



Ineffective biodiversity policy due to five rebound effects

Sara Maestre Andrés^a, Laura Calvet Mir^a, Jeroen C.J.M. van den Bergh^{a,b,c,f,*},
Irene Ring^d, Peter H. Verburg^e

^a Institute for Environmental Science and Technology, Universitat Autònoma de Barcelona, Edifici Cn – Campus UAB, 08193 Bellaterra, Cerdanyola, Spain

^b ICREA, Barcelona, Spain

^c Department of Economics and Economic History, Universitat Autònoma de Barcelona, Edifici Cn – Campus UAB, 08193 Bellaterra, Cerdanyola, Spain

^d Department of Economics, UFZ – Helmholtz Centre for Environmental Research, Leipzig, Germany

^e Department Spatial Analysis and Decision Support, Institute for Environmental Studies, VU University, Amsterdam, The Netherlands

^f Faculty of Economics and Business Administration & Institute for Environmental Studies, VU University Amsterdam, The Netherlands

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ABSTRACT

We explore the relationship between biodiversity, ecosystem services and conservation policy. A framework for studying their interdependence is proposed. We argue that a necessary (though not sufficient) condition for making a transition to a truly sustainable economy is that biodiversity conservation and its analysis take into account unwanted and avoidable indirect – i.e. rebound – effects of all kinds of biodiversity policy. We identify five types of such rebound effects and propose the terms biodiversity (two types), ecological, service and environmental rebound for these. The service rebound is associated with the problem of incongruence or conflicts, and thus the potential need for trade-offs, between ecosystem services or between such services and biodiversity conservation. Effective biodiversity policy requires the minimization of these various rebound effects.

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1. Introduction

Much has been written on biodiversity policy, from the perspective of biology, ecology, economics and policy sciences. What is missing in most writings is serious attention for the potential ineffectiveness of such policies in terms of unintended, unwanted and avoidable indirect effects. Effectiveness of biodiversity policy can be interpreted in various ways, namely in terms of biodiversity conserved, ecosystem functions (functional diversity) maintained, or ecosystem services guaranteed. Effective biodiversity policy is a necessary condition for making a transition to a truly sustainable economy. In order to develop our thinking about this issue we propose a framework that connects various types of diversity, ecosystem functions and services, values and biodiversity protection policies. This framework will allow us to identify potential unwanted, avoidable effects of biodiversity policies on each of these components. We refer to

such effects here as rebound, inspired by the literature on energy conservation and rebound (Sorrell, 2007; van den Bergh, 2011).

The drivers of biodiversity loss include many, such as hunting, land use, deforestation, fragmentation due to infrastructure, water use causing desiccation, and environmental pollution with climate change as a very important case. In addition, loss is enhanced by existing policies in sectors like agriculture, infrastructure and fisheries. An example is subsidies for biofuel production that promote conversion of tropical forest to tilled fields, which may reduce the area with habitats that support unique biodiversity (Kinzig et al., 2011). The complex set of drivers of biodiversity loss makes the analysis of effective policy not easy.¹

For addressing effectiveness well, the analysis of biodiversity policy needs to consider indirect, avoidable effects of biodiversity policy. We identify five categories of such effects, and propose the terms biodiversity (two types), ecological, environmental and (ecosystem) service rebound for these. These terms reflect that certain strategies aiming at conserving specific biodiversity have unintended effects which partly undo the direct conservation benefits, causing them to be less effective than is possible. These

* Correspondence to: Institute for Environmental Science and Technology, Universitat Autònoma de Barcelona, Edifici Cn – Campus UAB, 08193 Bellaterra, Cerdanyola, Spain.

E-mail addresses: sara.maestre@uab.cat (S. Maestre Andrés), lcalvetmir@gmail.com (L. Calvet Mir), jeroen.bergh@uab.es (J.C.J.M. van den Bergh), irene.ring@ufz.de (I. Ring), peter.verburg@vu.nl (P.H. Verburg).

¹ Of course, to achieve a sustainable economy it would also be necessary to address other policies that negatively affect biodiversity, for instance in areas like energy, agriculture, fisheries and infrastructure.

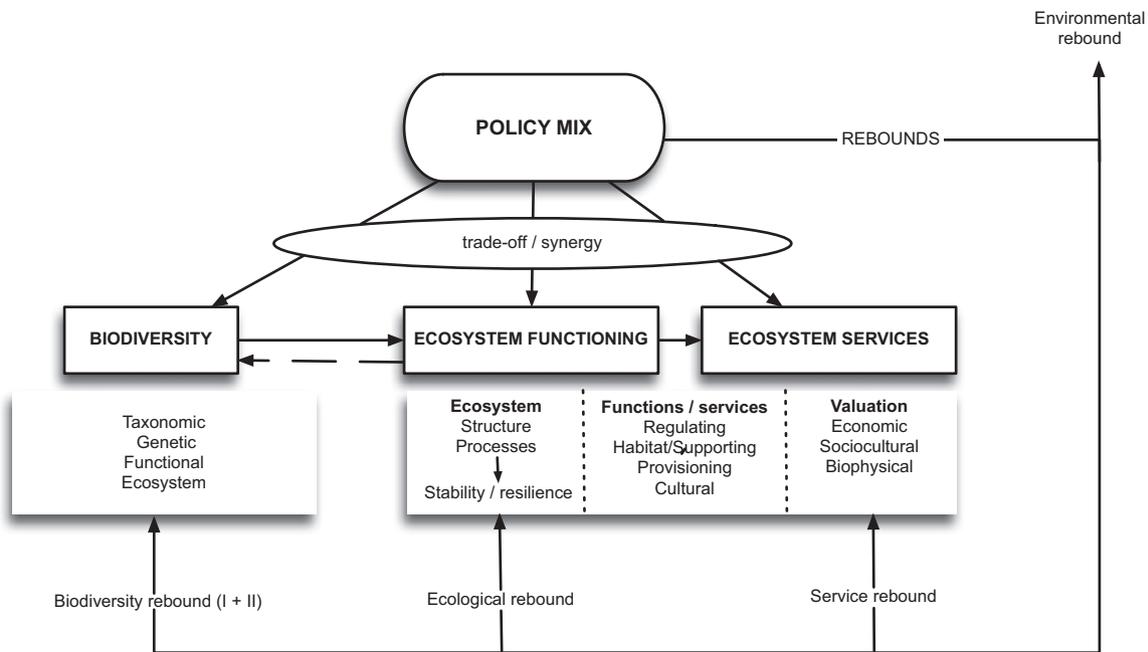


Fig. 1. A framework for biodiversity policy, ecosystem services and rebound effects.

rebounds can be reduced by appropriate design of policies and strategies. It is evident that effective biodiversity policy requires the minimization of these rebound effects.

Solving the rebound of biodiversity policies is not easy, however, as the ineffectiveness is not always transparent. Recognizing rebound will evidently depend on the precise interpretation of biodiversity and the type of biodiversity indicator used. Moreover, biodiversity-related rebound can follow different mechanisms, as exemplified by the five types of rebound of biodiversity policy. In order to understand the chain of cause-effect relationships from biodiversity through ecosystem function to ecosystem services, values and policy, and the place of the five types of rebound, we present the scheme in Fig. 1. Its elements will become clear in due course. Note that the aggregate classification of ecosystem functions and services is identical because the latter are appropriated functions. This does not deny that at a more disaggregate level there can be a distinction between the two, i.e. some functions are not appropriated or do not directly generate a service to humans.

In line with the aim of an opening issue of a new journal, we want to raise relevant research questions – both in terms of research and policy – about this theme. This includes discussing different notions of biodiversity, their connection with ecosystem services, how to compare policy options, and the role of ecosystem valuation concepts and methods to assess biodiversity loss or protection. Our discussion aims at providing arguments for broadening the analysis of biodiversity policy design by considering various types of indirect or rebound effects. Ultimately, this may give rise to distinct and new views on effective policy options and instruments.

2. Interpretations of biodiversity, ecological significance and policy relevance

Biodiversity has been defined as: “... the variability among living organisms from all sources, including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species, and of ecosystems” (Convention on Biological Diversity,

1993). Four fundamental facets of biodiversity can be distinguished, namely taxonomic, genetic, functional and ecosystem diversity. Taxonomic diversity refers to different taxa (e.g., class, order, family, genus, species). One specific type is species richness, that is, the number of species at a particular site or at a global scale. Genetic (or phylogenetic²) diversity is the genetic variation within and between species which is the fundamental level of diversity underpinning the other types. Functional diversity measures the number, type, and distribution of functions performed by organisms within an ecosystem, and thus reflects the diversity of morphological, physiological and ecological traits within biological communities and their interactions. It further indicates a degree of complementarity and redundancy³ of co-occurring species (Díaz and Cabido, 2001; Hooper et al., 2005). Finally ecosystem diversity refers to the diversity of assemblages and their environments over a defined landscape, ecological zone or at global scale (Swift et al., 2004).

Another way of classifying biodiversity is based on spatial characteristics. A common distinction is based on the spatial focus of analysis being local or a habitat (alpha diversity) versus regional or a landscape (gamma diversity). In addition, a more contentious notion, beta diversity or spatial turnover, captures among-site components or number of sub-units (habitats) (Hooper et al., 2005). Sometimes the relation between these diversity notions is summarized as $\gamma = f(\alpha, \beta)$, but this involves implicit assumptions.

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

² Phylogenetic diversity relates to the evolutionary history of a species (Faith, 1992).

³ Species that are redundant for one ecosystem process may not be redundant for others and species considered redundant under certain environmental conditions may become important under changed conditions (Harrington et al., 2010).

that local loss of non-unique species sometimes implies a loss in biodiversity value (Hammond et al., 1995).⁴

There is debate about which dimension of biodiversity is better in order to analyse the diversity of an area. Traditionally, taxonomic diversity has been the more used indicator. However, it is now well recognized that functional and phylogenetic relationships are also important indicators of biodiversity (Strecker et al., 2011). Consequently, functional diversity is now often assumed to be a better predictor of ecosystem functioning than other measures of diversity (Hooper et al., 2005). Nevertheless, one should not forget that interactions and functional roles of species involve complex and often unknown aspects, suggesting that phylogenetic diversity might capture species assemblages better than functional diversity to explain ecosystem productivity (Devictor, 2010).

Recent studies demonstrate that regions of high taxonomic diversity may be incongruent with regions of high functional or phylogenetic diversity (Forest et al., 2007; Strecker et al., 2011). Moreover, Devictor et al. (2010) found that phylogenetic and functional diversities were uncorrelated in many cases. However, this depends on the spatial scale of study of biodiversity, i.e. alpha, gamma or beta diversity. For instance, using beta-diversity patterns (among sites), functional and phylogenetic diversity were found to be positively related to taxonomic diversity, while the correlation between functional and phylogenetic beta-diversity was even higher.

These patterns of (non-)congruence of distinct indicators suggest that species occurring locally may derive from regional species pools with similar as well as different biogeographical and evolutionary histories (Cumming and Child, 2009). Moreover, for a given regional pool, species may respond to environmental gradients in different ways, which affects the spatial distribution of functional and phylogenetic diversity and can generate a spatial mismatch between taxonomic, functional and phylogenetic diversities (Prinzing et al., 2008). However, strong environmental filters could restrict species composition to a relatively restricted range of functional characteristics, thereby limiting the degree of functional diversity capable of influencing different ecosystem properties (Grime, 2001). Increasing species richness would then just lead to a finer division of the available niche space rather than to greater functional diversity (Díaz and Cabido, 2001). Mapping beta-diversities reveals coherent transitional zones between regions with different pools of species, functional or phylogenetic diversity. This approach can thus help to identify and delimit ecological boundaries around areas of particular interest. On its own, beta-diversity is however silent on the amount of diversity of a given region. For instance, high beta-diversity can be found in highly fragmented landscapes with low gamma diversities if few species or little functional or phylogenetic diversity is found in these landscapes. Therefore, gamma and beta diversities offer complementary information on biodiversity patterns (Devictor et al., 2010).

This is all not just interesting for theoretical reasons but also shows a clear connection with policy. For example, in protected areas networks areas having the highest taxonomic diversity were protected whereas areas having the highest phylogenetic and functional diversity received less protection. A similar analysis for beta-diversity revealed a different pattern: Areas having the highest beta-taxonomic and phylogenetic diversity values were well protected, while areas with the highest functional diversity received less protection (Devictor et al., 2010). Measuring each of

these complementary biodiversity components is necessary for understanding ecosystem functioning in terms of the complete structure, composition and dynamics of natural communities. Associating these with the ecosystem services provided in the relevant area allows one to develop a systematic conservation planning that accounts for multiple aspects of biological diversity, reflecting taxonomic, functional, and evolutionary perspectives (Bello et al., 2010; Hooper et al., 2005; Strecker et al., 2011).

2.1. *The mutual relation between biodiversity and ecosystem functioning*

Biodiversity both responds to, and influences, ecosystem functioning (Holling et al., 1995; Hooper et al., 2005). Variations in ecosystem functioning can result from fluctuations in the environment from year to year, directional changes in conditions, abiotic disturbance, or biotic disturbance. Although there is no a priori reason to expect that different ecosystem properties have a single pattern of response to changes in different components of biodiversity, some studies show that the most important dimension of biodiversity which influences ecosystem functioning are species' functional characteristics. These include effects of dominant species, keystone species, ecological engineers, and interactions among species (e.g., competition, facilitation, mutualism, disease, and predation). In addition, comparisons of distinct ecosystems suggest that abiotic conditions, disturbance regime, and functional traits of dominant species have a larger effect on many ecosystem properties than species richness.

Hooper et al. (2005) summarized the main responses of ecosystem functioning to changes in species or functional diversity. As shown in Box 1, the patterns depend on "... the degree of dominance of the species lost or gained, the strength of their interactions with other species, the order in which species are lost, the functional traits of both the species lost and those remaining, and the relative amount of biotic and abiotic control over process rates ..." (p. 9).

3. **Linking ecosystem functioning, ecosystem services and biodiversity**

There is a broad consensus in the scientific community about the relationship between ecosystem functioning and ecosystem services. Ecosystem functions can be defined as all aspects of the structure and processes of ecosystems with the capacity to produce services that satisfy human needs directly or indirectly (Hooper et al., 2005). Gómez-Baggethun and de Groot (2010) distinguish between potential benefits associated with ecosystem functions and real benefits, which are the potential ones concretized once they are used or enjoyed by people. One can regard ecosystem services somehow as a simplifying translation of ecological complexity to a limited number of functions and ecosystem services. Various classification of these have been proposed in the past (e.g., Turner et al., 2000), which have converged to a quite uniform view of main categories and detailed services. Influential classifications have been proposed by de Groot et al. (2002), the Millennium Ecosystem Assessment (MEA, 2003) and The Economics of Ecosystems and Biodiversity (TEEB, 2010b). These divide ecosystem functions and services into four main categories: regulating services, such as the regulation of climate, maintenance of soil fertility and waste-water treatment; habitat or supporting services, such as habitats for species and maintenance of genetic diversity; provisioning services, which include food, raw materials, water and medicinal resources; and cultural services, like recreation and aesthetic appreciation.

However, there is not yet agreement on the conceptualization of biodiversity and its relationship with ecosystem functioning and services. According to MEA (2005), biodiversity represents the foundation of ecosystems that, through the services they

⁴ Stirling (2007) proposes to regard diversity as a multidimensional concept, consisting of three dimensions, namely variety, balance and disparity. Variety denotes the number of different identities (functions, physical appearances, or behaviours) in a population of elements. Balance (or equality) captures the distribution or frequency of the various identities. Disparity refers to the degree of distance or qualitative difference between the identities in a population.

Box 1—Possible responses of ecosystem functioning to biodiversity change.

- (1) Diversity might have no effect: changing relative abundance or species richness might not change process rates or pool sizes. Lack of response could occur for several reasons, such as primary control by abiotic factors, dominance of ecosystem effects by a single species that was not removed, or strong overlap of resource use by different species.
- (2) An increase in diversity originates a change in ecosystem functioning, associated with two main mechanisms: increasing species richness increases the likelihood that those key species, which have a dominant effect on ecosystem properties, are present; species or functional richness contributes to ecosystem properties through positive interactions among species.
- (3) An increase in diversity implies a saturating response in ecosystem functioning. This is the most commonly hypothesized pattern, where complementarity, facilitation, and sampling effects for high productivity (or other properties) are all expected to show a similar saturating average response as diversity increases.
- (4) Complementarity and selection or sampling effects are not necessarily mutually exclusive. There can be a continuum of diversity effects, ranging from the probability of sampling one dominant species to the probability of selecting several complementary species.
- (5) Ecologists disagree over whether sampling effects are relevant to natural ecosystems. Some ecologists argue that sampling effects are artefacts of certain experimental designs because of their dependence upon the debatable assumption that communities are random assemblages of species from the total species pool. Others assert that they are simply an alternative mechanism by which species richness might affect ecosystem properties in natural communities, pointing out that there are many stochastic factors that can influence community composition.
- (6) Adding trophic levels is expected to lead to more complex responses of ecosystem properties to a change in biodiversity.

Based on Hooper et al. (2005).

provide, affect human well-being. Thus biodiversity is an abstract notion that affects the generation of a multitude of ecosystem services, and is associated with notions like integrity, stability and resilience of complex systems.

Another view is expressed by Mace et al. (2012). They conceptualize biodiversity as (a) a regulator of ecosystem functioning, (b) a final ecosystem service, and (c) a good that has value of its own. The first role is argued to be the most important one. For example, the dynamics of many soil nutrient cycles are determined by the composition of biological communities in the soil while resilience to pests or environmental change improves with more diverse biological communities. The second characterization responds to the argument that biological diversity at the level of genes and species contributes directly to some goods and their values. For instance, the potential value of wild medicines and the potential benefits from bioprospecting for medicinal purposes increase directly with the number and genetic distinctiveness of species. The third conceptualization follows from biodiversity being itself the direct object valued by humans. Many components of biodiversity may be seen to have cultural value, including appreciation of wildlife and scenic places and spiritual, educational and recreational values. Nevertheless,

biodiversity is difficult to disentangle and measure which suggests that considering it directly as a service or a good to which instrumental value is assigned can be problematic. This is further discussed in Section 4 on biodiversity values.

In addition, there is debate on the role of biodiversity in delivering or enhancing ecosystem services provision. Some authors state that biodiversity can enhance ecosystem productivity (production of ecosystem services) and ecosystem stability. Generation of ecosystem services has been related to biological characteristics and more specifically to functional traits of ecosystems. Recent studies have argued that the multiple associations between functional traits and services, so-called trait-service clusters, can form the basis for ecosystem management and decision-making (Bello et al., 2010). For instance, for plants there is increasing evidence about the effects of community-level functional traits on ecosystem functioning that underlies important ecosystem services. A given ecosystem property could contribute to several ecosystem services: for example, diversity of flowering onset dates contributes to agronomic, cultural and pollination services (Lavorel et al., 2011). Furthermore, an ecosystem service is related to many ecosystem properties. For instance, high cultural value is related to high species diversity and highly diverse flowering characteristics. Some studies indicate that changes in biodiversity probably affect more regulating and cultural services, and long term resilience of ecosystem processes, and less provisioning services, at least in the short term (Mace et al., 2012; Lavorel et al., 2011).

In the specific case of agroecosystems, research suggests that their capacity to deliver a variety of ecosystem services depends on the intensity of use and on the diversity of croplands. For example, Sandhu et al. (2010) attribute a larger flow of ecosystem services to organic than to conventional agriculture, defined as agriculture based on monoculture and intensive use of agrochemicals, fuel, and machinery. In the same vein, Altieri (1999) and Jackson et al. (2007) argue that agriculture based on traditional practices like intercropping, agroforestry, or shifting cultivation delivers more ecosystem services than conventional agriculture, for various reasons. First, traditional agriculture largely relies on the maintenance of agrobiodiversity (Altieri, 1999; Jackson et al., 2007), thereby combining agricultural productivity with the delivery of other regulating services that biodiversity provides (MEA, 2005). Second, maintenance of agrobiodiversity in agricultural landscapes enhances the resilience of agroecosystems, i.e. their capacity to reorganize after disturbance, thereby enhancing the likelihood of maintaining the supply of ecosystem services over time in the face of variability and change (Jackson et al., 2007). Third, the adaptation of traditional agriculture to site-specific biological, edaphic, and climatic conditions reduces the dependence on inputs of machinery, agrochemicals, and fuel, thereby reducing related disservices in terms of soil compaction, water pollution, and greenhouse gas emissions (Altieri, 1999). For instance, maintenance of high biodiversity levels in specific taxonomic groups (maintenance of landraces) improves the performance of ecosystem services by enhancing pest control, pollination, or soil fertility (Altieri, 1999; Jackson et al., 2007). In addition, the habitat service “maintenance of landraces” is tightly connected with important cultural services, such as “heritage value of home gardens and associated traditional ecological knowledge” and “place for creating and enhancing social networks” since both landraces and knowledge are spread throughout seed exchange networks (Calvet Mir et al., 2012).

As has been shown, there is growing consensus among ecologists that, in general, biologically diverse ecosystems provide a greater flow of ecosystem services than non-diverse systems (Hooper et al., 2005; Lavorel et al., 2011). Nevertheless, characterizing multiple ecosystem services and biodiversity across the same region has only recently emerged as a field of study, which

means that levels of congruence are still poorly understood. The little quantitative evidence available to date has led to mixed conclusions (Chan et al., 2006). Comparing eco-region distribution data for biodiversity and a limited set of ecosystem services, Naidoo et al. (2008) find that optimizing for individual ecosystem services (carbon sequestration, carbon storage, grassland production of livestock, water provision) conserved only 22–35% of the species for a given area as did optimizing for species, that is, no more than were conserved by selecting ecoregions at random. They also found that maximizing species representation for a given area captured only 17–53% of maximum ecosystem service provision, depending on which service was considered and at which area limit the comparison was made. These levels of ecosystem service capture from species optimization were, again, no greater than those from a random selection of ecoregions (Naidoo et al., 2008).

Other studies exploring spatial patterns in the distribution of ecosystem services across landscapes analyze the spatial concordance between ecosystem services and biodiversity. They find that ecosystem services and biodiversity are interdependent (Egoh et al., 2008; Goldman and Tallis, 2009). However, there remains disagreement about whether spatial congruence of ecosystem services and biodiversity is rare or not, and what this implies for ecosystem management. Without knowledge about relationships between biodiversity and ecosystem services provision and among ecosystem services, we are at risk of designing policies that imply unwanted trade-offs.

4. Values of biodiversity and ecosystem services

Valuation can be seen as the process of assigning importance to objects and actions. Pascual et al. (2010) mention two major types of valuation, namely (a) ecological valuation based on bio-physical accounting which neglects human needs or wants and (b) economic valuation based upon consumer preferences. The latter takes the form of monetary valuation using market and non-market valuation approaches. In addition, one can identify socio-cultural valuation using a subjective evaluation approach (e.g., with a Likert scale) (Brondizio et al., 2010; Calvet Mir et al., 2012). The relevance of group-based, social and cultural valuation in relation to biodiversity and ecosystem services has gained recognition (EPA-SAB, 2009).

Valuing biodiversity economically is controversial (Ring et al., 2010). An optimistic perspective is based on the idea that one is able to disentangle or decompose the total economic value of biodiversity into different types of values (as discussed in Box 2). The economic value of ecosystem services refers to instrumental values, resulting from the interaction of a human subject willing to pay for a (change in) an object (the ecosystem service), as opposed to intrinsic values in which case the subject plays no role. Most environmental economists consider that valuing biodiversity is a necessary step to make rational and accurate choices and trade-offs. Pavan Sukhdev, coordinator of the TEEB report, considers valuation in the broadest sense, including cultural and social approaches, as a key tool for conserving biodiversity: “lack of valuation is, we are discovering, an underlying cause for the observed degradation of ecosystems and the loss of biodiversity” (TEEB, 2008, p. 4). Any decision or policy affecting biodiversity implicitly assigns a value to it. Moreover, despite its shortcomings, monetary valuation of welfare impacts – particularly when using a referendum type of format – might be considered as a democratic approach to decide about public policy regarding biodiversity, that is, as long as certain conditions are fulfilled, such as having a not too uneven income distribution and equal access to ecosystem services.

Box 2—A typology of economic values of biodiversity.

Different types of economic value have been proposed in the literature (Turner et al., 2000; Nunes and van den Bergh, 2001; Pascual et al., 2010). *Utilitarian or (direct) use value* of components of biodiversity refers to the productive and consumptive uses of organisms or genes that are part of the local diversity as inputs into consumption and production processes. These are the subsistence and commercial benefits of species or their genes.

Indirect use value can be seen as the value of biodiversity contributing to ecosystem life support functions and the preservation of ecological structure and integrity (Swift et al., 2004). It can also denote biodiversity at a certain location affecting through complex ecosystem links a value at other locations. Barbier (1994) defines it as “... support and protection provided to economic activity by regulatory environmental services ...” (p. 156). Different terms for the same notion are *contributory value*, *primary value*, and *infrastructure value* of biodiversity (see Farnworth et al., 1981; Norton, 1986; Gren et al., 1994).

Option value is the value (a kind of use value) of keeping an option open for potential future use. *Quasi option value* is the value of being able to obtain information by keeping an option open, such as learning about unique species in the future by preserving all tropical forests. *Bequest value* represents the value of biodiversity for use by our offspring, or more generally future generations. *Philanthropic (altruist) value* is the value associated with use by others in our generation.

Non-use value is the value that biodiversity has on its own, without a (human) subject using it. According to some this value comprises cultural and social benefits, although the exact separation with use values is debatable (that is why some prefer the term *passive use value*). Indeed, use is often implicit, like in the case of watching movies or photos of species or nature. One has to distinguish here between *intrinsic* and *existence values*. The first is really another value concept (without a subject, so non-transformable into monetary units, but instead often taking the form of a “right”). *Existence value* of an environmental entity reflects humans capturing its intrinsic value or the instrumental value it has for ecosystems or non-human species (Attfield, 1998).

There are many arguments in favor of economic valuation of biodiversity with which one can agree or disagree. If one strives to support public policy with information about biodiversity values, then one needs a clear understanding of the relationships between biodiversity types, ecosystem functioning, ecosystem service categories, and biodiversity policies as these define the scenarios to be valued. Fig. 1 already provided a schematic perspective on these relationships. But it is good to realize that biodiversity policies, even incentive-based economic instruments, do not necessarily require economic valuation; one can instead use information about responses by economic agents to prices, or just experiment with price incentives to find out which price levels regulate behavior within safe, desirable biodiversity standards.

Decomposing the total value of biodiversity into direct and indirect use, non-use, option or quasi-option values as in Box 2 is difficult for a number of reasons (Nunes and van den Bergh, 2001). One is that there are different types or levels of diversity as discussed in Section 2, so the question is which one needs to be valued. In addition, valuation will lead to an under-estimation of the ‘real’ value because so many links between biodiversity and value categories are easily overlooked or simply cannot be empirically assessed. Some feel uncomfortable with putting an instrumental value on biodiversity and argue that biodiversity mainly has intrinsic value (Ehrenfeld, 1988). This view regards biodiversity as an abstract notion that is associated with notions

Table 1
Illustrating the most relevant relationships between ecosystem services, biodiversity dimensions and economic values.

Ecosystem service	Most relevant biodiversity dimensions	Most relevant economic value types
Regulating (climate regulation, waste water treatment, pollination, etc.)	Functional, ecosystem	Indirect use, option value
Habitat/supporting (habitats for species, maintenance of genetic diversity)	Genetic, functional	Indirect use, quasi-option, existence
Provisioning (water, raw materials, food, medicinal resources, etc.)	Genetic, taxonomic, functional	Direct use, option, quasi-option, bequest
Cultural (knowledge, recreation, aesthetic, etc.)	Taxonomic, ecosystem	Direct use, option, existence, bequest

like integrity, stability and resilience of complex systems, and thus difficult to disentangle and measure. One may see the value of ecosystems and their services as a metaphor, which is useful in communicating science-based insights to policy makers. The success of the much debated “Value of Nature” article by Costanza et al. (1997) can perhaps be understood in this way. Kosoy and Corbera (2009) point out that monetary valuation runs a risk of leading to partial or incomplete sets of values of ecosystems upon which policies and strategies will be based, which then neglect non-monetized values.

All in all, it is unavoidable that there are different opinions on biodiversity value. In the United States, where executive orders often require economic cost-benefit analyses, the Environmental Protection Agency is now actively promoting the use of a wider range of valuation methods, including measures of attitudes, preferences and intentions, civic valuation, decision science approaches, ecosystem benefit indicators, and biophysical ranking methods (EPA-SAB, 2009; Ring et al., 2010). From a collective choice perspective, social norms and institutions are crucial for societal decision making (Vatn and Bromley, 1994). As alternatives, consensual, multi-criteria, multi-stakeholder and group-based deliberative valuation processes have been suggested as more appropriate. Here people act as citizens, not (only) as consumers (Funtowicz et al., 1998; Lienhoop and MacMillan, 2007; Spash, 2008a,b; Spangenberg and Settele, 2010). Others feel that laypersons cannot judge the relevance and complexity of biodiversity-ecosystems-functions-services relationships and thus are unable to value biodiversity and associated ecosystem services appropriately. Instead, judgments about biodiversity changes are then better left to experts, like biologists. An intermediate solution is to let experts inform laypersons before confronting the latter with valuation questions (Nunes and van den Bergh, 2001). Surprising perhaps, Calvet Mir et al. (2012) found a high statistically significant correlation between the responses by laypersons and a panel of scientists on the socio-cultural valuation of ecosystem services provided by home gardens. Finally, according to Pearce (1999), “... much of the literature on the economic valuation of ‘biodiversity’ is actually about the value of biological resources and it is linked only tenuously to the value of diversity...” But whereas biodiversity refers to the variety of life, biological resources refer to the manifestation of that variety.

Table 1 illustrates possible connections between the most relevant biodiversity dimensions that contribute to the provision of each of the four types of ecosystem services and the most relevant economic values associated with these services.

5. A typology of rebound of biodiversity policies

After exploring the main definitions of biodiversity in the literature and establishing the link between biodiversity, ecosystem services and their values to society, in this section we aim to characterize the potential rebound effects of biodiversity policies. This will allow us to advance in the promotion of more effective policies. A biodiversity policy can be considered to be effective if it will produce the conservation benefits as desired *ex ante* (Doremus, 2003). One might also define effectiveness in a more abstract way as attaining the highest marginal environmental

benefit associated with a given instrument (OECD, 2007; Ring and Schröter-Schlaack, 2011).

Different biodiversity conservation policies can deal with particular causes of biodiversity loss, such as hunting, habitat destruction (land use, deforestation, fragmentation), water use (causing desiccation), and environmental pollution (with climate change as a special and very important case). At a more fundamental level, one can identify environmental externalities, myopia (a high rate of time discounting), a lack of adequate property rights, and to a lesser extent market power and asymmetrical information, as the indirect causes of biodiversity loss. Different types of biodiversity conservation instruments can be designed to deal with these various causes. On the one hand, policies may aim to provide prohibitions, barriers, standards (e.g., land tenure and use rights), or negative incentives like prices (subsidies, land or product taxes, access fees) to alter behavior and projects that negatively affects biodiversity (TEEB, 2010a, 2011). On the other hand, policies may provide positive incentives such as payments for environmental services or ecological fiscal transfers that aim to direct behavior towards biodiversity conservation (ten Brink et al., 2011). The most important biodiversity policy instruments are summarized in Table 2.

The types of policies in the table are likely to score differently in terms of effectiveness, depending on the context and application. This section aims to draw attention to a kind of government or policy failure that affects effectiveness, namely the unintended indirect effects and potential ineffectiveness of policies. Biodiversity policies can have a number of unintended, unwanted and avoidable rebound effects.⁵ We propose the following typology of rebound:⁶

1. *Biodiversity rebound I (spatial spillover)*: Policy to protect one type of biodiversity in a certain area has a negative impact on such biodiversity elsewhere, i.e. in another region. This rebound operates through spatial spill-over effects which some have called displacement or leakage (Lambin and Meyfroidt, 2011). An example is restricting outdoor recreation in one nature area that leads to recreationists moving to other areas so that environmental pressure there increases with potentially negative impacts on biodiversity. Or deviating water flows in the landscape to assist in the protection of biodiversity in a wetland can lead to water shortage and desiccation in other nature areas with consequences for respective biodiversity. Evaluations of the effectiveness of protected areas showed that associated conservation policies may lead to an increase in deforestation rates outside these protected areas (Lambin and Meyfroidt, 2011) reducing habitats for biodiversity conservation. In the case of Sumatra, a

⁵ If they would not be avoidable one could think of compensation measures.

⁶ We are not assuming more than 100% rebound here. The latter is also known as the “Jevons paradox” in the context of energy rebound, because of the English economist William Stanley Jevons who in his 1865 book “The Coal Question” drew attention to the risk that a more efficient steam engine would increase rather than decrease the demand for coal. Of course, more than 100% biodiversity rebound should withhold one from implementing the respective biodiversity conservation policy in the first place.

Table 2

Biodiversity conservation instruments and their characteristics.

Source: Adapted from Ring and Schröter-Schlaack (2011).

Instruments	Incentive	Direct target
<i>Regulatory instruments</i>		
Direct regulation and spatial planning	Coercion	Various behaviors that negatively affect biodiversity
<i>Economic instruments</i>		
Biodiversity offsets and mitigation banking	Avoiding a fine	Planning of projects that harm biodiversity
Environmental taxes	Tax	Various behaviors that negatively affect biodiversity
Tax reliefs	Avoiding a tax	Various behaviors positively affect biodiversity
Ecological fiscal transfers	Payment	Various behaviors that positively affect biodiversity
Environmental subsidies	Payment	Various behaviors that positively affect biodiversity
Government financed payments for environmental services	Payment, contract	Compliance with terms of contract to protect biodiversity
Market-based payments for environmental services	Payment, contract	Compliance with terms of contract to protect biodiversity
<i>Voluntary and information-based instruments</i>		
Voluntary instruments	Prevention of coercive regulation	Compliance with voluntary agreement or pledge to protect biodiversity
Certification	Regulating access to a market or gaining a good reputation	Compliance with code of conduct to protect biodiversity

reduction of deforestation in adjacent unprotected areas was observed, probably due to urban migrations (Gaveau et al., 2009). By contrast, a study in the Peruvian Amazon (Oliveira et al., 2007) found that although forest subject to legal concessions experienced a large reduction in deforestation, after enactment of stringent timber harvest legislation the rates of forest clearing and disturbance outside areas with concessions increased rapidly. Protection of public forests in the US Pacific Northwest also displaced timber harvests on private timberlands in the region and further away, with a total displacement of 84% of the reduced public harvest timber because of conservation programs (Wear and Murray, 2004). A similar leakage effect was found for cropland in the United States, where the purchase of conservation easements on farmland brought non-cropland into crop production elsewhere, for about 20% of the cropland area that was retired from cultivation (Wu, 2000).⁷

2. *Biodiversity rebound II (incongruence between protection of different types of biodiversity)*: Policy to protect one type of biodiversity (e.g., genetic) can negatively affect another type of biodiversity (e.g., taxonomic or functional). As has been discussed in the previous sections, areas of high conservation interest are traditionally defined as biodiversity hotspots, but sometimes they are based upon rather arbitrary criteria. In fact, both past and current conservation strategies have frequently focused on giving priority to certain taxa or areas to protect rarity, endemism and distinctiveness (Hooper et al., 2005). For instance, French protected areas have underrepresented functional diversity, having been established following taxonomic diversity patterns (Devictor et al., 2010). Another example is providing incentives for habitat protection through creating corridors between protected areas which may increase disease risks by promoting contact between wild and domesticated animals (Kinzig et al., 2011). Only if all types of biodiversity are perfectly correlated will protection of one imply protection of the others, so that there are no conflicts and trade-offs required. However, as indicated by the conclusions of Section 2, this is unlikely to be the case.

3. *Ecological rebound*: As has been shown in Section 2, changes in biodiversity may lead to various responses in ecosystem functioning, some intended and foreseen but others not. As a result, biodiversity conservation policy might through its effect on particular biodiversity work out negatively on certain ecological relations. For example, red-list species conservation schemes can lead to population growth of particular species, in turn giving rise to a loss of equilibrium between different species in the ecosystem, because of food scarcity or predator pressure. This is discussed in more detail in the illustration of the Weitzman assessment of biodiversity policy below. When ecological changes affect ecosystem functional diversity, this rebound type overlaps with biodiversity rebound II.

4. *Service rebound (trade-off between biodiversity and ecosystem services)*: Although biodiversity conservation policy is increasingly justified based on the ecosystem services provided, there is still incomplete empirical evidence that there exists a strong relationship between biodiversity conservation and supply of ecosystem services. In fact, biodiversity policy may protect a certain type of biodiversity while degrading or sacrificing a particular ecosystem service. We recognize here that two different types of trade-offs exist, one between ecosystem service provision and biodiversity and another between different types of ecosystem services (Raudsepp-Hearne et al., 2010; Ring et al., 2010; Willemen et al., 2010). Since we are concerned with biodiversity conservation the first type of trade-off is more relevant here. An example is the case in which biodiversity conservation implies a transformation of landscapes formed by a combination of culture and nature to more pure nature, with a loss of cultural values as a result. Another example is provided by the study done by Chan et al. (2006). It examines the potential trade-offs between goals for biodiversity and for certain ecosystem services. The authors find that there is a low average correlation between biodiversity and the six services studied (carbon storage, flood control, forage production, pollination, recreation and water provision). Moreover, crop pollination and forage production show a negative correlation with biodiversity. Another case of a trade-off between biodiversity and ecosystem services is conserving certain species that need dense, old-growth or primary forests, such as the boreal owl (*Aegolius funereus*), and provisioning ecosystem services, like grazing and timber production. An example of service rebound in the context of marine

⁷ We do not claim that all biodiversity policy is subject to rebound. For instance, Andam et al. (2008) find in a study for Costa Rica that deforestation spillovers from protected to unprotected forests are negligible. Our aim here is merely to classify potential channels or mechanisms of rebound of biodiversity policies.

ecosystems is a protected area policy aimed at forbidding fishing in order to conserve certain protected species. This can reduce provisioning services for fishermen while increasing biodiversity in the protected zone, although there are examples where marine protected areas may considerably increase fish catches close to their edges due to regenerating fish stocks within the protected areas (Flogarty and Botsford, 2007). There is a fundamental incongruence, and thus a conflict and need for trade-off, between (maintenance of) biodiversity and ecosystem services. As a result, this type of rebound can never be completely removed. Therefore, it is likely that in case of provisioning services such a trade-off will lead to increased demands on services in other locations in order to fulfil the (worldwide) demand for them. Such displacement of service demand (somewhat resembling biodiversity rebound I: spatial spillover) may negatively affect ecosystems elsewhere. In case of services that are more location specific such displacement is less likely.

5. *Environmental rebound*: Biodiversity policy can generate a negative impact on certain environmental indicators. This is also known in the literature as shifting or cascading (Lambin and Meyfroidt, 2011) of one to another environmental problem. For example, biodiversity conservation leading to less use of tropical hardwood may lead to a shift in consumption and associated industries to other construction materials that involve chemicals or toxic components, or use a lot of CO₂-intensive energy. This then means a shift to other types of environmental problems. This will not always be easy to empirically demonstrate, as it involves 'invisible' behavioral and economic mechanisms. Harvey et al. (2010) mention carbon leakage as a potential risk of REDD (Reducing Emissions from Deforestation and Degradation) aimed at a combination of carbon sequestration and biodiversity protection.

Note that rebound can occur through physical processes and displacement: e.g., using water to maintain wetlands may create drought conditions elsewhere (e.g., on a river trajectory) and put pressure there on biodiversity. Alternatively, rebound can involve an economic mechanism: e.g., spending limited budget on biodiversity protection in one spot may lead to deviating money from conservation elsewhere, or behavioral change stimulated by conservation policy (e.g., through environmental taxes or ecolabels) leads to new consumption and production activities that cause pressure on biodiversity or other environmental media. In addition, rebound can be local or nearby (like when water use affects adjacent ecosystems) or distant in space (because of economic or large-scale environmental processes).

Two other, related aspects of biodiversity loss and conservation matter for rebound of particular policies: the combined effects of multiple factors and pressures behind biodiversity loss, and the interaction or synergy of multiple, simultaneously active policies (policy mix) (Schröter-Schlaack and Ring, 2011). This complex nature of the interaction of causes and policies should be addressed when one aims to completely assess the potential rebound of biodiversity policy.

Finally, rebound also can result from global agreements on biodiversity that have all kinds of local effects, some of which are unintended. Such agreements need to develop effective mechanisms to eliminate such rebound effects to make the policies more effective. For instance, international policies that express support for conservation, such as the Convention on Biological Diversity, typically fail to have adequate precision and clarity to save many of the unique, agrobiodiversity-rich areas on the planet (Harrop, 2007). Instead, village level and regional institutions often may assure more biodiversity conservation through the engagement of local communities in activities that improve their livelihoods (Bawa et al., 2007; Jackson, 2007).

5.1. Weitzman on biodiversity policy: genetic distinctiveness and ecological rebound

Systematic conservation planning has traditionally focused on identifying priority areas that ensure adequate representation of measures of taxonomic diversity, such as species richness (Margules and Pressey, 2000). Consequently, as has been shown in many studies, functional diversity has been significantly under-represented whereas taxonomic diversity has been significantly over-represented in protected areas (Devictor et al., 2010). Also in economic models of biodiversity loss, biodiversity is mainly considered at the species level, paying attention to taxonomic diversity, or in some cases prioritizing species conservation based in genetic information (Eppink and van den Bergh, 2007).

To illustrate potential ecological rebound (rebound type 3) of policies that focus on specific biodiversity, consider the approach of Weitzman (1998). He studied the problem of protecting biodiversity under a limited budget constraint, but without considering certain ecological dynamics. It is relevant here as it is a well-known approach that is regarded by many economists as useful for biodiversity policy assessment.

He derives the following criterion for setting priorities among biodiversity-protecting projects:

$$R_i = (D_i + U_i) \frac{\Delta P_i}{C_i} \quad (1)$$

Here R_i represents the performance index of species i , D_i is the (genetic) distinctiveness of species i (meaning roughly how unique or different a species is), U_i denotes the direct utility associated with preservation of species i , and C_i is the cost of the protection project that increases the probability of survival of species i by ΔP_i . Uncertainty of extinction is introduced by defining P_i as the probability of survival of species i , so that $1 - P_i$ is the probability of extinction of species i . These probabilities are exogenous, i.e. they originate from outside Weitzman's framework.

van der Heide et al. (2005) draw attention to the lack of ecological considerations in Weitzman's criterion. They suggest that the ecological interdependence among species can in the context of Weitzman's criterion be modeled by defining mutually dependent rather than exogenous, independent survival probabilities. For their survival species depend very much on other species, through food web and ecosystem relationships. This implies that, generally, the extinction of one species will have an impact on the survival probabilities of certain other species. The conclusion is that Weitzman's ranking criterion generally holds only under very limited conditions: namely, when the probabilities of extinction of species are exogenous and constant. This assumption seems to hold mainly, and perhaps only, for ex situ conservation, which severely limits application of the criterion. Applying Weitzman's criterion to in situ conservation can provide an incorrect ranking of biodiversity policies leading to ecological rebound because it misunderstands ecological relationships between species. Note that this in turn means that biodiversity rebound is relevant here, as ecological (species) relationships determine functional diversity.

6. Concluding remarks

We have presented a framework of relationships between biodiversity, ecosystem functioning, ecosystem services and various types of rebound of biodiversity policy. The concern behind identifying potential rebound mechanisms is to design effective policies for biodiversity protection. Making sure that biodiversity policy is effective is a necessary condition for realizing a transition to a truly sustainable economy.

We have provided a preliminary classification into five types of rebound (biodiversity-spatial, biodiversity-incongruence, ecological, environmental and service) and have provided a preliminary set of illustrations of these rebound mechanisms. Some types of rebound relate to conflicts and the need for trade-offs between different types of biodiversity, or between certain types of biodiversity and certain ecosystem services. We do not claim any definite results, but merely offer a starting point for research. We hypothesize that including rebound effects in the analysis of biodiversity will alter policy conclusions.

Which particular research approach is needed to study these various types of rebound? It will require close collaboration between natural and social scientists, a good understanding of the various direct and indirect (fundamental) causes of biodiversity loss, a clear choice of relevant biodiversity measures, and a translation of past research in clear conclusions about connections between biodiversity, ecosystem functioning and services. One very likely will need to use systems models for concrete cases to assess all the unwanted and avoidable rebound effects of particular policies of biodiversity protection. In addition, it is useful to examine which policies, and in which settings, are functioning relatively well in terms of generating a low rebound and thus having a high effectiveness. Against this background, it would seem useful to connect the instruments in Table 2 to the rebound typology. This involves further conceptual thinking along the lines as sketched here, as well as studying different cases and ecosystems to understand the relevance of particular combinations of instruments and contextual factors for the magnitude of rebound.

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