, it provides a wide range of direct and indirect benefits to mankind, which occur on both local and global scales. Second, many human activities contribute to unprecedented rates of biodiversity loss, which threaten the stability and continuity of ecosystems as well as their provision of goods and services to mankind. Consequently, in recent years much attention has been directed towards the analysis and valuation of the loss of biodiversity, both locally and globally.

Clearly, the valuation of biodiversity loss can be approached from an ecological, economic or combined perspective (see Polasky and Segerson 2009). The present study addresses all three options. This includes attention to the ecological and economic foundations of biodiversity analysis and valuation. Relevant concepts and valuation methods are identified and discussed. In addition, empirical applications are reviewed. Finally, the study addresses the opportunities offered by multidisciplinary economic-ecological modelling. This allows for the description of the complexity of ecosystem functions and processes, which can be integrated in a transparent way with solid economic valuation approaches.

In order to arrive at this stage, a number of biological, ecological and economic issues and questions need to be dealt with. For example, what are the implications of biodiversity for the structure and functions of ecosystems? Which underlying driving forces influence the loss of biodiversity? Which direct and indirect roles does biodiversity have for human society? Which considerations are relevant in making decisions about the conservation of biodiversity? These are important questions that will guide the present study.

It should be added that valuation and indicator information play a crucial role in assisting policymakers in the design of resource reallocation plans, contributing to ensure the sustainable use of biodiversity. From many studies it is unclear, however, whether the available information always specifically addresses biodiversity. The reason is that biodiversity is often associated with complex ecosystem functions and processes that relate only very indirectly to human welfare. As a result, 'resource valuation' and 'biodiversity valuation' are often confused. In the present study, biodiversity indicator and valuation techniques will be reviewed with a focus on

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Clearly, there are many approaches to studying the relationship between the economy and the environment. This involves linking economics and ecology (Costanza et al. 1997). An important stream is based on materials accounting, using the principle of material balance to describe the chain of extraction, transformation, consumption and emission (Ayres et al. 1999). Another approach has focused on building economic and social accounting systems that can incorporate the measurement of economic welfare together with the measurement of environmental indicators. Here, we will present an ecological economics approach to the study of biodiversity. This is motivated by the fact that biodiversity is a multidimensional concept linked to biological, ecological, cultural and economic entities. A formal mathematical approach to the valuation of biodiversity from an integrated economic perspective can be found in Brock and Xepapadeas (2003). Our contribution differs from other related and earlier studies, such as Barbier et al. (1994), Pearce and Moran (1994), Rapport et al. (1998), and Van Kooten and Bulte (2000), in the following ways: (1) it presents a stronger focus on the analysis of biodiversity rather than ecosystems or natural assets; (2) it offers a multidisciplinary and integrated approach to shed light on the value of particular biodiversity elements; (3) it explores the use of applicable valuation methods and empirical studies; and (4) it analyses the role biodiversity indicators and value information can play in biodiversity policy and management. Some of these other studies provide interesting complementary information, such as on the economics of renewable resources (notably Van Kooten and Bulte 2000).

The sequence of sections in the paper follows the idea that analysis of biodiversity policy involves a number of steps, relating to the identification, measurement and aggregation of biodiversity values. Against this background, the paper is structured as follows. Section 2 provides an exposition on the notion of biodiversity, identifies different levels of life diversity, and discusses alternative perspectives on biodiversity value. Next, Section 3 focuses on general aspects of the economic valuation of biodiversity benefits, offering a discussion on alternative perspectives on biodiversity values. Section 4 then examines which valuation methods can be used for specific value types. In addition, it presents a survey of valuation studies for different levels of diversity, and critically discusses the range of empirical findings. In Section 5 a review is offered of frameworks and methods of integrated modelling of the relationship between biodiversity, ecosystem structure, ecosystem functions, economic activity, and human welfare. Section 6 addresses the implications of the use of biodiversity value information for policy design, devoting special attention to certification and ecolabelling mechanisms, while Section 7 concludes with the role of ecology, economics and their integration in biodiversity analysis.

2. Multilevel Diversity and Types of Biodiversity

2.1. Biodiversity as a Complex Environmental Resource

Biodiversity is a multifaced concept with both ecocentric and antropocentric characteristics showing a great variety in all regions of our world. The analysis of biodiversity is, consequently, rooted in the domain of both the natural and the social sciences. Its modelling implies knowledge of the relationships between biodiversity, the dynamics of ecosystems, and the level of human economic activities. In this context, it is noteworthy that Baumgärtner has argued that measuring diversity presupposes prior value judgements. One reason why biodiversity modelling has been so difficult is the complex and partly unobservable character of these relationships. The geographic diversification in biodiversity and its interrelatedness to socio-economic and physical-climatological conditions make it also difficult to develop and apply an operational methodology for biodiversity analysis. This strand of research is certainly still in its infancy. Essentially, the biological organization base of an ecosystem is made up of three main interrelated classes: (1) biotic resources emerging from the soil or water (such as vegetation and animal populations); (2) abiotic resources with a productive or consumptive nature (such as minerals and energy); and (3) environmental components needed for human wellbeing (such as clean water or fresh air). In general terms, modelling such a complex biological organization has emerged from two streams of ecology, namely community and ecosystem ecology (Holling 1992, Holling et al. 1995, and Schindler 1988, 1990). Community ecology emphasizes the study of the interrelations between species. Some applied studies have, however, called attention to situations where the abiotic environment plays an important role in (re)shaping the relationships of the ecological community. These situations offered a new impetus to ecological thought, giving birth to a second stream in ecology literature: ecosystem ecology. Ecosystem ecology takes biotic and abiotic elements as variable and interactive. For instance, abiotic diversity (e.g. physical characteristics of the landscape such as soil pH and salinity) is expected to be linked to the prevalence of endemic species and thus to biotic diversity and rarity in a natural way (Bertollo 1998). An illustrative example of analysing in a broader context of land use and biodiversity was provided by Haines-Young (2009).

In this context, ecological valuation methods are not only aimed at assessing diversity and rarity of species, but also at assessing the complex interactions between the biotic and abiotic environments, based on the assumption that the variety of abiotic conditions is equally important as the variety of species. In other words, ecosystem ecology aims to identify and characterize the impact of biotic-abiotic interactions on food webs and species interrelations and to assess the role of nutrient flows. Independent of the ecological modelling approach, an important aspect of ecosystem ecology is the recognition that the variability of the biological resources influences the functioning and structure of ecosystems. An interesting and informative overview of the effects of biodiversity on ecosystem functioning can be found in Balvanera et al. (2006) and Hooper et al. (2005).

2.2. Variability of the Biological Resources

There is an abundance of definitions of biodiversity. The United Nations Convention on Biological Diversity (UNEP 1992) defines biodiversity as *"... the variability among living organisms from all sources, including terrestrial, marine and the ecological complexes of which they are part ..."* (art. 2, p. 5). Biodiversity encompasses four levels. At the most basic level is genetic diversity, which corresponds to the degree of variability within species. Roughly speaking, it concerns the information represented by genes in the DNA of individual plants and animals (Wilson 1994). Species diversity refers to the variety of species. Empirical estimates of this are associated with a large degree of uncertainty. In fact, only about one and half million species have been described so far (see Parker 1982 and Arnett 1985), while scientists estimate that the earth currently hosts 5 to 30 million species (see Wilson 1988). Less than half a million have been analyzed for potential economic uses (Miller et al. 1985). Since genetic and species diversity are directly linked, the distinction between them is sometimes blurred. In this sense phenotypic diversity versus genotypic diversity is relevant. Thirdly, ecosystem diversity refers to diversity at a supra-species level, namely at the community level. This covers the variety of communities of organisms within particular habitats as well as the physical conditions under which they live. A long-standing theoretical paradigm suggests that species diversity is important because it enhances the productivity and stability of ecosystems (Odum 1950). However, recent studies acknowledge that no pattern or determinate relationship needs to exist between species diversity and the stability of ecosystems (Johnson et al. 1996). Folke et al. (1996) instead suggest that a system's robustness may be linked to the prevalence of a limited number of organisms and groups of organisms, sometimes referred to as 'keystone species'. It is also possible that the specific relationships depend very much on whether the abiotic environment is stable or not (Holling et al. 1995). Functional diversity refers to the capacity of life-support ecosystems to absorb some level of stress, or shock, without flipping the current ecosystem to another regime of behaviour, i.e. to another stability domain (Turner et al. 1999). This has been originally referred to as 'resilience' (Holling 1973). Unfortunately, a system's functional robustness is still poorly understood and we often do not know the critical functional threshold associated with the variety of environmental conditions at different temporal and spatial scales (Perrings and Pearce 1994). From a management point of view, a safe strategy seems to be to require a minimum level of biodiversity for any ecosystem to be sustained. A low level of ecosystem resilience can cause a sudden decrease in biological productivity, which in turn can lead to an irreversible loss of functions for both current and future generations (Arrow et al. 1995). Finally, functional diversity expresses the range of functions generated by ecosystems, including ecosystem life support functions, such as the regulation of nature major cycles (e.g. water and carbon) and primary ecosystem processes, such as photosynthesis and biogeochemical cycling (Turner et al. 2000). The task of evaluating the structure and functioning of an ecosystem requires that much be known about what the ecosystem does and what that is worth for both biodiversity and for humans. The value of ecosystem structure is generally more easily appreciated than that of ecosystem functioning. Assessing ecosystem functions, such as nutrient retention and pollution absorption for any given region, is extremely difficult. But ecosystem structure is also incompletely known. To assess the value of, for instance, the insect fauna and soil fungi, when many of these species have never even been described taxonomically, pushes human knowledge beyond its current limits (Westman 1985). The preservation of ecosystem processes and their consequent functioning is as important a goal for conservation as is the preservation of ecosystem structure. Ecology has now come to understand ecosystem processes to the extent that some management principles are evident, even if many questions remain unsolved. In recent years, meta-analysis has helped to create a quantitative synthesis of various empirical findings on ecosystems and biodiversity valuation (see Brander et al. 2006, and Nijkamp and Nunes 2008).

2.3. Biodiversity, Ecosystem Services and Human Wellbeing

How important is biodiversity for human wellbeing? The Millennium Ecosystem Assessment (MEA), an international consortium of over 1300 scientists, has focused intensively on the status of biodiversity and ecosystem services because of their contribution to human wellbeing, and has produced several technical volumes as well as thematic summary reports – see MEA (2003, 2005). The MEA wanted to assess the status of ecosystems and ecosystem services (‘the benefits people obtain from ecosystems’) because of their contribution to human wellbeing. This conceptual framework states that biodiversity underpins ecosystems and ecosystem services, which in turn contribute to human wellbeing. Against this background, a new conceptual framework was produced: ecosystem services (including supporting, provisioning, regulating and cultural services) are the cornerstone of wellbeing, which provision, in turn, shall be anchored inter alia in the levels of biodiversity. When subscribing this MEA approach, economic valuation of biodiversity is to be characterized by a three-step approach. The first step is the modelling and assessment of the role of biodiversity in the provision of ecosystem services. The second step is the estimation of the bio-physical impact of changes on the levels of biodiversity on the quantity, and quality, of these ecosystems services. The third, and final, step refers to the welfare assessment of changes in the levels of supply of the ecosystem services, portraying as much as possible these changes in monetary terms. The general acceptance of the MEA framework is a major step in explicitly linking biodiversity, ecosystems and human wellbeing. For the same reason, nowadays it is often proposed to use this framework as a basis to value biodiversity benefits, in particular when working on the policy agenda and management of biodiversity resources. A recent example refers to the 'Potsdam Initiative', which was launched at the G8+5 environment ministers meeting in Potsdam, in March 2007. This meeting called for a study on the economic significance of the global loss of biological diversity, looking at the costs of the loss of biodiversity and the failure to take protective measures versus the costs of effective conservation (see Markandya et al. 2008, Sukhdev 2008). In a more general sense, the MEA issues have led to a plea for a more focussed scientific research effect (see Carpenter et al. 2009).

3. Economic Analysis of Biodiversity Values

3.1. Alternative Perspectives on Biodiversity Values

Given the above described four levels of diversity, it is clear that there is no single notion of biodiversity. Therefore, this section presents various perspectives which suggest that biodiversity value can be interpreted in several ways:

(1) *Instrumental* vs. *intrinsic valuation*. Many people, including biologists and natural scientists, do not feel comfortable with placing an instrumental value on biodiversity. The common argument is that biodiversity has a value on its own, without being used by humans – also know as intrinsic value. A more extreme version of this perspective even claims that that instrumental valuation of biodiversity, often translated in monetary terms, is a nonsense exercise (Ehrenfeld 1988). Many people, however, accept to place a monetary value on biodiversity arguing that, like any other environmental good or service, it is an outcome of an anthropocentric, instrumental point of view, bearing in mind the benefits of biodiversity for humans in terms of its production and consumption opportunities (Fromm 2000). Two specific motivations are the following. First, making public or private decisions that affect biodiversity implicitly means attaching a value to it. Second, monetary valuation can be considered as a democratic approach to decide about public issues. Finally, some subscribe an intermediate attitude by arguing that monetization of biodiversity benefits is possible, but that this will always lead to an under-estimation of the 'real' value since 'primary value' of biodiversity cannot be translated in monetary terms (Gren et al. 1994). As Gowdy (1997) has recently claimed *"... although values of environmental services may be used to justify biodiversity protection measures, it must be stressed that value constitutes a small portion of the total biodiversity value…".*

(2) *Monetary* vs. *physical indicators*. Monetary valuation of biodiversity is anchored in an economic perspective, based on biological indicators of the impacts of biodiversity on human welfare (see Randall 1988). The economic value of biodiversity can be traced to two important sources. First, biodiversity can serve as an input into the production of market goods. For example, the case of bioprospecting, i.e. the search for new pharmaceutical products (Simpson et al. 1996). In addition, biodiversity can be interpreted as a contributor to individual utility or wellbeing. For example, the human pleasure derived from experiencing nature. Economic valuation of biodiversity always leads to monetary values (or indicators), interpreted by economists as a common platform for comparison and ranking of alternative biodiversity management policies. On the contrary, physical assessments of biodiversity value are based on non-monetary indicators. These include, for example, species and ecosystems richness indices (see Whittaker 1960 and 1972), which have served as important valuation tools in the definition of 'Red Data Books' and 'Sites of Special Interest'. It is not guaranteed, however, that monetary and physical indicators point always in the same direction. In this sense, they should at best be regarded as complementary methods for assessment of biodiversity changes.

(3) *Direct* vs. *indirect values*. The notion of direct value of biodiversity is sometimes used to refer to human uses of biodiversity in terms of production and consumption. Conversely, the notion of direct value of biodiversity is associated to a minimum level of ecosystem infrastructure, without which there would not be the goods and services that are provided by the same system (Farnworth et al. 1981). Later on, the term 'indirect value' of biodiversity was proposed by Barbier (1994) and described as *"... support and protection provided to economic activity by regulatory environmental services…"* (p. 156). Nevertheless, in the literature, one can find different terms, such as 'contributory value', 'primary value', and 'infrastructure

value' of biodiversity, to point at the same notion (see Norton 1986, Gren et al. 1994 and Costanza et al. 1998).

(4) *Biodiversity* vs. *biological resources*. Whereas biodiversity refers to the variety of life, at whatever level of interest, *biological resources* refer to the manifestation of that variety. According to Pearce (1999), *"... much of the literature on the economic valuation of 'biodiversity' is actually about the value of biological resources and it is linked only tenuously to the value of diversity…"*. The precise distinction is not always clear, and the two categories seem to be at least overlapping.

(5) *Genetic* vs. *other life organization levels*. Scientists face an important decision when valuing biodiversity: which level of diversity is under consideration? Some scientists, especially from the natural sciences domain, tend to focus on genetic and species levels, whereas others, including social scientists, tend to study species and ecosystems levels. Naturally, such a decision is crucial for the assessment of biodiversity value since it anchors the choice of the most appropriated indicators, a cornerstone of any valuation study of biodiversity.

(6) *Local* vs. *global diversity*. The design of a valuation context involves important decisions about the spatial frame of analysis (Norton and Ulanowicz 1992). Whereas biodiversity loss is usually discussed at a global or worldwide level, valuation biodiversity studies frequently address policy changes or scenarios defined at local, regional or national levels. Although this seems contradicting, it can be argued that biodiversity and its loss are relevant at multiple spatial levels, from local to global

(7) *Levels vs. changes of biodiversity.* One can also focus the assessment on levels of biodiversity. Such a valuation is highly data demanding and trade-offs, the anchor of any economic valuation exercise, will be extremely hard to set. Examples are thus difficult to find in the empirical valuation literature and the ones that exist are often target of a fierce scientific debate (e.g. Costanza et al. 1998). In an extreme perspective, one can always argue that the value of biodiversity is the summed value of the GNPs of all countries from now until the end of the world (Norton 1988). Alternatively, the valuation context can involve the design of policy management options, or scenarios, based on explicit changes in biodiversity levels.

(8) *Holistic* vs. *reductionist approaches*. According to a holistic perspective, biodiversity is an abstract notion, linked to the integrity, stability and resilience of complex systems, and thus difficult to disentangle and measure (Faber et al. 1996). In addition, the insufficient knowledge and understanding of the human and economic significance of almost every form of life diversity further complicates the translation of physical indicators of biodiversity into monetary values. For these reasons, economic valuation of biodiversity is by many scientists regarded as a hopeless task (Ehrenfeld 1988). On the contrary, a reductionist perspective is based on the idea that one is able to disentangle, or disaggregate, the total value of biodiversity into different economic value categories, notably into use and passive use or non-use values (Pearce and Moran 1994).

(9) *Expert* vs. *general public assessments*. Economic valuation starts from the premise that social values should be based on individual values. Therefore, when deciding for a general public valuation context, it is agreed that all taxpayers, with every educational level and varied life experiences, are involved in the valuation exercise. Such a valuation assessment benefits form a clear and legitimate democratic support. Another view assumes that laypersons cannot judge the relevance and complexity of biodiversity-ecosystems functions relationships. Instead, therefore, judgments and evaluation of biodiversity changes in this view should be left to experts, notably biologists. An example of an intermediate 'solution' is to use experts to inform laypersons sufficiently before confronting them with a valuation questionnaire (NOAA 1993).

It is clear that many different biodiversity value perspectives can be distinguished based on the above nine considerations. Evidently, it is crucial to know the perspective being adopted [15]. The next section will clarify this point for the subsequent assessment of empirical valuation studies.

3.2. The Concept of Economic Value

Economic valuation aims to provide a monetary expression of biodiversity values. The reason for this is that the theoretical basis of economic valuation is monetary (income) variation as a compensation or equivalent for direct and indirect impact(s) of a certain biodiversity change on the welfare of humans. Both direct and indirect values, relating to production, consumption and non-use values of biodiversity are considered when pursuing an economic valuation of biodiversity. Explicit biodiversity changes, preferably in terms of accurate physical-biological indicators, should be related to these. Biodiversity changes must be marginal or small for economic valuation to make sense. The economic valuation of biodiversity changes is based on a reductionist approach value. This means that the total economic value is regarded as the result of aggregating various use and non-use values, reflecting a variety of human motivations, as well as aggregating local values to attain a global value, i.e., a bottom-up approach (Nunes and Schokkaert 2003).

Moreover, the economic valuation of biodiversity starts from the premise that social values should be based on individual values, independently of whether the individuals are experts in biodiversity-related issues or not. This can be considered consistent with the democratic support of policies. A more detailed discussion and evaluation of monetary biodiversity valuation is offered in Subsection 3.3.

3.3. A Classification of Economic Values of Biodiversity

It is possible to identify and characterize different value categories of biodiversity. Figure 1 shows a classification of biodiversity values that is the basis for the analysis of valuation studies. A first category, denoted by link $1 \rightarrow 6$, depicts biodiversity benefits that run through ecosystem life support functions and preservation of the ecological structure in natural systems. The diversity of functions generated by ecosystems, in turn, links to the demand for goods and services. This value category can represent, for example, the benefits of flood control, groundwater recharge, nutrient removal, toxic retention, and biodiversity maintenance (Turner et al. 2000). A second biodiversity category, denoted by link $1 \rightarrow 4 \rightarrow 5$, captures the value of biodiversity in terms of natural habitat protection. This can relate, for example, to tourism and outdoor recreational demand. A third value category, denoted by link $2 \rightarrow 5$, captures the benefits of an overall provision of species diversity. This value category represents the indirect value of biodiversity in biological resources in terms of inputs to the production of market goods. Well-known examples are the pharmaceutical and agriculture industries, which use plant and animal material to develop new medicines and new products (Myers 1988, Simpson et al. 1996).

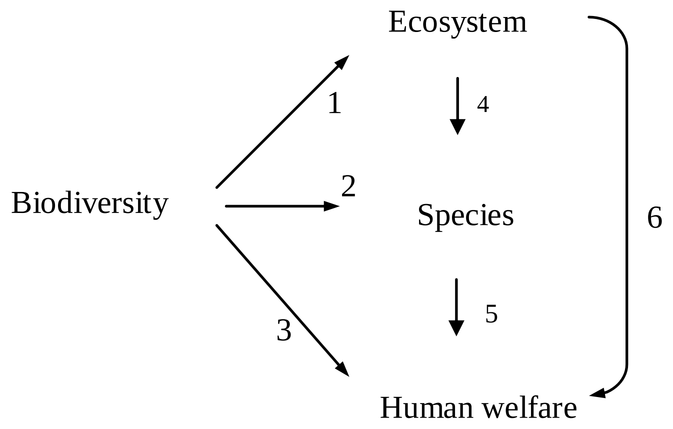

Figure 1. Economic values of biodiversity. Source: Nunes et al. (2001).

Finally, a fourth category, captured by link 3, denotes a passive or non-use component of biodiversity value, which reflects moral considerations to other species (bioethics), human philanthropic, or bequest considerations. The latter relates to the knowledge that biodiversity will be available to the next generations. We refer for a systematic survey to Table 1.

This chapter has tried to clarify various basic economic aspects of biodiversity evaluation. A solid micro-economic welfare-theoretic foundation seems to be a sine qua non condition for a proper biodiversity evaluation. Clearly, in an operational sense, still many advances need to be made. This will be further discussed in Section 4.

Table 1. Total economic value of biodiversity

Biodiversity value category (see Figure 1)	Economic value interpretation	Biodiversity benefits	Methods for economic valuation (and their applicability)
2 → 5	Genetic and species diversity	Inputs to production processes (e.g., pharmaceutical and agriculture industries)	CV: + TC: – HP: + AB: + PF: + Market contracts: +
1 → 4 → 5	Natural areas and landscape diversity	Provision of natural habitat (e.g., protection of wilderness areas and recreational areas)	CV: + TC: + HP: – AB: – PF: + Tourism revenues: +
1 → 6	Ecosystem functions and ecological services flows	Ecological values (e.g., flood control, nutrient removal, toxic retention and biodiversity maintenance)	CV: – TC: – HP: + AB: + PF: +
3	Non-use of biodiversity	Existence or moral value (e.g., guarantee that a particular species is kept free from extinction)	CV: + TC: – HP: – AB: – PF: –

Note: the sign + (–) means that the method is more (less) appropriate to be selected for the design of the valuation context of the biodiversity value category under consideration.
Legend: CV=contingent valuation; TC = travel cost; HP = hedonic pricing; AB = averting behaviour; PF = production function.

4. Measuring Economic Values of Biodiversity Benefits

In this section, we will present a set of illustrative applications of evaluation methods for different types of biodiversity issues.

4.1. Genetic Diversity and Bio-prospecting

Recent years have shown a sharp increase of interest in bio-prospecting, i.e., a search among the genetic codes contained in living organisms in order to develop chemical compounds of commercial value in agricultural, industrial, or pharmaceutical applications (Simpson et al. 1996). This is dominated by pharmaceutical research since most prescribed drugs are derived or patented from natural sources (Grifo et al. 1996). This section considers assessments of willingness to pay by the pharmaceutical industries for genetic diversity as input into commercial products. The marginal value of such input, often translated in terms of genetic information for medicinal purposes, is measured by its contribution to the improvement of health care. For example, research by the US National Cancer Institute on screening of plants over the last two decades has yielded various highly effective anti-cancer drugs (e.g., *paclitazel* and *camptothecin*) and anti-leukemia drugs (e.g., *homoharringtonone*) (Cragg et al. 1998).

Recent registrations and applications of bio-prospecting contracts and agreements between states and pharmaceutical industries represent important benchmarks of monetary indicators for these types of biodiversity values. Illustrations of estimates are shown in Table 2.[1] The most notable of these agreements is the pioneering venture between Merck and Co., the world's largest pharmaceutical firm, and Instituto National de Biodiversidad (INBio) in Costa Rica. At the moment of the contract's signature, in 1991, Merck paid Costa Rica about $1 million and agreed to pay royalties whenever a new commercial product was explored. Since then, INBio has signed contracts on the supply of genetic resources with Bristol-Myers Squibb and other companies and non-profit organizations (Ten Kate and Laird 1999; INBio 2001). Another illustration of the market value of genetic diversity is the commercial agreement signed in 1997 between Diversa, a San Diego-based biotechnology firm, and the US National Park Service. Diversa paid $175,000 for the right to conduct research on heat-resistant microorganisms found in hot springs in Yellowstone National Park (Sonner 1998, Macilwain 1998). More recently, a Brazilian company, Extracta, signed a $3.2 million agreement with Glaxo Wellcome, the world's second-largest pharmaceutical company, to screen 30,000 samples of compounds of plant, fungal and bacterial origin from several regions in the country (Bonalume and Dickson 1999).

Table 2. Valuation of bio-prospecting agreements: examples

Contractors	Study	Value
INBio & Merck (1991)	2,000 samples of the Costa Rica genetic pool	$1 million
Yellowstone National Park & Diversa (1998)	Thermostable enzyme *Taq* polymerase and bacterium *Thermus aquaticus*	$175,000
Brazilian Extracta & Glaxo Wellcome (1999)	30,000 samples of Brazil biota	$3.2 million

Despite the fact that these agreements show a positive economic value of genetic diversity, concern remains about the fairness of such deals. Indeed, some environmental groups have been very critical, claiming that these are unequivocally 'biopiracy' actions (see RAFI 2001). Furthermore, genetic diversity may also give rise to a number of existence and moral values. These however, are not the basis for the pharmaceutical industry's willingness to pay, and therefore are not captured by the market prices of the agreements.

[1] All estimates in Tables 2-7 are in nominal values.

4.2. Species Diversity and Species Values

Most of the valuation studies of species preservation have focused on single animal species. Table 3 lists some recent studies, all applications in the US, except for a Swedish contingent valuation (CV) study of wolves (Boman and Bosdedt 1995). The estimates are derived from CV applications and obtained from the individual willingness to pay (WTP) to avoid the loss of a particular species. Most welfare gains accrued to individuals are based on recreational activities such as watching threatened or endangered species in their natural habitat, or simply reflect the well-being derived from the knowledge that such a species exists. The later case can be interpreted as relating to non-use or passive use values. For example, Van Kooten (1993) assessed the economic value of waterfowls in a wetland region in Canada; Loomis and Larson (1994) valued ‘emblematic’ endangered species, namely the gray whale; and Stevens et al. (1997) valued the restoration of Atlantic salmon in one river in the state of Massachusetts – see Van Kooten and Bulte (2000) for more examples.

Table 3. Valuation of single species

Author(s)	Study	Mean WTP estimates (per household/year)
Stevens *et al.* (1997)	Restoration of the Atlantic salmon in one river, Massachusetts	$14.38 to $21.40
Jakobsson and Dragun (1996a)	Conservation of the leadbeater’s possum, Australia	$29 (Australian $)
Boman and Bosdedt (1995)	Conservation of the wolf in Sweden	700 SEK to 900 SEK
Loomis and Larson (1994)	Conservation of the gray whale, US	$16 to $18
Loomis and Helfand (1993)	Conservation of various single species, US	From $13 for the sea turtle to $25 for the bald eagle
Van Kooten (1993)	Conservation of waterfowl habitat in wetlands region, Canada	$50 to $60 (per acre)
Bower and Stool (1988)	Conservation of the whooping crane	$21 to $141
Boyle and Bishop (1987)	Conservation of the bald eagle and the striped shiner, Wisconsin	From $5 for the striped shiner to $28 to the bald eagle
Brookshire, Eubanks and Randall (1983)	Conservation of the grizzly bear and the bighorn sheep, Wyoming	From $10 for the grizzly bear to $16 for the bighorn sheep

Alternatively, economists can pursue valuation studies of species preservation that focus on more than one species, as shown in Table 4. The estimates are higher than the single species value estimates, though not as high as one would expect, bearing in mind the initial single species estimates. For example, the WTP of the wolf study in Sweden alone corresponds to more than 70 per cent of the WTP for 300 Swedish endangered species.

The interpretation of such estimation results may be, however, heavily criticized because of the CV’s design and execution (see Carson 1997). Nevertheless, some authors prefer to work with other categories of biodiversity value, namely value categories related to natural habitat, ecosystem functions and services flows protection. These are discussed in the following subsections.

Table 4. Valuation of multiple species

Author(s)	Study	Mean WTP estimates (per household/year)
Jakobsson and Dragun (1996b)	Preservation of all endangered species in Victoria	$118 (Australian $)
Desvousges et al. (1993)	Conservation of the migratory waterfowl in the Central Flyway	$59 to $71
Whitehead (1993)	Conservation program for coastal nongame wildlife	$15
Duffield and Patterson (1992)	Conservation of fisheries in Montana rivers	$2 to $4 (for residents) $12 to $17 (for non residents)
Halstead et al. (1992)	Preservation of the bald eagle, coyote and wild turkey in New England	$15
Johansson (1989)	Preservation of 300 endangered species in Sweden	1,275 SEK
Samples and Hollyer (1989)	Preservation of the monk seal and humpback whale	$9.6 to $13.8
Hagemann (1985)	Preservation of threatened and endangered species populations in the US	$17.73 to $23.95

4.3. Species Diversity and Habitat Values

A recurrent problem with the interpretation of the value estimates of species preservation is the frequently missing link between the value assigned to a particular (set of) species and the area needed to protect (their) habitats. Some studies instead, link the value of biodiversity to the value of natural habitat conservation. Some examples are listed in Table 5. For example, Bateman et al. (1992) undertook a contingent valuation study to assess the monetary value of conserving the Norfolk Broads, a wetland site in the UK that covers three National Nature Reserves. The estimation results from a mail survey show that respondents living in a zone defined as 'near-Norfolk Broads' had a WTP of ₤12, whereas those living in the 'elsewhere UK' zone had a WTP of ₤4. In the context of the Netherlands, Hoevenagel (1994) asked 127 respondents for an annual contribution to a fund from which farmers in the Dutch meadow region would receive a government grant if they managed their land in a way that enhances wildlife habitat. The average WTP was between NLG 16 and NLG 45. Brouwer (1995) found similar results.

More recently, Nunes (2002ab) used for the first time a national CV application in Portugal to assess the willingness to pay for the protection of natural parks and wilderness areas. The mean WTP results ranged from $40 to $51. In the US context, Mitchell and Carson (1984) used the CV method to value the preservation of water ecosystems and the aquatic-related benefits provided by all rivers and lakes in the US. Loomis (1989) used CV to value the preservation of Mono Lake, California – see the valuation figures in Table 5. Kealy and Turner (1993) estimated the benefits derived from the preservation of the Adirondack aquatic system. The WTP estimates ranged between $12 and $18. Boyle (1990) valued the preservation of the Illinois Beach Nature Reserve. The estimation results show that the average WTP ranged between $37 and $41. Silberman et al. (1992) studied the existence value of beach ecosystems for users and non-users of New Jersey beaches. The results show that the mean WTP for a user is about $15.1, while the mean WTP for a non-user is about $9.26.

Table 5. Valuation of natural habitats

Author(s)	Study	Mean WTP estimates (per household)
Nunes (2002a,b)	Protection of natural parks and wilderness areas, Portugal	\$40 to \$51
Wiestra (1996)	Protection of ecological agricultural fields, The Netherlands	NLG 35 (single-bounded)
Richer (1995)	Desert protection in California, US	\$101
Brouwer (1995)	Protection of peat meadow land, The Netherlands	NLG 28 to NLG 72
Carson et al. (1994)	Protection of the Kakadu conservation zone and National Park, Australia	\$52 (minor impact scenario) \$80 (major impact scenario)
Hoevenagel (1994)	Enhancing wildlife habitat in the Dutch peat meadow region, The Netherlands	NLG 16 to NLG 46
Kealy and Turner (1993)	Preservation of the aquatic system in the Adirondack region, US	\$12 to \$18
Hoehn and Loomis (1993)	Enhancing wetlands and habitat in San Joaquin Valley in California, US	\$96 to \$184 (single program)
Diamond et al. (1993)	Protection of wilderness areas in Colorado, Idaho, Montana and Wyoming, US	\$29 to \$66
Silberman et al. (1992)	Protection of beach ecosystems, New Jersey, US	\$9.26 to \$15.1
Bateman et al. (1992)	Protection of a wetland site, the Norfolk Broads, UK	£4 to £12
Boyle (1990)	Preservation of the Illinois Beach State Nature Reserve, US	\$37 to \$41
Loomis (1989)	Preservation of the Mono Lake, California, US	\$4 to \$11
Smith and Desvousges (1986)	Preservation of water quality in the Monongahela River Basin, US	\$21 to \$58 (for users) \$14 to \$53 (for non-users)
Bennett (1984)	Protection of the Nadgee Nature Reserve, Australia	\$27
Mitchell and Carson (1984)	Preservation of water quality for all rivers and lakes, US	\$242
Walsh et al. (1984)	Protection of wilderness areas in Colorado, US	\$32

Other studies link the value of biodiversity to the value of protection of natural areas with high tourism and outdoor recreational demand. In this biodiversity value category, biodiversity has been assessed by various methods, including contingent valuation, the travel cost method and market prices such as tourism revenues. Some examples are listed in Table 6. For example, the World Tourism Organization (WTO 1997) estimated that Ecuador earned \$255 million from ecotourism in 1995. A major sum accrued to a single park, the Galapagos Islands. In Rwanda, gorilla tourism in the Volcanoes National Park generated directed revenues of \$1.02 million annually until 1994, or \$68 per ha (AG Ökotourismus/BMZ 1995). Studies of less popular parks indicate lower values. The recreational value of Mantadia National Park in Madagascar was estimated to range between \$9 and \$25 per ha (Mercer et al. 1995). One particularly interesting valuation result is shown in the study by Norton and Southey (1995). This study calculates the economic value of natural habitat for biodiversity protection in Kenya by assessing the associated opportunity costs of foregone agricultural production, which is estimated to be \$203 million. This is much higher than the \$42 million of net financial revenue from wildlife tourism. Layman et al. (1996) explored the travel cost method to estimate the recreational fishing value of Chinook salmon in the Gulkana river, Alaska. The estimates of the mean consumer surplus per day range from \$17 to \$60 for actual trips, depending upon the wage rate. In a different context, Chase et al. (1998) studied ecotourism demand in Costa Rica. The value estimates result from a survey of foreign visitors to three national parks: Volcan Irazu, Volcan Poas, and Manuel Antonio. Manuel Antonio National Park registered the highest WTP, viz. \$24.90. Finally, Moons (1999) used the travel cost method to assess the economic value of recreational activities in the Meerdal-Heverlee forest in Belgium.

Table 6. Valuation of tourism and outdoor recreation

Author(s)	Study	Measurement method	Estimates
Moons (1999)	Enjoyment received in forest-related recreational activities in Flanders, Belgium	Travel cost	BEF 1,030 per trip
Chase et al. (1998)	Protection of recreation opportunities in three national parks, Costa Rica	Contingent valuation	$21.60 to $24.90 per visitor
WTO (1997)	Ecotourism in Ecuador	Tourism revenue	$255 million annually
Layman et al. (1996)	Chinook salmon in the Gulkana river, Alaska	Travel cost	$17 to $60 per trip
AG Ökotourismus (1995)	Gorilla tourism in Volcanoes National Park, Rwanda	Tourism revenue	$1.02 million annually
Mercer et al. (1995)	Recreational value of Mantadia National Park, Madagascar	Tourism revenue	$9 and $25 per ha
Norton and Southey (1995)	Biodiversity conservation in Kenya	Production function	$203 million annually

4.4. Biodiversity, Ecosystem Functions and Service Flows

The contingent valuation (CV) method has been widely used for valuing biodiversity benefits around the world, in terms of both species diversity and natural habitat protection. Nevertheless, when it comes to the monetary valuation of ecosystem functions, CV may not always be the best choice. This is because ecosystem functions, such as ecosystem life support, are not an issue that the general public is familiar with. In addition, the complexity of the relationships involved makes their accurate and comprehensive description in a survey extremely difficult. Researchers frequently end up using valuation methods based on travel costs, averting behaviour or production functions. In this context, valuation studies based on soil and wind erosion, water quality, and wetland ecosystem functions can be distinguished. These are listed in Table 7, and will concisely be discussed.

One particular category of valuation of ecosystem functions and services relates to soil erosion. Veloz et al. (1985) performed an economic analysis and valuation of soil conservation in the Dominican Republic. They estimate that, for a 25-year land use interval, the net returns from the introduction of erosion control programs are about DR$ 260 per hectare. Walker and Young (1986) estimate the damage caused by soil erosion in terms of (loss of) agricultural revenue in the Palouse region of northern Idaho and western Washington to be equal to $4 and $6 per acre, for scenarios with slow and rapid technological progress, respectively. Holmes (1988) studied the impact of water turbidity due to soil erosion on the costs incurred by the water treatment industry. Estimates show that mitigation costs ranged from $4 to $82 per million gallons of water for conventional and direct filtration systems, respectively. When applying these estimates to the American Water Works Association figures on total surface water withdrawal, the nationwide damages induced by turbidity are estimated to fall in between $35 and $661 million annually. King and Sinden (1988) have explored the use of the hedonic price method in order to capture the value of soil conservation in the farmland market of Manilla Shire, Australia. The hedonic land market price regression results show that soil condition (e.g., depth of topsoil) has an implicit marginal price of $2.28/ha. More recently, Huszar (1989) studied erosion due to wind in New Mexico. According to this study, wind erosion costs to households are due to increased cleaning, maintenance and replacement expenditures, and also to reduced consumption and production opportunities. A household cost function was estimated on the basis of 242 survey respondents. The total household costs were estimated to be $454 million per year.

In the valuation field, also various sectoral studies have been undertaken, e.g., on water quality valuation (see Ribaudo 1989a,b, Torell et al. 1990, Abdalla et al. 1992, Laughland et al. 1996, Choe et al. 1996). It is noteworthy but not surprise that many valuation studies address ecosystem's functions in agriculture and forestry (see for recent studies Gallai et al. 2009, Priess et al. 2007). It should be added that in recent years also meta-analytical methods have been used extensively (see e.g. Nijkamp et al. 2008).

Table 7. Valuation of ecosystem services.

Author(s)	Study	Measurement Method	Estimates
Choe et al. (1996)	Value of a public health program at Times Beach, Philippines	Travel cost	$1.44 to $2.04 per trip
Laughland et al. (1996)	Value of water supply in Milesburg, Pennsylvania	Averting expenditures	$14 and $36 per household
Turner et al. (1995)	Life-support value of a wetland ecosystem on a Swedish island, Baltic Sea	Replacement costs	$0.4 to $1.2 million
Barbier (1994)	Preservation of Hadejia-Jama'are wetlands, Nigeria	Production function	N 850 to N 1,280 per ha
Abdalla et al. (1992)	Groundwater ecosystem in Perkasie, Pennsylvania	Averting expenditures	$61,313 to $131,334
McClelland et al. (1992)	Protection of Groundwater Program, US	Contingent valuation	$7 to $22
Andreasson-Gren (1991)	Nitrogen purification capacity of a Swedish island in the Baltic, Gotland	Replacement costs	SEK 968 per kg
Torell et al. (1990)	Water in-storage in the High Plains aquifer	Production function	$9.5 to $1.09 per acre-foot
Tobias and Mendelsohn (1990)	Tourism and ecotourism based on non-consumptive uses of wildlife in Costa Rica	Tourism revenue	$1.2 million per ha
Ribaudo (1989a, b)	Water quality benefits in ten regions in the US	Averting expenditure	$4.4 billion
Huszar (1989)	Value of wind erosion costs to households in New Mexico	Replacement costs	$454 million per year
King and Sinden (1988)	Value of soil conservation in the farm land market of Manilla Shire, Australia	Hedonic price	$2.28 per ha
Holmes (1988)	Value of the impact of water turbidity due to soil erosion on the water treatment	Replacement costs	$35 to $661 million annually
Walker and Young (1986)	Value of soil erosion on (loss of) agricultural revenue in the Palouse region	Production method	$4 and 6$ per acre
Veloz et al. (1985)	Soil erosion control program in a watershed in the Dominican Republic	Production function	DR$ 260 per ha

5. Integrated Ecological-Economic Modelling and Analysis of Biodiversity

5.1. Background

The analysis and modelling of biodiversity are rooted in the domains of the natural and social sciences; they require the study of human economic activities, their relationships with biodiversity, and with the structure and functions of ecosystems. The combination or integration of the two approaches implies in practice often a somewhat qualitative, formal, sequentially integrated framework. Interdisciplinary work involves economists or ecologists transferring elements or even theories and models from one discipline to another and transforming them for their specific purposes (see also Carpenter et al. 2009, Polasky and Segerson 2009). The objective of the present section is to develop a common way of thinking about the modelling and valuation of biodiversity. This may require activities such as reduction, simplifying or summarizing. This section provides a survey of frameworks and methods of integrated ecological-economic modelling and the valuation of biodiversity. It ends with an illustration of a regional integrating modelling exercise.

5.2. Frameworks and Theories Underlying Integrated Modelling

Before discussing specific methods and models we will address the frameworks and conceptual perspectives underlying the integration of economics, ecology and other disciplines. The literature shows various examples of such simple frameworks. Surveys are amongst other offered by Barbier (1990), Van den Bergh and Nijkamp (1991), Van den Bergh (1996), Costanza et al. (1997), Ayres et al. (1999) and Turner et al. (2000).

A very general and almost non-theoretical ('no assumptions') framework is the Driver-Pressure-State-Impact-Response (DPSIR) framework, a variation on the framework proposed for environmental data classification by Turner et al. (2000) and Rotmans and de Vries (1997) for integrated analysis and modelling. The components can be interpreted as follows: 'Driver' = economic and social activities and processes; 'Pressure' = pressures on the human (health) and environmental system (resources and ecosystems); 'State' = the physical, chemical and biological changes in the biosphere, human population, resources and artifacts (buildings, infrastructure, machines); 'Impact' = the social, economic and ecological impacts of natural or human-induced changes in the biosphere; 'Responses' = human interventions on the level of drivers (prevention, changing behaviour), pressures (mitigation), states (relocation) or impacts (restoration, health care). According to Rotmans and de Vries (1997) integration can be of various types. Vertical integration means that the causal chain in the PSIR or DPSIR framework is completely described in one model. Horizontal integration (of subsystems) in this context is defined as the coupling of various global biogeochemical cycles and earth system compartments (atmosphere, terrestrial biosphere, hydrosphere, lithosphere and cryosphere). Full or total integration means a combination, leading to the complex linking, of various drivers, pressures, states, impacts and responses, thus allowing for various synergies and feedback. The integration frameworks proposed in environmental and ecological economics represent more specific theoretical choices than the DSPIR model. We will discuss several of these in the following subsection.

5.3. Integrated Model Assessment

A very general method of developing integrated models is the systems well-known approach (also 'systems dynamics'). This includes a wide range of model types: linear versus nonlinear, continuous versus discrete, deterministic versus stochastic, and optimizing versus descriptive. The systems approach allows us to deal with concepts like dynamic processes, feedback mechanisms, and control strategies (see Bennett and Chorley 1978, Costanza et al. 1993). One can integrate two subsystems, or have a hierarchy or nesting of systems. The fixed elements in the system can either be considered black boxes or be described as empirical or logical processes themselves. The systems approach is suitable for integrating existing models, and can incorporate temporal as well as spatial processes.

Costanza et al. (1993) distinguish between economic, ecological and integrated approaches on the basis of whether they optimize: (1) generality, characterized by simple theoretical or conceptual models that aggregate, caricature and exaggerate; (2) precision, characterized by statistical, short-term, partial, static or linear models with one element examined in much detail; and (3) realism, characterized by causal, nonlinear, dynamic-evolutionary, and complex models.

These three criteria are usually conflicting, so a trade-off between them is inevitable. A distinction between analytical and heuristic integration is relevant here. Analytical integration means combining all aspects studied in a single model (and therefore model type). Heuristic integration can proceed by using the output of one model as input to another, and vice versa, as well as by extending this through (finite) iterative interaction. In this case different model types, such as optimization models and descriptive models, can be combined. If one desires to attain a great deal of analytical power, analytical integration seems attractive, whereas striving for realism would imply the use of a heuristically linked set of models from different disciplines. Striving for empirically sound models often implies modest approaches to improving precision, which usually goes at the cost of model use in a wider context or with a wider range of parameter values. The development of integrated models, through the joint efforts of economists and ecologists, is based on bringing together elements, theories or models from each discipline and transforming these for the purpose of integration. This may require operations such as reduction, simplification or summarizing. The results may not always be greeted with enthusiasm within the disciplines, especially when they neglect certain nuances or different viewpoints.

Many integrated models defined at the level of ecosystems are based on the standard systems-ecological approach (Patten 1971, Jørgenson 1992). They include ecosystem modules that describe the effects of environmental pollution, resource use and other types of disturbance. A main problem is modelling the effects of multiple stress factors, since the empirical basis for this is often lacking. Various integrated models have been developed for terrestrial and aquatic systems. Surveys are presented in Braat and Van Lierop (1987), Van den Bergh (1996) and Costanza et al. (1997). Some studies have paid much attention to spatial aspects, focusing on spatial disaggregation into zones (for instance, Giaoutzi and Nijkamp 1995, Van den Bergh and Nijkamp 1994) or on land-use planning in interaction with landscape ecology (see Bockstael et al. 1995). Formal theoretical approaches in ecology that provide a basis for these approaches have been described by Watt (1968), Maynard-Smith (1974), Roughgarden et al. (1989) and Jørgenson (1992). Perrings and Walker (1997) consider resilience in a simple integrated model of fire occurrences in semi-arid rangelands such as those found in Australia. The model describes the interaction between extreme events (fire, flood, and droughts), grazing pressure, and multiple locally stable states. Carpenter et al. (1999) develop and explore water and land-use options in an integrated model of a prototypical region with a lake that is being polluted. This model combines rationally bounded behaviour, supposedly in accordance with the reality of regional resource and environmental management, and a nonlinear ecosystem module describing processes occurring at different speeds. The model generates multiple locally stable states as well as ‘flipping’ behaviour (see also Janssen and Munda 1999). Swallow (1994) integrates theoretical models of renewable and non-renewable resources to address multiple uses and tradeoffs in wetland systems. A special category of integrated modelling is sometimes referred to as the biophysical or energy approach. This aims to integrate economic and environmental ecological processes in energy-physical dimensions, based on the notion that any system is constrained by energy availability (Odum 1987). These models include energy and mass balances. A central concept in this approach is ‘embodied energy’, which is defined as the direct and indirect energy required producing organized material structures. Applications of these energy-inspired models cover ecosystems, economic systems, and environment-economy models (Odum 1987). An extended application to a regional system is presented by Jansson and Zuchetto (1978) (see also Zuchetto and Jansson 1985).

The recent focus on integrated assessment of the enhanced greenhouse effect (a potential climate change) can be regarded as the new wave in ‘world models’, where (again) economists and others have tended to rely on different model approaches (Bruce et al. 1995). The integrated climate assessment models combine results from the natural sciences (physics, chemistry, biology, earth sciences) and the social sciences (economics, sociology, political science), and have so far given rise to a continuation of the trend in world models towards increasing detail and disaggregation. These climate assessment models have a multilayered conceptual structure that distinguishes physical and environmental effects of human activities from adjustments to climate change by humans (individuals, firms, organizations) and policy responses (mitigation, aimed at the causes) at various spatial levels (Parry and Carter 1998).

5.4. Specific Methods and Models

Integrated models can have different formats. Table 8 illustrates some characteristics of integrated models and provides general examples. One important distinction is between policy optimization and evaluation (usually numerical simulation) models. One of the first and famous integrated assessment models used in policymaking is the RAINS model (Alcamo et al. 1990). This includes an optimization algorithm for calculating cost-effective acidification strategies in Europe, aimed at realizing deposition targets throughout Europe, and taking account of sensitive natural areas (forests and lakes). This model is a rare case of direct science-policy influence, as it was used in the negotiations on transboundary air pollution in Europe. Castells (1999) offers an informative analysis of the institutional and evolutionary dimensions of the interaction between scientists, research institutions and negotiations on international environmental agreements, with special attention given to the RAINS model and the acid rain context in Europe. In the area of integrated assessment models for CO_2 emissions (climate) strategies, one can find both economic optimization (Nordhaus 1994) and detailed descriptive model systems like IMAGE and TARGETS (Alcamo 1994; and Rotmans and de Vries 1997). DICE by Nordhaus (1991) is the first example of a policy optimization model for climate change. The model essentially combines economic growth theory with a simplified climate change model. Tol (1998) provides a short account of the evolution of the economic optimization approach to climate change research. He emphasizes the attention placed upon the analysis of uncertainty and learning from a cost-effectiveness perspective, which has given rise to various model formulations and analyses. More recently, Janssen (1998) and Van Ierland (1999) present informative surveys and categorizations of macroeconomic-cum-environment and macro-level integrated models, including the climate-oriented integrated assessment models. Van Ierland devotes special attention to the various ‘regionalized world’ models (with acronyms like RICE, CETA, MERGE, DIALOGUE, FUND). Van den Bergh and Hofkes (1998) have collected distinct approaches to integrated modelling with an economic emphasis that focus on sustainable development questions in theory and in practice, as well as at global and regional levels.

Table 8. Characterizing integrated models

Model criterion	Range of choice	Examples of distinct approaches
Analytical integration	Optimization (benevolent decision maker); Equilibrium (partial or general); Game-theoretical; Dynamic-mechanistic; Adaptive (multi-agent & dynamic); Evolutionary (irreversible, bounded rationality)	Many theoretical models: growth theory, renewable resource economics (fisheries, forestry, water quality/quantity), systems models (Limits-to-growth), cost-effectiveness models (RAINS), welfare optimization (DICE)
Heuristic integration	Satellite principle; Multilayer subsystems; Sequential; Parallel consistent scenarios; Aggregation of indicators; Evaluation	Regional environmental quality models (Resources for the Future), world models (Club of Rome), integrated assessment model
Spatial coverage	World; National; Regional; Urban; Local Ecosystem	Ecosystem modelling, macroeconomic modelling, regional modelling, urban modelling, world models
Spatial disaggregation	Single region; Multiregion; Spatial grid (GIS)	Integrated assessment models (climate change), land use models
Aggregation level	Micro (individuals, households); Macro (national economy, main sectors, global); Sectorial; Interest groups; Homogeneous land plots; Spatial grids; Temporal (days, seasons, years)	Computable general equilibrium models, macroeconomic models (Keynesian), multisector models, land use models, landscape models

Source: Van den Bergh (1996)

5.5. Interaction between Integrated Modelling and Monetary Valuation

Progress in improving models to provide economic information, particularly predictive information, will require a vital and dynamic interdisciplinary dialogue. At this level, integrating modelling and monetary valuation can present important advantages for guiding policy by presenting important interactions. First, values estimated in a valuation study can be used as parameter values in model studies. Benefit or value transfer (e.g., meta-analysis exercises) can be used to translate value estimates into other contexts, conditions, locations or temporal settings that do not allow for direct valuation in 'primary studies' (due to technical or financial constraints). Second, models can be used to generate values under particular scenarios. In particular, dynamic models can be used to generate a flow of benefits over time and to compute the net present value, which can serve as a value relating to a particular scenario of ecosystem change or management. Third, models can be used to generate detailed scenarios that enter valuation experiments. An input scenario can describe general environmental change, regional development or ecosystem management. This can be fed into a model calculation, which in turn can provide an output scenario with more detailed spatial or temporal information. The latter can then serve, for example, as a hypothetical scenario for valuation, which is presented to respondents in a certain format (graphs, tables, story, diagrams, pictures) so as to inform them about potential consequences of the general policy or exogenous change. User-friendly computer software can be used in such a process. Finally, the outputs of model and valuation studies can be compared. For instance, when studying a scenario for wetland transformation, one can model consequences in multiple dimensions (physical, ecological and costs/benefits), and aggregate these via a multi-criteria evaluation procedure, with weights being set by a decision-maker or a representative panel of stakeholders. Alternatively, one can ask respondents to provide value estimates, such as willingness to pay for not experiencing the change. If such information is available for multiple management scenarios, then rankings based on different approaches can be compared.

5.6. Advantages and Disadvantages of Integrated Modelling

Using integrated economic-environmental models for the analysis and evaluation of biodiversity issues has both advantages and disadvantages. Three main advantages are: (1) handling data, information, theories, and empirical findings from various contributing disciplines in a systematic and consistent way; (2) being explicit about assumptions, theories and facts; and (3) addressing complex phenomena, interactions, feedback, laborious calculations and temporally, spatially and sectorally detailed and disaggregate processes. An argument against non-formal approaches to integrated research is that these fail to provide for a systematic and consistent linking of data, theories and empirical insights from various disciplines. Instead, these approaches tend to result in a battle of perspectives based on distinct and usually implicit premises and information bases. Models force researchers to be explicit about at least the latter two inputs to integrated research. Most of the disadvantages of integrated modelling apply to non-model-based integrated research as well. They include: an unclear synergy of approximations and uncertainties; rough application of monodisciplinary theories and empirical insights; simplification of complex phenomena (e.g., by treating them as a black box); misinterpretation or arbitrary choice of disciplinary perspectives by the model, and a lack of systematic or complete linking of subsystems or submodels. Complex or high-dimensional models have the extra disadvantage of being difficult to calibrate and validate, and of lacking transparency.

The main disadvantage of models perhaps is that they are trusted too much, so that they run the risk of being interpreted as objective representations of reality, and are then taken too seriously, especially by laypersons and policymakers. On the other hand, policymakers often express their doubts about formal modelling. Shackley (1997) states that numerical models have, despite their long tradition of development and widespread use, not achieved the epistemological status; the controlled laboratory experiment has in natural sciences (and more recently in the social sciences and in environmental economics in particular). This relates to the fact that modelling results never 'prove' anything, since they do not generate real or physical processes. The best way to view theoretical and especially empirical models is to consider them tools for hypothetical experiments with complex systems, which serve as analogies or pictures of real-world systems that do not allow – technically, morally or politically – for experimentation. In other words, complex model systems, notably integrated economic-ecological models, are heuristic devices for learning about the real-world system, rather than for predicting its real course of behaviour. In addition, integrated modelling is restricted by the model type.

If economic and ecological models fit within a (general) systems framework, they may be blended into a single model structure, where compartments or modules may represent the original models, and certain outputs of one module serve as input for another. Nevertheless, it is often not easy to link models directly. For instance, if both the economic and ecological systems are represented in the form of programming or optimization models, several options are available: look for a new, aggregate objective; adopt a multiobjective or conflict analysis framework; or, when possible, derive multiple sets of optimality conditions and solve these simultaneously. Alternatively, when the economic and ecological systems are represented by different model types, it is difficult to suggest how they could be linked to one another. When economic models have an optimization format and ecosystem models have a descriptive format, direct technical integration seems feasible; otherwise heuristic approaches are needed.

6. Biodiversity Policy

6.1. Public Policy Strategies

As argued above, biodiversity comprises functions that affect the wellbeing of individuals and societies in all regions of our world. The mainly public good character of biodiversity, combined with the presence of many externalities, evidently gives rise to a market failure. In particular, market prices fail to capture the full range of biodiversity benefits to individuals and society. This contributes to the rapid depletion of biodiversity, leading to important welfare losses. Therefore, there is a clear scope for public biodiversity policy. A successful public policy design aimed at ensuring the conservation and sustainable use of the full range of biodiversity benefits implies the use of environmental instruments that: (1) protect biodiversity private values, such as benefits in terms of species and genetic diversity, through the provision of proper market incentives, such as taxes and charges or the assignment of well-defined property rights; and (2) protect biodiversity public values, such as benefits in terms of the knowledge of the continued existence of ecosystems diversity and bequest values related to maintaining them for the enjoyment of future generations, through the use of institutions and the creation of market mechanisms such as the provision of information. Therefore, biodiversity policy-related measures are often applied in a policy mix, combining standard-setting, direct and indirect market intervention and the provision of information. Table 9 presents a concise list of the strategies available to governments involved in biodiversity policy design. These are discussed in more detail in the subsequent subsections.

Table 9. Strategies for government involvement in public biodiversity policy

Direct government interventions:

1. Price incentives: fees, charges, taxes and tradable permits.
2. Command-and-control strategy: quantity standards, technology regulation, access restrictions.
3. Assignment of property rights.

Provision of information:

4. Development of market mechanisms: certification, ecolabelling, and institutional building.

Source: OECD (1999)

6.2. Direct Government Intervention

One possible way of addressing market failures is through direct government intervention. This involves the use of policy instruments such as taxes, command-control policy, and the definition of property rights. The best-known price instrument is the optimal or Pigouvian tax, which restores a situation with biodiversity externalities to a social optimum. For example, in 1995 the Dutch government introduced a groundwater tax so as to minimize the desiccation effects associated with the excessive use of groundwater reserves, one of the most important causes of biodiversity loss in the country (Bellegem et al. 1999). However, the implementation of Pigouvian taxes for policy design is very difficult in view of the large degree of uncertainty associated with determining the social costs of biodiversity loss and the high financial expenses of the activities involved in translating such social costs into monetary terms. As a result, it is difficult to find other situations where policy instruments make the use of Pigouvian taxes to internalize the full non-market benefits of biodiversity.

Alternatively, the government can impose strict command-control policies. This means that the government directly dictates clear quantity targets, i.e., quantity standards, which have to be followed by producers and consumers. The government may have to set up a regulatory body, which monitors whether the restriction is being complied with by the firms and which enforces it by punishing violators. An example is to set a limit on the number of daily visitors to nature areas, such as sensitive wilderness areas, or on the number of animal species that can be caught by hunters or fishers. Adopting such quantity standards is especially attractive from the perspective of policy effectiveness. However, command-control policy generally implies embracing high monitoring and enforcement costs. Moreover, uniform control does not sufficiently address the heterogeneity of agents, and thus misses out on potential efficiency gains.

Third, the government can opt for the provision and enforcement of well-defined property rights. This instrument is particularly efficient in addressing the market price internalization of the private values of biological resources. An example of this type of public policy is the assignment of property rights to ship ballast waters (see Van den Bergh et al. 2002). However, the public value of biological resources, such as existence and moral values cannot be internalized in the market price through the provision and enforcement of property rights, thus hindering the effectiveness of this strategy.

Independent of the policy instrument used, direct government intervention often involves administrative costs, for instance, the government may have to establish a monitoring and enforcement agency, thus hampering the effectiveness of this strategy. In addition, the strategy may also be ineffective in the presence of important information asymmetries. This is because information plays a crucial role in the design of an effective Pigouvian tax, particularly when firms have an incentive to conceal true information. In the literature we distinguish two types of informational problems, i.e., hidden information and hidden action problems. Hidden action refers to a post-contractual problem in which one party knows more about his or her type, or effort, after the contract is signed. This is also known as an adverse selection problem. Hidden information refers to a pre-contractual problem, in which one party knows more about his or her true type than the other party before the contract is signed. This is also known as a moral hazard problem – see Akerlof (1970) for a detailed analysis.

Finally, public policies based on direct government intervention may create bureaucratic inefficiency. Bureaucrats may pursue rents and are prone to influences from lobbying activities by market participants. In fact, in the absence of rent-seeking behaviour, such direct government involvement may instead create a disincentive for market participants to innovate or to employ the most efficient method of production.

6.3. Information Provision

Information provision is an integral part of a public policy directed to the use of market forces without direct government involvement in supply and demand forces. In such a context, policy instruments based on market creation mechanisms have proven to be a valid alternative to direct market intervention policies. The provision of information works within one of two basic conceptual frameworks: certification or ecolabelling. Biodiversity ecolabelling refers to the act of providing information to the consumer that a product, or a product's attributes, possesses specific characteristics regarding the product's origins or ecological, social and economic specifications. Biodiversity certification refers to the act of provision of information with respect to alternative management systems, based on the ability to create a product in an environmentally sound and sustainable manner. Assessing the integrity of a product, or a product's attributes, and the underlying management system involves an evaluation of management practices with respect to defined standards, generally fixed at the management unit level.

In both cases, a credible scheme must evaluate the integrity of the producer's claim and the authenticity of product origin. The assessment of the authenticity of the product's origin involves the identification and monitoring of the supply chain, including raw materials transport and processing, secondary manufacturing and, finally, retail distribution. Therefore, the success of certification and ecolabelling may prove to be difficult to achieve. This strategy often goes hand-in-hand with direct government intervention-oriented public policies, giving rise to 'mixed policies'. The goal of this policy is to circumvent the weaknesses and inefficiencies that may occur when adopting either the command-and-control policy or the market mechanism approach and therefore to achieve higher policy effectiveness. In the next section we explain the use of the certification and ecolabelling policies for conservation and sustainable use of biodiversity and how they can be combined with other instruments to improve their effectiveness.

6.4. Biodiversity Certification and Ecolabelling

Biodiversity certification and ecolabelling refers to an act of provision of information to the consumer about a product, or product characteristics, creating a separate market segment for the product. The participation of consumers in markets for these differentiated products usually permits the market price internalization of some biodiversity benefits. As a matter of fact, consumers are willing to pay a price premium for these benefits. According to the US Environmental Protection Agency, several market surveys indicate that a majority of consumers consider themselves to be environmentalists and would prefer to buy products with a reduced environmental impact when the quality and costs are comparable (EPA 1993). The question is then: can consumers, who purchase biodiversity-friendly products, internalize the full range of benefits of biodiversity? If the answer to this question is yes, then biodiversity certification and ecolabelling can effectively create a segment of the market that is a market for biodiversity friendly products and services. An organic vegetable product, i.e., a vegetable product that is planted without chemical fertilizers, could be an example of such a market segment (Van Ravenswaay 1995, 1996). The underlying steering engine for the creation of such a market relies on the fact that consumers believe that there is a difference in taste between organic and non-organic vegetables. It is often argued that organic vegetables taste better than non-organic ones. In addition, consumers believe that organic products are healthier than non-organic ones and, most of the time, they are able to distinguish between the two products by their appearance. In this setting, the role of certification and ecolabelling is to inform and provide assurance to consumers. Hence, consumers are able to internalize the benefits of consuming the good. Therefore, ecolabelling works as an instrument for resolving the standard hidden information problem.

Most of the time, however, the benefits from biodiversity certification and ecolabelling largely accrue to society at large, and are not explicitly internalized by the individual consumer who purchases the good. In this setting, where benefits from biodiversity certification and ecolabelling are characterized by a public good nature, it is harder to achieve an effective biodiversity certification policy. There are three important factors that determine the success of this type of policy. These are consumer awareness, firm incentives to undertake certification and ecolabelling, and the sensitivity of consumer demand to production costs. These will concisely be discussed.

Consumer awareness

Consumers' awareness is a necessary condition for the creation of an effective policy certification of public biodiversity benefits. In the extreme scenario of 'no consumer awareness' about environmental and biodiversity protection benefits, which are indirect for the consumption and use of the goods and services, there will be no willingness to pay, or price premium, for such biodiversity benefits. Once there is sufficient consumer awareness about the need for a biodiversity-friendly environment, there will be a significant willingness to pay a price premium for ecolabeled or certified products. However, consumer awareness may take many years to develop (see Van Ravenswaay and Blend 1997). Hence, policymakers may want to launch extensive information campaigns, targeting the general public, as well as initiate formal education programs about the benefits of having a clean and biodiversity-protected environment.

Incentives of firms to undertake certification and ecolabelling

If firm costs are not sensitive to the costs of undertaking certification and ecolabelling regimes, then producers may have sufficient incentive to incorporate such policies. However, in most cases the adoption of certification and ecolabelling policies would increase firms' production costs because producers may have to install new production technologies or may have to utilize certain inputs in order to satisfy the environmental standards that are stipulated by the product label – see Van Ravenswaay and Blend (1997) for more details. Therefore, adopting certification implies incurring higher production costs. These, in turn, damage the firm's market competitiveness, eventually leading to reductions in the firm's profits. Therefore, hardly any producer would like to adhere to certification and ecolabelling regimes. In other words, there are simply not enough market mechanism incentives to make the adoption of certification and ecolabelling policies successful. In such a setting, policymakers may need to complement (or combine) certification policy instruments with other policies aimed at providing enough incentives for producers to adopt certification and ecolabelling. In other words, policymakers may need to launch mixed public policies. For example, it may be effective to combine biodiversity certification and ecolabelling with input subsidies, technical assistance provision, and R&D subsidy regimes.

It is worthy of note that in situations where firms' production costs do not change with the adoption of a certification or ecolabelling regime, it does not mean that a certification or ecolabelling policy is always advisable. In this setting, two markets co-exist, i.e., the market for the conventional product and the market for the certified product. Mattoo and Singh (1994) show that market complementarity between certified and non-certified products can stimulate investment in the technology of the non-certified, or conventional, products – a so-called 'benefit spillover' to the non-certified products. This may lead, in turn, to an increase in the output of the conventional product, in contrast with the original goal of the certification and ecolabelling policy, i.e., gradually increasing the market share of the environmentally friendly product. To avoid such a situation, policymakers are advised to implement certification and ecolabelling schemes together with other public policies, such as the introduction of environmental quality standards and only awarding certificates and ecolabels to those who meet the standards – see Dosi and Moretto (1998) for additional details.

Sensitivity of consumer demand for biodiversity price premiums

If consumers are not willing to pay a premium for certified and ecolabelled products while, at the same time, the introduction of such products boosts firms' production costs, then producers' profits will inevitably decrease. Without any further developments, the certification and ecolabelling regimes would be predestined to fail. In this context, the success of the biodiversity certification and ecolabelling requires that it be combined with other policy strategies. Once again policymakers may want to launch a mixed policy. For instance, policymakers can introduce a certification regime followed by a strong environmental information campaign, stressing the benefits of biodiversity certified products. NGOs also have an important role in building up consumer awareness and disseminating information, for instance through the design and content of ecolabels and certificates. It should be clear to consumers what benefits they can obtain from buying ecolabelled products.

In any situation where consumers are aware of biodiversity benefits, are willing to pay a price premium to consume and use them and the firms' production costs are not too sensitive to the adoption of certification and ecolabelling schemes, policymakers can rely on certification and ecolabelling as effective instruments for protecting biodiversity products, without having the need to combine these instruments with other public policies.

Evaluation of biodiversity certification and ecolabelling policy instruments

It can be concluded that the success of certification and ecolabelling as policy instruments for the creation of markets for biodiversity benefits, which is a crucial tool for protecting biodiversity products and service flows, depends on several crucial factors. These include the ability of the proposed policy instrument to internalize a wide range of the biodiversity benefits, which ultimately depend on the public good nature of the biodiversity benefit under consideration. In addition, three other factors determine the success of biodiversity certification and ecolabelling policy instruments. These are consumer awareness, firm incentives to undertake certification and ecolabelling, and the sensitivity of consumer demand to production costs. From this discussion, it emerges that certification and ecolabelling policy instruments alone are not sufficient to guarantee the successful protection of biodiversity products and services flows. The Dutch energy market, which includes green-energy certification and direct government intervention in the market forces, has shown the crucial significance of combining public policies in order to create an effective means of protecting biodiversity products.

Finally, the certification schemes need to be sufficiently flexible to allow for mutual recognition among the agents involved, to meet the demands of weak and sensitive markets, and to avoid encouraging unfair international trade. To achieve this, it is important to explore each country's unique environmental and cultural characteristics. Through mutual understanding and learning from the past, certification and ecolabelling can positively contribute to the creation of markets for biodiversity and thus are expected to assist in the development of effective and broadly accepted sustainable management policies for scarce natural resources.

7. Conclusions

How can we use the ideas presented here to formulate an integrated, effective framework to assess the value of biodiversity? And what can we learn from the large number of available studies? The answers to these questions require, inter alia, that a clear life diversity level be chosen, that a concrete biodiversity change scenario be formulated, that biodiversity changes – notably losses – be within certain pre-specified boundaries, and that the particular perspective on biodiversity value be made explicit.

So far, most studies lack a uniform and clear perspective on biodiversity as a distinct, univocal concept. In addition, at present we have insufficient knowledge about, for example, how many species there are; for this reason alone, it is very difficult, if not impossible, to assess the total economic value of biodiversity. To completely answer the question, 'What is the value of biodiversity?', we would have to include the value of genetic variation within species across populations, the value of the variety of interrelationships in which species exist in different ecosystems, and the functions among ecosystems. Without any doubt, full monetary assessment is impossible or would be subject to much debate. An additional problem is that, at the global level, biodiversity values can differ significantly, even for similar entities, due to unequal international income distribution. All in all, the available economic valuation estimates should be considered, at best, as a lower bound to an unknown value of biodiversity, and are always contingent upon the available scientific information as well as their global socio-economic context.

As we have seen, biodiversity can be dealt with at different levels: genetic, species, ecosystem, and functional diversity. For the analysis and valuation of biodiversity at the ecosystem and functional levels, which may be regarded as the cornerstone of the analysis and valuation of biodiversity, an active interdisciplinary dialogue is necessary, with emphasis on the complex interface between natural science and social science approaches. A comprehensive assessment of ecosystem biodiversity characteristics, structure, and functioning requires the analyst to take various important steps. First, the socio-economic causes and consequences of biodiversity degradation or loss should be determined. Second, the negative impacts on biodiversity caused by human activities should be assessed. The range and degree of biodiversity functioning should be estimated, especially in terms of ecosystem-functional relationships. Finally, alternative biodiversity management strategies should be ranked and a joint spatial and temporal systems analysis of each policy scenario should be carried out.

The physical assessment of the functions performed by biodiversity is an essential prerequisite of any ecological evaluation. However, simply identifying these functions is insufficient if we want to present resource managers and policymakers with relevant policy response options. It is necessary to develop criteria for the expression of the functions in a form that allows for evaluation. For example, one can identify the range of biodiversity management strategies by exploring the use of methods such as Red Data Species Lists and biological value indexes. Recently, computer models have become available to aid making decisions about species conservation. Models have been applied to calculate minimum dynamic areas that support the minimum viable population of a certain species. In addition, computer models have been used for habitat evaluation, predicting ecological conservation values under different development scenarios. This approach to ecological evaluation allows for a direct comparison of management or conservation strategies.

From an economic perspective, certain aspects of biodiversity are scarce and highly desirable, which is the reason why they have economic value. The concept of economic value is founded in welfare economics, which developed around the theory of consumer behaviour. Economic valuation assumes interaction between a subject – a human being – and an object – for example, an element of biodiversity. As a result, economic value is distinct from the notion of intrinsic value, which assumes that an object has or can have value in the absence of any (human) subject. It is important to recognize that economists do not pursue absolute value assessment of environmental systems or all the biodiversity they contain, but always focus attention on valuing environmental system changes. This means that the terms ‘economic value’ and ‘welfare change’ are two sides of the same coin. Economics can thus assess the human welfare significance of biodiversity changes, namely through the determination of changes in the provision of biodiversity-related goods and services and their consequent impacts on the wellbeing of humans who derive – use or non-use – benefits from their provision. Further note that, although many economic studies employing monetary valuation claim to have assessed biodiversity values, they often confuse biodiversity with biological resources.

Different instruments are available for assessing the monetary value of biodiversity. Stated preference methods have often been used, because the use of revealed preference methods would leave out important biodiversity value types, notably non-use and quasi-option values. This can lead to a significant value measurement bias, especially since biodiversity conservation is associated with many non-use and indirect use (or primary ecological) values. Alternatively, researchers can combine distinct valuation techniques. Special attention however, should be given to the aggregation of resulting values so as to avoid double counting. From our review of economic valuation studies, it is clear that the assessment of biodiversity values does not lead to an unambiguous monetary value of biodiversity. Instead, available economic valuation estimates should generally be regarded as providing a very incomplete perspective on, and at best lower bounds to, the unknown value of biodiversity changes.

Integrated economic-ecological modelling can contribute to, and may even be essential for, a thorough understanding of the intricate relationship between biodiversity and ecosystem and economic dynamics. Although integrated modelling has somewhat of a tradition, both at the ecosystem level and at the global level, applications to biodiversity-related problems are scarce.

Integrated modelling can be linked to biodiversity valuation and evaluation in various ways. Integrated models can generate a set of biological and economic, possibly monetary, indicators that can be further aggregated through multicriteria analysis techniques. In addition, it is possible to provide for a closer, innovative connection between modelling and valuation, among other methods, by: generating conditional values for specific environment-economic scenarios; using scenario-modelling outcomes such as tables and graphs in valuation experiments (e.g., contingent valuation); and using spatial models to aggregate monetary values related to specific areas.

Biodiversity public policy entails the use of direct market intervention, including taxes and command and control instruments, the provision of information, such as certification and ecolabelling, or the combination of both in some sort of policy mix. The success of certification in internalizing biodiversity benefits in the market prices of goods and services – which means that it constitutes an effective instrument in protecting biodiversity – depends on three factors: the public good nature of non-market biodiversity benefits; the application of economic valuation methods to assess the monetary magnitude of non-market biodiversity benefits; and, supply and demand factors, which include the level of consumer awareness and sensitivity to certain products and the producer's propensity to embrace certification schemes. In some cases certification and ecolabelling policy instruments alone are insufficient to guarantee the successful internalization of biodiversity benefits and thus to contribute to a better allocation of biodiversity. Indeed, mixed policy strategies involving both certification and direct government intervention in supply or demand may sometimes be needed.

Finally, one needs to be aware of the limitations of biodiversity valuation and analysis. Biodiversity is a complex and abstract concept. It can be associated with a wide range of benefits to human society, most of them still poorly understood. In general terms, the value of biodiversity can be assessed in terms of its impact on the provision of inputs to production processes, on human welfare, and on the regulation of ecological functions. A complete understanding of these and their integration into multidisciplinary studies provides a great challenge for future research, in which economists, ecologists and others will have to work closely together. Only then can useful policy insights be expected. There is no doubt that the ecological economics of biodiversity will face many research and policy challenges in the years ahead of us.

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