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Ecological–Economic Analysis and Valuation of Biodiversity

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1. Introduction: context and scope of the study

In the spirit of the growing environmental awareness in the past decades, we witness nowadays an increasing interest in biodiversity, both locally and world-wide. Biodiversity requires our attention for two reasons. First, it provides a wide range of benefits to mankind and human activities. Second, many human activities have contributed 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. The valuation of biodiversity can be approached from an ecological, economic, or combined perspective. This contribution presents an overview of economic and ecological indicators of biodiversity, identifies the underlying valuation approaches, discusses basic concepts and theories, and reviews the respective applications.

The organisation of this study is as follows. Section 2 starts with a concise review of the most important frameworks that focus on the conceptualisation of biodiversity and its role in ecological processes. It continues then with the identification and discussion of alternative ecological valuation approaches of biodiversity, including biotic-richness and ecosystem health approaches. Section 3 addresses the economic concepts and methods relevant to the valuation of biodiversity. It starts with a discussion of the reasons why economists are interested in pursuing monetary value assessment of biodiversity and why some scientists disagree with that. A survey of empirical valuation studies is presented. The distinction between 'resource valuation' and 'biodiversity valuation' is discussed. Finally, respective valuation results and measurement methods are discussed. Section 4 identifies a combined economic-ecological valuation perspective. According to this perspective, biodiversity is analysed, modelled and valued from an integrated point of view, combining natural and social-economic aspects, and thus can enable the construction of mutually consistent indicators of biodiversity. Finally, Section 5 presents conclusions and outlines some recommendations for use of valuation methods and findings in environmental policy.

2. Ecological foundations for biodiversity analysis and valuation

2.1 Conceptualisation of biodiversity and its role in ecological processes

The analysis of biodiversity is rooted in the domain of both natural and social sciences and its modelling implies a review of knowledge on the relationships between biodiversity, the dynamics of ecosystems, and the level of human economic activities. One reason why biodiversity modelling has been so difficult relates to the complex and partly unobservable of the biodiversity-ecosystem relationships involved, such as biotic-abiotic interactions, food webs nutrient flows, and species interrelations. Independently of the complex nature of the biodiversity-ecosystem relationships, an important aspect is the recognition that the variability of the biological resources influences the functioning and structure of ecosystems. In the literature one can find three main approaches to modelling. These are reviewed in the present section.

Holling (1987 and 1992) proposed a model to describe and explain the dynamics of a terrestrial ecosystem in terms of a structure that is characterised by a sequential interaction between four basic functions or phases - the "4-box model". The functions are: (1) exploitation; (2) conservation; (3) release; and (4) reorganisation. Within this model, ecosystems develop from the exploitation phase during which systems capture easily accessible resources, to the conservation phase during which systems build and store increasingly complex structure, and then evolve to the release phase during which systems free some of the mature structures. The released structure is then available for reorganisation and uptake in the phase of the exploitation phase. The exploitation function refers to the ecosystem's processes that are responsible for "colonising disturbed sites". The conservation function refers to the ecosystem's processes that are responsible for "resource accumulation that builds and stores energy and material". The release or "creative destruction" function refers to an abrupt change in the ecosystem caused by external disturbance, releasing energy and material that have been accumulated during the conservation phase. Examples of the release phase are fire, storms, and pests (Costanza *et al.* 1995). Finally, the reorganisation function refers to the ecosystem's processes that are responsible for mobilising released energy and materials and making them available for the next exploitative phase.

Based on a categorisation of the ecosystem's functions initially made by Odum (1971), de Groot (1994) describes the relationship between biodiversity and ecosystem. In general terms, de Groot characterises the ecosystem structure in terms of four categories of biodiversity functions: (1) life support functions, (2) carrier functions, (3) production functions, and (4) information functions. Biodiversity is seen to have a life support function, i.e., a regulation of essential ecological processes. The life support functions refer to the group of the biodiversity service flows that contribute to the maintenance of a healthy environment, by providing clean air, water and soil, by providing flood control, and by providing carbon storage and waste absorption. Most of the life support functions are often fuzzy (e.g., provision of carbon storage), and thus not easily to be demarcated and identified. The carrier functions refer to the provision of space for human activities such as habitation, agriculture, and recreational activities. The production functions refer to the provision of environmental resources, ranging from raw materials for industrial use to water and energy resources. The information functions refer to the maintenance of mental health, providing opportunities for reflection, spiritual enrichment and aesthetic experience.

More recently, Norberg (1999) proposed an alternative approach to classify ecosystem functions and services of Nature. Norberg selected groups of ecosystem services to which common ecological concepts apply: (1) are the goods and the services internal to the ecosystem or shared with other systems?, or, (2) are the goods and the services of biotic or abiotic origin?, and (3) at which level of ecological hierarchy are goods and services maintained? Bearing in mind such selection criteria, ecosystem functions and services of Nature are classified in three categories: (1) maintenance of the populations; (2) regulation of material and energy flows; and (3) organisation of biological units through selective processes. These categories represent three major fields in ecology that have well-established theoretical foundations and refer to population/community ecology research, ecosystem research, and biological organisation, respectively (Levin *et al.* 1997, Levin 1998).

The first category corresponds to the group of ecosystem services that are "(…) associated with certain species or a group of similar species (…)" (Norberg, page 185). Examples of such services include valuable foods and goods as fish, timber, pharmaceuticals chemicals, and flowers. The second category consists of processes that regulate exogenous chemical or physical cycles, i.e. processes that drive material and energy flows in ecosystems. The biota takes a significant part in most global cycles of chemical compounds such as water, CO_2, and nitrogen. Finally, the third category of ecosystem services is related to the organisation of biotic entities. Organisation is virtually present at all scales: organisation of genes through natural selection, spatial distribution of a population through dispersal and competitive exclusion, or the development of food webs and ecosystems through invasion and extinction processes.

2.2 Ecological indicators of biodiversity: (1) biotic-richness approach

From a biotic approach, the definition of biodiversity 'value' indicates the range magnitude of biological products and services flows provided by nature. Traditionally, the measurement of biological diversity has been undertaken with the use of genetic, species, and ecosystem richness or variety indices.

Measurement of genetic diversity

The analysis, conceptualisation and measurement of genetic differences can be done in terms of (1) allelic frequencies, (2) phenotypic traits and (3) DNA sequences. The same gene can exist in different frequencies or variants. These variants are called alleles. Thus, allelic diversity measures the different gene composition variants of individuals. In general, the more alleles, the more diverse their frequencies, the greater the genetic diversity. Average expected heterozygosity, the probability that two alleles sampled at random are different, is commonly used as an overall measure. A number of different indices can be applied to the measurements to assess average expected heterozygosity (Antonovic 1990).

Phenetic diversity is a measure based on individual's phenotype traits, checking whether individuals share the same characteristics. This valuation method is focused on the measurement of the variance of certain traits and, in general, involves readily measurable morphological and physiological characteristics of the individual. However, individual genetic information is often difficult to assess, and comparisons are difficult, when the individuals or populations are measured in terms of different qualitative traits. To cope with that, scientists may now use DNA sequence variation so as to measure the genetic variety. The DNA sequence information is obtained by the use of the polymerase chain reaction. For this reason, only a small amount of material, ultimately one single cell, is required to obtain the DNA sequence data. Closely related species may share up to 95% of their DNA sequences, thus implying little diversity in the overall genetic information.

Measurement of species diversity

The measurement of species diversity, in its ideal form, consists of a complete catalogue of the distribution and abundance of all species in the area under consideration. However, this measurement is usually not possible unless the area under consideration is a small area. Therefore, in practice the measurement of species diversity is often based on samples. The central measures of species diversity are the a , b and g species diversity, as originally proposed by Whittaker (1960 and 1972). a -diversity refers to the number of species using only their presence (and not abundance) in a given area. It therefore measures the species richness of a given sample plot. The use such a species diversity measure would imply that an area containing a high number of species would be preferred to one with a fractionally smaller number of species (Huston 1994).

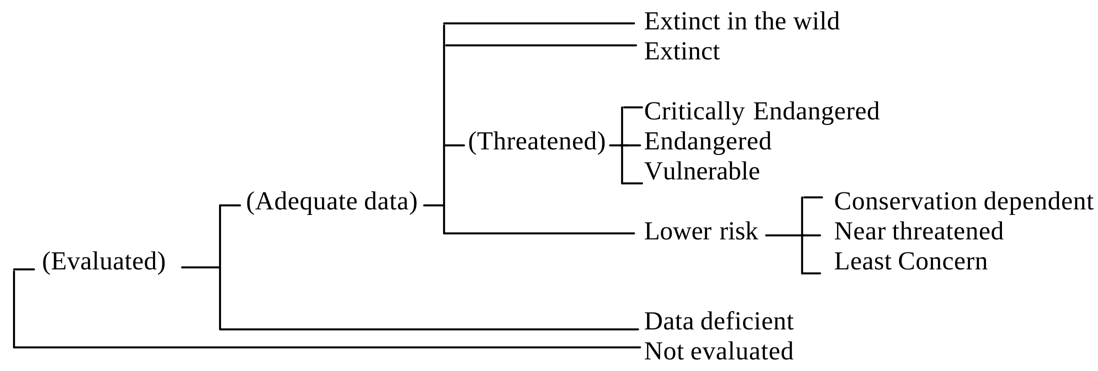
Another measure of species richness is g -diversity. It is often used to assess the overall diversity within a large region and its comprehension has direct implications when dealing with biodiversity at the landscape level (Noss 1983, Franklin 1993). National species lists, usually the only information available, can be treated as lower bounds on gamma diversity. Colombia and Kenya, for example, are the homes to over 1,000 species of birds, while the UK and the forests of eastern North America are homes to only about 200. A coral reef of northern Australia may be a home to 500 species, while the rocky shoreline of Japan may be a home to only 100 species (UNEP 1995). Finally, b-diversity measures the turnover of species between local areas, i.e. the rate of change in species composition among discrete sites or habitat units (Cody 1986 and 1993). As such it cannot be expressed in the number of species: it is represented in terms of an index and it is interpreted as a species turnover rate. b-diversity is generally used to estimate average changes in species in response to site or habitat heterogeneity.

Species richness measurement is useful, but it may depict a biased diversity assessment. First, scientists face great uncertainties about the total number of species. Therefore, species richness can be only be measured for some species (the species studied), but not for all the species in the area. As a matter of fact, only in a very few places on the planet are there rough estimates for the total number of species. Second, the size of the area is often arbitrary. Species diversity is associated to habitat scale in a complex way. Thus, one needs to be cautious when comparing the species diversity of areas that differ greatly in size. In addition, species diversity is a result of complex genealogical relationships that are not measured here. Alternative species diversity measures supplement species richness with measures of the degree of genealogical difference. Such diversity measures include the weighting of close-to-root species, higher-taxon richness, spanning tree length and taxomonic dispersion (Williams *et al.* 1991). However, and for the time being, practical difficulties with respect to the implementation of such measures confer reliance on the simpler indicators of species richness.

Measurement of ecosystem diversity

The measurement of biodiversity at the organisational level of ecosystem diversity encompasses a multi-complexity of relationships, both at the intra- and supra-species level, that play a crucial role in defining the overall distribution and abundance of species. For this reason, there are a number of factors that make the assessment of ecosystem diversity more blur and less clearly defined. Actually, at the ecosystem level many different units of diversity are involved, ranging from the patterns of habitats to the age structure of populations, including the patterns of communities on the landscape and patch dynamics. At these levels it is not clear where to draw boundaries delineating units of biodiversity. When a wetland, for example, is disturbed or degraded, we need to look at the impacts of the disturbance on the larger level of the landscape. Emphasis on a system-wide approach also serves to remind that the ecological value of an ecosystem may not be equivalent to the aggregate value of that same system's components. In other words, the system is more than just the aggregation of its individual parts; it possesses a primary value (Gren *et al.* 1994; Turner, Perrings and Folke, 1997). Furthermore, the conservation of biodiversity at the ecosystem level does not only underlie the preservation of species, but also contributes to the safeguard of the ecosystem functions and services. Thus, the full range of biodiversity values depends on the processes that support the functioning of larger-scale ecological systems. Given such unambiguous boundaries, there are different measurement approaches. These include bio-geographical realms or provinces, based on the distribution of species, and eco-regions or eco-zones, based on physical attributes such as soils and climates (UNEP 1995).

Table 1: Structure of species categories



Source: http://www.iucn.org/themes/ssc/redlists/categor.htm

Operationalisation of the biotic-richness approach: examples

Ecologists are often asked to contribute with their expertise and help policy makers to define conservation priorities. A considerable part of the ecologist's help is, in some way, related to Usher's ecological approach to environmental protection (Usher 1989). According to Usher's conservation assessment, decisions are characterised by three steps. First, attributes are identified and are used to reflect the conservation interest of the species or site. Second, criteria are developed for the expression of the attributes in a form that allows evaluation. Finally, values are attached to particular levels of criteria.

One important instrument for assessing species variety is the Red Data Book (e.g. ICN 1993). Since their introduction, Red Data Books were conceived as to identify threats or causes of decline of different species around the world (Fitter and Fitter 1987, Mace and Lande 1991, Mace *et al.* 1992, IUCN 1993, Mace and Stuart 1994). In short, Red Data Book is an evaluation technique characterised by the selection of a species list (the attribute) to assess the species richness (the criterion). Red Data Books classify species in one of eight different categories: extinct, extinct in wild, critically endangered, endangered, vulnerable, lower risk, data deficient and not evaluated – see Table 1. The goal is to provide an easily and widely understood method to order species in categories related to their threat of extinction under current circumstances[1] in order to (1) provide information on which to base conservation programs; (2) assist the drafting of legislation, and (3) to convey information comprehensible to a non-specialist. For this reason, Red Data Books are frequently used by numerous governmental and non-governmental organisations for policy guidance and the establishment of conservation priorities.[2]

Several problems however exist, rendering category assessment quite difficult to operationalise. While, the classification of a species to a category relies on an objective

[1] For example, "critically endangered" category refers to the species that is facing an extremely high risk of extinction in the wild in the immediate future. Alternatively, a species is "endangered" when it is facing a very high risk of extinction in the wild in the near future. Finally, a species is said to be "vulnerable" when it is not "critically endangered" nor "endangered" but yet is facing a high risk of extinction in the wild in the medium-term future (see Annex for more details).

[2] Most obviously these were required for selecting and designating the European Union 8,819 nature reserves under the Natura 2000 network, 6,977 within the Habitat Directive (92/43/CEE), and 1,842 within the Birds Directive (79/409/CEE).

evaluation, the actual definitions of these categories rely on a subjective view. In practice, the large number of available criteria (e.g. a , b and g criterion) used for evaluation in some way already reflect the difficulties that exist in conceptualising its value. Furthermore, the species category of threat is not necessarily sufficient to determine priorities for conservation action. Therefore, if Red Data Books were elected as the ecological method used in the OECD countries for the establishment of biodiversity priorities, it would be a risk that many sites would not have much of a value (and thus more likely not to be protected), since this evaluation technique simply provides an assessment of the likelihood of species extinction. Finally, given the scientific understanding of population and ecosystems, it is possible to develop alternative indicators, including the use of numerous other criteria concerning conservation action.

An early example of a multicriteria rating is the method proposed by Randwell (1969). The method was used to evaluate coastal habitats and combining the use of eight criteria into a single score, the Comparative Biological Value Index (CBVI). Each of these criteria are rated according the scale as described in Table 2 and the final score is obtained by summing up the scores for all the nine criteria:

$$CBVI = Ph + O + D + G + S + P + E + C$$

The maximum potential value is 28 and the minimum value is 7. The higher the CBVI value, the greater is the requirement for site protection. Since Randwell, the use of indices constitutes a popular practice in ecological valuation and management (see Spellerberg 1992 for an extensive review of CBVI assessment of landscape and urban habitats). However, this valuation approach relies on input criteria that require some subjective valuation. As such they may not be helpful to the decision-making process as originally intended. This problem is confounded as assessing and representing ecological values for policy guidance.

Recently, however, computer-based systems have been used to develop a general, integrated framework where one can simulate natural/management changes and assess the respective conservation evaluations. One specific output of this evaluation approach is to study population dynamics and estimate a minimum viable population, defined as the smallest population which as an acceptable probability of persisting over a given time period (Soulé 1987).[3] This in turn allows the calculation of the minimum dynamic area, i.e. the geographic area of suitable habitat required as to support the minimum viable population. Recent published studies on the leadbeaster’s possum (Lindenmayer *et al.* 1993), the eastern barred bandicoot (Lacy and Clark 1990), the Puerto Rico parrot (Howels and Edward-Jones 1997), and the giant panda (Zhou and Pan 1997) confirm the use of the population viability analysis in the policy decision circles and help conservation decisions. Another example of the application of computer system for ecological evaluation refers to the System for Evaluating Rivers for Conservation (SERCON) recently undertaken by the Scottish Natural Heritage, the nature conservation agency in Scotland (Boon *et al.* 1997). The overall objective was to predict the impact of different development scenarios on river ecological conservation value as well as to provide of a simple way of communicating such results to planners, developers and policy-makers.

[3] Several general models have been developed for this task, e.g. VORTEX (Lacy 1993) and METAPOP (Akcakaya 1994), and these can be used to consider the complex interactions between demographic, environmental and genetic influences on a population.

Table 2: *Rating of the criteria used by Randwell for evaluating coastal habitats*

Criteria	Type		Rating
Pysicochemical featuares (Ph)	High speciality		3
	Some special features		2
	Type example		1
Optimum populations (O)	Best populations of one or more local species		4
	Large populations of local species		3
	Large populations of common species and small populations of local species		2
	Representative populations		1
Diversity (D)	Outstanding populations		3
	High diversity		2
	Species ranges small		1
Geographic units (G)	Many species at limit		3
	Some species at limit		2
	Dew species or no species at limit		1
Size (S)	Mud-flats (ha)	Cliffs (km)	
	> 4000	> 80	5
	1600 – 3999	40-79	4
	800 – 1599	24-39	3
	400 – 799	8-23	2
	< 400	<8	1
Purity (P)	Little disturbance		3
	Moderate disturbance		2
	Much ground disturbed or polluted		1
Education and research use (E)	Much used		3
	Some use		2
	Potential use		1
Combinatory value (C)	Adjacent to another habitat of likely national value		4
	Adjacent to another habitat of likely regional value		3
	Adjacent to another coast habitat site not spoilt by development		1
	Surrounded by developed coastline		0

Source: Randwell (1969)

Ecological valuation based on computer modelling presents important advantages. First, it has encouraged greater rigour in data evaluation, since it permits the introduction of elements of subjectivity within a transparent and repeatable framework. Second, it allows a direct comparison of alternative conservation policies, independently of the number of the involved criteria and respective attributes (e.g. it permits the comparison of a conservation policy involving a criterion with ten attributes which could score a maximum of 50, with another strategy involving a criterion with two attributes that could only score a maximum of 10). Finally, it permits an evaluation even when some of the attribute data are missing, which is quite common in practice.

2.3 Ecological indicators of biodiversity: (2) ecosystem health/integrity approach

Ecological valuation methods are not only aimed at assessing diversity and rarity of species, but also the complex interactions between the biotic and abiotic environments, based on the assumption that the variety of abiotic conditions is equally important as variety of species. 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). Therefore, from an ecosystem perspective, the definition of biodiversity value is intrinsically related to ecosystem performance and integrity. Furthermore, the term "value" indicates how well an ecosystem is functioning when compared to its own potential and how important this is for the functioning of other ecosystems and, ultimately, for the functioning of the global ecosystem (Sijtsma *et al.* 1998).[4]

Ecosystem health

Ecosystem health is an overall indicator of the ecosystem functioning (or ecosystem integrity), taking into account both ecological and human processes. An ecological system is said to be healthy, if it is stable and sustainable, i.e. if it is active and maintains its organisation and vigour over time and is resilient to stress. In short then, ecosystem health can be defined as "(…) a measure of the overall performance of a complex system that is built up from the behaviour of its parts (…)" (Costanza 1992, pp. 241). Before we can measure the health of an ecosystem it is, however, necessary to proceed to identify biotic and abiotic parameters or indicators, target human economic activities, and the scale or hierarchy of analysis - see Figure 1.

When selecting the parameters, and indicators, it is important that these are relevant to the analysis. Most of the biotic and abiotic parameters or indicators, such as indexes describing the soil, flora, and fauna, have emerged from the ecological literature (Odum 1971). Furthermore, the measurement of the selected biotic and abiotic indicators needs to be a feasible task, so that it is possible to come up with plausible, valid figures. In addition, it is necessary to frame the scale, or hierarchy, of analysis. The choice of the scale relates to important decisions over boundary and temporal perspective of analysis (Norton and Ulanowicz 1992). It is common that boundaries are drawn in accordance with the ecosystem's land features or its geography (e.g. wetlands ecosystem). Finally, the measurement of ecosystem health also requires the identification of the human actions that influence the ecological structure and processes. The underlining idea is that human economic activity or target groups, e.g. consumers, industry, influence their environment. As a result, information on the impact of economic activity on the ecological structure and processes, in general, and on economic indicators, in particular, needs to be taken into account when assessing the ecosystem health.

It is also clear that both ecological and human dimensions are dynamic; they change in different ways and according to multiple frames of time. Consequently, biotic, abiotic, and economic indicators must be sufficiently dynamic to change accordingly. For example, even if we have clearly defined the spatial and temporal boundaries of analysis, there could remain important questions regarding the structure of the ecosystem, originally bounded and described as a whole. Therefore, there would be the need to represent boundaries at a smaller scale. Each part is characterised by its own set of indicators and is assessed individually. This set of indicators can differ substantially from part to part, i.e., from ecosystem to ecosystem (Costanza *et al.* 1992).

[4] From an ecological perspective, the definition of 'value' indicates how well an ecosystem is functioning compared to its own potential (Sijtsma *et al.* 1998). Furthermore, if two ecosystems functioning equally well with respect to their own potential, one system may be more valuable because of its positive synergies to the well-functioning of other systems.

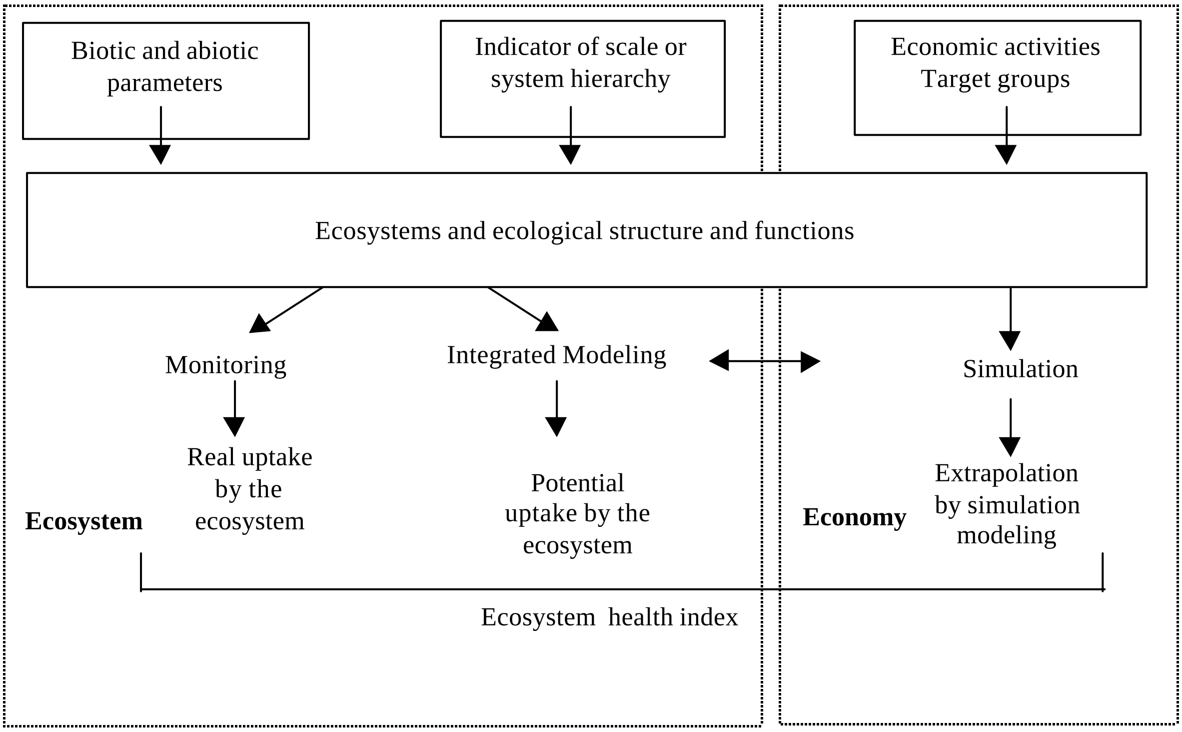


Figure 1: Definition of ecosystem health indexes

After having decided upon the biotic and abiotic indicators, scale or hierarchy of analysis and the target economic groups, the scientist is in a good position to proceed with the measurement of the overall ecosystem performance, i.e. ecosystem health. One possible approach is to use directly the available data provided by monitoring activities. This data will provide information with respect to the real uptake by the ecosystem and this way allow to measure the overall ecosystem health. Alternatively, one can integrate the relevant concepts and develop an analytical framework. The combination of the available data and such a model formulation will allow for a further step in the assessment of ecosystem health. At this stage, the scientist is not only to use the available data to deduce ecosystem health, but also explore the dynamics of integrated modelling so as to estimate the potential uptake by the ecosystem. Finally, the scientist is able to simulate different conservation scenarios by manipulating the observed characteristics, or ‘controlling variables’ - e.g., introduction of a new set of concentration ratios. For each conservation scenario the scientist is able to compute the associated ecosystem health index and thus provide crucial information as to rank the alternative management policy scenarios.

Examples of ecosystem health indicators

The construction of ecosystem health indexes allows policy makers to predict ecosystem response as a result of various specific management alternatives and natural changes. In practice, this formulation leads to the following form for an overall system health index (HI):

$$HI = V*O*R$$

Table 3: Indices of vigour, organisation and resilience

Components of health	Related concepts	Related measures	Field of origin	Measurement solution
Vigour	Function	GPP, NPP, GEP	Ecology	Monitoring
	Productivity	GNP	Economics	
	Throughput	Metabolism	Ecology	
Organisation	Structure	Diversity index	Ecology	Network
	Biodiversity	Mutual information predictability	Ecology	analysis
Resilience		Scope for growth	Ecology	Simulation modeling

Source: Costanza (1922)

where *V* designates system 'Vigour' and represents a cardinal measure of the ecological system activity, metabolism, or primary productivity; *O* designates system 'Organisation' and represents a 0-1 index of the relative degree of the ecological system's organisation, including its diversity and connectivity; and *R* designates system 'Resilience' and represents a 0-1 index of the relative degree of the ecological system's resilience – see the components in Table 3. In short, and to conclude, the overall ecological system health is given by its activity weight through indices for the relative system's organisation and resilience. To operationalise the vigour, organisation and resilience components, the health index will require the application of different measurement solutions to current data, involving the use of expertise from both economics and ecology.

Ulanowicz's ascendency index

One important example of ecosystem-health index, Ulanowicz's ascendency index, allows for an integrated, quantitative, hierarchical measurement of ecosystem health (Ulanowicz 1992). In simple terms, the ascendency index reflects any degradation of the system. However, such an indicator requires data on all transfers occurring in the ecosystem under consideration. The collection of this data is usually a laborious, expensive task. For this reason, fully quantified networks of ecosystems still remain scarce (Costanza 1992). In addition, it is important to remember the distinction between a scientific-orientated methodology and a policy-orientated approach. In practice one can find several valuation methods that combine a set of ecosystem integrity indicators into one common denominator that has a socio-politically appeal and thus a meaning for the policy makers. The following sub-sections review the operationalisation of some of the ecological valuation methods, most of them with an application to the Dutch regional and national policy agenda (see Ruijgrok 1999).[5]

The ecosystem classification method

Brink and Hosper (1989) developed a General Method for the Description and Evaluation of Ecosystems, known by the Dutch acronym AMOEBE.[6] The method was originally used to assess the quality of aquatic ecosystems by comparing the presence of selected species with a their presence in a benchmark situation of 1930. The selected species, which Brink and

[5] The characteristics of the Netherlands, a country with the highest population density in the European Union and home of important human economic activity, urge the use of this ecological valuation methods more than in any country.

[6] Algemene Methode voor OEcosysteembeschrijving en Beooordeling.

Hosper called ‘target variables’, were selected on the basis of: (1) their representativeness (i.e. do they represent a healthy aquatic ecosystem); (2) their flexibility (i.e. can they be influenced by human interventions), and, finally (3) their measurability and data availability (i.e. are they easy to measure and/or are there data-bases available). Since the AMOEBE does not indicate whether one ecosystem is more valuable than another (the value of the ecosystem is a function as a deviation from its own potential), the valuation method is not often selected with the goal of policy formulation guidance.

Nature measurement method

In 1995, the Dutch Centre for Agriculture and Environment in Utrecht developed the nature measurement method as to assess the natural values of agricultural areas. The natural values were measured in terms of species abundance and its deviation from its own diversity potential. Species were selected that are present on agricultural plots, easy to recognise and that represent natural quality (Buys 1995). A similar formula was developed by the Foundation for Spatial Economics of the University of Groningen to assess the costs and benefits of the National Ecological Network (Sijtsma and Strijker 1995). The costs were valued in monetary terms and the benefits mostly in ecological terms. The National Ecological Network used the identification of ‘nature target types’ (i.e. pre-defined types of nature such as the European CORINE network) that are assumed to provide the habitat mosaic for ‘target species’. To this purpose, digital thematic maps are generated as geographic information systems land cover habitats types. Target species are, in turn, classified and selected on basis of national and international rarity. Zurlini, Amadio and Rossi have recently applied this valuation method to depict the map of Italian nature (see Zurlini *et al.* 2000).

Ecological effect measurement method

In 1996, the Centre for Environmental Studies in Leiden developed an ecological effect measurement method to value the effects of housing development projects on nature and landscape. The method is as a first step characterised by the description of the reference situation and human intervention measures, followed by the determination of the effects on nature and, finally, proceeding to the aggregation of effects (Cuperus and Canters 1995). The main ecosystem biotic and abiotic characteristics were used for valuation, including spatial diversity (e.g. variables on soil and vegetation structures), abiotic functioning (e.g. variables on temperature, sediments, water and soil), fauna and flora communities as well as their relations with the surroundings (e.g. variables on hydrologic, geomorphologic and biomass relations). Since this valuation method takes into account both biotic and abiotic diversity assessed by means of the deviation from a reference situation, it can easily be transferred to the policy arena and used, for example, to determine compensation measures in the case of damage to existing natural areas.

Ecological capital index

More recently, the Dutch Environmental Planning Bureau in Bilthoven developed the ecological capital index with the objective to assess the state of both natural and cultural ecosystems in relation to human activities. This ecological capital index is calculated by multiplying the ecosystem’s quantity by its quality The abiotic environment is here regarded as a conditional variable for such a biodiversity reference situation (RIZA 1999). The international application of the ecological capital index respects the recommendations for a core set of indicators as proposed by the Convention on Biological Diversity (UNEP 1997) and thus is compatible with the international classification (IUCN 1991) of ecosystems on the basis of the degree of human influence.[7]

To conclude, ecosystem health indexes allow the scientist to assess the overall ecosystem performance. A number of examples have been identified here. These play a crucial for policy guidance since they let us to predict ecosystem response as a result of alternative management scenarios and, this way, making possible scenario's rankings to be compared.

3. Economic foundations for biodiversity analysis and valuation

3.1 Why economists pursue economic valuation

The economic valuation of natural resources, in general, and biodiversity, in particular, is among the most pressing and challenging issues confronting today's environmental economists. In short, economists value biodiversity because such a valuation exercise allows for a direct comparison with economic values of alternative options and facilitates, for example, cost-benefit analysis – a crucial tool for policy formulation. In addition, the monetary valuation of biodiversity allows economists to perform environmental accounting, natural resource damage assessment, and to carry out proper pricing. Moreover, the valuation exercise is revealed to be essential in the research of individual consumer behaviour and investigate what does the individual consumer think of certain biodiversity management objectives, or identify individual consumer's motivations with respect to biodiversity conservation.

Many people, however, do not accept to a place monetary values on biodiversity. Arguments against it are rooted in the human preference orientation that 'guide' consumer behaviour with respect to biodiversity (Ehrenfeld 1988, Lockwood 1999). At the risk of oversimplification, we can distinguish two broad ranges of value orientations - see Table 4. According to the 'anthropocentric' orientation, the value of biodiversity is an outcome of its role in human welfare, as humans conceive it "(…) whether they are selfish, altruistic, loyal, spiteful, or masochistic (…)" (Becker 1993). A second valuation perspective is rooted in an 'biocentric' or 'ecocentric' value orientation, which claims that nature has an intrinsic value[8] and therefore deserves protection. In reality, however, value orientations are overlapping and several versions of 'anthropocentrism' and 'ecocentrism' can exist within one individual. Q-altruism and stewardship are examples of such 'mixed' attitudes (Sagoff 1980, Norton 1982, Van der Veer and Pearce 1986). Stewardship is a form of altruism that is fully divorced from any explicit notion of consumption. It corresponds to a sense of responsibility – usually in a Christian perspective - for the conservation and maintenance of the resource. Q-altruism is rooted in the firm belief that living organisms are incapable of protecting themselves against human actions. Therefore, the conservation of living organisms merits humans' sympathy or compassion.[9]

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[7] The International Union for Nature Conservation, under the auspices of the United Nations, makes a distinction between natural, adapted, cultivated, built and deteriorated ecosystem.

[8] By definition an intrinsic value is the value of the resource *per se* without a subject attaching a value to it.

[9] Recently, Nunes (1999a) developed the use of multiple motivation items scales to measure consumers' level of "anthropocentricism" and integrate such a construct in a contingent valuation application. The empirical results confirmed that the individual level of "anthropocentricism" had a robust econometric role in the economic

Table 4: Value orientations and environmental attitudes

Value orientation	Valuation perspective	Ethical approach	Biodiversity attitude
Ecocentrism	rights conferred to all living organisms	Nature has intrinsic value, regardless of human recognition	Biodiversity first
Antrophocentrism	rights and interests conferred to individual humans	Value of nature is value conferred to humans	Humans first

Adapted from Lockwood (1999)

3.2 Biodiversity as a source of economic values

The concept of total economic value of an environmental resource has its foundations in welfare economics. It focuses on changes in the economic welfare of humans. Therefore, the terms 'economic value' and 'welfare change' can be used interchangeably.[10] The total economic value (TEV) of an environmental resource consists of its use value (UV) and nonuse value (NUV). Use values arise from the actual use. It can be further divided into direct use values (DUV), indirect use values (IUV) and option values (OV) (Pearce and Moran 1994). The nonuse values are usually divided into a bequest value (BV) and an existence value (XV). Bequest value refers to the benefit accruing to any individual from the knowledge that others might benefit from an environmental resource in the future. Existence value refers to the benefit derived simply from the knowledge that the resource is protected. This leads to the following equation

$$TEV = UV + NUV = (DUV + IUV + OV) + (XV + BV)$$

Economic values of changes in biodiversity require special attention. We classify such values into four categories. These are illustrated in Figure 2. A first category, denoted by link 1   6, depicts biodiversity benefits in terms of ecosystem functions and services (see Section 2.1). We can interpret this biodiversity value category as a direct use value component. A second category, denoted by link 1   4   5, captures the value of biodiversity in terms of supply of ecosystem space or natural habitat protection. We can interpret this biodiversity value category as an indirect use value component. A third category, denoted by link 2   5, captures the benefits conferred to society in terms of an overall diversity provision of biological resources, notably specific animal and species for use in agriculture and medicine. We can interpret this biodiversity value category as a direct use value component. A fourth category, captured by link 3, refers to the direct impact of biodiversity on human welfare. The economic value of biodiversity is then measured in terms of philanthropic considerations, independently of biodiversity use or consumption. One can interpret this biodiversity value category as a nonuse component. The nonuse values have a public good character for which no market price is available. As a consequence, most of the time policy makers have based their decisions on an undervaluation of biodiversity benefits, which has resulted in a misuse

valuation exercise. Moreover, the respective coefficient estimate show to vary with the object submitted to valuation.

[10] It should be clear that economists do not pursue total value assessment of an environmental system, but of system changes (preferably marginal or small).

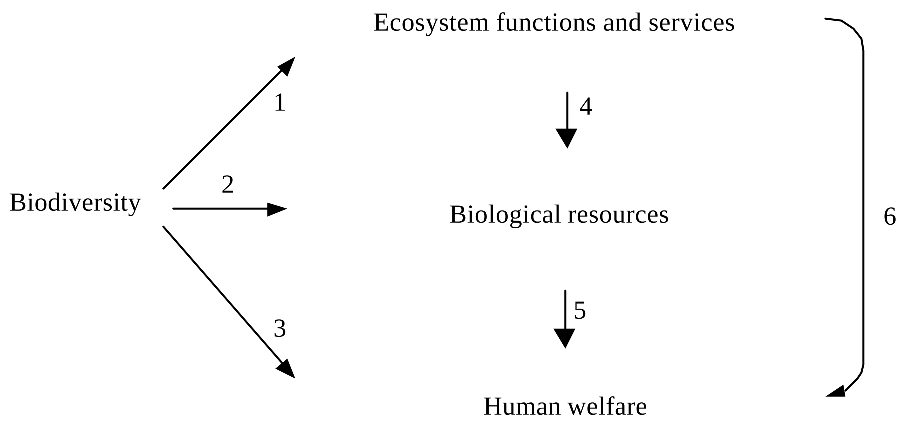

Figure 2: Economic values of biodiversity

and misallocation of scarce resources. The monetary assessment of biodiversity use and nonuse benefits requires the handling of special valuation tools. These are discussed in the following section.

3.3 Alternative monetary valuation methods

In the absence of market prices, certain techniques are needed to retrieve consumers' preferences for biodiversity benefits. On the basis of the biodiversity component to be valued, one can distinguish alternative valuation methods: travel cost, hedonic price, averting expenditures cost, production function, and contingent valuation,[11] - see Table 5. Travel cost is also a frequently used economic valuation method (Bockstael *et al.* 1991). It is especially designed to assess recreational benefits by looking at the generalised travel costs (site entrance fees, accommodation and journey costs) when visiting a natural park or reserves. Alternatively, when using the hedonic price method (Palmquist 1991) to estimate the economic value of a biodiversity benefit, say, good soil conditions, researchers explore the analysis of land market values and the soil quality characteristics (e.g. depth of topsoil). For example, King and Sinden (1988) explored the hedonic price method in order to capture the value of soil conservation in the farm land market of Manilla Shire, in New South Wales, Australia. The hedonic land market price regression results show that the implicit marginal price of land condition is estimated to be \$2.28/ha. Researchers can also estimate the economic value of biodiversity on the basis of consumer's expenditures made to avert or mitigate the adverse effects (Cropper and Freeman 1991) derived from the loss of biodiversity benefits. Huszar (1989) studied wind erosion costs that New Mexico households' incur from increased cleaning, maintenance and replacement expenditures, and reduced consumption and production opportunities. In addition, the economic value of biodiversity can be assessed through a production function. Recently, Barbier (1994) conducted a study focused on the value assessment of the economic importance of Hadejia-Jama'are wetlands, Nigeria. The valuation is based on the estimation of some key direct use values that the floodplain provides to local population through crop production, fuelwood and fishing.

[11] Many other classifications of valuation exist. See Freeman (1979).

Table 5: Identification of monetary valuation methods according to the different biodiversity value components

TOTAL ECONOMIC VALUE (TEV)	USE VALUES (UV)	Direct use value (DUV)	Recreation benefits, e.g. sight-seeing, fishing, swimming Methods: Travel cost, contingent valuation
		Indirect use value (IUV)	Ecosystem functional benefits, e.g. regulating local chemical composition of the water Methods: production function, averting behaviour, hedonic price
		Option Value (OV)	Insurance for having the asset on *stand-by*, e.g. future visits, future genetic manipulation Methods: contingent valuation
	NONUSE VALUES (NUV)	Bequest Value (BV)	Legacy benefits e.g. habitat conservation for future generations Methods: contingent valuation
		Existence value (XV)	Existence benefits, e.g. knowledge of existence of marine wildlife diversity Methods: travel cost, contingent valuation

Finally, the contingent valuation method collects data to retrieve individuals' preferences on biodiversity by means of questionnaires, nowadays often supported by multidisciplinary research teams (Carson *et al.* 1992, NOAA 1993). Contingent valuation presents important features. First, it is able to express the Hicksian welfare measure directly in monetary terms. Second, the CV method is the only valuation technique which is capable of shedding light on the monetary valuation of the nonuse values, i.e., the benefit value component that is not directly associated with its direct use or consumption. Ignoring such values will be responsible for a systematic bias in the estimation (essentially an underestimation) of the total benefits of biodiversity. Moreover, CV brings with it the advantage that environmental changes may be valued, even if they have not yet occurred (*ex ante* valuation), offering a greater potential scope and flexibility in specifying different states of nature that may even lie outside the current institutional arrangements or levels of provision. For the same reasons, CV can be used to assess almost all value components of biodiversity and thus is frequently used in environmental policy.[12] Contingent, and other, valuation studies will be reviewed and discussed in more detail in the following section.

3.4 Empirical valuation studies

This section reviews some biodiversity valuation studies, presenting a valuation range for each biodiversity value component.[13] The discussion is organised according to Figure 2. First, we focus on studies that perform a value assessment of biodiversity in terms of the benefits associated with the provision of species diversity (see link 2 5). Second, we focus on valuation studies that pursue the assessment of biodiversity benefits in terms of protection of natural habitat or ecosystem space (see link 1 4 5). Third, we discuss valuation studies that focus on biodiversity's benefits conferred to the society in terms of ecosystem functions and

[12] An exception is made for the ecosystem functional benefits since these are hard to describe in a questionnaire format and they are often related to issues not particularly familiar to most respondents.

services, see link 1 6. Finally, whenever the CV method is applied, one is able to assess the nonuse value component of biodiversity, and thus capture the monetary value of link 3.[14]

Single, multiple species valuation surveys

One of the first species valuation studies was conducted by Boyle and Bishop in 1987. They conducted a CV study to estimate the existence values for wildlife species. The estimated values suggest that substantial existence values are associated with the Bighorn Sheep in Wyoming (see results in Table 6). More recently, Van Kooten (1993) studied the economic value of waterfowl in Canada. According to the results, the shadow values of marginal land converted to waterfowl habitat were estimated to be $50 to $60 per acre. Loomis and Larson (1994) valued an “emblematic” endangered species, namely the Gray Whale. The willingness to pay (WTP) was estimated between sixteen and eighteen dollars per household. Boman and Bosdedt (1995) carried out the economic valuation of the conservation of the Wolf in Sweden. The estimation results show that the mean WTP is about $126 per year.

Table 6 shows also multiple species valuation studies. Johnansson (1989) conducted a CV study that focus on the preservation of 300 endangered species in Sweden. As we can see, the final estimate, $ 194 per year, is higher than economic valuation of the conservation of the Wolf in Sweden, about $126 per year, though not so high as one would expect. This is partly because the single species valuation study refers to popular, socially attractive, charming and beautiful species. Therefore, when preserving the Wolf *in situ*, as one of the faunal emblems of Sweden, the final value estimate may also embed other characteristics that not directly related to the species to be valued. In other words, the valuation results may also reflect individual social esteem motives or warm glow (Nunes 2000a).[15] Furthermore, the estimate valuation for single species can be affected by the availability of related species, i.e. single species values can be affected by the respondent’s perception of substitutes and complementary species (Samples and Hollyer 1989). For example, preserving a Wolf *in situ* also preserves the habitat for other endangered species; namely the ones that share the same environment as a Wolf. Therefore, the WTP for Wolf may be interpreted as a result embedding a valuation of a wider range of species. For this reason, some authors prefer to work with a higher aggregation level of species diversity and carry out valuation surveys at a habitat protection level.

Diversity at the ecosystem level

The majority of CV studies that focus on of biodiversity at the ecosystem level link directly the valuation of biodiversity to the non-use or recreational valuation of habitat protection programs. The main reason for such a link is primarily the difficulty associated with defining in a survey such an abstract concept such as ecosystem diversity. Indeed, some CV studies indicate that the concept of biodiversity is ill understood among the general population (Hanley *et al.* 1995). However, a number of valuation studies have attempted to value biodiversity conservation policies by using other methods. Generally speaking, we can find

[13] Nunes and van den Bergh (2000) list a more extensive bibliography of work related to valuation of biodiversity.

[14] Since nonuse values have no behavioural market trace, economists cannot retrieve information about these values by relying on market-based valuation approaches.

[15] This is a well-known critique supported by economists such as Jerry Hausman, Peter Diamond, William Desvousges and Paul Milgrom, who express their doubts with respect to the suitability of CV results for inclusion in benefit-cost analyses (Hausman 1993).

Table 6: Valuation studies

	Valuation study	Method
Single species		
Minimum range: $5	Stripped Shiner, endangered species in Wisconsin, US Boyle and Bishop (1987)	Contingent valuation
Maximum range: $126	Wolf, endangered species in Sweden Boman and Bosdedt (1995)	Contingent valuation
Multiple species		
Minimum range: $18	Preservation of threatened and endangered species populations in the US, Hageman (1985)	Contingent valuation
Maximum range: $194	Preservation of 300 endangered species in Sweden, Johnansson (1989)	Contingent valuation
Habitat: Terrestrial (non-use)		
Minimum range: $27	Protection of the Nadgee Nature Reserve, Australia, Bennett (1984)	Contingent valuation
Maximum range: $101	Desert Protection in California, US Richer (1995)	Contingent valuation
Habitat: Coastal (non-use)		
Minimum range: $10	Protection of New Jersey beaches, US Silberman *et al.* (1992)	Contingent valuation
Maximum range: $51	Protection of a wilderness coastal area, Portugal Nunes (2000b)	Contingent valuation
Habitat: Wetland (non-use)		
Minimum range: $8	Protection of the Norfolk Broads, UK Batemann *et al.* (1992)	Contingent valuation
Maximum range: $96	Enhancing wetland habitat in California, US Hoehn and Loomis (1993)	Contingent valuation
Habitat: Ecosystem space (recreation)		
Minimum range: $23/trip	Forest recreation activities in Flanders, Belgium Moons (1999)	Travel cost
Maximum range: $23 million/year	Tourism in Ecuador WTO (1997)	Tourism revenue
Ecosystems functions		
Minimum range: $1.2 million	Life-support value of a wetland ecosystem in the a Swedish island, Baltic Sea, Turner *et al.* (1995)	Production function
Maximum range: $4.4 billion	Water ecosystem benefits in ten regions in US Ribaudo (1989)	Averting behaviour

studies that focus on ecosystems functions and the value assessment of life-support, soil and wind erosion, or water quality benefits. Some of these studies are listed in Table 6.

During the 1980s many contingent valuation studies dealt with the measurement of the non-use benefits derived from the conservation of terrestrial parks and nature reserves - see Bennett (1984) and Richer (1995). The valuation applications continued through the 1990s –

see Silberman *et al.* (1992), Batemann *et al.* (1992), and Hoehn and Loomis (1993) – but now also tackling the valuation of non-use benefits of coastal and wetland habitats. Silberman *et al.* estimated the existence value for users and nonusers of New Jersey beaches. The results show that the mean WTP for a user is about $15.1, while the mean WTP for a nonuser is about $9.26. Batemann *et al.* (1992) undertook a contingent valuation to assess the monetary value of conserving the Norfolk Broads, a wetland site in the UK gathering three National Nature Reserves. A mail survey across Britain showed that non-visitors respondents were willing to pay, on average, 4 pounds (circa $8). More recently, Nunes (2000b) used for the first time the CV method in Portugal to assess the national WTP for the protection of a coastal natural area. The mean WTP results ranged from $40 to $51.

In the recreation domain, the World Tourism Organisation (WTO 1997) estimated that Ecuador earned $255 million from eco-tourism in 1995. A major sum accrued to a single park, the Galapagos Islands. Studies of less popular areas indicate lower values. The recreational value of a Regional Forest Park in Belgium was estimated to be around $23 per trip (Moons 1999). Norton and Southey (1995) calculated the economic value of biodiversity protection in Kenya by assessing the associated opportunity costs in terms of forgone agricultural production, which is estimated to be $203 million. This valued can be compared with $42 million in net financial revenues from wildlife tourism and forestry. More recently, Chase *et al.* (1998) studied the eco-tourism demand in Costa Rica. The value estimates result from the survey of foreign visitors to three national parks: Volcan Irazu, Volcan Poas, and Manuel Antonio. The highest WTP registered was about $25 for the Manuel Antonio national park.

When it comes to the monetary valuation of ecosystem functions, CV may not be the first method choice. This is because ecosystem life support is not an issue that the general population is familiar with. In addition, the complexity of the relationships involved makes an accurate and comprehensive survey description harder. Researchers frequently end up with the use of valuation methods such averting behaviour, production function, or hedonic pricing. In 1991 Andreasson-Gren (1991) estimated the costs of nitrogen abatement via wetlands restoration with the costs for using conventional technologies. The estimated nitrogen purification capacity of wetlands was based on the results of a Swedish island in the Baltic Sea, Gotland. According to the study results, the total value of a marginal increase in nitrogen abatement on Gotland was about SEK 968 per kilogram. Turner *et al*. (1995) addressed the valuation of a wetland ecosystem in the Swedish island in the Baltic Sea exploring the use of production function. Their value estimations confirmed that a considerable amount of industrial energy would be necessary to substitute for the loss of wetland life-support functions – see results in Table 6. Ribaudo (1989) is responsible for one of the most comprehensive study valuing water ecosystems. The author valued the economic benefits from the reduction in the discharge of pollutants in waterway systems for nine impact categories: recreational fishing, navigation, water storage, irrigation ditches, water treatment, industrial water use, steam cooling, and flooding. Benefits were defined in terms of changes in defensive expenditures, changes in production costs, or changes in consumer surplus, depending on the damage category and the availability of data. The total water quality benefits were estimated to be $4.4 billion.

4. Integrated ecological-economic modelling and valuation of biodiversity

4.1 Economics – Ecology interface

The analysis and the modelling of biodiversity are rooted in both natural and social science domains and thus imply the study of human economic activities, its relationships with respect to biodiversity, the structure and functions of ecosystems. The combination or integration of the two approaches will imply 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 purpose (Perrings *et al.* 1995). The underlying objective of this approach is the development of a common way of thinking about modelling and valuation of biodiversity. For instance, if economic and ecological models fit in a (general) systems frame, then they may be blended in 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. Alternatively, if both the economic and ecological systems are represented in the form of programming or optimisation models then several options are available: look for a new, aggregate objective; adopt a multi-objective or conflict analysis framework; or, when possible, derive multiple sets of optimality conditions and solve these simultaneously. Finally, when economic and ecological systems are represented by different model types, it is harder to suggest how they can be linked to one another. Where economic models have an optimisation format and ecosystem models have a descriptive format, then a direct technical integration seems feasible, otherwise heuristic approaches are needed. This may require operations such as reduction, simplifying or summarising. For example, one can come up with a simple dynamic model summarising and simplifying some of the causal relationships of the spatial hydrological model and the statistical vegetation model, and linking the outcomes to a simplified economic interaction and values model.

Before discussing specific methods and models it seems useful to say a few words about pros and cons of integration frameworks and respective conceptual perspectives. These are discussed in the following section.

4.2 Frameworks and theories underlying integrated modelling

A general method to develop integrated models is a systems approach (also 'systems dynamics'). This covers a wide range of model types: linear versus nonlinear, continuous versus discrete, deterministic versus stochastic, and optimising versus descriptive. Such system approaches allow dealing with concepts like ecological dynamic processes, feedback mechanisms, and controlling 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 be either considered as black boxes or described as empirical or logical processes themselves. The systems approach is suitable to integrate existing models, and to incorporate temporal as well as spatial processes. Costanza *et al.* distinguish economic, ecological and integrated approaches on basis of the criteria: (1) generality, characterised by simple theoretical or conceptual models that aggregate, caricature and exaggerate; (2) precision, characterised by statistical, short-term, partial, static or linear models with one element examined in much detail; and (3) realism, characterised by causal, non-linear, dynamic-evolutionary, and complex models. These three criteria are usually conflicting, so that a trade-off between them is inevitable. 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.* (1999) and Rotmans and de Vries (1997) for integrated analysis and modelling. The components have the following interpretation:

- '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 artefacts (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 behavior), pressures (mitigation), states (relocation) or impacts (restoration, health care).

According to Rotmans and de Vries (1997) integration can include various types. Vertical integration means that the causal chain in the PSIR or DPSIR framework is completely described in a model ("close the PSIR loop"; p. 25). 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). An alternative and relevant distinction is between analytical and heuristic integration. 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 extending this by an (finite) iterative interaction. In this case different model types can be combined, such as optimisation models and descriptive models. If one desires to attain a great deal of analytical power, the analytical integration seems attractive, whereas striving for realism would imply the use of a heuristically linked set of models from different disciplines.

Formal modelling and evaluation in integrated economic-environmental studies 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. Most disadvantages of integrated modelling apply to non-model based integrated research as well.[16] They include: unclear synergy of approximations and uncertainties; a rough application of monodisciplinary theories and empirical insights; a simplification of complex phenomena (e.g., by treating them as a black box); a misinterpretation and 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.

[16] 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 at least to be explicit about the latter two inputs to integrated research.

The main disadvantage of models perhaps is that they are trusted too much, that they run the risk of being interpreted as objective representations of reality, and then are taken too seriously by especially laypersons and policy makers. On the other hand, policy makers often indicate 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 that the controlled laboratory experiment has in natural sciences (and more recently in social sciences and environmental economics in particular; see Shogren 1999). 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 as 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. In addition, integrated modelling is also restricted by the model type.

4.3 Integrated assessment

Full or total economic-environmental 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 literature shows various examples of such integrated economic-environmental frameworks. Surveys are 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.* (1999).

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, as 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, 1993; van den Bergh and Nijkamp 1994) or 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 found in Australia. The model describes the interaction between extreme events (fire, flood, and draughts), 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 bounded rationally behaviour, supposedly in accordance with the reality in 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 *et al.* 1999). Swallow (1994) integrates theoretical models of renewable and non-renewable resources to address multiple use 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, 1983). These models include energy and mass balances. A central concept in this approach is 'embodied energy', defined as the direct and indirect energy required to produce organised 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 Jansson and Zuchetto (1978) (see also Zuchetto and Jansson 1985).Biophysical assessment models, and other integrated empirical applications, are reviewed and discussed in the following section.

4.4 Empirical applications

Striving for empirically sound models often implies modest approaches to improve precision, which usually goes at the cost of model use in a wider context or wider range of parameter values. The development of integrated models, by 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 steps like reduction, simplifying or summarising. The results may not always be greeted with enthusiasm within the disciplines, especially when they neglect certain nuances or different viewpoints. For example, the recent focus on integrated assessment of the enhanced greenhouse effect (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 provide AN integration of natural sciences (physics, chemistry, biology, earth sciences) and 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 disagregation. 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) – see Table 7. As we can see, integrated models can have different formats. One important distinction is between policy optimization and evaluation (usually numerical simulation) models. One of the first and famous integrated assessment models used in policy making is the model RAINS (Alcamo *et al.* 1990). This includes an optimisation algorithm for calculating cost-effective acidification strategies in Europe, aimed at realising 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 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

Table 7: Characterising integrated models

Model criterion	Range of choice	Examples of distinct approaches
Analytical integration	Optimisation (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, Meadows), 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, the present study
Spatial coverage	World; National; Regional; Urban, Local, Ecosystem	Ecosystem modelling, macroeconomic modelling, regional modelling, urban modelling, world models
Spatial disaggregation	Single region; Multi-region; Spatial grid (GIS)	Integrated assessment models (Climate change), land use models
Aggregation level	Micro (indiduals, households); Macro (national economy, main sectors, global); Sectoral; 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 (2000)

to climate change research. He emphasizes the attention for 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 models with an economic emphasis focusing on sustainable development questions, in theory and practice as well as at global and regional levels.

4.5 Integrating modelling and monetary valuation

Progress on improving methods for providing such economic information (particularly predictive information) will require a strong and dynamic interdisciplinary dialogue. At this level integrating modelling and monetary valuation can present important advantages for policy guidance, presenting important interactions. First, values estimated in a valuation study can be used as a parameter value in a model study. Benefits or value transfer (e.g. meta-analysis exercise) can be used to translate value estimates to 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 a general environmental change, regional development or ecosystem management. This can be fed into a model calculation, which in turn provides 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. Computer software can be used in such a process. Finally, the output of model and valuation studies can be compared. For instance, when studying a scenario for wetland transformation one can model the 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 a willingness to pay for not experiencing the change. If such information is available for multiple management scenarios, then rankings based on either approach can be compared.

5. Conclusions and recommendations

How can we use the ideas presented in Sections 2, 3 and 4 to formulate an integrated, effective framework addressing to the economic valuation of biodiversity? And what do we learn from the empirical valuation studies focused on biodiversity? A crucial, and initial step, is to carefully describe the object of analysis and valuation. Therefore, researchers face two important decisions when valuing biodiversity: (1) at which level should biodiversity be examined; and, (2) what biodiversity value types should be measured?

Assessment of the physical interactions

Clearly, it is necessary to attain information about the nature, type, and persistence of stress or shocks experienced by ecosystems, their functions and stability, and their impacts on human welfare (loss). A comprehensive assessment of ecosystem biodiversity characteristics, structure and functioning requires the analyst to undertake various important actions. First, the causes and consequences of biodiversity degradation/loss should be determined in order to improve understanding of socio-economic impacts on biodiversity processes and attributes. Second, the sustainability of biodiversity uses and negative impacts on the biodiversity caused by off-site human activities should be assessed. the range and degree of biodiversity functioning should be assessed, especially in terms of ecosystem-functional relationships. Third, the range and degree of biodiversity functioning should be assessed, especially in terms of ecosystem-functional relationships. Finally, alternative biodiversity conversion, development and conservation management strategies should be assessed and spatial and temporal systems analysis (via a range of methods and techniques) of alternative biodiversity change scenarios should be carried out.

Ecological evaluation

The physical assessment of the functions performed by biodiversity is an essential prerequisite to any ecological evaluation. However, simply identifying these functions is insufficient, if we want to present resource managers and policy makers with the relevant policy response options. It is necessary to develop criteria for the expression of the functions in a form that allows evaluation. Finally, values are attached to particular levels of criteria. The provision of such information is essential if an efficient level of biodiversity resource conservation is to be determined.

Ecologists contribute to the identify the range of policy decisions with respect to the biodiversity management/conservation strategies, by exploring the use of ecological valuation methods such as red data species lists and biological value indexes. More recently, computer models have been often used by ecologists to aid conservation evaluation decisions, namely in the domain of species population. Several general models have been developed for this task (e.g., VORTEX and METAPOP) and applied to calculate minimum dynamic areas, i.e. the geographic area of suitable habitat required to support the minimum viable population (e.g., the giant panda). Finally, computer models have also been used for habitat evaluation with the objective of simulating the impact of different development scenarios and predicting the respective ecological conservation values. This approach to ecological evaluation has attractive features. First, it permits the introduction of evaluation criteria within a transparent and repeatable framework. Second, it allows a direct comparison of management, or conservation, strategies. Finally, it permits an evaluation even when some of the data concerning the functions performed by biodiversity is are missing, which is quite common in practice.

Economic valuation

The concept of economic value has its foundations in welfare economics. It focuses on the welfare of humans, and in turn depends on the theory of consumer behaviour. Therefore, valuation in an economic sense is always the result of an interaction between the subject and an object. Moreover, economists do not pursue total value assessment of an environmental system, but rather system changes. This means that the terms ‘economic value’ and ‘welfare change’ can be used interchangeably. The goal is then to assess the human welfare significance of biodiversity changes, through determination of the changes in the provision of biodiversity related goods and services and consequent impacts on the well-being of humans who derive use or non-use benefits from such a provision. Biodiversity has been, however, frequently tackled in the empirical valuation studies with the measurement of the economic value of a particular set of biological resources, such as changes of the range of species on human recreation activities. Though, biodiversity has indirect links with various ecosystem goods and services. One reason why this monetary value component is not frequently assessed relates to the complexity involved in the ecosystems and thus their functions and processes can not be easily characterised. Therefore, one needs to have these two concepts clear and do not confuse the biodiversity economic valuation exercise with the economic value (of a specific) biological resource.

Different instruments are available to assess the economic value of biodiversity. Survey valuation studies have often been used because the use of revealed preference methods will leave out important biodiversity value types, notably non-use and quasi-option values. This can lead to a significant value measurement bias, especially if species conservation decisions are characterised by a high irreversibility. Alternatively, researchers can combine valuation techniques. Special attention, however, should then be given to value aggregation across the resulting values so as to avoid double counting.

Integrating modelling

The analyses and valuation of biodiversity are rooted in both natural and social science domains and thus imply the study of human economic activities, its relationships with respect to biodiversity, and the structure and functions of ecosystems. In practice, there is a clear need to obtain information about the cause, type, and persistence of stress on biodiversity and the estimation of the respective impacts on human welfare. The combination or integration of the ecological and ecological to assess and value biodiversity leads to an integrated framework. The underlying objective of this approach is the development of a common way of thinking about modelling and valuation of biodiversity. Interdisciplinary work is therefore required, involving economists or ecologists transferring elements or even theories and models from one discipline to another and transforming them for their specific purpose. This may require analytical operations such as reduction, simplifying or summarising. For instance, one can come up with a simple dynamic model by simplifying some of the causal relationships of the spatial hydrological model and the statistical vegetation model, and linking the outcomes to a simplified economic interaction and values model.

Last but not least, economists need to be aware of the limitations of any proposed valuation exercise. Despite all research efforts in an integrated, multidisciplinary analysis, modelling and valuation, it should be recognised that not all biodiversity value types can be made explicit nor that all explicit biodiversity value types can be measured in monetary terms. A valuation study is a partial and characterised by strict temporal and spatial boundary. As Gowdy (1997) has recently said "… 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…". In other words, monetary estimates of biodiversity should at best be interpreted as conservative estimations and thus regarded in terms of lower bounds. Clearly, more valuation studies need to be pursued to improve the information available to decision makers about the nature, magnitude and value on the impacts of human activities on the overall level of biodiversity.

Annex

The Criteria for Critically Endangered, Endangered and Vulnerable. Prepared by the IUCN Species Survival Commission As approved by the 40th meeting of the IUCN Council, Gland, Switzerland 30 November 1994.

Critically Endangered (Endangered/ Vulnerable)

A) Population reduction in the form of either of the following:

1. An observed, estimated, inferred or suspected reduction of at least 80% (50%/20%) over the last 10 years or three generations, whichever is the longer, based on (and specifying) any of the following:
    - (a) direct observation
    - (b) an index of abundance appropriate for the taxon
    - (c) a decline in area of occupancy, extent of occurrence and/or quality of habitat
    - (d) actual or potential levels of exploitation
    - (e) the effects of introduced taxa, hybridisation, pathogens, pollutants, or parasites.
2. A reduction of at least 80%, (50%/20%) projected or suspected to be met within the next ten years or three generations, whichever is the longer, based on (and specifying) any of (b), (c), (d) or (e) above.

B) Extent of occurrence estimated to be less than 100 km2 (5,000 km2/ 20,000 km2) or area of occupancy estimated to be less than 10 km2, and estimates indicating any two of the following:

1. Severely fragmented or known to exist at only a single location.
2. Continuing decline, observed, inferred or projected, in any of the following:
    - (a) extent of occurrence
    - (b) area of occupancy
    - (c) area, extent and/or quality of habitat
    - (d) number of locations or subpopulations
    - (e) number of mature individuals.
3. Extreme fluctuations in any of the following:
    - (a) extent of occurrence
    - (b) area of occupancy
    - (c) number of locations or subpopulations
    - (d) number of mature individuals.

C. Population estimated to number less than 250 (2,500/10,000) mature individuals and either:

1. An estimated continuing decline of at least 25% (20%/10%) within 3 years or one generation (5 years or two generations/10 years or three generations), whichever is longer, or
2. A continuing decline, observed, projected, or inferred, in numbers of mature individuals and population structure in the form of either:
    - (a) severely fragmented (i.e. no subpopulation estimated to contain more than 50 mature individuals)
    - (b) all individuals are in a single subpopulation.

D. Population estimated to number less than 50 (250/1,000) mature individuals.

E. Quantitative analysis showing the probability of extinction in the wild is at least 50% (20%/10%) within 10 years or 3 generations (20 years or 5 generations/100 years), whichever is the longer.

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