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Final report on relationships between biodiversity, ecosystem services and values in case studies

PART I: Literature review of the relationships between biodiversity, ecosystem services and values

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Literature review of the relationships between biodiversity, ecosystem services and values

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1. Introduction

Although ecosystem services are generated from myriad interactions occurring in complex systems, improving understanding of at least some of the key relationships between biodiversity and service provision will help guide effective arguments for biodiversity conservation. Considerable research has been undertaken on the contribution of biodiversity to selected ecosystem processes, and scientists have begun to extend such work to include ecosystem services. Nevertheless, there are still numerous uncertainties and gaps in scientific knowledge on the relationship between biodiversity and ecosystem services. It is unclear how these uncertainties affect decision-makers' perceptions of the value of biodiversity and their subsequent decisions related to biodiversity conservation.

BESAFE has advanced the state-of-the-art in this area by undertaking a systematic review of the contribution of relevant aspects of biodiversity (e.g. key species abundance, functional traits) to the delivery of specific ecosystem services and their associated economic and social values. Existing methods for evaluating the relationship between biodiversity and ecosystem services and the implications for the valuation of biodiversity have been reviewed and their relevance for the BESAFE case studies assessed. These include approaches which aim to determine the importance of biodiversity for ecosystem service delivery, such as the identification of Ecosystem Service Providers, Service Providing Units and functional traits which can be linked to service provision (Section 1.1). Furthermore, the project has reviewed methodological approaches relevant for the valuation of different types of ecosystem services (Section 1.2). The review focuses on scientific literature to gather the best available evidence, but many of these papers result from previous EU-funded projects, such as the FP6 RUBICODE project (Section 2). The results have been analysed to assess general patterns of linkages across ecosystem services (Section 3.1). A typology of biodiversity – ecosystem service relationships has also been created to structure existing evidence using network analysis (Section 3.2). Full results of the literature searches for each ecosystem service are presented in the Annex to the main report.

1.1 Methods for studying the relationship between biodiversity and ecosystem services

Over two decades of research has demonstrated a clear link between biodiversity and key ecosystem functions such as nutrient cycling or biomass accumulation (Cardinale et al., 2012). More recently, this has been complemented by a growing body of research exploring the role of biodiversity in the provision of ecosystem services of value to humans, such as climate regulation and water purification.

As this research field has evolved, awareness has grown that biodiversity cannot be described simply by the number of species present in an ecosystem (species richness). The number, abundance, composition and spatial patterns of species, genotypes, populations and functional groups are all important aspects of biodiversity (Díaz et al., 2007a). Research has revealed that the mix of functional traits in an ecosystem is often a better predictor of ecosystem function than species richness. For example, a plant community with a diverse range of root depths and canopy heights may be more efficient at capturing light and water, and therefore more productive, than one with the same number of species but less functional diversity: this is known as niche complementarity (Cardinale et al., 2012).

The analysis of functional traits has helped in revealing and understanding the nature of the relationship between biodiversity and ecosystem functioning (Hooper et al., 2005; Diaz et al., 2007a; Kremen et al., 2007; De Bello et al., 2010; Lavorel & Grigulis, 2012; Dias et al., 2013; Lavorel, 2013, Luck et al., 2012). However, approaches to quantifying the contribution of biodiversity and ecological functioning to ecosystem service outputs have been slower to develop (Haines-Young & Potschin, 2009; Carpenter et al., 2009). Two recent reviews show that considerable progress has been made, but important knowledge gaps and uncertainties persist (Cardinale et al., 2012; Balnavera et al., 2014). For example, experimental studies on the impact of biodiversity on ecosystem functions often produce valuable data, but they fail to reproduce the complex species assemblages present in natural ecosystems. Studies linking biodiversity to ecosystem services, on the other hand, typically observe larger scale natural ecosystems, but suffer from the presence of confounding factors such as climate or soil type. These reviews identify a need to develop models and analytical approaches that can link the analysis of functional traits more closely to the delivery of ecosystem services. The BESAFE project addresses this need through a systematic literature review of the way in which ecosystem services depend on different aspects of biodiversity, including a detailed breakdown of the relevant functional traits and ecosystem service providers for each service reviewed.

Trait-based approaches

Extensive research on plant traits has shown that trait-based approaches may help elucidate the complexity of mechanisms in the field (Lavorel, 2013). Given their effects on underlying ecosystem services, several studies have used information on functional traits to quantify ecosystem service delivery (Kremen, 2005; De Bello et al. 2010; Diaz et al., 2011). Knowledge on associations and trade-offs between plant traits is well established, but the study of the consequences of these for ecosystem functioning and the resulting services is less developed (Lavorel & Grigulis, 2012). De Bello et al. (2010) suggested that the multiple associations between traits and services across different trophic levels result in what they call ***trait-service clusters***. Their review groups well-documented trait-service associations into clusters of ecologically-related services (Figure 1.1). They propose that this approach will allow for the assessment of combined biotic effects on the simultaneous delivery of multiple services. Trait-service clusters would potentially serve to manage trade-offs of services associated with traits within a trophic level and across multiple trophic levels (Lavorel, 2013), as well as facilitating the monitoring of clusters of services at different spatial scales.

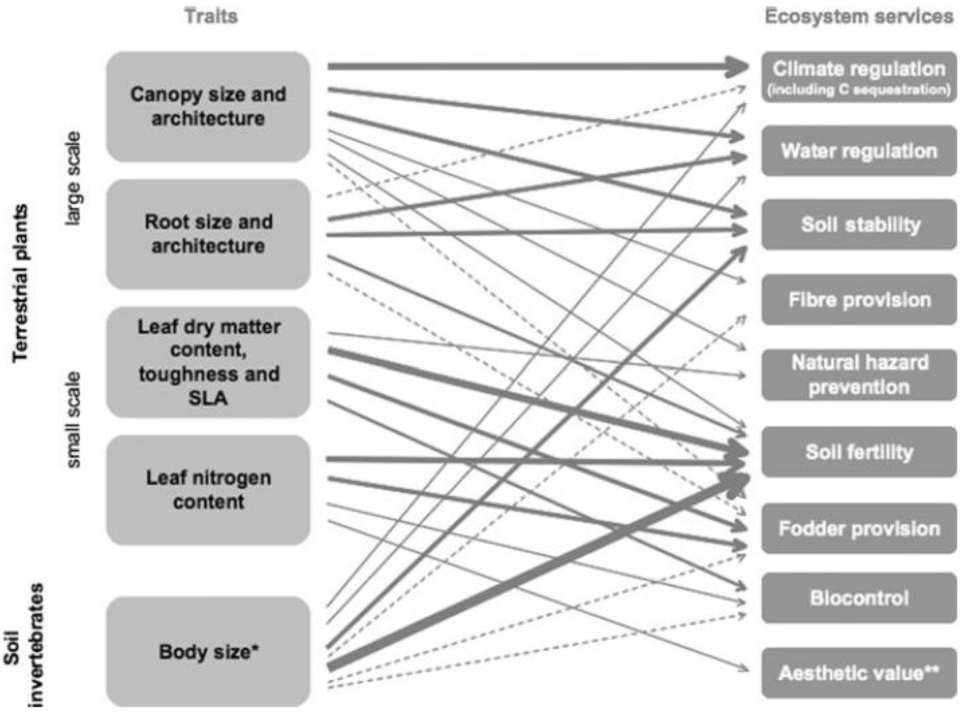


Figure 1.1: *Most commonly reported plant and invertebrate traits and their involvement in multiple ecosystem service delivery. Larger arrow thickness for a given trait-service relationship indicates that more statistically significant associations were found in the literature (extracted from de Bello et al., 2010).*

Until recently, most trait-based research has focused on the effects of plant traits on primary production (Lavorel, 2013). There is a need to investigate a wider range of ecosystems, services and organisms. Luck et al. (2012) sought to develop a systematic framework for selecting traits that are important either for providing an ecosystem service or for responding to environmental change, and applied their approach to vertebrates (birds). They suggested that although the use of statistical methods for selecting traits may be more objective, it may also exclude ecologically relevant traits. Therefore, they call for vertebrate ecologists to develop more coherent and systematic trait-based approaches.

In one of the few practical field studies to date, Conti & Díaz (2013) analysed the links between functional diversity and ecosystem properties. In their study of semi-arid forest ecosystems, they tested the notion of functional diversity as a major driver of ecosystem output (in this case, carbon accumulation). They found that all three components of functional diversity (dominant trait values, variety of trait values, and the presence of particular species in the community) contributed to explain carbon storage at the ecosystem level. Hooper et al. (2005) also assessed the effects of variations in functional traits, functional types and functional diversity on biodiversity, concluding that functional characteristics can strongly influence ecosystem properties and output of ecosystem services. Furthermore, Dias et al. (2013) offer a novel methodological framework for designing experiments that decouple community-weighted mean and functional diversity (quantified by functional richness, evenness and divergence) on ecosystem processes, to assess the relative importance of each of these two community functional components on ecosystem services.

Lavorel & Grigulis (2012) suggested a novel approach that focuses on the direct interactions between ecosystem services resulting from fundamental functional mechanisms or trade-offs. Using existing knowledge on plant traits, they present a conceptual framework linking environmental change, ecosystem functioning and changes in ecosystem services. They proposed scaling up trait relationships at the individual plant level to ecosystem properties and, subsequently, to ecosystem services. To test the hypotheses on the mechanisms underpinning or constraining the ability of ecosystems to provide multiple services, they suggest the use of structural equation modelling (SEM). SEM facilitates testing the significance of the overall model (the relationships between the components of an ecosystem), as well as specific parameters of the model (e.g. the strength or direction of the interactions between those components).

Existing quantitative models of ecosystem services built from plant traits and environmental variables (Díaz et al. 2007b), to quantify and project ecosystem services, use unique values for trait means or divergence and abiotic factors within a given land use. Lavorel et al. (2011) highlighted that finer scale biotic and abiotic variation within each land use need to be considered for a landscape analysis. They proposed an approach for the analysis, mapping and understanding of multiple ecosystem service delivery in landscapes. By combining spatially-explicit single ecosystem service models based on plant traits and abiotic characteristics, they identify ‘hot’ and ‘cold’ spots of multiple ecosystem service delivery, and the land use and biotic determinants of such distributions. ‘Hot’ spots represent areas of high delivery of multiple services, while ‘cold’ spots represent low delivery across services.

Ecosystem Service Providers and Service Providing Units

Luck et al. (2003) highlighted that species populations are the fundamental unit in the provision of ecosystem services, and that there is a need to understand the links between population dynamics and service output. They offer the concept of a Service Providing Unit (SPU), in which instead of defining a population or organism along geographic or genetic lines, they suggest it can be done in terms of the services it generates at a particular scale. It is somewhat of an ‘ecological footprint’ of the biophysical mechanisms that give rise to the service (Haines-Young & Potschin, 2009).

Kremen (2005) extended the SPU concept and proposed identifying key Ecosystem Service Providers (ESP). Kremen (2005) also suggested defining ESPs in terms of their functional traits and how the dynamics of functional groups of species may impact service provision. This was extended by Kremen et al. (2007) into a framework for understanding the impact of broadscale interactions between the distribution of resources, traits and land use change on service delivery.

The SPU and ESP concepts were combined by Luck et al. (2009) into the ***SPU-ESP continuum*** to show how the service-provider concept can be applied at the population, functional group and community levels. This produced a more nested approach to the understanding of service functions and processes and offered a detailed categorisation of outputs and their relationship to human well-being. By using examples from existing literature, they provided a classification specifying the type of ecosystems concerned, the ecological unit providing the service or SPU, its attributes and a response measure to describe the relationship between the components of biodiversity and the level of service provision (Table 1.1). Furthermore, Kontogianni et al. (2010) used the SPU concept in a spatial and population approach as a way to systematically quantify the key components of nature that provide services and link them with measureable outcomes for human well-being.

To understand and quantify how service outputs vary across landscapes and their process-related use, Syrbe & Walz (2012) extended the SPU concept to include place-based assessments and structure metrics. They focus on three concepts: Service Providing Areas (SPAs) are the areal basis for service provision; Service Benefiting Areas (SBAs) to determine where the services are needed (complementary to service generating areas); and the connecting space between providing and benefiting areas, or Service Connecting Areas (SCAs). By using landscape metrics as indicators of landscape services, they sought to estimate and evaluate landscape units and indicate spatial trade-offs between services.

Table 1.1: *Illustration of the SPU-ESP continuum approach using examples from the literature. Extracted from Luck et al. (2009).*

Service	Ecosystem [level of organization]	Service provider [level of organization]	Service-provider characteristics	Supporting element	Response measure	Relationship
Biological control	Agroecosystem [apple orchards]	Great tit [population]	Density of breeding pairs[a]	Density of nest boxes[b]	Caterpillar damage to apples	Control vs. treatment
Biological control	Agroecosystem [coffee plantation]	Azteca ant	Green scale [population]	Activity level[c]	Shade trees[d] Number of scale[e]	Time to removal[f] Linear[g]
Biological control	Agroecosystem [rice fields]	Egg parasitoids [functional group]	Abundance of predators and parasitoids[h]	Presence of parasitoid and absence of predator	Leaf and plant-hopper abundance	Control under negative impact of predators on parasitoids[i]
Pollination	Agroecosystem [watermelon crops]	Native bees[j] [functional group]	Functional group, species-specific visitation rates and efficiencies[k]	Upland habitat[l]	Pollen deposition[m]	Saturating, exponential increasing[n]
Pollination	Agroecosystem [coffee plantation]	Native and exotic bees [functional group]	Functional group dynamics[o]	Tropical forest[p]	Seed mass, fruit set, peaberry frequency, pollen deposition (number of visits per flower), bee species richness	Comparative[q]
Pollination	Agroecosystem [atemoya crops]	Nitidulid beetles[r] [functional group]	Functional group dynamics[s]	Rainforest	Beetle species richness[t]	Exponential decay[u]
Pollination	Agroecosystem [canola fields]	Wild bees [functional group]	Functional group dynamics[v]	Uncultivated land[w]	Bee abundance, seed set	Linear[x] Saturating[y]
Waste decomposition	Agroecosystem [rice fields]	Mallard [population]	Population density[z]		Residual surface straw[aa], structure of surface straw[ab], chemical composition[ac]	Control vs. treatment Control vs. treatment Control vs. treatment
Water regulation	Forest/terrestrial	Terrestrial vegetation [community]	Soil-slope-vegetation complex	Water regulation, hydroelectricity generation		Comparative[ad]
Water filtration	Freshwater	Forest [community]	Forest cover[ae]		Water and sediment nutrients	Various
Seed dispersal	Oak forest	Eurasian jay [population]	Population abundance[af]	Oak and coniferous forest[ag]	Oak saplings	n/a
Seed dispersal	Tropical forest	Insular flying fox [population]	Flying fox abundance index[ah] = 0.77 to 0.81		Chewed diaspores[ai]	Threshold

Discussion

Ecosystem services are not homogeneous or static across landscapes and seascapes (Fisher et al., 2009). Balvanera et al. (2006) suggested that more diverse systems have more temporal stability, as well as greater resistance to external forces such as alien species. In their meta-analysis of existing studies, they also found that the effect of increasing biodiversity on 'consumption stability' (i.e. the result of variations in biodiversity at one trophic level on the next) was strongest at the community rather than the whole ecosystem level. This suggests that ecosystems may sometimes have the capacity to buffer disturbances at one level and minimise impacts. However, the buffering effects can be specific and are not generalisable.

Menzie et al. (2012) called for a holistic assessment of ecosystem services. They argued that an accurate assessment is based on the holistic evaluation of ecosystems founded on solid expertise in ecosystem dynamics. They explored the complexity of ecosystem services and highlighted that a reductionist approach often fails to capture ecological dynamics that are vital to the functioning and provision of services. They concluded that developing an understanding of the interconnections and complexity of ecosystems and the associated trade-offs of services in time and space is key, although likely the most challenging aspect.

Thus, even though several studies and meta-analyses have furthered knowledge of the role of biodiversity in the supply of ecosystem services (Hooper et al., 2005; Balvanera et al., 2006; Nelson et al., 2008; Luck et al., 2009; Harrison et al., 2010; von Haaren, 2012; Cardinale et al. 2012; Bastian, 2013), the complexity of ecosystem functioning still poses uncertainty about the role of many species and other components of biodiversity, especially when coupled with social-ecological systems. The review described in Section 2.1 builds on the current state-of-the-art frameworks and concepts for linking biodiversity and ecosystem service provision by combining the identification of ESPs and evidence of their key attributes or traits for service delivery for 11 ecosystem services. This aims to first reveal the complexity of the interconnections and second explore the possibility of reducing this complexity to reveal different typologies of relationships through the network analysis reported in Section 2.2.

1.2 Methods for studying the relationship between ecosystem services and values

In this part of the report we shift our attention from the biophysical aspects of the services provided by ecosystems to the contribution these services make to human well-being, e.g. benefits and associated values. There are many different ways of defining well-being and the linkages between well-being and ecosystems. In this report, however, we restrict ourselves to the approaches aimed at quantifying values or at least ranking values. This biases the review towards concepts developed in economics, but we include evaluation methodologies which do not use monetary values as the basis for quantification or ranking.

Economic valuation builds on an extensive literature developed over the past 30 years attempting to identify and quantify values of environmental goods and services when market transactions and associated prices do not exist to infer values. This highlights the importance of distinguishing between "value" and "price". Often ecosystem goods and services are priced at zero, even though their value in terms of the impact on human well-being is non-negligible. Outdoor recreation is often used to illustrate this point. While the access to recreational opportunities is often free, and the price therefore zero, people choose to spend their time and expenditure on travel in order to benefit from the service. This fact is a simple manifestation of the discrepancy between price and value, and environmental valuation research has developed from the need to demonstrate and quantify the values from ecosystems in order to account for such value in environmental policy development and evaluation. Early literature on this topic tended to focus on the development of robust approaches to value individual services such as, for example, recreational opportunities from individual sites.

However, research is now tending to focus more on the more complex task of valuing multiple and interacting sets of ecosystem services from entire landscapes.

Typology of values

It has become customary to distinguish between use and non-use values as a first differentiation between the different sorts of economic values arising from ecosystems. Use value is a measure of the relative satisfaction, happiness, pleasure or fulfilment of preferences derived from the consumption of a good or service (Naidoo & Adamowicz, 2005). Table 1.2 below provides definitions for the different types of use values. Non-use values (also known as passive values) are values that are not associated with actual use of a good or service (Brouwer et al., 1999) (Table 1.3).

In the ecosystem service valuation literature the focus has so far mainly been on the first three types of use values: consumptive and non-consumptive use value and indirect use value (Table 1.2); and existence value (Table 1.3). While it is often argued that insurance value (Table 1.2) is an essential ecosystem service (Baumgartner & Quaas, 2008), the challenges involved in estimating such values have so far proved too complex for most case studies. In the following section we therefore focus on the types of values and methodologies applicable to the majority of ecosystem services valuations.

Table 1.2: *Different aspects of use value.*

Direct use value (consumptive)	The value derived from the actual consumptive use of a good or service. For example, the harvesting of timber is a direct use consumptive value of a forest.
Direct use value (non-consumptive)	The value derived from the actual non-consumptive use of a good or service, e.g. recreation is a direct non-consumptive use value of a forest.
Indirect use value	The indirect (non-consumptive) value that is derived from ecosystems, such as the role of ecosystems in maintaining clean water supplies.
Insurance value	The value derived from the reduction of risk to which an individual or society is exposed, e.g. biodiversity may add to resilience of the provision of ecosystem services, which would be an insurance value of biodiversity.
Option value	We are currently unaware of many aspects of biodiversity. Option values refer to the potential future value that may be derived from aspects of biodiversity we are yet ignorant about.

Table 1.3: *Different aspects of non-use value.*

Bequest values	The value derived from the satisfaction gained from preserving a natural environment for future generations.
Existence value	The value conferred by the existence of an organism, or organisms, independent of their utility to humans. It is the value that people place on simply knowing that something exists, even if they will never see it or use it.
Intrinsic value	This is based on the object being valued for itself rather than because it serves a valued purpose.

Methodologies for ecosystem service valuation

Various methods have been applied to value ecosystem services. The different methods and their applications are summarized in Table 1.4.

Table 1.4: *Valuation methods applied to ecosystem services (ES). Source: Bateman et al. (2011).*

Valuation method	**Use type**	**Applications**	**ES valued**
Adjusted market prices: Market prices adjusted for distortions such as taxes, subsidies and non-competitive practices.	Direct use value (consumptive)	Provisioning services	Crops, livestock, woodland, etc.
Production function approach: Estimation of production functions to isolate the effect of ecosystem services as inputs to the production process.	Indirect use values	Regulating services	Maintenance of beneficial species, productive ecosystems and biodiversity; storm protection; flood mitigation; air quality; peace and quiet; workplace risk.
Damage cost avoided: Calculates the costs which are avoided by not allowing ecosystem services to degrade.	Indirect use values	Regulating services	Drainage and natural irrigation; storm protection; flood mitigation.
Averting behaviour: Examination of expenditures to avoid damage.	Indirect use values	Regulating services	Pollution control and detoxification.
Revealed preference methods: Examine the expenditure made on goods related to ecosystem (e.g. travel costs for recreation; hedonic (typically property) prices in low noise areas).	Direct use value	Provision of space often classified as cultural services	Maintenance of beneficial species, productive ecosystems and biodiversity; storm protection; flood mitigation; air quality; peace and quiet; workplace risk.
Stated preference methods: Uses surveys to ask individuals to make choices between different levels of environmental goods at different prices to reveal their willingness to pay for those goods.	Use and non-use value	Applications to most types of ecosystem services	Water quality, species conservation, flood prevention, air quality, peace and quiet.

Valuation method	Use type	Applications	ES valued
Deliberative valuation: Uses focus groups to ask communities to make choices between different levels of provision of environmental goods.	Use and non-use value	Applications to most types of ecosystem services	Conservation priorities.
Expert valuation	Use and non-use value	Applications to most types of ecosystem services	Mainly applicable to issues where scientific knowledge Is essential for judgement of alternatives.
Multi-criteria evaluation: Mainly applicable when many attributes are affected from the policy action to be valued.	Use and non-use value	Applications to most types of ecosystem services	Conservation and environmental policy priorities.
Benefit Transfers: Based on transferring values from similar policy settings in different locations.	Use and non-use value	Applications to most types of ecosystem services	Often used in hedonic valuation studies.

The value of an ecosystem service should reflect the marginal utility to society of the service. For services that are traded, e.g. supply of crops, timber and in some cases water, market prices exist. If consumers are assumed to maximize their utility - a usual assumption in economics – the market prices of goods and services can be used as indicators of their marginal utility to consumers. This is the basic principle in economic valuation of goods and services, including the different ecosystem services (Johannsson, 1993; Freeman, 2003; Bateman et al., 2011).

Thus, market prices can be used to value provisioning services that are traded, as shown in the first line of table 1.4. The market prices reflect the direct consumptive use value of the services consumed by households or enterprises.

The production function approach is also based on market prices, but it is used to value ecosystem services that are not directly traded, i.e. they do not have a market price (see Hanley and Barbier, 2009). This includes a number of regulating services that have an indirect use value, such as flood mitigation. They contribute to maintain the production, consumption or welfare level in society. This contribution is estimated on the basis of production functions that describe how the market value of a good is determined by a number of inputs including regulating ecosystem services. The value of each regulating services is determined by its effect on the total market value of the good.

However, the indirect use value of regulating services can also be estimated by two other market price based valuation methods. Firstly, the damage cost avoided because of regulating services can be used to value services that protect against damage (e.g. flood mitigation). This valuation method is related to the production function approach, but in many cases the damage costs avoided are estimated without the use of a production function (see Bateman et al., 2011). The second method especially related to regulating services is the averting behaviour approach, where the value of the service is estimated on the basis of the costs of alternative protection against damages (e.g. the cost of building flood protection infrastructure). So, if a regulating service is eliminated and the current production or welfare level is to be sustained then alternative protection measures have to be established. The cost of these measures is an indicator of the value of the regulating service, assuming that the welfare level should be sustained.

The last category of market price based valuation methods are the so-called revealed preference methods (see Braden & Kolstad, 1991; Randall, 1994 and Day et al., 2007). These methods include travel cost methods and hedonic methods which are mostly used to evaluate the use value of a number of cultural services, but also the value of noise protection, air quality changes and workplace risks. The basic idea of revealed preference methods is that consumers indirectly reveal their willingness to pay for an ecosystem service by the expenses they pay for traded goods that make it possible to get the ecosystem service. For example, households pay transport expenses and expend time to get to a nature area and these costs are an indicator of the value of the cultural services they get from the area. Similarly, housing prices may depend on how close the houses are to nature areas, and therefore the differences in housing prices can be used to estimate households' willingness to pay for cultural services supplied by the nature areas. This hedonic housing price method is also often used to estimate willingness to pay for reduced noise level or air quality improvements.

Revealed preference methods indirectly reveal households willingness to pay on the basis of existing market prices for traded goods and services. In contrast, stated preference methods reveal willingness to pay by asking individuals directly about their willingness to pay for different environmental goods or by asking them to make choices between different levels of ecosystem services that involve different levels of costs to supply (Mitchell & Carson, 1989; Hanley & Barbier, 2009). These methods can be used to estimate use value as well as non-use value for almost all types of ecosystem services. However, the values estimated by use of these methods are also in many cases surrounded by a higher degree of uncertainty than the market based methods.

Deliberative valuation is very closely related to stated preference methods. Instead of asking individuals to make choices between different levels of ecosystem services, focus groups representative of different communities are asked to make the choices. The method can also be used to estimate use and non-use values of most types of ecosystem services.

The last three valuation methods shown in Table 1.4 are not economic valuation methods as such. They do not necessarily lead to an economic value of the ecosystem services. Thus, expert valuation includes a number of methods where experts are asked to evaluate the relative value or importance to society of different ecosystem services. The answers can have very different forms, including different quantitative measures of importance and qualitative evaluations. Multi-criteria evaluation includes a large range of quantitative as well as qualitative evaluation methods (Zeliny, 1982; Janssen, 1996). The evaluators also vary from single individuals to groups of decision-makers or experts. Each multi-criteria method represents a technique by which evaluators are asked to make quantitative or qualitative weighting of different criteria including supply of specific ecosystem services. Finally, benefit transfer includes techniques for transferring the valuation results from one or several earlier completed analyses to a new similar valuation setting (Brouwer, 2000; Bateman et al., 2009). Until now, benefit transfer methods have mostly been used to transfer economic valuation results, but they are also relevant in relation to results of expert valuation and multi-criteria valuation. All methods can be used in estimating use as well as non-use value of most types of ecosystem services.

2. Methods

2.1 Literature review on the relationships between biodiversity, ecosystem services and values

Eleven ecosystem services were included in the review, chosen to represent the key groups of services from the classifications of the Millennium Ecosystem Assessment (MA) and the Common International Classification of Ecosystem Services (CICES) (Table 2.1).

Table 2.1: *The 11 ecosystem services included in the literature review and their association with the MA and CICES classifications.*

Ecosystem service	MA classification	CICES division/group
Provisioning services:		
Timber production	Fibre (timber and wood fuel)	Materials/Biomass (timber)
Freshwater fishing	Food (capture fisheries and aquaculture)	Nutrition/Biomass (freshwater fish and marine fish)
Freshwater provision (quantity)	Freshwater	Nutrition / Water
Regulating services:		
Water purification (quality)	Water purification /waste treatment	Mediation of waste, toxics and other nuisances / Mediation by biotic and ecosystems
Water flow regulation (flood protection)	Natural hazard regulation	Mediation of flows / Liquid flows
Mass flow regulation (erosion protection)	Erosion regulation	Mediation of flows / Mass flows
Atmospheric regulation (carbon sequestration)	Climate regulation	Maintenance of physical, chemical, biological conditions / Atmospheric composition and climate regulation
Pest regulation	Biological control	Maintenance of physical, chemical, biological conditions / Pest and disease control
Pollination	Pollination	Maintenance of physical, chemical, biological conditions / Lifecycle maintenance, habitat & gene pool protection (pollination)
Cultural services:		
Recreation (species-based)	Recreation and ecotourism	Physical and intellectual interactions with biota, ecosystems, and land-/seascapes / Physical and experiential interactions
Landscape aesthetics	Aesthetic values	Physical and intellectual interactions with biota, ecosystems, and land-/seascapes / Intellectual and representative interactions (aesthetic)

In order to review and consolidate existing research on the linkages between biodiversity and these 11 ecosystem services, a literature search was conducted between July 2012 and August 2013 using Web of Science or Web of Knowledge. The primary aim of focusing on peer-reviewed academic literature was to find the best available knowledge reported by the scientific community. A systematic methodology was adopted in order to ensure that a rigorous and repeatable method was applied to each ecosystem service. The method consisted of three stages: (i) the generation of keywords, (ii) a systematic search, and (iii) extraction of the data.

The literature review was conducted in two parts, which are described in the following sections:

1. Review of links between biodiversity and ecosystem services
2. Review of links between ecosystem services and values

Review of linkages between biodiversity and ecosystem services

Keywords were generated based on the results of a pilot test (conducted from February to April 2012) which showed that 'ecosystem services' is a relatively new term and, hence, only using this term in a literature search is likely to miss relevant papers. Thus, keywords specific to each ecosystem service were selected, accompanied by appropriate biodiversity terms which could be related to the given ecosystem service. We included both synonyms (i.e. the service) and antonyms (i.e. the disservice) in the search terms to enable negative, as well as positive, impacts of biodiversity on ecosystem service supply to be captured. Additional service-related terms were used if necessary to refine results when large numbers of papers were found for the initial search terms (see Appendix 1 for a full list of search terms).

The objective was to find 50 relevant papers for each service in order to have a wide range of relationships and studies. For many ecosystem services, however, the number of relevant results using the above methodology was too few. In these cases, additional intelligent search approaches were utilised. These included: (i) searching the reference lists of relevant articles for secondary references which may be of interest (termed snowballing); and (ii) searching papers that have cited the relevant papers (termed reverse snowballing). In total 50 papers were found for all services except timber production and freshwater fishing, where only 35 and 45 papers could be found, respectively, after applying all search approaches. This reflects the limited number of studies that have examined how biodiversity influences timber and fish production, despite the large amount of literature on the impact of best management and/or harvesting practices on wood yield/quality and the impact of fishing on fish attributes.

Data from the 530 papers were extracted into a database, with parameters covering:

(i) the ecosystem service;
(ii) the reference;
(iii) the location of the study;
(iv) the spatial scale;
(v) the temporal scale;
(vi) the ESP,
(vii) the important biotic attributes of the ESP; and
(viii) abiotic factors which affect service delivery.

ESPs were categorised into seven groups: single population; two or more populations; single functional group; two or more functional groups; dominant community; single community/habitat; and two or more communities/habitats (see Appendix 2 for definitions of these terms).

The biotic and abiotic attributes were determined from the pilot test which identified those attributes cited as being important within a selection of papers across the 11 ecosystem services. It also took into account biodiversity-related indices from other studies (e.g. Feerst, 2006; Feerst et al., 2010; Hooper et al., 2005). The final list of biodiversity attributes (see Appendix 2 for full definitions) included:

- species attributes (presence of a specific species type, species abundance, species richness, species population diversity, species size or weight, population growth rate, mortality rate, natality rate, life span/longevity);
- functional group attributes (presence of a specific functional group type, abundance of a specific functional group, functional richness, functional diversity, flower-visiting behavioural traits, predator behavioural traits);

- community/habitat attributes (presence of a specific community/habitat type, community/habitat area, community/habitat structure, primary productivity, aboveground biomass, belowground biomass, sapwood amount, stem density, wood density, successional stage, habitat/community/stand age, litter/crop residue quality, leaf N content).

The final list of abiotic factors included: temperature, precipitation, evaporation, wind, snow, soil, geology, water availability, water quality, nutrient availability (soil minerals) and slope (angle, aspect).

The direction of each relationship between the biodiversity attributes of the ESP and the ecosystem service was also classified as being predominantly positive, negative or unclear (i.e. both positive and negative, or authors unsure of the relationship). Where quantitative information on the relationship was provided in the literature, this was also extracted into the database. Furthermore, it was noted whether the paper discussed the ESP as being an ecosystem service antagoniser (ESA[1]), and where biodiversity could also have a negative effect on the ecosystem service concerned.

Finally, all papers were evaluated for the strength of the presented findings. This was based on five questions:

(i) is the evidence qualitative, quantitative or both?
(ii) is the evidence based on single or multiple observations?
(iii) is the evidence direct or indirect (i.e. through a surrogate)?
(iv) is the link explicitly mentioned or only inferred; and
(v) is the evidence based on empirical data, modelled information, or both?

Responses to these questions were then combined with equal weighting into a five class qualitative scale ranging from 1 (very weak) to 5 (very strong). It should be noted that this scale does not take account of the actual strength of the relationship identified in the paper, i.e. the correlation coefficient and significance level.

Review of linkages between ecosystem services and values

The literature review of the values of ecosystem services included the same 11 ecosystem services and followed the same procedure as the review of relationships between biodiversity and ecosystem services described above. The values review also started by generating keywords, followed by systematic review in Web of Science and extraction of data.

The keywords chosen included, in addition to biodiversity and ecosystem services, terms such as economic value, valuation, social value, multi criteria, evaluation, economic* and different expressions for each of the ecosystem services. The focus was on finding studies that were relevant in valuing each ecosystem service even if the studies did not use the ecosystem service concept or have any explicit reference to biodiversity. The number of studies reviewed varied between the different services (see Section 3.1.4).

Information was extracted on the following parameters:

(i) the ecosystem service;
(ii) the reference;

[1] ESAs are defined as the populations, functional groups or communities which disrupt the provision of other ecosystem services and the functional relationships between them and ESPs (see Harrington et al., 2010 for a more detailed definition).

(iii) the ecosystem service provider;
(iv) biodiversity linkage(s) on which valuation is based and, if so, how the link is defined and used;
(v) ecosystem services which are or are not included in valuation;
(vi) quantified information on the nature of the ecosystem service and the valuation: units of measurement used for ecosystem service delivery (valued as a bundle or a single service) and valuation;
(vii) types of value measured;
(viii) valuation approach used;
(ix) ecosystem service beneficiaries;
(x) temporal, spatial and locational setting of the valuation;
(xi) evidence: reliability of the results; and
(xii) policy context and relevance of valuation as claimed in the publication.

The ecosystem service provider and location is important for the interpretation and use of the valuation results. Many valuation studies are related to a very specific area, e.g. a national park, and therefore the results are only relevant for that area or very similar areas. Valuation results can only in very few cases be used as a general value of an ecosystem service.

To ensure correct interpretation of the results, in some cases it may be important to bring out which ecosystem services are or are not included in the valuation study. For example, freshwater areas supply regulating services related to the decomposition of several nutrients, and therefore it is important to know which nutrients are and are not included in valuation studies of these regulating services. Similarly ecosystems supply a number of different recreational services and valuation studies may vary with regard to which services are included.

For further use of the results of valuation studies, e.g. in cost benefit analysis, information on the nature of the ecosystem service and the units of measurement used for ecosystem service delivery is necessary. For example, is the value of nutrient decomposition related to the amount of nutrients discharged to the freshwater areas, or to the estimated amount removed? It is also important to know if the services are valued as a bundle or as single services.

As discussed in Section 1.2, values of ecosystem services include several types of use and non-use values. Therefore, the review of valuation studies also records which of these values are being measured. This shows which types of values are more frequently valued and which are difficult to measure.

Very central to the review is information about which valuation approaches have been used in relation to each of the eleven ecosystem services reviewed. This information is important because it indicates which valuation approaches can be used in relation to each ecosystem service and which approaches are used most often. However, the reason for one valuation approach being used more often than others is not always clear: it could be the most satisfactory theoretical approach, or it could be the easiest to use in practice.

The review also includes information about who are the beneficiaries of the ecosystem services. Of course, it can be argued that all services represent benefits to society in general, but in many cases it is only a part of society – certain individuals or enterprises - that is the real beneficiary of a service. The information throws light on possible distributional questions related to the supply of a service. As mentioned above, most valuation studies are carried out in a specific temporal, spatial and locational setting, and do not necessarily provide values of each ecosystem service that can be applied more generally. Recording information about these circumstances makes it possible to assess in which other contexts the results can be used.

Evaluation of the reliability of the results is an important aspect of the review. The evaluation may show how reliable each the valuation approaches is in general. It can also be used as basis for further evaluation of how difficult and resource-demanding it is to make reliable valuation studies. Finally, the evaluations are important in relation to use of the valuation results in a policy context. Only reliable results can be expected to be used as basis for policy recommendations.

The policy context and relevance of the valuation study as claimed by the authors is the last subject of the review. This information is collected to evaluate to what degree valuation of ecosystem services is meant to contribute to policy formulation.

2.2 Mapping the empirical evidence of relationships between biodiversity, ecosystem services and values using network analysis

The results of the two literature reviews were synthesised into a series of network diagrams, one for each of the ecosystem services studied. This enables clear visualisation of the typology that has been created, based on the linkages between biodiversity attributes, ecosystem services and values. The sections below describe how the data were extracted and processed from each of the two reviews, and how the diagrams were created.

Biophysical review

For the biophysical review, the collected data were compiled and three variables summarising the linkages between biodiversity attributes and each ecosystem service were calculated:

(i) the level of support given – this reflects the number of papers providing the same evidence for a particular linkage, as a proportion of the total papers analysed for that service;
(ii) the strength of evidence – an average ranking of the strength of evidence for the particular linkage ranging from 1 (very weak) to 5 (very strong) and
(iii) the direction of the evidence for the relationship – predominantly positive, negative, or unclear (i.e. uncertain). This was calculated as follows:

$$Predominant\ Direction = \frac{\sum\begin{pmatrix} strength\ of\ evidence\ for \\ all\ positive\ relationships \end{pmatrix} - \sum\begin{pmatrix} strength\ of\ evidence\ for \\ all\ negative\ relationships \end{pmatrix}}{Total\ number\ of\ papers\ showing\ evidence\ of\ relationship}$$

The parameter "strength of evidence for positive or negative relationships" was chosen rather than raw counts of direction to ensure that those records offering weak evidence had a smaller influence on the overall direction than those identified as having strong evidence.

Following this assignment of trend, the trend direction was classified into four classes:

- No trend data/ no relationship between ESP and biodiversity attribute;
- Neutral relationships – those classed as "Uncertain" or "Both" and examples where predominant direction is exactly zero (equal number of records of the same strength on both sides) or all records are assigned to be either "uncertain" or "both" with none to either positive or negative;
- Positive (where Predominant Direction > 0); and
- Negative (where Predominant Direction < 0).

Note that the trend direction was not recorded for relationships between the ecosystem service and ESPs and between the ESPs and abiotic factors. Hence, linkages for these relationships were incorporated into the networks based on the number of papers citing a particular linkage.

Valuation review

For the review of values, the collected data were compiled and three variables summarising the linkages between the ecosystem service, the ecosystem service beneficiary and the method used to obtain the data were calculated.

Beneficiaries were classified according to the following typology:

(i) households or individuals;
(ii) firms;
(iii) communities or societies;
(iv) groups of stakeholders; and
(v) "other" (i.e. groups or individuals not covered by the previous four classes).

The typology for the valuation methods identified by the literature search was based on nine classes:

(i) stated preference;
(ii) revealed preference;
(iii) market price;
(iv) avoided costs;
(v) benefit transfers and reviews;
(vi) multi criteria analysis;
(vii) the production function approach;
(viii) biophysical ranking; and
(ix) non-economic value ranking.

A final class, "other", was used to capture methods not included in the nine above; in practice, these were mostly simulation modelling studies.

A variable for the level of support (evidence) for each linkage was also calculated. This was based on the number of papers providing evidence for a particular linkage between: (i) the ES and the beneficiary; and (ii) the beneficiary and the valuation method used, as a proportion of the total papers analysed for that service (following the same approach as for the biophysical attributes).

Presenting the data

Network diagrams were created based on the above variables using the Pajek software (http://pajek.imfm.si/doku.php) to explore the linkages between the individual ecosystem services and both the biotic attributes and the values data.

3. Results

3.1 Literature review on the relationships between biodiversity, ecosystem services and values

3.1.1 Linkages between Ecosystem Service Providers (ESPs) and ecosystem services

The 11 ecosystem services investigated in the literature review were found to be underpinned by different ESPs (Table 3.1), with certain services tending to be linked with certain ESPs. The services freshwater provision, water purification, water flow regulation, mass flow regulation, atmospheric regulation and landscape aesthetics were discussed in at least 70% of papers as being facilitated by a provider at the community level, such as an entire forest, grassland, prairie wetland, high-country landscape or hay meadow. In contrast, the provisioning services of timber production and freshwater fishing were most often facilitated by two or more specific species populations, such as particular species of fish for freshwater fishing or certain tree species for timber production. A particular functional group was often the provider for the regulating services of pollination (in 70% of papers, e.g. flower visiting insects) and pest control (in 30% of papers, e.g. parasitoids). Water flow regulation was the only ecosystem service in this review for which a dominant community was identified as the ESP, although this was mentioned in only two papers.

In general, regulating services were associated with many different ESPs covering the species, functional group and community levels (Table 3.1). The provisioning services were facilitated by ESPs covering two levels: the species and community levels for freshwater fishing and freshwater provision, and the species and functional group levels for timber production. Not surprisingly, the cultural services were almost exclusively provided at one level: the species level for species-based recreation and the community level for landscape aesthetics.

Table 3.1: *Percentage of papers citing a linkage between a specific ESP and ecosystem service. The seven ESP classes are: SP1 = single population; SP2+ = two or more populations; FG1 = single functional group; FG2+ = two or more functional groups; DC = dominant community; CH1 = single community/habitat; CH2+ = two or more communities/habitats). The ESP cited by the greatest percentage of papers is highlighted in grey for each ecosystem service.*

Ecosystem service	**SP1**	**SP2+**	**FG1**	**FG2+**	**DC**	**CH1**	**CH2+**
Provisioning services:							
Timber production	0	80	0	20	0	0	0
Freshwater fishing	27	69	0	0	0	4	0
Freshwater provision	2	8	0	0	0	42	48
Regulating services:							
Water purification	6	10	0	2	0	54	28
Water flow regulation (flood protection)	8	20	0	0	4	50	18
Mass flow regulation (erosion protection)	4	10	2	10	0	46	28
Atmospheric regulation (carbon sequestration)	6	4	2	4	0	56	28
Pest regulation (biological control)	20	12	30	14	0	20	4
Pollination	6	16	70	6	0	0	2
Cultural services:							
Recreation (species-based)	30	66	0	0	0	4	0
Landscape aesthetics	0	0	0	0	0	84	16

3.1.2 Linkages between biodiversity attributes and ecosystem services

A large range of biodiversity attributes (24 out of the 28 listed in Section 2.1) were cited in the papers reviewed as being important for the provision of one or more of the 11 ecosystem services (Figure 3.1). The most common were community/habitat area (31% of papers), species abundance, (27%), species richness (25%) and community/habitat structure (24%). Second to these were species size or weight (12% of papers) and community/habitat age (10%). Biomass, including above- and belowground components, and litter were also mentioned in a number of papers, as was species and functional diversity. In contrast, attributes such as wood density, sapwood amount and leaf N content were mentioned in very few papers (less than 2%).

Table 3.2 provides a breakdown of the important biodiversity attributes by ecosystem service. Timber production and freshwater fishing were most frequently linked to species richness and species abundance, although the latter was also highly related to species size/weight and to a lesser extent to community/habitat area. Pollination was also predominately linked to species richness and abundance, as well as flower-visiting behavioural traits. Species-based recreation was the only other ecosystem service that was mainly linked with species level attributes, with species abundance being the most frequently cited followed by species richness, species size/weight and species diversity.

Freshwater provision, water purification and water flow regulation were most frequently linked with community/habitat area, although community/habitat structure and age were also cited quite often. Landscape aesthetics was also predominantly associated with community level attributes, specifically community/habitat structure and, to a lesser extent, community/habitat area. The services of mass flow regulation, atmospheric regulation and pest regulation show more varied links across different biodiversity attributes. All show the greatest percentage of citations with community level attributes: above- and belowground biomass and community/habitat area and structure for mass flow regulation; community/habitat age and structure and aboveground biomass for atmospheric regulation; and community/habitat structure and area for pest regulation. However, linkages with species level attributes were also found for all three services and with functional group attributes for pest regulation (particularly, functional richness and predator behavioural traits).

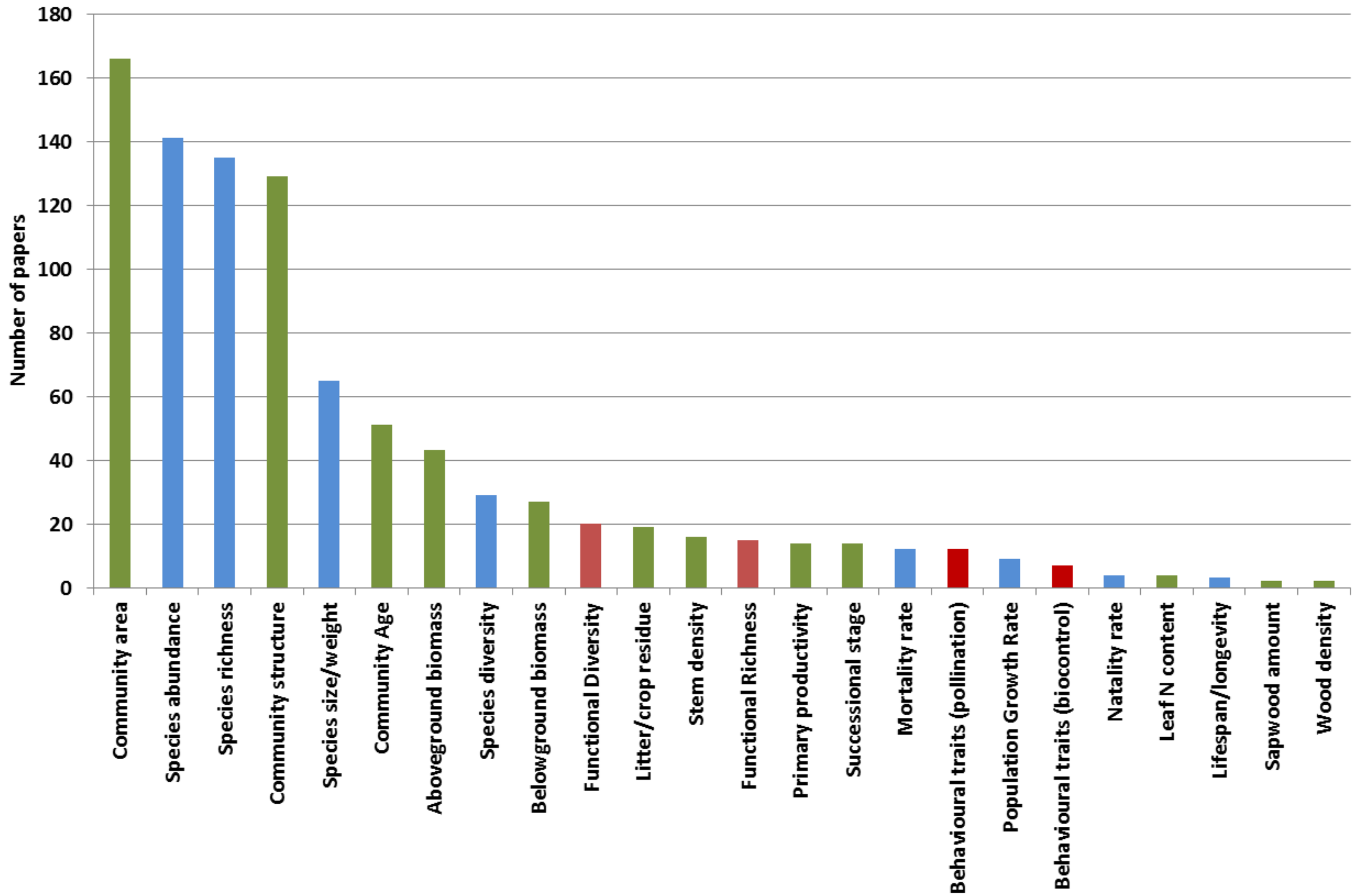


Figure 3.1: *Number of papers citing a linkage between each biodiversity attribute and the 11 ecosystem services investigated. Bars are colour coded as: blue = species level attributes; red = functional group level attributes; green = community or habitat level attributes.*

Table 3.2: *Percentage of papers citing a linkage between a specific biodiversity attribute and ecosystem service. The biodiversity attributes cited by the greatest percentage of papers is highlighted in grey for each ecosystem service. Note that papers may cite more than one biodiversity attribute linked to an ecosystem service.*

	Species-level attributes								**Functional group attributes**				**Community attributes**											
Ecosystem service	Species abundance	Species richness	Species diversity	Species size/weight	Population growth rate	Mortality rate	Natality rate	Life span / longevity	Functional richness	Functional diversity	Behavioural traits (pollination)	Behavioural traits (biocontrol)	Community / habitat area	Community / habitat structure	Primary production	Aboveground biomass	Belowground biomass	Sapwood amount	Stem density	Wood density	Successional stage	Community/habitat/stand age	Litter / crop residue quality	Leaf N content
Provisioning services:																								
Timber production	29	89	3	3	0	0	0	0	6	9	0	0	0	3	3	3	0	0	9	0	6	0	0	9
Freshwater fishing	64	29	2	60	4	11	0	7	0	0	0	0	22	4	13	13	0	0	0	0	0	2	0	0
Freshwater provision	0	2	2	2	4	0	0	0	0	0	0	0	56	12	2	2	6	4	14	0	0	34	0	0
Regulating services:																								
Water purification	2	12	4	4	2	0	0	0	2	0	0	0	62	16	0	6	2	0	4	0	0	6	2	0
Water flow regulation	4	0	0	10	0	0	0	0	0	0	0	0	78	28	0	2	2	0	2	0	4	26	6	0
Mass flow regulation	10	14	2	6	0	0	0	0	0	4	0	0	26	22	0	26	28	0	4	0	10	2	8	0
Atmospheric regulation	4	16	16	14	2	12	0	0	0	10	0	0	8	28	6	26	16	0	0	4	6	30	12	0
Pest regulation	40	18	8	6	6	0	4	0	16	6	0	14	40	52	6	10	0	0	2	0	2	2	10	2
Pollination	70	80	2	2	0	0	0	0	4	6	22	0	4	6	0	0	0	0	0	0	0	0	0	0
Cultural services:																								
Recreation (species)	72	34	20	30	0	2	4	0	4	8	2	0	4	2	0	0	0	0	0	0	0	0	0	0
Landscape aesthetics	2	6	0	0	0	0	0	0	0	0	0	0	34	86	0	0	0	0	0	0	2	0	0	0

Table 3.3 summarises the direction of relationships between biodiversity attributes and ecosystem services.

Table 3.3: *Summary of positive and negative relationships between biodiversity attributes and ecosystem services. Note only relationships found in greater than 10% of papers are included. Arrow direction indicates positive (↑) or negative (↓) relationship. Arrows in bold based on ≥ 50% of papers; arrows not in bold based on 10-49% of papers.*

	Species abundance	Species richness	Species diversity	Species size/weight	Mortality rate	Functional richness	Behavioural traits (pollination)	Behavioural traits (biocontrol)	Community / habitat area	Community / habitat structure	Primary production	Aboveground biomass	Belowground biomass	Stem density	Community/habitat/stand age	Litter / crop residue quality
Provisioning services:																
Timber production	↑	**↑**↓														
Freshwater fishing	↑	↑		↑	↓						↑					
Freshwater provision									↑↓					↓	↓	
Regulating services:																
Water purification		↑							**↑**							
Water flow regulation									**↑**	↑					↑	
Mass flow regulation		↑							↑	↑		↑	↑			
Atmospheric regulation		↑	↑	↑	↓					↑		↑	↑		↑	↑
Pest regulation	↑	↑				↑		↑	↑	↑						↑
Pollination	↑↓	**↑**					↑									
Cultural services:																
Recreation (species)	↑↓	↑	↑	↑												
Landscape aesthetics									↑	**↑**						

Positive relationships

The majority of relationships between biodiversity attributes and ecosystem services had a dominant positive direction (Table 3.3). Species level attributes, such as abundance, were found to benefit species-based recreation, pollination and pest regulation. For example, a higher number of insectivorous birds had a positive effect on pest control (Koh, 2008). Species richness was particularly important for timber production and freshwater fishing, where polycultures were found to be more productive than monocultures (e.g. Erskine et al., 2006 for timber production; and Papoutsoglou et al., 1992 for freshwater fishing). However, it should be noted that although the predominant direction for timber production was positive, eight papers cited a negative relationship with species richness compared to 19 which reported a positive relationship. The size and weight of species is another attribute which positively affected service provision, including freshwater fishing, atmospheric regulation and species-based recreation. In forest environments, it was found that larger trees, such as those with a diameter at breast height (DBH) ≥ 10cm account for over 90% of the aboveground carbon stocks in forest and agroforest habitats in eastern Panama (Kirby & Potvin, 2007).

Functional group attributes, such as functional richness and diversity, also displayed a predominantly positive relationship across the services, most commonly discussed for pest regulation and pollination. For these services, the benefits of behavioural traits such as flower visiting behaviour for pollination (Biesmeijer et al., 2006; Hoehn et al., 2008) and natural pest control (Drapela et al., 2011; Lee & McCracken, 2005) were particularly noted.

Community level attributes, such as community/habitat area, were also found to benefit many ecosystem services, including water purification, water flow regulation, mass flow regulation and landscape aesthetics. Water flow regulation was significantly improved as a result of increased forest area through reducing runoff and providing greater water storage (Farley et al., 2005; Thomas and Nisbet, 2006). A number of papers also discussed a positive relationship with stand age. For atmospheric regulation, larger carbon storage was found in older tree species due to a combination of (a) the time period over which they have sequestered carbon, and (b) the result of tree size increasing with age (e.g. Hantanaka et al., 2011; Keeton et al., 2010; Kirby & Potvin, 2007; Zhao et al., 2010). Numerous papers cited the importance of biomass for carbon sequestration. For example, higher levels of aboveground biomass were linked to increased carbon storage in an alpine meadow (Sun et al., 2011) and a larger green biomass was found to increase soil nitrogen content, in turn increasing soil carbon, in sub-alpine grasslands (Lavorel and Grigulis, 2012). Landscape diversity (or complexity) was also found to benefit landscape aesthetics. For example, Van den Berg et al. (1998) found that beauty ratings were positively related with perceived complexity in The Netherlands, whilst Yao et al. (2012) reported that perceived visual quality was positively influenced by the variety of vegetation in China.

Negative relationships

Negative relationships between biodiversity attributes and ecosystem services constitute only a small part of the overall linkages and are generally based on fewer papers than the positive relationships (Table 3.3). Freshwater provision was the only ecosystem service which shows predominantly negative relationships with different biodiversity attributes. In general it was found that increases in community/habitat area, structure, stem density, aboveground biomass and age increased water consumption and, hence, reduced the provision of this ecosystem service (e.g. Bren & Hopmans, 2007; Farley et al., 2005; Petheram et al., 2002; Zou et al. 2008). For example, increases in afforestation were associated with an average water yield reduction (e.g. Rey Benayas et al. 2007; Sun et al., 2006), as demonstrated by Buytaert et al. (2007) where afforestation of natural grasslands with *Pinus patula* decreased base flow and reduced the water yield by about 50% in a study site on the Andean highlands of Ecuador. Furthermore, in a study of a grassland catchment afforested with *Eucalyptus grandis*, a significant reduction in stream flow was observed with the stream becoming completely dry after a period of 9 years (Scott & Lesch, 1997).

A few negative relationships were found for other ecosystem services, particularly with mortality rate. For example, atmospheric regulation was reduced in grassland communities due to increased mortality of root and rhizome tissues from grazing (Klumpp et al., 2009). Not surprisingly, mortality rate was also found to negatively affect species-based recreation and freshwater fishing by lowering fish stocks (e.g. Lorenzen, 2001). In addition, biodiversity was found to reduce pest regulation in a number of papers. Reasons cited for this were that (a) in species-rich systems, alternative prey can be used as food for predators or parasitoids, decreasing the suppression of pest species (Oelbermann & Scheu, 2009), and (b) that some predators can either become prey themselves (Xu et al., 2011) or protect pests (Mody et al., 2011); both of which decrease the effectiveness of pest control.

Negative impacts on ecosystem service provision often involved invasive species. For example, rapid growth rates and extensive root systems of species such as *Tamarisk* and *Kandelia candel* were found to increase sedimentation rates, raising surface water elevation and increasing the likelihood of flooding (Foote et al., 1996; Lee & Shih, 2004; Zavaleta, 2000). Hence, increases in the area, size and abundance of habitats containing these species negatively affected water flow regulation (Erskine & Webb, 2003). Several studies also reported invasive species antagonising carbon storage and thus, affecting the service of atmospheric regulation. This was found in forests as a result of bark beetle (*Ips typographus* L.) induced damages (Seidl et al., 2008) and also in Californian grasslands where non-native species, such as *Avena barbata*, were invading native grasses (Koteen et al., 2011). In addition, high densities of the invasive plant dandelion (*Taraxacum officinale*) in the Chilean Andes were found to reduce pollination (Munoz & Cavieres, 2008). Furthermore, the introduction of alien fish species in Mediterranean freshwaters was found to negatively affect native fish populations due to competition for resources and habitat degradation (Hermoso et al., 2011).

Unclear or unknown relationships

The review found that the relationship between biodiversity attributes and ecosystem service provision is not always simple (i.e. predominantly positive or negative). Hence, there were instances where the relationship between certain attributes and a given ecosystem service was classified as unclear. This could occur due to the existence of a threshold after which the direction of the relationship with the biodiversity attribute changed, such as was reported in 34% of papers for the effect of species abundance on freshwater fishing. Often a higher abundance of fish species increased the provision of this service, however, once a certain level or ecological carrying capacity was breached, fish yield was found to decrease (e.g. De Silva et al., 1992; Hasan & Middendorp, 1998; Lorenzen, 1995; Smith et al., 2012). A similar effect was observed in a study of mass flow regulation by Cammeraat et al. (2005) in which root systems of early successional vegetation on steep slopes were associated with increased erosion, but this effect was reversed after a period of 40 years when later successional plant communities with different root systems had established.

Unclear relationships may also result from conflicting evidence both within and between papers. This was found for two papers considering linkages between species diversity and atmospheric regulation (e.g. Potvin et al. 2011; Sharma et al. 2010), where this biodiversity attribute has a positive relationship with a single community ESP and a negative relationship when two or more communities were identified as the ESP. Interactions between species can also lead to unclear relationships. For example, bees, particularly managed honey bees, can have indirect negative effects on pollination services by competing with wild pollinator species for flower resources and reducing flower visitations by the latter (Allsopp et al., 2008; Shavit et al., 2009). This led to 14 and 10% of relationships for pollination with species abundance and species richness, respectively, being defined as unclear. The effect of community/habitat structure on pest regulation was also unclear in 18% of papers. For example, Bianchi et al. (2010) found that dispersal and gathering behaviour of predator groups influenced their performance, although this was in turn influenced by landscape structure.

3.1.3 Linkages between abiotic factors and ecosystem services

The influence of abiotic factors, in addition to biotic attributes, on ecosystem service provision was also considered in the review. Although the search strategy was directed towards literature focusing on biotic attributes affecting the ecosystem services, 22% of papers specifically mentioned a link between an abiotic factor and service delivery. They are an integral part of the ESP (Harrington et al., 2010) and it is hardly surprising that climatic parameters such as temperature and precipitation were commonly cited, particularly for the services of atmospheric regulation, mass flow regulation, water flow regulation, water purification and freshwater provision, where these abiotic parameters often have direct, clearly observable influences. Further abiotic factors that were identified in greater than 10% of papers for at least one ecosystem service were soil properties such as porosity (cited across many services), water quality and nutrient availability (particularly cited for freshwater fishing), and slope (particularly cited for mass flow regulation).

3.1.4 Linkages between ecosystem services and values

The main focus of the review of valuation studies has been on the choice of valuation approach for each of the 11 ecosystem services. The results of this central part of the review are summarised in Table 3.4.

Timber production

Ten studies about timber production have been reviewed. As timber production is a provisioning service they all value the extractive use value of timber. One study also claims to value the option value of forests used for timber production. In all studies market prices of timber are used to value timber production. If market prices are assumed to reflect marginal utility to society, the market price approach is an appropriate valuation method to use for a service that is traded.

Freshwater fishing

Commercial freshwater fishing is another provisioning service, and it is not surprising that the ten reviewed studies all value the use value of fishing. The catch is traded and the market prices are the correct basis for valuation of this service. However, the market price approach is only used in four of the ten studies. Two studies use the production function approach, and therefore indirectly the market price approach. The use of the production function approach is due to the focus of the studies. They value preservation of freshwaters for fishing, and good quality freshwater can be regarded as a service or input that contributes to the production of fish, i.e. the production function approach. The value of the produce is measured in market prices. One study values protection of freshwater fishing using the avoided cost approach. As long as the possibility of fishing is preserved, the costs of freshwater regulation are avoided. However, avoided costs can only be regarded as an indicator of the benefits or value of freshwater fishing. Finally, three studies use the benefit transfer approach. This is the least resource demanding approach to valuation, but it does not represent an original contribution to the valuation of freshwater fishing.

Freshwater provision

Very few studies exist valuing the quantity of water provided from ecosystems. The six reviewed papers show that freshwater provision can be valued in a number of different ways. When market prices exist, valuation can be based on these prices as has been done in one paper. However, if market prices are heavily regulated then this valuation approach is debatable. When freshwater is used as an input to production, the production function approach could be preferable: the value of freshwater is assumed to be equal to its marginal contribution to production. Two papers use this approach.

Freshwater may also be in short supply, and there may be problems in maintaining the actual supply. In such cases it may be argued that valuation should be based on either the avoided cost approach or the revealed or stated preference. The avoided cost approach shows the costs that must be paid to maintain the water supply, or the costs avoided if no initiatives for maintaining the water supply are necessary. The revealed and stated preference approaches show individuals’ willingness to pay for an ample freshwater supply. Each of these valuation approaches are used in one paper.

Table 3.4: *Summary of the valuation methods used to evaluate the different types of values related to the 11 ecosystem services.*

	Market prices	Production function	Avoided costs	Averting behaviour	Revealed preference	Stated preference	Deliberative valuation	Expert valuation	Multi-criteria	Benefit transfer	***Total***
Provisioning services											
Timber production											
- extractive use value	9	1									**10**
- option value	1										**1**
Freshwater fishing											
- use value	4	2	1							3	**10**
Freshwater provision											
- use value	1	2	1		1	1					**6**
Regulating services											
Water purification											
- use	1		1	1	2	15			1	2	**23**
- use and non-use						12			1		**13**
Flood protection											
- indirect use			5			1			2	1	**9**
Erosion protection											
- direct use						1					**1**
- indirect use	1	1	5			3				9	**19**
- non-use						1					**1**
Carbon sequestration											
- indirect use				5		1				1	**7**
- use and non-use						1				3	**4**
Pest regulation											
- indirect use		4	10								**14**
Pollination											
- indirect use	8	29	1	2				1			**41**
- option value		1	1								**2**
Cultural services											
Recreation											
- use value	11				7	13				3	**34**
- non-use value						2					**2**
Landscape aesthetics											
- use and non-use					1	2			7		**10**

Water purification

Water purification is a regulating service which is related to the quality of freshwater. Therefore, in most of the 23 reviewed studies valuation is based on individuals' willingness to pay for clean water. Fifteen studies base the valuation on the stated preference approach while two studies use the revealed preference approach. When the stated preference approach is used it is difficult to determine if use value or non-use value is measured. In 12 studies the measured water purification value includes both types of value and it is not attempted to distinguish between the two values.

The value of clean water can also be measured on the basis of the costs of water protection measures, but only two of the reviewed studies use this approach. One study measures the avoided costs of water protection and the other study measures the costs of averting behaviour. However, it is not always clear if it is avoided costs or the cost of averting behaviour that is measured. Both represent costs of water protection and of supplying clean water.

Two studies use the benefit transfer approach to value the water purification service and one study uses the multi-criteria approach. However, benefit transfer should only be used if the other valuation methods are too resource-demanding to use, and the results of multi-criteria analysis are difficult to use in connection with traditional welfare economical valuation. Multi-criteria analysis is mainly appropriate when studying decision-making, if the decisions valued have multiple attributes.

Flood protection

Flood protection is a regulating service that has an indirect use value. The ecosystem service provider is not used directly, but its protecting service creates a basis for welfare generating human activities. Nine studies are reviewed and five of them use the avoided cost approach to value flood protection services. The avoided costs are typically avoided economic losses because of flooding or the costs of alternative flood protection measures.

The avoided cost approach is an appropriate choice for valuation of flood protection, but the stated preference approach can also be used. One study does this. Stated preference methods are mainly relevant when the focus of the research is people's perceptions of risks and evaluation of potential financing schemes. As can be seen from Table 3.4, benefit transfer and multi-criteria valuation have also been used in the literature.

Erosion protection

The erosion protection service is very similar to flood protection, and of the eleven original valuation studies reviewed five use the avoided cost approach. This can be justified in the same way as the use of the approach in relation to flood protection. Four studies use the stated preference approach, which can be justified when studying individual preferences.

The market price and production function approaches have also been used, but in this case they are very similar to the avoided cost approach. Thus, market prices are used to value the economic loss avoided because of erosion, and the protection service can be regarded as an input to production whose marginal value can be measured on the basis of an estimated production function. This marginal value reflects the avoided production loss because of erosion.

Nine of the studies reviewed base the valuation of erosion protection on benefit transfer. As mentioned above this approach should only be used when other possibilities are ruled out.

Carbon sequestration

Eleven studies related to carbon sequestration have been reviewed. Seven of them only value the indirect use value of this regulating service, while four studies value both use and non-use values - however without distinguishing between these.

Two alternative approaches are used to value the carbon sequestration service. One is based on the costs of alternative measures to reduce greenhouse gas emissions, and the other is based on the costs of climate change caused by greenhouse gas emissions. The latter is the theoretically most satisfactory, but also the most difficult to use in practice.

Four of the reviewed studies base the valuation of carbon sequestration on the estimated costs of climate change. The values are transferred from other studies that estimate these costs. The benefit transfer approach is fully justified in this case as carbon sequestration directly contributes to restricting climate change. However, estimating the costs of climate change is associated with great uncertainty.

Two studies use the stated preference approach. The measured values reflect individuals' willingness to pay for greenhouse gas mitigation or willingness to pay for avoiding climate change. Indirectly, willingness to pay may reflect the expected costs of future climate change, but this is far from certain. However, when studying potential ways of financing climate change mitigation, people's perceptions of these costs are relevant, even if people are unaware of the true value of climate change mitigation. When interpreting stated preference estimates for use in economic analysis, it is important to know the methods used to derive the value estimates.

Valuation of carbon sequestration in the last five studies is based on the costs of greenhouse gas mitigation. A target for reduction of greenhouse gas emissions is assumed and the measured mitigation costs reflect the marginal cost of meeting that target. The marginal costs can be estimated in cost-effectiveness studies or the market price of CO_2 quotas can be used as an indicator of these costs. Both approaches have been used in the reviewed studies. They are relatively easy to use in practice and they are also widely used in economic analyses to value changes in CO_2 emissions. However, the measured costs are not a satisfactory measure of the value of carbon sequestration. The costs do not reflect the avoided costs of climate change, but the costs of meeting a more or less haphazardly chosen CO_2 reduction target.

Pest regulation

Pest regulation is a regulating service provided by nature, e.g. insects, birds and mammals. The service has an indirect use value as it controls attacks of pests, fungal infections and weeds. Pest control results in higher yields or a reduced need for alternative pest control.

The fourteen studies reviewed value the pest regulation service on the basis of its importance for the value of yields and the costs of alternative pest control. Ten studies estimate both the avoided yield loss and alternative pest control costs. Therefore, they are categorised as using the avoided cost approach for valuation.

However, it is debatable whether avoided costs should include both yield loss and alternative pest control costs. The point is that it could be argued that natural pest control either controls yield losses or leads to less need for alternative pest control. Only one of these avoided costs should be included in the valuation - or a combination if no double-counting can be assumed.

Four studies only estimate the importance of pest control for yield and they are categorised as using the production function approach. In these studies there are no problems of double-counting.

Pollination

Pollination is a regulating service which has an indirect use value and all 41 reviewed studies estimate this value. Two of them also claim to value the option value of preserving pollinators. Most studies analyse the relationship between pollination and yield of different crops or fruits. This is why they can be said to use the production function approach. Not all of them value the yield, but those of them that do value it use the market price approach to value the produce. This is the correct approach and in fact it is very closely related to the production function approach. Some valuation studies include the value of honey produced by the pollinating bees as a part of the value of pollination. However, honey production is not a part of this ecosystem service. It is a provisioning service supplied by bees.

One study uses the avoided cost approach to value pollination and two studies use the averting behaviour approach. If the calculated costs reflect the value of the avoided production loss caused by lack of pollinators then the avoided cost approach is similar to the production function approach. On the other hand, if the estimated values reflect the costs of preserving pollinators then the avoided cost approach cannot be recommended. The estimated costs are not necessarily equal to the value of the pollination services. This criticism also applies for the averting behaviour approach.

Finally, one study bases the valuation of pollination on expert evaluation. However, this study cannot be regarded as a real valuation study, as it only includes experts' qualitative evaluations of the importance of pollination.

Recreation

Thirty four studies have been reviewed in relation to recreational services. Recreation is a cultural service which has a use value to individuals who use nature for recreational purposes. All the reviewed studies estimate use value, and two of them also claim to estimate non-use value even if it is not obvious what is meant by the non-use value of a recreational service that always involves activities by individuals.

The majority of the reviewed studies use stated or revealed preference approaches to value recreational services. These are also in most cases the only possible economic approaches as many recreational services are non-traded goods.

Eleven studies value recreational services on the basis of the tourist expenditure they are expected to generate. In Table 3.4 these studies are categorised as having used the market price approach. Tourist expenditure reflect visitors' willingness to pay for visiting an area and using the recreational services it supplies, but expenditure may include payment for many other goods and services. Therefore, this valuation approach should be used with care.

Three studies use the benefit transfer approach which can be a resource-saving solution in cases where results from original and relevant valuation studies exist.

Landscape aesthetics

Seven of the ten reviewed studies about landscape aesthetics do not value this service. In Table 3.4 they are categorised under the multi-criteria approach as they contain conceptual discussions about what is meant by landscape aesthetics as an ecosystem service, and which factors or nature elements determine its value. The discussion reflects the prevalent uncertainty with regard to how valuation of this service should be handled.

Three studies use the stated or revealed preference approaches which also seem to be the only feasible approaches to use in relation to valuation of landscape aesthetics. However, the valuation methods have to be further developed both in relation to how aesthetic services are described and in relation to the way individuals' preferences are revealed.

All services

The majority of the reviewed valuation studies are regional (41%) or local (32%) studies. Only 18% and 9% of the studies are national and global, respectively. This is not surprising as the value of an ecosystem service in most cases depends on local or regional conditions. For example, the value of flood protection depends on how the protected area is used - how much of the area is farmland, what is grown, how many individuals live in the area and what infrastructure is found in the area. The same is true for most other regulating and cultural services. The only exceptions where globally valid values of ecosystem services can be meaningfully estimated are provision of timber which has a world market price and carbon sequestration where the value depends on the avoided global damage costs of climate change. However, for most other ecosystem services the possibility of benefit transfer between similar areas exists.

The reliability of the reviewed valuation studies has been evaluated with regard to the amount and quality of data, and the uncertainty and distribution of value estimates. With regard to the amount and quality of data, the review distinguishes between three categories:

1. Weak data foundation and data collection not designed to address ecosystem service valuation;
2. Solid data foundation and coherent data collection;
3. Thorough, high quality and especially designed data collection.

Of almost two hundred studies reviewed, 64% are evaluated as being of category 2, 22% as category 1 and the remaining 14% as the highest quality category 3. Of course, evaluations like these are uncertain - especially because many reviewers have been involved - but overall the results show that a solid data foundation for valuation of ecosystem services can be found in most cases.

With regard to the uncertainty and distribution of value estimates, the review has distinguished between the following categories:

1. Superficial analysis;
2. Solid analysis and results;
3. Thorough analysis, clear results in terms of magnitude of values (and potentially distribution of value).

The evaluation with regard to value estimates is very similar to the evaluation of data foundation. Again 64% of the studies are evaluated as being solid and even 20% of the studies as being thorough analyses. This means that only 16% of the studies are regarded as being superficial by the reviewers. Of course, the same reservations pertain to these evaluation results as those put forward in relation to data foundation.

3.2 Mapping the empirical evidence of the relationships between biodiversity, ecosystem services and values using network analysis

The network diagrams resulting from the literature review are presented in Figures 3.2 to 3.12; there is one diagram for each of the 11 ecosystem services. Three summary variables are presented on the network diagrams using different aspects of the arcs (lines) joining the nodes:

- **Width of line:** the width of the line is drawn as proportional to the ***Level of support*** (i.e. the number of records).

- **Colour of line:** The colour of the line reflects the ***Direction of the evidence***, with green lines for predominantly positive relationships and red lines for predominantly negative ones; grey is used for relationships that are classified as neither or both ("neutral").

- **Depth of colour:** A three class colour system is used to differentiate the ***Strength of the evidence*** with lighter shades of green, red and grey reflecting weaker positive, negative and neutral relationships and darker shades reflecting stronger ones. See Table 3.5.

Table 3.5: *Key to network line shading.*

		Positive	Neutral	Negative
Biophysical Review	Strength <3.5			
	Strength ≥3.5			
Values Review	All linkages			

Overview of networks

When interpreting the network diagrams it is important to bear in mind the limitations inherent in basing the study purely on the number of citations for each linkage, which does not necessarily reflect their biophysical functional importance. Some links may be considered so obvious that they receive little research attention; others may be under-investigated due to funding constraints or technical difficulties in designing appropriate experiments. Also, as noted above, simply ranking a linkage as being "positive" or "negative" does not take account of the biophysical strength of that linkage.

Nevertheless, the network-based approach is a useful tool to provide a quick overview of the connections between an ecosystem service, the biophysical attributes and the valuation in a single diagram. An exploration of Figure 3.2 for timber production highlights this. It is quickly clear from the figure that the only types of ecosystem service providers mentioned in the biophysical review are "multiple species" and "multiple functional groups", and of these "multiple species" commands by far the greater proportion: the species (2+) line is considerably thicker than the functional group (2+) line. It is also clear that the majority of relationships identified with biophysical attributes are positive (there is a lot more green than red, in contrast to freshwater provision, Figure 3.4). It is also clear that the evidence at the functional group level is on average considered to be of greater strength than that at the species level (the green is darker). All attributes, both biotic and abiotic, highlighted by the review can be identified; however, it is easy to identify those which are most commonly mentioned within the papers (those with the thickest lines: species richness, species abundance). At the same time it is also quickly apparent where the majority of work on valuation within the literature has been performed: "Groups of stakeholders", "Communities/societies" and "Firms" are the classes most commonly investigated, and of these "Groups of stakeholders" are the most commonly investigated. Conversely, "Firms" are considerably less commonly investigated than the other classes. It is also clear that market price is the valuation method most commonly applied (it is the thickest line for each of the beneficiaries) but that "benefit transfer and reviews" and "other methods" are also applied.

Looking at the full diversity of the 11 ecosystem services (Figures 3.2 to 3.12) it is clear that the relationships between ecosystem service, ESPs and their attributes varies significantly, from very simple networks such as timber production and landscape aesthetics, with only one or two classes of ESPs and few associated attributes, to considerably more complex networks such as atmospheric regulation, mass flow regulation and pest regulation, each of which links to six classes of ESPs. It is also very apparent from the diagrams that negative relationships with biophysical attributes are far less commonly identified than positive ones for the majority of the ecosystem services: most have only one or two negative relationships, and landscape aesthetics has none. However, not all ecosystem services are characterised in this way: the freshwater provision network is very much dominated by *negative* relationships between biophysical attributes and the service of water provision (because vegetation intercepts rain and absorbs groundwater). We can therefore conclude that the direction of the ESP-attribute relationship is specific to the ecosystem service in question.

A comparison of the values networks across the ecosystem services again shows variety in terms of the complexity of the networks. Some ecosystem services, such as freshwater fishing, pest regulation and water purification are very much dominated by studies of a single type of beneficiary (communities/societies, groups of stakeholders and households/individuals, respectively). At the other extreme, studies of atmospheric regulation and landscape aesthetics seem better balanced between a number of different beneficiary classes. The vast majority show that the focus of studies has been on a single main beneficiary class, but supported by a smaller number of studies that have looked at one or two alternative beneficiary classes. In terms of methods it is clear that some methods are strongly preferred for particular services (e.g. market price for timber production) while others show a wide range of methods applied, (see “atmospheric regulation”; Figure 3.8). In many cases different methods are applied to different beneficiary types: e.g. for mass flow regulation the main beneficiary is “communities/societies” and this is shown to be studied using “benefits transfer and review”, “avoided costs” and “other”. However, the secondary beneficiaries are both shown to be studied using “stated preference”; this may reflect the fact that certain methodologies are more appropriate for certain types of beneficiary.

Figure 3.2 Network for the service of timber production. *(1) indicates that the class is for a single ESP; (2+) indicates two or more of the same class; (+) and (-) after the name of an ESP or attribute indicates the direction; "BT and reviews" = "benefit transfer and reviews"; "AF" = abiotic factor.*

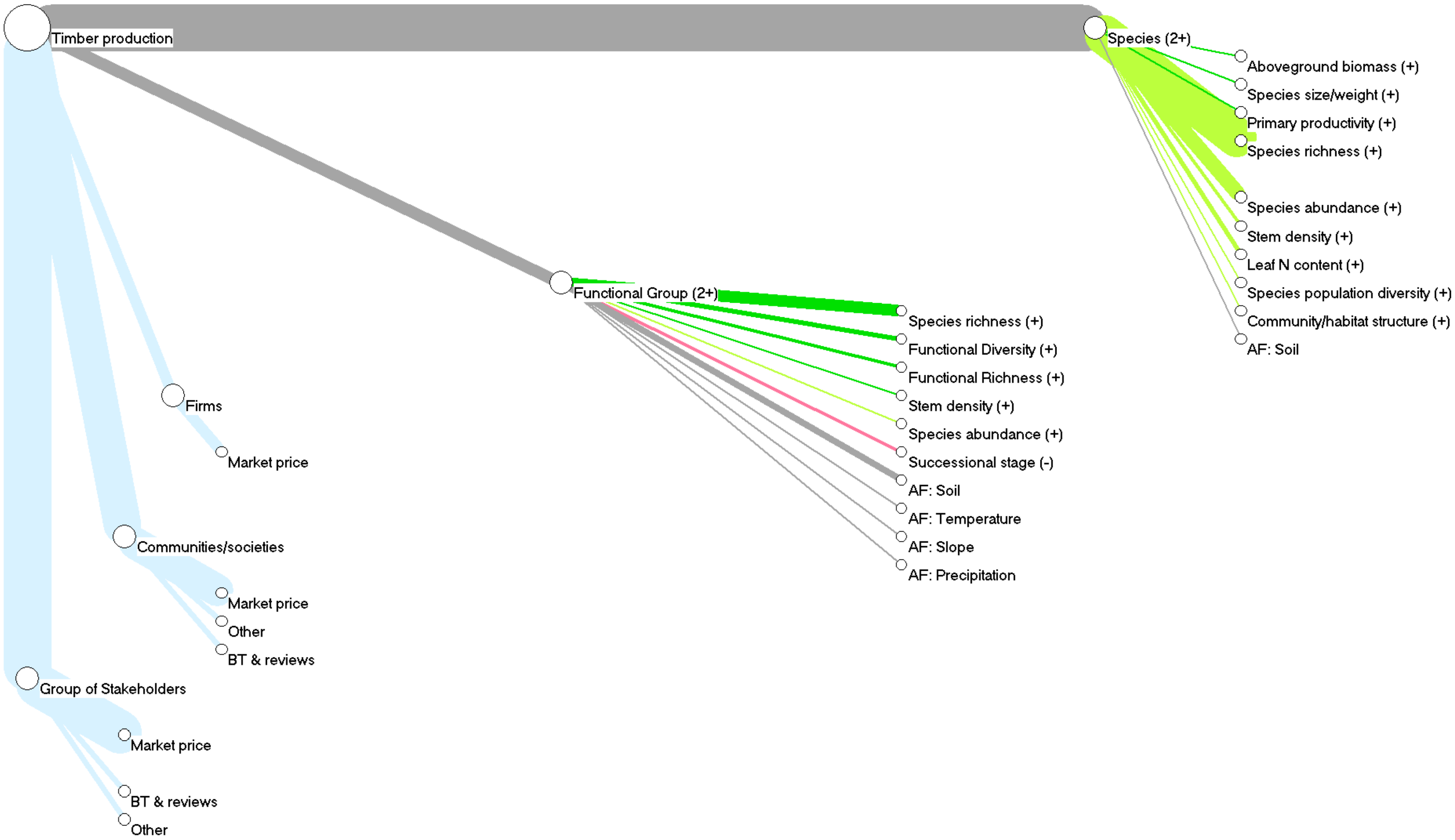


Figure 3.3 Network for the service of freshwater fishing. *(1) indicates that the class is for a single ESP; (2+) indicates two or more of the same class; (+) and (-) after the name of an ESP or attribute indicates the direction; "BT and reviews" = "benefit transfer and reviews"; "AF" = abiotic factor.*

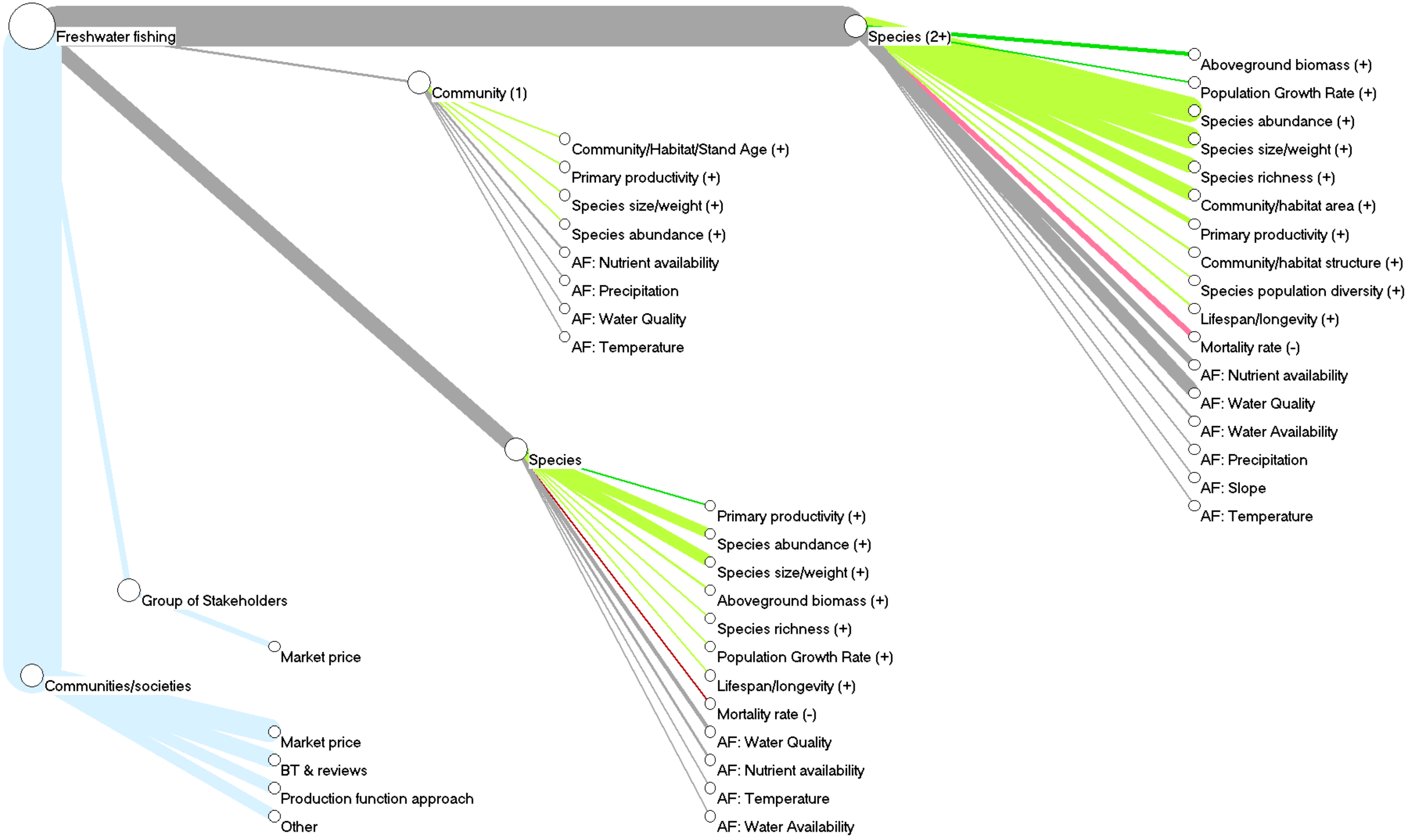


Figure 3.4 Network for the service of freshwater provision. *(1) indicates that the class is for a single ESP; (2+) indicates two or more of the same class; (+) and (-) after the name of an ESP or attribute indicates the direction; "BT and reviews" = "benefit transfer and reviews"; "AF" = abiotic factor.*

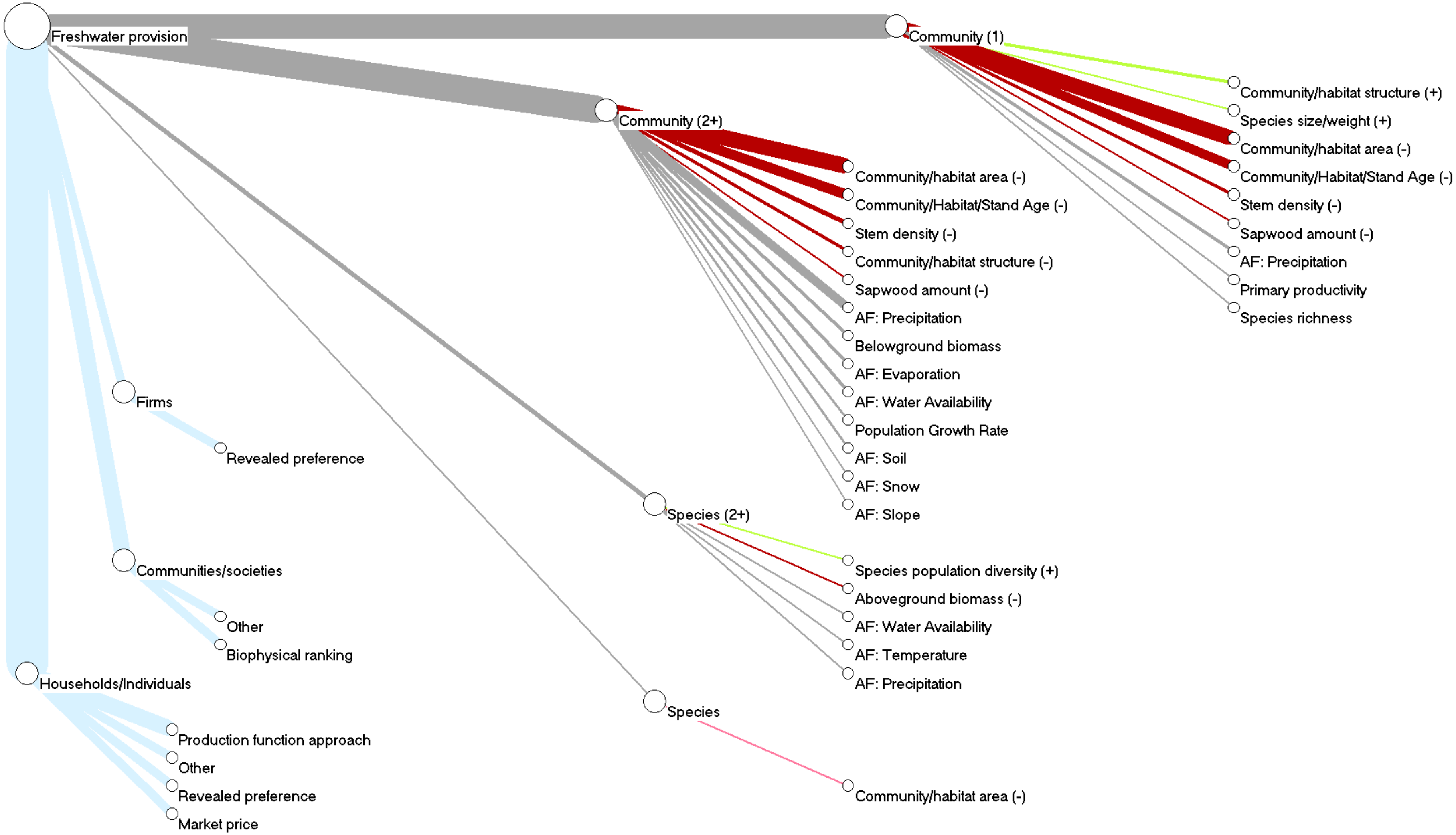


Figure 3.5 Network for the service of water purification. *(1) indicates that the class is for a single ESP; (2+) indicates two or more of the same class; (+) and (-) after the name of an ESP or attribute indicates the direction; "BT and reviews" = "benefit transfer and reviews"; "AF" = abiotic factor.*

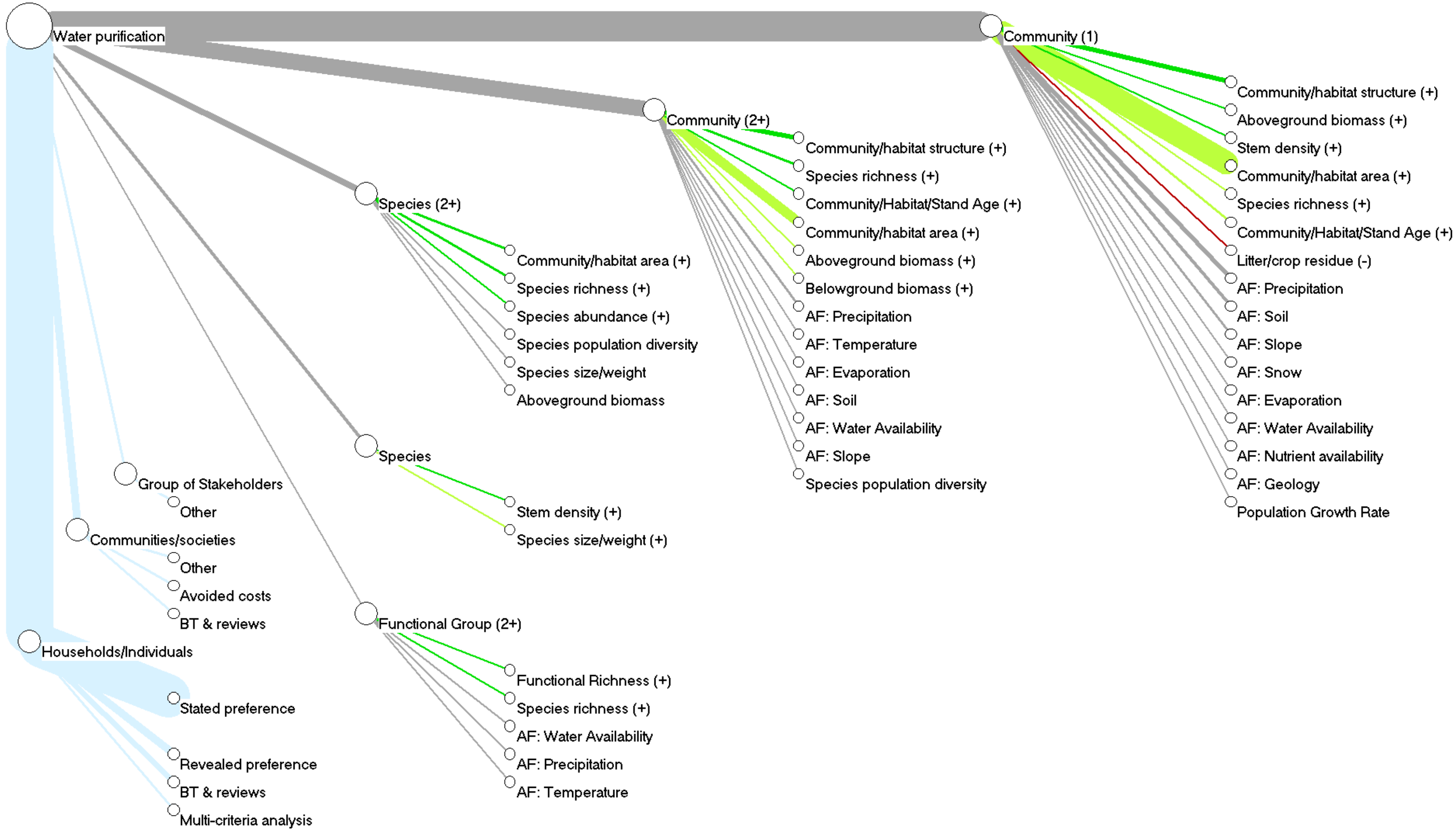


Figure 3.6 Network for the service of water flow regulation. *(1) indicates that the class is for a single ESP; (2+) indicates two or more of the same class; (+) and (-) after the name of an ESP or attribute indicates the direction; "BT and reviews" = "benefit transfer and reviews"; "AF" = abiotic factor.*

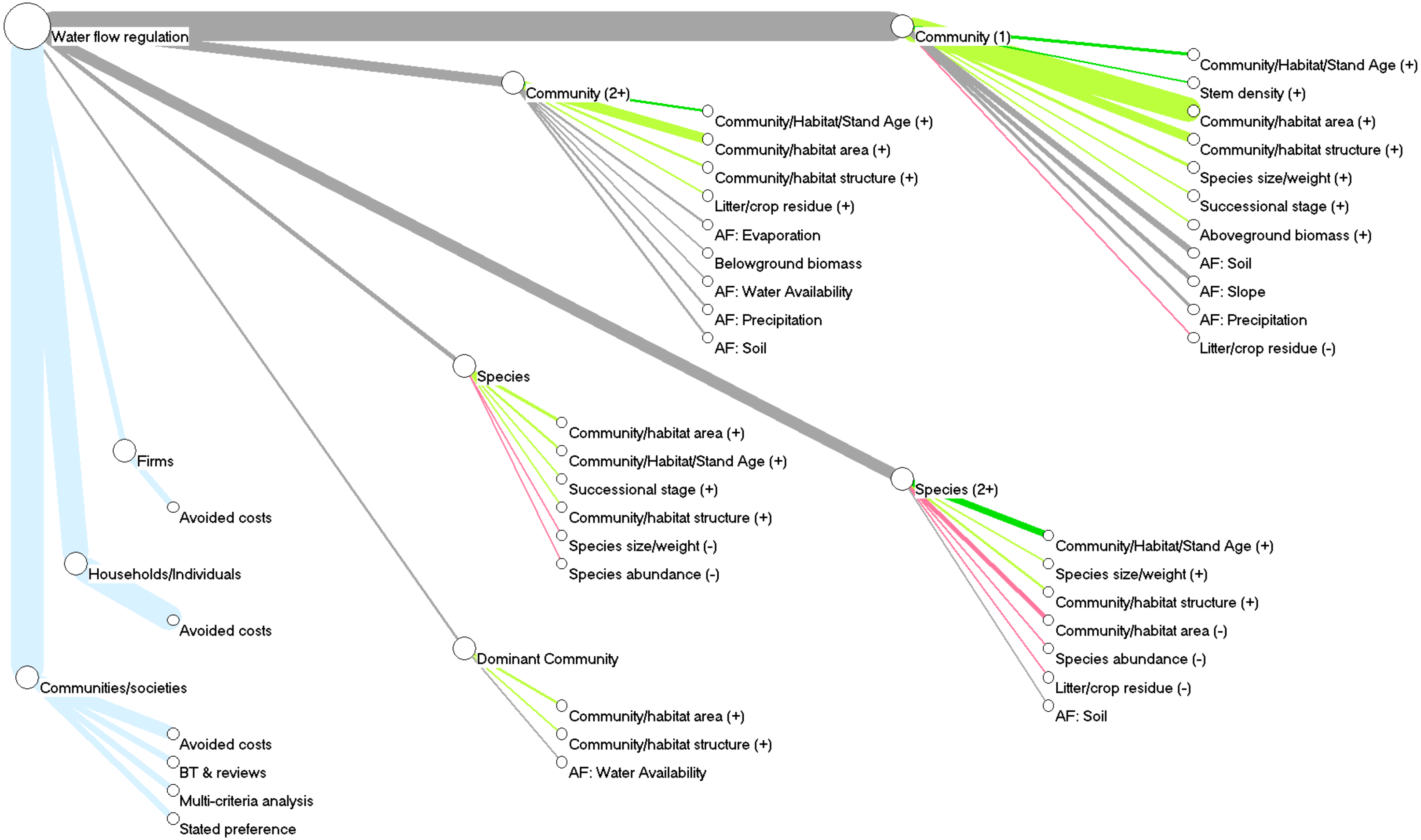


Figure 3.7 Network for the service of mass flow regulation. *(1) indicates that the class is for a single ESP; (2+) indicates two or more of the same class; (+) and (-) after the name of an ESP or attribute indicates the direction; "BT and reviews" = "benefit transfer and reviews"; "AF" = abiotic factor.*

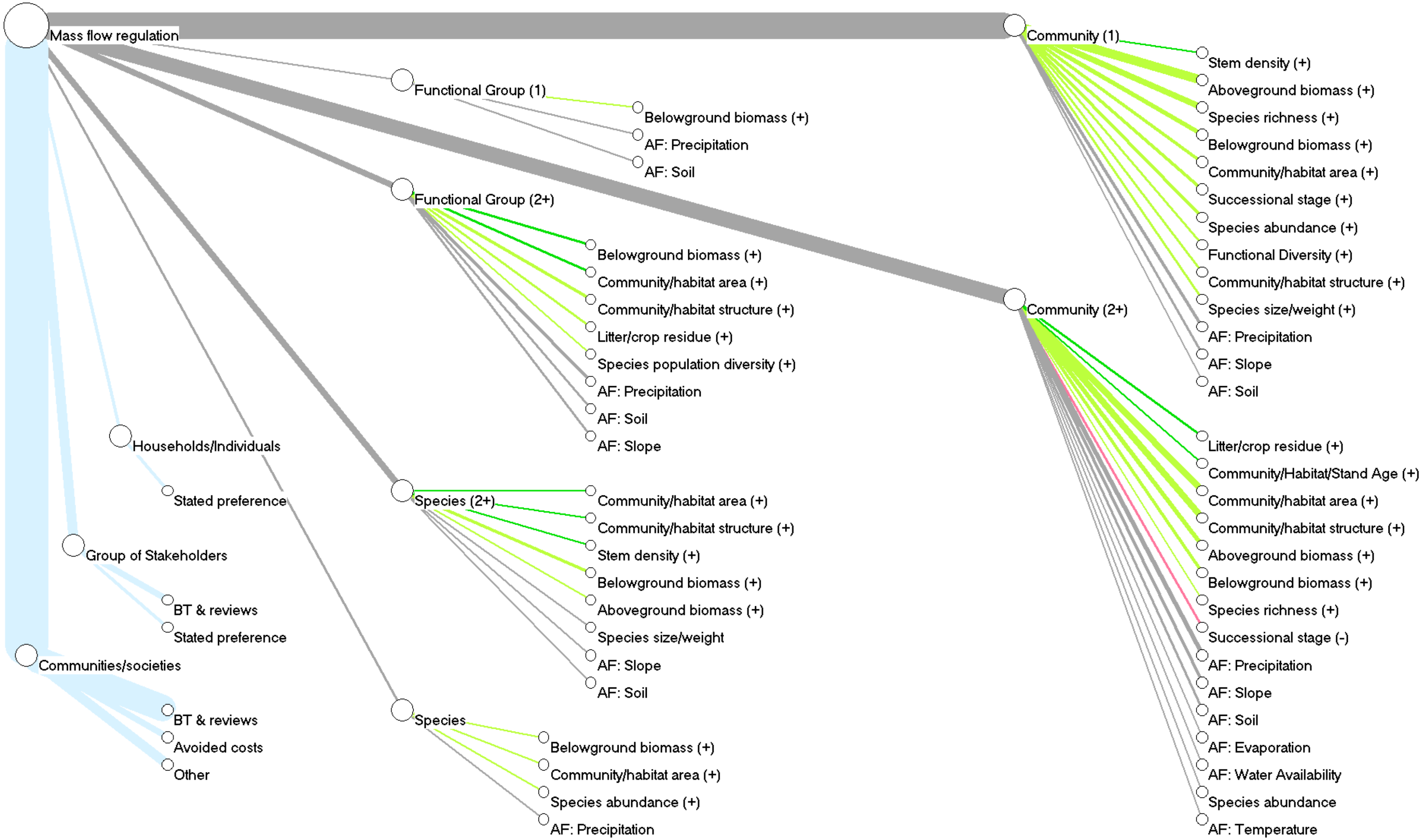


Figure 3.8 Network for the service of atmospheric regulation. *(1) indicates that the class is for a single ESP; (2+) indicates two or more of the same class; (+) and (-) after the name of an ESP or attribute indicates the direction; "BT and reviews" = "benefit transfer and reviews"; "AF" = abiotic factor.*

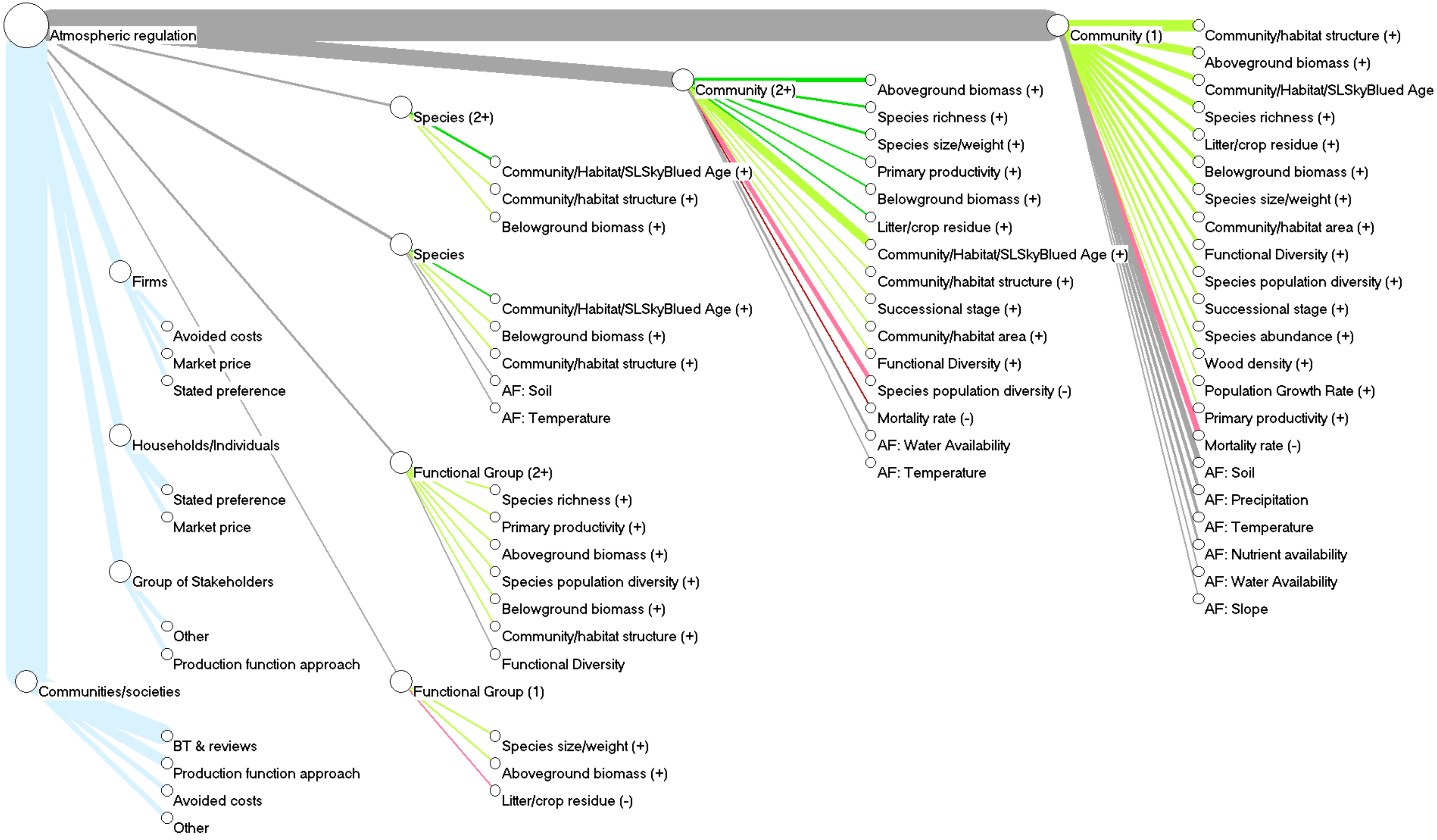


Figure 3.9 Network for the service of pest regulation. *(1) indicates that the class is for a single ESP; (2+) indicates two or more of the same class; (+) and (-) after the name of an ESP or attribute indicates the direction; "BT and reviews" = "benefit transfer and reviews"; "AF" = abiotic factor.*

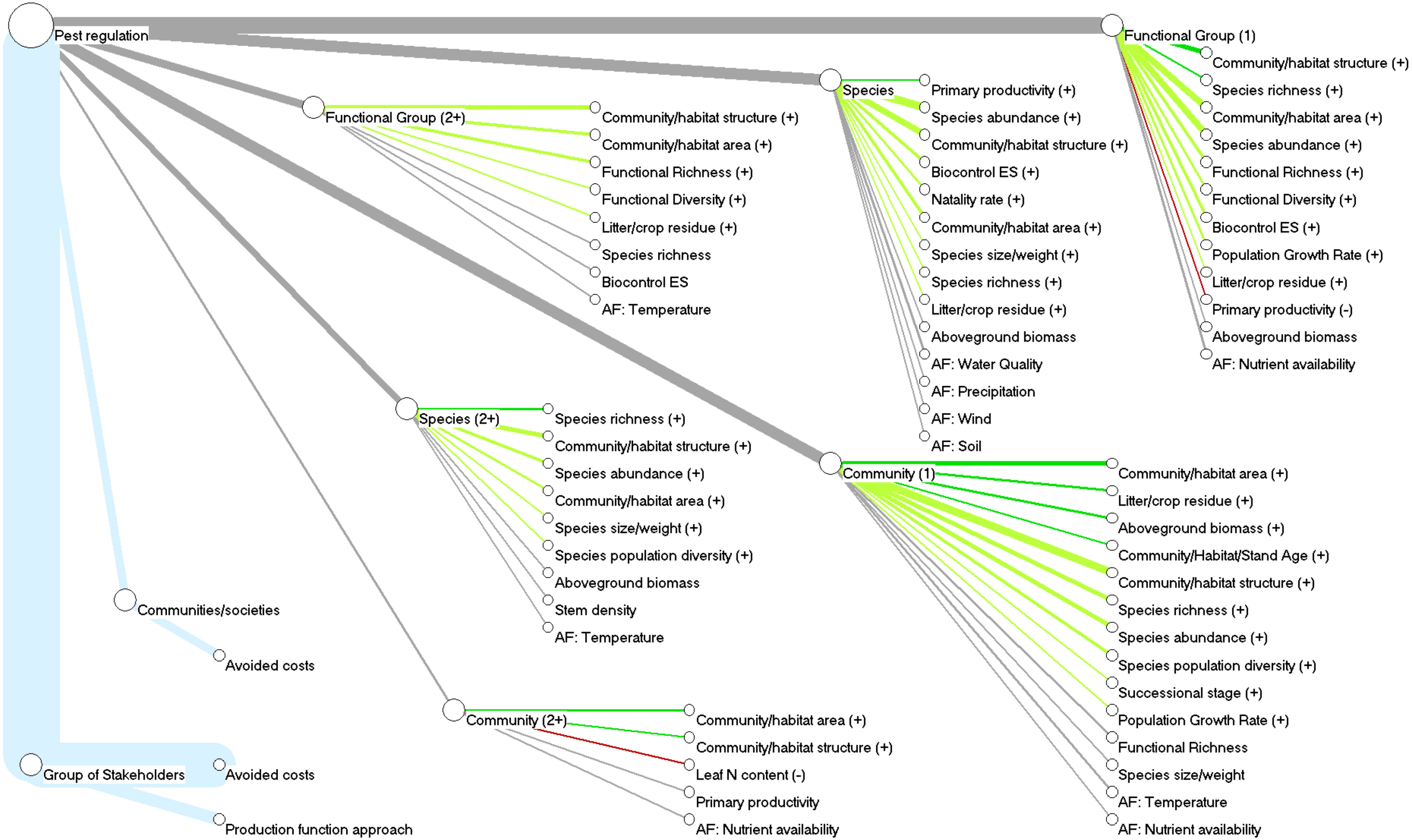


Figure 3.10 Network for the service of pollination. *(1) indicates that the class is for a single ESP; (2+) indicates two or more of the same class; (+) and (-) after the name of an ESP or attribute indicates the direction; "BT and reviews" = "benefit transfer and reviews"; "AF" = abiotic factor.*

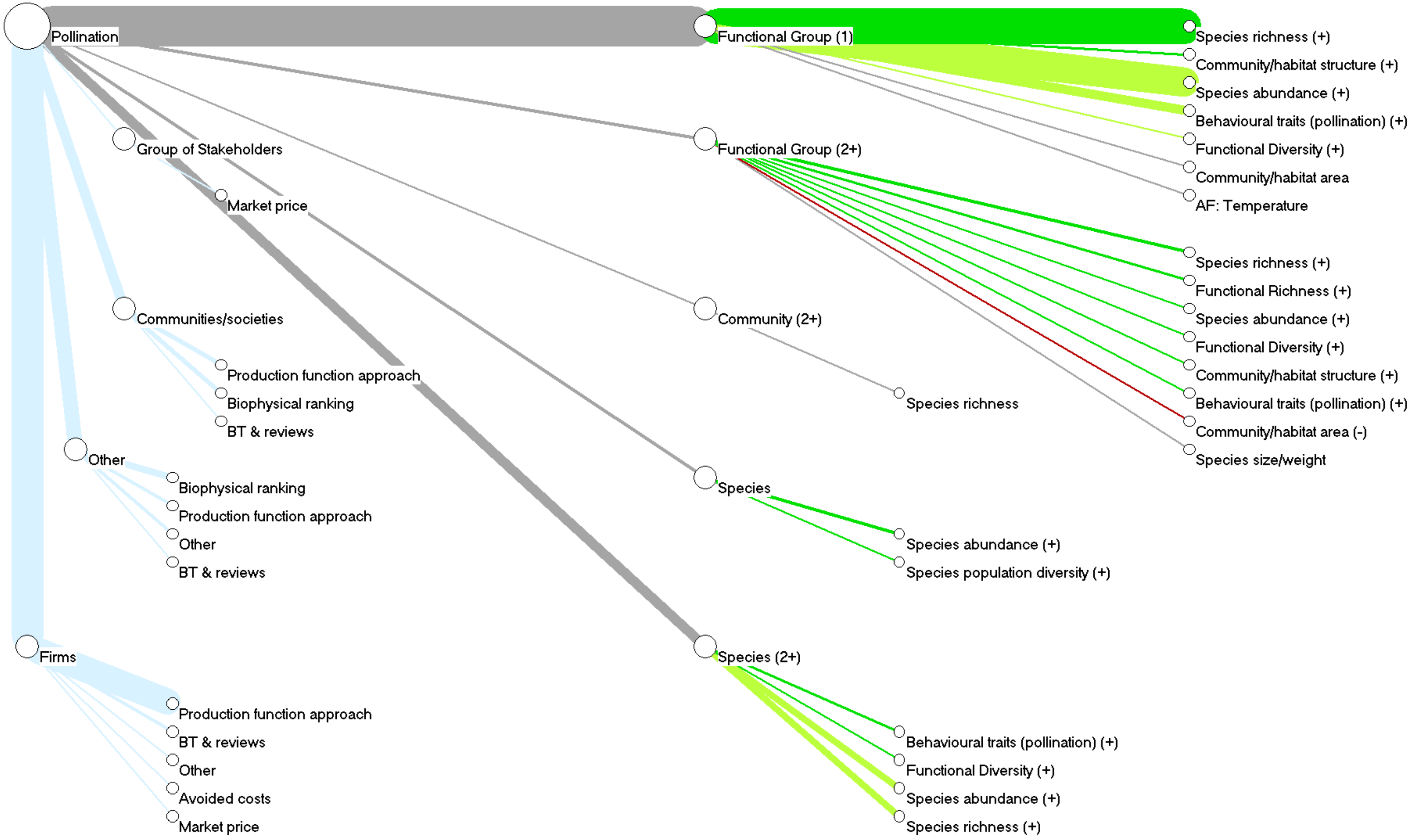


Figure 3.11 Network for the service of recreation (species-based). *(1) indicates that the class is for a single ESP; (2+) indicates two or more of the same class; (+) and (-) after the name of an ESP or attribute indicates the direction; "BT and reviews" = "benefit transfer and reviews"; "AF" = abiotic factor.*

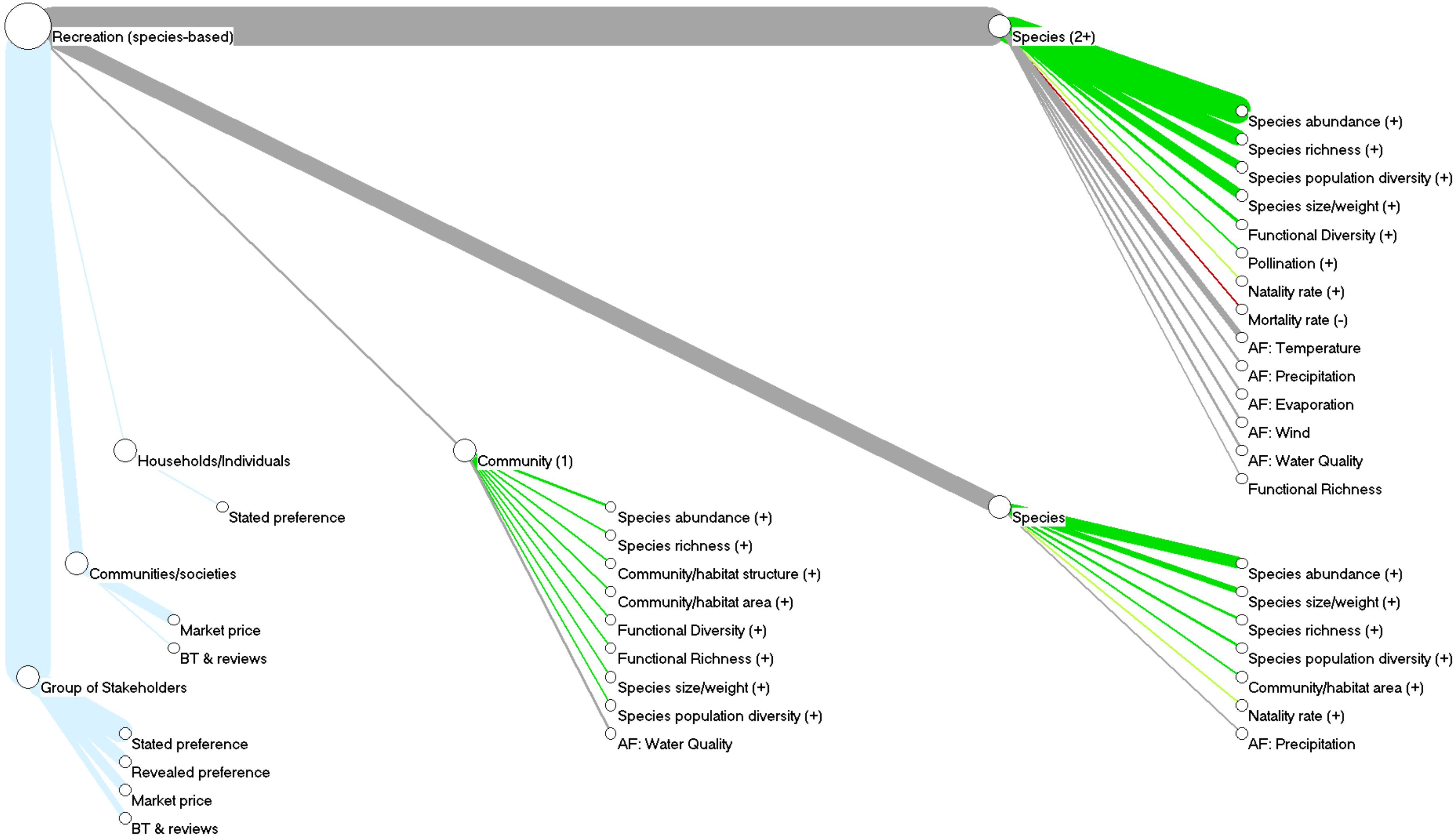
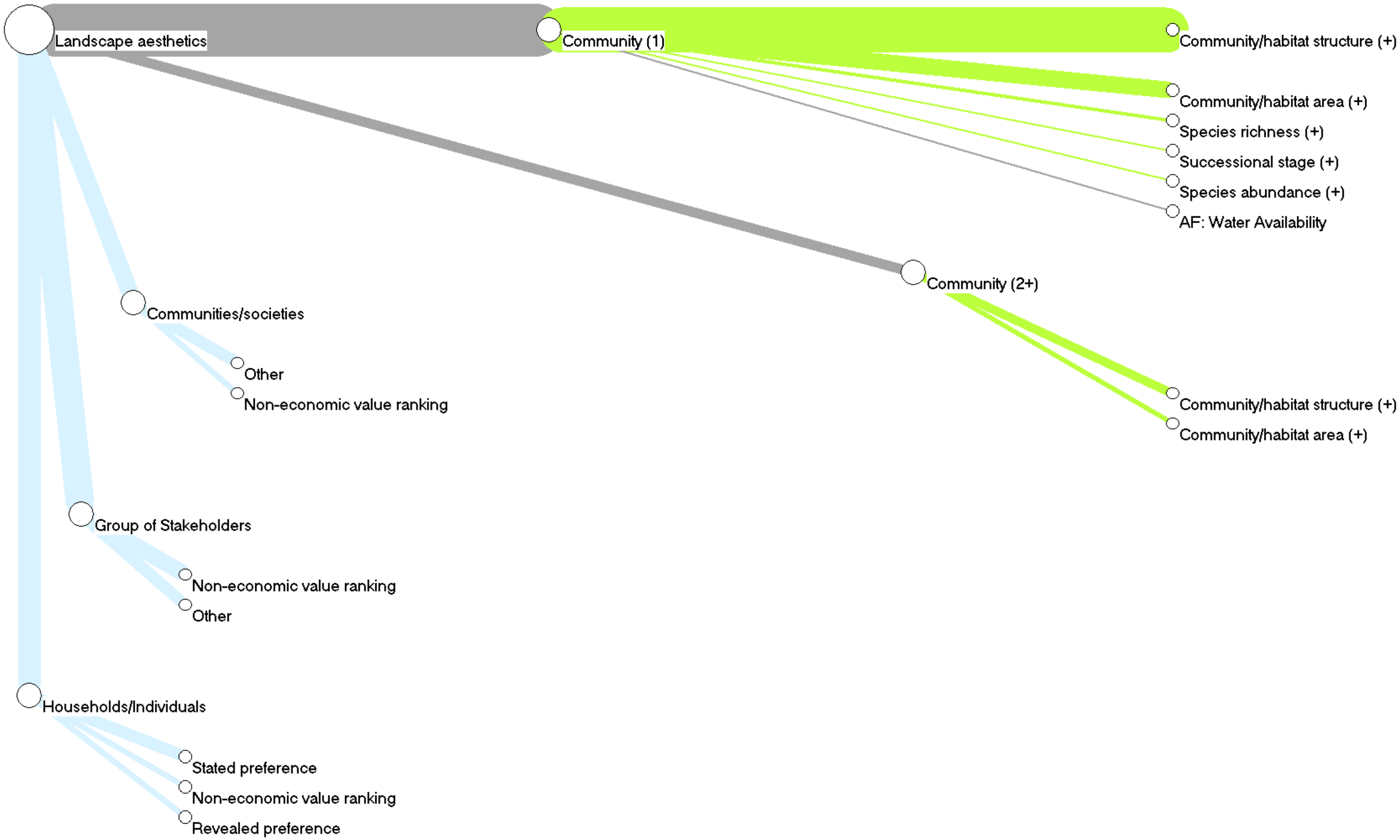


__Figure 3.12 Network for the service of landscape aesthetics.__ (1) indicates that the class is for a single ESP; (2+) indicates two or more of the same class; (+) and (-) after the name of an ESP or attribute indicates the direction; "BT and reviews" = "benefit transfer and reviews"; "AF" = abiotic factor.



4. Discussion and conclusions

Previous studies have provided valuable information on the role of biodiversity in ecosystem service delivery from a theoretical perspective (Mace et al., 2012), explored the links between functional traits and ecosystem services (de Bello et al., 2010) or examined how biodiversity influences the functioning of ecosystems and, thus, their ability to provide ecosystem services (Cardinale et al., 2012). Other studies have estimated the value to society of ecosystem services (UK National Ecosystem Assessment, 2011; Bateman et al., 2011). In this study we have gone a step further to build up the scientific knowledge base on ecosystem service providers (ESPs), their beneficiaries, their biodiversity attributes, the direction and strength of evidence for these relationships, the influence of abiotic factors, and the visualisation of the linkages using network analysis. The results show an intricate array of linkages between biodiversity attributes related to ESPs and ecosystem services and their value to society. Overall, our results add weight to the emerging realisation that the relationships between biodiversity and the provision of ecosystem services are highly complex and involve many uncertainties (Balvanera et al., 2014). The results also show that ecosystem services in many cases are highly valued by society, which may further strengthen arguments for biodiversity protection.

The detailed networks for each of the selected ecosystem services demonstrate particular hierarchies and the immense complexities of the relationships between biodiversity and service provision. Nevertheless, some dominant trends emerge. Five biodiversity attributes stand out as being particularly important with each being cited in over 50% of papers for at least one ecosystem service (Table 3.2). These are species abundance (freshwater fishing, pollination, species-based recreation), species richness (timber production, pollination), species size/weight (freshwater fishing), community/habitat area (freshwater provision, water purification, water flow regulation) and community/habitat structure (pest regulation, landscape aesthetics). Three further biodiversity attributes are notable, being reported in between 25 and 50% of papers for at least one ecosystem service: community/habitat age (freshwater provision, atmospheric regulation, water flow regulation), and above- and belowground biomass (mass flow regulation and atmospheric regulation). These dominant attributes tend to be at the species and community levels, but functional group attributes were also frequently cited as being important for pollination and pest regulation (in 14 to 22% of papers). Some of these relationships may appear to be commonplace or a matter of trivial logic, but we believe that this is the first time that such linkages have been clearly documented in a comparative context.

Although it may be tempting to consider only the “simpler” pattern of thicker lines, representing the most often cited links in the network diagrams, it is the inclusion of the less frequently cited linkages of thinner lines, which may be functionally just as important, that reveals the full degree of interdependence and complexity across organisational levels. A summary of the linkages between broad categories of biodiversity attributes, ESPs and ecosystem services for all the services included in this review is provided in Figure 4.1. This emphasises the range of organisational levels involved in the overall relationship between biodiversity and ecosystem services with each service being linked to a number of ESPs and biodiversity attributes that together span multiple spatial scales. The attributes identified at a particular organisational level can be of importance to ESPs operating at different levels of organisation. For example, an attribute of a species population such as “abundance of individuals” can have important links to ESPs that are defined at the functional group level (e.g. pollinators) or at entire community levels (e.g. atmospheric regulation). Similarly, functional group or community level attributes (e.g. functional richness, community/habitat area) can be important for ESPs that work on one or more species population levels. Note that it is equally valid to view these links from the opposite direction, so that it may be said that ESPs operating at any particular scale draw upon biodiversity attributes that are characteristic of a variety of different organisational levels. The important point is that there is an interdependence between ESPs and their attributes that feeds the delivery of the different ecosystem services via a very complex network that is a consequence of the range and variety of linkages within the detailed networks associated with the single services (Figures 3.2 to 3.12). This is supported by the literature review of de Bello et al. (2010) who provide a similar diagram to Figure 4.1, highlighting the multiple associations between traits and ecosystem services across different organisational levels (see Figure 1.1).

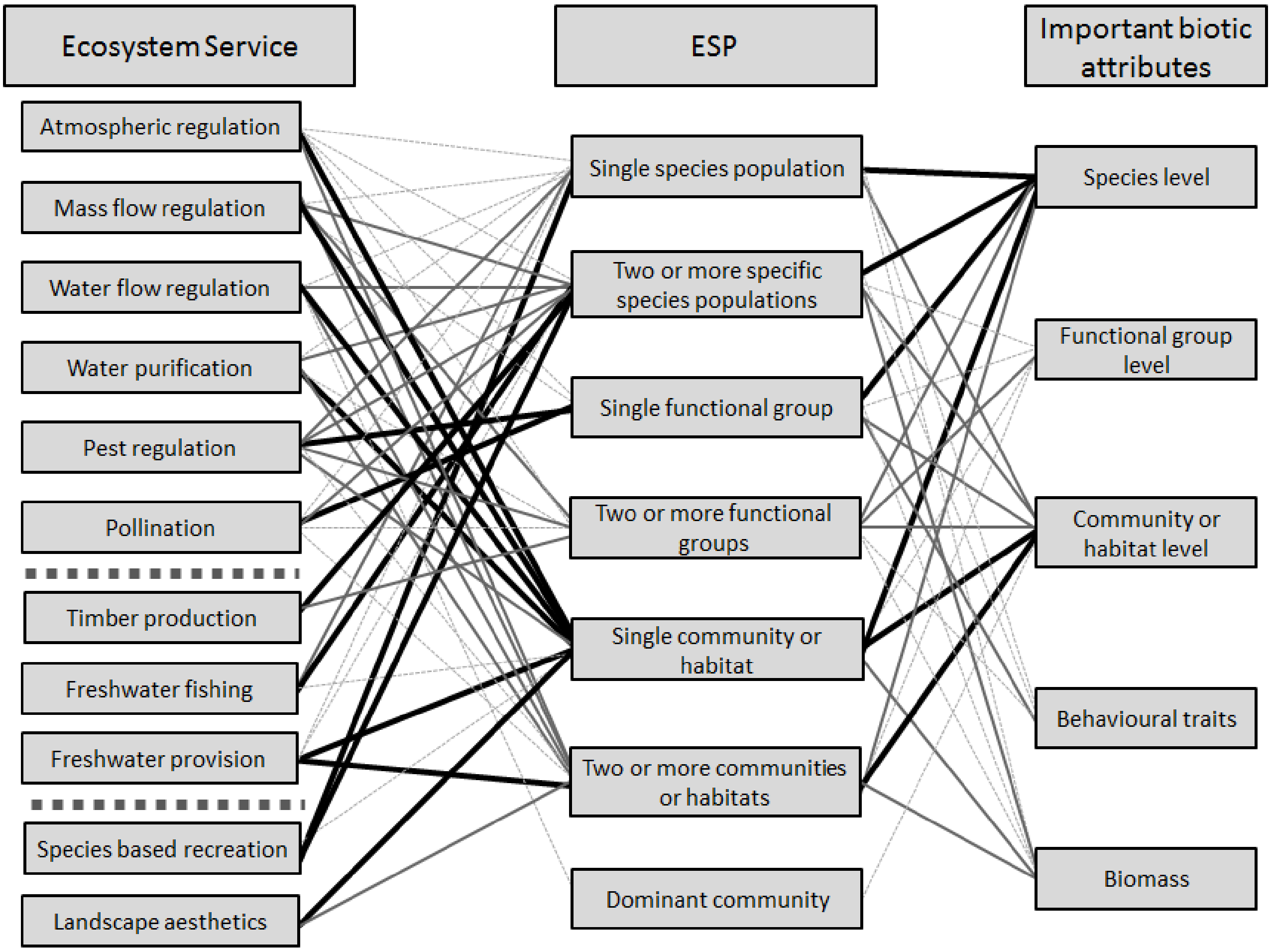


Figure 4.1: *Linkages between broad groups of biodiversity attributes, ESPs and ecosystem services for the 11 ecosystem services included in the literature review. Species level attributes include species richness, diversity, abundance, size and weight; functional group level attributes include functional diversity and functional richness; community or habitat level attributes include community/habitat area, age, structure and successional stage; behavioural traits include flower visiting behaviour and biocontrol; biomass attributes include above and belowground biomass and litter or crop residue. The thickness of the connecting lines reflects the number of papers providing evidence for that linkage equally divided into three categories with the thickest lines representing the most frequently cited linkages.*

In addition to identifying the linkages between biodiversity attributes and ecosystem services, the direction of the relationship (positive, negative or unclear) was captured wherever possible (as summarised in Table 3.3 for relationships cited in at least 10% of papers). This confirms the strong evidence base supporting a positive relationship between community/habitat area and water purification and water flow regulation, between community/habitat structure and landscape aesthetics, and between species richness and pollination. It also shows that considerable uncertainties still exist for some relationships with evidence being spread across different attributes, or evidence being mixed (i.e. showing both positive and negative trends). This is supported by

Cardinale et al. (2012) who also found that evidence for effects of biodiversity on ecosystem services was often mixed. In their review of relationships between different diversity levels (species, genetic and trait) and a range of provisioning and regulating ecosystem services, positive trends were found for wood, fisheries, carbon storage, pollination and freshwater purification and negative trends for pest control. The results are not directly comparable with our study, but together they considerably increase the evidence base and highlight gaps in knowledge.

Although attention is usually focussed on the positive linkages, and these dominate in the present literature review (Table 3.3), the negative relations between biodiversity and ecosystem service provision are also very important to the overall dynamics. The components of biodiversity responsible, the ecosystem service antagonisers (ESAs), can be defined at any level and may have direct or indirect disruptive effects on service provision (Harrington et al., 2010). Importantly, what is a provider for one service (i.e. an ESP) can at the same time act as an ESA for other services. For example, many links between biodiversity attributes and the provision of freshwater were predominantly negative (Figure 3.4) because vegetation parameters such as increased stem density, biomass and age, particularly in relation to trees, have a direct negative effect by sucking water out of the system. However, these same vegetation attributes may be important for the provision of many other services, such as atmospheric regulation or landscape aesthetics.

We also found that regulating services were often associated with more ESPs and biodiversity attributes than other categories of ecosystem services as shown in Tables 3.1 and 3.2 (based on the 11 services investigated in this study). This is supported by analysis of the network diagrams which shows that the linkages were generally more complex and branched (i.e. each node was associated with a greater number of connections) for regulating services (as shown in Figures 3.8 and 3.9 for atmospheric regulation and pest regulation) compared to provisioning services (as shown in Figure 3.4 for freshwater provision). This underscores the significant role that biodiversity plays as a regulator of ecosystem processes (Mace et al., 2012). It also highlights the importance of further research to understand how different services interact with each other and the biodiversity attributes that underpin them, as the condition of regulating services (often referred to as intermediate services by some authors; e.g. Fisher et al. 2009) can be critical in mediating the delivery of other services.

The review of valuation studies has demonstrated that several valuation approaches have been employed to estimate the value of ecosystem services. It also demonstrates that for each ecosystem service only a few valuation approaches should be preferred (as shown in Table 3.4). For provisioning services (timber production, freshwater fishing and freshwater provision) that have use value, market prices should be used when they exist. If market prices do not exist, the production function approach can be used.

With regard to regulating services, which mainly have indirect use value, the picture is not so clear. For protecting services (flood protection, erosion protection and pest regulation) the avoided cost approach is the most natural approach to use. It is used in most of the reviewed studies. The production function approach can also be used in relation to pest regulation. The water purification service has both use and non-use value and, therefore, the stated preference approach is and should be used, as it is the only valuation approach that can estimate non-use values. The ecosystem service of carbon sequestration is very difficult to value. In principle, it should be valued on the basis of avoided damage costs of climate change. Some international estimates of these costs have been made and the results have been transferred to some of the reviewed studies (the benefit transfer approach). However, the damage cost estimates are very uncertain and therefore in other studies the costs of alternative mitigation measures (the averting behaviour approach) have been used to value carbon sequestration. Finally, in most of the reviewed studies the value of the pollination service is estimated by use of the production function approach. This approach is preferred because pollination has indirect use value to society by its contribution to food production.

The cultural services include recreation and landscape aesthetics. Recreation activities have direct and indirect use value to society and their value can be estimated on the basis of market prices if these exist (e.g. payment for whale watching and hunting licenses). This is done in some of the reviewed studies. For recreation services where no market prices exist, the stated preference and revealed preference approaches are the most used valuation approaches. The service of landscape aesthetics has both use and non-use value (heritage areas) to society. Very few proper valuation studies of landscape aesthetics have been made until now. Perhaps this reflects that it is a very intangible service which is difficult to handle within the traditional and relevant valuation approaches (the stated preference and revealed preference approaches).

Two conditions characterise the work with valuation of ecosystem services up till now. First, in most cases the values have been estimated locally or regionally. This means that the results have limited geographical validity and great care should be exhibited in transferring of values from one area to another. General prices can only be expected for services such as timber production (world market prices) and carbon sequestration (global damage costs of climate change). Secondly, valuation estimates are very uncertain. Thus, valuation results show great dispersion even between studies valuing the same service for the same area. Therefore, the results should be used with care and much more work needs to be done before valuation results become reliable for all ecosystem services.

It is clear from the review that for many ecosystem service valuations there are very few studies that explicitly make the link to biodiversity. There are exceptions to this, as pollination and pest control specifically value aspects of biodiversity. Recreation service studies also link directly to biodiversity by valuing recreation to view specific species. For most of the other studies reviewed, authors rarely mention biodiversity explicitly.

Our review focused on a sample of ecosystem services across the different service categories and did not consider interrelationships between services. Information on such aspects could be extracted from the papers which emerged from this review, but a different approach specifically focused on assessing synergies and trade-offs between ecosystem services and their relationship to the underlying biodiversity and values may reveal new insights. Further research to expand the search to cover more services and to explore how the linkages identified differ by ecosystems or biogeographical region would also be useful for targeting the management of biodiversity and service provision. The complex relationships we have reported here are based on only the frequencies of citation in the literature, which is not necessarily the same as functional importance (see section 2.2). Nevertheless, the results could be used to guide which biodiversity attributes should be the focus of future research to advance understanding of the functional importance of biodiversity for ecosystem service supply. More specifically, additional research is needed to better understand the linkages represented by the thin lines in the networks – the linkages exist, but how strong are their functional roles in joining the different aspects of biodiversity with the provision of ecosystem services in the amounts required by beneficiaries?

Our review focuses on the relationships between biodiversity, ecosystem services and values. However, to support management and protection strategies, future research should also take account of effects of other socio-economic factors and land use decisions on different components of biodiversity and, as a result, ecosystem service delivery. Incorporating traditional conservation strategies for species and habitat protection within the broader context of social-ecological systems and ecosystem service delivery can lead to added benefits for biodiversity through closer integration of conservation policy with policies in other sectors (Haslett et al., 2010). This approach usefully extends conservation effort beyond the borders of protected areas, to encompass many species with presently widespread distributions as well as other aspects of biodiversity occurring in non-protected areas. But the social-ecological system approach also has the capacity to considerably improve conservation management effectiveness within protected areas (Palomo et al., 2014). Biodiversity is not (just) a good to be conserved for its intrinsic value, but has a critical role in ecosystem processes (Mace et al., 2012) that provide essential services to humans (Cardinale et al., 2012). Improving understanding of when the goals of biodiversity conservation and ecosystem service maintenance are compatible or interdependent (Balvanera et al., 2014) requires a strong knowledge base on biodiversity – ecosystem service – value linkages within different socio-ecological systems. Further expansion of this knowledge base through closer examination of the less well studied relationships depicted in this review may help to reveal additional or new arguments for the need to conserve biodiversity in all its guises.

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Appendix 1: Keywords used in the literature review on linkages between biodiversity and ecosystem services

Table A1.1: *Keywords used for the systematic search of linkages between biodiversity and the 11 ecosystem services. All searches used the same biodiversity terms: Biodiversity, "Biological diversity", Species, Habitat*, Genetic, Trait*, Function*, Landscape, Richness, Abundance.*

Ecosystem service/disservice	Additional service related terms
Timber production:	
Wood	"Tree diversity"
"Wood production"	
Timber	
"Timber production"	
Freshwater provision (quantity):	
"Water provision"	Forest*
"Fresh water provision"	Vegetation
"Freshwater provision"	Soil
"Water suppl*"	Sapwood
"Drinking water provision"	Leaf*
"Drinkable water provision"	
"Potable water provision"	
Water flow regulation (flood protection):	
"Flood protection"	Tree*
"Flood defence"	Biomass
Wetlands	Topography
"Flood storage"	"Vegetation cover"
Flooding	
"Flood generation"	
"Flood detention"	
"Flood event"	
"Natural retention"	
"Flood attenuation"	
Atmospheric regulation:	
"Carbon storage"	Tree*
"Carbon sequestration"	Soil*
"Carbon loss"	Biomass
"Carbon emissions"	
Pollination:	
"Pollination services"	"Wild plants"
Pollinators	"Crop plants"
Freshwater fishing:	
"Freshwater fishing"	Trout
"Quantity of caught fish"	Salmon
"Fish yield"	Carp
"Fisheries management"	Freshwater eel
"Aquaculture performance"	Perch
"Impact of fish polyculture on yield"	
"Impact of fish monoculture on yield"	
"Effect of stocking density on freshwater fishing"	
"Relationship between fish abundance and yield"	
Water purification (quality):	
Water quality	Tree*
Water regulation	Soil*
Water purification	Forest*
Nutrient* retention	Vegetation
Nutrient* translocation	Plant*
	Pollutant*
	Wetland*
	Microorganism*
	Accumulation
	Sediment*
Mass flow regulation (erosion protection):	
"Erosion protection"	Mountain
"Soil erosion"	Riverbanks
"Soil stability"	"Road side"
"Sand stability"	"Sand dune"
	Farmland
Pest regulation:	
"Natural pest control"	None
"Pest control"	
"Biological control"	
"Biological pest control"	

Ecosystem service/disservice	Additional service related terms	Ecosystem service/disservice	Additional service related terms
"Pollinator efficiency"	Fruit	*Recreation (species-based):*	
"Flower visitors"	"Seed set"	Recreation	Angling
Zoophilous	Horticulture	Tourism	Hiking
Landscape aesthetics:		Aesthetic	Birding
Tourism	None	"Aesthetic value"	Hunting
Recreation	*Note: only*		Photography
Appreciation	*Biodiversity,*		Bat
Preference*	*Habitat* and*		Fishing
Perception	*Landscape used*		
Aesthetic	*for the Biodiversity*		
Valuation	*terms*		

Note: The wildcard [*] is used to pick up all keywords starting with the preceding characters and apostrophes are used to ensure that multi-word searches use the combined terms [e.g. "Biological Diversity"].

Appendix 2: Definitions of ESPs and biodiversity attributes used in the literature review

Table A2.1: *Definition of ESPs.*

Ecosystem service provider (ESP)	Definition
Single specific species population	A group of organisms, all of the same species, which occupies a particular area (geographic population), is genetically distinct (genetic population) or fluctuates synchronously (demographic population).
Single functional group	A collection of organisms with similar functional trait attributes in the study area, i.e. a feature of an organism, which has demonstrable links to the organism's function. Sometimes referred to as a guild, especially when referring to animals.
Entire community or habitat	An association of interacting populations, usually defined by the nature of their interactions or by the place in which they live.
Dominant community	Defined either qualitatively or quantitatively based on the paper.

***Table A2.2**: Definition of biodiversity attributes or traits.*

Biodiveristy attributes or traits	**Definition**
Species abundance	Number of individuals of a species expressed per unit area or volume of space. Synonymous with species population density.
Species richness	Number of different species represented in a set or collection of individuals.
Species population diversity	The number, size, density, distribution and genetic variability of populations of a given species.
Species size or weight	Includes body size or weight, diameter at breast height (DBH) for trees, species/vegetation/tree height, basal area defined as the cross section area of the stem or stems of a plant or of all plants in a stand, generally expressed as square units per unit area) (there is a free text box to specify the type of measurement).
Functional richness	The number of functional groups or trait attributes in the community.
Functional diversity	Range, actual values and relative abundance of functional trait attributes in a given community.
Community/habitat type	Name of habitat or ecosystem.
Community/habitat area	Includes width or diameter, i.e. for buffer zones.
Community/habitat structure	In terms of complexity (amount of structure or variation attributable to absolute abundance of individual structural component) and heterogeneity (kinds of structure or variation attributable to the relative abundance of different structural components).
Primary productivity	Rate at which plants and other photosynthetic organisms produce organic compounds in an ecosystem.
Aboveground biomass	The total mass of aboveground living matter within a given area.
Belowground biomass	The total mass of belowground living matter within a given area.
Sapwood amount	Include allocation of carbon to sapwood and sapwood area.
Stem density	Measured as the number of stems/specified area
Wood density	Measured as the weight of a given volume of wood that has been air-dried.
Successional stage	Changes in the number of individuals of each species of a community and by establishment of new species populations that may gradually replace the original inhabitants; categorised into early and late stages.
Habitat/community/stand age	Includes young and old-growth forests, even and uneven-aged forests, or can specify the age.
Population growth rate	Change in the number of individuals of a species in a population over time.
Mortality rate	Number of deaths of individuals per unit time.
Natality rate	Number of new individuals produced per unit time.
Life span/longevity	Duration of existence of an individual/expected average life span.
Litter/crop residue quality	Quality of plant litter with respect to decomposition often defined by the C:N ratio, but ratios of C, N, lignin and polyphenols are other chemical properties and particle size and surface area to mass characteristics are physical properties.
Flower-visiting behavioural traits	Well suited to the system to provide pollination ES. Free text box will allow the behavioural type/preference/strategy to be entered.
Predator behavioural traits	Well suited to the system to provide biocontrol ES. Free text box will allow the behavioural type/preference/strategy to be entered.

Appendix 3: Full literature review results

A3.1 Timber production

A3.1.1 Results – Biodiversity and timber production linkages

Ecosystem type and study location

Timber production can be realised in (natural) forests or in plantations. Both habitats are the subject of this review. 51% of the case studies (18 entries) were in forests and 49% (17 entries) were in plantations. Where relevant we split the results.

The locations of the case studies were well spread over the climate types and continents of the world (Table A3.1.1). However, papers dealing with plantations were mainly focussed on the tropics.

Table A3.1.1: *Locations of the cases, classified by climate type, continent, country and habitat type (forest and plantation).*

Climate type	N	Continent	Country	Forest	Plantation
Boreal	3	Europe	Finland	1	
		North America	Canada	2	
Temperate	16	North America	USA	2	2
			Canada	3	
		South America	Chile	3	
		Oceania	Australia		3
		Europe	Germany	1	
			Sweden	2	
Mediterranean	5	Europe	Spain	3	
		North America	USA	1	
		Asia	Iran		1
Tropical	11	North America	Costa Rica		6
			Puerto Rico		1
		Oceania	Australia		3
		Asia	Philippines		1
Total	35			18	17

Spatial and temporal scale

60% of the studies were at the local scale, whilst 37% were at the sub-national scale. Only one study (in Canada) was at the national/sub-continental scale (Paquette & Messier, 2011). 51% of the entries in the database have a decadal time scale and 49% describe relationships on an annual base.

Ecosystem service provider (ESP)

For most of the studies (28, i.e. 80 %) the ESP was two or more specific species populations (Figure A3.1.1). In these studies more than half of them (16 or 57%) were plantations for timber production. They consisted of two or a few more tree species, planted in pure and mixed proportions. In the case of two tree species (A and B), these trials often use the replacement series design of:

- 100% A, 75% A + 25% B, 50% A + 50% B, 25% A + 75% B and 100% B, such as in the Australian studies of Bauhus et al. (2004) with *Eucalyptus globulus* ssp., *Pseudoglobulus* and *Acacia mearnsii* and Bristow et al. (2006) with *Eucalyptus pellita* and *Acacia peregrine*.

- Pure and 50% A/50% B mixtures, such as in Amoroso & Turnblom (2006) with Douglas-fir (*Pseudotsuga menziesii*) and Western hemlock (*Tsuga heterophylla*) in the Pacific Northwest of the USA.

The remaining cases with two or more species populations were forests (12 or 43%). They consisted of a limited number of species, such as Scots pine (*Pinus sylvestris*) and silver birch (*Betula pendula*) in the boreal forest study in Southern Finland (Hynynen et al. 2011). However, there were cases where a larger number of tree species were important, such as in the case of the spruce-dominated forests of Canada which contained the following most frequent tree species: black spruce (*Picea mariana*), white spruce (*Picea glauca*), red spruce (*Picea rubens*) and balsam fir (*Abies balsamea*) (Lei et al., 2009).

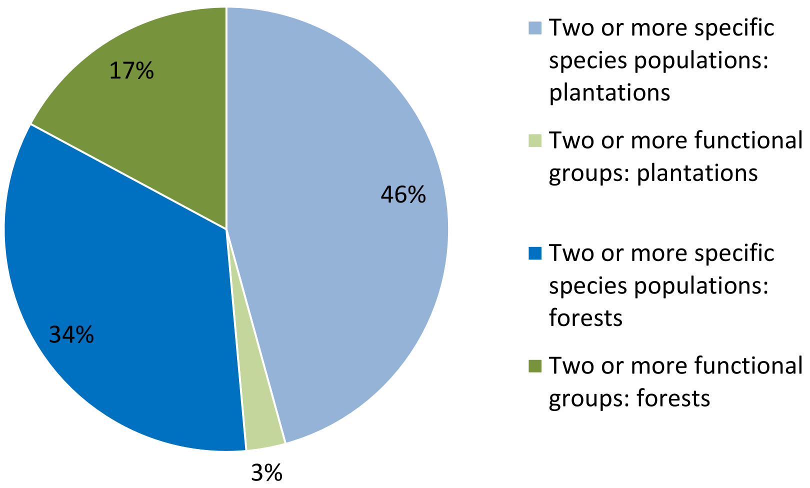

Figure A3.1.1: *Categorisation of the ecosystem service provider (ESP) for the service of timber production, showing the distinction between forests and plantations.*

For seven studies (20%) the ESP was two or more functional groups. Most of them (6 or 86%) were forests rather than plantations. They consisted of at least two, but most often more, dominant tree species, and a great number of accompanying tree species. Examples are the pine-dominated Mediterranean forests in Catalonia, Spain (Vila et al., 2003) and the tropical plantation in the Philippines which consists of 77 species belonging to 32 families and 57 genera (Nguyen et al., 2012).

Important attributes of the ESP

All studies focused on species type, and studied forests and plantations with economically important tree species, such as (among others) conifers and eucalyptus. Species richness was the most important attribute (in 31 or 89% of cases), with a positive relationship in 19 (61%) studies, evenly divided over forests and plantations (Figure A3.1.2). One example of a strong positive relationship between species richness and timber production in forests was the study by Liang et al. (2007) in the Douglas-fir/western hemlock (*Pseudotsuga menziesii/Tsuga heterophylla*) forests of Oregon (USA) (coexisting with many other tree species in natural stands, in particular *Alnus rubra, Thuja plicata* and *Acer macrophyllum*) and in the mixed-conifer forests of California (USA) with the dominant tree species *Pinus ponderosa, P. jeffreyi, P. lambertiana, P. menziesii, Abies concolor* and *Libocedrus decurrens*. Further, the study by Piotto et al. (2010) in tropical forest plantations with native tree species on abandoned pasture land in Costa Rica demonstrated that in the long-term (15 - 16 years), regardless of species composition, mixed plantations performed better than pure stands for all variables considered, including basal area, total volume and aboveground biomass.

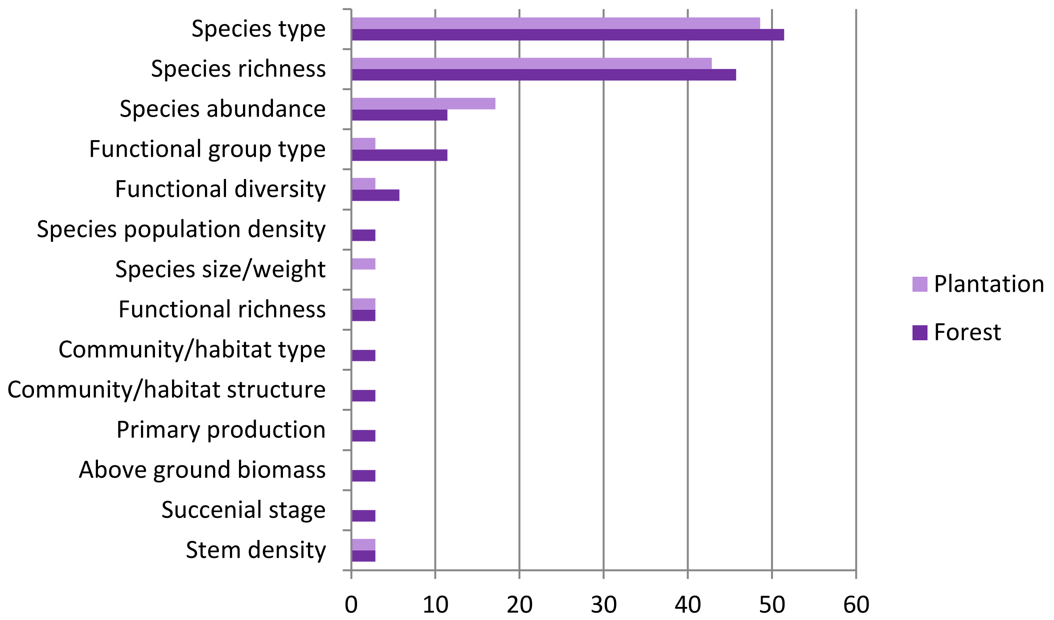

Figure A3.1.2: *Categorisation of the ESP attributes for the service of timber production (%), showing the distinction between forests and plantations. Note: some studies used multiple attributes.*

A negative relationship was found in 8 (26%) studies (5 in forests, 3 in plantations). In forests, the negative relationship between species richness and timber production is mainly recorded in rather old-growth forests. This was the case in the boreal forest in Southern Finland (Hynynen et al. 2011) with a negative impact of birch competition on the growth of pine trees. Jacob et al (2010) came to the same conclusion in the temperate deciduous forests in Germany, where they investigated forest stands differing in tree taxa diversity which were grouped into three diversity levels: (i) four forest stands of European beech *Fagus sylvatica* contributing to 85 - 100 % of the total tree basal area; (ii) four stands mainly consisting of beech, lime (*Tilia cordata and T. platyphyllos*) and ash *Fraxinus excelsior*; (iii) four stands with five dominant tree taxa (beech, lime, ash, hornbeam (*Carpinus betulus*) and maple (*Acer pseudoplatanus* and *A. platanoides*). Furthermore, the impact of the conifer species Douglas-fir (*Pseudotsuga menziesii*) and Western hemlock (*Tsuga heterophylla*) on the productivity of mixed plantations in the northwest of North America was obviously negative in comparison with pure stands (Amoroso & Turnblom, 2006). In 4 (13%) cases the relationship was unclear.

Species abundance was the second most important attribute, with a positive impact in 7 (20%) cases (3 in forests, 4 in plantations), a negative impact in 1 case (forest) and an unclear relationship in 1 forest case and 1 plantation case.

The important attributes when the ESP was two or more functional groups were functional richness which was an important positive attribute in 1 case and negative in 1 case, and functional diversity which was positive in 2 cases and negative in 1 case.

Other attributes were of minor importance (often only 1 case per attribute), and were mostly mentioned indirectly, and this was mostly for forests.

Discussion

The evidence for linkages between the ESP, its attributes and the ecosystem service of timber production was mainly based on the interpretation by the author. This was mostly derived from quantitative interpretations, supported by statistical analysis and graphs. As already mentioned above, most cases gave a positive relationship between timber production and biodiversity.

The cases with strong evidence are equally shared between plantations and forests. 10 records (29%) showed a positive relationship between timber production and species richness or abundance. The 2 negative cases concerned the rather old-growth forests in Finland (Hynynen et al., 2011) and Germany (Jacob et al., 2010).

Table A3.1.2: *Classification of the cases by strength of evidence.*

Strength of evidence	Records (N)	Records (%)
Very weak	0	0
Weak	4 (2 +ve)	12 (6 +ve)
Average	19 (12 +ve)	54 (34 +ve)
Strong	12 (10 +ve)	34 (29 +ve)
Very strong	0	0

The relationship between timber production and biodiversity was mostly rather clear and evident. However, where the relationship between timber production and biodiversity was positive, this was not always purely linked to species diversity, but also to diversity in species properties, such as shadow tolerance and intolerance, resulting in niche complementarity. The study of Chen et al. (2003) who compared three distinct situations is a good example:

- Study 1: western red cedar - western hemlock (*Thuja plicata - Tsuga heterophylla*): both shade tolerant: impact of species richness was unclear.
- Study 2: lodgepole pine - western larch (*Pinus contorta - Larix occidentalis*): both shade intolerant: impact of species richness was negative.
- Study 3: lodgepole pine - black spruce (*Pinus contorta - Picea mariana*): shade intolerant pine and shade tolerant spruce: impact of species richness was positive.

Finally, the impact of abiotic factors was rarely included in the studies.

A3.1.2 Results – Timber production and value linkages

The results from the review of valuation of timber production are summarised in Table A3.1.3.

It can be seen from Table A3.1.3 that forests in all of the 12 reviewed studies are listed as the ecosystem service provider. The results also show that all 12 studies assess direct consumptive use value. Several studies, however, assess several values; hence, in addition to the direct consumptive use values, 5 studies also assess non-consumptive use values, 2 assess indirect use value, 6 assess non-use value and finally 1 study assesses option value. In terms of spatial scale, 6 studies are local, 2 are regional and 3 are national while 1 is not specified. The beneficiaries are listed as firms (2 studies), groups of stakeholders (8) and society (6 studies). The market price method is seen to be the most common valuation method (used in 10 out of 12 studies), and this corresponds well with the fact that the main emphasis is on valuation of direct consumptive use values.

Table A3.1.3: *Results from the review of valuation of timber production.*

	Market prices	**Benefit transfer and reviews**	**Other**	**Total**
Ecosystem service provider				
- forest	10	1	1	12
Value				
- direct consumptive use	10	1	1	12
- direct non-consumptive use	4		1	5
- indirect use	2			2
- non-use	4	1	1	6
- option	1			1
Spatial scale				
- local	5		1	6
- regional	2			2
- national	2	1		3
- other	1			1
Beneficiaries				
- firms	2			2
- communities/society	6	1	1	8
- group of stakeholders	8	1	1	10

Discussion

There are few papers in the peer reviewed forest economics literature that “just” value timber production, without specific methodological twists. Standard estimation of timber production values is a task carried out by public or private evaluators, from a profit maximisation point of view. However, the peer reviewed articles include some that value several types of services, in addition to timber provisioning services.

Some observations:

- Naturally, almost all studies use some estimate of market prices to value the provisioning services of timber. These studies, in the forest economics tradition, value the discounted sum of timber values over time.
- These studies evaluate the provisioning service from a forest manager point of view.
- Other services are valued using different types of methods, including benefit transfer, market prices (carbon), stated preference surveys or as a cost of a constraint arising from specific regulations that take such services into account.
- When other services are included, the studies usually have a society point of view.
- There are only a few studies that evaluate biodiversity in combination with timber provisioning services.
- Biodiversity is usually quite vaguely defined, and almost always linked to species/wildlife abundance, etc.
- Note that there is a large literature that values forest biodiversity and other non-timber services more directly (i.e. not in connection with estimates of timber values), especially using stated preference surveys.

A3.2 Freshwater fishing

A3.2.1 Results – Biodiversity and freshwater fishing linkages

Spatial and temporal scale

Very few articles (only 10%) discussed the temporal scale at which this ecosystem service was provided. There were, however, cases which discussed annual catch or performance (e.g. Tucker et al., 1994; Rahman et al., 2008), or seasonal fishing on floodplains (e.g. Mollot et al., 2003). One entry concerning recreational fishing considered freshwater fishing on a daily timescale (Adamowicz et al., 1994); however this was the only study to do so. Many studies (42%) discussed fishing as being provided at a local scale, examining individual fisheries (e.g. Frei & Becker, 2005; Milstein et al., 1988). The second most common scale was sub-national, discussed in 31% of articles. Examples include a study by Hasan & Middendorp (1998) who discuss optimal stocking densities in oxbow lakes in western Bangladesh, and Sugunan & Katiha (2004) who examine the impact of stocking on yield in a number of reservoirs in Andhra Pradesh, India. Around 16% of articles examined freshwater fishing at a global level or in global review studies (e.g. Birkeland & Dayton, 2005; Downing et al., 1990; Milstein, 1992). Very few articles discussed freshwater fishing at the national or continental scales.

Ecosystem service provider (ESP)

The majority of articles found the service of freshwater fishing to be provided by either two or more specific species populations (69%) e.g. bighead carp, silver carp, grass carp and blunt snout bream (De Silva et al., 1992); or a single specific species population (27%) e.g. rainbow trout (Trzebiatowski et al., 1981) or striped bass (Bosworth et al., 1998) (Figure A3.2.1). Only two articles described this service as being provided by a larger ecological unit such as the entire community or habitat, e.g. freshwater lakes (Fernando & Holčík, 1982).

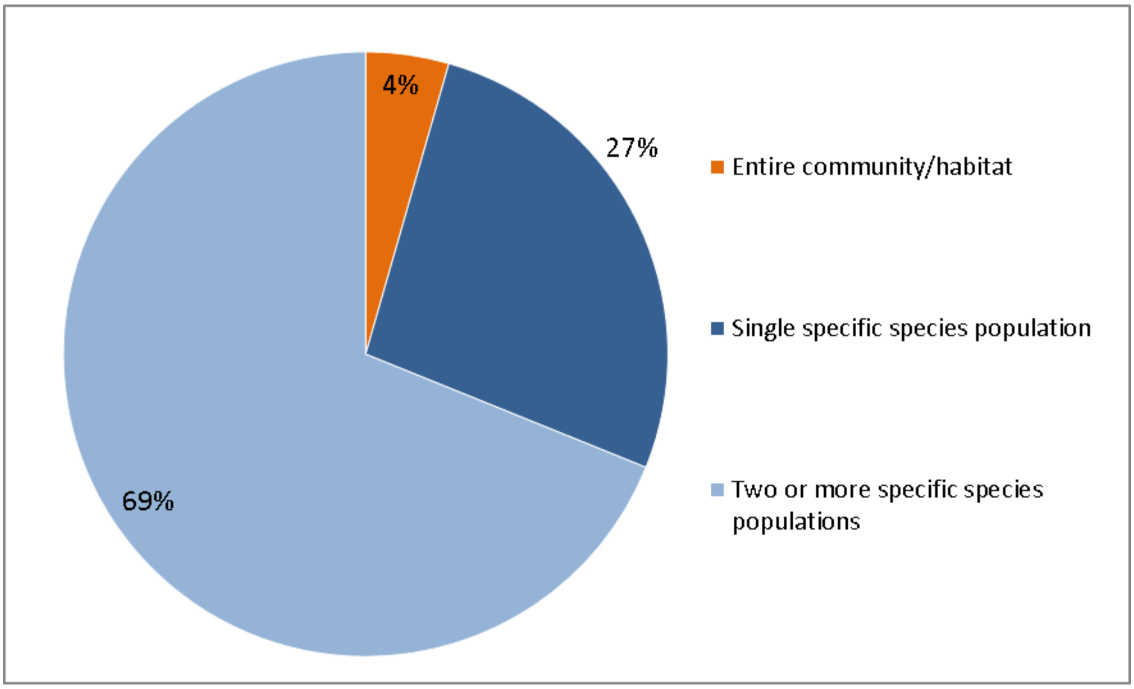

Figure A3.2.1: *Categorisation of the ecosystem service provider (ESP) for the service of freshwater fishing.*

Important attributes for the ESP

The majority of articles (64%) discussed species abundance as being an important attribute for the provision of freshwater fishing (Figure A3.2.2). Of these, only one article found high fish abundance to negatively impact on fishing (Milstein et al., 1988). In this case it was found that a high density (abundance) of silver carp in polyculture ponds negatively affected carp growth and yield. Of the remaining articles 38% provided evidence to suggest that higher abundance increased yield, with for example enhanced stocking being related to a significantly increased number of recreational catches of carp in the Czech Republic (Boukal et al., 2012), and a significant positive correlation identified between stocking density and fish yield in freshwater reservoirs in India (Sugunan & Katiha, 2004). However, the predominant direction of this relationship was unclear, as was found in 59% of studies which cited abundance. In general, studies pointed towards the existence of an ecological carrying capacity within freshwater systems, where reductions in yield took place when stocking density breached a certain threshold (e.g. De Silva, 2003; Hasan & Middendorp, 1998; Wohlfarth et al., 1985). A study by De Silva et al. (1992) found that in a number of Chinese reservoirs, once stocking density exceeded 6000-8000 fish ha^{-1}, yield decreased, whereas before this threshold yield was found to increase almost linearly with increasing stocking density. Similarly, in a study of Australian bass (Smith et al., 2012), yield increased to a point with increasing abundance of bass, however, once the carrying capacity was reached for the site (i.e. individuals were no longer able to meet their minimum energy requirements) mortality was found to significantly increase, and hence stocking programs usually aim to stock at the site's carrying capacity.

Species size/weight was the second most popular attribute to be discussed in the literature, mentioned in 60% of articles. This is essentially fish biomass. The majority of studies found that increased weight benefited the provision of freshwater fishing. The first reason for this is that for commercial fishing, fish need to be of marketable size and weight (e.g. Milstein et al., 1988; Rutten et al., 2005; Tucker et al., 1994; Wohlfarth et al., 1985). This is therefore a particularly important trait in terms of economics (Lorenzen, 2000). A number of studies also highlighted that larger fish are more likely to survive than smaller individuals, and hence can increase yield (Li, 1999; Lorenzen, 2000). One study also suggests that protecting larger individual species can improve the sustainability of species in fisheries (Birkeland & Dayton, 2005).

Species richness was found to have a predominantly positive relationship with fish production and yield, and on the whole polycultures were found to perform better than monocultures. The addition of tilapia, for example, to experimental ponds containing rohu and common carp resulted in additional production without affecting growth (e.g. Rahman et al., 2008). Similarly, significantly higher fish yields and growth of species such as *Cirrhinus mrigala* were reported for ponds containing common carp (Wahab et al., 1995). For prawns, it was found that growing species in monoculture at high stocking densities was not economical. In contrast farming prawns in a polyculture with fish led to an increase in mean species size and proportion of marketable animals (Wohlfarth et al., 1985). Polycultures were also associated with additional benefits; (i) improving environmental conditions, e.g. preventing large algal blooms from forming (Wohlfarth et al., 1985) or improving water quality (Dos Santos & Valenti, 2007), and (ii) by increasing available food resources. For example, Milstein et al. (1988) found that in polycultures common carp help to recirculate nutrients into the water column, by stirring up mud at the lakebed. This can aid phytoplankton development and, hence, provide more food for silver carp if also included in the polyculture (Milstein et al., 1988).

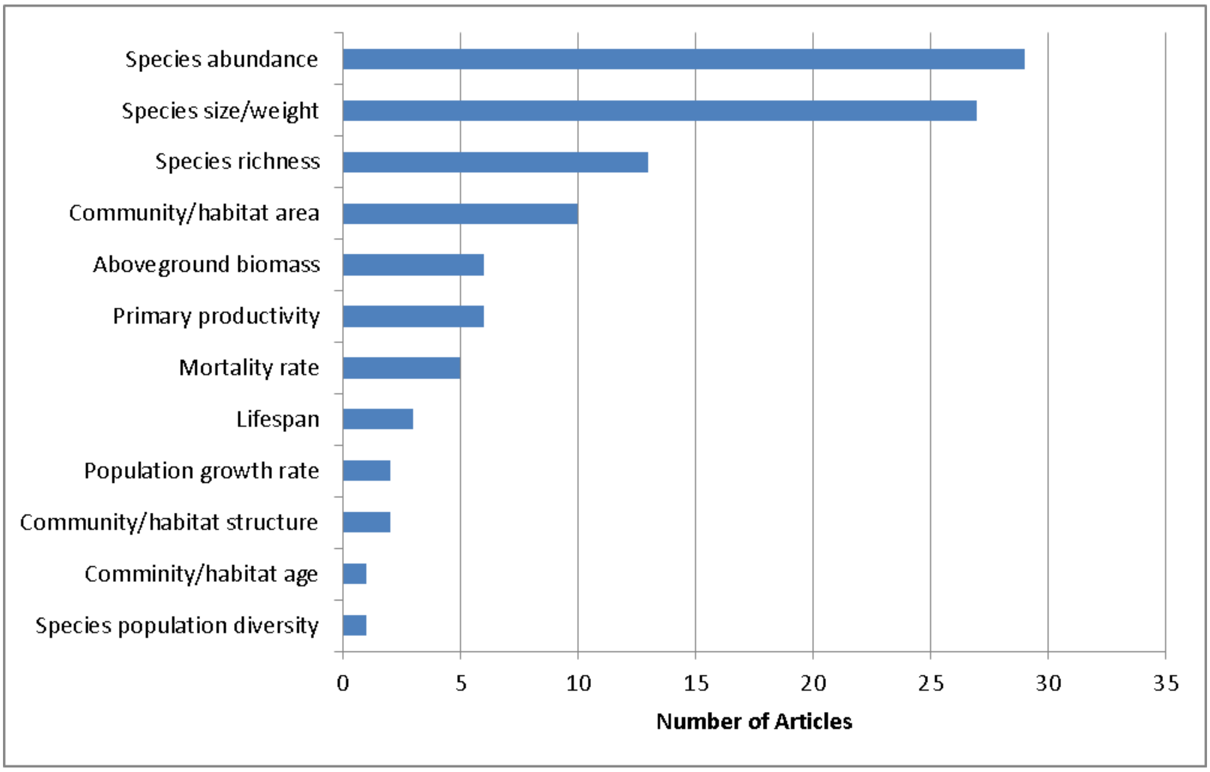

Figure A3.2.2: *Categorisation of the ESP attributes for the service of freshwater fishing. Note: some studies used multiple attributes.*

The effect of habitat area on freshwater fishing was mentioned in 22% of studies, however, this was found to be predominantly unclear in the literature. Mollot et al. (2003) found that flooding, by increasing the area of aquatic habitat available to fish in the Mekong basin made it possible to harvest larger quantities of fish. Other studies find watershed and lake area to have no significant correlation with fish production (e.g. Downing et al., 1990; Quiros, 1998), or that it significantly impacts some species and not others. For example, in a study of freshwater ponds in Bangladesh, Kadir et al. (2007) found that pond area explained 20-30% of variability in rohu performance, with a positive relationship found between the two, whereas area had no significant effect on catla and common carp.

Fish production and yield (cited in 13% of entries) also correlated positively with primary productivity in a number of studies (e.g. Downing et al. 1990; Lavrentyeva & Lavrentyev, 1996). In 11% of database entries, mortality was mentioned to negatively affect freshwater fishing by substantially reducing yield and recapture rates (e.g. Lorenzen, 2001; 2000). The same proportion of studies examined the effect of aboveground biomass, i.e. biomass contained within the freshwater lake or reservoir. In general, a higher food biomass (phytoplankton and zooplankton) helps to increase yield (Li, 1999). Research has found that a higher biomass of macrobenthos can explain 20% of the variation in fish biomass observed in North-temperate lakes (Hanson & Leggett, 1982). However, results for macrophyte cover were unclear, as Maceina & Reeves (1996) found that the amount of effort to catch large fish was inversely proportional to macrophyte abundance, although in general catch rates of largemouth bass were greater with a higher macrophyte cover.

As shown in Figure A3.2.2, further attributes were discussed in very few studies. These do, however, include longevity (discussed in three studies) for which it was found that extending the length of culture period in ponds was found to increase harvest yields and fish size (e.g. Garcia-Perez, 2000).

Discussion

The literature search

Many of the papers found in the search were not relevant for this review because they did not analyse directly the relationship between biodiversity and freshwater fishing, or tended to focus on the ecological impact of fishing. Hence, the process of snowballing and using very specific search terms were found to be particularly useful for finding more relevant papers.

Abiotic factors

42% of entries mentioned one of multiple abiotic factors which affect freshwater fishing. Of these, 31% discussed the impact of water quality on fish yield, with factors such as dissolved oxygen content and water clarity being important (e.g. Tucker et al., 1994). Similarly, nine entries mentioned the impact of nutrient availability. Kadir et al. (2007), for example, found a significantly negative relationship between secchi disk depth (an indicator of biological productivity) and yields of silver carp and mrigal. Wohlfarth et al. (1985) suggest that nutrient availability is especially important for common carp.

In terms of meteorological factors, three articles referred to the effects of temperature (e.g. Fernando & Holčík, 1982; Wahab et al., 1995) and water availability. A good example of how the latter impacts yield was provided by Boukal et al. (2012), who observed up to a ten-fold increase in fish catch after major flooding events in the Czech Republic as a result of carp being washed down from areas upstream. Similarly, two articles discussed precipitation, with Mollot et al. (2003) researching floodplain fishing in southern Lao which is facilitated by the monsoon.

Finally, one study discussed the impact of slope, as it was found that recreational fishermen preferred freshwater sites to be located in mountains and foothills rather than flatter land such as prairie (Adamowicz et al., 1994).

ESAs and a negative impact of biodiversity on freshwater fishing

Only two cases were found in this review where fish species antagonised the service of freshwater fishing. Both instances were found in polycultures: increased competition for food can reduce yields of certain species in Indian carp polycultures (Wahab et al., 2002), and similarly Wahab et al. (1995) found that in Bangladesh polycultures silver carp interacted negatively with catla and rohu for feed and reduced their growth. However, it is important to note that this adverse reaction was limited when common carp were present (Wahab et al., 1995).

Only two papers found in the review mentioned biodiversity having a negative impact on this ecosystem service. Both concerned the impact of birds predating on prawns, which reduced stocks and hence yield (Dos Santos & Valenti, 2007; Garcia-Perez, 2000).

Strength of evidence

No entries found in this review provided very strong evidence for the links between biodiversity and freshwater fishing. The majority of papers had weak (29%) or average (64%) evidence, based on empirical studies with few or no repeated experiments at a single site only.

A3.2.2 Results – Freshwater fishing and value linkages

The results from the review of valuation of freshwater fishing are summarised in Table A3.2.1

Table A3.2.1: *Results from the review of valuation of freshwater fishing.*

	Market prices	**Production function**	**Benefit transfer and reviews**	**Other**	**Total**
Ecosystem service provider - wetlands - other areas - species	3 1	1 1	1 2	1 1	6 4 1
Value - direct consumptive use - other	3 1	2	3	1 1	9 2
Spatial scale - local - regional - national - global - other	1 2 1	2	1 1 1	1 1	4 4 1 1 1
Beneficiaries - communities/society - group of stakeholders	4 1	2	3	2	11 1

Ecosystem service provider (ESP)

The ESP most represented in the studies reviewed for values were “wetlands” (in 6 studies out of 10). One study also included several ESPs including freshwater marshes, bay swamps, river/lake swamps, saltwater marsh, bays and estuaries, mixed shrub-shrub wetland, wetland coniferous forests, wetland forested mix, mixed wetland hardwoods, and wet prairies.

Valuation method

The most frequently used valuation method was based on market prices (used in 4 studies), followed by “benefit transfers” (3 studies) and “the production function approach” (2 studies). The emphasis is on the assessment of direct consumptive use values. Hence, only 2 studies also assess ‘other values’. Society is listed as the beneficiary in all studies; in addition to this 1 study also assesses the value accruing to a group of stakeholders. In 8 of the studies the values are either assessed on a local (4) or regional (4) scale; in the remaining 3 studies the values are assessed on a national (1), global (1) and ‘other’ scale.

Biodiversity linkage on which valuation is based

An explicit link to biodiversity was mentioned in only 1 of the 10 studies reviewed.

A3.3 Freshwater provision

A3.3.1 Results – Biodiversity and freshwater provision linkages

Spatial and temporal scale

Most of the studies in the review focused on the sub-national (44%) or local (40%) scale. Thus, the studies concerned either whole catchments (or several catchments) or individual streams and/or particular local plots, e.g. forest stands. Four papers investigated water supply at a national scale, two of which (Egoh et al., 2008; 2009) analysed the spatial overlap of biodiversity and water provision in important areas of a whole country (South Africa). For 8% of the studies the spatial scale was global.

Ecosystem service provider (ESP)

In most of the studies the ESP was the entire community or habitat (42%) or two or more communities or habitats (48%) (Figure A3.3.1). In most cases different land covers were compared with regard to water yield. For example, a review by Petheram et al. (2002) revealed that across a broad range of locations, water recharge was higher under shallow-rooted annual vegetation than deep-rooted vegetation. According to a review by Bruijnzeel (2004), eucalypts are "voracious consumers of water" when compared to the native vegetation of Southeast Asia. Similarly, Nosetto et al. (2005) showed that tree plantations evaporated on average 180% more water than grasslands. Ruprecht & Schofiel (1989) investigated the effects of clearing the native vegetation of southern Australia dominated by jarrah (*Eucalyptus marginata*) and establishment of agricultural plants, and concluded that such a change resulted in a large stream flow increase (ca. 30% rainfall/year), due to the decrease in both transpiration and interception loss.

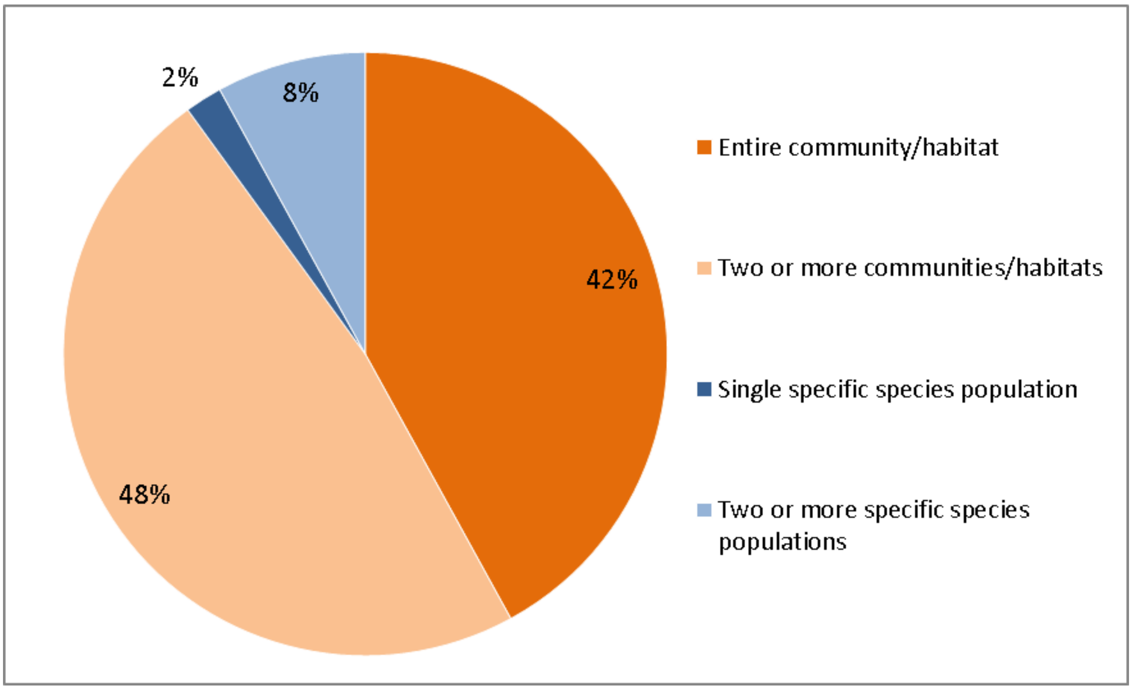

Figure A3.3.1: *Categorisation of the ecosystem service provider (ESP) for the service of freshwater provision.*

Usually the studies focused on forests, particularly on the question of forest harvesting and/or afforestation, including changes from native forests to plantations. For example, Buytaert et al. (2007) demonstrated that afforestation of natural grasslands with *Pinus Patula* decreased base flow and reduced the water yield by about 50% in a study site on the Andean highlands of Ecuador. Rowe & Pearce (1994) investigated changes to stream flow after harvesting evergreen mixed beech-podocarp-hardwood native forest and planting *Pinus radiata*. They concluded that directly after harvest, stream flow increased significantly, however with time the growth of pine led to rapid decreases in the flow that finally stabilised at a level of about 250mm/yr lower than pre-harvest levels. Komatsu et al. (2008) in their study in Japan on the effect of the transformation of native broadleaved forest to coniferous plantation concluded that water yield from a coniferous plantation that was 30 years old was lower than from broadleaved forest, even if initial clear-cutting increased the yield. Some studies also compared water yield from native forests versus plantations. For example, a study by Kagawa et al. (2009) found that native forests on Hawaii Island used much less water than alien tree plantations.

A few studies concerned atmospheric moisture interception from clouds and fog by forests that contributed to water input. A review by Hamilton et al. (1995) underlined that cloud forest enhances net precipitation by direct canopy interception of cloud water, which results in net additions to the water yield of the watershed. Also studies by Gomez-Peralta et al. (2008) and Brauman et al. (2010) confirmed that the atmospheric moisture from clouds and fog captured by forests contributes to the general water input. Studies have even found that cloud forests can help to generate a substantial amount of water supply in tropical areas (Reid, 2001).

One study focused on a single specific species population and three other studies on more than one species. A study by Nie et al. (2012) investigated the impact of encroachment by mesquite trees on water yield in a semi-arid area in Mexico. Le Maitre et al. (2002) showed the influence of invasive species on natural river flows, while Dierick & Hölscher (2009) demonstrated that species choice plays a role for water use rates. Cavaleri & Sack (2010) compared water use by native and invasive plants.

For 8% of the studies the ESP was assigned at a global scale by a number of review studies (e.g. Blumenfeld et al., 2010; Cavaleri & Sack, 2010). One examined the effect deforestation has on water yield (Sahin & Hall, 1996), whereas another was a global meta-analysis comparing water use in invasive and native species (Cavaleri & Sack, 2010).

Important attributes of the ESP

Most of the studies did not focus on some specific attribute of biodiversity, but rather compared areas with some particular land cover with areas of other cover, or investigated effects of changing the forest cover: hence 56% of articles referred in some way to habitat area (Figure A3.3.2). In general, harvesting of the forest led to a significant increase in water yield (e.g. Waterloo et al. 2007; Komatsu et al., 2008), but this effect disappeared fast as new vegetation was introduced, such as through afforestation (e.g. Hornbeck et al., 1993; Rowe & Pearce, 1994; Duncan, 1995; Yao et al., 2012), although this usually took a few years. For example, in a study by Fahey & Watson (1995), planting pine on tussock grassland sites led to a 20% decrease in water yield annually, after the initial 7-years period with no change.

There were also a number of studies that paid attention to other particular attributes (Figure A3.1.2). The age of trees/forest planted for example was cited in 34% of articles as influencing water yield. In general the higher the age of a forest stand, the larger its water use and thus the lower the water yield (Rowe & Pearce, 1994; Huang et al., 2003; Komatsu et al., 2008). For example, Muller (2009) demonstrated that water seepage to the groundwater level was 29%, 12% and 0% for pine, and 43%, 28% and 22% for beech stands at ages of 8, 14 and 28 years, respectively. This was also confirmed by a comprehensive global review by Farley et al. (2005) concluding that runoff losses increased significantly with plantation age for at least 20 years after planting.

In some of the studies, however, the relation with age was not clear as there were other factors that also mattered. For example, in a study by Jayasuriya et al. (1993) the effect of thinning of the mountain ash (*Eucalyptus regnans*) forest causing an increase in stream flow due to lower stem density was still visible 11 years after treatment in the uniformly thinned catchment, but this effect had completely decayed in the patch cut catchment.

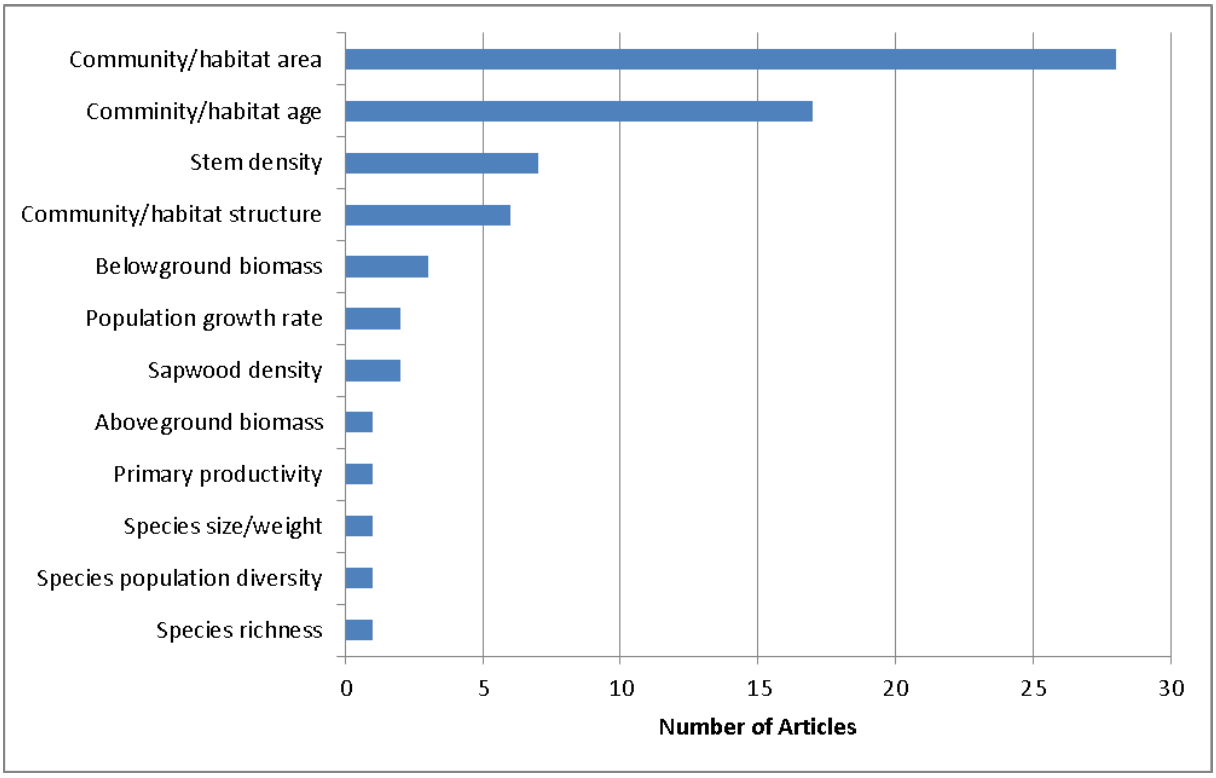

Figure A3.3.2: *Categorisation of the ESP attributes for the service of freshwater provision. Note: some studies cited multiple attributes.*

Moreover, some of the studies were too short-term to capture the long-term influence of age on water yield. Most of the studies encompassed a maximum of one or two decades. In some cases new stream flow equilibrium was reached (e.g. Ruprecht & Schofiel, 1989) or water yields returned to pre-disturbance levels (Ruprecht & Stoneman, 1996) several years after treatment/harvesting/planting, but in other cases we simply do not know enough about the long-term influence of age. However, a study by Muller (2009) suggests that from some older age this effect is not so large. This study showed that water seepage to the groundwater level was 43%, 28%, 22% and 21% in ages of 8, 14, 28 and 135 years respectively in beech stands, indicating the decreasing level of impact with age.

There was also one study (Singh & Mishra, 2012) that presented evidence of the positive impact of age on water yield, showing that old forests positively and highly significantly influenced runoff coefficient (i.e. a measure of water yield). However, in this study an old-growth forest defined as "primary forest, mature secondary forest and undisturbed mature plantations" was compared with mixed forests, i.e. "disturbed forests and to lesser extent naturally occurring open forests", thus probably other factors than age also played a role here.

14% of the studies addressed stem density as being an important forest attribute that negatively influenced the water yield. Zou et al. (2008) in their study comparing low and high-density stands of ponderosa pine concluded that there was a significantly greater soil water content in the low-density stands over a wide range of conditions. A study by Takahashi et al. (2011) looked into canopy water storage capacity and concluded that this capacity was more than twice as great at the native site compared to the site invaded by alien species due to both morphological characteristics and high stem density of the invasive trees. Kagawa et al. (2009) compared water use by native and alien species on Hawaii Island and also mentioned that lower stem densities result in lower water use by trees, which leads to more water in the catchment. Webb & Kathuria (2012) showed that catchment afforestation with *Pinus radiata* caused lower stream flow when compared to pasture catchment, but that the thinning (i.e. decreasing tree density) at age 14 had a significant positive effect on stream flow that persisted for at least six years. Also two other thinning experimental studies confirmed this stem density influence. Stoneman (1993) demonstrated that stream flow increased from 0.5% of rainfall before thinning to 7.6% of rainfall nine years after thinning of the jarrah (*Eucalyptus marginata*) forest in Western Australia. Jayasuriya et al. (1993) reported a stream flow increase of 25 to 30% after thinning in the mountain ash (*Eucalyptus regnans*) forest in South-east Australia.

Structural components of habitats were mentioned in 12% of studies as affecting the provision of freshwater. Interception by forest canopies was cited the most often (e.g. Brauman et al., 2010; Hamilton, 1995). One study also mentioned the influence of crown architecture on seepage and soil water availability (Muller, 2009). The effect of structure was predominantly unclear from the literature. Two studies suggested this negatively affected freshwater supply (Brauman et al., 2012; Muller, 2009), two suggested a positive relationship as a result of trees intercepting cloud water (Brauman et al., 2010; Hamilton, 1995), and two suggested a negative relationship due to increased water use (Blumenfeld et al., 2010; Putuhena & Cordery, 2000).

A few individual studies focused on morphological characteristics of some species. Two studies indicated that sapwood amount negatively influenced water yield, as higher sapwood area meant higher water use by trees, leading to lower water yield in the whole catchment (Kagawa et al. 2009). This is explained through the decline in sapwood area index (SAI) producing a decrease in stand transpiration, and thus an increase in water yield (Vertessy et al. 2001). A study by Muller (2009) also gives some indication that tree structure may influence water provision. In this study, water seepage to the groundwater level was much lower in pine stands than in beech stands and the probable explanation for this was the crown architecture of the trees, as pine has an open, awkward shaped crown that more easily intercepts water.

Leaf area index may also have an influence on water yield. In a study by Gomez-Peralta et al. (2008) on forest cloud and fog interception, the quantity of apparent fog interception was related to the canopy leaf area index, as plots with higher leaf surface area intercepted greater amounts of fog. Another study on cloud forests (Brauman et al. 2010) also indicated that a forest site with taller trees and denser mid-canopy has much higher throughfall than the other site.

One study, by Dierick & Hölscher (2009), also mentioned aboveground biomass indicating that maximum water use rates of different plants were significantly related to estimated biomass. A study by Cornish & Vertessy (2001) also positively relates water use to the growth rate of trees.

Discussion

The literature search

It was not easy to find relevant papers, as the searches returned very many hits. Many of them concerned the impact of water provision, groundwater level or water supply on biodiversity and not the other way around. Many also focused on the response of different species to water availability or changes in water regime. A lot of papers also considered only the influence of abiotic factors, such as precipitation or wind, on freshwater provision. Therefore, only 10 papers from the main search were found relevant for the aims of the study (and 4 papers from the pilot study), and the rest were obtained through snowballing and reverse snowballing. Moreover, even among the 14 chosen papers, some individual papers were not fully relevant. Some of them showed rather weak evidence for the relation between water supply and biodiversity (see section about strength of evidence).

Abiotic factors

In several cases abiotic factors were mentioned as being important for potable water provision. Particularly they related to actual water availability in the place of the study, which is also related to precipitation, snow and temperature (and connected to that the intensity of evaporation). For example, Rey Benayas et al. (2007) underlined that a decrease in water yield due to reforestation is most accentuated in dry regions with low water availability and precipitation. Huang et al. (2003) in their study of the effect of tree planting on grassland concluded that a reduction in monthly runoff occurred mainly in months with greater rainfall. Holdsworth & Mark (1990) who compared water yield from three different kinds of land cover observed that the highest water yield in snow tussock grassland was especially visible in the "snow-free" six months. Cavaleri & Sack (2010) showed that the differences in water use of native and invasive species depended strongly on climate, with the greater water use of invasives enhanced in hotter, wetter climates at the coarser scales. Also some studies underlined that the annual water yield is a function of annual rainfall, as low annual rainfall generated low run-off or water yield (Bren & Hoppmans, 2007; Webb 2009).

Among other abiotic factors, wind, slope and elevation also played some role. For example, Bruijnzeel (2004) in a review comparing water yield from different land covers reported that the differences in initial response of water yield to forest clearing can be explained by not only precipitation levels but also by differences in elevation and distance to the coast (affecting evaporation), catchment steepness and soil depth (governing the residence time of the water and the speed of baseflow recession), the degree of disturbance of undergrowth and soil by machinery or fire (determining both the water absorption capacity of the soil and the rate of regrowth) and the fertility of the soil (influencing post-clearing plant productivity and water uptake). Also a study by Gomez-Peralta et al. (2008) underlined that both total annual rainfall and net precipitation increased with elevation, which indirectly influenced water yield. In their study also, only the higher elevation forest site received significant water inputs from fog interception.

Strength of evidence

Most of the reviewed papers (as much as 60%) presented strong evidence for the links between freshwater provision and some aspect of biodiversity. Thirteen papers had average, but still relatively good strength of evidence. Most of the remaining five papers with weak strength of evidence investigated spatial overlap or congruence between water supply and some aspects of biodiversity, e.g. species richness and vegetation diversity in Egoh et al. (2009) or primary productivity in Egoh et al. (2008). In a similar manner Kai et al. (2006) used a rarity-weighted richness index and correlated it with water provision. These studies focus rather on correlation or spatial overlap than on the actual link between biodiversity and water provision.

Scale

The question of scale may also be important in some cases. An interesting study by Cavaleri & Sack (2010) focused on the question of scale in relation to native and invasive species. They compared water use by native and invasive plants and concluded that at the leaf scale, invasive species had greater stomatal conductance than native species, but at the plant scale natives and invasives were equally likely to have the higher sap flow rates. At the ecosystem scale, invasive-dominated ecosystems were more likely to have higher sap flow rates per unit ground area; while on the other hand, ecosystem-scale evapotranspiration was equally likely to be greater for systems dominated by invasive and native species of the same growth form.

A3.3.2 Results – Freshwater provision and value linkages

The results from the review of valuation of freshwater provision are summarised in Table A3.3.1.

Table A3.3.1: Results from the review of valuation of freshwater provision.

	Production function	Revealed preference	Stated preference	Biophysical ranking	Other	Total
Ecosystem service provider						
- wetlands	1					1
- forests	1	1	1	1	2	6
Value						
- direct consumptive use			1			4
- indirect use	1		1	1	1	7
- non-use	2	1	1	1	2	1
- option value			1			1
Spatial scale						
- local	2	1	1	1	2	7
Beneficiaries						
- households/individuals	2	1				5
- firms		1	1	1		1
- communities/society					2	2

It can be seen from Table A3.3.1 that the main ecosystem service provider is forests (6 studies) while one study has wetlands as the service provider. In terms of the valuation method, different approaches are used. Two studies use the production function approach, one revealed preferences, one stated preferences, one biophysical ranking and indirect use value and two use other methods. The emphasis is on assessment of direct consumptive use value and indirect use value. Households/individuals are listed as the beneficiaries in most studies, but two studies value benefits to communities/society. In all studies the values are assessed on a local scale.

Discussion

Many independent studies exist on valuing changes in water/biodiversity quality from a human welfare point of view. However, few studies attempt to value benefits of water quality/quantity in relation to changes in biodiversity/ecological functions.

- Many studies applied stated preference techniques based on hypothetical scenarios compared to revealed preference methods.
- In many economic valuation studies, socio-economic, demographic and environmental characteristics/ecological functions are treated as external drivers and were not explicitly taken into account. Hence, a more multidisciplinary approach is needed in order to take this into account.
- More attention on socio-hydrological linkages is needed, to generate policy relevant insights.
- It is questionable whether stated preferences methods can really measure the actual social benefits merely in terms of willingness to pay and willingness to avoid. Therefore, it is necessary to apply other techniques to enhance the validity of such studies.
- There are fewer studies which apply other qualitative approaches such as multi-criteria decision analysis (MCDA) and agent-based models (ABM).
- There are many payment for ecosystem services studies, but fewer that discuss how to implement such schemes in a policy context, with institutional and regulatory aspects.

- There are many studies on valuing the benefits of water/biodiversity under hypothetical scenarios, but only a few compare those estimates with the cost of implementing such alternatives via cost-benefit or cost-effectiveness analysis.

A3.4 Water purification

A3.4.1 Results - biodiversity and water purification linkages

Spatial and temporal scale

Most of the studies included in the review concerned the sub-national spatial scale (54% of articles). These usually dealt with the whole catchment or several catchments. As many as 40% of articles focused on the local scale; a few of these cases were local experiments, whilst others simply dealt with a particular lake or river fragment.

Most commonly, the studies encompassed several months or a period of a few years (40% of articles), however, in most of the cases (58% articles) the temporal scale was not explicitly stated. Only a study by Greiner & Hershner (1998) considered a longer period. In this study the authors analysed long term phosphorus retention in wetlands over a 30 year period.

Ecosystem service provider (ESP)

In over half of the studies (52%) included in the review, the ESP was the entire community or habitat, while 28% of studies focused on two or more communities or habitats (Figure A3.4.1). In most cases the habitats in question were either wetlands or some kind of vegetated/forested buffers (including experimental and constructed wetlands and buffers) along rivers or streams, whose role was to remove nutrients from the water. For example, in a literature review regarding vegetated buffers, Vought et al. (1995) concluded that a buffer strip 10 m wide can reduce the phosphorus (P) load, typically bound to sediment, by as much as 95%. A study in Illinois, USA, compared the potential of forest and grass vegetated buffers to reduce nutrient inputs from agriculture to streams and concluded that both forested and grass vegetated buffer strips significantly reduced local nitrate loadings; forested buffers being more effective than grass buffers at reducing the concentration of nitrate-N, but less efficient at retaining total and dissolved P (Osborne & Kovacic, 1993). On the contrary, the study by Jia et al. (2006) of agricultural land showed that vegetative buffers reduced the average concentration of P by about 30%, but the average concentration of N was not affected. Moreover, in their study irrigation scheduling and proper management were more important to water quality than vegetative buffers.

Several studies illustrated the role of wetlands, both natural and constructed, in water purification. For example, in their review of the global literature, Verhoeven et al. (2006) showed that many studies at the site scale have demonstrated that wetlands have a high and long-term capacity to improve water quality, mainly by removing nitrate and phosphorous. The main mechanisms for water quality improvements are denitrification by bacteria and nutrient uptake by vegetation. Most of the reviewed studies on wetlands simply provide rates of accumulation or storage of different nutrients by wetlands, but do not compare wetlands with other land cover types. For example, in a study by Hernandez & Mitsch (2006), constructed wetlands over a 10-year period stored an average of 43–47 Mg sediment/ha/yr, 162–166 kg N/ha/yr and 33–35 kg P/ha/yr in soil, and removed an average of 410–470 kg N/ha/yr of nitrate-N via denitrification. Greiner & Hershner (1998) estimated sediment accumulation rates to be from 3.0 to 5.6 mm/yr and calculated total P burial rates to range from 0.05 to 1.30 g TP/m2/yr. In a study from Taiwan more than 96% of total coliforms, 48% of biochemical oxygen demand, and 40% of nutrients (total N, total P) were removed via the constructed wetland system between 2007 and 2009 (Wu et al., 2010).

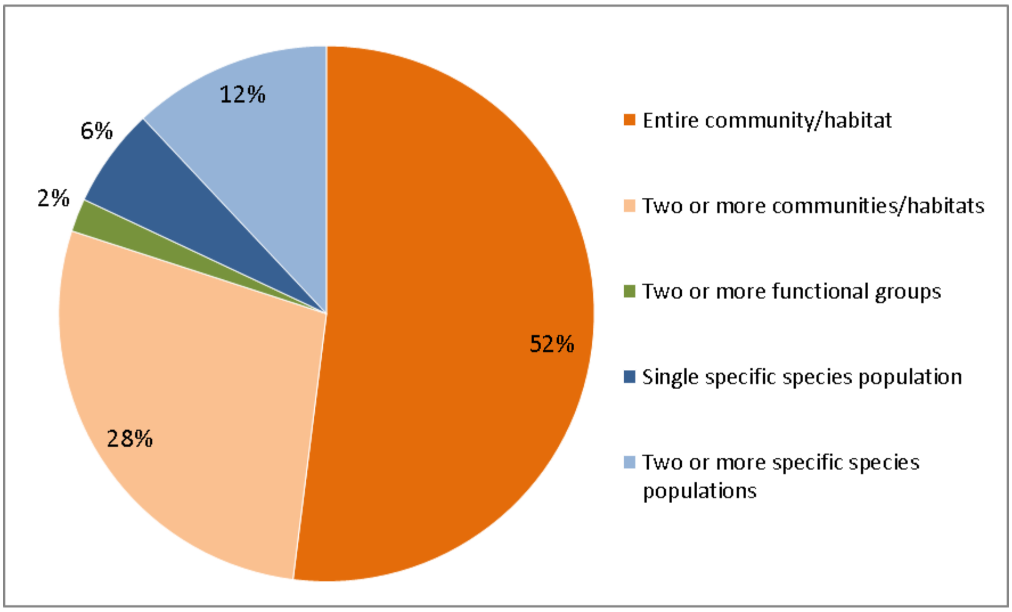

Figure A3.4.1: *Categorisation of the ecosystem service provider (ESP) for the service of water purification.*

Only 4% of the reviewed studies compared wetlands vegetated with different species or vegetation forms. Fisher et al. (2009) showed that beds planted with iris (*Iris pseudacorus* and *Typha latifolia*) and mixed vegetation removed the greatest proportion of ammonium, and increased nitrate concentrations in the water to a lesser extent than the *Phragmites* bed. Weisner & Thiere (2010) demonstrated that N retention was consistently higher in emergent vegetation wetlands than in wetlands dominated by a mixture of submerged vegetation and filamentous green algae or by only filamentous green algae.

Most of the studies where the ESP was a community or habitat did not describe biodiversity in any quantitative way (through number of species, structural diversity, etc.); neither did they explicitly provide any attributes of biodiversity that may be important for water quality. Instead, they tended to focus on land cover and whether a particular cover existed or not, in some cases comparing different covers. The land cover/land use studies were usually conducted at the scale of whole catchments. For example, Swaine et al. (2006) showed that catchments with higher forest cover yielded more oligotrophic and less turbid water in a study from Ghana; whilst Miller et al. (2011) showed that forest land cover had a significant impact on base flow water quality, reducing NH_4–N and NO_3–N in the base flow in a US study. Further, according to a study of pasture colonisation by trees by Carroll and Tucker (2000) areas with low surface cover had large sediment loss from the soil and spoil material (>70t/ha), but this declined (<0.5t/ha) after pasture colonization by trees.

For the remaining studies, the ESP was a single specific species population (6% of articles), two or more species populations (6% of articles) or two or more functional groups (2% of articles). These studies usually compared the efficiency of different species in water purification. For example, Miyazaki et al. (1999) in their experiment of a floating culture system showed that *Cyperus alternifolius* had higher rates of N absorption from water, and thus higher water purification efficiency than *Oryza sativa*. In a study by Lee & McNaughton (2004) comparing *Nuphar variegatum* (Water lily) and *Zizania palustris* (wild rice) beds, variation occurred in vegetated areas and was species dependent.

Finally, in a study by Fisher et al. (2009), the ESP was defined as “two or more communities or habitats”, as it dealt with vegetation beds of different species communities. However, species richness was also shown to be an important attribute in this study (see “Attributes” section below).

Important attributes of the ESP

Most of the studies in the review where the ESP was one or more communities or habitats did not present any particular attributes of biodiversity that would impact water quality regulation, but focused on simply cover / no cover differences; hence, habitat area was mentioned in 62% of entries as affecting this service (See Figure A3.4.2). A few studies focused on the effect of the area of a particular community/habitat on water purification possibilities. For example, a study by Sweeney et al. (2004) demonstrated that increased channel width in forested reaches due to lower encroachment of herbaceous plants (and, hence, more streambed area per unit length of stream) prevents nonpoint source pollutants from entering small streams and enhances the in-stream processing of both nonpoint and point source pollutants. Further, a review by DeSteven & Lowrance (2011) showed that relatively narrow stream buffers (8 to 20 m) can provide measurable water quality benefits.

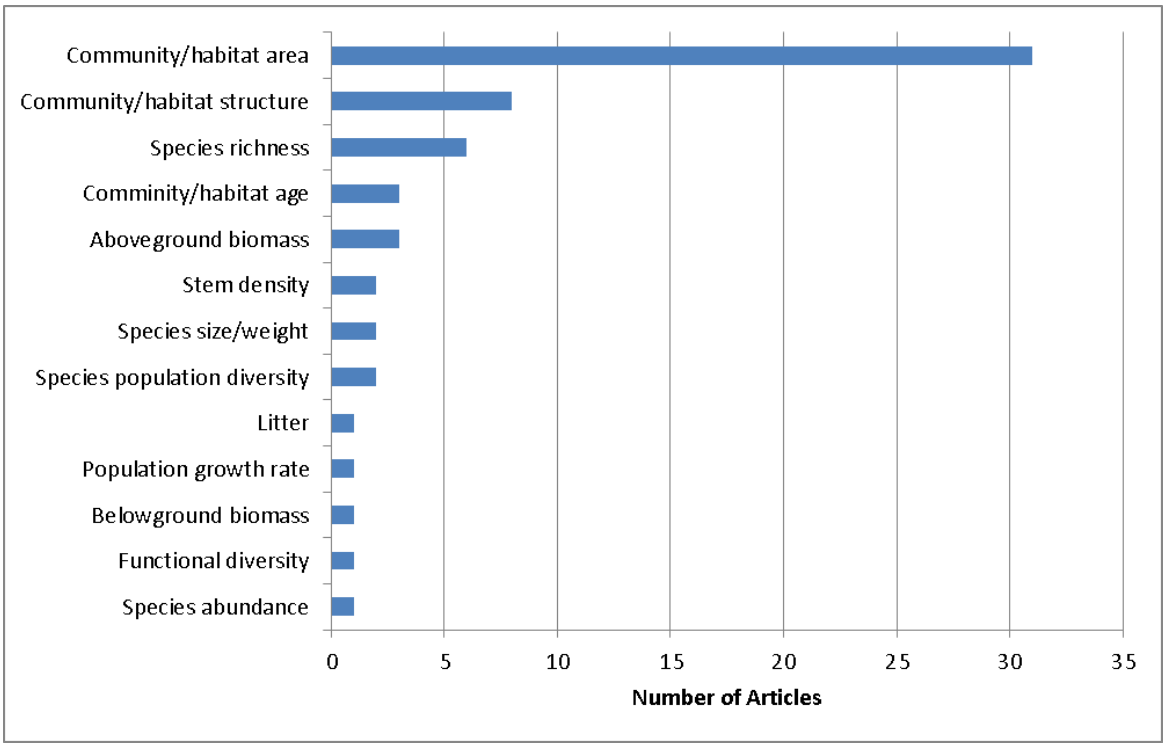

Figure A3.4.2: *Categorisation of the ESP attributes for the service of water purification. Note: some studies used multiple attributes.*

Some studies suggested that wetland area is a major factor controlling hydrologic assimilative capacity (Helfield & Diamond 1997). For example, in a study by Moreno-Mateos et al. (2008), wetlands ratio (% of wetlands) was negatively correlated with nutrient concentration (and thus positively correlated with water quality). Verhoeven et al. (2006) showed that to efficiently remove N, the area of wetlands should be between 2-7 % of the total catchment area. Kovacic et al. (2006) calculated that a wetland area of 450 ha (i.e. 5 % of the watershed) would be required in the Lake Bloomington watershed in the US to reduce N loading by 46%. Helfield & Diamond (1997) estimated that about 110 ha of wetland area would be required to reduce P levels to meet regulatory objectives in the lower Don River. Mitsch et al. (2001) estimated that creation and restoration of 2.1–5.3 million ha of wetlands in the Mississippi River Basin (0.7–1.8% of the basin) could reduce nitrate loadings to the Gulf of Mexico by 300–800 000 metric tons/yr, or by 18–50%, based on annual loads of 1.5–1.6 million Mt/yr.

Several other attributes were also cited in the literature, for example, a few highlighted the importance of the community, habitat or landscape structure for water purification. Martinez et al.

(2009) showed that larger structural diversity of the tropical montane cloud forests leads to higher water quality when compared to less diverse coffee plantations and grasslands. According to Martin et al. (1999), there is a positive relationship between natural denitrification activity and microbial community structure measures, such as richness, evenness and diversity. Moreno-Mateos et al. (2008) investigated the structure of the landscape and concluded that large patch size (i.e. a more homogenous landscape) is related to higher concentration of nutrients and higher salinity in waters (i.e. lower water quality). Additionally, patch density also has a positive effect on N concentration (i.e. lower water quality). A study by Liu et al. (2012) also mentioned landscape structure as being probably important for water quality; however, their analysis does not provide evidence for any clear general relationship between the complexity of the landscape and water quality regulation.

A study by Adhikari et al. (2011) on different cattail and bulrush species found a positive relationship between both above-ground and below-ground biomass of plants and water quality. The above-ground plant parts were more efficient at taking up total N in all investigated wetlands when compared to below-ground parts, while the below-ground plant parts were more efficient for total P uptake.

Those studies which considered species as the ESP usually focused on the efficiency of particular species on water purification. In most cases these effects were explained by some particular attribute of the species. Species richness was one such factor, mentioned in 12% of studies. For example, in the study by Fisher et al. (2009) the largest reduction in ammonia concentrations in the Iris vegetated bed may be attributable to the more floristically diverse condition of this bed when compared to other beds. This may, according to the authors, be associated with a greater diversity of plant-associated microbes related to increased N functioning, a greater vegetation biomass, or more efficiently filled rhizosphere and above-sediment spaces that increase the surface area of microbial biofilms, plant uptake and sedimentation. Cardinale et al. (2011) in their review of the international literature on the role of producer diversity in ecosystems concluded that producer species richness increases the efficiency by which plants and algae assimilate inorganic resources and convert these into standing biomass. Thus, the concentrations of nutrients and litter in soil or water both decrease with increasing richness. However, these concentrations are still higher than what is achieved by the most efficient monoculture, thus the relation between species richness and water quality is not so straightforward and depends on the species in question. Another study by Cardinale (2011) demonstrated that species richness of algae may have a positive influence on water quality. According to his analysis, N uptake rates increase linearly with species richness and are driven by niche differences among species.

Species abundance may also have a positive influence on water quality, as demonstrated in an experiment with submerged macrophytes by Nakamura et al. (2008). They showed that water quality in a harvested pond with higher abundance of macrophytes (percent water volume infested with macrophytes = 10 %) was better, i.e. it had a lower nutrient level, than in a harvested pond with lower abundance (PVI = 3%).

Another attribute that may be related to water quality is size or weight of species, but evidence was only found in two studies in the review. The study by Miyazaki et al. (1999) comparing N absorption by *Oryza sativa* and *Cyperus alternifolius* grown in the floating culture system demonstrated that water purification efficiency of *C. alternifolius* was further improved with fertilizer application as it increased both the root weight and N absorption rate. On the other hand, *O. Sativa* had a negative effect on water purification under the fertilized condition, thus the relationship is not very clear. Another study by Moore (2004) definitely showed that species size matters in the case of the effect of seagrass beds on water quality.

Only in one study, by Oelmann et al. (2011), was the ESP related to functional groups. In a grassland experiment it was demonstrated that mixtures containing legumes had significantly higher above-ground N storage than other grassland species mixtures. On average of all sampling campaigns legume-containing mixtures stored 237% more above-ground N than mixtures without legumes.

Discussion

The literature search

The search of the literature was not an easy task, as the search terms seemed to be too general to pin-point papers relevant for the aims of the study, and thus the number of hits obtained was very large. Because of that, refining terms were necessary. Still, even using refined terms, the number of hits was very large. Many studies returned by the search concerned the issue of how water quality influences some aspect of biodiversity or investigated water quality together with biodiversity, but without indicating any relationship between the two. Thus, finding 50 relevant papers was difficult and time-consuming. Snowballing proved to be a good technique to find the papers that did not come up during the search.

What was surprising is that in some articles dealing with vegetated buffers or wetlands, no details were given on these habitats/communities (e.g. species composition). For example, it could be stated that there were 13 native species, but no names of species were given.

Abiotic factors

The influence of abiotic factors on water quality regulation was not the main point of this review. However, in several cases there was an obvious influence of some abiotic factors on water quality. Particularly, temperature, evaporation and water availability were important factors. For example, in a study by Oelmann et al. (2011) showing that legumes had better N storage capacity than other plants, they claimed that rainfall, soil temperature and moisture influenced productivity and thus, above-ground N storage. In a study by Li et al. (2008) total dissolved solids in water showed lower values in the rainy season, which was related to their dilution by precipitation. Detenbeck et al. (2004) demonstrate that increased precipitation may heighten effects of deforestation because it increases transport capacity of particulates and raises the water table and frequency of overland flow, and thus reduces interaction of groundwater with mineral soils as compared with surface organic layers.

There were also some seasonal differences in water quality due to temperature, for example, ammonium reduction in vegetation beds was greater in the spring and summer, mainly due to microbial activity, which controls the denitrification processes and is larger during warmer temperatures (Fisher et al. 2009).

In individual cases soil was also an important factor. For example, Sliva & Williams (2001) in their study of water quality in forested versus urban areas concluded that NH_4 in water was positively correlated with silt-clay in soils. In a study by Greiner & Hershner (1998), clay content was also significantly correlated with total P in waters. Landscape configuration, e.g. slope also sometimes has an impact. For example, in a study by Reed & Carpenter (2002) P-yield increased with increasing average slope in the watershed.

Negative impact of biodiversity on water quality

Some individual studies showed that the ESP may also have a negative impact on the ecosystem service of water purification. In a study by Osborne & Kovacic (1993) during the dormant season both grass and forested buffers released dissolved and total P to the groundwater, even if they acted as a sink for much of the year. Fisher et al. (2009) indicated that wetland vegetation beds not only reduce P loading to waters but are also net exporters (particularly in the autumn) of orthophosphorus which is biologically available and therefore may lead to eutrophication. Also Helfield & Diamond (1997) underlined that the nutrient uptake process in wetlands does not provide permanent removal, but only represents temporary storage, as nutrients may be released at senescence. Therefore they suggested periodic harvesting of aquatic vegetation to remove the nutrients from the whole system.

A study by Weisner & Thiere (2010) demonstrated that in some cases lower diversity may be favourable for water quality. In their investigation of experimental wetlands, wetlands dominated by tall emergent vegetation were more efficient in removing N, but also had lower diversity than wetlands dominated by filamentous algae or submerged macrophytes. In such a case management that promotes diversity of species by suppressing dominant emergent plant species would impair the water quality regulating function of the ecosystem.

A study by McDowell (2008) concerning a particular species, deer, and conducted in experimental conditions on a deer farm demonstrated that a deer can be an ecosystem service antagoniser, as wallowing by deer and deer's direct excretal input and runoff caused higher concentration of nutrients. In the experiment mean concentrations of NO_3-N, NH_4+, N, SS and TP improved after fencing-off from deer and riparian planting, with loads of these constituents decreasing by 78 - 98 %.

Strength of evidence

For most of the studies in the review, the evidence provided was strong (48% of articles) or average (46%). Only two studies seemed to be not satisfactory in providing evidence for the link between biodiversity and water quality. In the study by Swaine et al. (2006) the predictive power of the relationship between different vegetation groups and water quality was weak. In the study by Mitsch et al. (2001) correlations of changes between buffer and control reaches for water quality showed some significant relationships, but the result was based on one site that had the greatest change in physical variables between the buffer and control reaches and also the largest and widest buffer zone.

A3.4.2 Results - water purification and value linkages

The results from the review of valuation of water purification are summarised in Table A3.4.1. It can be seen from Table A3.4.1 that 21 of the reviewed studies value direct and indirect use value while 11 studies value both use and non-use value. The ecosystem service providers are coastal waters (10), rivers (6), coral reefs (2), wetlands (2) and watersheds (3). These ecosystem service providers are beneficial to society because they provide water of good quality for drinking, industrial use or recreational purposes.

The spatial scale of the studies is mainly local (14), but four studies are regional and national. The beneficiaries of water purification are either specific households/individuals or communities/society depending on the valuation method used. If stated preference methods are used the estimated willingness to pay is mainly related to the households and individuals interviewed even if these are regarded as representative for the society in general. The other valuation methods estimate the value of the purification service for a community or for society in general.

The benefits of water purification include both use and non-use values and this is reflected in the choice of valuation method. The stated preference methods are the only economic approaches by which non-use values can be estimated. Therefore, the ten studies that value the total use and non-use value of water purification use this method. Of course, benefit transfer can also be used to transfer non-use values. The non-economic approaches which are used in two studies can also value use as well as non-use value. If valuation of water purification is limited to use value its value can be estimated by other economic valuation methods. Two studies use the avoided cost approach and three studies the revealed preference approach.

Table A3.4.1: *Results from the review of valuation of water purification.*

	Avoided costs	Revealed preference	Stated preference	Benefit transfer and review	Other	Total
Ecosystem service provider						
- coastal water, fjord	1	1	7			10
- river			4	1	1	6
- coral reef		1	1		1	2
- wetland			1	1		2
- watershed	1	1	1			3
Value						
- direct consumptive use		2	14	2	2	20
- non-consumptive use					1	1
- indirect use	1	2	14	2	2	21
- non-use value			10	1		11
Spatial scale						
- local			12			14
- regional		2	2	1	1	4
- national	1		1	1	1	4
Beneficiaries						
- households/individuals	1	1	12			14
- community/society	1	2	2	2	2	10

A3.5 Water flow regulation (flood protection)

A3.5.1 Results – Biodiversity and flood protection linkages

Spatial and temporal scale

The scale of the ecosystem service provider (ESP) was variable; however the majority of studies discussed flood protection at a local (40%) or sub-national (34%) scale. Very few articles discussed flood protection at the national (6%) and continental scales (6%), whereas 14% of studies discussed this on a global scale. Bullock & Acreman (2003), for example, reviewed the role of wetlands in the hydrological cycle. Other global scale studies tended to focus on the effect of afforestation and deforestation on hydrological flow curves using results from multiple locations (e.g. Bradshaw et al., 2007; Lane et al., 2003).

Flood protection was discussed in almost all studies (96%) at an annual-seasonal scale. Examples include the effect of forest harvesting on seasonal and annual peak flows (Iroumé et al., 2005; Lin & Wei, 2008); and the change in annual peak discharge associated with wetland restoration scenarios in Canada (Yang et al., 2010). A study by Farley et al. (2005) was the only one in this review to consider flood storage on a decadal timescale, examining the time period over which run-off losses increase with plantation age. This was found to increase for at least 20 years after planting for Eucalypt forests, with afforested grasslands achieving a 50% reduction in run-off a decade after planting. In contrast to this long timescale, a study by Schmittner & Giresse, (1996) used a daily timescale, examining the environmental controls on flash flood flows. Examples of parameters used include the hourly surface water discharge, and the timelag induced by vegetation on the scale of minutes.

Ecosystem Service Provider (ESP)

Figure A3.5.1 shows the distribution of ESP classes found in the review. The main ESP discussed in the literature is the entire community or habitat (50%); with studies examining the influence of habitat on hydrological parameters such as run-off, interception, lag-time and flood recurrence intervals. One example of this ESP in the literature is given by Qi et al. (2007), who find the function of forest communities to be greatest in reducing the effects of peak rainfall for short rainfall events. Research showed that both mixed conifer-broadleaf and mixed broad-leaved forests reduced peak flow and surface run-off, although the mixed conifer-broadleaf forest communities were associated with the largest reductions (Qi et al., 2007).

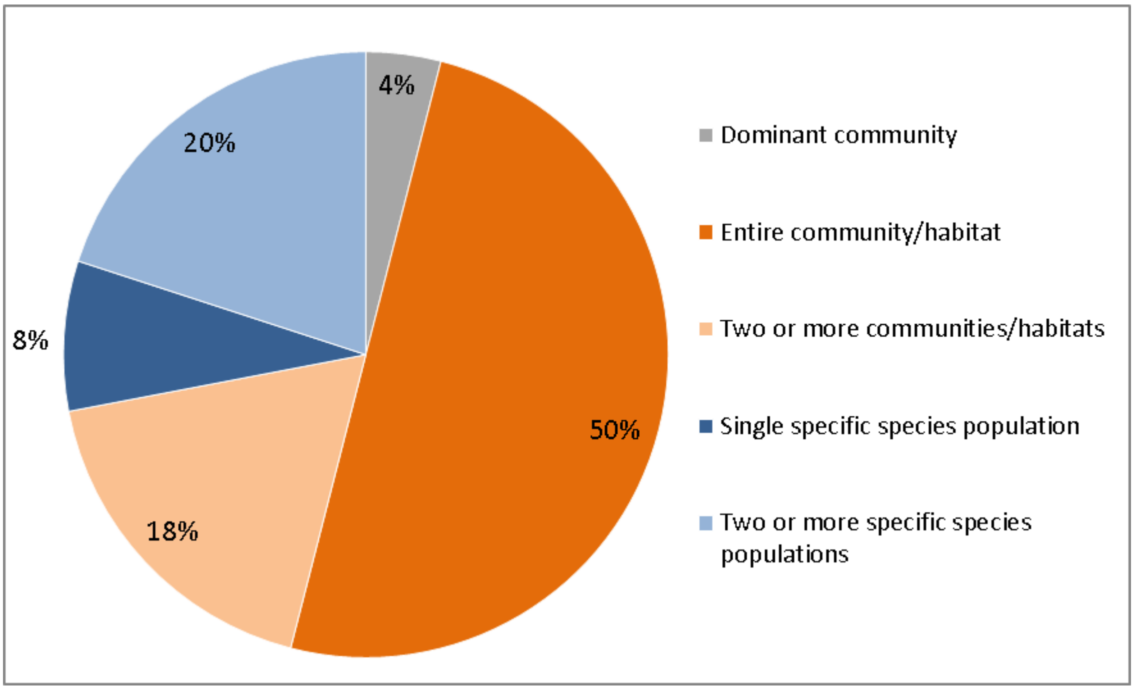

Figure A3.5.1: *Categorisation of the ecosystem service provider (ESP) for the service of flood protection.*

The second most common unit providing this ecosystem service was two or more specific species populations (20%). Such studies examined themes such as the hydrological effects of converting eucalyptus forest to *Pinus radiata* (Puhuhena & Cordery, 2000); and a study by Bren & Hopmans (2007) compares mature eucalypt and immature radiata pine plantations in Australia, finding the pine plantations to be associated with greater run-off.

The category "two or more communities or habitats" was the third most popular ESP (18%). Robinson et al. (2003) provide a good example here, studying the effects of various forest types in Europe, covering Atlantic northwest European coniferous forest, continental central European mixed hardwood forest and Mediterranean south European open forests to see how these influenced peak and base flows.

A few studies (8%) discussed single specific species populations as providing flood protection, with some examples being studying various forest plantations, e.g. pine (Fahey & Jackson, 1997), and the effect of Tamarisk, an invasive woody shrub, on flooding (Zavaletta, 2000). Very few studies (4%) examined a dominant community as the ESP, one of those being Lin & Wei (2008) who examined the effects of harvesting on the dominant sub-boreal spruce community in a Willow watershed in British Columbia.

Important attributes of the ESP

The ability of wetlands to regulate water flow is highlighted in a number of studies (e.g. Acreman et al., 2007; Ming et al., 2007; Posthumus et al., 2010), with a global review study concluding that most floodplain wetlands reduce or delay flooding (Bullock & Acreman, 2003). The presence of wetland vegetation delays surface run-off into waterways and the storage provided by these areas can reduce the magnitude of downstream flooding (Acreman et al., 2011). In contrast to this flood mitigation role, wetlands located at the headwaters of river systems can actually increase flood peaks and generate flood flows as these are often saturated, quickly transporting rainfall to waterways (Bullock & Acreman, 2003). Furthermore, although the presence of vegetation has been found to delay surface run-off, one study finds that for managed woodland plantations in northwest Europe and Eucalyptus in southern Europe with poorly drained soils, forestry is likely to have a limited role in managing regional and large-scale flood risk or drought flows (Robinson et al., 2003).

The majority of studies compared areas of different land-use and cover, discussed the effects of changing forest cover, or multiple attributes of biodiversity. The most common of these examined in the literature include community or stand age (26%), structure (28%), habitat area (78%) and species size (10%), as shown in Figure A3.5.2. Studies examine how these attributes affect hydrological parameters, such as run-off and infiltration rates, and hence how they impact on flooding.

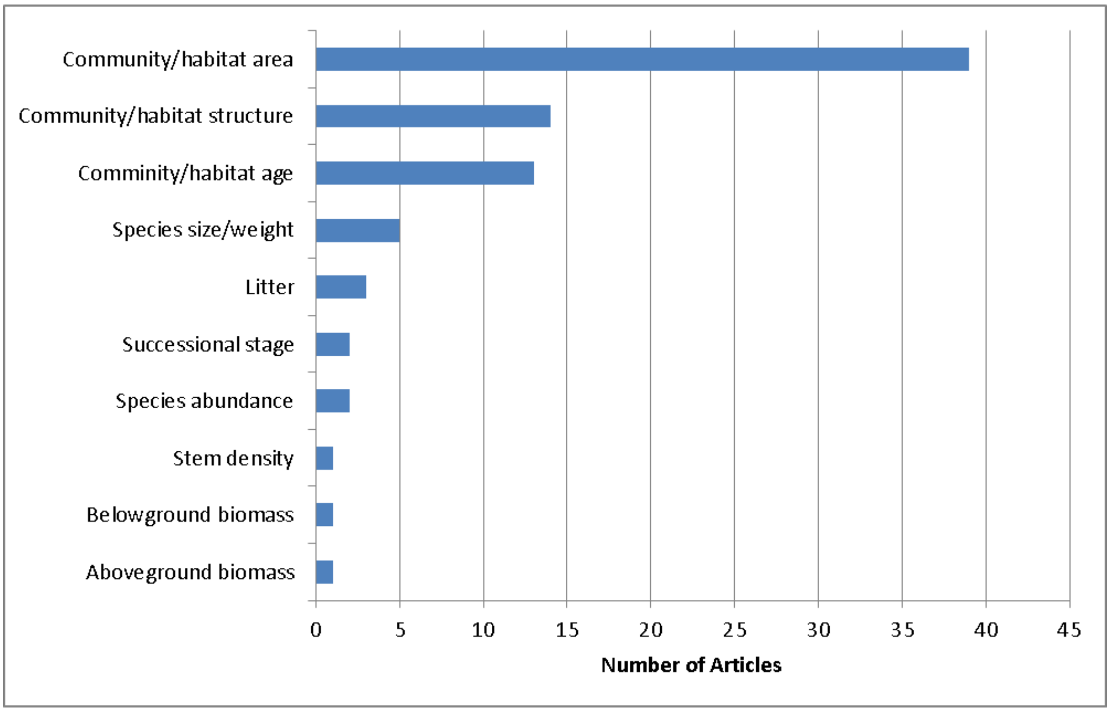

Figure A3.5.2: *Categorisation of the ESP attributes for the service of flood protection. Note: some studies used multiple attributes.*

Habitat area was the most common attribute and was discussed in 78% of studies to some extent, with a positive relationship existing between this parameter and flood protection as a result of decreases in run-off and bank discharge (Clark, 1987). This may be due to (a) the interception and evaporation of water from the tree canopy (around one-third of rainfall), and (b) the increased hydraulic conductivity of the soil as a result of tree roots and soil organisms enhancing soil structure (Clark, 1987). Much of the literature discussed the effect of habitat area in terms of wetland and forest loss. Bradshaw et al. (2007), for example, conducted a global study finding flood frequency to be negatively correlated with the extent of remaining natural forest. Modelling results from the study show that a 10% decrease in natural forest area increased flood frequency by between 4 and 28%, with a 4-8% increase in total flood duration (Bradshaw et al., 2007). The same relationship exists for coastal habitats, although a study based in Louisiana found that a 50% reduction in marsh habitat is still able to provide substantial flood attenuation (Temmerman et al., 2012). For the loss of wetland habitat in south Louisiana, the largest simulated increase in storm-surge height was 0.4 m caused by a wetland loss of over 10 km. Modelling results from two study sites in the area suggest that the ability to attenuate storm surges is dependent on location; one site having a high rate of storm surge attenuation (1 m per 25 km), and the other having a rate of only 1 m per 50 km of wetland. It is important to note that there is no evidence to suggest that the relationship between the area of any habitat (e.g. forest (Lane et al., 2003) or coastal wetland (Temmerman et al., 2012)) and flow reduction is linear. Many studies highlight the loss of wetland vegetation as a key factor in reducing the ability of wetlands to provide flood protection. Removing this natural buffer increases the velocity of run-off and soil moisture content, as the natural moisture holding capacity of these habitats is removed (e.g. Foote et al., 1996). As a result, the conversion of natural wetland vegetation is of gross concern. In India, for example, the large scale conversion of many natural wetlands to rice paddies for agriculture has greatly reduced the ability of this habitat to buffer against flooding (Foote et al., 1996). Similarly, both scrub clearance in Somerset (Acreman et al., 2011) and the conversion of native forests and grasslands in New Zealand to pine plantations (Fahey & Jackson, 1997) reduce surface friction and hence increase flood conveyance. In contrast, the restoration of wetlands (i.e. increasing habitat area) substantially increases the area of wetland drainage (Yang et al., 2010). This reduces sedimentation and peak discharge, resulting in reduced flooding; average annual reductions in peak discharge in the Canadian prairies being estimated at between 1.6 and 23.4% (Yang et al., 2010). In a global study, afforestation has been shown to reduce run-off rates over a large range of climates, the largest reductions reported for the wettest sites (Farley et al., 2005).

Surface vegetation cover is known to impact interception rates and, hence, flood run-off (Farley et al., 2005). This impact is illustrated by a study in the Roussillon area of France which examined the effect of vegetation on flash flood flows (Schmittner & Giresse, 1996). The authors identified a positive correlation between surface water discharge during heavy rainfall events and percentage vegetation cover, with a calculated correlation coefficient, r, of 0.58 ($P<0.001$). A more complete vegetative cover effectively raises the threshold for the amount of precipitation required to initiate flow (Cosandey et al., 2005). As well as the percentage cover, the type of vegetation cover is also important when considering flood protection with natural (i.e. protected woodland) or nearly natural types (i.e. not agricultural) having the highest infiltration rates and, hence, improving the effectiveness of this ecosystem service (Schmittner & Giresse, 1996). Despite this, natural cover is unable to regulate surface run-off effectively during heavy rainfall events, with even the highest performing protected forest catchment unable to mitigate flooding completely (Schmittner & Giresse, 1996). Furthermore, a review of French research on hydrological parameters in the Mediterranean found little difference in flooding and other hydrological parameters between forest and other vegetation types; a forested catchment having only slightly lower annual peak discharge than a reference grassland (Cosandey et al., 2005). The review finds that the ratio of bare to covered soil is the most important factor influencing hydrological behaviour in the region (Cossandey et al., 2005).

The age of a specific habitat was another attribute that was recorded as impacting positively on flood protection in 26% of studies, as was successional stage. Research has shown that the greatest reduction in peak flows over time occur during the first 10-years of forest growth, after which the rate of reduction slows (Robinson et al., 1991). Forest age is also known to impact on lag-time, with a 2-year old forest stand having a lag-time of 6.1 h, and a 22-year old stand a lag-time of 7.4 h (Robinson et al., 1991). The impact of stand age has been quantified for a number of afforested sites. For example, a study by Iroumé et al. (2005) found the relationship between annual run-off and age in a plantation forest to have a correlation coefficient of 0.73. The authors identified a clear reduction in annual run-off as plantations age resulting from their increased water consumption capacities through higher interception and transpiration rates (Iroumé et al., 2005). This occurs as canopy cover increases or a closed canopy has been established (Putuhena & Cordery, 2000; Robinson et al., 2003). Habitat age also has a pronounced effect on the ability of the soil to hold moisture, with one study finding a 34% difference in water storage capacity between a young and old mixed pine-beech forest stand, the oldest stand having the highest water storage capacity (Wahlet al., 2005).

In terms of the effect of forest structure, Wittmann & Junk (2003) found that a later successional stage in the white water forest communities of the Amazon relates to increased flood protection. This arises from a more complex forest structure which allows the establishment of large-stemmed or buttress building trees, impeding water flows and increasing the deposition of fine-grained sediments (Wittmann & Junk, 2003). On a smaller scale species type (e.g. in forest stands) has a considerable effect on flood protection, with contrasting stand types displaying different flood run-off coefficients. For example, Wahl et al. (2005) found pure beech stands to have a 5% higher water holding capacity than Scots pine stands; thought to be related to the larger quantities of humus associated with European Beech. However, beech is not always associated with a higher water holding capacity as modelling results suggest that the conversion of a pure Scots pine stand to a mixed pine-beech stand could increase overland and subsurface flow for up to 100-years post-conversion (Wahl et al., 2005).

Finally, species size can affect a number of hydrological parameters as noted in 10% of studies, with a positive relationship existing between species size and interception rate. Larger tree species in the Rousillon area of southeastern France intercept more water than their smaller counterparts, inducing a run-off time-lag of up to 3 minutes on small plots during heavy rainfall events (Schmittner & Giresse, 1996). In addition, a taller vegetation height is associated with an increase in surface roughness which has been shown to benefit flood mitigation; both reducing flow velocity and flood duration (Morris et al., 2005).

Discussion

The literature search

The methodology for this particular search was complex. A large proportion of papers found were not relevant to BESAFE, for example, discussing the impact of flooding on biodiversity. Therefore, many of papers used in the database were obtained through a combination of snowballing and Google Scholar searches of various keywords found in the relevant papers. This methodology was very time-consuming, and the majority of relevant papers found, provide weak to average evidence (see section on strength of evidence), focussing mainly on hydrological parameters such as run-off as opposed to referring to flooding directly. The methodology used, although not being the most efficient, was able to produce 50 relevant hits from the current literature, which appears from this review to be highly centred around the effects of flooding on biodiversity, the reverse being poorly explored.

Abiotic factors

Abiotic factors were discussed in almost a third of articles as influencing the service of flood protection. Slope inclination, for example, impacts hydrology in two ways. Firstly, areas with steeper topography generally have shallower soils and, hence, reduced vegetation cover (Cheng et al., 2002). Secondly, precipitation falling on steeper slopes has a higher kinetic energy, reaching waterways more rapidly (Schmittner & Giresse, 1996). Therefore, in areas with steeper slopes, it is likely that maximum peak flood flows will exceed the capacity of river channels (Bruijnzeel, 2004; Cheng et al., 2002) and, hence, such locations are more likely to experience flash flooding during intense rainfall events (Schmittner & Giresse, 1996). This relationship between slope inclination and flood occurrence has been quantified by Schmittner & Giresse (1996) to have a correlation coefficient of 0.52 ($P<0.001$).

Soil properties are another crucial determinant of flood run-off (Hümann et al., 2011). Soils with high porosity, such as wetland soils (porosity of 40-80%), have a high capacity to mitigate flooding (Ming et al., 2007). For example, in the Momoge National Nature Reserve in northeastern China, the amount of flood storage provided by wetland soils was calculated to be 1.03×10^{10} m^3 (Ming et al., 2007). Human land-use can affect soil porosity; with uses such as agriculture compacting the soil, preventing the vertical infiltration of water and promoting subsurface flow above the compacted layers which contributes to flood creation (Hümann et al., 2011). Age also has an effect on soil properties with established forests, for example, having relatively porous soils and high infiltration rates compared to recently afforested areas which have higher surface run-off (Hümann et al., 2011).

The effect of fire on hydrological parameters was examined by Aronica et al. (2002) for two Sicilian basins. Although previous studies report an increase in peak discharge after forest fires, both basins in the Sicilian study saw an increase in the frequency of low peak discharge events post-fire, which for one basin can be explained by a low amount of forest cover and for the other remains unclear (Aronica et al., 2002). Cosandey et al. (2005) find that sites affected by fire in the French Mediterranean have a higher peak discharge associated with heavy rainfall events compared to those which are not. The collected data suggest that during the first three-years after fire, run-off can increase by around 15% although this trend is not as pronounced across all study sites (Cosandey et al., 2005).

The negative impact of biodiversity on flood protection

Invasive species with rapid growth rates were found to negate the service of flood protection. Their presence was identified in numerous studies as having a negative relationship with flood water storage by obstructing waterways (e.g. Foote et al., 1996; Lee & Shih, 2004). For example, a study conducted by Lee & Shih (2004) concluded that a 20% reduction in *Kandelia candel* (L.) Druce after two years of flooding conditions would reduce water surface elevation by 4-76 mm in the Guandu mangrove wetland, Taiwan. This is because of their extensive root systems which increase sedimentation rates and the presence of this mangrove species which raises surface water elevation. Similarly the spread of the invasive woody shrub Tamarisk (*Tamarix sp.*) in arid- and semi-arid areas of the western USA is associated with increases in river channel sedimentation and, hence, increased frequency and severity of flooding (Zavaleta, 2000). The removal of invasive species such as these will reduce sediment deposition rates and lower flood risk (Foote et al., 1996; Lee & Shih, 2004; Zavaleta, 2000).

Strength of evidence

The majority of articles found in this review presented average (50% of studies) to strong (<25% of studies) evidence for a link between biodiversity and flood protection. Despite this, numerous studies demonstrated weak (16%) or very weak evidence of this link (8%). These studies were based on a linkage or theory without conclusive evidence, stating only 'general knowledge' without referencing the source of information. Foote et al. (1996), for example, discuss the process of wetland loss in India, acknowledging that the ability of wetlands in India to provide flood protection through storage is reduced when turned into rice fields, but providing very little evidence in terms of referenced articles to support this and no direct observations or quantitative results. This study attempts to give a wide overview, but in doing so the evidence for links between biodiversity and flood protection are very weak, for example, the study states that the use of terraces for crops such as pulses, nuts and oranges on the slopes of the Himalayas is thought to increase run-off rates and reduce snow-holding capacity, but no evidence of this impact is given (Foote et al., 1996). Acreman et al. (2011) examines the trade-off in various ecosystem services provided by the Somerset levels and Moors wetlands, stating that they provide flood water storage. The authors identify a negative relationship between scrub clearance and flood storage due to reduced friction and increased flood conveyance, but fail to provide reasoning for this or references to studies which found similar results.

Factors not captured in the database
A number of factors were raised in the studies reviewed that were not captured in the database. These include, for example, the effect of management, including soil management techniques (e.g. no-tillage; Martin, 1999) and the removal of woody debris (Erskine & Webb, 2003). In addition to these, actions such as strip cropping can also mitigate against flooding although crop type is an important factor here, with sorghum, maize, wheat and barley able to provide sufficient resistance to flood flows (Smith et al., 1990).

In addition to management, the importance of location is another factor poorly represented in the database. For example, as far as the loss of coastal marsh habitat is concerned, the location of die-back in reference to river channels impacts highly on flood conveyance, with a 50% die-off of vegetation directly adjacent to the channels resulting in a 0.8 m rise in peak water levels based on modelling compared to a considerably lower rise if the die-off were to occur far from the channels or at random (Temmerman et al., 2012).

A3.5.2 Results – Flood protection and value linkages

The results from the review of valuation of flood protection are summarised in Table A3.5.1.

Table A3.5.1: *Results from the review of valuation of flood protection.*

	Avoided costs	Stated preference	Benefit transfer	Multi-criteria	Total
Ecosystem service provider					
- wetland	1			1	4
- mangrove	3	1	1		3
- forest	2				2
Value					
- indirect use value	6	1	1	1	9
Spatial scale					
- local	2				2
- regional	4	1		1	6
- national			1		1
Beneficiaries					
- households/individuals	4				4
- firms	1				1
- communities/society	2	1	1	1	5

It can be seen from Table A3.5.1 that all nine reviewed studies estimate indirect use value. The ecosystem service providers are wetland (4), mangrove (3) and forests (2). These ecosystem service providers are beneficial to society because they protect society from damages caused by floods.

The spatial scale of the studies is mainly regional (6) with only two local and one national study. The beneficiaries of flood protection are either specific households and firms (farmers) or society in general.

The benefits of flood protection are the avoided damage cost, and this is reflected in the choice of valuation method. Six studies use the avoided cost method, including two studies where the avoided costs are equal to avoided costs of alternative protection (averting behaviour). Only one study used each of the stated preference method, benefit transfer and multi-criteria method for valuation.

Discussion

Three methods for the valuation of water flow regulation (flood protection) are used in the scientific literature:

- Avoided damage costs by the ecosystems' protection of land against flooding;
- The costs of alternative protection measures against flooding; and
- Revealed willingness to pay for protection against flooding.

In all studies, except the meta-study by Newton et al. (2012), the valuation concerns a geographically specific area. The value of the flood protection function of ecosystems necessarily depends on the specific conditions of the area protected - how land is used, how many households live in the area, the infrastructure, etc.

In five of the reviewed papers the valuation of the ecosystem service is based on the calculated damage costs of floods. This also seems to be the methodologically most reasonable approach. Of course it not an easy task to calculate these costs and, in fact, valuation in all the reviewed studies can be criticised for just being based on the damage costs of a single flood incident.

A correct way to calculate avoided expected damage costs would be, first, to evaluate the annual risk (probability) of floods if the protecting ecosystem was destroyed. This is a hypothetical situation and therefore difficult to describe. Second, the expected mean damage cost of a flood incident should be estimated - possibly based on earlier experiences. Finally, expected annual avoided damage costs can be calculated as annual flood risk multiplied by expected mean damage costs.

Correct calculation of expected annual avoided damage costs may not be an easy task. Therefore, calculation of the costs of alternative protection measures against flooding, which is undertaken in two of the reviewed papers, may be a more practicable approach. These costs do not represent the value of the protected area. If it should pay to build flood protection installations, e.g. breakwaters, the value of the protected area should be bigger than the installation costs. However, if it is assumed that the area with its production activities, houses, infrastructure and non-market goods is so valuable that it should be protected in any case then the costs of alternative protection measures may be a reasonable indicator of the value of the ecosystems' water flow regulation services.

The value of water flow regulation (flood protection) can be based on willingness to pay for flood protection expressed in stated preference studies (one paper) or relative value flood protection expressed in multiple criteria analysis (two papers). Willingness to pay by individuals or households expresses their own evaluation of how much flood incidents will cost them in the future. The problem with this is that these expectations may be wrong and, therefore, it may be better to base the valuation on scientists' objective evaluation of risks and costs. It is also a problem that the expressed willingness to pay also may cover willingness to pay for other services than just flood protection. It may be difficult for respondents to distinguish between the different services and value each of them singularly. For example, inhabitants may have a willingness to pay for protection of a mangrove area, but how much of this is willingness to pay for flood protection by the mangrove may be difficult to state.

The same objections may be raised against the results of multiple criteria analysis. In addition, the relative values stated are not necessarily in monetary terms, which makes it difficult to use the results in connection with valuation of ecosystem services.

A3.6 Mass flow regulation (erosion protection)

A3.6.1 Results – Biodiversity and erosion protection linkages

Spatial and temporal scale

The majority of articles (76%) included in the review considered erosion protection at the local spatial scale; the studies often confined to small valleys or catchments in mountain or hilly areas and/or single farms or sand dune systems. Studies at a sub-national (22%) spatial scale usually covered larger mountain regions or catchments or were review articles. Only one article included within the database covered erosion protection at a global scale in a review study (Stokes et al., 2009).

The longer term nature of many erosion processes and erosion protection systems is reflected in the decadal or longer time scale addressed by most (64%) of the studies. Some studies (32%) fit within an annual time scale and often involved experimental measurements (e.g. De Baets et al., 2006) or field surveys (e.g. Isselin-Nondedeu & Bedecarrats, 2007). Similarly, the 4% of studies that involved very short (daily) time scales were experimental manipulations involving particular plants and their effects on soil loss and runoff under different conditions (Andry et al., 2007; Casermeiro et al., 2004).

Ecosystem service provider (ESP)

For populations of most plant species, the "habitat" comprises the set of abiotic conditions that the individuals experience, together with the "community" of other plants with which they coexist and interact. Thus there is considerable overlap in the meaning of these two terms in the present context of erosion protection service provision.

Keeping this in mind, it is clear from FigureA2.6.1 that in most of the studies the ESP was either the entire community or habitat (46%) or two or more communities or habitats (28%). A considerable number of studies focused on erosion in farmland situations. These encompassed erosion protection on abandoned and poorly managed farmland. For example, Quinton et al. (1997) demonstrated that protection from erosion on abandoned sites can be achieved by establishing a community of shrubs with a least of 30% canopy cover. Further, Erdogan et al. (2011) modelled the impact of land-cover change, increasing the amount of cropland, on soil loss potential based on rainfall, soil type, topography, land use and management. Zhang et al. (2008) assessed the benefits of terracing on sloped land, and Keay-Bright & Boardman (2007) and Sansom (1999) investigated erosion in grazing systems.

It is clear that there must be overlap between some of the above studies involving farmland habitats and those, also numerous, studies in which mountain or uplands are the habitats/communities considered. Indeed, vegetation on a sloping topography is perhaps the most common theme of erosion control in the studies within the present review. Mountain plant communities on ski slopes (Dinger, 2002) or in avalanche zones (Krautzer et al., 2011), or successional vegetation after forest clear-cutting (Novakova & Krecek, 2006b) provide some of the diverse range of non-farming mountain examples. Studies of the stabilizing effects of plant communities on sand dunes give one of the classical examples of plant diversity and erosion control and a number of such studies are included in the review (e.g. Henriques et al., 1984; Malkinson et al., 2003).

Studies that involved a single specific species population (4%) or two or more specific species populations (10%) as the ESP for erosion control were seldom in the review and were restricted to studies in which only one plant species was present (Gadgil, 2002) or where one or only a small number of plant species were the subjects under investigation (e.g. Andry et al., 2007; Zuazo et al., 2006; 2008).

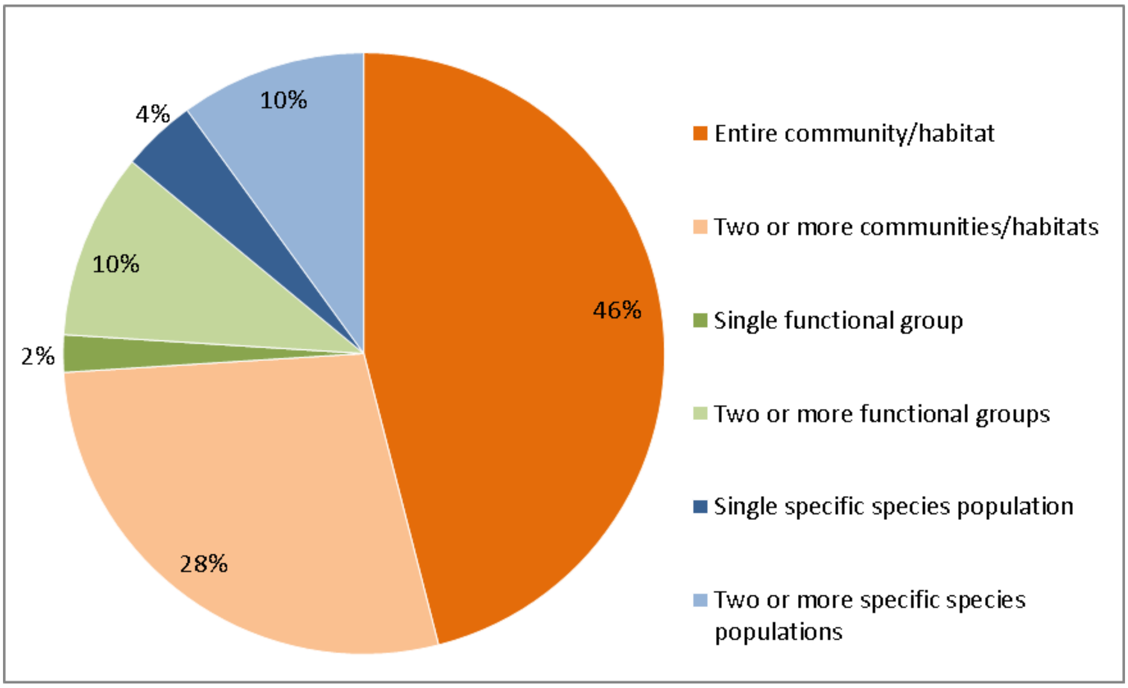

Figure A3.6.1: *Categorisation of ecosystem service providers (ESP) for the service of erosion protection.*

Important attributes of the ESP

Many of the studies in the review did not focus on, or even identify, particular attributes of species or other biological groupings associated with the provision of erosion protection services. Indeed, the level of reporting was often so general that it was difficult to establish even inferred relations between biodiversity and erosion protection. However, some studies did include details of one or more relevant attributes, as summarised in Figure A3.6.2.

In considering the species-associated suite of attributes, species abundance (10% of studies), measured by degree of ground cover, was found to be important for erosion protection in a number of situations, for example, in road embankment experiments using different degrees of cover of *Sedum sediforme* (Andry et al., 2007). Evidence for the contribution of species richness to erosion protection (14%) was variable. Although species richness may be beneficial (e.g. Wang et al., 2005), the study by Casermiero et al. (2004) concluded that biodiversity did not seem to be important in soil protection in the studied area (Spain). Species size was implicated in 6% of studies, as it is associated with the amount of shelter given to the ground beneath. For example, Imao (1982) showed that the runoff ratio of rainfall on sloped farmland decreased with crop growth.

The same principle applies to aboveground biomass, as noted in 26% of studies. For example, Casermeiro et al. (2004) showed that plant cover is the main factor reducing surface runoff and the movement of sediments. Belowground biomass (28% of studies) may be intuitively one of the most important attributes for erosion protection and indeed the evidence appears to support this. For example, a study by De Baets et al. (2007) investigated plant species and their erosion-reducing potential by measuring root density, root length density and root diameter, and a study by De Baets et al. (2006) investigated grass root density and soil detachment rate.

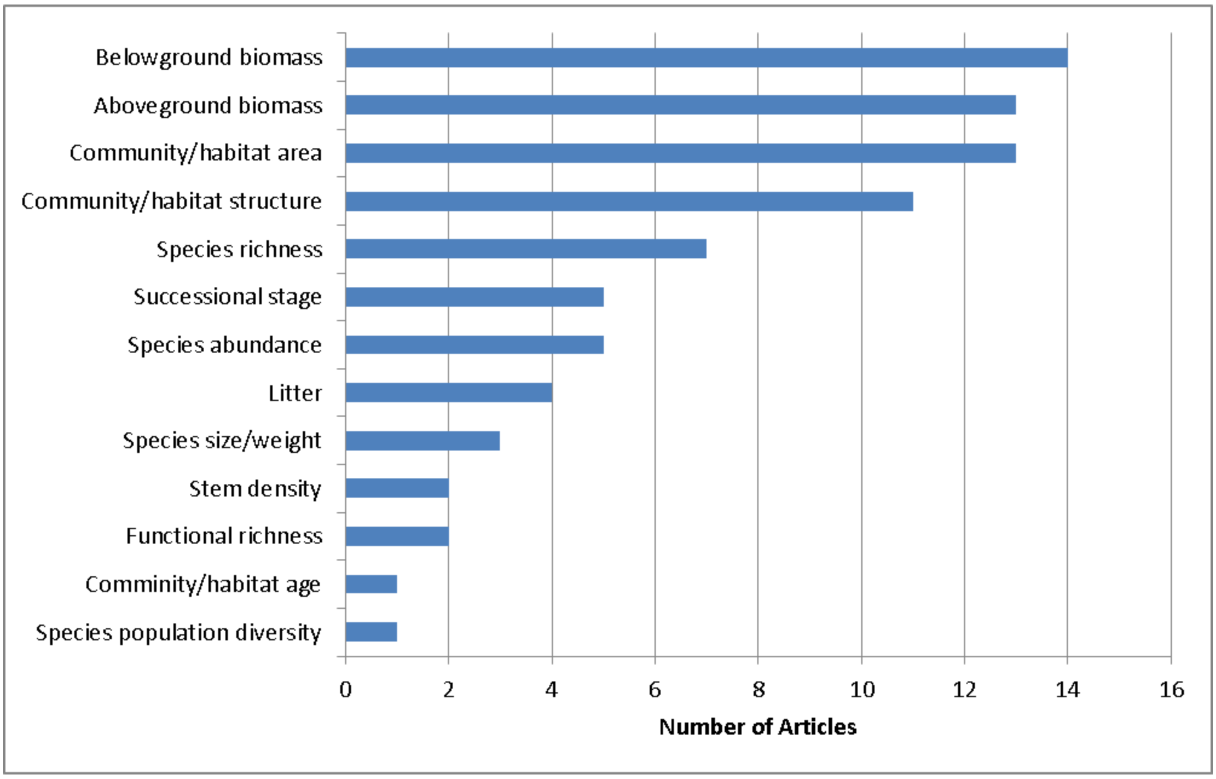

Figure A3.6.2: *Categorisation of the ESP attributes for the service of erosion protection. Note: some studies used multiple attributes.*

Successional stage of the vegetation was mentioned in 10% of the studies, and later successional stage (and hence more complex vegetation) is beneficial in sand dunes (Malkinson et al., 2003), in landslide sites (Walker et al., 2009) and on abandoned, previously farmed steep slopes (Cammeraat et al., 2005). The role of functional diversity for erosion protection was poorly documented (4% of studies) and seemed unclear (e.g. Lemauviel & Roze, 2003). Finally, a number of studies (37%) discussed attributes which are classified under “other” in the database, covering a diversity of parameters that were sometimes clear (e.g. the plant communities occurring on terraced slopes as reported by Zhang et al., 2008 and Yu et al., 2012), but were often less well defined, such as the influence of land management on soil erosion on farmland (Keay-Bright & Boardman, 2007).

Discussion

The literature search

Although the search provided very many hits, it quickly became clear that most were irrelevant to the aim of relating aspects of biodiversity to erosion protection. Indeed, many of the search terms, though present in the articles, were used merely as “advertisements” and bore no relation to the content of the study. Thus, it proved extremely difficult to reach the target of 50 articles and a considerable number of those studies that are included are of less relevance than desired. However, the articles selected do illustrate the present state-of-the-art for research on provision of erosion protection services as related to biodiversity.

Abiotic factors

Almost a third of studies (30%) relevant to the BESAFE review mentioned abiotic factors as influencing the provision of this ecosystem service. The most commonly mentioned factor was precipitation (mentioned in 24% of entries); with an experimental study by Andry et al. (2007) treating contrasting vegetation covers with different simulated rainfall intensities. Slope was cited in 16% of articles, and soil related variables in 14% of articles.

The negative impact of biodiversity on erosion protection

Only a single study indicated a negative impact of plant species communities in erosion protection. Cammeraat et al. (2005) investigated the complexity between succession, root systems and landslide activity on steep abandoned farmed slopes in Spain. Mass wasting processes (downward movements of soil, etc.) increased over time after abandonment with increasing cover of vegetation following natural succession. Thus, erosion appeared to increase with the colonisation of early successional plant communities. However, this effect was reversed after 40 years when mass movement activity was reduced with the establishment of later successional plant communities. Two further studies described some aspect of biodiversity as having a negative impact on erosion protection. One described how rabbits fed on seedlings of the sand-binding marram grass, *Ammophila arenaria (L.)*, and hence it was only in sites where rabbits had been eradicated that sand stability able to be established and maintained (Gadgil, 2002). A second study by Keay-Bright & Boardman (2007) discussed how past grazing in the Sneeuberg Mountains of South Africa was likely to be the most causal factor behind the badlands and gullies in the study area.

Strength of evidence

Almost half (48%) of the studies were judged to provide weak or very weak evidence for a relation between biodiversity and erosion protection services. This is in comparison to 36% providing average, but valid, evidence and 16% providing strong evidence. No studies were judged to provide very strong evidence. This skew towards weaker evidence provided by the studies in the review may be taken to reflect the research state-of-the-art, as yet, not much attention has been paid to the relation between biodiversity and erosion protection. Of course, the relations between soil stability and vegetation, particularly root complexity and aboveground cover, are well known, but apparently these have not been primary subjects of recent research for purposes of erosion protection service provision. Instead, many studies, including a considerable number in the present review, report on the causes of erosion and were not designed to consider its remedies.

A3.6.2 Results – Erosion protection and value linkages

The results from the review of valuation of erosion protection are summarised in Table A3.6.1.

Table A3.6.1: *Results from the review of valuation of erosion protection.*

	Avoided costs	Stated preference	Benefit transfer and reviews	Other	Total
Ecosystem service provider					
- agriculture/rural areas				1	2
- river basin		1	1		2
- forest	2	1	4	2	8
- other			5		5
Value					
- direct non-consumptive use		2			2
- indirect use value	2	2	10	3	17
- non-use		1			1
Spatial scale					
- local		1	3		7
- regional			3	3	3
- national		1	1		2
- global	2		3		5
Beneficiaries					
- households/individuals		1			1
- communities/society	2		8	3	13
- group of stakeholders		1	2		3

As seen from Table A3.6.1 the main emphasis of assessment is on indirect use values; hence, only three studies also assess other values (two assess direct non-consumptive use values and one assesses non-use value). While society is the prime beneficiary considered in the studies (13 out of 17 studies), three studies assess the value accruing to groups of stakeholders and one study assesses the value accruing to households or individuals. The spatial scale varies across the studies with seven studies focusing on the local scale, three on the regional, two on the national and five on the global scale. In terms of the ecosystem service provider, two studies focus on agriculture, two on river basins, eight on forests and five on 'other'. The most utilised valuation method is the benefit transfer method which is used in ten of the studies. The other studies rely on stated preference methods (2), avoided costs (2) and 'other' (3).

A3.7 Atmospheric regulation (carbon sequestration)

A3.7.1 Results – Biodiversity and atmospheric regulation linkages

Spatial and temporal scale

The spatial scale at which this ecosystem service was considered varied from continental to local. Over half (64%) the database entries considered the latter. Zhao et al. (2010), for example, examined the carbon stored by urban forests in the Hangzhou metropolitan area, eastern China; and Potvin et al. (2011) studied the effects of biodiversity on carbon storage for a tropical tree plantation in Panama. The second most common scale was sub-national, considered by 30% of studies, such as Bai et al. (2011) who investigate carbon storage in the Baiyangdian watershed, an area of 31,200 km^2 spanning 33 counties and a range of environments. Other sub-national studies include the study of forest plantations along the Pearl River Delta in southern China (Zhang et al., 2012) and the study of boreal forests in northern Sweden (Yurova & Lankreijer, 2007). The national to sub-continental scale was considered in very few studies (4%), but includes a study by Caspersen & Pacala (2001) who utilised data from the US Forest Service to study 24,670 natural forest plots across the US. Only one study was found which considered carbon storage on a continental scale, this examining the carbon balance of six shrublands across Europe (Beier et al., 2009).

The literature found in this review considers carbon storage on either an annual (28%) or decadal scale (72%). Annual or shorter timescales include themes such as the seasonal changes in soil organic carbon storage in response to grazing (Sun et al., 2011); and the net annual C-sequestration rates for various forest types (Kaul et al., 2010). Steinbiss et al. (2008) provide another example of a short temporal scale, examining the impact of plant diversity on soil carbon storage in experimental grassland plots over a 4-year period. In contrast to these shorter timescales, Mills & Cowling (2006) consider carbon sequestration on a decadal scale, estimating the total carbon storage at two thicket restoration sites in South Africa. The literature review also found evidence of the impacts of climate change on carbon storage being considered. For example, Seidl et al. (2008) simulate the potential impact of bark beetle invasion on forest carbon stocks over the next century, and Bunker et al. (2005) examine the potential impact of climate change related species extinctions on carbon storage in a tropical forest.

Ecosystem Service Provider (ESP)

This review found that studies examining carbon storage considered a broad range of ESPs (Figure A3.7.1). Over half the articles (56%) examined the service as being provided by an entire community or habitat. Examples include temperate forest habitats in Australia (Hantanaka et al., 2011), the Alaskan boreal spruce ecosystem (Hollingsworth et al., 2008) and tropical forest in Panama (Bunker et al., 2005).

Many studies (28%) compared the carbon storage provided by two or more communities or habitats. This includes Kirby & Potvin (2007) who examined the difference in carbon storage between trees in tropical moist forest, agro-forest and pasture in Panama. The authors identify that different species in each land-use type have variable carbon storage potential, with espave (*Anacardium excelsum*), for example, contributing around 16% of the total carbon stocks per hectare in the tropical moist forest habitat. Other authors compared carbon storage in different plantations, including Moso bamboo (*Phyllostachys heterocycla*) and China fir (*Cunninghamia lanceolata*) (Yen & Lee, 2011), as well as *Michelia macclurei* (Dong et al., 2009).

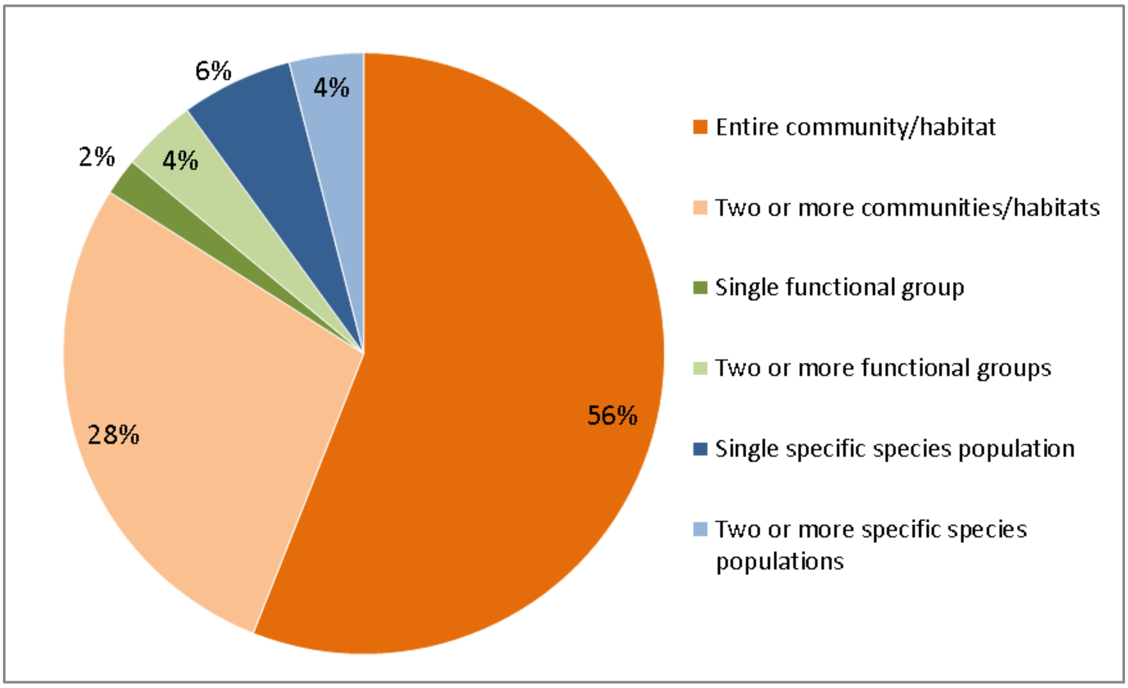

Figure A3.7.1: *Categorisation of the ecosystem service provider (ESP) for the service of atmospheric regulation.*

The carbon storage achieved by a single specific species population was rarely considered in the literature (6% of articles in the review). Two examples, however, include the carbon sequestration in soils by the bioenergy crop *Miscanthus x giganteus* (Zimmermann et al., 2012), and a Chinese fir plantation (Zhao et al., 2009).

The ESP class "two or more functional groups" was explored in only 4% of studies. Firstly by Steinbeiss et al. (2008) who compare 16 experimental grassland plots, and secondly by Cahill et al. (2009) who compare the role of C4 and C3 grasses in the Wisconsin prairies, USA. Tolbert et al. (2000) is one of the few studies found to examine carbon storage by multiple species populations. The authors studied carbon sequestration provided by various biomass crops, including Sweetgum (*Luqyudanver stracifula*) and cottonwood (*Populus deltoides*). Finally, only one study in this review discussed a single functional group as the ESP, this being an alpine grassland in the central French Alps which was studied for plant functional trait relationships (Lavorel & Grigulis, 2012).

Important attributes for the ESP

A variety of links between aspects of biodiversity and carbon storage were identified in the literature. The most common attribute discussed was the effect of stand age (cited in 30% of entries, Figure A3.7.2). Overall larger carbon storage was found in older tree species due a combination of (a) the time period over which they have sequestered carbon, and (b) the result of size increasing with age (e.g. Hantanaka et al., 2011; Keeton et al., 2010; Kirby & Potvin, 2007; Zhao et al., 2010). In grassland habitats, a positive relationship has been identified between plant height, aboveground biomass and carbon storage (Lavorel & Grigulis, 2012). In forest environments, large trees, such as those with a diameter at breast height (DBH) ≥ 10cm were found to account for over 90% of the aboveground carbon stocks in forest and agroforest habitats, and for over 50% of the aboveground carbon stock in pasture in eastern Panama (Kirby & Potvin, 2007). As a result of this relationship, it is possible to estimate carbon storage as a function of DBH, although this needs to be tested under a range of site conditions, age and size ranges (Laclau, 2003). In contrast to the above studies, Hollingsworth et al. (2008) found that the relationship between DBH and carbon storage may not be valid for all forest stands. The authors examined regional soil carbon storage in Alaskan boreal forests and found only a weak relationship between the basal area and total soil carbon (r^2=0.16) for mature stands (between 44 and 295 years) (Hollingsworth et al., 2008). Furthermore, across all stands studied, the basal area consistently displayed a negative relationship with organic, mineral and total soil carbon content (Hollingsworth et al., 2008).

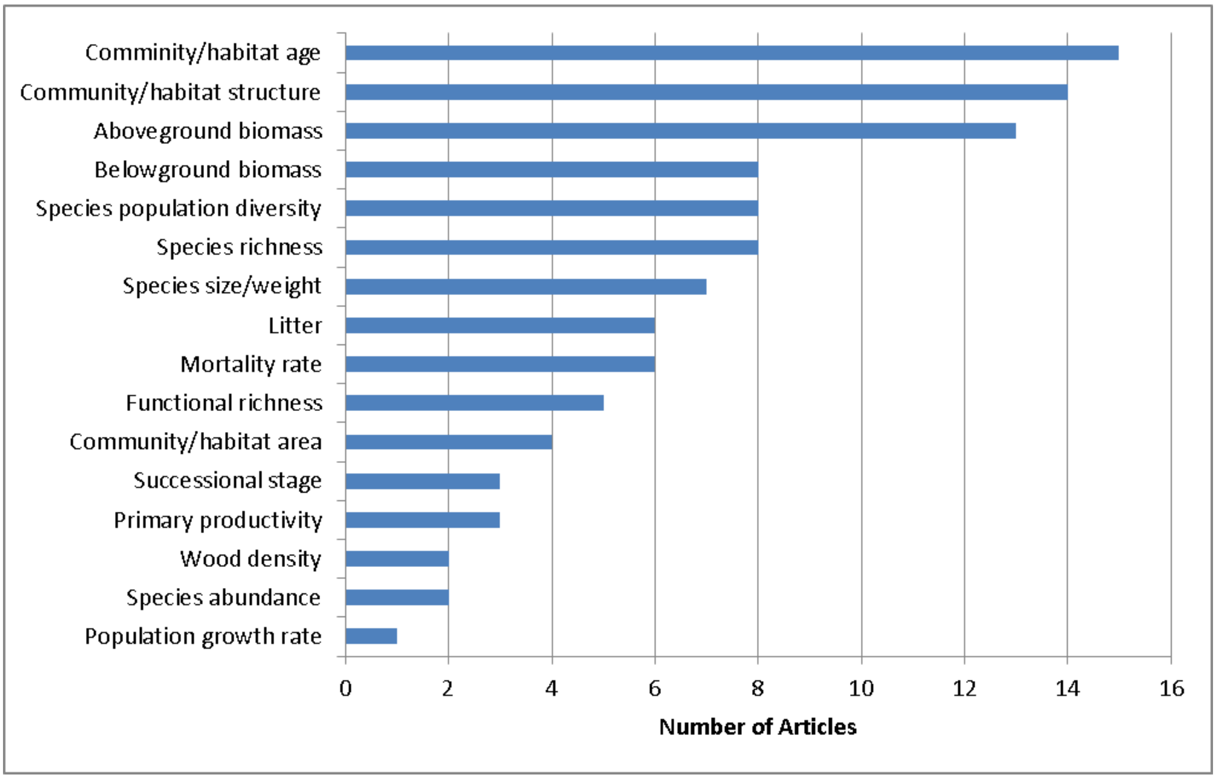

Figure A3.7.2: *Categorisation of the ESP attributes for the service of atmospheric regulation. Note: some studies used multiple attributes.*

The effect of age on carbon sequestration appears to be variable. For example, Law et al. (2003) found carbon storage in live mass to reach a maximum at around 200-years in ponderosa pine stands. In addition to increasing carbon storage with age, the ratio of aboveground biomass displays a strong correlation with total ecosystem carbon, which itself is correlated to stand age (r^2=0.91) (Law et al., 2003). Hantanaka et al. (2011) also studied the effect of age on carbon storage, highlighting firstly that carbon stocks do not increase linearly with age, with younger forests having higher rates of carbon uptake (see also Keeton et al., 2010); and secondly that mature forests have a qualitatively different form of carbon organisation than their younger counterparts (Hantanaka et al., 2011). For example, mature temperate forest was found to have similar levels of carbon storage to some older regrowth forest (>20 years) estimated at >300 t ha^{-1} (Hantanaka et al., 2011). In contrast, younger regrowth forest stands were found to store substantially less carbon (<180 t ha^{-1})

(Hantanaka et al., 2011). Research has also shown older growth forests to store more carbon in the arboreal layer, Qi et al. (2010) finding these to store 164 Mg C ha^{-1} compared to the 25-71 Mg C ha^{-1} stored by young larch plantations in northeast China. In contrast with this increase in carbon storage with age, one study examining China fir forests found the mean carbon sequestration to decrease with age class (Yen & Lee, 2011).

Species richness was highlighted in a number of studies as having a positive relationship with soil carbon. This varied in strength, with Kirby & Potvin (2007) finding an r^2 correlation coefficient of 0.59 ($p<0.001$) for forests in eastern Panama, and Steinbiss et al. (2008) reporting a correlation, r, of 0.34 ($p=0.002$) for experimental grassland plots in Germany. Chen (2006) found similar results for an old-growth mixed broadleaf Korean pine forest at a small scale, with tree species evenness having a significant linear relationship with the natural log of total tree carbon storage. In contrast, at a larger scale, stands with the same species richness can have considerably different amounts of carbon storage (Chen, 2006). As far as soil carbon storage is concerned, fungal biodiversity is an important parameter to consider; having a strong positive relationship with carbon storage. The strength of this relationship is such that Persiani et al. (2008) estimate that a 10-fold increase in fungal richness in Mediterranean grassland soil equates to a 27-fold increase in carbon accumulated from litter and roots. However, further research shows that if more than 20 species of fungi taxa are present, the soil exhausts its capacity for carbon storage and, hence, soil carbon does not increase after this point (Persiani et al., 2008). It is important to note that there is evidence that this saturation phase in species richness also occurs in natural forest environments (see Ruiz-Jaen & Potvin, 2011). As far as the rate of increase in carbon storage is concerned, Balvanera et al. (2005) found an asymptote distribution with cumulative carbon storage increasing rapidly as species richness begins to increase in tropical forest in Mexico. Finally, as the effect of species richness on carbon storage is so large, some authors conclude that this attribute is, at least in the short-term, more important than differences in plant biomass inputs for carbon storage (Steinbiss et al., 2008).

The literature was somewhat conflicting over the impact of diversity on carbon sequestration. Where species diversity was concerned, Sharma et al. (2010) found that tree diversity in the Himalayas was significantly negatively correlated with total carbon density of forest stands ($r = -0.25$, $p = 0.05$), implying that stands with higher diversity are not rich in carbon density. In contrast to this finding, work by Potvin et al. (2011) suggests that maximum aboveground carbon pools in tropical plantations increase significantly less over time in monocultures than in mixture plots, indicating that low species diversity reduces the rate at which trees sequester carbon. In agreement with this, spatial correlation over the Baiyangdian watershed, China, finds the relationship between species biodiversity and carbon storage to be significantly positive ($r = 0.55$) (Bai et al., 2011).

As far as successional diversity is concerned, research shows that forest stands in the US with high successional diversity fix and store more carbon than those with a low successional diversity, regardless of successional composition (Caspersen & Pacala, 2001). The effect of functional diversity was also examined in the literature. This has an important influence on soil carbon stocks with, for example, the diversity of *Sphagnum* moss species being a good predictor of soil carbon sequestration in Alaskan black-spruce forests (Hollingsworth et al., 2008). Additionally, Persiani et al. (2008) suggest that morpho-functional diversity in the Mediterranean influences the capacity of the soil to sequester carbon with a low rate of soil carbon turnover in grasslands reflecting a high fungal biodiversity.

Discussion

The importance of specific species or ecosystem types

A number of studies highlight the importance of certain species for carbon storage. For example, Kirby & Potvin (2007) find that for a tropical moist forest in Panama, some species store substantially more carbon than others. These include espave (*Anacardium excelsum*), which accounted for approximately 16% of total C-stocks per hectare, with *Cavanillesia plantanifolia*, *Castilla elastica* and *Quararibea asterolepis* also storing large amounts of carbon compared to other species (Kirby & Potvin, 2007). In China, research has shown that only five species in an old-growth Korean pine forest (*Fraxinus mandshurica*, *Pinus koraiensis*, *Quercus mongolica*, *Tilia amurensis, Acer mono*) contribute 85% of stand C-storage (Chen, 2006). Similarly in a conserved forest in Mexico, over 95% of species were classified as being functionally unimportant for carbon sequestration, with 90% of the total C-storage provided by only 13% of species (e.g. *Dialium guianense*) (Balvanera et al., 2005). Forest type has also been found to affect carbon sequestration with conifer-dominated forest types found to have a higher carrying capacity of carbon stocks than broadleaf-dominated forest types (Sharma et al., 2010). As a result, it is suggested that protecting the conifer-dominated forest has the largest impact on reducing emissions from deforestation (Sharma et al., 2010).

Abiotic factors

Climate (temperature and precipitation) was found in 16% of studies to influence carbon storage. For example, a European scale study of carbon sequestration in shrubland found the largest storage and sinks of carbon to be associated with wet and cold climates, which have a particularly large amount (95%) of their carbon stored in the belowground biomass (Beier et al., 2009). Considering future climate change, model simulations suggest that reductions in the optimum temperature for growth of many species may result in reduced carbon storage. For example, a 10% decrease in the optimum temperature for growth for Chinese fir has been simulated to result in a 4.5, 7 and 10% decrease for stem, root and foliage carbon storage, whereas if optimum temperature is decreased by 20%, this could reduce carbon storage in the above vascular plant parts by as much as 15% (Zhao et al., 2009).

Research indicates that soil carbon sequestration (in 20 cm and 100 cm profiles) is positively correlated with precipitation; model simulations showing that a 96 mol C m^{-2} reduction in soil carbon could occur in response to a reduction in precipitation of 100 mm yr^{-1} (Meier & Leuschner, 2010). This effect has been observed along a transect of beech stands in central Germany; those receiving <600 mm yr^{-1} precipitation having on average 23% less soil carbon storage than stands receiving >900 mm yr^{-1} precipitation (Meier & Leuschner, 2010). Heavy rainstorms can also have an effect on forest carbon sinks. For example, Chen et al. (2012) found the carbon sequestered in a tropical mountain rainforest to increase with the number of heavy rainfall storm events, making these forests serve as a carbon sink during wet months. In contrast, during dry months, high amounts of evapotranspiration were found to suppress tree growth, decreasing the amount of carbon sequestration provided by the trees, and in some cases switching the studied tropical forest from a carbon sink to carbon source (Chen et al., 2010).

The effect of soil pH was examined in a number of studies. For example, an investigation into regional soil carbon storage in Alaskan black spruce forests found acidic communities to have larger total soil C-storage as a result of them having significantly higher amounts of carbon in the organic layer than in non-acidic black spruce communities (Hollingsworth et al., 2008). In contrast, modelling results from Zimmermann et al. (2012) for fields planted with the bioenergy crop *Miscanthus x giganteus* suggest that a positive relationship should exist between soil pH and SOC stocks, although field study results indicate no significant evidence of this effect whereas control sites planted with barley display the opposite trend (Zimmermann *et al.,* 2012).

In addition to pH, soil texture was also found to influence soil carbon sequestration. A study of the prairie ecozone in Canada found SOC to be higher in fine- (10.4±0.6 kg m^{-2}) rather than coarse-textured soil (6.2 ± 0.4 kg m^{-2}), with the largest contrasts seen at 0-3 cm and 3-10 cm depths (Bai et al., 2009). Furthermore, soil temperature has been found to have a significant effect on soil carbon dynamics in a range of soil environments in northeastern Spain (Emran et al., 2011). For example, the soil CO_2 flux from pine tree stands (*Pinus pinea*) and cultivated vines (*Vitis vinifera*) were both found to correlate highly with soil temperature (r=0.82; r=0.95 respectively) (Emran et al., 2011).

The impact of wildfire on carbon storage was discussed in a number of articles (e.g. Jonsson & Wardle, 2010; Mack et al., 2011; Richards et al., 2011) as a major driver of ecosystem carbon sequestration (Conard & Ivanova, 1997). Multiple studies discussed the effect of the fire return interval (FRI) on carbon emissions. High FRIs are associated with increases in carbon storage (Jonsson & Wardle, 2010). For example, modelling results suggest that a doubling of the FRI in Russian boreal forests would lead to a 15% increase in carbon sequestration, whereas a decrease is estimated to cause a 10% reduction in carbon sequestration over a 50-year period (Conard & Ivanova, 1997). This increase in carbon storage with increasing FRI occurs as a result of the interplay between (a) an increased soil mineral nitrogen and tree net primary productivity under a low FRI, and (b) greater charcoal and grass inputs as a result of higher FRIs, increasing the size of the passive carbon pool (Richards et al., 2011).

The negative impact of biodiversity on carbon storage

Invasive species were the only instance in the literature of biodiversity negating this ecosystem service. Invasive species in Californian grasslands, for example, are causing an average reduction in carbon storage of 40 Mg ha^{-1} in the upper 50 cm of the soil, which is suggested to be a result of the low net primary productivity rates in non-native annuals relative to perennial grasses (Korteen et al., 2011). As far as fauna are concerned, the abundance of invasive bark beetles is expected to increase with climate change (Seidl et al., 2008). This will increase bark beetle induced damages to trees causing a marked decrease in above and below ground carbon stocks (Seidl et al., 2008), with severe bark beetle damage scenarios indicating that this could cause the carbon balance of the trees to become negative (Seidl et al., 2008).

The importance of management

Many articles highlighted the impact of management including felling, afforestation and grazing activities on carbon storage. For forestry, it was found that the felling of larger trees more clearly reduces forest carbon stocks (Glenday, 2008), and although avoided deforestation would have the largest impact on landscape-level carbon stocks, actions such as increasing the rotation length can also increase C-sequestration by trees (Kirby & Potvin, 2007). In Indian Sal forests, for example, it is estimated that an increase in rotation length from 120 to 150 years could increase carbon storage by 18% (Kaul et al., 2010).

In grassland habitats, grazing activity and agriculture impact strongly on carbon storage. This has been found to alter plant traits and trigger faster carbon cycling (Klumpp et al., 2009), with increases in the intensity of grazing able to shift alpine meadow soils in the Tibetan Plateau from being carbon sinks to sources (Sun et al., 2011). A few studies also examined the impact of soil management on atmospheric carbon regulation with, for example, practices which reduce soil disturbance (e.g. no-till) increasing carbon sequestration in soils (Mishra et al., 2010).

Strength of evidence

The strength of the link between biodiversity and carbon sequestration in the majority of articles was average (76%), whereas a few articles (6%) provided weak to very weak evidence of this link. These include a study by Hector et al. (2011) which is an introductory paper concerning the Sabah Biodiversity Experiment, a long-term field experiment in Borneo. The paper states some current knowledge and the experimental design of the project, with the aim being to understand the relationship between tree species diversity and the functioning of a lowland rainforest. As this experiment is not yet complete, the authors are only able to report previous findings, without primary results to support these. Other papers where the evidence was rather “weak” include a study by Conard & Ivanova (1997) on the potential impacts of fire regime on carbon emissions and a study by Bai et al. (2011) who used spatial correlation to identify that biodiversity hotspots often co-occur with ecosystem services such as carbon sequestration, with no explanation about the cause and effect of this relationship. In contrast to this weak evidence, 18% of studies provided strong evidence for a link between biodiversity and the service. For example, Kaul et al. (2010) examined the carbon sequestration potential of certain tree species in India using a dynamic growth model to estimate the carbon storage potential of sal, Eucalyptus, poplar and teak forests. This study includes a detailed literature review, examines carbon storage in forest biomass, soil and wood products as well as interactions between these components. Furthermore, the authors compare model-simulated data with field data from previous studies and include an uncertainty analysis.

A3.7.2 Results –Atmospheric regulation and value linkages

The results from the review of valuation of carbon sequestration are summarized in Table A3.7.1.

Table A3.7.1: *Results from the review of valuation of carbon sequestration.*

	Averting behavior	Stated preference	Benefit transfer	Total
Ecosystem service provider				
- grassland	1		1	2
- forest	2		2	6
- marine organism		2	1	1
- habitat restoration	1		1	2
Value				
- indirect use value	4	2	5	11
Spatial scale				
- site specific			3	4
- regional	2	1	1	4
- national	2	1		2
- global			1	1
Beneficiaries				
- households				2
- firms	1	2		1
- society	3		4	7
- specific stakeholders			1	1

It can be seen from Table A3.7.1 that all eleven reviewed studies estimate the indirect use value. The ecosystem service providers are grassland (2), forests (6), marine organisms (1) and diverse habitats that are restored (2). These ecosystem service providers are beneficial to society because sequestration of carbon means either that the damage costs of climate change or the costs of averting behaviour are reduced.

It is difficult to extract information on the spatial scale of the studies because the carbon sequestration effect is related to a piece of land and an ecosystem service provider, while the benefits of carbon sequestration are global (if measured by the reduced damage costs of climate change). If the benefits are measured by saved averting costs or stated preferences, they might be estimated for any spatial scale. Therefore, in Table A3.7.1 it has been chosen to indicate if a study estimates carbon sequestration for a specific site or at regional, national or global level. Four studies have been categorised as site specific, four studies as regional, two as national and one as global.

The same problem exists in relation to indication of beneficiaries. The benefits of the avoided costs of climate change go to the global society, while the costs of averting behaviour and stated preferences are related to the national society or specific stakeholders. This is reflected in Table A3.7.1. The five studies that are based on benefit transfer all use transferred estimates of avoided damage costs of climate change for valuation. The benefits therefore go to the global society or specific stakeholders of the world. The valuation in studies that use the costs of averting behaviour reflect costs either of the national society or firms. Results of stated preference studies reflect households' willingness to pay for reduced costs of climate change.

Discussion

Three methods for the valuation of carbon storage of ecosystems (or CO_2 fixation) are used in the scientific literature:

- The global damage costs of climate change;
- The expected price of CO_2 emission permits (marginal costs of CO_2 mitigation for the quota covered sectors); and
- The marginal cost of CO_2 mitigation for the country analysed.

Global damage costs of climate changes
The avoided global damage costs of climate change are in principle the correct measure of the value of carbon sequestration. However, there are several very severe problems connected with the measure:

- It is highly uncertain how and where climate will change as a consequence of the increased amount of greenhouse gases in the atmosphere.
- It is highly uncertain which welfare-relevant consequences the expected climate change will have.
- The valuation of global welfare-relevant consequences is almost an impossible task to carry out.
- The results from partial valuation studies cannot be used in connection with valuation of climate change consequences. This is because results from partial studies cannot be aggregated. This represents a huge, not yet solved, methodological problem.

Nevertheless, several studies have been carried out to calculate the global damage costs of climate change. The results from a great part of these studies are reviewed in Tol et al. (2005). They have also been used as a basis for valuation of carbon sequestration in five of the eleven reviewed papers. In Table A3.7.1 they are categorised as benefit transfer studies.

Expected price of CO_2 emission permits
The expected price of CO_2 emission permits can be used as an indicator of the marginal cost of CO_2 mitigation in the quota covered sectors. This method has been used in three of the eleven studies reviewed. The total amount of permits reflects a politically decided target for the CO_2 reduction in these sectors and, therefore, the emission permit price depends on this target.

Therefore, use of the permit price to value the amount of carbon stored in biomass should be interpreted as a calculation of the saved costs of buying CO_2 quotas. This can be regarded as a benefit to society - scarce resources can be used for other purposes - but it has nothing to do with the value of any avoided damages of climate change.

The calculation also presumes that the sector where carbon is stored in biomass is a part of the quota covered sector. If this is the not the case it is difficult to argue for using the CO_2 permit price in valuing carbon sequestration. Instead it would be better to value carbon sequestration by calculated marginal costs of fulfilling CO_2 targets for the non-quota covered sector.

Marginal cost of CO_2 mitigation for the country analysed
Marginal costs of CO_2 mitigation have been used to value carbon sequestration in three of the eleven studies reviewed. The method presumes that a target for CO_2 emissions is decided. Otherwise it is difficult to argue that carbon sequestration will lead to saved reduction costs for the society which is the reason for valuing carbon sequestration in this way.

A3.8 Pest regulation

A3.8.1 Results – Biodiversity and pest regulation linkages

Spatial and temporal scale

The majority of the papers reported on studies in agricultural landscapes. The most frequent spatial scale was the local scale (72%) and the next frequent the sub-national scale (14%). Many of the studies were based on a single field with its surrounding habitats or landscape. Sometimes they concerned the farm level and surrounding landscape. A few relevant papers were laboratory experiments and, hence, were at a very small spatial scale.

Agricultural fields were most frequently the focal ecosystem type, mainly annual crops (wheat, rice, oil seed rape, potato, soybean), but sometimes tree crops (coffee, cacao, apples). In many studies the relationship between the cropped field and the surrounding landscape was studied. The most common type of non-crop area studied was forested areas or herbaceous field margins. Another frequently mentioned environmental parameter studied was the farming type: conventional *vs.* organic farming systems.

Almost half of the papers originated from European studies (44%), followed by Asia (14%) and the USA (14%). A few papers reported on the global scale. These were mainly reviews which searched for general patterns in several aspects of natural pest control.

The seasonal nature of pest control is reflected by the temporal scale. Almost all studies had a temporal scale of one or a few years (92%). Seasonal variety was included in a number of papers (e.g. Chaplin-Kramer & Kremen, 2012). Very few papers studied the behaviour of predators or parasitoids on a very short temporal scale in laboratory or greenhouse studies (e.g. Pérez-Lachaud et al., 2002).

Ecosystem Service Providers (ESPs)

Natural pest control is provided by species, mostly parasitoids or predators. Pest control by nematodes, bacteria or fungi were not considered in the review. There is a lot of literature on these organisms and they are applied quite a lot. However, the literature is generally on (industrial) processing of the beneficial compounds of these organisms. Therefore, the review focused on parasitoids and predators. The papers had a large range of focal ESPs (Figure A3.8.1). The functional group is most often mentioned (single: 30%, two or more: 14%), followed by the species level (single species: 20%, two or more species: 12%) and the community level (single community: 20%, two or more: 4%).

The majority of the papers focused on arthropods (80%) either as specific species, e.g. *Coccinella septempunctata* (Bianchi et al., 2007), *Notonomus gravis* (Nash et al., 2008) or *Tersilochus heterocerus* (Vinatier, 2012), or functional groups, e.g. parasitoid wasps (Boccaccio et al., 2012), generalist natural enemies (Rand & Tscharntke, 2007) or entire communities (Olson et al., 2007). Other species groups found in the papers that contribute to natural pest control were birds (Koh, 2008; Kellermann, 2007), bats (Lee & McCracken, 2005) and frogs (Xi et al., 2011). The studies with birds were related to tree crops (oil palm and coffee).

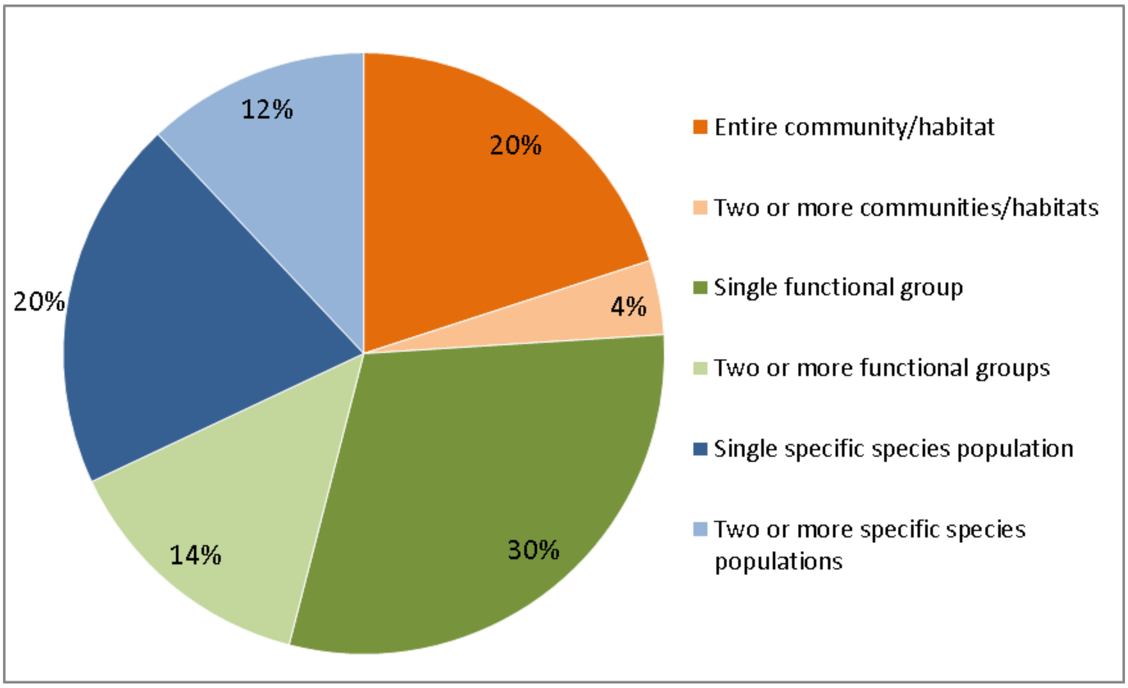

Figure A3.8.1: *Categorisation of the ecosystem service provider (ESP) for the service of natural pest control.*

Important attributes for the ESP

An overview of the important attributes for the ESP is given in Figure A3.8.2. For pest control, attributes ranging from the species level to the community level are important.

The attributes of community/habitat area and structure were found most frequently (40% and 52%, respectively). These attributes were mainly related to the non-cropped area in the surroundings of farm fields. Structure is related to landscape complexity or environmental variety surrounding the arable field, which can be expressed by e.g. the diversity of plant species (Olson & Wäckers, 2007), landscape complexity (Bianchi et al., 2006) or landscape connectivity (Boccaccio & Petacchi, 2009). Non-cropped areas provide shelter and alternative food for natural enemies. The differences in mobility and seasonal movement of pests and their enemies are important factors in explaining the importance of landscape structure.

In most studies the relation between pest control and community/habitat area or structure was positive (Figure A3.8.3). However, the relationships are complex and the number of papers with an unclear relationship to these attributes was relatively high (20% for area and 35% for structure). A few papers found a negative relationship (Olson & Wäckers, 2007; Rand et al., 2007; Bianchi et al., 2007). One explanation for this is that non-cropped areas function as a sink for natural enemies in these cases. Another explanation is that crops are richer in nutrients, resulting in more nutritious prey, which stimulates the population of predators (*Coccinella septempunctata*). This effect is lacking, or less strong, in complex systems due to the lower amount of annual crops. Different functional groups (e.g. specialists *vs.* generalists) seem to respond differently to these attributes, but this remains a challenge for further study.

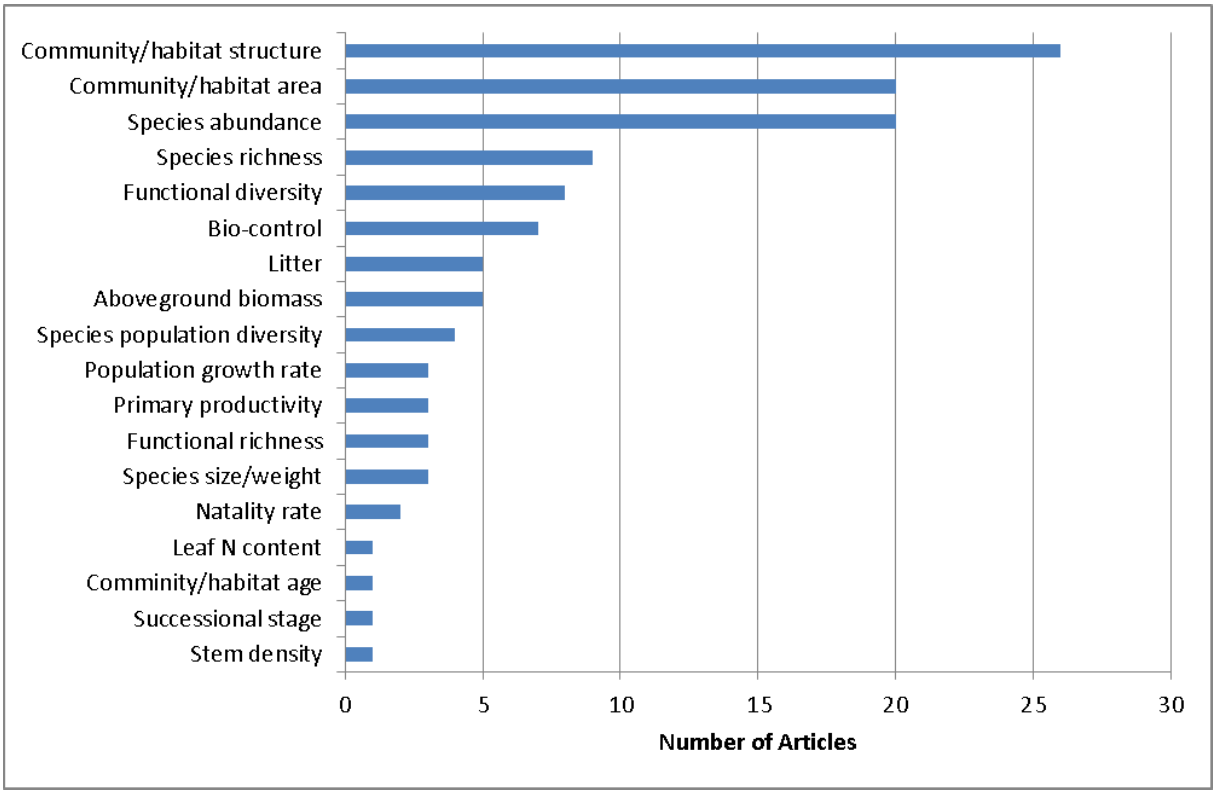

Figure A3.8.2: *Categorisation of the ESP attributes for the service of natural pest control. Note: some studies used multiple attributes.*

Functional groups were often the focal ESP in the papers. The attributes directly related to functional groups are, however, not very often used as attributes to explain the functioning of pest control. Species richness was cited as an important attribute to explain pest control in nine papers. The authors found a positive relationship in about half of these papers, whilst the relationship was unclear in the other papers. A number of the papers for arthropods concluded that probably only the abundance of a few species is relevant for pest control (e.g. Macfayden et al., 2012).

The category 'other' scored quite high for this ecosystem service. A closer look at this category shows that different farming system-related factors influence the effectiveness of ESPs. In particular, seven papers discussed the difference between organic and conventional farming. In most cases natural pest control functions better in organic farming than in conventional farming (e.g. Öberg et al., 2007). This can be explained by the higher organic matter in the soil (e.g. Drinkwater et al., 1995) or the absence of chemical insecticides (Geiger et al., 2010; Kraus et al., 2011). In some studies the effect of organic farming *vs.* conventional farming remains unclear (Schmidt et al., 2003), or organic scores less than conventional because of the higher tillage frequency which disturbs ground dwelling predators (mainly carabids) (Nash et al., 2008; Legrand et al., 2011). A positive effect of agri-environmental schemes was mentioned by Geiger et al. (2010).

Another factor mentioned in a number of papers is the availability of alternative food sources in the arable fields or close surroundings. The presence of flowering plants providing nectar and pollen stimulates the activity of parasitoid wasps, which obviously stimulates pest control.

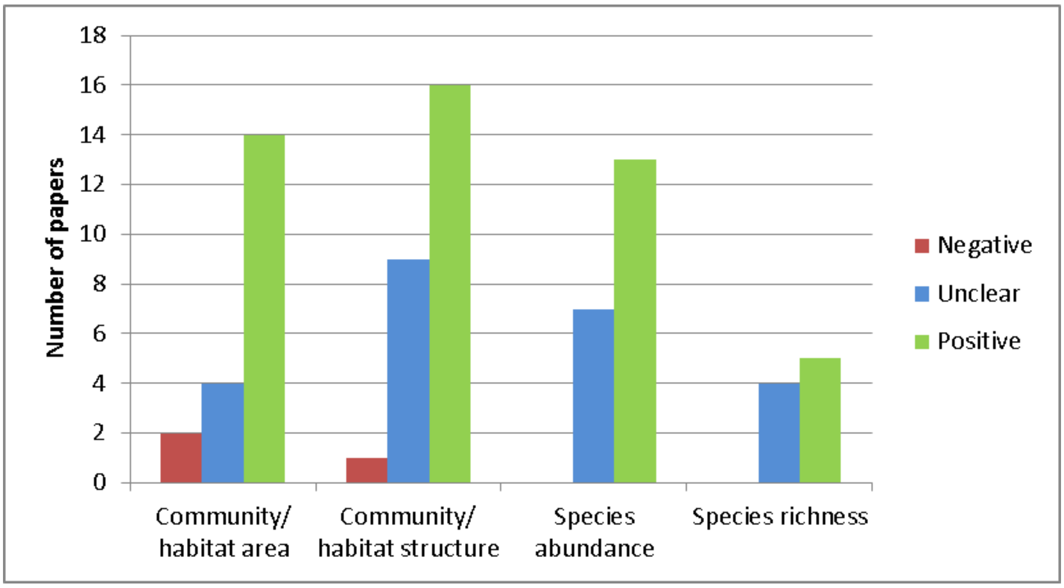

Figure A3.8.3: *Direction of the relationship between the ESP attributes and ecosystem service for natural pest control. Only four attributes frequently mentioned in the papers are shown.*

Discussion

The literature search

The literature search focused on those aspects of pest control that relate to biodiversity in terms of species that occur in the wild. A particular focus was on open field crop systems. Some papers that modelled the relationship between biodiversity and natural pest control or that were based on laboratory or greenhouse experiments were also included. The literature on pest control and biodiversity is, however, much broader than our selection. Pest control by (entomofagous) nematodes, bacteria or fungi can also be effective and is quite often applied. A considerable amount of the literature deals with pest control based on these functional groups.

The papers that were relevant for this study in general focus on the ESP, that is, the species providing pest control. Such papers were relatively easy to find. Numerous papers about the ecology of natural enemies mention the importance of the species, functional group or community/habitat for pest control in their conclusions, but without supporting data. These were eliminated from the review. Studies which aimed to keep pest pressure and crop damage below an (economic) threshold without the use of chemicals were rare.

Abiotic factors

Abiotic factors were mentioned in a few studies (20%). In many cases the influence on the natural enemies is indirect. For example, the pest species coffee berry borers are more abundant in dry and warm conditions, and it is expected that natural pest control by birds is more effective (Kellerman et al., 2007). Thermals influence the spatial and temporal distribution and availability of pest species, which are prey for bats. Bats follow those thermal movements (Lee & McCracken, 2005). A direct relationship between temperature and natural enemy population growth was found by Xi (2012).

Nutrient availability is another abiotic factor mentioned in a number of studies (e.g. Wäckers et al., 2008). One study showed a positive effect of fertilization of crops with pest control: *Coccinella septempuntuata* more effectively suppressed aphids in fertilized fields, probably because the aphids had good nutritious quality in these conditions. The absence of pesticides was crucial in this case (Bianchi et al., 2007).

The negative impact of biodiversity on pest regulation
A number of studies explicitly mention the negative effect of biodiversity related attributes on pest control. In more species rich systems, alternative prey can be used as food for predators or parasitoids, thus decreasing the suppression of pest species. Oelberman & Scheu (2009) observed this effect for spiders at a high spider density. Nevertheless, many natural enemies do need alternative prey in order to survive periods with low pest density.

Another mechanism where biodiversity was shown to have a negative effect is described by Mody et al. (2011), where ants defend aphids against parasitism. Finally, natural enemies can compete with each other, or even be prey themselves for other predators, which again decreases the effectiveness of pest control (e.g. Xu et al., 2011).

Strength of evidence
The strength of evidence of the majority of the papers (60%) was classified as average. One of the criteria used for the classification of strength of evidence was on which part of the multitrophic interactions in the food webs of natural enemies, pests and crops was the evidence the strongest. The majority of the studies classified as average focused on natural enemies and pest species, but not, or only slightly, on crop damage (e.g. Krauss et al 2011). A smaller number of papers (24%) provided empirical evidence on the relation of biodiversity with crop yield (e.g. Geiger et al., 2010; Kellerman et al., 2007), which were classified as strong evidence. Other papers were classified as weak evidence (total 16%), for example, when the focus was merely on the natural enemy, but no data on the real contribution of pest control was provided (e.g. Nash et al., 2008), when the behaviour of natural enemies was modelled without information of the functioning of natural pest control (e.g. Drapela et al., 2008), or when existing knowledge of the natural pest control system was applied in a tool or design of landscapes (e.g. Venatier et al., 2012). Summarising the review, the evidence is average. This indicates that some evidence on the relation between biodiversity and pest regulation is available, but there is still quite a lot of discussion on the functional relationships between biodiversity aspects and pest control, which is also reflected by Figure A3.8.3. Another conclusion of the evaluation of the strength of evidence is that many studies lack information on the final effect on crop yield, and whether natural pest control effectively can replace chemical pest control.

A3.8.2 Results – Pest regulation and value linkages

The review is based on a total of 14 papers. Common for all studies is that the pest control ecosystem service is valued by the use of avoided costs. The avoided costs are manifested in the form of reduced costs for pesticides, avoided yield loss and/or avoided crop damage. In most of the studies, where reduced use of pesticides are included as a benefit of natural pest control, the reduced pesticide costs used in the valuation only include the costs for buying pesticides and application costs. Hence, it is only in a few of the studies that the external costs of pesticide use in relation to health and the environment also are included; common for the studies including external costs of pesticide use is, however, that they rely on values obtained from other studies. In terms of the type of values assessed these are all classified as indirect use values, although it can be argued that the values assessed in the few studies including the environmental costs of pesticide use also may include some non-use values. It is not very surprising that the studies focus on assessing the indirect use values provided by natural pest control, since many of these values translate into marketable benefits which are fairly easy to put a price on. A consequence of the focus on indirect use values related to market goods is that the values assessed in the studies primarily reflect values that accrue to private stakeholders, in this case farmers.

In terms of the pests and crops studied there are some repetitions; three of the studies focus on natural enemies of aphids in soybean, two deal with birds' control of the coffee berry borer in coffee and three deal with bats controlling pests in cotton. Four of the remaining studies also focus on specific pests and/or crops, while the remaining two studies adopt a more overall perspective and attempt to assess the aggregate value of pest control as an ecosystem service on a national and a forest reserve level, respectively.

In all the reviewed studies it is concluded that ecosystems provide, or have the potential to provide, services of significant value in relation to pest control. The value depends on many factors such as management practices (e.g. crop choice, degree of pesticide use), input and output prices, and the characteristics of the surrounding environment. Hence, a common message from the studies seems to be that pest control represents an important - and potentially valuable, also from a private economic perspective - ecosystem service. However, it is also clear from the studies that pest control is a service whose provision is severely limited by conventional farming practices. Realising the full potential benefits of natural pest control therefore requires that the service is taken explicitly into account by farmers/planners.

A3.9 Pollination

A3.9.1 Results – Biodiversity and pollination service linkage

Spatial and temporal scale

The majority of articles included in the review (62%) considered pollination at the sub-national spatial scale, the studies often involving sampling replications and/or comparisons across the wider landscape of a region. Landscapes include those that are primarily agricultural (e.g. Winfree & Kremen, 2009) or more natural (e.g. Potts et al., 2006; Blanche & Cunningham, 2005) or a mixture of the two (e.g. Albrecht et al., 2007), with examples from Europe, as well as other continents. About 16% of studies were undertaken at the local scale, including experimental work within a small area of natural habitat (Munoz & Cavieres, 2008) or within standardised field plots (e.g. Fontaine et al., 2006; Jauker et al., 2012). At the other extreme, studies at the national to continental scales (14%) provided comparative information from more than one country (mainly within Europe, e.g. Biesmeijer et al., 2006) or summarised information for a single country (e.g. Ashworth et al., 2009). The global scale studies (6%) provided overviews of particular aspects of pollination services in a search for general patterns (e.g. Ricketts et al., 2008).

The seasonal nature of many plant-pollinator systems is reflected in the annual/seasonal time scale of most (78%) of the studies. A few studies (14%) adopted a historical or otherwise longer term perspective covering decadal time scales (e.g. Bommarco et al., 2012; Breeze et al., 2011), while a single study emphasised the very short (daily) time scale of mass flowering within a season (Diekotter et al., 2010).

Ecosystem Service Providers (ESP)

By definition, animal species that pollinate plants form a functional group of organisms. Thus, most of the studies in the review (71%) involved this single functional group as the ESP (Figure A3.9.1). An additional 6% of studies differentiated between particular details of this general function and defined further functional groups (e.g. Hoehn et al., 2008). Some other studies (16%) took a more species-oriented approach and considered pollination service provision by populations of species from particular taxonomic groups, such as bumble bees (Bommarco et al., 2012), or a particular beetle (Blanche & Cunningham, 2005) or moth (Martins & Johnson, 2009) species. A small number of studies (6%) considered only a single species as the service provider and referred to honey bees, *Apis mellifera* (Breeze et al., 2011; Geerts & Pauw, 2011; Vergara & Badano, 2009). A single study by Vamosi et al. (2006) examined pollination service provision by the communities of organisms found in biodiversity hotspots worldwide.

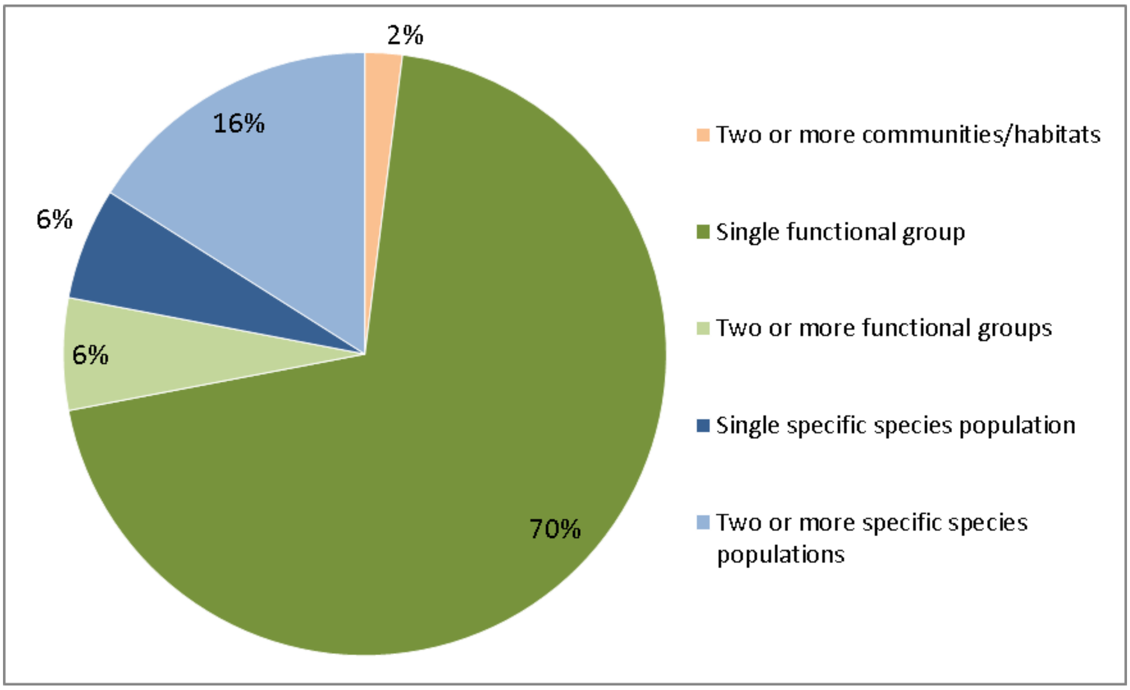

Figure A3.9.1: *Categorisation of ecosystem service providers (ESP) for the service of pollination.*

Important attributes for the ESP

A variety of different attributes were identified in the studies included in the review as having an influence on the provision of pollination services (Figure A3.9.2). However, a complication arises within the pollination literature because species and functional groups are often treated rather synonymously. As mentioned in the description of the ESPs, those populations of flower-visiting animals that deliver pollination services are a single functional group. But in addition to this, the vast majority of studies clearly identified the pollinator species concerned, as well as the plants they pollinate. Thus, pollination may be impacted by species and functional group attributes simultaneously. This is reflected in the description of the attribute results and the associated Figure A3.9.2.

In considering the species-associated attributes, a further confounding factor arises through the inclusion or omission of managed and/or wild honey bees, *Apis mellifera*, in the studies. All studies in the review referred to, and identified, the pollinator species type, but this needs to be divided into those studies that included honey bees in the system (32 studies or 63%) and those in which honey bees were excluded (the remaining 19 studies or 37%). Both of these groups of studies included examples using crop plants and/or wild plants in agricultural landscapes, near natural habitats and mixed natural/crop plant landscape mosaics. Amongst those studies in which honey bees were included, some treated this managed species simply as an element of the system being investigated, for example, Holzschuh et al. (2011) who examined cowslip (*Primula veris*) pollination as a phytomer species within oilseed rape crop landscapes. Other studies compared the contribution of honey bees to pollination with contributions from other pollinator species. For example, Potts et al. (2006) found that honey bees, though one of the two most common bee species encountered in Greek olive grove vegetation, were not the primary pollinators of wild plants in this type of habitat. Three studies focussed on honey bees alone. Breeze et al. (2011) reported on the falling capability of honey bees to meet crop pollination demands in the UK. In a very different and somewhat more natural setting, Geerts & Pauw (2011) showed that farming involving native honey bees in South Africa can have negative effects on nectar-feeding bird biodiversity in fynbos vegetation. The third honey-bee only study, again in South Africa, referred to improved honey yield in relation to closer proximity to forest wild flower resources (Sande et al., 2009).

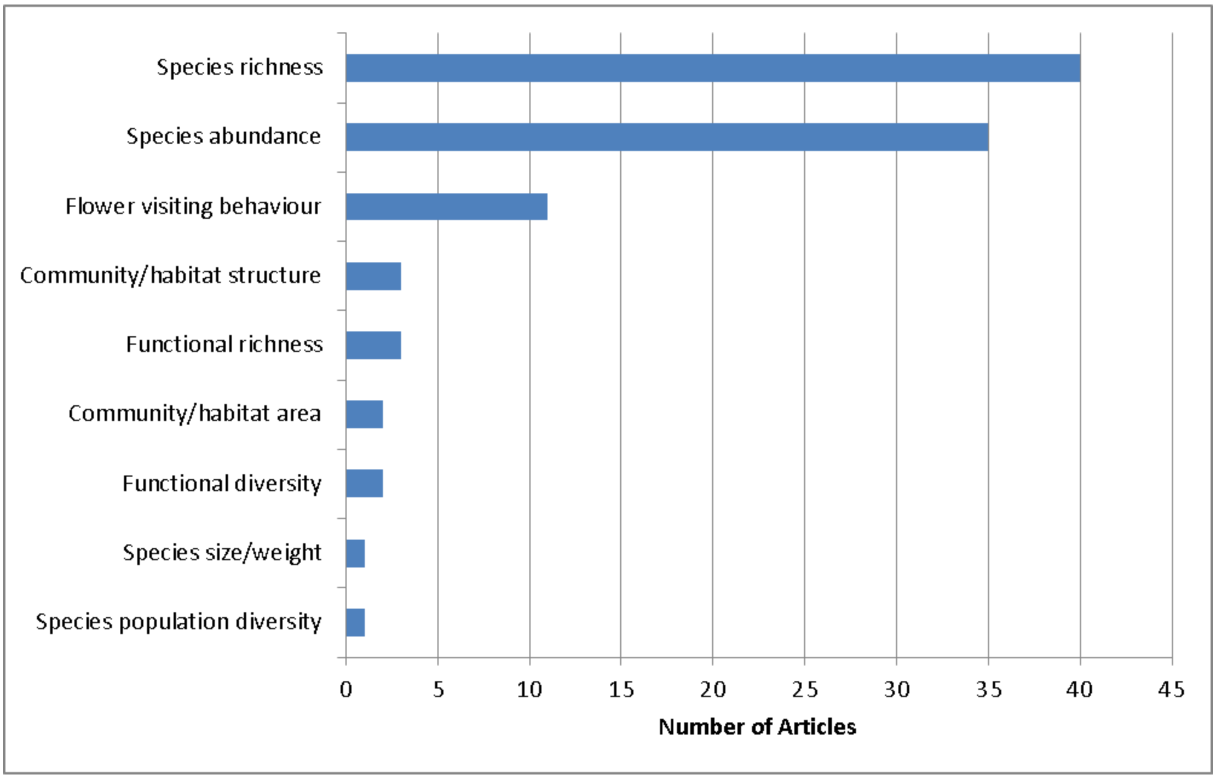

Figure A3.9.2: *Categorisation of the ESP attributes for the service of pollination. Note: some studies used multiple attributes.*

The 37% of studies in the review that did not include honey bees reflect particular research interests, mainly focusing on the impact of the variety and abundance of wild pollinator species on pollination services in different situations involving crop and wild plant pollination systems.

Species richness (the number of species) and species abundance (the numbers of individuals of each species) are the two major components of classical measures of biodiversity. Both parameters are also shown to be highly relevant in the context of the provision of pollination services. Species abundance was a relevant attribute in 70% of the studies reviewed. These often involved investigations with a main focus on measurement of pollination success/efficiency of crop or wild plants, without or including honey bees. Studies that identified species richness as a relevant attribute (80%) overlapped to some extent with those involving abundance, but tended to be more oriented to the importance of maintaining a palette of different pollinator species for providing services to target crops or wild plants.

Turning to the functional group attributes and bearing in mind the synonymy with species attributes, most studies (76%) indicated relevance of the pollinator functional group of animals. The remaining 24% of studies were simply those that involved a strongly species/taxonomic oriented approach. An increase in the diversity of pollinator functional groups was demonstrated to be beneficial to pollination service provision, as addressed in a small number of studies (4%), such as that of Biesmeijer et al. (2006) who concluded that functional diversity of pollinators is positively related to maintenance of diversity of wild plant communities in Britain and the Netherlands. Furthermore, Fontaine et al. (2006) showed that experimentally increasing the functional diversity of both plants and pollinators led to the recruitment of more diverse plant communities and, thus, the functional diversity of pollination networks may be critical to ecosystem sustainability. Hoehn et al. (2008) showed that both diversity and richness of functional groups of pollinators defined on the basis of species functional (behavioural) traits enhance pollination efficiency and crop yield of pumpkin.

Community/habitat type was identified in 8% of studies as being of relevance to pollination, as well as the other related attributes of structure (6%) and habitat area (4%). For example, Brittain et al. (2010) found that the proportion of uncultivated land surrounding Italian vineyards negatively affected fruit set and seed weight of potted Petunia plants in both organic and conventional vine fields, whilst Albrecht et al. (2007) compared hay meadows to intensively managed meadows in Switzerland. Furthermore, food webs of flower and pollinator species in sunflower farms were investigated by Cavalheiro et al. (2011) who found that more complex food webs which include maintenance of a diversity of flowers help to maintain pollinator communities and increase sunflower pollination. Cavalheiro et al. (2010) also investigated food webs of flower and pollinator species in mango farms, where in addition it was demonstrated that larger farmed areas increase the distance to natural habitats and therefore reduce pollination services.

A number of studies in the review (22%) addressed specific behavioural/morphological parameters of pollinators, in particular plant-pollinator systems, and were able to identify relevant flower-visiting behavioural traits as influencing pollination services. For example, Hoehn et al. (2008) measured three behavioural traits of pumpkin-pollinating bee species - flower height preference, daily time of flower visitation and within-flower behaviour - as related to body size, and linked these to seed set of pumpkins within a surrounding land use intensity gradient. Spatial and temporal complementarity in the traits measured were found to enhance pumpkin pollination efficiency and crop yield. In a comparative investigation of wild bee species and honey bees pollinating sunflowers, Greenleaf & Kremen (2006b) found, perhaps rather surprisingly, that interference competitive interactions of a honey bee with a wild bee on a male flower increases the probability that the honey bee will move to a female flower, and hence promote sunflower pollination.

A single paper in the review was concerned with honey yield and quality provided by managed honey bees in South Africa (Sande et al., 2009) and has been placed in the “other” attribute category. Honey yields were found to be substantially higher close to the forest where there are more diverse flower resources. The authors recommend maintenance of high apiflora species diversity and abundance, which happens naturally inside the forest. Thus managed honey bees and honey yield provide an alternative argument for conserving natural forest fragments and their floral diversity.

Discussion

The literature search

Pollination is one of the most widely used examples of the importance of ecosystem service provision for human well-being, to the extent that pollination has now achieved something of a “flagship” status as a regulating service. Such popularity has undoubtedly had a certain influence on how research on the subject has been conducted and reported over the last decade, but it also means that there is already a considerable body of literature that addresses the services aspect of plant-pollinator systems. This permitted the decision here to include ecosystem services as a search term, rather than use a wider search as has been required for other services. Supplementary scanning of recently published and in-press papers and their reference lists ensured an up to date coverage. Such extra scanning of references provided 10 relevant studies not picked up by the computer searches.

Abiotic factors

Only a very few studies (6%) mentioned any abiotic factors as relevant to pollination. Klein et al. (2003) reported that in highland coffee agroforestry systems in Suvwalesi the diversity of solitary bees increased with light intensity and this diversity, not abundance, explained variation in coffee fruit set. Similarly, Munyuli (2012) reported on the importance of light intensity and that it is positively related to bee species richness in coffee plantations in Uganda. This author also found the optimal degree of shade cover for bee foraging to be 10-50%. A study by Dauber et al. (2010) implied some influence of abiotic factors in that it was designed to cover four climatic zones within Europe: Atlantic, Boreal, Continental and Mediterranean.

The negative impact of biodiversity on pollination
A number of studies (10%) define some form of antagonising or negative impact of species in pollination systems, mostly involving honey bees. It is clear that this important pollinator species can also be an ecosystem service antagoniser of pollination or other services. Allsopp et al. (2008), in a survey of deciduous fruit pollination in South Africa, concluded that managed honey bees can negatively affect wild pollinator species diversity. The results of a study by Shavit et al. (2009) provided partial evidence for behavioural competition between honeybees and native bees in Israel. For this reason, these authors recommend prohibiting introduction of beehives to all nature reserves in Israel. This may help pollination of common and rare native plants as well as crops. Breeze et al. (2011) cast doubt on long held beliefs that honeybees provide the majority of pollination services to UK agriculture, and highlight the importance of measures aimed at maintaining both wild and managed species.

A different form of negative impact is reported by Munoz & Cavieres (2008), who show that in the alpine vegetation of the Chilean Andes high densities of the invasive plant *Taraxacum officinale* (dandelion) have negative effects on pollination and reproductive output of the two related native plant species.

Strength of evidence
A large proportion (52%) of the studies are judged to provide very strong or strong evidence for a relation between biodiversity and pollination services, with a further 26% providing average, but valid evidence. This significant skew towards stronger evidence provided by the studies in the review indicates that the positive relation between biodiversity and pollination services is real, and should not be ignored. Certainly the strength of the evidence is also a reflection of the intense, high quality, clearly designed research that provides the information, but this only strengthens the evidence base. The 22% or so of studies that show only weak or very weak evidence do so mainly because of their focus or approach rather than because of weak science. Thus, studies that are based on a modelling approach (e.g. Lonsdorf et al., 2009), or only provide indirect evidence of biodiversity-pollination services relations (e.g. Jauker et al., 2011) or discuss pollinator diversity, but not pollination success (e.g. Tscheulin et al., 2011) cannot be said to provide strong evidence for the biodiversity-pollination services relation.

A3.9.2 Results – Pollination service and value linkage

The results from the review of valuation of pollination service are summarised in Table A3.9.1. As can be seen in Table A3.9.1, all reviewed studies focus on the services provided by either wild or managed pollinators. The production function approach is used in more than half of the studies, and the second most used method is biophysical ranking; other methods used are the market price approach (2), avoided costs (1), benefit transfer (4) and 'other' (3). In terms of the types of values assessed, the main emphasis is on indirect use values (assessed in 30 studies). Other types of values assessed are option value (7) and 'other' (6). In seventeen of the studies, values are assessed on a local scale, while in the remaining studies they are assessed on a regional (8), national (8) and global (8) scale. Firms are listed as the beneficiaries in twenty three of the studies; in the remaining studies the beneficiaries are society (7), group of stakeholders (1) and 'other' (10).

Table A3.9.1: *Results from the review of valuation of pollination service.*

	Market price	Production function	Avoided costs	Benefit transfer and reviews	Biophysical ranking	Other	Total
Ecosystem service provider - wild and/or managed pollinators	2	23	1	4	8	3	41
Value							
- indirect use value	1	18			6		30
- option		4	1	4		3	7
- other	1	3			2		6
Spatial scale							
- local		7		2	7		17
- regional		5		1		1	8
- national	2	5	1			2	8
- global		6		1	1		8
Beneficiaries							
- firms	1	18		2		1	23
- communities/society		3	1	1	3		7
- group of stakeholders	1						1
- other		2		1	5	2	10

Discussion

The selected papers were published between 1998 and 2012 (Figure A2.9.3), although no papers were selected between 1999 and 2001. The majority of the papers were published in 2006 (21%), followed by 2009 (16%).

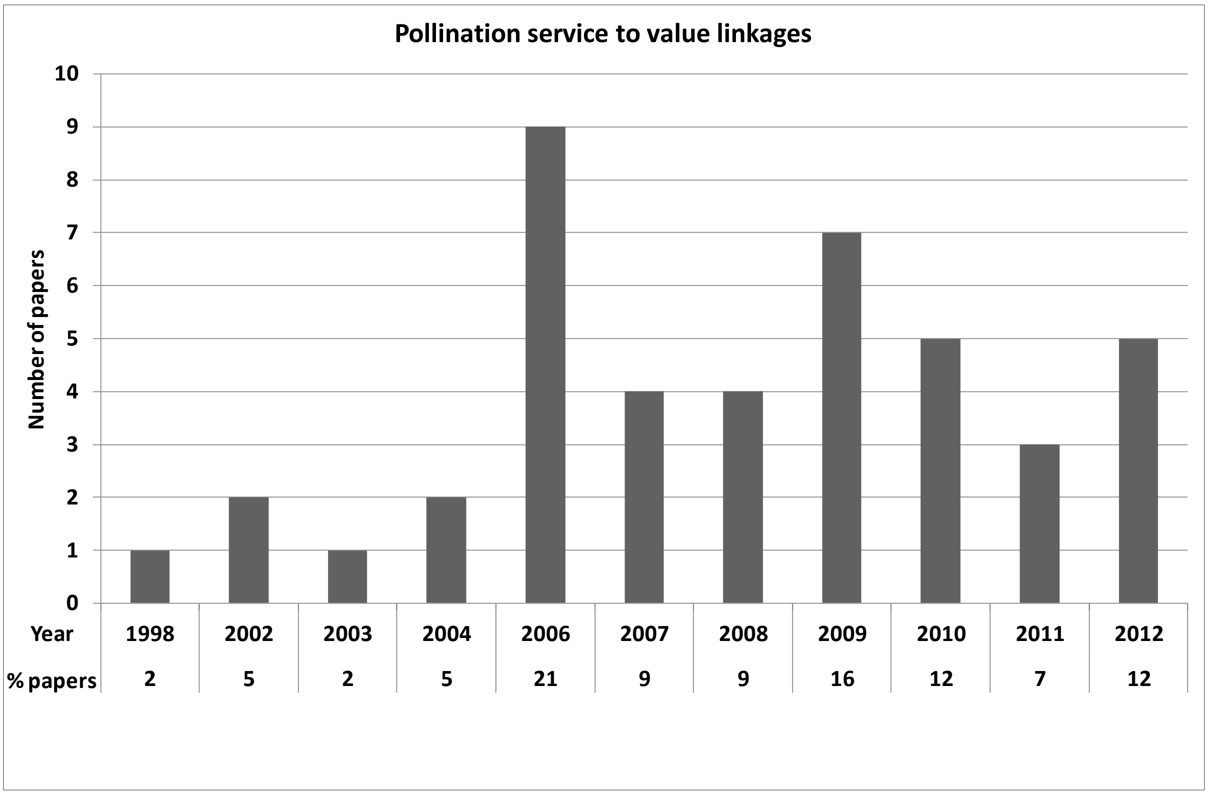

Figure A3.9.3: *Temporal distribution of the studies reviewed for the linkages between pollination service and value.*

The types of values assessed by the reviewed papers fall within three main categories; some studies measured more than one type of value. As mentioned above, the vast majority of the studies measured “indirect use”.

There was a noticeable degree of variation between the valuation approaches adopted within the papers, and about 6% of the papers included more than one method. The most common approaches were “replacement value”, relations based on empirical observations, crop production estimates and crop dependency on insect pollination.

With the exception of “households”, all the other categories of beneficiaries identified by the proposed classification were represented. In almost half of the studies, more than one beneficiary group could be identified.

Due to the diverse nature of this service, a quantitative summary is not appropriate. For a qualitative summary, all the studies reviewed made at least some claim on how their evaluation could be used within a policy context, in particular for conservation management measures. In some cases there were specific references to existing policies (for instance support programs), or attempts to discuss the cost of implementing certain management measures.

A3.10 Recreation activities

A3.10.1 Results – Biodiversity and recreation service linkages

Type and location of recreational activities

Recreational activities included wildlife viewing (58%), recreational fishing (20%), hunting (18%) and others (4%). Recreational activities related to biodiversity were found in five different continents with most of the studies conducted in Europe (Figure A3.10.1). Recreational activities linked to these studies included wildlife viewing such as bird watching, hunting of wild animals and swimming with dolphins. In most of the studies carried out in Australasia, the marine ecosystem is the main provider of recreational services, while for those in North America and Africa, terrestrial ecosystems were the main ecosystems providing the services (e.g. wild life viewing in a national park or hunting of wild animals such as deer). Europe and Asia had an almost equal mix of recreational activities carried out in both marine and terrestrial systems.

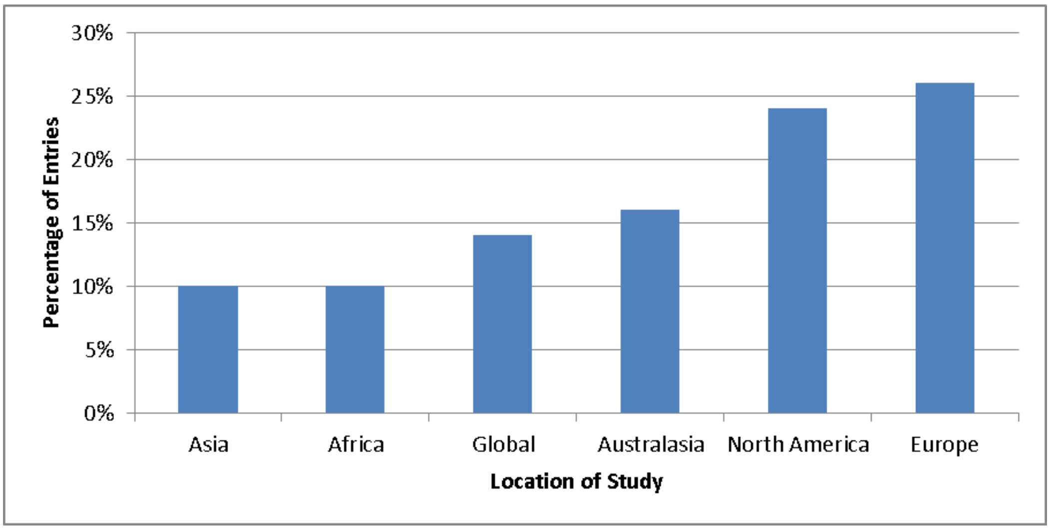

Figure A3.10.1: *Percentage of studies in five different continents and globally with evidence of recreational use of biodiversity.*

Spatial and temporal scale

Most studies were focused on the local scale (48%), while 20% were on the national scale and 20% on the sub-national scale. As recreation is more of a local ecosystem service, but with beneficiaries on a global scale, it is common to have the study at a particular study site where visitors are asked about their experience and preference or tour operators are asked questions about preference of the different types of tours. For example, in order for Fennell & Weaver (1997) to understand the potential of ecotourism in vacation farms, they sampled the vacation farm tour operators in Saskatchewan in Canada. In another local study, Munn et al. (2011) looked at hunter's willingness to pay for hunting leases in the state of Mississippi in the United States. National level studies were mostly drawn from national statistics or focused on a particular group of tourists (e.g. anglers or hunters). For example, Lewin et al. (2006) evaluated the catch orientation of German anglers focusing on their motivation and satisfaction.

Most recreational studies are carried out through questionnaires from daily visits of tourists, tour operators or postal surveys. Studies aimed at tourists on a recreational trip were mostly done on a daily basis while those aimed at tour operators were seasonal, where records could be used to answer questions from one or more seasons. More annual/seasonal studies (50%) were found, therefore, than daily studies (48%). One study by Broch et al. (2012) which looked at farmer's willingness to participate in reforestation contracts that deliver ecosystem services, such as recreational hunting, had a temporal scale of about 10 years. The authors modelled farmers' preferences and biodiversity aspects, such as species richness, over a long period.

Ecosystem service provider (ESP)

Biodiversity elements that were important for the service of recreation included fauna and flora species, although fauna species such as birds (e.g. Kerlinger et al., 1994; Sari et al., 2011), fish (e.g. Butler et al., 2009) and mammals (e.g. Lindsey et al., 2012) were the most common in the review (Figure A3.10.2). According to Fennel & Weaver (1997), birds and mammals are the most important species for wildlife viewing in Saskatchewan, Canada. Bird species listed for recreation included, but were not limited to, waterfowl, robins, eagles, humming birds, grouse and the collared dove. Mammal species included ungulates (e.g. deer), carnivores (e.g. lion and coyote) and rodents (e.g. beaver). Fish species included rainbow trout, brown trout, brook trout, lake trout (*Salvelinus narnaycush*), perch and salmon (*Oncorhynchus tshawytscha*). A few reptile species included frogs (*Dendrobates tinctorius*) and gecko (*Phelsuma gecko*). Other organisms listed were insects such as dragonflies and some plant species.

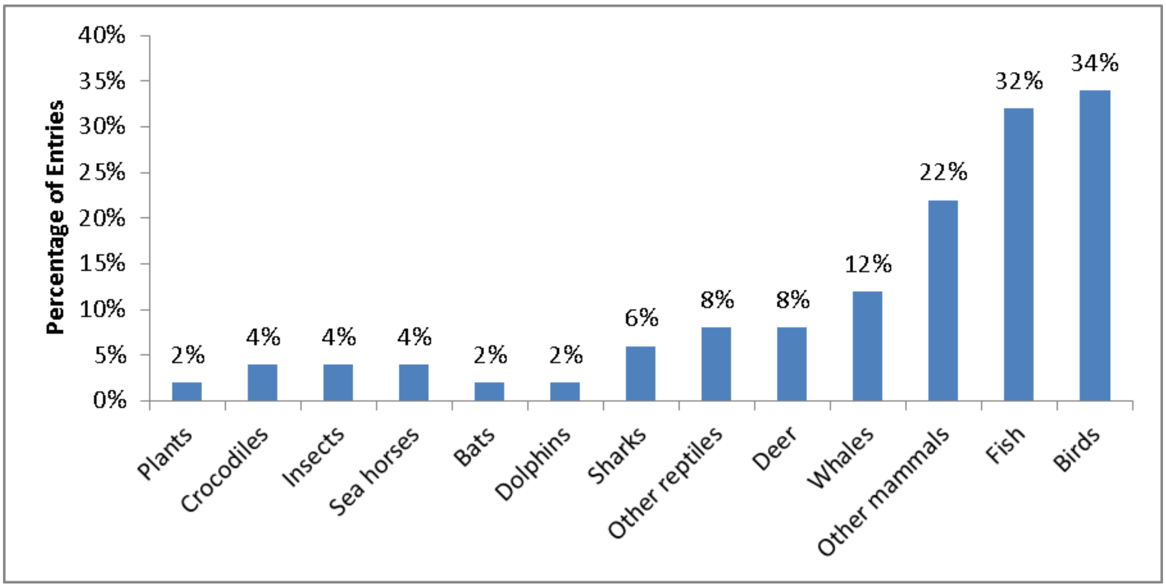

Figure A3.10.2: *Percentage of studies that recorded different biodiversity features as important for recreation. Note that some studies considered more than one species type.*

In many cases, the ecosystem service of recreation was provided by a single species in one location (Figure A3.10.3). In such a situation, tourists will visit a particular location to see a particular species (e.g. whales, dolphins or penguins). For example, Ratz & Thompson (1999) reported on the impact of tourism on penguins at the Yellow-eyed Penguin Conservation Reserve, Dunedin, New Zealand. At this site, visitors view breeding yellow-eyed penguins (*Megadyptes antipodes)* at close range from hides. In such places, species abundance is important to ensure visitors satisfaction of viewing the species. Nevertheless, tourists could also encounter other species which are not the main reason for visiting the location. A positive relationship was reported between species abundance and recreational value in 34% of all studies, and it was found that of these the focus was on a single species population.

However, in most of the studies (66%), the ecosystem service was provided by more than one species population. About a third of all studies (30%) reported a positive relationship between species richness and the recreational service. For example, visitors who go to the Amboseli National Park in Kenya are interested in a variety of species (Okello et al., 2008). Although the park management usually markets the "big five" (elephants (*Loxodonta africana*), buffalo (*Syncerus cafer*), rhinos (*Ceratotherium simum* and *Diceros bicornis*), lions (*Panthera leo*) and leopards (*Panthera pardus*)), the authors showed that visitors were interested in species other than those being marketed. Lindsey et al. (2007) also found among tourists in South Africa that high mammal diversity is the most important feature of a protected area, followed by the presence of large predators. It would appear this relationship does not only end with recreational activities directly linked to biodiversity, but extends to other activities. For example, Ruiz-Frau et al. (2012) found a positive correlation between kayaking route popularity and the presence of wildlife on the popularity of the route (R^2=0.6, p<0.001). The authors also found that the level of marine biodiversity at the dive location is one of the most important factors in determining diving location for scuba divers.

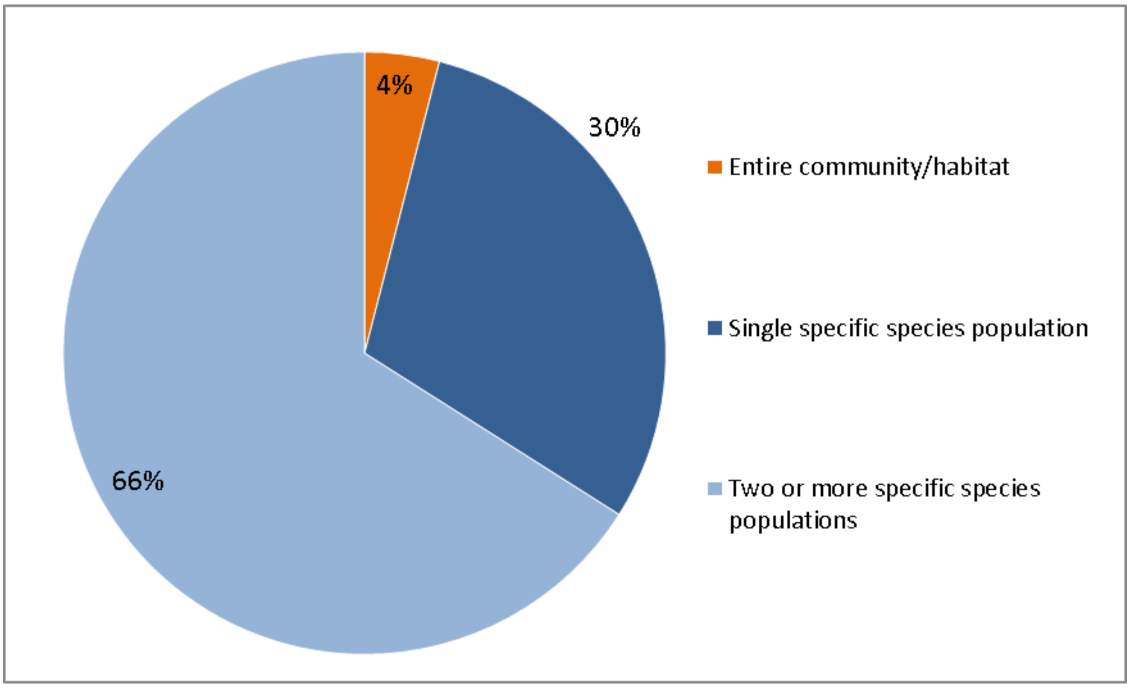

Figure A3.10.3: *Categorisation of ecosystem service providers (ESP) for the service of recreation.*

Important attributes for the ESP

Recreational activities in general do not focus on particular traits of species. However, some traits can be important depending on the activity being carried out (Figure A3.10.4). Species abundance for example is particularly important for the service of species based recreation, and was mentioned in 72% of articles. This is mainly as wildlife viewing and recreation requires species to be present. For example, marine wildlife tourists at the Stingray City Sandbar in Grand Cayman prefer to see higher number of rays (Semeniuk et al., 2009). Similarly, visitors to the Amboseli National Park in Kenya want to see a lot of species (Okello et al., 2008). The viewing of rare birds (Booth et al., 2011; Schanzel & McIntosh, 2000), whales (Hoyt, 2005) and butterflies (Lemelin, 2009), for example, all require these species to be present.

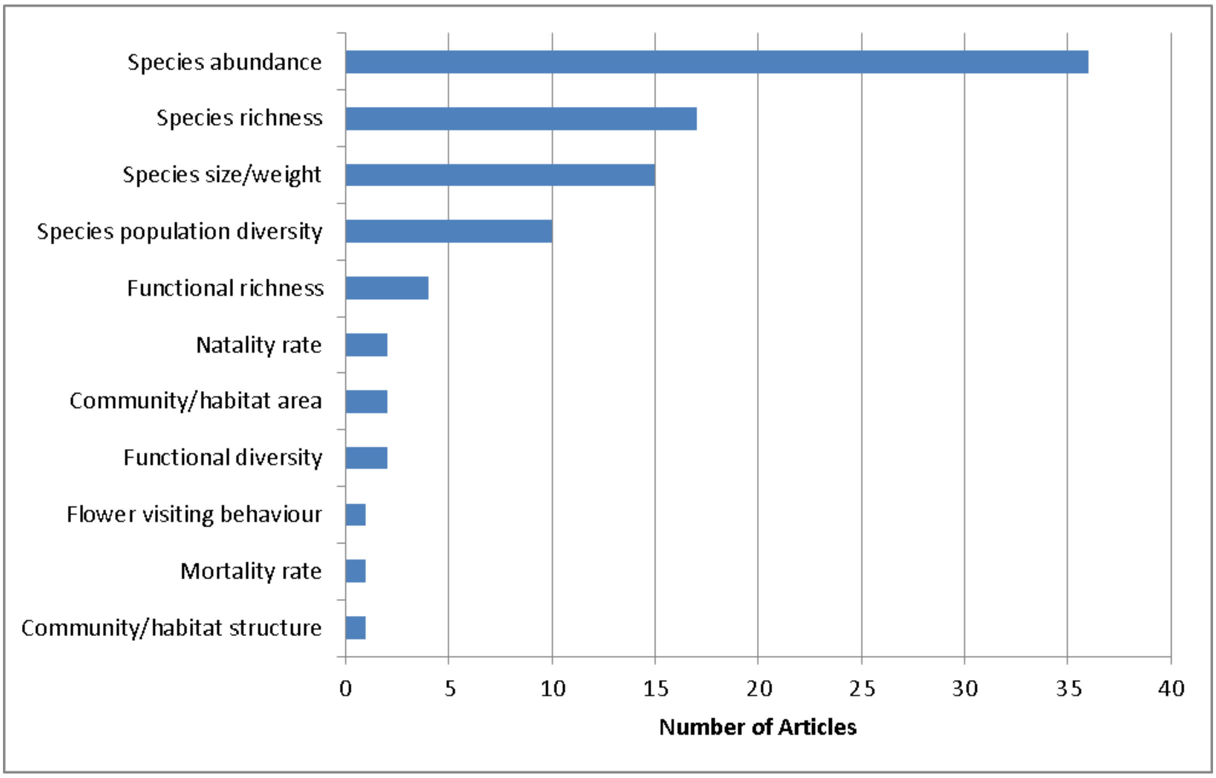

Figure A3.10.4: *Categorisation of the ESP attributes for the service of recreation. Note: some studies used multiple attributes.*

It is also common knowledge that species traits determine species richness (Seddon et al., 2008). The importance of species richness for recreation was specified in 34% of studies. Visitors love to see a variety of species, as do bird watchers (Lee et al., 2010). In addition to richness, 40% of all studies in the review also specified the importance of species size in recreation. Some anglers prefer larger fish and for many recreational hunters, the size and quality of trophy matters. For example, Gallagher & Hammerschlag (2011) mentioned in their study that the large size of the shark is an important parameter for the profitability of the ecotourism industry according to the authors own experience. In another study, Lindsey et al. (2007) evaluated viewing preferences of tourists in five protected areas in South Africa and found that large carnivores were more popular with first time visitors. One study in the review evaluated the importance of cryptic species for recreation (Uyarra & Côté, 2007). Although not particularly mentioned in the literature review, many species traits could indirectly be important in recreation. Species traits could influence species distribution in many ways. According to Suding et al. (2003), tolerance and colonization abilities appeared to be important translators that linked organismal traits to abundance patterns. Certain species flourish in certain environments because of their ability to adapt and colonize these areas which become important places where tourists could come in close contact with the species. An example is the penguin population in the western Cape of South Africa.

Finally, species population diversity was mentioned in 20% of articles as being important for recreation. Dragonfly enthusiasts for example enjoy recognizing many different types of Odonata (Lemelin, 2007), and according to a survey of divers 40% said that fish diversity was the single most important factor influencing satisfaction (Rudd & Tupper, 2002).

Discussion

The literature search

As mentioned above conducting a literature search with the search terms tourism and recreation coupled with biodiversity resulted in a large number of studies because there are many studies that look at the effects of tourism or recreation on biodiversity. This made finding studies that looked at the relationship between biodiversity and recreation very difficult. Many studies may also have been picked up that were not related to biodiversity because there are many other recreational activities such as biking that are not linked to biodiversity. However, including specific recreational terms such as "hunting" resulted in fewer studies more focused on recreational activities linked to biodiversity. Snowballing helped also in finding other studies on the subject. Nevertheless, finding 50 relevant papers on the subject was difficult. Very few studies that had looked specifically at the relationship between recreation and biodiversity were found. Rather, most of the studies recorded acknowledged the use of biodiversity in recreation.

Abiotic and other factors

In most studies, abiotic factors were not recorded as important in the service of recreation, but these factors definitely play a role in recreational activities being mentioned in 26% of entries. Only 4-8% of the studies indicated that precipitation, wind and water quality were important. In general recreational activities could be influenced by climatic conditions such as temperature and precipitation. Therefore, many recreational activities are seasonal partly because tourists prefer to go out during warm and dry weather conditions or during times of interesting animal behaviour. Indeed, 12% of the studies recorded temperature as an important factor. According to Fennell & Weaver (1997), wildlife viewing occurs most commonly during spring and summer in their study area in Canada. Good weather conditions were also cited as important in enhancing tour experience especially when viewing nocturnal animals (Wolf & Croft, 2012). Aside from how weather affects tourist's movements and times of tours, such abiotic factors affect animal behaviour. For example, air temperature and wind affects bird migration patterns which are an important aspect of bird watching (Burger et al., 1995). The authors also found a decrease in bat populations and species richness with an increase in wind speed suggesting a negative effect of wind on recreation.

Other factors that affect recreational services include proximity to target species, species behaviour and the rarity of species. A few studies recorded proximity to species as an important factor. Tourists love to be close to animals during game viewing and hunters definitely have a better chance at aiming at nearby targets. According to Ryan et al. (2000), an important aspect that gives visitor satisfaction is coming close to a large number of species in one place in their natural environment. While tourists are satisfied with coming close to species, the behaviour of individual species or a group is usually fascinating. Interestingly, 14% of studies cited animal behaviour as an important factor for their satisfaction. Amongst these, one study cited a tourist describing "the huge turtles lumbering out to sea after depositing the eggs" as fascinating (see Ballantyne et al., 2011), while another mentioned other fascinating activities, such as wallabies fighting, birds swimming, lizards coming to drink or birds swooping down to the water (Ryan et al., 2000). Tourists are most interested in carnivores and species that are interacting (fighting, mating, grooming or dancing birds) (Lee et al., 2010; Okello et al., 2008). In fact, some authors seem to suggest that the behaviour of the species is more important to tourists than the species themselves. For example, Okello et al. (2008) suggested that the wildebeest migration in the Maasai Mara is such a spectacle that tourists are possibly interested in the phenomenon of migration in practice rather than the wildebeest *per se*. Ryan et al. (2000) also found that tourist's fascination with crocodiles was based on the attributes of potential threat, danger, power, links with the prehistoric, and survivorship, but when asked to describe the Northern Territory, crocodiles did not immediately come to mind even though, arguably, the reptile is etched upon the ethos of the 'Top End' of the state.

Interestingly, 16% of studies mentioned that rare species were important in attracting tourists suggesting that many tourists could also be interested in particular species. Some studies have shown that visitors place more value on rare species suggesting a potential negative effect known as "Anthropogenic Allee Effect", a phenomenon where exaggerated value on rarity fuels disproportionate exploitation of rare species, rendering them even rarer and thus more desirable, ultimately leading them into an extinction vortex (Courchamp et al., 2006; Angulo & Courchamp, 2009).

The negative impact of biodiversity on recreation

No study recorded any negative impact of biodiversity on recreation in the review. However, some studies have reported injuries from wildlife attack or even death (Burns et al., 2003). In contrast, several studies have reported negative effects of recreational activities on biodiversity. For example, recreational fishing such as catch and release could have negative effects on fish physiology which may lead to higher mortality rates and a reduction of fish abundance (Cooke et al., 2005). Other effects of recreational fishing include: loss of genetic diversity, evolutionary changes due to selective angling, truncating size and age structures, reducing biomass, and altering community composition (Coleman et al., 2004; Lewin et al., 2006).

Strength of evidence

In general, there was a positive relationship between biodiversity and recreational services as 48% of the studies were classified as very strong and another 30% as strong. The remaining studies were either very weak, weak or average (4%, 8% and 10%, respectively). Some of the weak relationships were credible relationships between recreation and biodiversity. For example, Broch et al. (2012) showed that a greater number of hunted animals had a positive effect on the compensation level required for farmers to enter into a reforestation contract ($p < 0.01$) recognising the importance of species richness in hunting activities. However, this relationship was classified as being weak because it was a potential recreational activity and lacked empirical data.

A3.10.2 Results – Recreation service and value linkages

The results from the review of valuation of recreation service are summarised in Table A3.10.1. It can be seen from Table A3.10.1 that almost all reviewed studies either value direct consumptive use value or non-consumptive use value. The ecosystem service providers are animal species which are either hunted, fished or just watched. These ecosystem service providers are beneficial to society because they give individuals good recreation experiences.

The spatial scale of the studies is mainly local (21); others included six regional, four national and 4 global studies. The beneficiaries of recreation services are mainly specific groups of stakeholders (27) and society in general (7).

The benefits of recreation services are consumptive and non-consumptive experiences and this is reflected in the choice of valuation method. Twelve studies use the market price approach and eleven studies the stated preference approach. The choice between these two methods depends on the existence of relevant market prices. Seven studies use the revealed preference approach (travel cost method), which is possible in cases where the recreation service can be obtained by travelling to a recreation area.

Table A3.10.1: *Results from the review of valuation of recreation service.*

	Market price	Revealed preference	Stated preference	Benefit transfer and reviews	Total
Ecosystem service provider					
- animals (hunting, angling)	4	4	5	3	16
- animals (viewing)	7	2	4	2	15
- natural or artificial reefs	1	1	1		3
- forests			1		1
Value					
- direct consumptive use	4	4	5	3	16
- non-consumptive use	8	3	8	2	21
- non-use			1	1	2
Spatial scale					
- local	7	5	8	1	21
- regional	2	1	1	2	6
- national	2	1	1		4
- global	1		2	1	4
Beneficiaries					
- households/individuals			1		1
- communities/society	6			1	7
- group of stakeholders	5	7	11	4	27

Discussion

A substantial number of studies focus on local economic impacts of recreation (Butler et al., 2009; Duffus & Dearden, 1993; Kumar & Kumar, 2010; Wilson & Tisdell, 2003; Grado et al., 2007; Pennisi et al., 2004; Chen et al., 2003). This is clearly policy-relevant information, especially in the context of rural and coastal communities that are often relatively deprived, short of economic opportunities and facing depleted resource bases. From the biodiversity argumentation perspective, it can be important to demonstrate that conservation for recreational use brings jobs and income to local populations, in order to establish such uses as alternatives to (over-)harvesting for provisioning services (consumption or sale) and to justify conservation expenditures. These studies do not, however, deliver 'values' in the total economic value framework: they focus on direct expenditures, and often on indirect impacts via economic multipliers, and do not attempt to estimate any willingness to pay values.

It should also be stressed that the expenditure values are generally primarily of interest from a local or regional perspective, because expenditures are likely to be displaced from similar activities elsewhere in a country. This is not the case for major attractions that draw in international visitors, where there is a net boost to national economies.

Furthermore, we cannot assume that the value to recreational users is necessarily greater than the expenditures - the gross value of the recreation (hunting, angling, watching…) may be greater or less than their expenditures. For example, spending on food/restaurants is a contribution to the local economy, but not directly an indicator of any value from species-based recreation.

Some studies present both expenditure values and willingness to pay (WTP) values, generally through stated preference studies. Again, both are policy-relevant, from slightly different perspectives, but they should not be considered additive. Stoll et al. (2006) examine both expenditure and WTP among bird-watchers for conservation of Sandhill Cranes. Hvenegaard & Butler

(1989) explore birdwatching expenditures in some detail, coupled with a very basic contingent valuation (CV) question (What is the most your costs on this trip could have risen before deciding not to come birding?) and a basic question on potential sales (What items might you have purchased had they been available?).

Stated preference methods

Stated preference has been widely used to value hunting and angling. Non-consumptive uses have also been valued in this way, though often it is harder to identify specific species underpinning values. Some CV studies capture specifically the use value associated with hunting by focusing the valuation question on the loss of the hunting experience. Hunting does include other use values associated with the general environmental quality of the experience, jointly consumed with the hunting, but when the WTP expressed is for the hunting permit, as in the study by Boman et al. (2011) of moose, this can justifiably be considered as directly associated with hunting and with the specific target species. Similarly, Marangon & Rosato (1998) look at WTP specifically for hunting in particular reserves in Northern Italy, and are able to distinguish between hunting primarily for ungulates from hunting primarily for hare and birds. Signorello (1999) has a similar question regarding WTP an entrance fee for bird-watching areas in an Italian wetland.

Studies can also seek to explore values for changes in conditions. Boyle et al. (1998), for example, use CV to study values per trip for angling trout, bass or walleye, and hunting deer, elk or moose, and also to estimate the marginal values of catching/bagging one additional animal.

Other studies ask more general questions for which it is not possible to separate out specific aspects, for example, Clayton & Mendelsohn (1993) who use a question based on WTP for a 4 day visit permit to a bear watching area. Part of the WTP will be specifically related to bears, but it is not possible to tell exactly how much.

To varying degrees, there remains a possibility in these studies that some part of the values expressed may relate to a non-use 'donation' beyond a strict use-value – this depends on details of how the survey is constructed and administered. Sometimes it is clear that use and non-use are both covered. For example, Stoll et al. (2006) elicit wildlife-watchers' WTP for different conservation programmes: they demonstrate that birders preferred abundance over diversity, in their survey, but do not separate out use and non-use. Other studies cover both use and non-use, but seek explicitly to distinguish between them, as for example Bosetti & Pearce (2003) do in a study of grey seal populations.

Bosetti & Pearce also distinguish WTP for viewing in the wild from viewing captive animals, which is potentially important in the context of the argument that biodiversity can be adequately protected in zoos and parks: the recreation values are likely to be higher for natural settings. Similarly, Oh et al. (2008) use CV methods to contrast diving on artificial and natural reefs: while both display significant value, the natural reef experiences are valued substantially more.

Conjoint analysis studies offer an alternative to CV that makes it easier to separate out different components of value. Rudd (2001), for example, examines the value to divers of spiny lobster abundance: attributes in the study included size of dive group, price of the dive presence of macrofauna, Nassau grouper abundance and mean Nassau grouper size. The macrofauna attribute could be one or more spiny lobster, one or more sea turtles, one or more reef sharks, or none of these animals, allowing estimation of the marginal value of spiny lobster presence.

Revealed preference and combined methods

Travel cost methods have also been used. Edwards et al. (2010) use single-site travel cost to estimate values for bird-watching trips, and check this against a simple CV question, with results somewhere between the travel cost values for zero time value and time valued at 1/3 of the wage. Gurluk & Rehber (2008) use zonal travel cost to estimate values for a bird-watching site.

Random utility models used with travel cost or contingent behaviour data allow the value of recreational experiences to be broken down into component parts, helping tease out the contributions of specific species and general environmental quality, if the underlying data are rich enough. Knoche & Lupi (2007), for example, are able to develop marginal values for increases in deer populations and for increase in access to agricultural land for hunting, in both cases deriving separate values for firearm and archery hunters. Kragt et al. (2006) develop a contingent behaviour model to estimate changes in reef visits, consumer surplus and expenditures under a scenario of degradation of the Great Barrier Reef.

Studies have *compared* stated and revealed preference methods, for example, Charbonneau & Hay (1978) who derive broadly similar results from CV and travel cost studies of angling. More recent studies have *combined* methods to improve analysis. Prayaga et al. (2009) use a pooled revealed preference and contingent behaviour model, enabling them to study both current values of angling in part of the Great Barrier Reef Marine Park, and changes in values in a scenario of increased fish populations in future. Similarly, Gillig et al. (2003) use combined analysis of travel cost and CV data to assess WTP to maintain angling catch rates and for fish catch improvements for red snapper in the Gulf of Mexico, demonstrating how the joint model allows improved precision.

Value transfer

Valuation is an expensive business, but biodiversity arguments can often be made on the basis of value transfer from previous studies. This can apply to expenditure-based arguments as well as to value-based arguments. Some studies use value transfer methods to predict likely tourist expenditures under changed conditions. For example, Loomis (2006), estimates expenditures and existence values associated with expanding sea otter populations in California and uses his results to argue that these benefits exceed costs to fisheries.

Ritz & Ready (nd) use value transfer to estimate both costs and benefits of changing deer populations in Pennsylvania, distinguishing between the value to hunters and the value to viewers. There is a conceptual problem here that you can't shoot your deer and look at it too, and this is a general difficulty that must be considered if combining marginal values for consumptive and non-consumptive uses. Helvoigt & Charlton (2009) use value transfer to estimate market values, recreational values, and non-use values of NW pacific salmon and steelhead.

Value transfer can often be improved by using multiple studies (function transfer) rather than single studies (point transfer), although this is not axiomatic. Where enough studies exist, meta-analysis can be useful. Johnston et al. (2006) present meta-analysis of marginal WTP per fish from 48 angling studies. WTP per fish over the sample ranged from US$0.048 to US$612.79, with a mean of US$16.82. The meta-analysis found WTP is systematically influenced by resource, context, and angler characteristics; a small proportion of the variance in WTP is also accounted for by methodological variables.

Discussion

Three broad themes can be identified:

- Measurement of the local economic impact through estimation of user expenditures;
- Stated preference; and
- Revealed preference.

Stated preference methods have varied success in separating use values from non-use: this is not necessarily a problem, but if estimates specifically of recreational use values are required then the stated preference study needs to be designed with this in mind. Similarly, not all studies attempt to derive marginal values for changes in conditions. Revealed preference methods are widely used for recreation, and are increasingly combined with stated preference methods, especially contingent behaviour, to help them explore potential future changes.

All these methods can also be used in value transfer to new situations. Much depends on the specific argument context and care is needed to ensure that appropriate figures are selected to match the context, including the types of values covered (expenditures; recreational use of specific species; generalised recreation; recreation plus non-use), marginal or total values, and whether the values apply to a particular species or to a more general habitat or community.

One generally weak feature in the studies is the lack of detailed understanding of the links between biodiversity / ecological processes and the population status of the particular species of interest. This can limit the ability to use values to argue conservation points. It is not clear whether this weakness is because the knowledge is truly lacking, or more simply because the studies do not report this knowledge in detail. Greater emphasis on inter-and trans-disciplinary approaches to studying the value of biodiversity-based recreation would help to fill this gap.

A3.11 Landscape aesthetics

A3.11.1 Results – Biodiversity and landscape aesthetics linkages

Spatial and temporal scale, location of studies and methods used

Most of the studies in the review focused on the sub-national and local scale (85%), usually certain regions, typical landscapes or land use types (Table 3.11.1). The other 15% focussed on a specific land use type at the national level (e.g. forests, aquatic, rural areas).Most landscape appreciation studies were conducted in Europe (58%) and North America (22%), while the other 20% were conducted in Asia, Oceania, Africa and South America. The most studied land use types are: forests, rural areas and (peri)-urban areas.

Table A3.11.1: *Overview of the number of papers about landscape aesthetics covering different land use types and continents.*

Land-use	Europe	North-America	South-America	Asia	Oceania	Africa	Total
Forest (+grassland)	6	4	1	2	1	1	**15**
Protected nature	2			1			**3**
Rural	11			2			**13**
(Peri)-urban	4	5		1			**10**
Aquatic/watershed	1	1			1		**3**
Mixed/diverse landscapes	3	1			1		**5**
Total	**27**	**11**	**1**	**6**	**3**	**1**	**49**

The temporal scale of the studies is limited as the results are only representative for the period of the study. In only two studies, the time scale was extended by using scenarios (Black et al., 2010; Snep et al., 2009).

The most frequent method used to identify preferences for landscape aesthetics was through questionnaires with photographs (52%), followed by questionnaires and literature review (Figure A3.11.1).

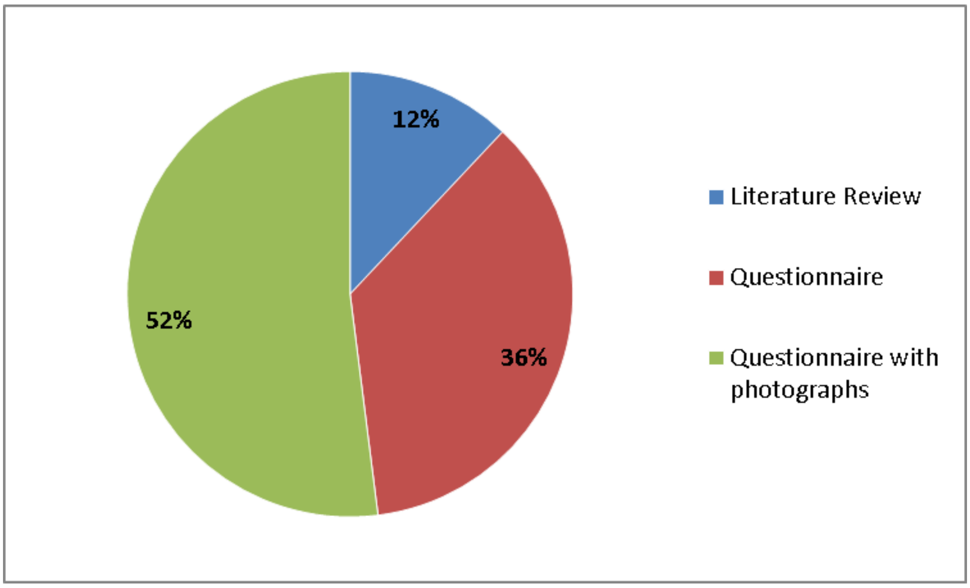

Figure A3.11.1: *Methods used to identify preferences for landscape aesthetics (% of total entries).*

Ecosystem service provider (ESP)

In most of the studies, the ESP was the entire community or habitat (84%) or two or more communities or habitats (16%, Figure A3.11.2). In the first case, a target group is asked why they appreciate a certain land use type or habitat; while in the second case, a target group is asked to compare the preferences between different land use types or habitats.

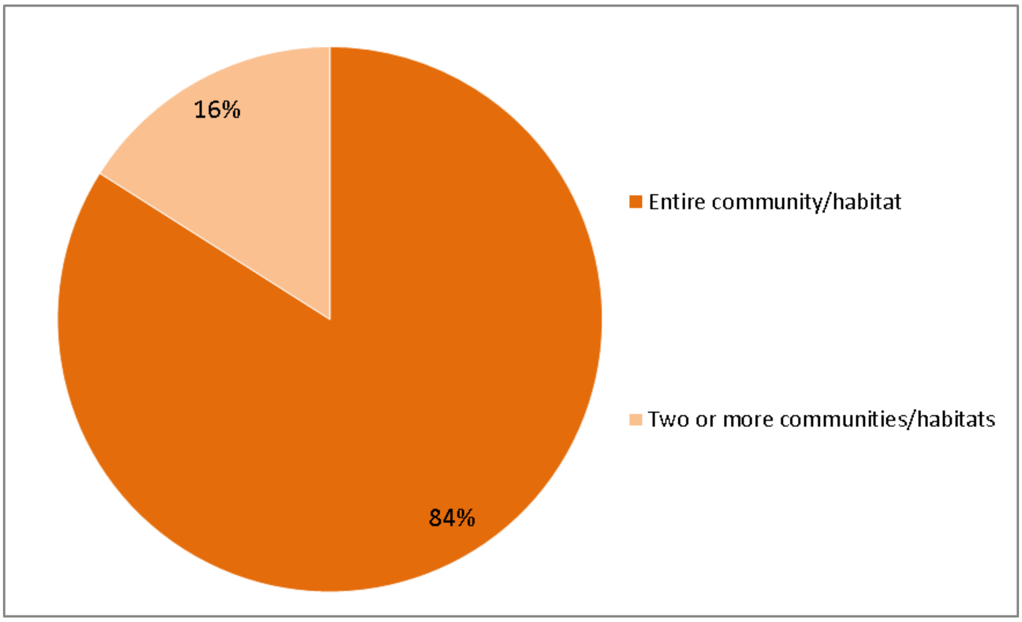

Figure A3.11.2: *Categorisation of the ecosystem service provider (ESP) for the ecosystem service of landscape aesthetics.*

Attributes of the ESP

The most important attribute of the ESP is community/habitat structure (43 entries or 86% of the entries, Figure A3.11.3). Other attributes of the ESP that affected landscape appreciation are: habitat area (17 entries), species richness (3), species abundance (1) and successional stage (1).

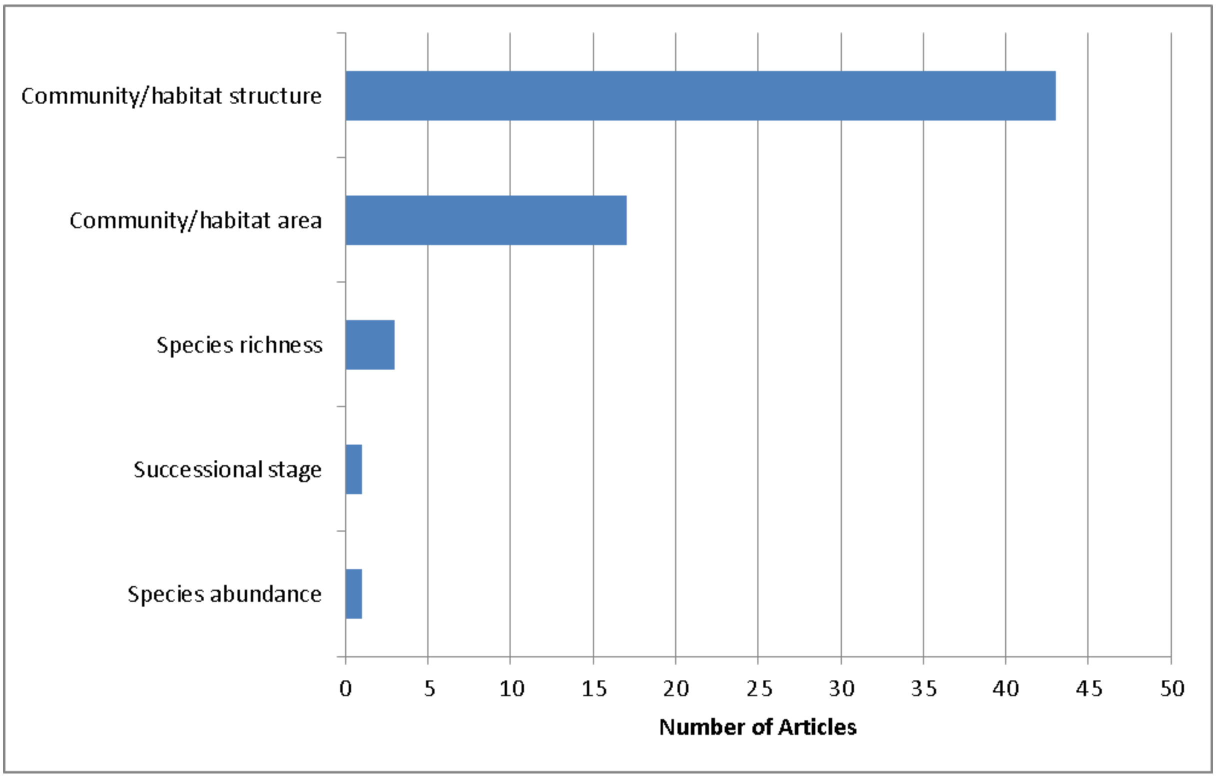

Figure A3.11.3: *Categorisation of the ESP attributes for the service landscape aesthetics.*

Habitat structure is defined in terms of complexity and/or heterogeneity: amount of structure or variation attributable to absolute abundance of individual structural component, or kinds of structure or variation attributable to the relative abundance of different structural components.

Habitat structure was not often mentioned as such, but could appear under many different labels:

- *Complexity and/or landscape diversity*: These or similar terms were used explicitly in seven entries: Van den Berg et al. (1998) found in the Netherlands that beauty ratings were positively related to perceived complexity; Huang (2013) found in Taiwan that participants liked landscapes more with increasing complexity, while Yao et al. (2012) found in China that perceived visual quality was positively influenced by the variety of vegetation. In the field of recreation, Kienast et al. (2012) detected in his Swiss study that preference for recreation areas is positively correlated with the number of land uses. Finally, in the field of well-being, Daniel et al. (2012) found that habitat diversity is positively correlated with the psychological well-being of visitors.
- *Uniqueness*: Uniqueness can also be an indicator of landscape diversity. It was mentioned twice: In the context of forest recreation tourism, the uniqueness of forest landscapes was an important attribute determining its attractiveness in Taiwan (Lee et al., 2010). For high-country communities in New Zealand, landscape features that make a locality distinctive were important (Swaffield & Foster, 2000).

- *Naturalness and natural features*: This was the most common term appearing in the selected studies (19 entries). For example, in urban areas, natural features provided positive effects on the visual quality of landscapes (Acar, 2008; Kaplan et al., 2006; Heyman, 2012). For tourism, natural features are found attractive by tourists, and leave a positive impression (Schmitz, 2007; Oğuz et al., 2010; Mikulec & Antoušková, 2011; Hadwen et al., 2012). In the case of forests, broadleaved woodlands were preferred above plantations in the study by Natori & Chenoweth (2008).
- *More dense vegetation*: The role of vegetation was mentioned in 18 of the entries. In the case of forests, those with a developed understory were appreciated more than those without (Holgén et al., 2000; Brown & Daniel, 1986; Püschel-Hoeneisen & Simonetti, 2012); older forests and those with larger trees were perceived to be more beautiful (Ribe, 2009), and forests with higher retention rates were preferred above clear-cuts (Ribe, 2006; 2009). Also, having some trees (especially bigger trees) in the working and living environment strongly affected employees' satisfaction with their workplace area (Kaplan, 2007). Furthermore, forest and public parks were features that respondents were most willing to pay for according to the study by Bowman et al. (2012).

However, it is very important to note that these positive appreciations cannot be generalised, and strongly depend on the type of stakeholder group. In the case of agricultural land, farmers and local people usually prefer more open landscapes, whereas recreationists show mostly a preference for low-intensity managed, species-rich landscapes with denser vegetation (Gómez-Limón & de Lucío Fernández, 1999; Swaffield & Foster, 2000; Dramstad et al., 2006; Pinto-Correia et al., 2011).

Such contrasting appreciation was also found between different ethnic groups. For example, for urban parks whites appreciate trees and other park vegetation; Latino's the cool refreshing “lake effect”; Asians especially mentioned the park's scenic beauty; while Blacks favour settings with a sense of openness and visibility, or which have built components (Kaplan & Talbot, 1988; Gobster, 2002). In the Netherlands, native Dutch people prefer the wilderness image, whereas immigrants from Islamic countries generally support functional nature and show low preferences for wild and unmanaged landscapes, like marshes and dunes (Buijs et al., 2009).

These findings make it clear that great care is needed in generalising results of landscape preference studies and that the effects of the context and the needs of stakeholder groups should be taken into account. This highlights that verification is needed in every case study.

Discussion

The literature search

It was not easy to find relevant papers, as the search returned a large number of hits, while only 7% of screened abstracts (497 in total) were relevant for this review. Many of the papers found by the search engine concerned the impact of tourism and recreation <u>on</u> biodiversity and nature in general, and quite a number of papers were purely focused on methodological aspects. The term “ecosystem service” was not very relevant for this search, as this term is not often used in this type of research (only two recent entries in our database used this term: García-Llorente et al., 2012 and Daniel et al., 2012).

Abiotic factors

A number of abiotic factors affect the delivery of the ecosystem service of landscape aesthetics or preference. The presence of water is very important in landscape appreciation. It was mentioned in 16 entries (30%). From the recreational point of view, most visitors appreciate the water element in the landscape (Swaffield & Foster, 2000; Dramstad et al., 2006; Oğuz et al., 2010), such as water courses (de Aranzabal et al., 2009; García-Llorente et al., 2012), river and lake shores (Kienast et al., 2012), lakes and ponds (Gobster, 2002), oceans, estuaries and river pools (Cocks et al., 2012) and water clarity, water quality and accessibility to water (Hadwen et al., 2012). On the other hand, traditional pastoralists strongly favour open, well watered land (Swaffield & Foster, 2000).

Climatic phenomena were mentioned in two papers only. Together with some specific natural elements, they were important attributes determining the attractiveness for tourism in Spain and Taiwan (de Aranzabal et al., 2009; Lee et al., 2010).

Topography and rockiness was also mentioned, with steeper reliefs, dramatic terrain, rock outcrops and summits (vistas) being cited as favourite landscape views (Swaffield & Foster; 2000; Pinto-Correia et al., 2011; García-Llorente et al., 2012; Kienast et al., 2012). One exception was found in China, where landscape preference increases as the variety of topography decreases (Yao et al., 2012). On the other hand, farmers obviously prefer land without rock outcrops (Pinto-Correia et al., 2011).

The negative impact of biodiversity on landscape aesthetics
Some landscapes are preferred above others, but no negative effects of biodiversity on landscape preference were reported.

Strength of evidence
Most of the reviewed papers (98%) presented average to strong evidence for the links between landscapes features and landscape preference. 'Average strength' was selected when sufficient people were interviewed, different cases and/or pictures were used, and background data were collected to justify the interpretations of the results. A 'strong score' was given for reviews which based their conclusions on several studies. A 'weak score' was given when very few people where interviewed, or when the methodology was not very strong to justify the conclusions (e.g. when conclusions were based on a simple comparison of two locations which differed on one parameter, or when the result of a non-representative group - for example students - was generalised over the entire population).

A3.11.2 Results – Landscape aesthetics and value linkages

The results from the review of valuation of landscape aesthetics service are summarised in Table A3.11.2.

It can be seen from Table A3.11.2 that all eleven reviewed studies value non-consumptive use value. Some of them (7) also value non-use values. Among the ecosystem service providers almost all landscape types are represented. These ecosystem service providers are beneficial to society because they give individuals good aesthetical experiences.

The spatial scale of the studies is mainly local (8), with others including two regional and one national study. The beneficiaries of recreation services are households/individuals (4), specific groups of stakeholdes (5) and society in general (3).

The benefits of landscape aesthetics services are non-consumptive experiences and non-use values which are reflected in the choice of valuation method. Eight studies use non-economic valuation approaches. This is because the relevant economic valuation approaches are not yet fully developed in relation to valuation of landscape aesthetics services. However, in three studies the revealed preference (1) and stated preference (2) approaches have been used.

Table A3.11.2: *Results from the review of valuation of landscape aesthetics service.*

	Revealed preference	**Stated preference**	**Non-economic value ranking**	**Other**	**Total**
Ecosystem service provider - heritage sites - mountain areas - rural landscapes - rivers - forests - city - other	1	1 1	1 2 1 1	1 1 1	2 2 2 2 1 1 1
Value - non-consumptive use - non-use	1	2 2	5 2	3 3	11 7
Spatial scale - local - regional - national	1	2	3 1 1	2 1	8 2 1
Beneficiaries - households/individuals - communities/society - group of stakeholders	1	2	1 1 3	2 2	4 3 5

Discussion

Observations from the review:

- There is a broad literature and several types of professions involved in evaluating values of aesthetic and cultural services (landscape planners, philosophers, anthropologists, economists, etc) – most of them are not from economics.
- Most of the studies classified here are non-economic and do not value the ecosystem service in monetary terms, rather they use e.g. rating methods, etc. from the landscape planning literature to rate aesthetic factors, etc.
- A few economic studies value aesthetic and heritage services. Those who do, typically use survey based approaches (CV or CE), or there are rare applications of hedonic valuations.
- Surprisingly few link aesthetic/heritage services directly to biodiversity, and those who study natural heritage have a more indirect link to wildlife, etc.
- Most values assessed are non-consumptive use values, related to recreation, enjoyment of views and landscapes. Some discuss non-use values and more subtle values such as those related to identity.
- There seem to be many different types of terminologies used (i.e. few papers classify within the ecosystem service framework), within different disciplines. The missing link to ecosystem services is partly because the concept is new.
- It is generally difficult to assess the “reliability of results” for this diverse literature, using many different methods and traditions.

A3.12 References

A3.12.1 Timber production references

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