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Ecological Structure and Functions of Biodiversity as Elements of Its Total Economic Value

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Abstract. Rational economic decisions regarding the conservation of biodiversity require the consideration of all the benefits generated by this natural resource. Recently a number of categories of values (inherent value, contributory value, indirect value, infrastructure value, primary value) have been developed, especially in the literature of Ecological Economics, which, besides the individual and productive benefits of biodiversity, also include the utilitarian relevance of the ecological structure and functions of biodiversity in the, so-called, total economic value. For the question of including the ecological structure and functions of biodiversity in the total economic value it is of crucial importance to note, that these categories of values are not only terminologically different, but also relate to different ecological levels of biodiversity and – most importantly – to specific complementary relationships – between species, between elements of ecological structures and between ecological functions and their contribution to human well-being. This paper analyses these complementary relationships, discusses their implications for the total economic value of biodiversity and draws conclusions for decision making in environmental policy.

Key words: biodiversity, cost-benefit-analysis, ecological structure, ecological functions, total economic value, nonsubstitutability, ignorance, safe minimum standard

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1. Introduction

From an economic point of view biodiversity, i.e. “the variety of life at all levels of organization, from the level of genetic variation within and among species to the level of variation within and among ecosystems and biomes” (Tilman 1997, p. 93), must be seen as an asset, with biodiversity conservation as an investment and the neglect of conservation to be interpreted as a de-investment, which leads to a reduction of valuable services and, in turn, to economic costs. Therefore, the economic foundation of a decision for or against the protection of biodiversity requires the consideration of all costs and benefits related to it. It is well known that the main problem of including the range of biodiversity services in economic choices is, that many of these services are not valued on markets. There is a gap between market valuation and the economic value of biodiversity.

In order to fill these valuation gaps, those services of natural assets that are not valued on markets, but must nevertheless be considered beneficial, must first be *identified* and then – as far as possible – monetarised. In the case of biodiversity, the first step of analysis, i.e. the identification of economic-relevant services, is of special importance, because over the course of time those benefits not allocated by the market have continuously gained in importance. This becomes evident when looking at the historic development of the economic valuation of non-market services of biodiversity. Many services of biological resources were long considered intangible, were viewed as being irrelevant in economics or were entirely unknown:

- Over the course of time the importance of recreation values increases due to the increasing scarcity in supply and the in tendency positive income elasticity of demand. Their monetarisation only became possible with the development of the travel cost method in the 1960s.[1]
- Only in 1967 were existence values identified as a potential benefit of natural assets. They were altogether unknown before that date (Krutilla 1967).
- The possibility of putting the diversity of species to productive use, e.g. for the industrial development of pharmaceutical products, became known to a wider public only at the end of the 1970s (Myers 1979; Prescott-Allen and Prescott-Allen 1986).[2]
- The ecological structure and functions[3] of biodiversity within the ecosphere and the earth's major element cycles (e.g. of carbon, nitrogen and sulphur) and their resulting relevance for man's existence were not systematically discovered by natural sciences before the early 1970s (Mooney and Ehrlich 1997). In economic valuation studies they have only recently and sparsely found consideration. They are at the centre of the following analysis.

The consideration of ecological structure and functions of biodiversity in its total economic value is an attempt to apply economic thinking to the entire range of biodiversity services. This essay, therefore, attempts to analyse the economic relevance of the ecological structure and functions of biodiversity. For this purpose, these structures and functions, which are described in great terminological divergence throughout economic literature, shall be discussed and structured. The main focus of the analysis is on *specific non-substitutabilities resp. complementary relationships* on different levels of biodiversity and their implications for the total economic value. Methods for the monetary valuation are taken into consideration only in so far as it is necessary for the analysis of these relationships (section 3). Based on this, conclusions are drawn for decision making in environmental policy in section 4. Beforehand, a short overview of the total economic value of biodiversity is given (section 2).

2. The Components of the Total Economic Value of Biodiversity in an Overview

Within the literature on economic valuation the value of natural assets for analytical purposes is defined by the components of the so-called total economic value.[4] The identification of the total value shows *which* values in the economic sense are generated by natural assets. The adjective “economic” implies, that those services of biodiversity are to be considered that possess a value from an anthropocentric, instrumental and utilitarian point of view. Economic relevance is, therefore, only gained by segments of the environment which touch on the welfare position of at least one individual directly as an argument of utility functions and/or via the production process. The following analysis remains restricted to this perspective and so ignores other arguments for the protection of biodiversity, like intrinsic values of nature.

Since the question of the protection of biodiversity generally requires marginal decisions the following analysis of the integration of the ecological importance of biodiversity into the total economic value will mainly focus on the marginal value of biodiversity (as in contrast to a total valuation of all the services of biodiversity, e.g. for supplementing or correcting national income accounts or for defining the total value of a protected area) (Barbier 1995, p. 154).

In literature there is a wide range of systematisations of the total economic value of the environment. They differ both on the services included and in the definitory and terminological comprehension. From a welfare economics point of view this diversity can, in a first step, be traced back to two fundamental categories of values: production and individual values (Freeman 1986).

Production values of biodiversity are an argument of production and cost functions of market-allocated goods. Qualitative or quantitative changes of this production input affect individual welfare via changing prices of goods or other inputs. Besides the use of ecosystems for agricultural and forestry production it is mainly the pool of genes and species that allows productive uses. A diverse pool of genes and species allows, by falling back on wild species, the improvement of cultivated plants in terms of yield increases and to preserve their resistance against diseases. It serves as a basis for the development of pharmaceutical products and is also increasingly put to productive use in the field of biotechnology (Hampicke 1991; Pearce and Moran 1994; Swanson 1996). As opposed to the production values, *individual values* of biodiversity are a direct argument of individual utility functions. For example, recreational values or aesthetic values of biodiversity may be subsumed here and also those benefits called passive use value (NOAA 1993), non-use value (see for example Pearce 1993) or existence value (Krutilla 1967).[5] The latter are based on the preferences of individuals for the protection of natural assets regardless of whether this asset now or in the future generates a productive, recreational or aesthetic use for the evaluating individuals themselves. Accordingly, existence values result merely from the information that a resource

Table I. Terminological considerations of the ecological structure and functions of biodiversity in the total economic value.

Terminology	References
Inherent value	Farnworth et al. 1981
Contributory value	Norton 1986
Indirect use value	Barbier 1989 a, b; 1994; Aylward and Barbier 1992; Pearce, Moran and Fripp 1992; Wood 1997
Primary value	Turner/Pearce 1993; Gren et al. 1994
Infrastructure value	Perrings et. al. 1995b; Costanza et al. 1997

Own collection

exists.[6] In this sense existence values are nowadays widely accepted in literature as a component of the total economic value (Kopp 1992; Smith 1993).

The ecological importance of biodiversity as a component of the total economic value. Some authors, who mostly belong to the research field of Ecological Economics, supplement production and individual values with another category, which is to consider the ecological importance of biodiversity. This value component describes the ecological-functional role of biodiversity in natural systems. Generally, those services of the biological resources that stabilise the ecological system and thereby perform a protective and supportive function for the economic system can be included here (Barbier 1989a, p. 96). A look at the recently developed approaches for considering the ecological functions of biodiversity displays a vast terminological diversity, however (see Table I).

- The "inherent value" describes those services "without which there would not be the goods and services that are provided by the system" (Farnworth et al. 1981, p. 281).
- The "contributory value" focuses on the economic-ecological importance of species diversity: "Even species that have been unsuccessfully examined for human uses are still useful, because they contribute to increases in diversity which, in turn, contribute to the generation of more species" (Norton 1986, p. 117).
- "Indirect use values" are described as "support and protection provided to economic activity ... by regulatory environmental services" (Barbier 1994, p. 156).[7]
- The "primary value" is defined as: "The existence of the wetland (or generally: ecosystem; the author) structure (all its components, their interrelationships and the interrelationships with the abiotic environment) is prior to the range of function/service values" (Turner and Pearce 1993, p. 186).

- “Infrastructure values” of ecosystems result from a “minimum level of ecosystem ‘infrastructure’ . . . as a contributor to its total value” (Costanza et al. 1997, p. 254).

For the question of implementing the ecological relevance of biodiversity into the total economic value it is important to note that these categories of values not only differ terminologically, but that services of biodiversity are considered that are different in content. These differences can be traced to *specific complementary relationships*. Three patterns can be distinguished:

- A complementary relationship of species in their habitat (3.1).
- A complementary relationship of biotic and abiotic components of ecosystem structures (3.2).
- The complementary relationship of the ecological functions of ecosystems and the contribution of the services of ecosystems to human welfare (3.3).

In the following chapter these relationships are discussed and their implications for the total economic value, especially for marginal consideration, are depicted.

3. Structure and Functions of Ecosystems as an Element of Total Economic Value

3.1. COMPLEMENTARY RELATIONSHIPS OF SPECIES WITHIN THE INTERDEPENDENCIES OF SPECIES DIVERSITY

3.1.1. *Marginal Production and Individual Values of Species*

In recent years a number of studies have been conducted on the quantification of the *production value* of species diversity. These studies mostly focus on one segment of the production value, namely the use of species for the development of pharmaceutical products.[8] In this, the probability that a species will lead to the development of a marketable pharmaceutical product is multiplied with the value of such an invention. For this latter value different valuation approaches are used. The valuations are comparatively low when the value of a species is deduced from the price that the pharmaceutical industry pays for the prospecting of the species. The valuations are, on the other hand, decisively higher when they are based on the market value of the pharmaceutical products developed on the basis of the species, or beyond that on the positive effects for human health induced by the pharmaceutical products (Pearce and Moran 1994). This in the end accounts for the major differences in the determined production values; Simpson/Sedjo/Reid (1996) report in a review of existing studies values between 44 $ and 23.7 million $ per species. This proves a significant production value of species and gives quantitative evidence for the economic relevance of this value category.

Nevertheless, the studies have only limited relevance for the valuation of the marginal production value of species. The reason for this is, that for the valuation of the marginal production value of species diversity the possible substitutability of species, or more precisely: the substitutability of the production value of species must be considered. This is an aspect which is not taken into account in the above-mentioned studies on the production value of species diversity (Simpson, Sedjo and Reid 1996). Simpson/Sedjo/Reid argue that there are great redundancies concerning the production value within the species pool, which stem from the fact, that different species can each produce identical chemical compounds and that different chemical compounds of different species may have similar characteristics concerning their use for the development of pharmaceutical products. Considering these possibilities for substitution, the value of a species depends not only on the probability that the prospection of a species will lead to the development of a marketable product, but also on the probability that all other species cannot contribute to the development of this product. Starting off from this assumption Simpson/Sejdo (1996) show that the marginal value of a species depends on the number of species available for bio-prospecting, the costs of bio-prospecting, the probability that a tested species will lead to the development of a pharmaceutical product and the financial reward from new pharmaceutical products. It is obvious, that the value of a marginal species is higher, the smaller the number of available species is, the lower the costs of bio-prospecting are and the higher the probability of developing a new product and this product's market revenue is. On the basis of plausible assumptions for these parameters Simpson/Sedjo come to very low valuations. For an assumed number of 1,000,000 species for example a marginal value of less than 1/10 Cent results (Simpson and Sedjo 1996, p. 23).[9] Following this, the marginal production value of species diversity is negligible.

Likewise there exist a range of valuation studies on the *individual value* of single species. They were mainly conducted in the first half of the 1980s in the USA, with both recreational and existence values considered by contingent valuation studies. In these studies a significant willingness to pay for popular animal species could be ascertained. For the protection of grizzly bears, for example, existence values of 15 to 24 dollars on average were offered (Brookshire, Eubanks and Randall 1983), for the protection of bald eagles in Wisconsin a total value of, on average, 10 to 75 dollars. But also for obscure species on which the test persons were only informed during the questioning a positive willingness to pay could be discovered (Boyle and Bishop 1987).[10]

The significance of these studies must be considered rather sceptically, however, for different reasons. It can be assumed that they tend to lead to an exaggeration of the individual value of species. Concerning the validity of the results Freeman (1993) remarks that the studies were conducted during a time when the contingent valuation method experienced its first practical application in environmental economics and these studies, therefore, do not conform to modern standards for guaranteeing the validity and reliability of contingent valuation studies (see for example NOAA 1993). With this background it is questionable whether the determined monetary values can be understood as a willingness to pay in the economic sense and actually mirror the relative weight that individuals attribute to these resources within their limited budget or whether these payments should not rather be considered an indicator for the general wish for the protection of nature (Freeman 1993b, p. 297; Brown and Shogren 1998, p. 12). The studies would then be subject to distortion by the so-called “Warm Glow”-effect of contingent valuations, that means they would have to be interpreted like contributions for charities and not as a willingness to pay for a specific natural asset (Kahneman and Knetsch 1992; Degenhart and Gronemann 1998).[11]

More important for the analysis of the relationship of species is the question as to whether the possibility of undamaged substitutes for the endangered species has been adequately considered in the design of the contingent valuation surveys (see generally to this problem of contingent valuations Cropper and Oates 1992, p. 720; Bishop and Welsh 1992, p. 411; NOAA 1993, p. 4604). The articulated willingness to pay is not plausible, if the individuals did not include the multitude of other species that could be possible substitutes for an endangered species in their valuation. The substitutability of species, or more precisely: of the individual value of species can in the end only be determined empirically. However, some speculations may be permissible: There may be species that possess a unique identity. That means, that by deselecting its use a specific benefit would be lost, i.e. a benefit that could not be provided in the same way by other species (Pearce, Markandya and Barbier 1989; Beirat Umweltökonomische Gesamtrechnung 1995). Generally speaking it can be assumed that this holds true for popular animal (mostly at the end of food chains, so-called “charismatic megafauna” (Brown and Shogren 1998, p. 13)) and plant species (mostly higher plants). These are then limited in their substitutability and a significant positive willingness to pay seems plausible. Unknown plants on the other hand or even micro-organisms must rather be viewed as substitutes concerning their recreational or existence value. Although it is not altogether impossible that even for these species, given complete information, a positive willingness to pay could be articulated (Bishop and Welsh 1992; Kopp 1992), the multitude of possible substitutes supports the assumption that their marginal value, analogous to the production value, tends to be low.

As an intermediate result it is to be noted, that the *individual and production value* of marginal species is comparatively low due to great substitutabilities, unless species are unique in this regard.

3.1.2. *On the Ecological Importance of Species Diversity*

From an ecological point of view the importance of the diversity of species is that the co-existence of species within a habitat is defined by complex relationships of interaction and interdependence, based for example on food chains. This means that the survival of one species depends on the existence of other species, whose existence in turn depends on yet other species etc. Norton tried to outline the economic relevance of these ecological interactions in the concept of the “contributory value”.

Contrary to the conventional economic way of thinking Norton does not support an isolated consideration of species (restricted to the production and individual value of single species), since biodiversity depends on their complexity, i.e. a large number of highly specialised elements and the interdependencies between these elements. Increases in diversity can, in this context, induce further increases in diversity, in the same way as the loss of a species can cause further reductions due to the existing interrelationships. The “contributory value” focuses on the fact that species can only survive within the web of interactive relationships and, therefore, each species contributes to the survival of other species. In this it is especially the diversity of plant species and micro-organisms often unknown to the public, that are “intermediate goods” (Crocker and Tschirhart 1992) for the productive and individual use of animals and plants (Ehrlich and Ehrlich 1992). Wood (1997) illustrates the importance of the “contributory value” in the example of the productive use of wild species for the preservation of the resistance of cultivated plants: because of their limited genetic diversity cultivated plants, especially those in single-crop farming, can only perform minor adaptations to changing environmental conditions. Wild species, on the other hand, possess a higher adaptability to changing environmental conditions, because of their greater genetic diversity (which is a result of a higher species diversity). The diversity of wild species is a prerequisite for their evolutionary development. This in turn forms the basis for preserving or improving the resistance of cultivated plants against diseases or pests, by cross-breeding with wild species.[12] Therefore, those elements of biodiversity which currently and in the future have no productive or individual use value must still be considered useful, for they contribute to biodiversity and thereby indirectly to the direct use values. Following these considerations even the loss of a not directly useful species can be relevant from a utilitarian point of view, because dependant, possibly directly beneficial species can be lost as well, by cascading effects (Norton 1986). In this, it is assumed that every species is unique in the sense that it takes a very specific position in ecological systems (Mooney et al. 1995). In economic terms, one therefore has to assume extensive complementary relationships of species within their habitat. Knowledge of these ecological interdependencies is, however, highly incomplete (Perrings, Folke and Mäler 1992).

3.1.3. *Implications for the Total Economic Value of Species Diversity*

The “contributory value” exposes the incompleteness of economic valuations of single species, if their value remains restricted to the productive and individual benefits. The importance of a total economic value which, besides the production and individual values also includes the ecological relevance of single species for species diversity, is therefore to counteract the resulting systematic undervaluation and point to the limits of the substitutability of species.

The different interpretations of the substitutability of species in the conventional economic view on the one hand and the ecological view on the other hand can be illustrated in a simple formula.[13] In ecology it is assumed that every species possesses a vector of n characteristics $[G_1 \ldots G_n]$, which is again the starting point of a vector of f functions (or services) $[F_1 \ldots F_f]$. These are a number of ecological functions (F_e), but within the total economic value recreational values (F_r), existence values (F_{ex}) or production values (F_{pro}) could as well be put forward. The limited substitutability of species results, from an ecological point of view, from the fact, that the combinations and the features of the characteristics of each species $[G_1 \ldots G_n]$ are unique and that these characteristics correspondingly generate a specific vector of functions $[F_1 \ldots F_f]$.[14] If a species was to substitute, it would have to possess all the characteristics $[G_1 \ldots G_n]$ of the species to be substituted, because only then could it fulfil the entire vector of functions. From this it follows logically that the species would have to be identical. Since every species performs very specific duties within ecological systems, a complete substitution of a species is generally impossible.

Substitutabilities arise only when the vector of functions, which originally comprises a great many components, is reduced. This is exactly the basis for the substitutability of the production or individual values of species. To assume that a larger number of species that produce a specific chemical exist, implies a reduction of the relevant functions of the species. The same holds true if the relevance of species remains restricted to individual values. In both cases only selected functions of the species are of interest (f [economic], with f [economic] $<$ f [ecological]). If the vector of relevant functions is narrowed in this way, the functions can correspondingly be fulfilled by a limited number of characteristics (n [economic] $<$ n [ecological]). By considering for example only (F_r) species become substitutable, because different species are marked by comparable recreational values. However, at the same time, the substitutability becomes limited when the recreational value of a popular species – and thereby, in the end, its specific characteristics – is considered unique. Species, therefore, can only be considered substitutable if one diverges from the assumption that every species possesses unique characteristics. In the conventional economic approach this is achieved by attempting to attribute single species with clearly definable and compartmentalised benefits, which are then considered interchangeable, i.e. substitutable, both amongst each other and with respect to man made goods. Norgaard criticised this method as "atomistic-mechanistic" (Norgaard 1985, p. 382) and unsuitable in an ecological context.

It generally shows that by restricting the functions and, thereby, the necessary characteristics a positive number of species can be determined that can perform a limited number of functions. These species are substitutable amongst each other. The substitutabilities become the greater, the fewer functions of the species are included in the consideration. The number of substitutive species is, therefore, of reversed proportionality to the number of functions of the species that are considered relevant. If all functions (f [ecological]) are considered, the number of substitutable species is zero.[15]

The conclusion that follows from this is, that the economic costs of the loss of a species are higher than the lost production and individual values. The problem of this potential valuation gap for the protection of species diversity can be illustrated at two examples:

- Model-theoretical studies by Simpson/Sedjo/Reid (1996) as well as empirical studies by Barbier/Aylward (1996) concerning the benefits and costs of pharmaceutical prospecting in the example of Costa Rica's National Biodiversity Institute (INBio) come to the conclusion, that it is unlikely that the private value of species for use in pharmaceutical research, i.e. the *production value*, will cover the opportunity costs of biodiversity protection.
- Recently Metrick/Weitzman (1998) have shown that the spending for the preservation of species under the US-Endangerd Species Act is significantly influenced by the size of the species: a ten percent increase in length leads to an 8.6 percent increase in spending. The physical characteristics of charismatic megafauna, from an economic point of view the *individual values* attributed to these species, effect the resource allocation much more than scientific characteristics like endangerment, recovery potential or the ecological function of species (Brown/Shogren 1998, p. 15).

So, consideration of production or individual values of single species alone leads to an inefficient resource allocation. Therefore, these values must be increased by the "contributory value". Unlike the production and individual values, however, there are difficulties in quantifying this value component in the economic sense, i.e. by its contribution to individual welfare. This would only be possible if complete knowledge about the ecological interrelations existed and the contributions of single species to directly beneficial species could be determined. The relevance of the "contributory value" lies therefore more in a qualitative evaluation. It reveals the complementary relationship of species and highlights an ecological phenomenon which tends to be neglected due to the "atomistic-mechanistic" method of the economic valuation approach.

The analysis has so far been restricted to the level of species diversity. In the following, the consideration expands to the importance of species as *carriers of ecological functions in ecosystems*.

3.2. COMPLEMENTARY RELATIONSHIPS OF ECOLOGICAL STRUCTURAL COMPONENTS OF ECOSYSTEMS

3.2.1. *On the Ecological Importance of Ecological Structural Components*

Beyond their function within the interdependencies of species diversity, the ecological importance of species results from their role as carriers of ecological functions in the ecosystem. *Species* and their interconnections *are*, in this context, *biotic elements of the ecosystem structure*, which, in combination with abiotic elements (and under input of energy), provide the basis of ecological functions of ecosystems at specific states of the system. These connections and the outstanding importance of species can be illustrated by the example of soil. The basis for the ecological soil functions are the different components of the soil structure (Wild 1993):

- As *abiotic components* on the one hand mineral substances (inorganic matter), on the other humus, i.e. the dead plant and animal substances along with their transformation products (organic matter) and finally the soil texture, which results from the spatial location of these variably shaped and sized organic and inorganic components. It is filled with soil water and soil gases.
- As *biotic components* the soil organisms and the system of their interrelations, generated by food webs and the resulting relationships of competition and co-operation.

Complex causal relationships exist between the structural components: The abiotic structural components determine the physical-chemical properties of soils, which can be quantified by attributes like clay content, pH, nutrient concentration, particle size, content of organic substance etc. The diversity of soil organisms depends, on the one hand, on the particular soil properties, but, on the other hand, the soil organisms also influence the abiotic structure components and, therefore, the soil properties. The soil properties again function as regulators for the chemical, physical and biological processes within the soil, such as ab- and desorption, the mechanical containment or the abiotic/biotic decomposition and restructuring of substances (Fränzle et al. 1993). The abiotic and biotic components of soil texture and the processes taking place between them are therefore the basis for the various ecological functions of the soil: the buffering and moderation of the hydrological cycle, the renewal of biological productivity, the regulation of global major element cycles, like the carbon, nitrogen and sulphur cycles etc. (Rat von Sachverständigen für Umweltfragen 1987; Daily, Matson and Vitousek 1997). The supreme importance of species diversity for the ecosystem functions will be evident by the following: 90 percent of the matter transformation processes in soil are conducted by the soil fauna and flora (Fränzle et al. 1993, p. 5).

The biotic elements are, therefore, attributed a central importance for the maintenance of the full range of ecosystem services. However, in ecology it is also assumed that not all species are of the same relevance in this context. It is on the opposite evident, that the loss of certain species has only minimal effect on ecosystem functions, while the loss of others can carry severe consequences. The reason for this is that the functionality of ecosystems depends on a limited number of biotic and physical processes. These processes are again directed by different groups of species with complementary functions. These species are called “keystone species” (Schulze and Mooney 1994; Mooney et al. 1995; Perrings 1995).

These findings could lead to the conclusion, that an ecosystem contained a larger number of ecologically redundant species. This assumption is, however, short-sighted for the direction of critical processes is under different environmental conditions conducted by different species. The importance of species diversity in ecosystems is, therefore, based on the fact that species that seem redundant under certain environmental conditions can become “keystone species” under different conditions (Schulze and Mooney 1994; Perrings 1995; Perrings 1998). Thus, in the case of a loss of a species certain ecological functions of this species can be taken over by other species, which take the place of the extinct species (Tilman 1997). This balancing process within and between ecosystems remains intact as long as specific species can substitute each other. It ends therefore, when all the species that could fill a certain niche under changing conditions are extinct or a balancing has become impossible, for example due to the spatial separation of ecosystems. If no substitution by another species is possible, maintenance of a function is only guaranteed if the species, through evolutionary process, mutation and genetic alteration adapts to the new conditions in the habitat in the course of the changing generations. This means that a loss of species in an ecosystem or alterations in the mix of species make the ecosystem more susceptible to exogenous shocks (natural shocks like change in climate or man-made shocks like changes of relative prices) and rises the probability that the system changes discontinuously form one stable state to another, for example from one ecosystem type to another (Perrings and Walker 1995b; Perrings 1998).

The question, as discussed in ecology, of a possible critical level of species diversity, which is essential for the functionality of ecosystems, can, from an economic point of view, be interpreted as the search for substitutabilities: It is the attempt to reduce, from an ecological point of view, the relevant functions of species to a finite number. If the function of ecosystems is guaranteed by specific critical processes and these processes can, under changing environmental conditions, be conducted by a limited number of species, this also implies a limiting of the characteristics of species which are considered relevant. Then, and only then, could they also be considered substitutable with regard to their ecological functions. Still, this “substitutability” cannot, in the sense of the economic substitution paradigm, be equated with specific species being “dispensable” or “non-essential”: Knowledge of the functional substitutability of ecological structures is highly imperfect, so that a reliable definition of a critical (essential) ecosystem structure is impossible (Mooney et al. 1995; Bingham et al. 1995). Those species which are considered seemingly less relevant may possibly conduct subtle, unforeseeable functions in the ecological network of interrelations (Holling et al. 1995). For the ecological functionality of ecosystems it is therefore crucial “that there are species waiting in the wings” (Dasgupta 1995, p. ix). As opposed to inputs of man made production processes the input of species to ecological productions cannot be continuously adapted to changing conditions. In particular, allocation decisions can usually not be reversed, because the choice against the protection of species is marked by irreversibility.

Against this background there is a high level of agreement among ecologists on the fact that, “although there are many unanswered questions, . . . the ability of ecosystems to provide a sustainable flow of goods and services to humans is likely to be highly dependent on biodiversity” (Tilman 1997, p. 94).[16] It is assumed “that there are levels of biodiversity loss that can not be sustained without inducing catastrophic change/fundamental reorganisation in all ecosystems” (Perrings et al. 1995a, p. 5). With the reduction of species the possibilities for the substitutions necessary for the preservation of functions under changing environmental conditions diminish. Conclusively the probability of a transition of the ecosystem into different states rises (Perrings 1998). This is the central importance of *functional diversity* in ecosystems.

3.2.2. *Implications for the Total Economic Value*

Considering their relevance for the functionality of ecosystems, species have an entirely different function compared to their productive and individual uses. This can be elucidated by an economic interpretation of the structures and functions of ecosystems. In analogy to the terms used in economics the structural components of ecosystems can be identified as stocks, while the functions are flows. Taking the analogy between ecological and economic analysis one step further, the structural components obviously function as assets, whose combinations result in certain functions of ecosystems (see also Aylward and Barbier 1992; Fromm 1997). Single species must therefore be considered an *input* for the production of ecological functions of ecosystems (Perrings 1995), as opposed to their production or individual values, which are an *output* of biodiversity.

Analogous to the discussion of the interdependencies within species diversity, the complementary relationships of ecological structure components likewise displays an ecological relevance of species that goes far beyond their production and individual values. However, while the anthropocentric value of species diversity primarily results from their contribution to the preservation of single, productively and/or individually used species, the economic character of ecological structure components is not yet established by this line of arguments. In the sense of the above mentioned “*primary value*” of ecosystems it can so far only be noted, that the structure components (“all its components, their interrelationships and the interrelationships with the abiotic environment”) (Turner and Pearce 1993, p. 186) are preliminary to the ecological functions. At the same time, the substitutabilities among the structural components hint to the fact that – in the sense of the "*infrastructure value*" – a certain critical ecosystem structure is not to be undercut, if the functionality is to be guaranteed (Costanza et al. 1997, p. 254). However, the economic relevance of the structural components can in the end only be deduced from the economic value of the ecological functions, which again results from the fact, that they are inputs for human used services of ecosystems.

3.3. COMPLEMENTARY RELATIONSHIP OF ECOLOGICAL FUNCTIONS OF ECOSYSTEMS AND ECONOMIC SERVICES

3.3.1. *On the Connection Between Ecological Functions of Ecosystems and Their Contribution to Human Welfare*

The complementary relationship of ecological functions of ecosystems and their contribution to human welfare are subject of most of the value categories for defining the ecological relevance of biodiversity, mentioned in chapter 2. Early thoughts on these complementary relationships in literature can be found in an analysis on the values of ecosystems conducted by Farnworth et al. (1981). The aim of these thoughts was to identify the manifold values of ecosystems and to structure them according to their proxinity to market allocation. This was explicitly meant to point out the economic relevance of non-market ecological functions and draw attention to the necessity of considering them in decision making. Here, Farnworth et al. identify a value category which includes the basic biogeochemical cycles of nature, the natural processes of selection and evolution and the life-support functions of ecosystems in an all encompassing perspective. These ecological services are, according to Farnworth et al., inherently connected to the integrity of natural systems and embody the totality of structure and functioning of the system. Analytically innovative in this was the highlighting of specific complementary relationships. According to Farnworth et al. ecological systems possess "*inherent values*", "as values that support all other values" (Farnworth et al. 1981, p. 281). Thereby, Farnworth et al. for the first time draw attention to ecological functions, which in the first place provide the basis for those services then directly used and valued by individuals. They are, therefore, in a complementary relationship. For a long time, however, these thoughts were paid little attention in literature. Only with the recent differentiation by Gren et al., between primary and secondary values of ecological systems, was the approach of Farnworth et al. – though not explicitly – taken up and further developed.

Gren et al. divide the services of an ecosystem into three categories (Gren et al. 1994; see also Turner and Pearce 1993; Bateman and Turner 1993):

1. Services for the development and maintenance of the functionality of the system itself.
2. Services for other ecosystems.
3. Services for human uses.

The services first mentioned describe the self-organising capacity of the system. They include both the dynamic evolutionary processes of the system and the capability of the system, to absorb external disturbances (stability). This "*primary value*" is the source for the other, so-called exported or secondary values of the ecosystem. The secondary values therefore depend on the continuous preservation of the "ecosystem health".[17]

However, the differentiation between the primary and secondary values, as conducted by Gren et al., is imprecise so far as the services for other ecosystems are also considered secondary values. If these services are understood as a contribution to the preservation of ecological functionality within interconnected ecosystems, they must rather be integrated within the primary value. A further argument for this is that these services, other than the third category, are not directly used by man, but contribute to the preservation of functions of other ecosystems, which in turn generate secondary use values. It therefore seems suggestive to assign both the first and the second service category to the primary value. The total value of an ecosystem, according to Gren et al., is then made up of the primary and secondary values. Between these values a complementary relationship exists.

The contribution of the concept of the so-called "*indirect use values*" to the clarification between ecological functions and human welfare lies predominantly in the differentiation between the *support function* of ecological systems on the one, and their *protection function* on the other hand (Aylward and Barbier 1992). The support function is linked to the inherent and the primary value of ecosystems. However, Aylward/Barbier limit the complementary relationships to the production values of nature. It seems logically stringent, though, to also assume this connection for the individual values: The use of an ecosystem for recreational purposes depends, just like the productive use, on its functionality and therefore on the interdependence of the structural components. In addition to this Aylward/Barbier highlight the protection services of ecosystems. Ecological functions like the regulation of climatic processes, of the hydrological cycle or the filtration, buffering and transformation of human induced environmental pollution in the end appear as avoided material damages and avoided health damages. Therefore their economic value rises from their ability to protect assets like man made capital, human capital and natural capital itself (for example agro-economic ecosystems) against natural and man-made shocks or disturbances. The economic relevance of ecological structures and functions in this approach apparently goes far beyond the input function for the production and individual values of ecosystems.

3.3.2. *Implications for the Total Economic Value*

In the previous analysis, the economic relevance of structures and functions of ecosystems and the resulting necessity for including them in the total economic value was demonstrated: It follows from their input functions for the production and individual values and the protection services for human capital, man made capital and natural capital (as output). The conventional total economic value,

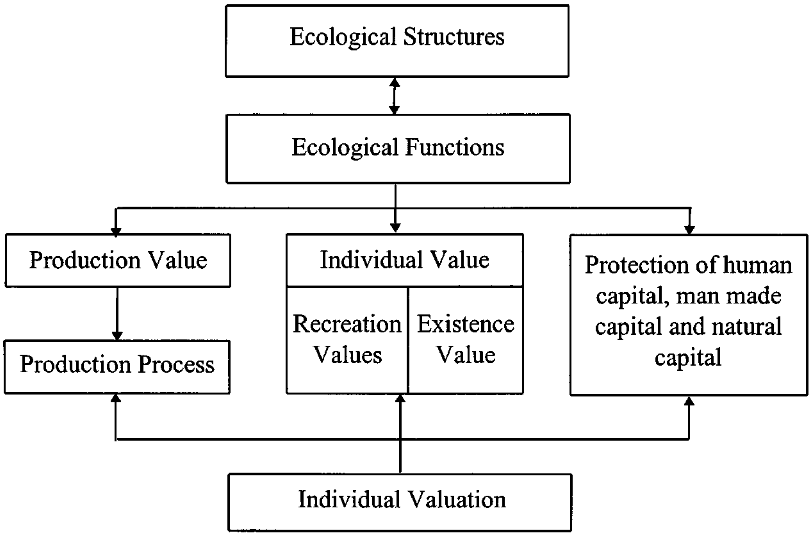

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Figure 1. Total economic value of biodiversity.

restricted to the individual and production values of biodiversity, contains no value component that gives credit to these connections and, therefore, remains incomplete.[18] Figure 1 shows an accordingly supplemented total economic value of biodiversity.[19]

Terminologically, the input and output functions of biodiversity are considered both within the differentiation between "direct and indirect use values" as well as in the differentiation between "primary and secondary values". The main difference between both concepts is that the "primary value" – as depicted in Figure 1 – explicitly includes the ecological structure components as carriers of ecological functions. Besides, the "primary/secondary" concept more strongly underlines the complementary character between ecological structures and functions, on the one hand, and human use of services on the other hand. In contrast, the term pair "direct/indirect" illustrates the different relationships of the value components to individual preferences.[20]

From an economic point of view, the rather trivial conclusion that results from the discussed complementary relationships is that the value of biodiversity for man in a total consideration tends towards the infinite (Costanza 1997, p. 255). Biodiversity as the basis for human life is, as a whole, indispensable under realistic technological and economic conditions and in this sense essential (Hampicke 1992, p. 318). Nevertheless, marginal decisions can be imagined and are reasonable: Specific ecological services can be impaired, although at the price of losing specific recreational values (e.g. forest damage), the qualitative and quantitative reduction of agricultural and forestry yields, the suffering of material damages (e.g. through floods) or health damages (e.g. through polluted foodstuff).

The analysis above has shown that human use of specific ecosystem services and respectively the infringement of these services can (at least theoretically) be traced back, each to specific ecological structures and functions. Against this background it could be asked, whether the stability of ecosystems *per se* can be ascribed an economic value. This would mean, that every change from one ecosystem type to the other, or, respectively, every function impairment of ecosystems would a priori be considered disadvantageous, by assuming a negative cost-benefit-saldo.

However, the central problem of marginal decisions is the fact that knowledge about the consequences of infringements on biodiversity is highly incomplete. Situations of decision-making are often, beyond risk and uncertainty, marked by ignorance about the consequences of the action. This ignorance in the end stems from unbridgeable gaps in our knowledge about ecosystem interrelations and regularities. This in turn results from the fact that it is practically impossible to grasp the complexity of an ecosystem in its entirety. Since the underlying processes that direct the development of ecosystems are unknown, changes of the system cannot be prognosticated (Faber/Manstetten/Proops 1996). Therefore, it is usually not known, by which structures and functions certain services of ecosystems are generated (Bingham et al. 1995). This holds especially true for the protection services of biodiversity. Their benefits are often only discovered, once they have been disturbed or lost (Daily 1997; Mooney et al. 1995).

The example of the nitrate decomposition capacity of soil can be used to illustrate the practical relevance of this problem. The microbiotically catalysed decomposition of nitrate in soil is of relevance especially with regard to the agricultural nitrogen fertilisation. It can lead to excessive nitrate in the soil, mostly due to the overly extensive input of liquid manure. The nitrate is then shifted to deeper layers within the soil, where it is denitrified, i.e. decomposed, under consumption of organic matter. The soil, therefore, has an important function for the protection of ground water, since nitrate pollution is reduced or avoided. The denitrification processes thereby contributes to the preservation of the drinking-water supply. In the past it was assumed that a steady state existed between the decomposition of nitrate and the continuous input into soils. The soil was assumed to be able to denitrify a certain amount of nitrate, with excessive nitrate inputs being passed on. From an economic point of view an optimum level of emissions could then be theoretically deduced from a comparison between the benefits of nitrate input, as increased agricultural yields, and the costs of nitrate input, as increased costs of drinking-water processing. However, more recent scientific findings seem to point out that the store of organic matter necessary for the decomposition of nitrate is exhaustible. The continuous input, therefore, leads to a reduction of the transformation capacities of the soil. The nitrogen decomposition capacity of the soil is therefore declining towards a minimum, due to the constant over-fertilisation (Umweltbundesamt 1990; Strebel, Böttcher and Duynisveld 1993). This would result in nitrates being passed on to the ground water unhindered. For the deduction of an economic optimum in the past, therefore, much higher damage costs (e.g. higher re-processing costs) would have had to be assumed. Ignorance regarding the functioning of the ecosystem “soil” therefore led to an underestimation of the costs of nitrate emissions at the outset, which now probably has dramatic consequences.

So far it can be noted, that a basically complementary relationship between ecological structures and functions on the one hand and the human use of services of biodiversity on the other hand can be assumed. However, there is very little knowledge about the specific complementary relationships. It follows, that reductions of biodiversity can lead to unpredictable effects on human welfare.

Following a system-theoretical interpretation of the ecosphere, which goes back to Norton/Ulanowicz (1992), this danger of unpredictable and probably irreversible welfare losses can be more thoroughly qualified. According to Norton/Ulanowicz the ecosphere is marked by a hierarchy of embedded subsystems on different scales. The larger an ecosystem the more subsystems it contains and, therefore, the greater is its complexity. It is assumed, that smaller systems (i.e. subsystems on a lower hierarchical level) react faster to external influences than higher systems, i.e. have a lower resistance. However, they also return to their original state faster, their resilience is, therefore, higher. Large systems accordingly possess a greater inertia and changes can less easily be reversed. From an ecological point of view, therefore, large-scale interferences must be considered more problematic than interferences into biodiversity that are restricted to a small scale. In an economic interpretation this points towards the conclusion that in the case of small-scale distrubances the possibility for substitutions within the ecosphere exist, whereas with large-scale interferences there is a higher danger of damaging potential substitutes as well. This approach again attempts to discover potential substitutabilities within biodiversity, in order to deduce priorities for protection.

It remains to be noted that it is basically the *combination of complementary relationships and ignorance* (or at least wide gaps of knowledge) that allows the deduction of a specific value for the stability of ecosystems, also from an economic point of view. The preservation of the resilience of ecosystems protects man from incalculable welfare losses. If one also follows the thesis that the resistance and resilience of ecosystems depends on the functional diversity, then the economic importance of the diversity of ecological structural components lies mainly in its function to minimise the risks of exogenous shocks: “The value of biodiversity includes a significant insurance component” (Perrings 1995, p. 842; also Tilman 1997, p. 107).

4. Conclusions for Decision-Making in Environmental Policy

The preceding arguments point out parallels between man made assets and natural assets (or biodiversity as a segment of natural assets). They can be viewed as an important starting point for decision making in environmental policy: There are man made *assets* intended *for direct individual use* (like houses, cars), others, which form the basis for productive uses (*productive assets*) and finally specific *security assets* which serve to secure these assets as well as individual welfare positions (for example private liability insurance and social security systems). Analogous to this, biodiversity must also be attributed a function as security asset, beyond the productive and individual uses. Although both the increasing productive use of biodiversity and their rising market values, and the improved feasibility of monetarising the individual values give important allocative incentives and reasons for the protection of biodiversity, an evaluation restricted to these value components remains nevertheless incomplete. The reasons for this are, that

- on the one hand, the probably most important, essential services of biodiversity for man result from its ecological protection functions and
- on the other hand, the mentioned benefits depend entirely on the ecological functionality of ecosystems.

Against this background Perrings/Walker (1995b, p. 204) stress that public and scientific attention concerning biodiversity preservation should not only be focused on global extinction of single species but also on the consequences of alterations of the mix of species in ecosystems. The potential costs of changes of ecosystem types for the current and future generations might be higher then the costs of a loss of single species.

What conclusions can be drawn for marginal decisions? As far as the total economic value would remain limited to productive and individual values the economic rationality of corresponding decisions could be checked by the cost-benefit rule, unless it would be restricted by irreversibilities as in the case of global extinctions. However, if beyond this the ecological functions of biodiversity are also considered, the application of the conventional cost-benefit-analysis is limited. This results from the specific characteristics of these services, which can be summed up, as discussed, in the key words “complementary relationships” and “ignorance”. Cost-benefit analysis of biodiversity accordingly shows a valuation gap, by which possibly essential services find no consideration in the economic weighing of choices. This problem is also given prominence in the discussed values of the ecological functions of biodiversity:

- Costanza/Farber/Maxwell (1989, p. 339) interpret the “contributory value (as) an estimate of the value individuals would place on environmental services if they were fully informed about the functioning of the environment on their behalf.” A lack of knowledge, therefore, causes too low willingness to pay.
- According to Farnworth et al. (1981, p. 281) the “inherent value” escapes individual valuation: “This concept . . . removes Man as the evaluator”.
- For the consideration of the primary and secondary values, different evaluation criteria must be used, according to Gren et al. The reference basis

for the "primary values" is the ecological functionality of the system; they are considered as a "non-preference, but still instrumental, type of value" (Turner and Pearce 1993, p. 186), that cannot be monetarily valued (Gren et al. 1994, p. 71). The secondary values on the other hand recur to the individual preferences and can, therefore, in principle be grasped by economic valuation methods.

- According to Costanza et al. (1997, p. 258) every monetarisation of ecosystems leads to an underestimation of its value, since the "infrastructure value" cannot be considered.

The question then is, how this valuation gap can be adequately considered within the framework of the economic valuation approach?

In valuation practice it is for example attempted to consider the ignorance about the damages of interferences into biodiversity, within the frame of cost-cost-benefit-analysis-analysis, by "generous" additions. An example for this is the cost-benefit-analysis on global warming by Nordhaus, in which one percent of the global gross national product is added to the costs of global warming, to cover all non-quantifiable damage, including ecological effects (Nordhaus 1991). However, such additions remain both more or less arbitrary and also – and this is the central point of criticism – the characteristics of "complementary relationships" and "ignorance" cannot be considered in this approach. Following the system-theoretical interpretation of biodiversity by Norton/Ulanowicz, these characteristics increase in importance with increasing geographical scale of infringements. The higher the probable nonsubstitutability of the ecological functions of biodiversity the more important it is to refer to a valuation mode which considers the characteristics of "ignorance" and "complementary relationships" in their interconnection (Toman 1994; Norton 1995).

The conclusion that can be drawn for the marginal valuation of biodiversity is not original, insofar as it was already formulated 40 years ago by Ciriacy-Wantrup, in the form of the so-called safe-minimum-standard (Ciriacy-Wantrup 1952 resp. 1968, p. 253). However, this decision rule has been gaining in acceptance in recent years (see for example Pearce 1991; Vatn and Bromley 1994; Beirat Umweltökonomische Gesamtrechnung 1995; Hampicke 1991a; Ready and Bishop 1991; Toman 1994; Turner, Adger and Brouwer 1998). The central idea of this rule is that on the one side uncertain (and in some cases irreversible) welfare losses should not be lightly accepted, while on the other side the ignorance about the consequences must not lead to the acceptance of unbearably high costs in the present. Decision criteria are, therefore, the amount of the current preservation costs. Interferences into biodiversity are to be avoided, as long as the opportunity costs for the current generation are not unacceptably high (Bishop 1978, p. 13).

As a result, this means, that conventional cost-benefit-analysis should be conducted, although their results must not be interpreted conventionally: If the cost-benefit-saldo in the concrete case speaks against the preservation of biodiversity, then this negative-saldo must be checked to see whether it is bearable for the current generation (see also Crowards 1998). Such an approach would guarantee that the costs for biodiversity conservation would be made explicit (rather than, as in the current practice, to continuously accept benefit losses implicitly) and a social decision concerning the limits of acceptable cost figures would have to be made (see also Beirat Umweltökonomische Gesamtrechnung 1995).

Notes

1. The travel cost method is considered the oldest valuation procedure for public goods. The origin of the method is traced back to thoughts by Hotelling in 1947. These were then further developed by Clawson/Knetsch in the 60s (see Clawson and Knetsch 1966).
2. The increasing economic relevance (in the private sector) of the productive uses of biodiversity becomes evident in the fact, that recently, although hesitantly, exclusive property rights on the biological resources of the tropical forests have been enforced and markets (and therefore market valuations) develop for the use of these resources (see for example Sedjo 1992; Lerch 1999).
3. The concept of ecological functions was introduced by Hueting (Hueting 1980; Hueting et al. 1998).
4. The concept of the "Total Economic Value" was established by Randall and Stoll 1983.
5. The reader might miss the option- and quasi-option-values in this list. Both are entangled in a several decades-long discussion, which led to the widely accepted result, that both values cannot be considered as separate value components of natural assets. The option value is merely an opportunity to grasp the influence of uncertainty on the individual valuation of natural assets. The quasi-option-value describes the welfare gain that can be realized by optimal environmental policy decisions in the face of uncertainty and irreversibility. It quasi equals the "value of efficient environmental policy". Therefore, the quasi-option-value is no value component of natural assets either (Freeman 1993a, p. 261; Shechter and Freeman 1994, p. 177).
6. On the motives for existence values see in detail Crowards 1997.
7. In literature, the term "indirect use values" is occasionally also used for the value category here called existence value (compare Steinhoff et al. (1987), p. 39). However, this classification has not prevailed.
8. See for example Principe 1989.
9. Estimates on the total number of existing species, of which only a fraction are known, however, range between 5 and 100 millions (Perrings et al. 1995a).
10. For an overview of other comparable studies see Aylward (1992), p. 43.
11. On further problems of contingent valuations of biodiversity, especially lexicographic preferences and individual information of biodiversity, see Hanley, Spash and Walker 1995.
12. On the importance of species diversity for productive benefits see also Swanson 1996.
13. For important hints on the following considerations I thank Dr. Rainer Brüggemann (Institute for Freshwater Ecology and Inland Fisheries, Berlin).
14. Strictly speaking, not only a specific vector of functions results from the characteristics, but also, as a further dimension, a species-specific shaping of each individual function.
15. However, it must generally be observed, that the classification of a species as non-substitutable with regard to its various functions and services is deduced from different evaluation criteria. With regard to the production and individual values non-substitutability results from the aggregation of individual willingness to pay. To classify a species as unique implicitly assumes, that the willingness to pay for its preservation is extremely high or even tend towards the infinite. The limited substitutability of the ecological functions of species, however, can only be deduced from their relevance for the functionality of ecosystems. In this respect they must be indispensable. This evaluation is not based on (economic) willingness to pay, though, but on ecological criteria.

16. These “unanswered questions” mainly concern the problem whether ecosystems are generally the more stabile the greater the species diversity is. In ecology many stability concepts exists. Two of the most important concepts are the resistance and the resilience of ecosystems. The resistance of an ecosystem is its ability to remain unchanged despite of an external disturbance. The resilience describes its ability to return to a referential state after a disturbance (Grimm/Schmidt/Wissel 1992, p. 136; see for a differing definition of “resilience” Holling et al. 1995, p. 50). Following Tilman current evidence shows that the resistance of ecosystems (but not necessarily on the level of communities) strongly depends on biodiversity whereas the connection between resilience and diversity is less clear (Tilman 1997, p. 109). There also exist ecosystems, like coastal and estuarine ecosystems, which possess a high resilience despite a small species diversity. In such cases disturbing environmental influences on the system are not buffered by a substitution of species, but by a high functional variability of individual species (Costanza/Kemp/Boynton 1995).
17. In this sense Wood (1997) differentiates between biodiversity and biological resources, the first being the source of the second. Freeman (1997) distinguishes between ecological ecosystem functions and economic ecosystem services.
18. The insufficient consideration of the functional value of natural assets within the total economic value is also pointed out by Vatn/Bromley (1994). In this context it must be stressed, that even the inclusion of the existence value does not resolve this deficit. Although the knowledge about the ecological relevance of a biological resource may enter into this value, it cannot be put on the same level with a systematic evaluation of the ecological-functional relevance of biodiversity, due to its methodical basis (individual preferences).
19. From a welfare-economic point of view the protection services are always element of a production function or an individual utility function and could therefore be integrated within these two value categories. However, to underline their importance as a value component of biodiversity they are listed in their own right here.
20. The other approaches mentioned in chapter 2 did make important contributions to the economic interpretation of the ecological service. However, for different reasons, they are no more referred to here for the terminological definition of the ecological relevance of biodiversity within the total economic value: the term “contributory value” describes only a segment of the ecological relevance of biodiversity (the contribution of one species for the preservation of other species) and is even by Norton himself no longer used in more recent publications. The term “inherent value” is unsuitable for describing the ecological functions, since, in literature, it is equated as well with the aesthetic or cultural values of nature (see for example Taylor 1986, p. 71). An unmistakable description of the ecological functions is therefore not guaranteed. The term “infrastructure value” is so far only very sporadically used in literature. It does not seem sensible to further enlarge the already existing multitude of terminologies.

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