aquacross



# Guidance on methods and tools for the assessment of causal flow indicators between biodiversity, ecosystem functions and ecosystem services in the aquatic environment

## Deliverable 5.1

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 642317.

## Authors

António Nogueira, Ana Lillebø, Heliana Teixeira, Michiel Daam (University of Aveiro); Leonie Robinson, Fiona Culhane (University of Liverpool); Gonzalo Delacámara, Carlos M. Gómez, Marta Arenas (IMDEA); Simone Langhans (IGB); Javier Martínez-López (BC3); Andrea Funk (BOKU); Peter Reichert, Nele Schuwirth, Peter Vermeiren (EAWAG); Verena Mattheiß (ACTeon)

## Contributors

Gerjan Piet, Ralf van Hal (Wageningen IMARES); Thomas Hein, Florian Pletterbauer (BOKU); Sonja Jähnig (IGB); Helen Klimmek (IUCN)

With thanks to:

Eeva Furman (SYKE; AQUACROSS Science-Policy-Business Think Tank); Tim O'Higgins (UCC); Manuel Lago, Katrina Abhold and Lina Roeschel (ECOLOGIC)

Project coordination and editing provided by Ecologic Institute.

Manuscript completed in December 2016

This document is available on the Internet at: http://aquacross.eu/project-outputs

| | |
|---|---|
| Document title | D5.1 Guidance on methods and tools for the assessment of causal flow indicators between biodiversity, ecosystem functions and ecosystem services in the aquatic environment |
| Work Package | WP5 |
| Document Type | Deliverable |
| Date | 29 December 2016 |
| Document Status | Final report |

## Acknowledgments & Disclaimer

This project has received funding from the *European Union's Horizon 2020 research and innovation programme* under grant agreement No 642317.

Citation: Nogueira, A., Lillebø, A., Teixeira, H., Daam, M., Robinson, L., Culhane, F., Delacámara, G., Gómez, C. M., Arenas, M.,Langhans, S., Martínez-López, J., Funk, A., Reichert, P., Schuwirth, N., Vermeiren, P., Mattheiß, V., 2016. Guidance on methods and tools for the assessment of causal flow indicators between biodiversity, ecosystem functions and ecosystem services in the aquatic environment. Deliverable 5.1, European Union's Horizon 2020 Framework Programme for Research and Innovation Grant Agreement No. 642317.

# Table of Contents

## List of Tables



## List of Figures

## List of Boxes

# List of Abbreviations

| | |
|---|---|
| ABM | Agent-based Modelling |
| AF | Assessment Framework |
| ARIES | Artificial Intelligence for Ecosystem Services |
| BBN | Bayesian Belief Networks |
| BD | Biodiversity |
| BEF | Biodiversity-Ecosystem Functioning |
| BES | Biodiversity-Ecosystem Services |
| CICES | Common International Classificaton of Ecosystem Services |
| CS | Case Study |
| DA | Discriminant Analysis |
| DPSIR | Drivers-Pressures-State-Impact-Response |
| EBM | Ecosystem-Based Management |
| EF | Ecosystem Function |
| ESS | Ecosystem Services |
| GDIM | Generalised Diversity-Interactions Models |
| GDM | Generalised Dissimilarity Models |
| IPBES | Intergovernmental Platform on Biodiversity and Ecosystem Services |
| MA | Millennium Ecosystem Assessment |
| MAES WG | Mapping and Assessment of Ecosystem Services Working Group |
| MSFD | Marine Strategy Framework Directive |
| SD | System Dynamics |
| SEMs | Structural Equation Models |
| SES | Social-ecological Systems |
| SNA | Social Network Analysis |
| TEEB | The Economics of Ecosystems and Biodiversity |
| WP | Work Package |

# About AQUACROSS

Knowledge, Assessment, and Management for AQUAtic Biodiversity and Ecosystem Services aCROSS EU policies (AQUACROSS) aims to support EU efforts to protect aquatic biodiversity and ensure the provision of aquatic ecosystem services. Funded by Europe's Horizon 2020 research programme, AQUACROSS seeks to advance knowledge and application of ecosystem-based management for aquatic ecosystems to support the timely achievement of the EU 2020 Biodiversity Strategy targets.

Aquatic ecosystems are rich in biodiversity and home to a diverse array of species and habitats, providing numerous economic and societal benefits to Europe. Many of these valuable ecosystems are at risk of being irreversibly damaged by human activities and pressures, including pollution, contamination, invasive species, overfishing and climate change. These pressures threaten the sustainability of these ecosystems, their provision of ecosystem services and ultimately human well-being.

AQUACROSS responds to pressing societal and economic needs, tackling policy challenges from an integrated perspective and adding value to the use of available knowledge. Through advancing science and knowledge; connecting science, policy and business; and supporting the achievement of EU and international biodiversity targets, AQUACROSS aims to improve ecosystem-based management of aquatic ecosystems across Europe.

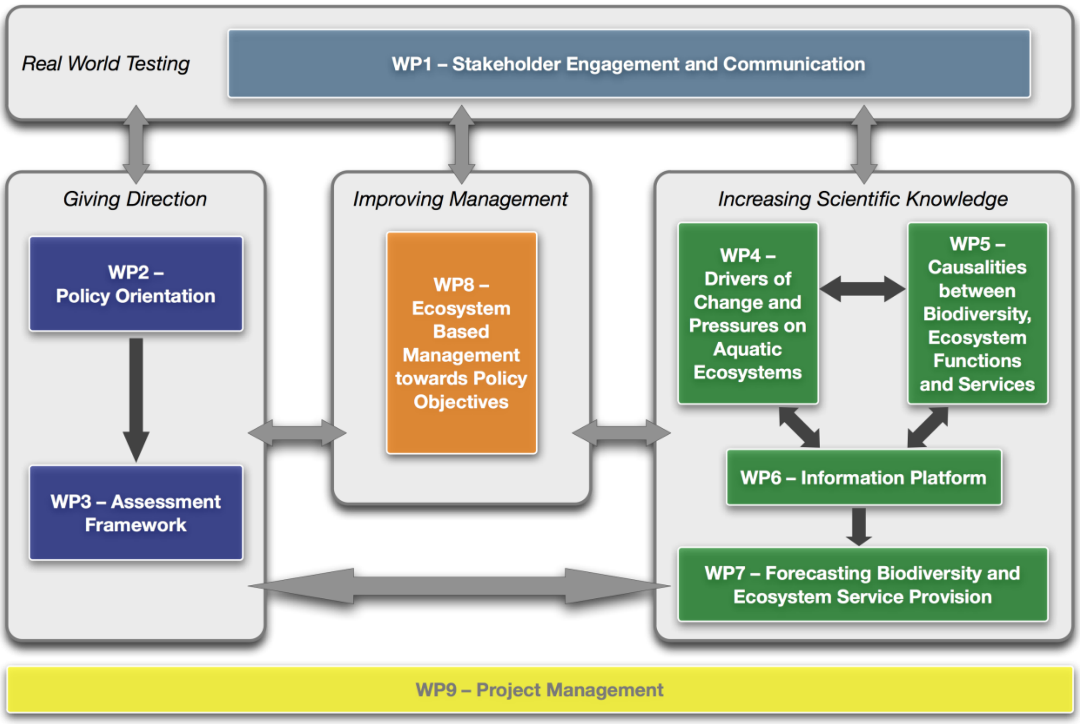
The project consortium is made up of sixteen partners from across Europe and led by Ecologic Institute in Berlin, Germany.

| | |
|---|---|
| **Contact** | **aquacross@ecologic.eu** |
| **Coordinator** | **Dr. Manuel Lago, Ecologic Institute** |
| **Duration** | **1 June 2015 to 30 November 2018** |
| | |
| **Website** | **http://aquacross.eu/** |
| **Twitter** | **@AquaBiodiv** |
| **LinkedIn** | **www.linkedin.com/groups/AQUACROSS-8355424/about** |
| **ResearchGate** | **www.researchgate.net/profile/Aquacross_Project2** |

# 1 Background

As part of Pillar 2, "Increasing Scientific Knowledge", of AQUACROSS (Figure 1), Work Package 5 (WP5) builds on the overarching Assessment Framework developed in WP3 to investigate in more detail the causalities between biodiversity, ecosystem functions and services dimensions (Task 5.1), and applies the framework in case studies to test and refine its applicability (Task 5.2). The impact of drivers and pressures (identified in WP4) will be incorporated in existing models and contribute to a correct definition of ecosystem status. The outputs of WP5 will contribute directly to WP6 (data analyses) and WP7 (forecasting of biodiversity and ecosystem services provisioning), and ultimately to WP8 (provide support to facilitate and promote science/policy communication). The results of the application of the AQUACROSS Assessment Framework to the case studies will be synthesised to feed back into the update of the framework and help formulate policy recommendations (Task 5.3).

Figure 1: AQUACROSS "four pillars" and work package structure




The objectives of WP5 include:

- Scope and design relevant and feasible indicators, methods and tools to assess changes in aquatic ecosystem status and service provision for the application of ecosystem-based management (EBM) (link to WP4, WP6, WP7 and WP8).

- Apply and test the AQUACROSS conceptual framework in regard to the investigation of the causalities between biodiversity and ecosystem functions and services across aquatic domains (link to WP3).
- Explore any existing causal links between biodiversity, ecosystem functions and services at different temporal and spatial scales for the case study areas, taking into account the drivers and pressures identified in WP4 (further link to WP7).
- Draw lessons to update the AQUACROSS conceptual framework and improved application of EBM of aquatic ecosystems (link to WP3 and WP8).

The work described in this report forms part of the AQUACROSS Assessment Framework (AF; Gómez et al., 2016a,b) and focuses on the causal links between biodiversity (BD) (directly measured or as captured by the state of ecosystems) and the ecological processes ensuring crucial ecosystem functions (EF) that enable the supply of ecosystem services (ESS). These are central themes to this stage of the AF that fit within the supply-side perspective (Figure 2) of the AQUACROSS Innovative Concept (Gómez et al., 2016a), and this document follows the conceptual definitions agreed by Gómez et al. (2016b).

The present report scrutinises the findings that have been achieved so far through a literature review on the current state of knowledge on links between biodiversity and ecosystem functions (BEF; Section 2) and ecosystem services (BES; Section 3). A brief reference is made to existing meta-analysis performed within the context of BEF and BES relationships (Section 4).

The concepts of biodiversity, ecosystem processes, ecosystem functions and ecosystem services have not always been addressed in the same way, namely in different pieces of legislation with implications for AQUACROSS objectives and work. As such, although we try to critically integrate the definitions of these concepts, in the context of AQUACROSS some definitions were agreed (Box 1). The background reasons behind these definitions will be presented in the next chapters.

The use of indicators for biodiversity, EF and ESS in the context of the AF is also discussed (Section 5), and sources of potentially useful indicators are listed, in order to provide examples for case studies (see Annex). The report concludes with an overview of methods to analyse causal links between biodiversity, ecosystem functions and services (Section 5.2.1), considering the AQUACROSS working framework supply-side, from state to benefits[1] (Figure 2).

[1] The assessment of Benefits and Values is not in the scope of the present report, which ends at the boundary of how the capacity to supply ecosystem services is affected by the state of the ecosystem.

## Box 1: Definition of Biodiversity, Ecosystem Process, Ecosystem Function and Ecosystem Services within AQUACROSS

**Biodiversity = Biological Diversity** means the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems (Convention on Biological Diversity, article 2). Biological diversity is often understood at four levels: genetic diversity, species diversity, functional diversity, and ecosystem diversity.

**Ecosystem Process** is a physical, chemical or biological action or event that link organisms and their environment. Ecosystem processes include, among others, bioturbation, photosynthesis, nitrification, nitrogen fixation, respiration, productivity, vegetation succession.

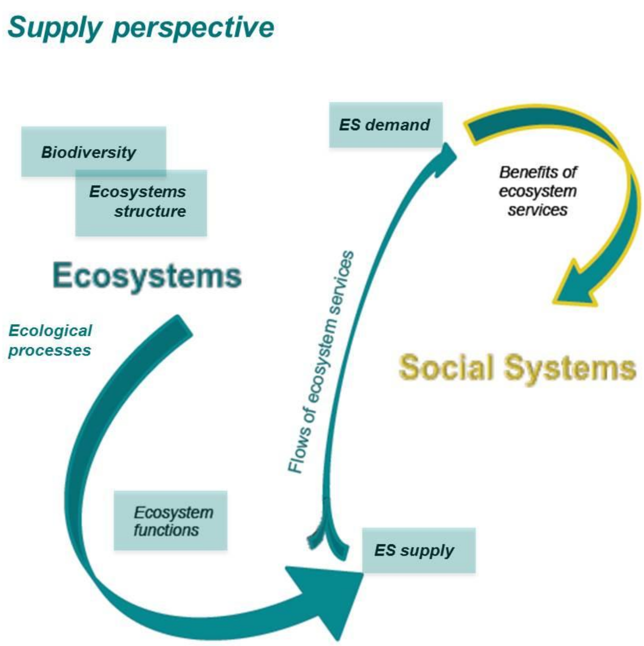
**Ecosystem Function** is a precise effect of a given constraint on the ecosystem flow of matter and energy performed by a given item of biodiversity, within a closure of constraints. Ecosystem functions include decomposition, production, nutrient cycling, and fluxes of nutrients and energy.

**Ecosystem Services** are the final outputs from ecosystems that are directly consumed, used (actively or passively) or enjoyed by people. In the context of the Common International Classificaton of Ecosystem Services (CICES), they are biologically mediated (human-environmental interactions are not always considered ecosystem services).

Example to integrate and differentiate concepts:

**Organic matter mineralisation** is an *ecosystem process* that leads to **carbon sequestration** (*ecosystem function*) contributing to **carbon storage** (*ecosystem service*) in the form of Green Carbon or Blue Carbon.

## Figure 2: The supply-side perspective of the AQUACROSS Architecture addressed in this report




Source: Gómez et al. (2016b)

# 2 Biodiversity-Ecosystem Functioning Relationships

The present chapter builds on a literature review on biodiversity and ecosystem functioning relationships.[2]

## 2.1 Introduction

Concern has grown over the past decades about the rate biodiversity is declining and its consequences for the functioning of ecosystems and the subsequent services they provide. This concern has triggered several international initiatives to ensure healthy ecosystems and, hence, the provision of essential services to people. Extensive scientific research was also initiated to better understand the link between biodiversity and ecosystem functioning (BEF) on one side and between biodiversity and ecosystem services (BES) on the other.

A vast number of existing experimental and observational BEF studies, and meta-analyses of data were generated by these studies, which tested the hypothesis that ecosystems with species-poor communities are functionally poorer, less resistant (capacity to resist change) and resilient (capacity to recover from change) to disturbance than systems with species-rich communities (Covich et al. 2004; Stachowicz, Bruno, and Duffy 2007; Strong et al., 2015). Reviewing the available BEF literature, Cardinale et al. (2012) concluded that "*There is now unequivocal evidence that biodiversity loss reduces the efficiency by which ecological communities capture biologically essential resources, produce biomass, decompose and recycle biologically essential nutrients.*"

One of the initial goals of AQUACROSS WP5 is to review the current state of knowledge on links between biodiversity, ecosystem functions and ecosystem services in aquatic realms (i.e., freshwater, coastal and marine). As a first step towards this, the present chapter aims at identifying the potential and the drawbacks of existing knowledge and BEF evaluations and their potential usefulness for the objectives of AQUACROSS. This chapter is organised as follows: the next part presents (i) underlying BEF mechanisms (Section 2.2); (ii) the shape of aquatic BEF relationships reported in the literature (Section 2.3); (iii) whether BEF relations are ecosystem-specific or whether they are interchangeable (Section 2.4); and (iv) current research limitations and needs in aquatic BEF studies (Section 2.5).

[2] Daam, M. A., Ana I. Lillebø, A. I.; Nogueira, A. J. A. Challenges in establishing causal links between aquatic biodiversity and ecosystem functioning (*in prep.*).

## 2.2 Underlying BEF mechanisms

Several mechanisms have been denoted to explain the influence of compositional diversity on ecosystem functioning, including: complementary niche partitioning, density-dependent effects, facilitation mechanisms, and identiy effects. These mechanisms are defined below using examples from aquatic realms:

- **Complementary niche partitioning**: occurs when several species coexist at a given site and complement each other spatially and temporally in their patterns of resource use (Truchy et al., 2015). Karlson et al. (2010), for example, showed that more diverse deposit-feeding marine macrofauna communities incorporated more nitrogen than a single-species treatment of the best-performing species, showing transgressive over-yielding through positive complementarity (practical aspects linked with transgressive overyielding concept are detailed in Schmid et al. (2008)). According to the authors, more diverse sediment communities showed more efficient trophic transfer of phytodetritus through niche partitioning among species from different functional groups, and a higher incorporation by surface feeders in multispecies treatments.
- **Density-dependent effects**: occur when species assemblage at a given site establish species-specific interactions (e.g., seagrass density has positive effects on crustaceans and fishes, but net effects could be negative through increased predation on small crustaceans by facilitating predatory fishes; Duffy 2006). In some cases, the expected prevailing processes, namely niche partitioning or competition, will be determined by the density of a specific species assemblage, and that will determine the magnitude of the ecosystem response (Sanz-Lázaro et al., 2015).
- **Facilitation**: occurs when activities of some species enhance or facilitate activities of others and, in turn, ecosystem process rates. For instance, within the suite of processes underpinning water purification in freshwaters, facilitation is seen when diverse assemblages of filter-feeding caddisflies capture more suspended material than they could in monoculture (Truchy et al., 2015). In this way, species diversity reduces 'current shading' (that is, the deceleration of flow from upstream to downstream neighbours), allowing diverse assemblages to capture a greater fraction of suspended resources than is caught by any species monoculture (Cardinale and Palmer, 2002). Facilitative changes in physical conditions induced by a facilitator produce a broadening of dependent species niches. For instance, on intertidal rocky shores, buffering from canopy-forming microalgae and mussels makes upper shore levels suitable for many species not able to tolerate environmental conditions in open areas (Bulleri et al., 2016). There might also be evolutionary aspects related to niche partitioning and facilitation that conditionates the ecosystem response (Reiss et al., 2009).
- **Identity effects**: occur in situations where particular species have a disproportionate functional role, and may subsequently also generate positive BEF relationships. When only a few species have a large effect on ecosystem functioning, increasing species richness increase the likelihood that those key species would be present (Hooper et al., 2005). This

form of non-transgressive over-yielding can also be called sampling or selection effects (Strong et al., 2015). For example, reduced nutrient recycling processes with declining fish diversity have been attributed to identity effects with relatively few species dominating nutrient recycling (McIntyre et al., 2007; Allgeier et al., 2014).

BEF research has explored multiple hypotheses for how organisms promote EFs: (i) the diversity hypothesis: mechanisms including niche complementarity and insurance (compensatory dynamics through space and time) and (ii) the mass ratio hypothesis (functional traits of dominant species chiefly promote EFs-identity effects) (Duncan, Thompson, and Pettorelli, 2015; Vaughn, 2010). Experimental BEF research focusing on species richness has provided broad support for the diversity hypothesis, whereas trait-based research has shown that many EFs are driven predominantly by mass ratio (Duncan, Thompson, and Pettorelli, 2015). Ultimately, both hypotheses are due to trait expression, and a combination of both species richness and identity may evidently play an important role (Fu et al., 2014; Vaughn, 2010). This also dictates that the sole evaluation of taxonomic changes is not sufficient to study BEF relationships, since i) species composition can change without concomitant functional changes, and ii) functioning can change even when species are unaffected, for example, through changed interactions or behaviours by the resident species (Truchy et al., 2015).

Examining species traits is also imperative since recent assessments have shown that global biodiversity loss preferentially affects species with longer life spans, bigger bodies, poorer dispersal capacities, more specialised resource uses, lower reproductive rates, among other traits that make them more susceptible to human pressures (Pinto, de Jonge, and Marques, 2014). Oliver et al. (2015) discussed that response traits (attributes that influence the persistence of individuals of a species in the face of environmental changes) and effect traits (attributes of the individuals of a species that underlie its impacts on ecosystem functions and services) of species also have a great influence on the resilience of ecosystem functions: "If the extent of species' population decline following an environmental perturbation (mediated by response traits) is positively correlated with the magnitude of species' negative effects on an ecosystem function (via effect traits), this will lead to less resistant ecosystem functions" (Oliver et al., 2015).

## 2.3 Shape of BEF relationships

After indications were derived from early BEF research that species richness was positively associated with ecosystem processes, several hypothetical associations between biodiversity and ecosystem function were proposed in the 1980s and 1990s (Naeem, 2008). Since the turn of the century, this was followed by various meta-analyses of data from experimental studies to unravel the shape and function of the BEF relationship (Balvanera et al., 2006; Worm et al., 2006; Schmid, Pfisterer, and Balvanera, 2009; Cardinale et al., 2011; Reich et al., 2012; Mora, Danovaro, and Loreau, 2014; Stachowicz, Bruno, and Duffy, 2007). Cardinale et al. (2011), for example, examined how species richness of primary producers influences the suite of ecological processes that are controlled by plants and algae in terrestrial, marine and freshwater ecosystems. By fitting experimental data to several mathematical functions (linear, exponential, log, power and Michaelis-Menten), they noted that the best fit was obtained by a Michaelis-Menten function[3] but that the difference was not considerable when compared to the power model.

Mora, Danovaro, and Loreau (2014) noted that BEF relationships in large-scale observational marine ecosystems generally yield non-saturating (convex) patterns with slopes on log-log scale ranging from 1.1 to 8.4, whereas ecosystem functioning rapidly saturates with increasing biodiversity in (concave) BEF functions from experimental marine studies that showed slopes on log-log scale ranging from 0.15 to 0.32. The authors attributed this to the fact that experimental studies fail to reveal the positive role of ecological interactions on species' production efficiency, as competition, instead of specialisation, is more likely to prevail in experimental settings. When species are put together in a contained artificial experimental setup, they are forced to compete or interact, which may lead to greater energy loss than under field conditions where specialisation may have already occurred (Mora, Danovaro, and Loreau, 2014).

The above indicates a serious limitation. As the Michaelis-Menten function is not adequate to describe concave relationships, such as those emerging from observational marine studies, it cannot be used for comparing different types of relationships (Mora, Danovaro, and Loreau, 2014). The authors provided three alternative hypotheses to explain this contrast between experimental and observational studies: i) the use of functional richness instead of species richness, ii) an increased production efficiency of species in producing biomass when more ecological interactions are present, and iii) the fact that communities are likely assembled in an ordered succession of species from low to high ecological efficiency.

Several other authors have also argued that different experimental designs will result in different BEF relationship results (Stachowicz et al., 2008; Byrnes and Stachowicz, 2009; O'Connor and Bruno, 2009; Campbell, Murphy, and Romanuk, 2011). Stachowicz et al. (2008), for example, argued that short-term experiments detect only a subset of possible mechanisms that operate in the field over the longer term, because they lack sufficient environmental heterogeneity to allow expression of niche differences, and they are of insufficient length to capture population-level responses, such as recruitment. Spatial heterogeneity of the physical environment has indeed been reported to play a key role in mediating effects of species diversity (Griffin et al., 2009). It should be noted, however, that resource heterogeneity must be accompanied by a broad enough trait diversity in order for resource partitioning to occur (Weis, Madrigal, and Cardinale, 2008; Ericson, Ljunghager, and Gamfeldt, 2009).

[3] A Michaelis-Menten function is a first order saturation curve (parabol) that can be used to describe the kinetics of a large number of biological processes.

In contrast to the above, Godbold (2012) and Gamfeldt et al. (2014) only encountered small, mostly non-significant, differences in marine BEF relationships between experiments performed in the laboratory, in mesocosms[4] and in the field. Causal effects of phytoplankton on functional properties in large-scale observational freshwater and brackish water studies have also been reported to be consistent with experimental and model studies (Ptacnik et al., 2008; Zimmerman and Cardinale, 2013). Furthermore, recently, a large-temporal experiment on BEF (Meyer et al., 2016) found evidence of a strong effect of biodiversity on ecosystem functioning due to "*both a progressive decrease in functioning in species-poor and a progressive increase in functioning in species-rich communities,*" with negative feedbacks, at low biodiversity, and complementarity among species, at high biodiversity, similarly contributing for biodiversity effects. They concluded, moreover, that species loss is likely to impair ecosystem functioning "*potentially decades beyond the moment of species extinction.*"

Regardless of the experimental design applied, BEF relationships appear to be best approximated by a power function: $Y \sim \kappa S^{\beta}$, where Y is the ecosystem functioning of a community with S species, and $\kappa$ and $\beta$ are constants (Isbell et al., 2015; Mora, Danovaro, and Loreau, 2014; Gamfeldt, Lefcheck, and Byrnes, 2014). The shape of the BEF curve changes depending on the value for the $\beta$ constant where curves are increasingly saturating as $\beta$ approaches 0 (Isbell et al., 2015). Reported values for the constants and, hence, shape and strength of the BEF relationships are highly variable. They appear i) to, at least partly, depend on the environmental context and on which species are lost, e.g. the loss of initially abundant species can reduce ecosystem functioning more than the loss of initially rare species; ii) to be stronger in longer experiments than those in short-term experiments and stronger in observational studies than experimental studies as discussed above; iii) to have $\beta$-values > 0.5 for some types of non-random biodiversity loss, and when considering the greater proportion of biodiversity that is required to maintain multiple ecosystem functions at multiple times and places such as large-scale observational studies; and iv) to show reduced slopes with increased disturbance level (Cardinale, Nelson, and Palmer, 2000; Biswas and Mallik, 2011; Mora, Danovaro, and Loreau, 2014; Isbell et al., 2015).

[4] "Aquatic mesocosms, or experimental water enclosures, are designed to provide a limited body of water with close to natural conditions, in which environmental factors can be realistically manipulated. Mesocosm studies maintain a natural community under close to natural conditions, taking into account relevant aspects from 'the real world' such as indirect effects, biological compensation and recovery, and ecosystem resilience" (https://www.mesocosm.eu/what-is-a-mesoscosm).

## 2.4 Do BEF relationships extrapolate over ecosystem types?

Several authors have reported a striking level of generality in diversity effects on ecosystem functioning across terrestrial, freshwater and marine ecosystems, and among organisms as divergent as plants and predators (Bruno et al., 2005; Moore and Fairweather, 2006; Handa et al., 2014; Hodapp et al., 2015; Lefcheck et al., 2015; Stachowicz et al., 2008; Cardinale et al., 2011; Gamfeldt, Lefcheck, and Byrnes, 2014). Stachowicz et al. (2008), for example, suggested that experimental design and approach, rather than inherent differences between marine and terrestrial ecosystems, underlie contrasting responses among systems.

Gamfeldt, Lefcheck, and Byrnes (2014) stated that, although BEF relationships appear to be non-ecosystem specific, it should be noted that marine and terrestrial realms differ in terms of their phylogenetic diversity at higher levels. For example, 15 phyla are endemic to marine environments, and the primary producers in the ocean belong to several kingdoms. On land, however, primary producers are mainly from the Plantae kingdom (Gamfeldt, Lefcheck, and Byrnes, 2014). Compared to terrestrial systems, aquatic ecosystems are also characterised by greater propagule[5] and material exchange, often steeper physical and chemical gradients, more rapid biological processes and, in marine systems, higher phylogenetic diversity[6] of animals (Giller et al., 2004).

These differences limit the potential to extrapolate conclusions derived from terrestrial experiments to aquatic ecosystems. According to Duncan, Thompson, and Pettorelli (2015), a focus on within-ecosystem type studies is hence crucial, as the nature of BEF linkages can be highly context-dependent, such as abiotic and climatic controls, disturbance and management. Hence, this also hampers the extrapolation of BEF relationships between different aquatic ecosystem types (freshwater, coastal and marine).

The mechanism behind BEF relationships also appears to be different between ecosystem types. For example, whereas complementarity is prevalent in terrestrial studies (Cardinale et al., 2007), positive BEF relationships examined in the marine environment are mostly driven by identity effects (Stachowicz, Bruno, and Duffy, 2007; Cardinale et al., 2012; Gamfeldt, Lefcheck, and Byrnes, 2014; Strong et al., 2015). In addition, aquatic and terrestrial systems are known to differ in the relative strength of top-down versus bottom-up effects (Srivastava

---

[5] In biology, a propagule is any material that is used for the purpose of propagating an organism to the next stage in their life cycle. In broader terms a propagule can be considered as the dispersive form of a organism (it can be a seed, a spore or even a larval form of an animal specie).

[6] Phylogenetic diversity measures the relative feature diversity of different subsets of taxa from a given phylogeny (i.e., the history of lineages of organisms as they change through time).

et al., 2009). Subsequently, BEF relationships may not directly extrapolate across ecosystem types, although BEF relations established in a certain ecosystem type may provide indications for further studies and/or additional evidence for their existence in other ecosystem types.

## 2.5 Research limitations and needs

### 2.5.1 Multiple EF relationships

The influence of compositional diversity on ecosystem function is a consequence of a range of mechanisms (see above), which become increasingly important as more ecosystem functions are considered (Isbell et al., 2011; Mouillot et al., 2011). For example, contrary to studies focusing on single ecosystem functions and considering species richness as the sole measure of biodiversity, Mouillot et al. (2011) found a linear and non-saturating effect of the functional structure, i.e. the composition and diversity of functional traits, of communities on ecosystem multifunctionality.

Greater levels of biodiversity may, thus, be required to support multiple EFs simultaneously, as the functional traits and importance of complementarity may vary for different EFs (Duncan, Thompson, and Pettorelli, 2015). This indicates that prior research has underestimated the importance of biodiversity for ecosystem functioning by focusing on individual functions and taxonomic groups (Andy Hector and Bagchi, 2007; Lefcheck et al., 2015). The need for considering multiple functions in BEF research has, therefore, often been discussed (Duncan, Thompson, and Pettorelli, 2015; Gamfeldt, Lefcheck, and Byrnes, 2014; Strong et al., 2015).

Accounting for interactions between ecosystem functions may complicate determining the response of individual ecosystem functions to biodiversity, since an increase in the functional output within one ecosystem function may change the availability of resources or substrate for use in other ecosystem functions – the so-called “spill-over” effect (Strong et al., 2015). Subsequently, the field of biodiversity and ecosystem multifunctionality is still relatively data poor due to the (i) complex issues generated by the analysis of multifunctionality, (ii) the effort to conduct experiments with many levels of species richness, and the (iii) difficulty of measuring more than a handful of functions (Byrnes et al., 2014).

This complexity may be illustrated with the fact that underlying diversity measures may vary among the different BEF relationships co-existing in natural ecosystems. Thompson et al. (2015), for example, showed that in natural pond communities, zooplankton community biomass was best predicted by zooplankton trait-based functional richness, while phytoplankton abundance was best predicted by zooplankton phylogenetic diversity. Similarly, Hodapp et al. (2015) showed that different aspects of biodiversity (richness, evenness) were significantly linked to different ecosystem functions (productivity, resource use efficiency).

Duncan, Thompson, and Pettorelli (2015) suggested grouping of EFs according to (i) the main contributing group (trophic level or functional group), (ii) functional traits, and (iii) underlying BEF mechanisms. By providing the underlying structure of species interactions, ecological networks may also aid in quantifying connections between biodiversity and multiple ecosystem functions (see Hines et al., 2015 for more detail).

### 2.5.2 Rare species and ecosystem connectivity

Although common species are typically drivers of ecosystem processes (Moore, 2006; Vaughn, 2010), the high functional distinctiveness of rare species indicate that they also support vulnerable functions, especially in species-rich ecosystems where high functional redundancy among species is likely (Jain et al., 2013; Mouillot, Bellwood, et al., 2013). For example, Bracken and Low (2012) showed that realistic losses of rare species in a diverse assemblage of seaweeds and sessile invertebrates, collectively comprising <10% of sessile biomass, resulted in a 42-47% decline in consumer biomass, whereas removal of an equivalent biomass of dominant sessile species had no effect on consumers. This also emphasises the importance of including system connectivity in experimental designs to allow an extrapolation of biodiversity ecosystem-functioning relationships to natural systems (Matthiessen et al., 2007).

Communities that are connected to a metacommunity via immigration are more diverse and stable than isolated communities; hence corridors in connected metacommunities can mitigate, and even reverse, local extinctions and disruption of ecosystem processes (Loreau, Mouquet, and Holt, 2003; Staddon et al., 2010; Downing, Brown, and Leibold, 2014). France and Duffy (2006), however, demonstrated that at the metacommunity level, grazer dispersal eliminated the stabilising effect of diversity on ecosystem properties, and at the patch level, grazer dispersal consistently increased temporal variability of the ecosystem properties measured.

Both results contradict the spatial insurance hypothesis, which is based on equilibrium metacommunities of sessile organisms with passive dispersal (Loreau, Mouquet, and Holt, 2003). In this way, habitat fragmentation, together with declining biodiversity, influence the predictability of ecosystem functioning synergistically (France and Duffy, 2006). The insurance hypothesis relies on the positive effect that biodiversity has on EF because of the variability of responses to changes in the environment (i.e. compensation); therefore, habitat fragmentation acts synergistically with biodiversity loss (decreasing this maintained level of processes). This is especially important for aquatic ecosystems, since barriers to dispersal are typically weak and flow of energy and materials is relatively rapid within and between habitats of these ecosystems (Hawkins, 2004; Giller et al., 2004).

### 2.5.3 Trophic levels

Large BEF evidence gaps align with several of the more functionally important trophic components (Strong et al., 2015). Microbial communities, for example, play key roles in maintaining multiple ecosystem functions and services simultaneously, including nutrient cycling, primary production, litter decomposition and climate regulation (Glöckner et al., 2012; Delgado-Baquerizo et al., 2016; Zeglin, 2015). Although positive effects of bacterial diversity on ecosystem functioning have previously been demonstrated, BEF studies into microbial communities are relatively scarce (Dell'Anno et al., 2012; Venail and Vives, 2013). This is, at least, partly due to the fact that defining and measuring biodiversity in consistent and meaningful units for the microscopic biological components, such as the microbial assemblages, and at the genetic scale, pose significant challenges (Strong et al., 2015).

Regarding genetic scale, a literature review by Hughes et al. (2008) revealed significant effects of genetic diversity on ecological processes, such as primary productivity, population recovery from disturbance, interspecific competition, community structure, and fluxes of energy and nutrients. Hughes and Stachowicz (2004), for example, showed that increasing genotypic diversity in a habitat-forming species (the seagrass *Zostera marina*) enhanced community resistance to disturbance by grazing geese. Thus, genetic diversity can have important ecological consequences at the population, community and ecosystem levels, and in some cases, the effects are comparable in magnitude to the effects of species diversity (Duffy, 2006; Latta et al., 2010; Massa et al., 2013; Roger, Godhe, and Gamfeldt, 2012; Hughes and Stachowicz, 2004; Hughes et al., 2008). In line with this, intraspecific variability has been discussed to be a key driver for biodiversity sustenance in ecosystems challenged by environmental change (De Laender et al., 2013). Given that many traits show a phylogenetic signal (i.e. close relatives have more similar trait values than distant relatives), the phylogenetic diversity of communities is also related with the functional trait space of a community, and thus with ecosystem functioning (Gravel et al., 2012; Srivastava et al., 2012; Best, Caulk, and Stachowicz, 2012; Griffin, Byrnes, and Cardinale, 2013). In addition, phylogeny determines interactions among species, and so could help predict how extinctions cascade through ecological networks and impact ecosystem functions (Srivastava et al., 2012).

Most research on biodiversity decline and ecosystem function has concentrated on primary producers (Messmer et al., 2014; Duncan, Thompson, and Pettorelli, 2015; Lefcheck et al., 2015). Biodiversity losses also include declines in the abundance of other taxonomic groups, and most extinctions in natural marine ecosystems have even been reported to occur at high trophic levels, i.e. top predators and other carnivores (Byrnes, Reynolds, and Stachowicz, 2007). Trophic composition of the predator assemblage (strict predators; intraguild predators: predators that consume other predators with which they compete for shared prey resources; or a mixture of the two) can play an important role in determining the nature of the relationship between predator diversity and ecosystem function (Finke and Denno, 2005). Griffin, Byrnes, and Cardinale (2013), for example, reported that richness effects on prey suppression in predator experiments were stronger than those for primary producers and detritivores, suggesting that relationships between richness and function may increase with trophic height in food webs. Predator diversity studies are also particularly relevant to conservation because they focus on the trophic group that is most prone to extinction, and because they nearly always measure diversity effects that span several trophic levels (Finke and Snyder, 2010). However, the magnitude and direction of these effects are highly variable and are difficult to predict since species at higher trophic levels exhibit many complex, indirect, non-additive, and behavioural interactions (Bruno and O'Connor, 2005; Bruno and Cardinale, 2008). For example, consumer diversity effects on prey and consumers strongly depend on species-specific growth and grazing rates, which may be at least equally important as consumer specialisation in driving consumer diversity effects across trophic levels (Filip et al., 2014). According to Duffy et al. (2007), the strength and sign of changes in predator diversity on plant biomass depends on the degree of omnivory and prey behaviour.

Gamfeldt, Lefcheck, and Byrnes (2014) indicated that mixtures of species generally tend to enhance levels of ecosystem function relative to the average component species in monoculture, although they may have no effect or a negative effect on functioning relative to the 'highest-performing' species. In addition to the number of species in a mixture, the structure of their interactions, therefore, needs to be accounted for to predict ecosystem productivity (Poisot, Mouquet, and Gravel, 2013). Subsequently, studies of single trophic levels are insufficient to understand the functional consequences of biodiversity decline (Thebault and Loreau, 2011; Reynolds and Bruno, 2012; Hensel and Silliman, 2013; Jabiol et al., 2013; Vaughn, 2010; Gamfeldt, Lefcheck, and Byrnes, 2014; Lefcheck et al., 2015). Community and food-web structure also influence species interactions and how species' traits are expressed, and both vertical (across trophic levels) and horizontal (within trophic levels) diversity are, hence, important (Duffy et al., 2007; Vaughn, 2010; Jabiol et al., 2013). For example, Ramus and Long (2015) demonstrated that higher marine producer (macroalgae) diversity directly increased consumer (benthos) diversity. This increased consumer diversity in turn enhanced consumer stability via increased asynchrony among consumers (i.e. species fluctuations are not in synchrony).

Stachowicz, Bruno, and Duffy (2007) concluded that multitrophic-level studies indicate that, relative to depauperate assemblages of prey species, diverse ones (a) are more resistant to top-down control, (b) use their own resources more completely, and (c) increase consumer fitness. In contrast, predator diversity can either increase or decrease the strength of top-down control because of omnivory and because interactions among predators can have positive and negative effects on herbivores (Stachowicz, Bruno, and Duffy, 2007). However, biodiversity modifications within one trophic level induced by non-random species loss (e.g. resulting from insecticide exposure) do not necessarily translate into changes in ecosystem functioning supported by other trophic levels or by the whole community in the case of limited overlap between sensitivity and functionality (Radchuk et al., 2015). Similarly, increased prey abundance may not pass up the food chain to higher trophic levels, if such prey is largely resistant to (or tolerant of) predators at these higher trophic levels (Edwards et al., 2010; Graham et al., 2015). Multitrophic interactions depend on the degree of consumer dietary generalism, trade-offs between competitive ability and resistance to predation, intraguild predation, and openness to migration (J. Duffy et al., 2007).

## 2.5.4 Random versus realistic species losses

While most studies of the relationship between biodiversity and ecosystem functioning have examined randomised diversity losses, several recent experiments have employed nested, realistic designs and found that realistic species losses may have larger consequences than random losses for ecosystem functioning (Larsen, Williams, and Kremen, 2005; Walker and Thompson, 2010; Naeem, Duffy, and Zavaleta, 2012; Bracken and Williams, 2013; Wolf and Zavaleta, 2015). According to Gross and Cardinale (2005), the difference in functional consequences of random and ordered extinctions depends on the underlying BEF mechanism:

> *"The model suggests that when resource partitioning or facilitation structures communities, the functional consequences of non-random extinction depend on the covariance between species traits and cumulative extinction risks, and the compensatory responses among survivors. Strong competition increases the difference between random and ordered extinctions, but mutualisms reduce the difference. When diversity affects function via a sampling effect, the difference between random and ordered extinction depends on the covariance between species traits and the change in the probability of being the competitive dominant caused by ordered extinction. These findings show how random assembly experiments can be combined with information about species traits to make qualitative predictions about the functional consequences of various extinction scenarios".*

Experiments with controlled (non-random) removal of species would, hence, be a good way forward to increasing our understanding of realistic species losses, although such experiments are fraught with practical obstacles and difficulties over interpretation of results (Raffaelli, 2004). In such experiments, the realistic order in which species are to be lost is determined by their susceptibilities to different types of disturbances (Solan et al., 2004; Raffaelli, 2006). Disturbance, in turn, can moderate relationships between biodiversity and ecosystem functioning by (1) increasing the chance that diversity generates unique system properties (i.e., "emergent" properties) or (2) suppressing the probability of ecological processes being controlled by a single taxon (i.e., the "selection-probability" effect) (Cardinale and Palmer, 2002). This becomes even more complex when multiple disturbances or pressures are considered. For example, Byrnes, Reynolds, and Stachowicz (2007) discussed that most extinctions (~70%) occur at high trophic levels (top predators and other carnivores), while most invasions are by species from lower trophic levels (70% macroplanktivores, deposit feeders, and detritivores). These opposing changes, thus, alter the shape of marine food webs from a trophic pyramid capped by a diverse array of predators and consumers to a shorter, squatter configuration dominated by filter feeders and scavengers (Byrnes, Reynolds, and Stachowicz, 2007). Changes in the food web with successive extinctions make it difficult to predict which species will show compensation in the future (Ives and Cardinale, 2004). This unpredictability argues for "whole-ecosystem" approaches to biodiversity conservation (implicitly incorporating the insurance hypothesis), as seemingly insignificant species may become important after other species go extinct (Ives and Cardinale, 2004).

### 2.5.5 Environmental conditions

The effects of biodiversity losses on ecosystem functions depend on the abiotic and biotic environmental conditions (Boyer, Kertesz, and Bruno, 2009; Capps, Atkinson, and Rugenski, 2015; Vaughn, 2010). Changes in water chemistry parameters (such as pH, temperature, alkalinity and water hardness), for example, may affect species life-history parameters and hence also directly or indirectly influence BEF relationships (Jesus, Martins, and Nogueira, 2014; Schweiger and Beierkuhnlein, 2014). In line with this, Boyer, Kertesz, and Bruno (2009) noted that species richness increased algal biomass production only at two of the four field sites that differed naturally in environmental conditions. de Moura Queirós et al. (2011) found that the effect of ecosystem engineers, through bioturbation, in EF was dependent on the presence of structuring vegetation, sediment granulometry and compaction. Belley and Snelgrove (2016) found evidence that environmental variables and functional diversity indices collectively explain the majority of the variation of benthic fluxes of oxygen and nutrients in soft sedimentary habitats, with both factors playing a similar role in the control of flux rates and organic matter remineralisation.

The main abiotic drivers of ecosystem functioning relevant for aquatic realms discussed by Truchy et al. (2015) include: temperature as a basic driver of metabolic processes (also Schabhüttl et al., 2013); light and nutrient availability, particularly important for primary producers (and nutrients also for decomposers); substrate composition; sediment loading, which can decrease light availability and hence limit primary production; hydrological regimes, which are fundamental organisers of temporal patterns in biotic structure and ecosystem process rates; and interactions between these various abiotic drivers. Under rapid global change, simultaneous alterations to compositional diversity and environmental conditions could have important interactive consequences for ecosystem function (Mokany et al., 2015). Despite this clear importance of abiotic condition on BEF relationships, many previously conducted BEF studies did not include testing of abiotic factors, which hampers interpretation of such study findings (Strong et al., 2015). There is, hence, a need for experimental studies that explicitly manipulate species richness and environmental factors concurrently to determine their relative impacts on key ecosystem processes such as plant litter decomposition (Boyero et al., 2014).

### 2.5.6 Spatio-temporal scale

The spatial-temporal scale of BEF evaluations has also often been indicated to influence study findings (Venail et al., 2010; McBride, Cusens, and Gillman, 2014; Vaughn, 2010; Isbell et al., 2011; Hodapp et al., 2015; Thompson, Davies, and Gonzalez, 2015). For example, strong species-identity effects at local scales can become species-richness effects at larger scales, as different species traits are favoured in different habitats (Vaughn, 2010). After evaluating 17 grassland biodiversity experiments, Isbell et al. (2011) reported that different species promoted ecosystem functioning during different years, at different places, for different functions and under different environmental change scenarios. The species needed to provide one function during multiple years were also not the same as those needed to provide multiple functions within one year (Isbell et al., 2011) and may also vary between seasons (Frainer, McKie, and Malmqvist, 2013). After studying nutrient recycling by freshwater mussels, Vaughn (2010) also concluded that this relationship was dynamic because both environmental conditions and mussel communities changed over the 15-year study period. Both the net effect of diversity and the probability of polycultures being more productive than their most productive species increases through time, because the magnitude of complementarity increases as experiments are run longer (Cardinale et al., 2007; Stachowicz et al., 2008; Reich et al., 2012). Similarly, species richness explained an increasing proportion of data variation as ecosystem processes complexity increased, and complementarity may be stronger as such complexity increases (Caliman et al., 2013).

What is now sorely needed is a new generation of experiments that target how spatial scale and heterogeneity, realistic local extinction scenarios, functional and phylogenetic composition, and other aspects of environmental change (especially temperature, acidification and pollution) influence the relationship between different dimensions of aquatic biodiversity and ecosystem functioning, under natural conditions across spatial and temporal scales (Kominoski et al., 2009; Narwani et al., 2015; Hensel and Silliman, 2013; Gamfeldt, Lefcheck, and Byrnes, 2014). Observational (i.e. correlational) field studies would provide one way forward because they do not require logistically-challenging manipulations, allowing the description of diversity-function relationships of entire sites and regions (Gamfeldt, Lefcheck, and Byrnes, 2014). Additionally, such studies would allow evaluating BEF curves likely to occur in the actual field and, hence, also aid in validating the way data and curves from experimental data are used to predict these real-world BEF relationships. However, successfully predicting linkages between biodiversity and ecosystem function requires using multiple empirical approaches across scales. Larger and consequently more complex approaches are ecologically more realistic than smaller systems (Vaughn, 2010). On the other hand, smaller-scale (experimental) approaches are easier to replicate and manipulate. Therefore, they have been proven more useful in elucidating the chain of events or evaluating a specific correlation between e.g. a certain (group of) species on a given ecosystem function.

Based on lessons learnt from previous experimental and theoretical work, Giller et al. (2004) suggested four experimental designs to address largely unresolved questions about BEF relationships: (1) investigating the effects of non-random species loss through the manipulation of the order and magnitude of such loss using dilution experiments; (2) combining factorial manipulation (i.e. including more than two patch types) of diversity in interconnected habitat patches to test the additivity of ecosystem functioning between habitats (i.e. to test whether the impact of biodiversity on ecosystem functioning in one kind of patch depends critically on biodiversity effects in another patch type); (3) disentangling the impact of local processes from the effect of ecosystem openness via factorial manipulation of the rate of recruitment and biodiversity within patches and within an available propagule pool; and (4) addressing how non-random species extinction following sequential exposure to different stressors may affect ecosystem functioning.

### 2.5.7 Trait-based evaluations

Species functional traits may provide an important link between the effect of human disturbances on community composition and diversity and their outcome for ecosystem functioning (e.g., Enquist et al., 2015; Frainer and McKie, 2015). Disturbance affects the distribution and composition of functional traits. Such shifts may, therefore, impact ecosystem functioning, particularly when traits that are crucial for ecosystem processes are impacted, but also due to changes in interaction between species (e.g. Huston, 1979; Osman, 2015).

Strong et al. (2015) evaluated the need for trait-based analysis in relation to the underlying BEF mechanism. They noted that BEF relationships underpinned by identity effects are often irregular when maintained in taxonomic (i.e. structural) biodiversity units and that such units may, hence, benefit from translation into functional diversity using traits-based analysis. For BEF relationships emerging from complementarity, direct (taxonomic) measures of biodiversity, such as species richness, may be sufficient to express the influence of biodiversity (Strong et al., 2015). Given that BEF relationships in the marine environment appear to be mostly driven by identity effects (c.f. section 2.4 above), trait-based analysis may be a promising way forward for these ecosystem types, although several constraints with such analysis have been reported, which include:

- Most studies of how biodiversity influences ecosystem function have examined single traits (e.g., the ability to break down leaves, rates of primary production), which is an oversimplification of species' roles, and very likely has led to underestimates of the impacts of species losses (Vaughn, 2010);
- The rate, efficiency or influence of a particular role is not coded within biological trait analysis, and this is understandable considering how the performance of any species can change depending on numerous factors, including age/life stage, season, abundance, habitat, community composition and environmental conditions (Reiss et al., 2009; de Moura Queirós et al., 2011; Vaughn, 2010; Frainer, McKie, and Malmqvist, 2013; Strong et al., 2015; Truchy et al., 2015);
- Efficient ways are needed to extrapolate information about key functional traits of known species to estimate the traits of poorly known species, which number in the millions, especially microbial species (Naeem, Duffy, and Zavaleta, 2012).
- Some species may be difficult to allocate to any broadly defined functional group, because they possess a high number of unique traits (Mouillot, Bellwood, et al., 2013; Truchy et al., 2015).
- Related with this, (freshwater) species are often placed into functional categories on the basis of shared autecological traits (i.e., trophic mode, behaviour, habitat, life history, morphology) that may not translate into shared ecological function. In addition, the degree of redundancy among species assigned to many of such functional groups or guilds is unknown (Vaughn, 2010).

## 2.6 Conclusions

Considering the aims of this chapter as outlined in the introduction (Section 2.1), it can be concluded that:

- Mechanisms and shape of aquatic BEF relationships are highly context-dependant, but that they appear to be best approximated by a power function;
- The shape of the power function (convex or concave) depends on the ecological function that the lost species play in the ecosystem and the likely redundancy linked with that function. As such ecosystems subject to large disturbancies are more likely to be affected by the disappearance of key species for ecosystem functioning. Thus, as biodiversity increases in highly disturbed systems, ecosystems are more likely to recover their function and become more resilient;
- A good understanding of the link between biodiversity and ecosystem functions is critical as it might determine management approaches that promote ecosystem resilience and adaptability essential to the delivery of ecosystem services;
- Species composition, in addition to species richness, is likely to also be very important, as ecosystem functions are very dependent on the role played by each species; as increasing biodiversity is likely to increase resilience and ESS delivery;
- Although a striking level of generality in diversity effects across terrestrial, freshwater, and marine ecosystems have been reported, BEF relationships may not directly extrapolate across ecosystem types due to intrinsic system-specific characteristics;
- Despite considerable research efforts and progress into BEF relations in the past decades, several research limitations and gaps still exist;
- Depending on the specific research question that is tackled, both observational and experimental studies may increase our understanding of BEF relationships.

# 3 Biodiversity-Ecosystem Services Relationships

The present chapter provides an overview of the information deducted so far from the literature review on BES relationships.

## 3.1 Introduction

Physical, chemical, and biological watershed processes are the foundation for many services that ecosystems provide to human societies (Villamagna and Angermeier, 2015). Since the composition of species communities is changing rapidly through pressures and impacts such as habitat loss and climate change, potentially serious consequences for the resilience of ecosystem functions on which humans depend may ensue (Oliver et al., 2015).

As discussed in detail in the previous chapter, there is now a firm evidence base demonstrating the importance of biodiversity to ecosystem functioning. However, there is less research available into whether biodiversity has the same pivotal role for ecosystem services, and hence whether protection of ecosystem services will protect biodiversity, and vice versa (Harrison et al., 2014; Bennett et al., 2015). Balvanera et al. (2014), for example, examined whether biodiversity, measured as species richness, drives ecosystem services supply for three provisioning services: forage, timber, fisheries; and three regulating services: climate regulation, regulation of agricultural pests and water quality. They cautioned that, while a positive link between biodiversity and ecosystem functioning (BEF) is now strongly supported, there is less evidence of a clear relationship between biodiversity and ecosystem services (BES) (Balvanera et al., 2014). Until present, it has therefore been challenging to turn the concept of ecosystem services into a practical conservation tool in the formulation of day-to-day policies on a national or regional scale (Burkhard et al., 2014; Cook, Fletcher, and Kelble, 2014; Heink et al., 2016; Mononen et al., 2015).

The aim of the present chapter is to provide a preliminary overview of existing knowledge on causal links between biodiversity and ecosystem services and aspects that need to be considered to operationalise BES.

## 3.2 Established biodiversity-ecosystem services relationships

Maes et al. (2012) mapped four provisioning services, five regulating services and one cultural service across Europe, and found that these tended to be positively correlated with biodiversity, although they noted that this relationship was affected by trade-offs, thus resulting in poorer correlations, in particular between the provisioning service of crop production and regulating services. For the regulating service water purification, Balvanera et al. (2014) summarised 59 experiments, showing that in 86% of the studies—spread across terrestrial, freshwater and marine ecosystems—increased species richness reduced nitrogen concentrations in water or soil.

The key role of biodiversity for regulating services has also been verified by other research (Mace, Norris, and Fitter, 2012; Harrison et al., 2014). For example, experiments have shown that bioremediation of contaminated groundwater and marine sediments is faster and more effective when bacterial biodiversity is higher (Dell'Anno et al., 2012; Marzorati et al., 2010). Harrison et al. (2014) conducted a systematic literature review to analyse the linkages between different biodiversity attributes and 11 ecosystem services. Although the majority of relationships were positive, biodiversity appeared to be negatively correlated with freshwater provision. This could be explained through increased water consumption resulting from increases in community/habitat area, structure, stem density, aboveground biomass and age increased water consumption and, hence, reduced the provision of this ecosystem service (Harrison et al., 2014). The review also showed that ecosystem services are generated from numerous interactions occurring in complex systems. Evidences and recent progresses in the field of systems ecology show, for example, that "hierarchical organization has an important damping effect in the higher levels on disturbances occurring in the lower levels and that the damping effect increases with increasing biodiversity" (Jørgensen, Nielsen, and Fath, 2016). Biodiversity may have a similarly complex role when it comes to its effect in ESS and, therefore, it is not straightforward to establish BD–ESS relationships. Improving understanding of at least some of the key relationships between biodiversity and service provision will help guide effective management and protection strategies (Harrison et al., 2014). However, a recent review (Ricketts et al., 2016) suggests that this task might not be that straightforward, as BD–ESS relationships seem to differ among ESS, and to depend on methods of measuring biodiversity and ESS, and on approaches to link them (spatially, management linkage, and functional linkage).

The difficulty in understanding the role played by BD in ESS is also due to the direct and/or indirect effects that BD can have in ESS provisioning: from regulator role (e.g., wetlands: hydrological cycle, carbon cycle), to supplier role (e.g., wetlands: drinking water), or as a good itself (e.g., wetlands: wood from mangroves; rice from rice fields) (Mace, Norris, and Fitter, 2012; Pascual, Miñana, and Giacomello, 2016). An example of a direct link is the demonstrated greater stability of fisheries yields when fish biodiversity increases (Cardinale et al., 2012). Indirect effects of biodiversity on ecosystem services act through interaction with ecosystem functioning and will, hence, both depend on as well as influence the abiotic state. For example, losses of algal diversity may affect the EF primary production and subsequently the regulating ESS carbon sequestration (Cardinale et al., 2012; Truchy et al., 2015). Regarding the latter, Duncan, Thompson and Pettorelli (2015) reviewed commonly studied ESS and the underlying EFs and main contributing trophic levels responsible for their delivery. Despite acknowledgements of a need for BES research to look towards underlying

BD-EF linkages, the connections between these areas of research remains weak (Duncan, Thompson, and Pettorelli, 2015).

Accounting for the relationships between ESS is also crucial to minimise undesired trade-offs and enhance synergies, as showed by Lee and Lautenbach (2016). These authors found sound evidence that synergistic relationships dominated within different regulating services and within different cultural services, whereas regulating and provisioning services often implied trade-off relationships. The increase of cultural services showed no evidence of affecting provisioning services (Lee and Lautenbach, 2016).

## 3.3 What is hampering establishing BES relationships

Despite a wealth of studies into biodiversity's role in maintaining ESS (BES relationships) across landscapes, we still lack generalities in the nature and strengths of these linkages, besides that they are unlikely to be linear (Barbier et al., 2008; Pinto and Marques, 2015). For example, often, an optimal ESS delivery may benefit from the integration of development (demand and supply of ESS) and biodiversity conservation, attaining to EBM goals (Barbier et al., 2008). Reasons for lack of stronger evidences are manifold, but can largely be attributed to (i) a lack of adherence to definitions and thus a confusion between final ESS and the EFs underpinning them, (ii) a focus on uninformative biodiversity indices and singular hypotheses and (iii) top-down analyses across large spatial scales and overlooking of context-dependency (Duncan, Thompson, and Pettorelli, 2015). In more detail, reported constraints in establishing BES links include: 1) dealing with multiple interconnected ESS and activities; 2) spatial-temporal scale; 3) type of ESS considered; 4) influence of climate change; 5) considering social-ecological systems, stakeholders and demand side; and 6) selection of relevant indicators.

### 3.3.1 Dealing with multiple interconnected ESS and human activities

Multiple interconnected ESS might result from the capacity of an ecosystem to support the joint-production of ESS that provide joint products or multiple benefits (Fisher, Turner, and Morling, 2009). This joint-production is a characteristic of ESS that results from the capacity of an ecosystem to deliver several services or the capacity of a service to provide several benefits. A relevant example to illustrate this concept of joint products or multiple benefits, is provided by wetlands, as they provide water for human consumption (provisioning service), regulate water cycle and mediate water quality (regulating services) and provide recreation opportunities (cultural services).

BES studies that have considered the direct influence of BD for only one ESS, only over a short time period, or without any influence of global change, are likely to underestimate its importance (Science for Environmental Policy, 2015). Indeed, evidence is now mounting to show that greater biodiversity is needed to maintain multiple ESS in the long term and under environmental change (Balvanera et al., 2014; Cardinale et al., 2012; Isbell et al., 2011; Naeem, Duffy, and Zavaleta, 2012). In addition, research is needed on the impacts to ESS from multiple human activities and their associated stressors ('impact-pathways'). In most cases, human actions to harvest ESS are likely to affect biodiversity (and hence potentially ecosystem services) negatively. For example, an integral part of agricultural intensification at the plot level is the deliberate reduction of diversity (Swift, Izac, and van Noordwijk, 2004). In other cases, there may be synergies, such as flood protection increasing soil quality, habitat provision, space for water and recreation (Rouquette et al., 2011).

Furthermore, human actions can also be translated into the production of ESS together with their social and ecological environment, named as co-production of ESS (e.g., Fischer and Eastwood, 2016). These authors specifically distinguish between three types of human contributions to ESS: i) the co-production of ecosystems structures, like artificial reefs or constructed wetlands; ii) the co-production of benefits, by producing something of use for themselves or others, such pieces of art or scientific knowledge; and iii) the the attribution of meaning to a service or benefit, apart from the tangible production of benefits, like the sense of place. The co-production of ESS, as consider by Fischer and Eastwood (2016), can also lead to aditional undesired disservices, namely unpleasant landscape resulting from the the co-production of ecosystems structures.

It is also important to integrate the history of ESS and their change over time, as well as understanding multi-relationships between ESS, since this can offer opportunities to foster synergies and avoid unnecessary trade-offs (Lee and Lautenbach, 2016; Tomscha and Gergel, 2016). Multi-activity trade-off evaluation and management will require a concerted effort to structure ecosystem-based research around impact-pathways (Mach, Martone, and Chan, 2015). This should include evaluating trade-offs between (i) a good ecological state or biodiversity, (ii) maximising provision of ESS, and (iii) low costs (Gómez et al., 2016a). There are several quantitative methods that come in hand for assessing such ESS associations, applicable to the identification and the understanding of supply-supply (i.e. simultaneously provided ESS), supply-demand (i.e. how stakeholders benefit from the ESS delivery), or demand-demand (i.e. interactions between stakeholders' needs) aspects, but also for the identification of drivers of ESS bundles (Mouchet et al., 2014).

Jopke et al. (2015) uncovered complex interactions between ESS using geographical analyses for attempting to optimise multiple ESS simultaneously. Similarly to Lee and Lautenbach (2016), they also found evidence that interactions of ESS occur in characteristic patterns, e.g., with trade-offs among agricultural production (i.e. provisioning) and regulating services. It is expected that similar patterns could occur for interactions of ESS pairs and bundles; however, synergies or trade-offs might also depend on whether the ESS analised share a common driver or location (Lee and Lautenbach, 2016; Jopke et al., 2015). Howe et al. (2014) found three significant indicators that a trade-off would occur: a private interest in the natural resources available, the involvement of provisioning ESS, and stakeholders acting at local scale. Their study suggests that accounting for why trade-offs occur (e.g., from failures in management or a lack of accounting for all stakeholders) is more likely to lead to synergies in the end.

### 3.3.2 Spatio-temporal scale

Studies relating biodiversity to ESS often focus on services at small spatial or short temporal scales, but research on the protection of services is often directed toward services providing benefits at large spatial scales (Howe et al., 2014; Birkhofer et al., 2015). AQUACROSS seeks to expand current knowledge and foster the practical application of EBM and, hence, BES for all aquatic (freshwater, coastal, and marine) ecosystems as a continuum. The meta-ecosystem concept provides a powerful theoretical tool to understand the emergent properties that arise from spatial coupling of local ecosystems, such as global source-sink constraints, diversity-productivity patterns, stabilisation of ecosystem processes and indirect interactions at landscape or regional scales (Loreau, Mouquet, and Holt, 2003). In this regard, a meta-ecosystem is defined as a set of ecosystems connected by spatial flows of energy, materials and organisms across ecosystem boundaries (Loreau, Mouquet, and Holt, 2003).

### 3.3.3 Type of ESS considered

Case studies often focus on provisioning as opposed to non-provisioning services (Howe et al., 2014). However, the significance of protecting regulating services and the biodiversity that underpins them should not be underestimated, as many other ESS are dependent upon them (Harrison et al., 2014; Science for Environmental Policy, 2015).

### 3.3.4 Influence of climate change

Much ecosystem monitoring and management is focused on the provision of ecosystem functions and services under current environmental conditions. Yet this could lead to inappropriate management guidance and undervaluation of the importance of biodiversity. The maintenance of EFs and ESS under substantial predicted future environmental change (i.e., their 'resilience') is crucial (Pedrono et al., 2015; Oliver et al., 2015).

### 3.3.5 Considering social-ecological systems, stakeholders and demand side

According to Bennett et al. (2015), answering three key questions will improve incorporation of ESS research into decision-making for the sustainable use of natural resources to improve human well-being: (i) how are ESS co-produced by social-ecological systems (SES), (ii) who benefits from the provision of ESS, and (iii) what are the best practices for the governance of ESS, considering both the supply- and the demand-sides (Mouchet et al., 2014; Balvanera et al., 2014; Bennett et al., 2015)? Acknowledging the role that both the ecological and the social-economic systems play in the provisioning of ESS, Mouchet et al. (2014) emphasise the importance of extending the analysis of these complex relationships beyond the trade-offs and synergies in simultaneously provided ESS (supply-supply), to include also how stakeholders can benefit from the ESS delivery (supply-demand), and also the interactions between stakeholders' needs (demand-demand, i.e. referring to the arbitration between different and divergent stakeholders' interests).

### 3.3.6 Selection of relevant indicators

A major challenge in operationalising ESS is the selection of scientifically defensible, policy-relevant and widely accepted indicators (Heink et al., 2016). For example, an analysis of the Marine Strategy Framework Directive (MSFD) revealed ambiguity in the use of terms, such as indicator, impact and habitat, and considerable overlap of indicators assigned to various descriptors and criteria (Berg et al., 2015). Hattam et al. (2015) highlighted some of the difficulties faced in selecting meaningful indicators, such as problems of specificity, spatial disconnect between the service providing area and the service benefiting area and the considerable uncertainty about marine species, habitats and the processes, functions and services they contribute to.

Despite that there are currently many monitoring programmes for biodiversity in aquatic systems, the extent to which they can provide data for ESS indicators is still not clear. Liquete et al. (2016) point out that, for an effective quantification of the link between biodiversity and ESS, the analysis of the delivery of ESS should be differentiated from the analysis of ecological integrity. Subsequently, an important challenge that has to be dealt with in AQUACROSS is the definition of relevant indicators for ESS in aquatic realms, within its conceptual Assessment Framework (Gómez et al., 2016b; see Section 5 herein).

## 3.4 Methodological challenges

Despite the fact that a surplus of methods and frameworks have been reported in the literature (Borja et al., 2016; Truchy et al., 2015), Villa et al. (2014) discussed that on the research side, mainstream methods for ESS assessment still fall short of addressing the complex, multi-scale biophysical and socio-economic dynamics inherent in ESS provision, flow, and use. Establishing BES relationships is challenging, because the multiple disciplines involved when characterising such links have very different approaches (common-language challenge). Additionally, they span many organisational levels and temporal and spatial scales (scale challenge) that define the relevant interacting entities (interaction challenge) (The QUINTESSENCE Consortium, 2016).[7] On the user side, application of methods remains onerous due to data and model parameterisation requirements. Further, it is increasingly clear that the dominant "one model fits all" paradigm is often ill-suited to address the

[7] The QUINTESSENCE Consortium aims at promoting a more unified framework for dealing with ecosystems services within research and management.

diversity of real-world management situations that exist across the broad spectrum of coupled human-natural systems (Villa et al., 2014). Network approaches are also a promising method for interdisciplinary research aimed at understanding and predicting ESS (The QUINTESSENCE Consortium, 2016). The choice of methods used to determine BES relationships is not a trivial aspect, as more studies indicate that it may influence detection and/or affect the direction of the relationships found (e.g. Lee and Lautenbach, 2016; Ricketts et al., 2016).

To foster the use of the empirical knowledge gathered in the last years, several authors support the development of broad registers of evidence on BES relationships (Ricketts et al., 2016). A good example is the database assembled by Pascual, Miñana, and Giacomello (2016) integrating available research results on BD-EF-ESS-Human well-being relationships, to support Bayesian Network modelling and scenarios development, accounting for uncertainty, in support of better informed decision-making processes.

The limitations and methodological challenges previously outlined will need to be addressed in AQUACROSS. Its several case studies may eventually shed light on the general applicability and adaptability of the overall proposals of the present report. Methods selected will need to be flexible, and adhere to the Assessment Framework (AF) developed under AQUACROSS (Gómez et al., 2016a,b). Eventually, the AF may be adapted based on lessons learnt from its application in the case studies.

Finally, a general preference for assessing ESS at terrestrial ecosystems as opposed to marine, coastal and freshwater ecosystems, is evident from the literature (Pascual, Miñana, and Giacomello, 2016). This emphasises the potential for AQUACROSS research to contribute meaningfully to the advances in this field of research.

## 3.5 Conclusions

Considering the aims of this chapter as outlined in the introduction (Section 3.1), it can be concluded that:

- A good understanding on how BD underpins ESS is of paramount importance, allowing decision-makers to consider the demand for ESS, the capacity of ecosystems to provide them and the pressures disabling directly or undirectly that capacity;
- BD is generally correlated with ESS, either positively or negatively depending on the type of the ESS, although the strength of the correlation might be reduced by the existence of trade-offs between ESS;
- The methods of measuring BD will affect the assessment of BD and ESS relationships;
- Indirect effects of BD on ESS will also act through interaction with EF, that is also dependent on the influence of the abiotic state;

- To minimise undesired trade-offs and enhance synergies between ESS, it is crucial to account for their spatial and temporal nature, as well as the management options that will condition those relationships;
- Although, a mismatch regarding ESS provided through joint-production (ESS that provide joint products or multiple benefits) or through co-production (human contributions to ESS) might occur, both concepts should be clearly defined and considered when dealing with interconnected ESS and human activities.
- The selection of indicators within the AF of AQUACROSS should take in consideration that the analysis of the delivery of ESS should be differentiated from the analysis of ecological integrity;
- Despite the advances in understanding and assessing BD and ESS relationships, the application of methods to address them remains onerous due to data and model parameterisation requirements;
- The complexity and broad spectrum of coupled human-natural systems relationships challenges the dominant "one model fits all" paradigm, making the choice of methods used to determine BD and ESS relationships context-dependent.

# 4 Evidence from Meta-analysis on BEF and BES Relationships

## 4.1 Introduction

The application of meta-analysis to ecological data, combining experimental data to test general hypotheses in ecology, emerged in the early 1990s (Hedges, Gurevitch, and Curtis, 1999). Meta-analyses integrates quantitative data presenting the “bigger picture” in terms of hypothesis testing, that is, meta-analyses allow data to be collected from a large number of publications, sites, taxa, etc., and permit the presentation of analysis in a standardised metric. Meta-analyses are a powerful approach for statistically testing hypotheses linked with multi-scale spatial and temporal patterns of dynamic populations, communities, and ecosystems (Cadotte, Mehrkens, and Menge, 2012).

Meta-analysis and validation of modelling approaches based on existing data, provided that they carefully consider the aspects discussed in the present report (spatio-temporal scale, number of EFs considered in the studies used, etc.), appear to be a good way forward to enable operationalising BEF research.

## 4.2 Meta-analyses of BEF and BES relationships

Early BEF syntheses were based on expert opinions or qualitative summaries and interpretation of data, which resulted in inconsistent conclusions, forcing researchers to confront their hypotheses with more quantitative forms of analyses (Cardinale et al., 2011; Naeem, Duffy, and Zavaleta, 2012). In the past decade, several meta-analyses on data obtained from manipulative experimental BEF experiments have been conducted to attain evidence for BEF relationships (Balvanera et al., 2006; Worm et al., 2006; Stachowicz, Bruno, and Duffy, 2007; Schmid, Pfisterer, and Balvanera, 2009; Cardinale et al., 2011; Reich et al., 2012; Mora, Danovaro, and Loreau, 2014). Since BEF evidence is mainly based on experimental studies, it has been debated in recent years as to whether these results are transferable to natural ecosystems; even more since BEF relationships may be different under both conditions. To date, only a few studies have addressed the challenge of validating experimentally derived theories with data from natural aquatic ecosystems (Duffy, 2009; Hodapp et al., 2015; Thompson, Davies, and Gonzalez, 2015). The development and application of integrated models of composition and function in natural ecosystems face a number of important challenges, including biological data limitations, system knowledge and computational constraints (Mokany, Ferrier, et al., 2015). For example, due to the multivariate nature of most ecological data, the methodology applied to assess fundamental mechanisms must accommodate the multivariate nature of these dependencies, as well as direct and indirect influences, e.g., by using structural equation models (SEMs) (Cardinale, Bennett, et al., 2009; Hodapp et al., 2015).

Integrated models could highlight priorities for the collection of new empirical data, identify gaps in our existing theories of how ecosystems work, help develop new concepts for how biodiversity composition and ecosystem function interact, and allow predicting BEF relations and its drivers at larger scales (Balvanera et al., 2014; Fung et al., 2015; Queirós et al., 2015; Strong et al., 2015; Mokany, Ferrier, et al., 2015). Integrated models are models which simulate and project simultaneous changes in biodiversity composition and EF over space and time for large regions, incorporating interactions between composition and function (Mokany, Thomson, et al. 2015). Such models could also form components within larger 'integrated assessment models', improving consideration of feedbacks between natural and socioeconomic systems (Mokany, Ferrier, et al. 2015). Ultimately this would aim at better informed management, as seen in the framework underlying the Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES) (Diaz et al., 2015).

Meta-analyses can be used to provide an integrated view of dispersed experiments that can be used to test a given hypothesis. Numerous examples of meta-analyses can be found in the literature involving different aspect of the causal flows involved in the chain off processes-BD-EF-ESS-benefits (see Annex II). An example of outputs from such analyses it is illustrated in the next section.

## 4.3 Example of some outputs from a meta-analysis involving BEF relationships

Griffin, Byrnes, and Cardinale (2013) tested the effect of predator richness on prey suppression using meta-analysis. Although their work focus only at one trophic level, predators in relation with their preys (usually herbivores), the supplementary information provided allow us to extend their approach to the underlying trophic levels (herbivores, producers and decomposers). For each experiment considered, the densities of prey/plants/nutrients/detritus (abundance per area or volume), reported in single-species treatments (monocultures), and the highest predator/herbivore/producer/detritivores richness treatment (polycultures), at the final time point of experiments (to maximise the potential for treatments of varying diversity levels to diverge), were considered. These pairs of values were used to calculate two metrics (log-response ratios) of the predator/herbivore/producer/detritivores richness effect on prey/plants/nutrients/detritus suppression.

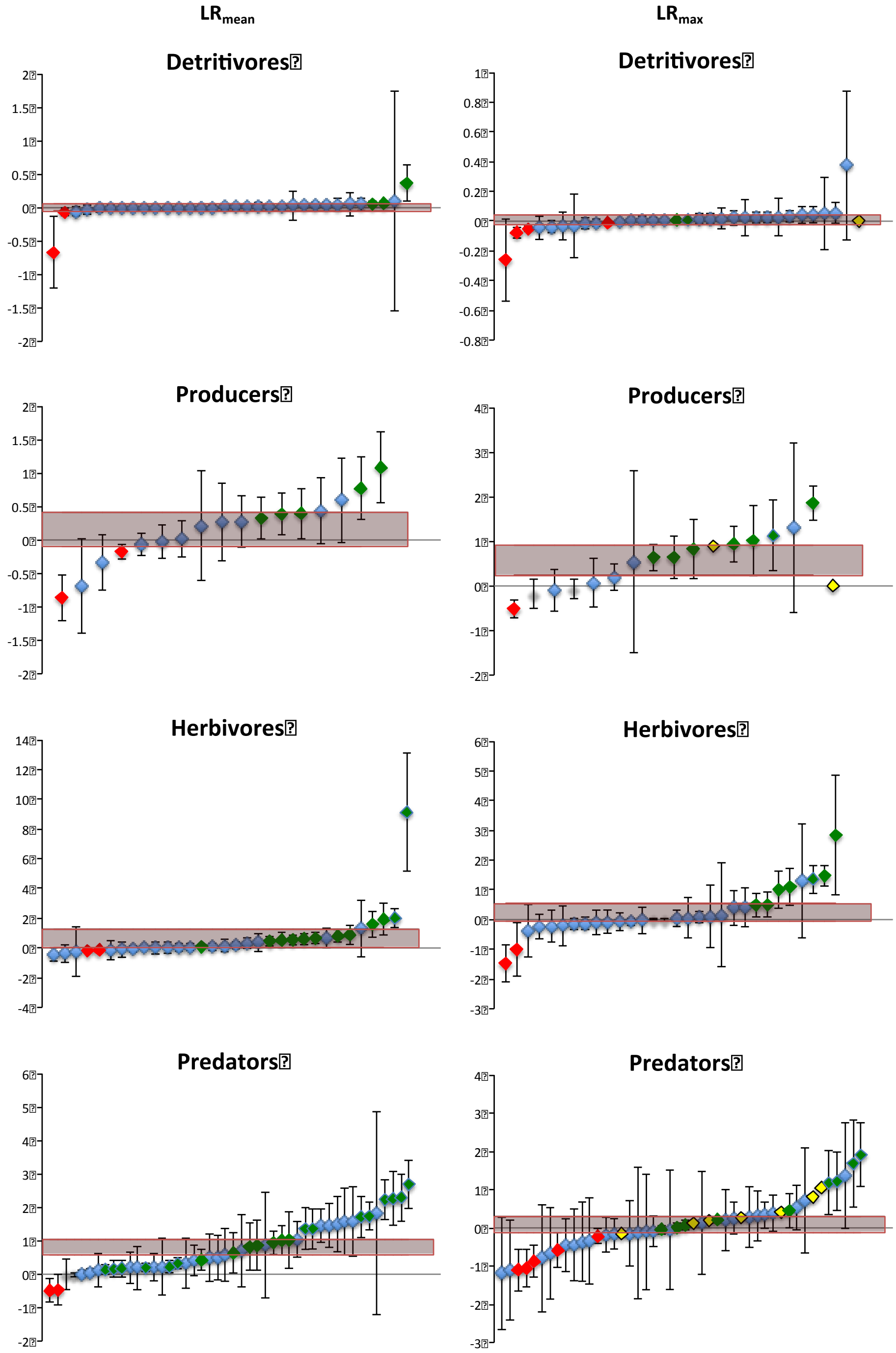
The first of these log-response ratios quantifies the mean richness effect ($LR_{mean}$) and measures whether the most species rich predator/herbivore/producer/detritivores mixture suppresses prey/plants/nutrients/detritus to a lesser or greater degree than the average of its component species in monoculture. The second log ratio, $LR_{max}$, gauges the performance of the polycultures relative to the predator/herbivore/producer/detritivores species that is most effective at suppressing prey/plants/nutrients/detritus (i.e., highest efficiency). These metrics were both reflected (multiplied by -1) to convert from measures of effects on final predator/herbivore/producer/detritivores density (the common response reported in studies) to effects on the level of prey/plants/nutrients/detritus suppression achieved by a predator/herbivore/producer/detritivores group. This meant that positive effects could be interpreted more intuitively as a positive effect of diversity on the magnitude of the aggregate process of interest (see Griffin, Byrnes, and Cardinale, 2013 for further details on the calculations involved). Results from an analysis involving the four trophic levels: predators, herbivores, producers, detritivores are depicted in Figure 3.

Individual experiments showed significant positive effects of predator richness on prey suppression in more than half of the cases, no significant effect in less than half of the cases, and significant negative effects in just 2 out of 46 cases. On average, species rich mixtures of predators suppress prey densities to a greater degree than their component species do alone, as 95% confidence interval for the grand mean indicates as it is located mainly to the positive part of $LR_{mean}$. Among individual experiments, there was a predominance of non-significant effects on $LR_{max}$ (28 of 40 effect sizes), with positive and negative significant effects equally rare (6 of 40 for each). Relative to the best-performing single species, that is, the predator species that reduces prey populations to the lowest level, diverse mixtures of predators were equivalent to the most efficient single predator species at suppressing prey, as shown by the 95% confidence interval for the grand mean. However, when we consider lower trophic levels although the number of significant negative effects continues to be very low (2 of 32 for herbivores, 2 of 17 for producers, and 2 of 32 for detritivores) the number of significant positive effects decreases with trophic level.

Predators seem to have a greater impact on their resources than lower trophic levels. Predator richness did not, however, strengthen prey suppression relative to the single most effective species ($LR_{max}$), perhaps implying that as long as the single most efficient predator is conserved, losses of predator richness may not affect prey suppression. However, the absence of a so-called "transgressive overyielding" effect should be interpreted cautiously. $LR_{mean}$ of predators are stronger than those of both plant richness and decomposer richness, indicating that species losses may have the strongest effects at higher trophic levels, where they are thought to most likely occur, as previously predicted. These results do not completely agree with results from Cardinale et al. (2006), as more studies were included in Griffin, Byrnes, and Cardinale (2013). Although a meta-analysis with more studies might imply higher variability in the results it makes the analysis more robust.

In conclusion, meta-analyses are important tools to analyse response patterns linked with BEF relationships measured in observational or experimental studies in order to derive, when possible, general rules. Nevertheless, the outcomes of this type of analysis is highly dependent on the number of studies involved and the trophic level considered. As previously noted, as we move towards upper trophic levels the impact of BD becomes more pronounced and relevant. Outcomes of meta-analyses will facilitate the establishement of suitable models to address BEF relationships or help identify situations where these relationships might be case-dependent.

## Figure 3: Effects of richness in upper trophic level on suppression of lower trophic level




Legend: (a) mean richness effect (log-response ratio, $LR_{mean}$) and (b) relative to best-performing individual species (log-response ratio, $LR_{max}$). Studies are arranged in order of effect size. Effects from each experiment are color-coded: negative effect (red), no effect (non significant, cyan), and positive effect (green). Yellow points indicate that confidence interval (and therefore statistical significance) could not be established. The shaded pink areas show the 95% confidence intervals of the grand means of each biodiversity effect.

Source: Graphs generated from supplementary data published by Griffin, Byrnes, and Cardinale (2013).

# 5 Indicators for Biodiversity, Ecosystem Functions and Ecosystem Services

As part of the development of an EBM operational assessment framework (AF), the social-ecological system needs to be deconstructed into a set of component parts (Elliott, 2011; Smith et al., 2016). Such a framework allows categorising a problem domain along the cause-effect chain (Patrício, Elliott, et al., 2016). In previous AQUACROSS deliverables (see Gómez et al., 2016a,b), we have identified and defined the key points and links within the SES that are relevant for the stages of implementation of the AQUACROSS AF presented in this document (Figure 2).

The AQUACROSS AF evolves from the traditional Drivers-Pressures-State-Impact-Response (DPSIR) cycle by explicitly considering ecosystem functions and services, human well-being, and both social and ecological processes (Gómez et al. 2016a). In this report, we focus on how ecosystems are linked to human welfare and, hence, in the main adaptations made by the AQUACROSS AF to the State-Impact stages of the DPSIR framework. The AQUACROSS AF approach allows better capturing the complex links between BD (as captured by measures of BD and ecosystem status) and the ecological processes ensuring crucial EF that enable the supply of ESS. Since these themes (i.e., Biodiversity - BD, Ecosystem Functioning - EF, and Ecosystem Services - ESS) are central to the stage of the AQUACROSS AF dealt within this report, a clear agreement on a definition of what an ESS is, and how this relates to EF and its BD components is required to allow the selection of appropriate and differentiated indicators (Böhnke-Henrichs et al., 2013; Liquete et al., 2013; Smith et al., 2016). Classification methods (next sections) applied to each of those compartments (i.e., BD, EF, and ESS) facilitate establishing links between each other, while the adoption of indicators will enable quantification of causal links along this BD-EF-ESS cascade. This means that indicators should be as stage-specific as possible and facilitate an articulated flow between the stages of an assessment framework, clarifying links, ideally allowing quantitative assessments, while avoiding overlap and double counting.

One of the advantages of having a set of indicators is that they aid organising the type of information needed for the assessment, and also allow quantifying the relationships between the different components and the flows across the AF (Gómez et al. 2016b). Indicators can also provide insight into variations in resilience by reporting e.g. on ecosystem recovery rates after disturbance (Lambert et al., 2014; Rossberg et al., 2017). This, in turn, can be used to assess the sustainability of human activities' impacts and support the development of appropriate management strategies (Lambert et al., 2014; Lillebø et al., 2016).

However, the complexity of the ecological systems, where structure and processes will combine in a myriad of ways to perform EF and to secure ESS supply, makes the selection of indicators a difficult process in practice (e.g. Maes et al., 2014; Lillebø et al., 2016; Liquete et al., 2016).

This report aims at providing guidance for selecting biodiversity components, ecological functions and ESS and respective indicators in ways that the assessment reflects the complexity of social-ecological interactions (Gómez et al., 2016a; Saunders and Luck, 2016) (Section 5.1). It is also crucial that the selection of indicators at this stage should be integrated and in line with relevant processes identified in the preceding stages of the AF (i.e., Drivers and Pressures; see Deliverable 4.1), in order to achieve a meaningful selection of ecosystem components and associated indicators and ensure a successful flow of information (see considerations under Section 5.2).

## 5.1 Classifications and indicators selection

Potential lists of indicators, indices and associated metrics, have been elaborated accounting for indicators outlined by key legislation identified in the project (see Deliverable 2.2 by O'Higgins et al., 2016) and identified in relevant scientific literature. For each main theme in the supply-side of the AQUACROSS AF (Figure 2) (i.e. BD, EF, and ESS, both ESS supply and ESS demand) the possible sources and examples of indicators are provided as an Annex. However, these are not intended to be prescriptive lists and each case study should select the indicators deemed most adequate for the context and purpose of study (i.e. the aquatic realm, the ecosystem features, the scale(s) of study, the identified pressure(s), the ESS being scrutinized; also see Section 5.2).

This guidance aims also at promoting consistency throughout the case studies, such that a standardized approach may ultimately allow a comparison of BEF and BES relations identified across aquatic realms, contributing to understand whether they are interchangeable or ecosystem-specific.

To operationalise this, the guidance focuses on:

- Defining comprehensive **classifications** (and developing relevant subcategories) pertinent for aquatic ecosystems, within each main theme: i.e. BD, EF and ESS, since such subcategories will allow building meaningful causal networks between the different components of the framework. The classification systems will be tailored to AQUACROSS needs, either by building on scattered approaches (as for BD and ecosystem state assessment), or by developing new ones (as in the case of EF), or by adapting existing ones (as the CICES ESS classification enlarged to accommodate abiotic outputs). *See Sections 5.1.2 to 5.1.4*
- Providing lists of **indicators**, and/or sources of indicators, and allocate indicators within each theme classification (i.e. BD, EF and ESS) and respective subcategories; preliminary

lists of indicators for BD, EF and ESS, structured into meaningful categories, as described in the following sections. *See Annex I*

- Identifying **criteria for the selection** of good indicators, relevant within each theme, and setting a *de minimum* approach to be applied across case studies. *See next section 5.1.1*
- Providing recommendations for applying a holistic approach to the BD-EF-ESS, accounting for interactions, synergies, and trade-offs, when identifying causal links. *See Sections 2 and 3.*

### Box 2: Definition of Indicators, Index, Metric and Measure within AQUACROSS

It is important to clarify how the concept of **indicator**, and the related terms **index**, **metric** and **measure** are understood and used within this document.

The term **measure** refers to a value measured against standardized units. A measure of something does not necessarily indicate something useful.

The term **metric** refers to a quantitative, a calculated or a composite measure based upon two or more measures. Metrics help to put a variable in relation to one or more other dimensions.

The term **index** refers to a metric whose final outcome should be easily interpreted by a non-specialist within a qualitative continuum. It can be a quantitative or qualitative expression of a specific component or process, to which it is possible to associate targets and to identify trends, and which can be mapped. It is how an indicator becomes an operational tool used within a management, regulatory or policy context.

The term *indicator* refers to a variable that provides aggregated information on certain phenomena, acting as a communication tool that facilitates a simplification of a complex process. It relates to the component or process responsive to changes in the social-ecological system, but does not possess a measurable dimension. Therefore it is not an operational tool in itself.

*An example of the use of the terminology above mentioned could be*:

Biogenic structures (such as coral reefs) are good *indicators* of seafloor integrity, for which specific *metrics* (e.g. biotic cover (%), maximum height) that can describe their features and are sensitive to pressures, need to be identified and incorporated into *indices* that allow evaluating their status and tracking progress in space and time.

## 5.1.1 Criteria for selecting indicators

Having a list of indicators, as comprehensive it may be, *per se* does not ensure a coherent evaluation of how the ecosystem state and functioning converge to secure the supply of ESS. In this sense, the tables in the following sections (5.1.2 to 5.1.4) provide guidance for selecting biodiversity components, EF and ESS, and how to link specific indicators to these proposed classifications (table in Annex) in ways that the case study assessments are able to integrate and reflect the complexity of social-ecological interactions (Gómez et al., 2016a,b; Saunders and Luck, 2016).

The selection of sound and relevant indicators has been the topic of prolific research, with several established criteria for identifying and testing the quality of indicators largely recognised as essential for building more robust assessments (Heink et al., 2016). Here, we point to the recent review and framework for testing the quality of indicators proposed by

Queirós et al. (2016) as a practical tool to guide the identification, comparison and selection of relevant, scientifically robust, cost-effective and sound indicators. This framework provides a scoring system that may be used as a basis to set minimum standards (i.e. quality criteria) for indicators.

Within AQUACROSS, it could be used, for example, for (a) selecting mandatory criteria that indicators need to fulfil or (b) agreeing on a minimum quality score to be achieved by an indicator before its use in case studies' assessments. In this sense, the AQUACROSS partners could select those criteria more relevant for the supply-side stages of the AQUACROSS AF (Figure 2), using them to set minimum quality standards across the eight case studies. Criteria cover aspects from *scientific basis* to *ecosystem relevance*, to *target setting*, to *cost-efficiency*, just to name a few (Queirós et al., 2016).

### 5.1.2 Biodiversity classifications

As introduced in Deliverable 3.2 of the AF (Chapter 2.5 in Gómez et al., 2016b), BD has an inherent multidimensional nature, spanning genes and species, functional forms, habitats and ecosystems, as well as the variability within and between them (Gonçalves et al., 2015; Laurila-Pant et al., 2015). Often regarded as a measure of the complexity of a biological system (Farnsworth, Lyashevska, and Fung, 2012; Farnsworth, Nelson, and Gershenson, 2013), BD is usually taken to be an abstract ecological concept (Bartkowski, Lienhoop, and Hansjürgens, 2015). Since preventing the loss of BD is increasingly becoming one of the important aims of environmental management, biodiversity must be understood and defined in an operational way (Laurila-Pant et al., 2015).

Farnsworth, Adenuga, and de Groot (2015) have defined BD as the information required to fully describe or reproduce a living complex ecological system; acknowledging like many others that, though a definition might be precise and 'concrete', it is still technically very demanding to calculate in practice (Bartkowski, Lienhoop, and Hansjürgens, 2015; Jørgensen, Nielsen, and Fath, 2016). To add complexity, all the dimensions of BD are tightly interconnected, affecting the state and functioning of the ecosystem as well as the ESS (Laurila-Pant et al., 2015). Ecosystems are complex functional units, encompassing not only the biotic and abiotic components of the environment (i.e., the biophysical environment), but their ecology, i.e. how living organisms interact with each other and with the surrounding environment. To offer a consistent theory about EF, a recent ecological sub-discipline has developed - systems ecology (Jørgensen, Nielsen, and Fath, 2016) that builds on the four pillars: (1) hierarchy, (2) thermodynamics, (3) networks and (4) biogeochemistry (Jørgensen, 2012). Because of such complexity, it is not straightforward to account for the role of BD or for the impacts of its decline on ESS in general (TEEB, 2010; Jorgensen and Nielsen, 2013; Laurila-Pant et al., 2015). So the question is: how to identify and select relevant proxies of BD that allow moving forward with current knowledge?

There is still not a clear understanding of the underlying role BD plays in ESS provision (Kremen, 2005; Hattam et al., 2015; and see also review above). In order to understand this role, the parts of the ecosystem which provide the services need to be identified. Most studies consider parts of the ecosystem, such as biotic groups (e.g. Grabowski et al., 2012), habitats (e.g. Burkhard et al., 2012) or functions (e.g. Lavery et al., 2013), in understanding the effect that changes in these have on the supply of ESS. Interactions between multiple biotic groups or habitats (thus overall BD) can influence service supply (Barbier et al., 2011). However, even where BD generally has been related to the supply of services, this has started with identifying the initial relationship between specific biotic groups and their supply of services and then considering BD of these groups at a regional scale (Worm et al., 2006).

Assessing BD and evaluating the state of ecosystems requires suitable indicators for tracking progress towards environmental goals, for quantifying the relation between BD and the function, and for establishing links with ecosystem provision (e.g. Pereira et al., 2013; Tittensor et al., 2014; Geijzendorffer et al., 2016; Teixeira et al., 2016).

If assessments aim, furthermore, at contributing to increase our understanding of the general causal links between BD-EF-ESS, it is then also crucial to ensure comparability of the BD measures adopted (Pereira et al., 2013; Gonçalves et al., 2015; GOOS BioEco Panel, 2016), by selecting at least a minimum set of common metrics for monitoring trends in BD and the integrity of the ecosystems.

In the process of selecting operational indicators it is, nevertheless, important to emphasise what Jost (2006) so clearly stated: "*a diversity index is not necessarily itself a 'diversity', and likewise the many measures used as proxies to grasp biodiversity, by themselves, are not biodiversity.*" This points to the need to use complementary measures that account for the complexity and many facets of BD (Kremen, 2005; Borja et al., 2014; Bartkowski, Lienhoop, and Hansjürgens, 2015).

In this report, several potential sources of indicators (and indices or associated metrics) are presented. It is, however, important to have present that the field of BD valuation is rather heterogeneous regarding both valuation objects and valuation methods (Bartkowski, Lienhoop, and Hansjürgens, 2015; Teixeira et al., 2016). The conservation and environmental management programmes have had different goals and approaches through time and have, therefore, selected different components to be assessed (see Deliverable 2.2 by O'Higgins et al., 2016), leading to different classifications and to the choice of different indicators. For example, earlier conservation initiatives (e.g. EU Nature and Water Directives) have focused traditionally on individual structural components, or on communities' composition and associations and habitats, which is reflected in the classifications adopted (such as e.g. the EUNIS biotopes classification, species red lists, biological quality elements). More recent EBM approaches (e.g. MSFD, EU 2020 Biodiversity Strategy) attempted to integrate the interplay between natural, social and economic systems, with their choice of indicators reflecting these different dimensions and the interactions between them (e.g. BD, food webs, commercial fish and shellfish, contaminants, improve knowledge of ecosystems and their services). Such inconsistency between existing approaches leads to a gap in standardised classifications for identifying the different and most relevant components of BD for selecting BD indicators.

Here, we bring together classifications used by different approaches in an attempt to facilitate the identification and assessment of parts of the ecosystem which, directly or indirectly, contribute to the delivery of ESS (Table 1).

Once the ESS providers have been identified, these can be the focus for identifying indicators of the functions, services, and benefits,[8] while maintaining a strong link with the state of the ecosystem. A typology of ecosystem components can also facilitate assessment of changes in ecosystem state due to drivers and pressures and consequent changes in the capacity to supply services, by linking it upstream to a typology of drivers and pressures (see Section 5.2 and Deliverable 4.1) and downstream to typologies of EF and ESS, such as those discussed in the following sections.

Regarding BD and ecosystem state evaluation, numerous indicators and indices are available for aquatic ecosystems (see for example the following reviews: Piet and Jansen, 2005; Piet et al., 2006; Birk et al., 2012; ICES 2014, 2015; Hummel et al., 2015; Piroddi et al., 2015; Maes et al., 2016; Teixeira et al., 2016), which are often developed in response to legal requirements, i.e. the Water Framework Directive, the MSFD, the EU 2020 Biodiversity Strategy SEBI indicators, the Red List Index for European species and the Habitats Directive. Thus, based on the requirements set by these legal frameworks, Member States will map and assess the state of their aquatic ecosystems, as required also by the EU 2020 Biodiversity Strategy Action 5. The objectives of the above-mentioned environmental policies differ, which is reflected in the distinct approaches adopted to assess ecosystem state (Zampoukas et al., 2013). Nevertheless, from conservation-oriented frameworks targeting particular species and habitats (as in the Nature Directives) to more encompassing EBM approaches (as in the MSFD), they have all contributed to the development of a wealth of methods for ecosystem assessments (Birk et al., 2012; ICES, 2014, 2015; Teixeira et al., 2016).

The adoption of existing indicators within case studies when applying the AQUACROSS AF not only favours a relevant link with European environmental policies in place, but ensures also that data are likely to be available for indicators and metrics referenced within those legal documents (Birk et al., 2012; Berg et al., 2015; Hummel et al., 2015; Teixeira et al., 2016; Patrício et al., 2016).

Indicators available for BD assessment include a variety of approahes from structural to functional, ranging from the sub-individual level to the ecosystem level, and capturing changes and processes operating at different spatial scales (see reviews by Birk et al., 2012; Teixeira et al., 2016). Thus, the scope of the indicators available is wide and, therefore, it should be able to cover case-study needs. Nevertheless, new indicator development could be justified within the AQUACROSS project, which would complement gaps in the existing resources.

[8] The assessment of Benefits and Values is not in the scope of the present report.

Table 1: Classification for biodiversity and the state of the ecosystem, applicable to aquatic ecosystems

| *AF stage* | **Biodiversity (BD)** *hierarchical (non-hierarchical)* | | | | |
|---|---|---|---|---|---|
| *Levels* | **Category (C )** | **Section (S)** | **Division (D)** | **Group (G)** | **Class (Cl)** |
| ***Instructions*** | two approaches are possible: | for each of the previous approaches, there are several alternatives: | for each of the three previous alternatives in Category 1 Diversity, any of the following three approaches is possible: | Diversity or Ecosystem State can be assessed at different levels (as suitable): | for the different levels grouped under Biodiversity components there are several detailed classifications available, linked to different environmental policies, as indicated below: |
| | **1. Diversity** | 1. genetic diversity<br>2. structural diversity<br>3. functional diversity | **(Diversity assessment scale)**<br>1. alpha diversity ("local")<br>2. gamma diversity ("regional")<br>3. beta diversity (turnover or dissimilarity) | **(Biodiversity components)**<br>1. species<br>2. population<br>3. community<br>4. habitat (includes abiotic features)<br>5. ecosystem | 1.1. WFD & MSFD taxonomic classific.*<br>1.2. MSFD functional groups classific.*<br>1.3. Functional traits classifications*<br><br>2.1. n.a. - *go to next level:* ***Indicators and /or indices (I;i)***<br><br>3.1. EUNIS classification level 4*<br><br>4.1. HD classifications 'level 3' (habitat type)*<br>4.2. EUNIS classification level 3 *(habitat)**<br>4.3. MSFD predominant habitats classification*<br>4.4. WFD supporting elements & Hydromorphological features*<br><br>5.1. n.a. - *go to next level:* ***Indicators and /or indices (I;i)*** |
| | **2. Ecosystem State** | *(taken from Teixeira et al. 2016; definitions therein)*<br>1. Indicator Species<br>2. Target Groups<br>3. Physiological Condition<br>4. Population Ecology<br>5. Community Structure<br>6. Life Traits<br>7. Foodweb<br>8. Thermodinamically Oriented<br>9. Biotope Features | n.a. - *go to next level:* ***Group (G)*** | | |

Legend: Assessment Framework (AF), Not Applicable (n.a.), biodiversity (BD), freshwater (FW), transitional waters (TW), coastal waters (CW) and marine waters (MW). Full classification (beyond Class level) is provided in Annex I (e.g. *go to next level*). This is a hierarchical classification, except for the Division level (under category Diversity) that is interchangeable with the Section level.

We draw attention to the importance of linking the indicators to the relevant ecosystem component(s) (as in Table 1) in order to facilitate the identification and quantification of flows during integrated modelling approaches and when linking this stage of the AQUACROSS AF to the remaining stages of the SES.

It is important to clearly distinguish between these different parts of the causal chain and have a common understanding of the categories in order to develop comparable outcomes of the relationships across geographical regions and across realms. This is regardless of the types of activities, pressures or ecosystem changes which may occur (Cooper, 2013). This will also ensure that AQUACROSS outcomes from the case studies may be comparable or at least interpretable within a common framework. An initial list of possible indicators for BD and ecosystem state assessment, along with links to other relevant sources of indicators, is provided as an Annex to this report.

### 5.1.3 Ecosystem functioning classifications

Recent research is thriving with new approaches and attempts to measure functionality (see Section 2). However, ecosystem functioning was not traditionally incorporated in applied management, which is reflected in the relatively reduced number of operational indicators found in the literature (Mouillot, Graham, et al., 2013; Hummel et al., 2015; Teixeira et al., 2016). For the aquatic realm, there are, nevertheless, good examples that demonstrate the potential of considering functional aspects within management contexts, namely through the use of species functional traits (e.g. van der Linden et al., 2016), or through the complementary use of functional variables like decomposition and sediment respiration in stream health monitoring practices (Feio et al., 2010). The EU MSFD has moved a step forward by incorporating functional aspects of ecosystems into its requirements, and the marine environmental assessments will now need to incorporate functional criteria.

As introduced in Deliverable 3.2 of the AF (Chapter 2.5 in Gómez et al., 2016b), any application of ecological models, selection of indicators, and quantification of ESS requires a sound knowledge of how ecosystems are functioning (Jørgensen, Nielsen, and Fath, 2016). However, the definition of ecosystem functioning and in particular the indicators used for measuring EF do not gather more consensus (Jax, 2005; Nunes-Neto, Moreno, and El-Hani, 2014; Dussault and Bouchard, 2016) than that found for BD (see previous section). The term "function" has been used in different ways within environmental science (Jax, 2005), and in particular within ecology (Dussault and Bouchard, 2016) and the ESS context (Jax, 2016).

In ecology, functions have privileged a contextual and relational aspect, i.e. "causal role" functions (see discussion by Dussault and Bouchard, 2016), over an evolutionary perspective. Based on the organisational theory of functions, function in ecology has been defined by Nunes-Neto, Moreno, and El-Hani (2014) as "a precise effect of a given constraint on the ecosystem flow of matter and energy performed by a given item of biodiversity, within a closure of constraints." This definition clearly distinguishes and links the different components of BD and EF (i.e. BEF). And in fact, in an EBM context, as that of the AQUACROSS AF, attributing functions to biotic and abiotic components of ecosystems facilitates the purpose of analysing processes of an ecosystem in terms of the causal contributions of its parts to some activity of an ecosystem (Jax, 2005), for example, related with ESS. Nevertheless, this approach may be insufficient with respect to some important aspects of BEF research, namely in the relationship between BD and ecosystem stability and resilience (Loreau and de Mazancourt, 2013; see discussion by Dussault and Bouchard, 2016).

From an evolutionary perspective, ecological functions should be defined relative to an ecosystem's more general ability to persist (i.e., both resistance and resilience). Accounting for how species traits enhance their present fitness, and therefore their propensity to survive and reproduce (Bigelow and Pargetter, 1987), might suit better the focus of BEF research on the relationship between BD and ecosystem resilience and sustainability, which in turn, when scaled-up to ecosystems level, can be interpreted as a propensity to persist, i.e. in terms of ecosystem stability and resilience (Bouchard, 2013a, 2014 in Dussault and Bouchard, 2016). In the context of AQUACROSS, ecosystem function[9] is defined as:

> *"a precise effect of a given constraint on the ecosystem flow of matter and energy performed by a given item of biodiversity, within a closure of constraints. Ecosystem functions include decomposition, production, nutrient cycling, and fluxes of nutrients and energy."*

Ecosystem functions differ from ecosystem processes,[9] as the latter are:

> *"physical, chemical or biological action or event that link organisms and their environment. Ecosystem processes include, among others, bioturbation, photosynthesis, nitrification, nitrogen fixation, respiration, productivity, vegetation succession."*

In the process of implementing an EBM approach, it is essential that the measures of ecosystem functioning can be correlated both with measures of BD of ecosystems (Cardinale et al., 2006; Hooper et al., 2005) on one side and with measures of ESS (Harrison et al., 2014) on the other side. In this sense, despite the fact that there might still be gaps in functional indicators and that further development of new indicators will be particularly relevant in this field, we list already some approaches that might be useful for applying AF in case studies. EFs and related indicators are usually divided into three main categories: (1) production, (2) biogeochemical cycles and (3) structural, although terminology may differ slightly depending on the source. The different ecological processes that ensure these EFs are listed in Table 2; where an ecological process can be associated to several EFs, and an EF may depend on several ecological processes.

---

[9] Ecosystem function and ecosystem processes definitions have evolved from the definition of **ecological process** in the AQUACROSS Innovative Concept, p. 80, where it was defined as "*the natural transformations resulting from the complex interactions between biotic (living organisms) and abiotic (chemical and physical) components of ecosystems through the universal driving forces of matter and energy*". Although these definitions are complementary, it was felt that it would be beneficial to treat them apart, for clarity and accuracy, but also to better support selection of specific indicators.

Table 2: Classification proposed for ecosystem functions and ecological processes

| *AF stage* | **Ecosystem Functions (EF)** *hierarchical* ( *non-hierarchical* ) | | | | |
|---|---|---|---|---|---|
| *Levels* | **Category (C )**<br>**Function category** | **Section (S)** | **Division (D)**<br>**Ecosystem Function** | **Group (G)**<br>**Ecological Processes** | **Class (Cl)** |
| ***Instructions*** | | n.a. | | (a Process can be associated to several EF) | n.a. |
| | **1. Production** | | 1.1. Primary production<br>1.2. Secondary production | 1.Bioturbation<br>2.Denitrification<br>3.Evapotranspiration<br>4.Grazing<br>5.Growth<br>6.Mineral weathering<br>7.Mobility<br>8.Mutualistic interactions<br>9.Nitrification<br>10.Nitrogen fixation<br>11.Nutrient uptake<br>12.Pellitization<br>13.Photosynthesis<br>14.Predator-prey interactions<br>15.Productivity<br>16.Respiration<br>17.Sediment food web dynamics<br>18.Shell formation<br>19.Structure building<br>20.Vegetation succession | *n.a. - go to next level:*<br>***Indicators and /or indices*** |
| | **2. Biogeochemical cycles** | | 2.1. Hidrological cycling (O and H)<br>2.2. Carbon cycling (C)<br>2.3. Nitrogen cycling (N)<br>2.4. Phosphorus cycling (P)<br>2.5. Sulfur cycling (S)<br>2.6. other element cycling<br>2.7. Nutrient retention<br>2.8. Carbon sequestration | | *n.a. - go to next level:*<br>***Indicators and /or indices (I;i)*** |
| | **3. Structural (Directly mediated by ecosystem structural components - Mechanically & Physically structuring)** | | 3.1. Habitat provision<br>3.2. Nursery function<br>3.3. Breeding grounds<br>3.4. Feeding grounds<br>3.5. Refugia<br>3.6. Dispersal<br>3.7. Biological control<br>3.8. Decomposition (mechanical&chemical)<br>3.9. Filtration<br>3.10. Sediment stability & formation | | *n.a. - go to next level:*<br>***Indicators and /or indices (I;i)*** |

Legend: Assessment Framework (AF), Not Applicable (n.a.), ecosystem functions (EF), freshwater (FW), transitional waters (TW), coastal waters (CW) and marine waters (MW). Listed Processes are transversal to several EFs. See Annex tables for full EF classification.

### 5.1.4 Ecosystem services classifications

The concept of ecosystem services (ESS)[10] has been evolving since the last century, even if the term ESS was not specifically employed. Ehrlich and Mooney published one of the first scientific publications referring to the term ESS in 1983 with a paper entitled: *Extinction, Substitution, and Ecosystem Services*. In 1997, Costanza and colleagues estimated *The value of the world's Ecosystem Services and natural capital*, publishing their results in Nature. The UN Secretary-General, Kofi Annan, called for the Millennium Ecosystem Assessment (MA) in 2000 in his report to the UN General Assembly, *We the Peoples: The Role of the United Nations in the 21st Century*. The objective of the MA was to assess the consequences of ecosystem change for human well-being and to establish the scientific basis for actions needed to enhance the conservation and sustainable use of ecosystems and their contributions to human well-being (MA, 2005). In 2010, the global initiative The Economics of Ecosystems and Biodiversity (TEEB) emerged, focusing on "making nature's values visible" and mainstreaming the values of BD and ESS into decision-making at all levels (TEEB, 2010). In 2013, the CICES working group published their final working report (CICES, version 4.3) (Haines-Young and Potschin, 2013). The proposed revised classification aimed to avoid double counting of ESS, namely between regulating and habitat or supporting services as foreseen in MA and TEEB, giving a special focus on those services which are underpinned by a connection to BD and the biological processes and functions of ecosystem. In the context of CICES' final version, ESS are biologically mediated, although CICES acknowledges the abiotic outputs from ecosystems. In this sense, the report includes a separate but complementary typology of abiotic outputs to facilitate their assessment. However, as highlighted in the report from the System of Environmental-Economic Accounting experimental ecosystem accounting working group (United Nations et al., 2014), CICES provides a structure to classify the flow of "final" ESS, but fails to provide a structure to classify ecosystem assets, ecosystem processes, and to link this information to economic and other human activities. Nevertheless, this working group acknowledges that the development of CICES will benefit from testing and use in the compilation of estimates of ESS.

At the EU level, in 2011, the EU Biodiversity Strategy to 2020 was publish, which aims to halt the loss of biodiversity and ESS in the EU and to help stop global biodiversity loss by 2020. This strategy also reflects the commitments taken by the EU in 2010, within the international Convention on Biological Diversity. In this context, the European Commission created a

[10] In the scope of AQUACROSS AF, ESS are the final outputs from ecosystems that are directly consumed, used (actively or passively) or enjoyed by people. In the context of CICES they are biologically mediated (human-environmental interactions are not always ESS, e.g. maritime traffic, tourism activities). This concept tries to bring together previous definitions. This definition has evolved from the early definition in the AQUACROSS Innovative Concept, p. 80, where it was defined as: "*Those benefits humans get from ecosystems*", thus making it more inclusive.

working group to support EU Member States reporting under Action 5 of the EU Biodiversity Strategy to 2020 – Mapping and Assessment of ESS (MAES WG). The MAES WG adopted CICES, which is the EU reference classification. While in previous reports CICES (Potschin and Haines-Young, 2011) included abiotic and biological mediated outputs as ESS, with a qualification specifying the level of dependency on BD, the final iteration of CICES (Haines-Young and Potschin, 2013) recommended that abiotic outputs are not considered as ESS with only those outputs reliant on living processes included. This focus on biologically-mediated services has been further emphasised through the adoption of the CICES classification system by the MAES WG, which, so far, only considers the biologically-mediated services for support of the EU Biodiversity Strategy,[11] i.e. those services which are associated with and dependent on BD (Maes et al., 2014, 2016).

As discussed in the AQUACROSS AF (Gómez et al., 2016b), despite this broad consensus in the current policy-relevant assessments of ESS, it is recognised that this definition of services (biologically-mediated) will not satisfy all, and that future assessments would benefit from being integrated, i.e. accounting for biological and abiotic outputs of ecosystems. There are important arguments supporting the inclusion of abiotic outputs of the ecosystem, as they can have implications for spatial planning, management and decision-making (e.g., Armstrong et al., 2012; Kandziora, Burkhard, and Müller, 2013; Sousa et al., 2016; Lillebø et al., 2016).

Following the evolution of the ESS concept, the definition of ESS has also evolved over the last decades. Table 3 highlights relevant examples that were taken into account in the AQUACROSS AF to reach the definition of ESS in the context of AQUACROSS (Chapter 2.5 in Gómez et al., 2016b).

In the scope of the AQUACROSS framework the final outputs include those resulting from mediated biological processes and/or from abiotic components of ecosystems, as illustrated in Table 4 to Table 6. The AQUACROSS definition of ESS encompasses more broadly the goods and services people get from the ecosystem, such as the abiotic outputs which are not affected by changes in the biotic aspects of ecosystem state (EEA 2015). It is, however, important to recall that Human environmental interactions are not always ESS, e.g. maritime traffic, tourism activities. These would be picked up as primary or secondary activities under the concepts described in Deliverable 4.1.

[11] Note: in CICES some of the regulating services provided by ecosystems acknowledges the combination of biotic and abiotic factors.

Table 3: Relevant examples of ESS definitions that were considered to reach the AQUACROSS definition of ESS

| Reference | Definition of ESS |
|---|---|
| Ehrlich & Mooney 1983 | Although a specific definition is not provided, authors refer to several ESS, e.g. (flood control, erosion prevention, filtration of atmospheric pollutants, supply of firewood and timber, climate-ameliorating services, public service functions of the systems, crops and pest control), and elaborate that: "The degree of alteration of services depends on the functional role(s) of the organisms that go extinct and on the pattern of extinctions (e.g., selective deletion from an ecosystem or destruction of most elements simultaneously)" |
| Costanza et al., 1997 | "the benefits human populations derive, directly or indirectly, from ecosystem functions" |
| MA, 2005 | "the benefits people obtain from ecosystems" |
| TEEB, 2010 | "direct and indirect contributions of ecosystems to human well-being"; the concept of ecosystem 'goods and services' is synonymous to ESS |
| Haines-Young and Potschin, 2013 (CICIES) | "Final ecosystem services are the contributions that ecosystems make to human well-being. These services are final in that they are the outputs of ecosystems (whether natural, semi-natural or highly modified) that most directly affect the well-being of people. A fundamental characteristic is that they retain a connection to the underlying ecosystem functions, processes and structures that generate them" |
| Maes et al., 2015, 2016 (MAES WG) | "the benefits people obtain from ecosystems" (MA); "direct and indirect contributions of ecosystems to human well-being" (TEEB); service flow refers to the 'actually used service',i.e., the 'final' services. The rationale for this division is to avoid the double counting of intermediate (or supporting) services in the valuation step of the process. |
| AQUACROSS AF (Gómez et al., 2016b: Chapter 2.5) | *"the final outputs from ecosystems that are directly consumed, used (actively or passively) or enjoyed by people"* |

As also discussed in the AQUACROSS AF, it is of paramount importance to consider ESS from the supply-side, considering the capacity of the ecosystem to supply services, and from the demand-side, including an economic perspective. As defined in Gómez et al. (2016b: Chapter 2.5) the supply side is "the potential or capacity of the ecosystem to supply services, whether or not it is used", whilst the demand side is "the services people ask from the ecosystems whether they are actually provided or not." Moreover, a 'supply-side' assessment based on ecosystem capacity considers how the state of the ecosystem is affecting the generation of the actually used services (Burkhard et al., 2012) and the potential to provide more and better services for present and future generations.

Ehrlich and Mooney discussed back in 1983 the links between extinctions of given elements of an ecosystem (populations, species, guilds) and the supply of ESS. These authors referred to the fact that extinctions in ecosystems occur continuously due to evolutionary and ecological processes. However, some of the human-driven extinctions have led to serious impairment of ecosystem functioning and of the services delivered to humanity. The provisioning of services should reflect changes to the ecosystem state (e.g. Böhnke-Henrichs, et al. 2013; Haines-Young and Potschin, 2013). This means that to be considered a service, a change in state of the ecosystem must result in a change in the supply of a service. This is true for biologically-mediated services; for example, a change in abundance of commercial fish populations has an impact on the supply of seafood, or a change in the wetland heath status (e.g. fragmentation) has an impact on the supply of clean water. However, a change or a difference in the abiotic conditions can also lead to a change in the supply of abiotic services; for example, a change in sand natural deposits, including beaches, due to a high energy storm event has an impact on mining of sand for construction or industrial uses, or even an impact on recreational activities on the beach. The exploitation of abiotic outputs, in addition to the use of the ecosystem for economic activities (i.e., space for activities to occur), can have an impact on the state of the ecosystem and, thus, the potential supply of services (Lillebø et al., 2016), even if they are not affected themselves by the state of the biological components of the ecosystem. However, whilst the capacity of the ecosystem to supply services is tightly linked to the state of the ecosystem (BD and ecosystem processes and functions), the demand and actual use of services can be decoupled from the state of the ecosystem, as they are a clear outcome of social processes. Also, a change in ecosystem state and BD can lead to a change in the supply of services but not in the demand of services. Moreover, the detrimental impacts of the use of services can, in turn, lead to a change in ecosystem state and BD and to a change in the supply of services. To build realistic scenarios for conservation and management purposes considering social-economic drivers, it is necessary to account for all services, namely the biologically-mediated ESS and the abiotic outputs (for more detailed information on Drivers and Pressures, see Deliverable 4.1).

In AQUACROSS, we aim to promote comprehensive assessments of the ESS and the benefits people get from nature, as much as they help with the understanding of complex systems for the identification and evaluation of appropriate responses (following the EBM concept). In this sense, partial ESS assessments may still be appropriate depending, for example, on objectives, scale or feasibility. Thus, to support different needs, we include both the services dependent on BD as well as those reliant on purely physical aspects of the ecosystem. Apart from the operational definition of ESS within AQUACROSS, it is also important to know how the concept relates to the ecosystem components, namely to its functions and processes. The work to be developed and tested within AQUACROSS WP5 will account for ESS and for the abiotic outputs from ecosystems combined with EFs and ecological processes. Table 4 to Table 6 illustrate some examples, meaning that lessons learnt from this application may lead to an adaptation of the AQUACROSS AF and/or the overall concepts of AQUACROSS.

The ESS classifications presented in Table 4 to Table 6 also include examples of ecosystem functions/ecological processes and abiotic functions/abiotic processes illustrating how to link this ESS classification to the EF classification proposed in the previous section.

Table 4: ESS and examples of EF and ecological processes, considering both biotic and abiotic dimensions, for the CICES Provisioning category

| Ecosystem services | | | | Abiotic outputs from ecosystems | |
|---|---|---|---|---|---|
| Provisioning | | | | Abiotic Provisioning | |
| Division | Group<br>(includes the respective classes) | Ecosystem functions<br>Ecological processes | Abiotic functions<br>Abiotic processes | Group | Division |
| Nutritional | **Biomass**<br>Wild plants and fauna; plants and animals from in situ aquaculture | **Production**<br>Primary production; Secondary production Photosynthesis; Respiration; Growth | **Production**<br>Evaporation; Crystallization | **Mineral**<br>Marine salt | Nutritional abiotic substances |
| | **Water**<br>Surface or groundwater for drinking purposes | **Structural**<br>(Directly mediated by ecosystem structural components)<br>Feeding grounds | **Structural**<br>(Directly mediated by the sun structural components)<br>Energy processes that makes the sun shine | **Non-mineral**<br>Sunlight | |
| Materials | **Biomass**<br>Fibres and other materials from all biota for direct use or processing; genetic materials (DNA) from all biota | **Production**<br>Primary production; Secondary production Growth | **Production**<br>Geochemical processes | **Metallic**<br>Poly-metallic nodules; Cobalt-Rich crusts, Polymetallic massive sulphides | Abiotic materials |
| | **Water**<br>Surface or groundwater for non-drinking purposes | **Structural**<br>(Directly mediated by ecosystem structural components)<br>Feeding grounds | **Structural**<br>(Directly mediated by the earth structural components)<br>Geochemical processes | **Non-metallic**<br>Sand/gravel | |
| Energy | **Biomass**<br>Plants and fauna | **Structural**<br>(Directly mediated by ecosystem structural components)<br>Productivity | **Structural**<br>(Directly mediated by the earth structural components)<br>Atmospheric and Ocean processes | **Renewable abiotic energy sources**<br>Wind and wave energy | Energy |
| | | | **Structural**<br>(Directly mediated by the earth structural components)<br>Geochemical processes | **Non-renewable abiotic energy sources**<br>Oil and gas | |

Source: adapted from Haines-Young and Potschin, 2013

Table 5: ESS, EF and ecological processes considering both biotic and abiotic dimensions, for the CICES Regulating and maintenance category

| Ecosystem services | | | | Abiotic outputs from ecosystems | |
|---|---|---|---|---|---|
| Regulating and maintenance | | | | Regulating and maintenance by abiotic structures | |
| Division | Group (includes the respective classes) | Ecosystem functions ecological processes | Abiotic functions Processes | Group | Division |
| Mediation of waste, toxics and other nuisances | Mediation by biota<br>Mediation by ecosystems Combination of biotic and abiotic factors | **Structural** (Directly mediated by ecosystem structural components) Decomposition Bio-physicochemical filtration/ sequestration/ storage/ accumulation of pollutants by biota and/or ecosystems; Mineralization processes; **Biogeochemical cycles** Adsorption and binding of metals and organic compounds in ecosystems, as a result of combination of biotic and abiotic processes | **Structural** (Directly mediated by ecosystem physical structural components) Screening by natural physical structures Atmospheric dispersion and dilution; Adsorption and sequestration of waters in sediments. **Geochemical cycles** Adsorption and binding of metals and organic compounds in ecosystems, as a result of abiotic processes | By natural chemical and physical processes | Mediation of waste, toxics and other nuisances |
| Mediation of flows | Mass flows<br>Liquid flows<br>Gaseous/air flows | **Structural** (Directly mediated by ecosystem structural components) Physical protection by vegetation (floods, wind and water erosion); Evapotranspiration; soil formation **Biogeochemical cycles** Nitrogen uptake; Denitrification; Carbon sequestration | **Structural** (Directly mediated by ecosystem physical structural components) Protection by sand and mud flats; topographic control by dunes and cliffs of wind erosion; **Global cycles** Adsorption/desorption processes; sedimentation; Diffusion; Precipitation | By solid (mass), liquid and gaseous (air) flows | Mediation of flows by natural abiotic structures |
| Maintenance of physical, chemical, biological conditions | Lifecycle maintenance, habitat and gene pool protection<br>Pest control<br>Soil formation and composition<br>Water conditions<br>Atmospheric composition and climate regulation | **Structural** Habitat provision; Nursery function; Breeding grounds; Feeding grounds; Refugia; Dispersal; Biological control; Filtration; Sediment stability & formation Predator-prey interactions; Grazing; Structure building; **Biogeochemical cycles** Mediation of geochemical cycles processes; Mediation of hydrological cycle processes; Mediation of atmospheric composition processes | **Structural** (Directly mediated by ecosystem physical structural components) Structure building **Global cycles** Global geochemical processes, atmospheric and Oceans circulation; Hydrological cycle; | By natural chemical and physical processes | Maintenance of physical, chemical, abiotic conditions |

Source: adapted from Haines-Young and Potschin, 2013

Table 6: Ecosystem services, ecosystem functions and ecological processes considering both biotic and abiotic dimensions, for the CICES Cultural category

| Ecosystem services | | | | Abiotic outputs from ecosystems | |
|---|---|---|---|---|---|
| Cultural | | | | Cultural settings dependent on aquatic abiotic structures | |
| Division | Group (includes the respective classes) | Ecosystem functions ecological processes | Abiotic functions processes | Group | Division |
| Physical and intellectual interactions with biota, ecosystems, and seascapes [environmental settings] | Physical and experiential interactions | Structural (Directly mediated by ecosystem structural components) Human perception processes of Physical and intellectual interactions with environmental settings | Structural (Directly mediated by ecosystem physical structural components) Human perception processes of Physical and intellectual interactions with physical settings | Physical and experiential interactions | Physical and intellectual interactions with land-/seascapes [physical settings] |
| | Experiential use of biota and seascapes; physical use of seascapes in different environmental settings | | | Experiential use of seascapes; physical use of seascapes in different physical settings | |
| | By physical and experiential interactions or intellectual and representational interactions | | | By physical and experiential interactions or intellectual and representational interactions | |
| | Intellectual and representational interactions Scientific; education, heritage; aesthetic; entertainment | | | Intellectual and representational interactions Scientific; education, heritage; aesthetic; entertainment | |
| Spiritual, symbolic and other interactions with biota, ecosystems, and seascapes [environmental settings] | Spiritual and/or emblematic Symbolic; sacred and/or religious | Structural (Directly mediated by ecosystem structural components) Human perception processes of natural ecosystem components | Structural (Directly mediated by ecosystem physical structural components) Human perception processes of natural physical structures | Spiritual and/or emblematic Symbolic; sacred and/or religious | Spiritual, symbolic and other interactions with land-/seascapes [physical settings] |
| | Other cultural outputs Existence; bequest | | | Other cultural outputs Existence; bequest | |

Source: adapted from Haines-Young and Potschin, 2013

## Identification of relevant indicators and associated metrics

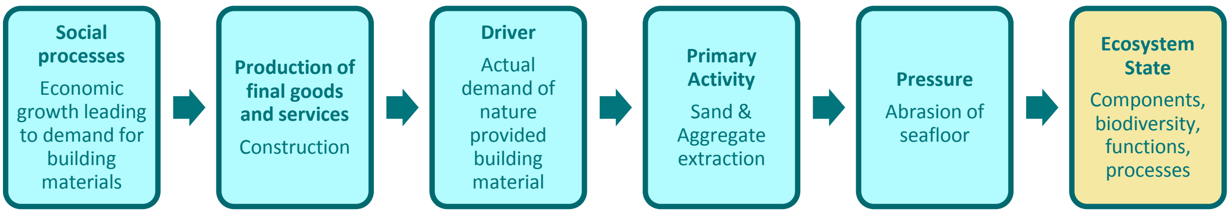
As presented in the previous tables, the indicators and metrics were categorised using the EU MAES ESS categories (Maes et al., 2014), which build on latest version (V4.3) of CICES (Haines-Young and Potschin, 2013; Maes et al., 2014, 2016): 1. Provisioning, 2. Regulating & Maintenance and 3. Cultural. This will ensure comparability with the approaches being followed by EU Member States.

An initial list of ESS indicators was obtained from the comprehensive review elaborated by Egoh et al. (2012) and complemented with the recent list of MAES indicators for ESS (Maes et al., 2014, 2016), and with Hattam et al. (2015) specific indicators for the marine environment. Also, to accommodate the inclusion of abiotic outputs, potential indicators will be identified and added to the lists. The selection of specific ESS indicators for applying the AQUACROSS AF (Gómez et al., 2016b), will be driven by the case studies context and needs. Lessons learnt from this application of indicators to the AQUACROSS case studies may lead to an adaptation of the AQUACROSS AF and/or the overall concepts of AQUACROSS.

# 5.2 Flows from Drivers and Pressures to Ecosystem State, Functions and Services

Under Deliverable 4.1 of AQUACROSS, the demand-side perspective on how use of ecosystem goods and services affects the ecosystem is covered in detail, but it is important to elaborate on this in terms of how the flows from social processes through drivers and their pressures to ecosystem state (see Figure 4) might then have causal links to changes in functions and supply of ESS. In other words, how might the effects of drivers on ecosystem state shown on the far right of Figure 4 below, lead to possible changes in the capacity to supply ESS?

Figure 4: Example of a single impact chain from a social process to its subsequent changes in ecosystem state




Legend: Drivers are the demand for the supply of ESS, resulting from social processes, such as economic growth, and the production of final goods and services, which require ESS from nature. Primary activities are directly involved in the exploitation of ESS and thus can directly cause pressures on ecosystem state. Ecosystem state (highlighted in blue here) then links through to the supply-side perspective on implications for supply of ESS, which is the focus of this broader report. For more information, see Deliverable 4.1. Source: Own Illustration

As described earlier in Section 5.1, there are many different potential classifications and indicators that can be selected to illustrate the state, and change in state, of BD, EF and ESS; likewise, under Deliverable 4.1, classifications and indicators have been described for activities associated with key drivers influencing aquatic ecosystems, and for the pressures they cause. In Table 7 below, the summarised classifications of broad activities and pressures taken from D4.1 have been added to those covered in more detail in Section 5.1 of this report. Considering these classifications and lists of possible indicators helps to establish the overall SES in which we may be considering evaluation of particular issues, and this can be formalised in a set of linkage matrices that describe the possible network of interactions relevant to a given study system (see Section 3.3 of Deliverable 4.1). As indicated in Table 7, the ecosystem state/biodiversity components form the common link between the demand-side (WP4) and the supply-side (WP5) perspectives, by linking upstream to a classification of drivers and pressures and downstream to those of ecosystem functions and services. Following Table 7 below, we go on to explore how the choice of indicators of BD-EF-ESS should be influenced by consideration of *both* how the pressure effect on ecosystem state is measured, and how any contributions to capacity to supply linked ecosystem services are measured.

Table 7: Combined broad classifications of activity types and pressures, ecosystem state/biodiversity, ecosystem functions and ecosystem services

| *AF stage Level* | Broad Activity Type | Pressure Categories | Ecosystem State / Biodiversity | Ecosystem Functions | | Ecosystem Services<br>***following CICES/MAES* | | | |
|---|---|---|---|---|---|---|---|---|---|
| | *Deliverable 4.1* | | | | | | | | |
| | 1 Agriculture & Forestry<br>2 Aquaculture<br>3 Fishing<br>4 Environmental Management Manufacturing<br>5 Waste Management<br>6 Residential & Commercial Development<br>7 Services<br>8 Mining, extraction of materials<br>9 Non-renewable energy<br>10 Renewable energy<br>11 Tourism, recreation & non-commercial harvesting | 1 Biological Disturbance<br>2 Chemical change, chemical & other pollutants<br>3 Physical change<br>4 Energy<br>5 Exogenou/ Unmanaged | ECOSYSTEM STATE*<br>1 Indicator Species<br>2 Target Groups<br>3 Physiological Condition<br>4 Population Ecology<br>5 Community Structure<br>6 Life Traits<br>7 Foodweb<br>8 Thermo-dinamically Oriented<br>9 Biotope Features<br><br>BIODIVERSITY<br>1 genetic diversity<br>2 structural diversity<br>3 functional diversity<br><br>**(definitions in Teixeira et al. 2016)* | 1 Bioturbation<br>2 Denitrification<br>3 Evapotranspiration<br>4 Grazing<br>5 Growth<br>6 Mineral weathering<br>7 Mobility<br>8 Mutualistic interactions<br>9 Nitrification<br>10 Nitrogen fixation<br>11 Nutrient uptake<br>12 Pellitization<br>13 Photosynthesis<br>14 Predator-prey interactions<br>15 Productivity<br>16 Respiration<br>17 Sediment foodweb dynamics<br>18 Shell formation<br>19 Structure building<br>20 Vegetation succession | **1. Production**<br>1.1 Primary production<br>1.2 Secondary production<br><br>**2.Biogeochemical Cycles**<br>2.1 Hidrological cycling (O and H)<br>2.2 Carbon cycling (C)<br>2.3 Nitrogen cycling (N)<br>2.4 Phosphorus cycling (P)<br>2.5 Sulfur cycling (S)<br>2.6 Other element cycling<br>2.7 Nutrient retention<br>2.8 Carbon sequestration | **A. Abiotic**<br>**B. Biotic**<br><br>(*for details on abiotic ESS see section 5.1.4*) | **1 Cultural** | 1.1 Physical and intellectual interactions with biota, ecosystems, and land-/seascapes [environ-mental settings]<br>1.2 Spiritual, symbolic and other interactions with biota, ecosystems, and land-/seascapes [environ-mental settings] | 1.1.1 Intellectual and representative interactions<br>1.1.2 Physical and experiential interactions<br>1.2.1 Other cultural outputs<br>1.2.2 Spiritual and/or emblematic |

| | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | | | | | | | 2Provisioning | 2.1 Energy<br>2.2 Materials<br>2.3 Nutrition | 2.1.1 Biomass-based energy sources<br>2.1.2 Mechanical energy<br>2.2.1 Biomass<br>2.2.2 Water<br>2.3.1 Biomass<br>2.3.2 Water |
| | | | | | **3. Structural (Directly mediated by ecosystem structural components**<br>3.1 Habitat provision<br>3.2 Nursery function<br>3.3 Breeding grounds<br>3.4 Feeding grounds<br>3.5 Refugia<br>3.6 Dispersal<br>3.7 Biological control<br>3.8 Decomposition mechanical & chemical<br>3.9 Filtration<br>3.10 Sediment stability & formation | | 3Regulating | 3.1 Main-tenance of physical, chemical, biological conditions<br>3.2 Mediation of flows<br>3.3 Mediation of waste, toxics and other nuisances | 3.1.1 Atmospheric composition & climate regulation<br>3.1.2 Lifecycle maintenance, habitat & gene pool protection<br>3.1.3 Pest &disease control<br>3.1.4 Soil formation & composition<br>3.1.5 Water conditions<br>3.2.1 Gaseous/air flows<br>3.2.2 Liquid flows<br>3.2.3 Mass flows<br>3.3.1 Mediation by biota<br>3.3.2 Mediation by ecosystems |

Note: In each case, more detailed lists are given in either Deliverable 4.1 (for WP4 classifications) and Tables 1,2 and 4,5,6 of this report (for WP5 classifications) (Broad Activity Types which can directly cause pressure in the ecosystem).

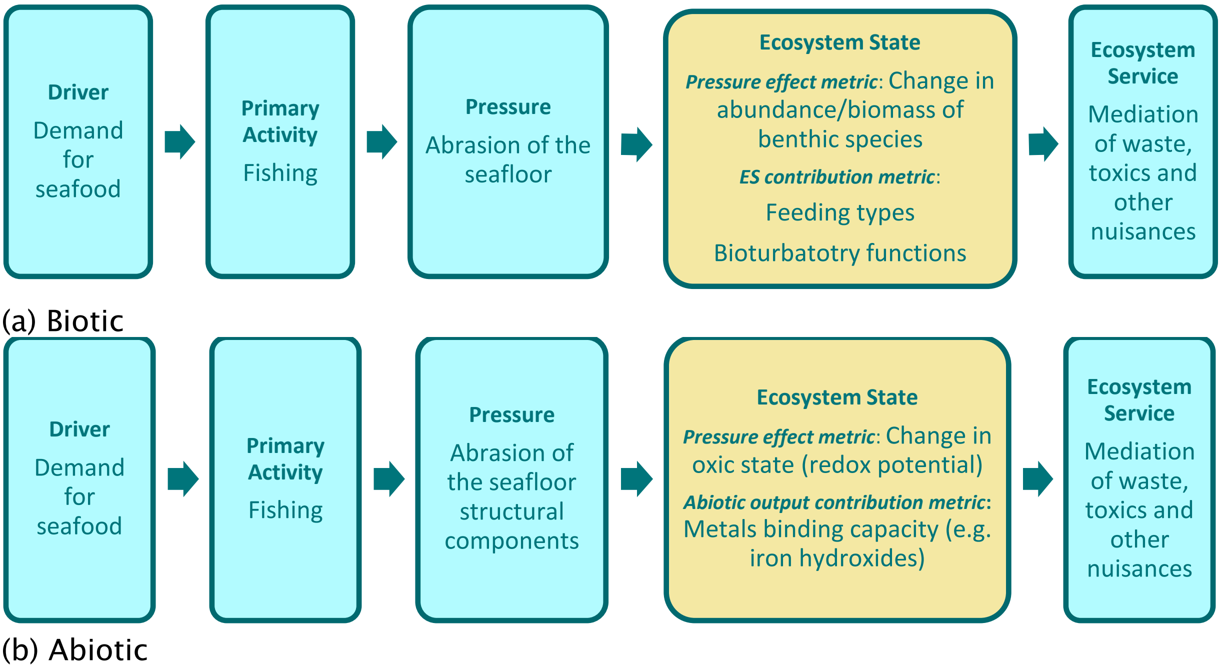
### 5.2.1 Linking the demand side to the supply side through ecosystem state metrics

The effects of pressures on the ecosystem have been explored both through field-based observations and experimental manipulations (see detail under Deliverable 4.1). These studies tend to inform us about the effects at the species or, sometimes, the process level. We need to understand how, or if, these changes will lead to any change in the capacity of the ecosystem to supply services. Here we consider how an understanding of the way in which pressures interact with the state of the ecosystem can affect options for evaluating the change in supply of ESS. In many cases, the metrics used to describe how pressures change ecosystem state may not, themselves, be the appropriate metrics to describe how the ecosystem contributes to the supply of services.

For example, most studies on the effects of abrasion pressure from trawling activity describe the effects in terms of changes in abundance or sometimes biomass of benthic invertebrate species (Kaiser et al., 2006) or aquatic submerged vegetation (Costa and Netto, 2014). In order to consider how these changes might lead to an effect on supply of ESS, we need to know more than this. Firstly, we need to know which services are at least, to some extent, underpinned by the functions and processes of the effected communities (benthic fauna and flora here), and, secondly, in what way do these communities supply those services, and do measurements of abundance and/or biomass capture this? To continue the example above, in order to consider the effect of abrasion on the capacity to supply the service *Mediation of waste, toxics and other nuisances* (see Table 5, Section 5.1.4), not only would we need to know about abundance and/or biomass, but we would also need to know how the different components of the benthic ecosystem can be described in terms of their role in supplying this regulating and maintenance service. This could be through consideration of biological traits that are associated with *Mediation of waste, toxics and other nuisances* e.g. the role of different fauna species in bioturbation or the role of seagrasses in phytoremediation (Figure 5a).

As such, pressures can have multiple effects and act on structures, processes and functions that support ESS, but they might also support abiotic outputs from the ecosystem. In this way, pressures can have direct and indirect effects on service provision, but also on the abiotic outputs from ecosystems (check ESS definition in the scope of AQUACROSS in Table 3). For example, comparable abiotic outputs from the ecosystem that might be affected by fishing trawling activity would be the regulating and maintenance services, specifically the mediation of waste, toxics and other nuisances, by natural chemical and physical processes taking place in the seafloor. Similarly, fishing causes abrasion of the seafloor, affecting in this way the seafloor structural components. This might have implications on the adsorption and binding of metals and organic compounds in seafloor, underpinned by abiotic processes, and therefore on the mediation of waste, toxics and other nuisances. A relevant example would be the exposure of anoxic layers to oxygen rich seawater and consequent changes in the redox potential, which will change the seafloor geochemistry (Figure 5b).

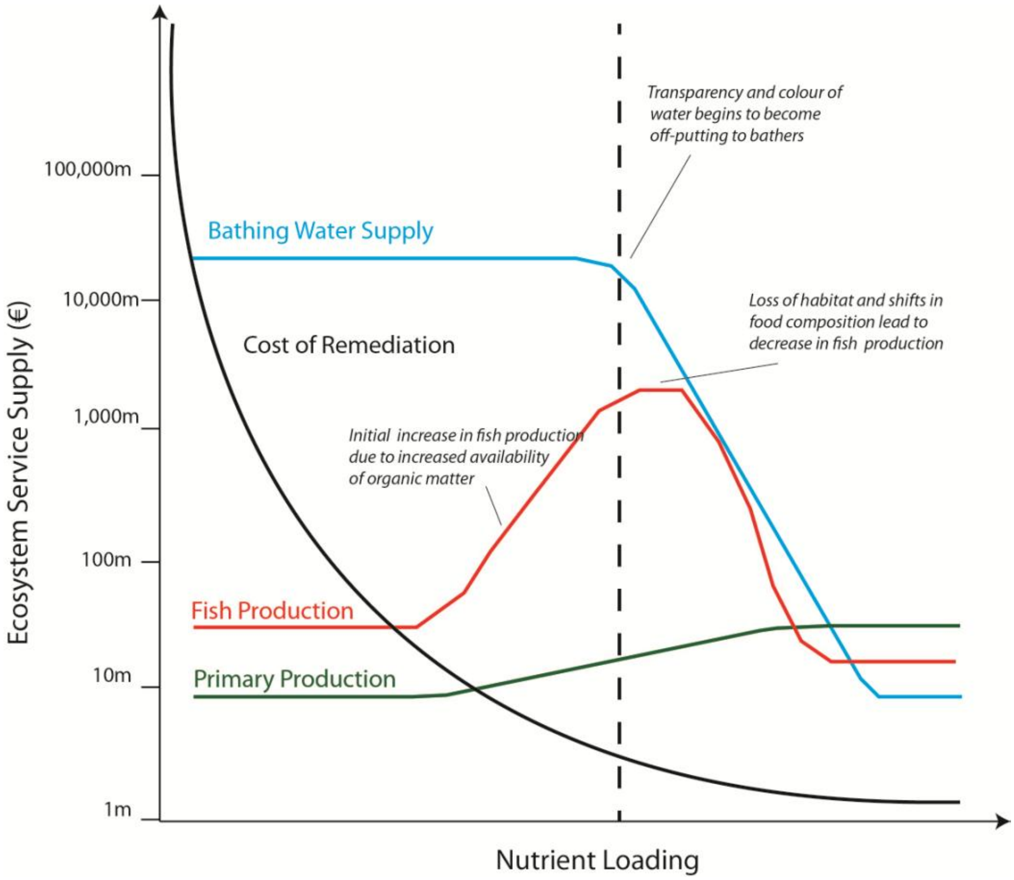
Figure 5: Fishing causes abrasion of the seafloor structural components, both biotic and abiotic




Legend: (a) Biotic: abrasion can affect e.g. the seagrasses and the benthic invertebrate species that live there, as well as the seagrass community, and is often assessed by measuring the effect on abundance and/or biomass of the species and the percentage of coverage and/or fragmentation (the pressure effect metric shown in the ecosystem state box highlighted in blue above). Benthic species, including seagrasses, contribute to the ESS remediation of wastes. However, in order to evaluate the effect of fishing abrasion on this capacity to supply this ESS, we would also need to know something about the bioturbation and feeding modes of benthic species in the communities affected (how they contribute to functioning that is relevant to supplying this ESS; the ESS contribution metric in the ecosystem state box above). (b) Abiotic: abrasion can also affect the sediment stability and redox potential equilibrium, and therefore the sediment profile oxic state. Sediment contributes to the ESS remediation of wastes, through, for example, its binding capacity for metals. However, in order to evaluate the effect of fishing abrasion on the capacity to supply this ESS, we would also need to know the sediment adsorption–desortion capacity for metals (how it contributes to function that is relevant to supplying this ESS; an example of the ESS contribution metric is given in the ecosystem state box above).

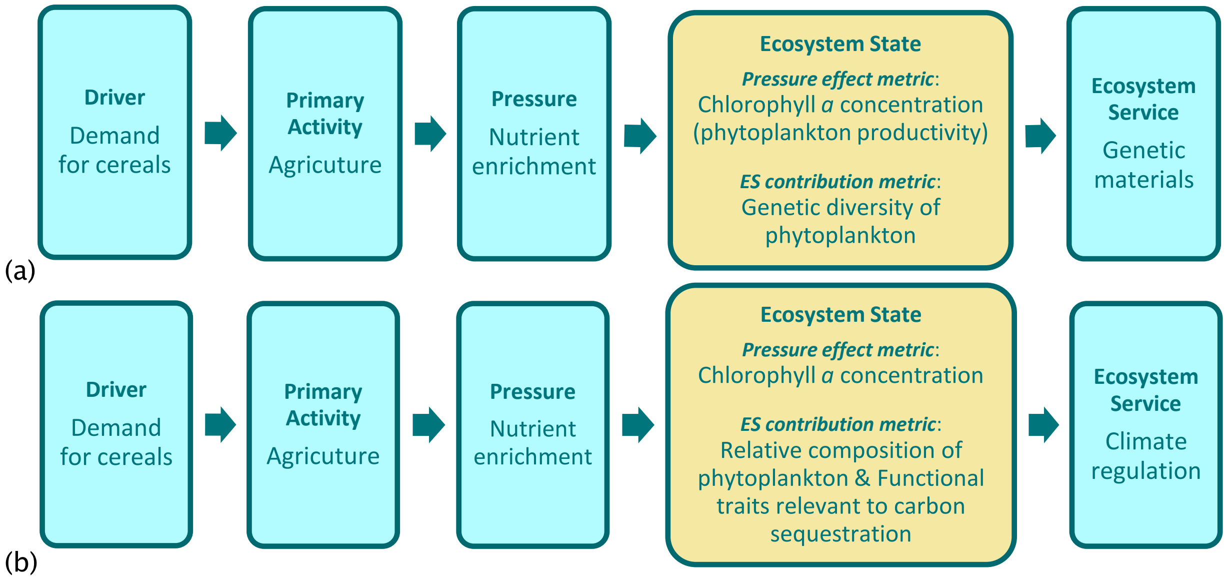
Taking another example, the assessment of nutrient enrichment in the aquatic environment is frequently assessed through measuring the chlorophyll *a* concentration of the water as an indication of the productivity of phytoplankton as a response to nutrient enrichment. Nutrient enrichment can also cause changes in species diversity and relative abundances of taxa in phytoplankton communities and the contribution of a change in relative composition, species diversity and overall productivity of the phytoplankton may then in turn have consequences for the capacity to supply certain ecosystem services, namely fish production and water for recreational purposes (O'Higgins and Gilbert, 2014), as illustrated in Figure 6. Dependent on the ESS, we may need to know different things about ecosystem state in order to evaluate if the capacity to supply the service being considered has in some way been affected by the pressure acting on the system (Figure 7). However, where there is only information available on how nutrient enrichment affects chlorophyll *a* concentrations for a given study system, there are then a number of assumptions that would need to be made in order to try to evaluate whether this would mean anything in terms of those metrics relevant to the supply of linked ESS.

Figure 6: Idealised trajectories for monetised recreational amenity value, carbon production/burial value and fish production value with changing nutrient load




Legend: amenity value (blue), carbon production/burial value (green) and fish production value (red). The curved black line indicates remediation cost; the dashed black line indicates a theoretical optimal solution. Source: O'Higgins and Gilbert, 2014

Figure 7: Agriculture releases nutrients through diffuse run-off into aquatic systems causing nutrient enrichment, which can affect phytoplankton communities in the water column




Legend: The effects are often assessed by measuring the chlorophyll *a* concentration of the water, which is taken as a proxy for phytoplankton productivity. Meanwhile, phytoplankton species contribute to a number of ESS, including (a) genetic materials and (b) climate regulation. However, in order to assess whether nutrient enrichment from agriculture could affect the capacity of the ecosystem to supply either of these ESS, we would also need to know something about the species or genetic diversity of the phytoplankton communities (a) and any change in relative composition of functionally relevant groups (b).

## 5.2.2 Summary

Understanding which part of the ecosystem (which ecosystem state components) is impacted by pressures can help lead to an understanding of how the ecosystem's capacity to supply ESS may be impacted, but the way a pressure affects ecosystem state and the way that this is measured, may not align with what needs to be known to assess the ecosystem's capacity to supply a service. Accordingly, it will be necessary to consider both the relevant metrics that describe the pressure effect on ecosystem state and those that describe how the ecosystem contributes to supply ESS in setting out to select relevant indicators to evaluate in the case study systems. Furthermore, linkage matrices that highlight all possible relational links between pressures and ecosystem state components, and state components with EFs and ESS, will help to provide a framework for exploring analyses across the whole SES (see further details in Deliverable 4.1). In the case studies, matrices will be developed under Task 4.2 to highlight linkages between drivers and pressures with different aspects of ecosystem state in the study systems. We recommend that under linked work through Task 5.2, the possible links out from the ecosystem state characteristics to EFs and ESS are also added to help provide a consistent framework in which analyses going forward can be explored.

# 6 Numerical Approaches to Analyse Causal Links

Ecological and biological systems dynamics are often governed by nonlinear interactions of environmental factors. Interactions between environmental variables can be so complex that the whole system achieves a broader functionality that cannot be deduced by considering individual environmental factors (Tan et al., 2006). Thus, analysis of these complex relationships requires the use of models and statistical tools capable of dealing with large datasets of environmental and biological variables.

Among the multitude of mathematical tools and approaches available, four of them can be used to assess causal links and environmental flows in case studies: discriminant analysis, generalised dissimilarity models, generalised diversity-interactions models, and tools integrating Bayesian approaches like ARIES.

The ordination and classification methods presented below can be applied in the case studies to better characterise and understand the flows between BD-EF-ESS. The outlined methodology is not exhaustive, instead it aims to illustrate some suitable approaches that can be used. The choice of methodology will ultimately depend on the objective of the study, and on the amount and quality of the available data.

## 6.1 Discriminant Analysis – DA

Discrimination methods include both classification ("predictive discriminant analysis") and separatory approaches ("descriptive discriminant analysis") with the linear combinations of descriptive discrimination known as linear discriminant functions or, more formally, canonical variates. Though predictive and separatory discrimination methods differ theoretically and operationally, they are nonetheless closely related (Williams, 1983).

Discriminant Analysis (DA) also known as Canonical Variate Analysis or Linear Discriminant Analysis is a multivariate approach to pattern recognition and interpretation that has been used extensively in ecological investigations, e.g. fish distributions (Olden and Jackson, 2002), freshwater habitat selection (Joy and Death, 2003), temporal patterns linked with physico-chemistry and the biology in aquatic systems (Fabrègue et al., 2014) and linking trophic guild with functional traits (Albouy et al., 2011).

DA is generally appropriate in problems with aggregated multivariate data, and has been applied by ecologists in areas as diverse as geographical ecology, social behaviour, niche structure, and organism morphology and physiology. This technique allows the classification of sites into classes or clusters using data from species composition and how it differs among sites of different classes. The abundance of several species may be combined to make the differences between classes clearer than is possible on the basis of the abundance values of a single species. However, the use of DA only makes sense if the number of sites is much greater than the number of species and the number of classes (Schaafsma and Vark, 1979). Thus, many ecological data sets cannot be analysed by DA without dropping many species.

Data for DA typically consist of observations for which there is a grouping index (e.g., habitat type, geographic characteristics) and an associated vector of measurements (species abundance, habitat structural characteristics). One objective of the analysis is to predict the group to which an observation belongs, based on its measurement values, from which predictive equations that are called discriminant functions can be derived. Such a formulation is called predictive DA. Alternatively, the objective may be to exhibit optimal 'separation' of groups, based on certain linear functions resulting from linear transforms of the measurement variables that are called canonical variates. This latter approach is called descriptive discriminant analysis. Both descriptive and predictive methods have been used in ecological studies, though most have had a descriptive orientation (Williams, 1983).

Predictive discrimination involves the classification of observations by means of the measured values x, thus it might be of interest to determine which of several congeneric species is most likely to utilize a given plot within some heterogeneous habitat. Habitat features measured on the plot can be used to predict species utilisation in an optimal manner.

Though the most active areas of statistical research in DA have traditionally concerned predictive evaluations, in ecology, most applications have taken a descriptive approach. Ecologists are generally interested in the parameters that separate populations, on the assumption that the operation of natural selection will be reflected in among-group differences of these parameters. It is felt that with sufficient biological insight the associated mathematical constructs can be given an ecological interpretation. In practice, this leads to an emphasis on the canonical variates, which are interpreted by means of the signs and magnitudes of their loadings (Williams, 1983). Nevertheless, a predictive approach based on the derivation of discriminant functions may be more suitable to address biodiversity related links. A detailed presentation of a stepwise approach to the use of discrimination functions in ecology can be found in Guisan and Zimmermann (2000).

To interpret the ordination axes, one can use the canonical coefficients and the intraset correlations. The canonical coefficients define the ordination axes as linear combinations of the environmental variables and the intraset correlations are the correlation coefficients between the environmental variables and these ordination axes.

The canonical coefficients give the same information as the intraset correlations, if the environmental variables are mutually uncorrelated, but may provide rather different information if the environmental variables are correlated among one another, as they usually are in field data. When the environmental variables are strongly correlated with one another, the effects of different environmental variables on the species composition cannot be singled out and, consequently, the canonical coefficients will be unstable (Williams, 1983).

The results of DA are affected by non-linear transformations of the species data, but not by linear transformations, although the later can be considered if there is some reason to do so (Jongman, Ter Braak, and van Tongeren, 1995).

Several tools are available to perform DA, e.g. the *lda()* function from the *MASS* package in R, the *DiscriMiner* R package hosts a range of functions for discriminant analyses, and the generic *predict()* function can be used to classify unknown objects into the classes of an Linear Discriminant Analysis R object. Several commercial statistical packages like CANOCO, SPSS, MINITAB, among others, also include tools to perform DA.

## 6.2 Generalised Dissimilarity Modelling

Generalised Dissimilarity Models (GDM) are statistical techniques for analysing and predicting spatial patterns of turnover in community composition (beta diversity) across large regions (Ferrier et al., 2007). The approach extends matrix regression to accommodate two types of nonlinearity commonly encountered in large-scaled ecological data sets: (1) the curvilinear relationship between increasing ecological distance, and observed compositional dissimilarity, between sites; and (2) the variation in the rate of compositional turnover at different positions along environmental gradients. Thus, GDM addresses the spatial variation in biodiversity between pairs of geographical locations to make predictions (in both space and time) and map biological patterns by transforming environmental predictor variables (Overton, Barker, and Price, 2009).

GDM can also be adapted to accommodate special types of biological and environmental data including, for example, information on phylogenetic relationships between species and information on barriers to dispersal between geographical locations. This modelling approach can be applied to a wide range of assessment activities including visualisation of spatial patterns in community composition, constrained environmental classification, distributional modelling of species or community types, survey gap analysis, conservation assessment, and climate-change impact assessment.

GDM software calculates the Bray-Curtis dissimilarities in species composition between sampled sites (in paired combinations of *i* and *j*), and then derives monotonically increasing functions for each of p environmental factors using a Generalised Linear Model and an exponential link function (Ferrier et al., 2007).

GDM software estimates the dissimilarities between other geographic locations based on their environmental conditions, and uses multidimensional scaling techniques to classify the study area into landscapes that were relatively homogeneous in environmental conditions and species composition (Ferrier et al., 2007). BIOCLIM predictors (Hijmans et al., 2005) that can be used as environmental predictors to generate GDM models are available at http://www.worldclim.org/. GDM software automatically removes environmental factors that do not significantly affect the turnover in species composition.

GDM software for the R statistical software environment developed by Irisson, mormede, and Raymond (2014) can be downloaded from https://github.com/jiho/atlasr and similar software developed by Fitzpatrick et al. (2013) can be downloaded from http://datadryad.org/resource/doi:10.5061/dryad.81p60 (official version) and https://github.com/fitzLab-AL/GDM (development version).

## 6.3 Generalised Diversity-Interactions Modelling

Two decades of experimental work has led to various approaches to the analysis of the relationship between biodiversity and ecosystem function (BEF) in experimental systems (Connolly et al., 2013). Researchers have tried to capture the shape of the function of richness (linear, log, square root, etc) that best describes the effect of species loss on community function (Cardinale, Srivastava, et al., 2009; Schmid, Pfisterer, and Balvanera, 2009; Cardinale, 2011). The best fitting relationship is then used to estimate the effects of species loss and the rate of deceleration of the response as richness increases. Usually, it is accepted that the BEF relationship with richness must have an upper bound, i.e. it saturates (Hooper et al., 2005). Connolly et al. (2013) recognise that while a saturating relationship is more appropriate for theoretical and physical reasons, many transformations of richness used to produce a linear BEF relationship give a decelerating but not saturating mathematical relationship.

Generalised Diversity-Interactions Models (GDIM), as proposed by Connolly et al. (2013), aim at unifying existing approaches to BEF relationship by providing a common framework within which to explore the effects of environment, space and time on ecosystem properties. The unification of several approaches within a single model is probably the most important outcome of their work. GDIM models follow the general equation when we consider an interaction between a pair of species *i* and *j* (Connolly et al., 2013):

$$y = \sum_{i=1}^{s} \beta_i P_i + \alpha A + \sum_{i<j}^{s} \delta_{ij} \left(P_i P_j\right)^{\theta} + \varepsilon$$

$y$ is the functional response, $s$ is the species pool, $\beta_i$ is the contribution of species of order $i$ to the ecosystem function, $P_i$ and $P_j$ are the proportion of species $i$ and $j$ respectively, $\delta_{ij}$ is the potential of species $i$ and $j$ to interact, $\delta_{ij}P_iP_j$ is the contribution of the interaction of species $i$ and $j$ to the functional response, $\epsilon$ is a measure of error, and $\theta$ is the index that shapes the relationship. Effects of changing factors can be assessed by comparing values of interaction coefficients and $\theta$ according to the principles outlined by Connolly et al. (2013).

A major limitation to apply this approach in the frame of AQUACROSS case studies is linked with the fact that, to the best of our knowledge, there are no public routines or computational tools available to apply the framework outlined by Connolly et al. (2013). Furthermore, most of the variables within this model remain unknown to different case studies in AQUACROSS.

## 6.4 Integrated Ecosystem Services modelling approach – ARIES

The integrated ESS modelling methodology named ARIES (ARtificial Intelligence for Ecosystem Services), aims at improving conceptual detail and representation of ESS dynamics, in support of more accurate decision-making in diverse application contexts (Villa et al., 2014). By using computer learning and reasoning, model structure may be specialised for each application context without requiring costly expertise. For these reasons, ARIES can be a powerful tool in the context of AQUACROSS and case studies modelling and scenarios testing.

ARIES is assisted by model integration technologies that allow assembling customised models from a growing model base. It currently integrates various techniques (Villa et al., 2014):

1. **Geographical Information Systems** (GIS), to model the geographical knowledge system that allows to both locate human and natural elements of SES and analyse topological networks of relationships between ESS and their beneficiaries.
2. **Bayesian Belief Networks** (BBN), to model the behavioural component of agents located in space. BBN are directed acyclic graphs that allow a concise causal representation of processes and influences. Nodes in a BBN correspond to random variables, whose potential values (outcomes) are defined by a probability distribution (McCann, Marcot, and Ellis, 2006).
3. **Social Network Analysis** (SNA), to model the multi-level formal and informal social networks of relations among social agents, defining pathways for exchange of information and materials. SNA can, for instance, model the dynamics of cooperation and mutual aid that can play important roles in coping strategies, affecting the resilience of the system (Entwisle et al., 2008).
4. **System Dynamics** (SD), to simulate a system temporal evolution by tracking values of aggregated variables and using process-driven "stock and flow" logics (Martínez-López et al., 2015). SD allows complex dynamic interdependencies to be captured, including non-linear feedbacks.
5. **Agent-Based Modelling** (ABM), computationally intense and micro-detailed models where many heterogeneous agents interact at multiple temporal and spatial scales (Balbi and Giupponi, 2010). Models can be geographically referenced using data retrieved from a GIS environment, and can incorporate links among agents to take social structure into account (see SNA). ABM can ultimately serve as an integration platform for all the techniques described.

The interdisciplinarity required for the study of ESS is best tackled using integrated modelling tools that are able to represent the wide variety of interactions that happen within SES, such as those based on behaviour, market prices, local versus global economy, etc. Moreover, in view of the ongoing climate change, there is certainly an urgent need to integrate the different elements that compose SES (processes, agents, events, etc.) in order to enhance governance, understand indirect and nonlinear causal links, and be able to predict future scenarios.

By means of integrated modelling tools, such as ARIES, ESS mapping can be studied in combination with other ecological and socio-economical interactions that might exert pressures on ecosystems, thus enabling EBM approaches. Value, in economic terms, is a marginal notion. The sorts of marginal values most common to economic analysis are those associated with unit changes of resources. On the contrary, most ESS assessment exercises focus on the value of a certain ecosystem *per se*. The avoided interpretation of value in many ESS assessments has made them scarcely credible from an economic point of view. The only way to reconcile economic value and ESS assessments is to consider marginal changes in ecosystems. This can be done in ARIES with simulated experiments where increasing portions of ecosystems are modified and the relative effect on ESS is measured. This is also a way to identify possible tipping points in ecosystem functioning for ESS and to include resilience.

ARIES ESS models are computational representations of the environment that allow biophysical, ecological, and/or socio-economic characteristics to be quantified and explored. Furthermore, as models can explore scenarios, trade-offs that result from different scenarios can be assessed as well. ARIES has also considerable potential to evaluate both the ecosystem structure and function underlying ESS and the supply and demand for ESS themselves.

Too often, models are standing monoliths developed for the purpose of one specific case study and are scarcely generalisable. Reusability and integration of data artefacts and models are becoming increasingly fundamental in interdisciplinary science. ARIES allows a flexible definition of the system boundaries (e.g., the context in terms of space and time) and of the main elements under analysis. Models, therefore, adapt to the selected context and produce context dependent results. Moreover, ARIES encompasses biophysical, ecological, and socio-economic dimensions through dynamic processes of very different nature.

ARIES models are built by the network of its users. Community driven knowledge is, therefore, promoted in order to increase model availability. Users are able to make their knowledge available across the ARIES network. In this regard, ARIES promotes community ownership rather than proprietary interests. Since ESS models can belong to different domains of expertise, it is important to enable approaches that capture the complexity of description necessary to the simulation of coupled human-natural systems, without burdening users. In this regard, ARIES focuses more on studying multiple interactions and scenarios, rather than on developing individual finer scale models with very precise outputs.

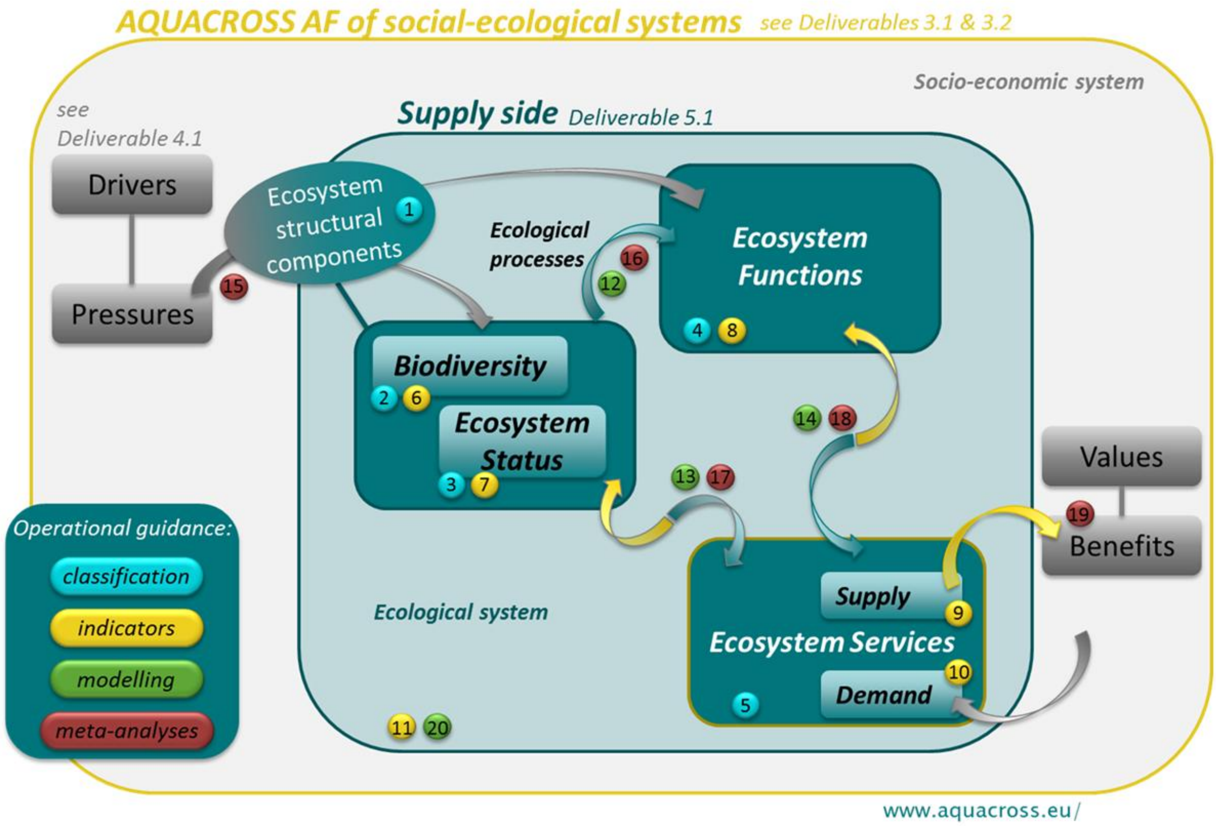
The nature of the targeted end-users' or developers' communities is also a key issue that must be taken into account when considering starting using a new modelling tool. Ideal modelling tools are as general and flexible as possible to suit the needs of both advanced and less skilled modellers (Martínez-López et al., 2015). In this regard, ARIES represents an adequately documented software tool that can be useful for non-programmers, and that is flexible enough so that advanced users can fully understand the role of each component and adapt them to case-specific requirements.

# 7 Guidance and Recommendations

The previous chapters discussed the different stages of the AQUACROSS AF that are related with the role of BD in maintaining functional ecosystems and warranting the provisioning of ESS. The aim was to provide a common understanding and highlight key points for the implementation of the AQUACROSS AF when assessing the supply side in the case studies, and eventual case scenarios beyond the project scope.

In this section, the essential aspects for operationalising this guidance are illustrated in Figure 8, through a conceptual diagram. This scheme provides an overview of the supporting information compiled in this report, regarding both research findings and resources available for effective assessments at specific stages of the supply side of the AQUACROSS AF (Table 8). The links to the socio-economic system are also considered in order to promote a fully integrated assessment.

Figure 8: Conceptual guidance for assessing ecosystems' integrity and ESS supply



The various **types of resources** compiled in this Deliverable 5.1 are linked to specific steps of the AF and are identified according to the nature of their content, namely: supporting classification schemes, potential indicators, suitable modelling approaches or evidences from meta-analyses (Figure 8).

To facilitate the use of those resources in AQUACROSS case studies, Table 8 includes links to the sections and/or annexes of this deliverable where detailed information is provided. In addition, the **main challenges** (C) identified for the implementation of the AQUACROSS supply-side conceptual diagram are indicated. The information generated at this stage of the project is also relevant for other tasks and stages of the AF, therefore such tasks are also identified in Table 8.

Finally, the AQUACROSS Project has selected eight case studies for testing the AQUACROSS AF in conjunction with stakeholders. Attending to the case studies main focus and objectives, some potentially relevant steps of the conceptual diagram and the respective available resources have been identified ( Table 9).

). These links are not exhaustive and intend to be suggestions to illustrate the concept applicability; requiring confirmation in the next stages of the process. Overall, Deliverable 5.1 implementation in the case studies will contribute to attain AQUACROSS goals regarding:

- showcasing specific elements of the objectives of the EU 2020 Biodiversity Strategy relevant for the management of aquatic ecosystems;
- understanding the most relevant challenges surrounding the protection of aquatic biodiversity; and
- maximising the lessons learnt and up-scale of results.

The eight **case studies in AQUACROSS** tackle a wide range of topics, which address the sustainability of nature resources' exploitation for the provisioning of different types of ESS (e.g., CS1 and CS2); the management of sectoral conflicts through integrated management (e.g., CS3, CS5 and CS8); or pressures and environmental impacts in biodiversity, such as those caused by invasive alien species or eutrophication processes (e.g., CS4, CS6 and CS7).

Although general biodiversity conservation concerns are core to all of them, the different case studies also take place within particular policy contexts and, therefore, target specific objectives set by the EU 2020 Biodiversity Strategy, by EU Directives and regulations (such as the Water Framework Directive, Habitat and Birds Directives, Common Fisheries Policy, and EU Invasive Alien Species Regulation), or conservation objectives for areas under special protection (such as Biosphere Reserves or Natura 2000 sites). In addition, for case studies operating in transboundary aquatic ecosystems (e.g., CS1, CS2 and CS3), the national-level environmental policies and goals need also to be harmonised and ultimately to converge. Often, these policies overlap spatially in the case studies' area; their requirements may not (see for example the AQUACROSS Deliverable 2.2 by O'Higgins et al., 2016).

Table 8: Main case study challenges identified for the implementation of the supply-side of the AQUACROSS AF and their relevant project tasks

| Link in conceptual diagram Figure 8 | Main challenges (*C*) | D5.1 resource | Relevance for tasks |
|---|---|---|---|
| **Classification schemes** | | | |
| 1 Ecosystem structural components (including abiotic features) | *C1* Identify the most relevant **components** of the **ecosystem** at different levels, from the sub-individual to the habitats, upon which **Pressures** may act, and which can be assessed and measured. These components are the **physical units** where Biodiversity and State changes are evaluated.<br>*C2* This classification targets one of the **boundaries** between the socio-economic and the ecological systems, thus it should enable linking them. | Annex I *Classification* | Tasks: 4.2, 5.2, 6.3 |
| 2 Biodiversity | *C3* Identify meaningful approaches for measuring Biodiversity attaining to AQUACROSS objective of unravelling **Biodiversity** role in supporting Ecosystem Functions and the provision of Ecosystem Services (**BEF and BES causal relationships**) across aquatic ecosystems.<br>*C4* This classification accounts for **scale issues** at different levels (e.g. at the ecosystem components, or at the spatial level) that enable identifying patterns at relevant scales. | Annex I *Classification* | Tasks: 5.2, 5.3, 6.3 |
| 3 Ecosystem status | *C5* Identify meaningful approaches for assessing **changes** in the state of the ecosystem caused by **anthropogenic activities**, which may alter the functioning of the ecosystems and compromise services provisioning, while taking on board assessment and requirements from **EU policies** in **aquatic environments** and biodiversity **conservation**. | Annex I *Classification* | Tasks: 2.5, 4.2, 5.2, 6.3 |
| 4 Ecosystem Functions | *C6* Identify meaningful ecosystem functions in **aquatic** ecosystems, along with the **ecological processes** and the ecosystem components that **sustain** such functioning.<br>*C7* Functions should facilitate a **link** with some ESS. | Annex I *Classification* | Tasks: 5.2, 6.3 |
| 5 Ecosystem Services | *C8* Adopt a classification that encompasses both **biological mediated** ESS but that considers also the **abiotic outputs** of the ecosystem; following as possible EU widely agreed approaches (e.g. CICES/ EU MAES). | Annex I *Classification* | Tasks: 5.2, 6.3 |

| Link in conceptual diagram Figure 8 | Main challenges (*C*) | D5.1 resource | Relevance for tasks |
|---|---|---|---|
| **Indicators** | | | |
| 6 Biodiversity<br>7 Ecosystem status<br>8 Ecosystem Functions<br>9 Ecosystem Services Supply | *C9* Make use of assessment tools implemented by MS within the context of existing **EU environmental policies** in **aquatic** biodiversity **conservation**.<br>*C10* Select stage specific indicators in order to promote complementarity and information flow and avoid overlap between these assessment stages.<br>*C11* **Distinguish** clearly between indicators of ecosystem **status**, ecosystem **functioning** and ESS **supply**, to avoid double reporting and overlap of information. | Annex I<br>*Indicators, indices & metrics* | Tasks:<br>2.5, 4.2, 5.2, 6.3, 8.1 |
| 10 Ecosystem Services Demand | *C12* **Distinguish** clearly between indicators of ESS **supply** and the **demand** for ESS.<br>*C13* Select indicators that establish a clear **link** with the **socio-economic system**. | Annex I<br>*Indicators, indices & metrics* | Tasks:<br>3.3, 4.2, 5.2, 6.3, 8.1 |
| 11 Selection of indicators | *C14* Overall, **selection** of **indicators** should follow minimum **quality criteria** to reduce uncertainty in the assessments along the different stages of the ecological system, and ultimately contribute to the robustness of the AQUACROSS AF outputs.<br>*C15* **Prioritise** the selection of **indicators** that can be integrated into **modelling approaches**.<br>*C16* Ensure that the selection of indicator meets **stakeholders** perception and **expectations**. | Chapter 5, in particular<br>Section 5.1.1<br>*Criteria for selecting indicators* | Tasks:<br>1.1, 3.3, 4.2, 5.2, 8.1 |
| **Modelling** | | | |
| 12 BEF causal relationships<br>13 BES causal relationships<br>14 EF-ESS causal relationships | For predicting outcomes of the interactions between the socio-ecological systems, the modelling approaches should account for:<br>*C17* Multi-species/habitats **complex interactions**;<br>*C18* The role of **environmental factors** (e.g. **abiotic** variables; anthropogenic **pressures**);<br>*C19* **Synergies** between ESS;<br>*C20* **Trade-offs** between ESS;<br>*C21* **Spatial** and/or **temporal heterogeneity** in the environmental domain (see C17) but also in the | Chapters 2,3<br>*BEF & BES relationships*<br>Chapter 4<br>*Meta-analyses*<br>Chapter 6<br>*Modelling* | Tasks:<br>5.2, 5.3, 7.2, 7.4 |

| Link in conceptual diagram Figure 8 | Main challenges (*C*) | D5.1 resource | Relevance for tasks |
|---|---|---|---|
| | exploitation of the natural resources by the **socio-economic** activities. | | |
| 20 Integrative modelling approaches | *C22* Provide that the **AQUACROSS** case studies' specific **models contribute** for **identifying** overall **BEF** and **BES patterns** across **aquatic ecosystems**.<br>*C23* Gather **sufficient data** from different stages of the AQUACROSS AF to allow an integrative modelling approach in each **case study**. | Chapter 4<br>*Meta-analyses*<br>Chapter 6<br>*Modelling* | Tasks:<br>5.2, 5.3, 7.2, 7.4 |
| **Meta-analyses** | | | |
| Flows from P to BD and EF; or from BD to EF or to ESS; and from BD or ESS to Benefits (15, 16, 17, 18, 19). | *C24* **Test** if the **hypothesis** and **evidences** found in **BEF** and **BES** overall research are **transferable** into **aquatic** domains.<br>*C25* Provide that the chosen analysis has **relevance** both for **stakeholders** and **Policy.** | Chapters 2,3<br>*BEF & BES relationships*<br>Annex II<br>*Compilation meta-analyses* | Tasks:<br>1.1, 2.5, 5.2, 7.2, 7.4, 8.1 |

Legend: challenges (C), pressures (P), biodiversity (BD), ecosystem functions (EF), ecosystem services (ESS), biodiversity-ecosystem functions (BEF), biodiversity-ecosystem services (BES), assessment framework (AF). This table provides an overview of case study challenges (see conceptual diagram of Figure 8) and different type of resources available in this Deliverable 5.1 for addressing those challenges. Tasks, in the AQUACROSS Project, for which information generated is relevant are also identified.

In practice, these heterogeneous scenarios are expected to lead to different type of assessments and approaches which are ideal for testing the robustness of the AQUACROSS AF concept. Focusing specifically in the **AF supply-side**, it is thus anticipated that a wide variety of BD components is assessed, focusing on diverse structure and/or functional features, and performed at different scales by each of the case studies. Similarly, the selection of EFs (and associated ecological processes) and of ESS to be investigated will differ in function of local or regional objectives and constraints, such as relevant pressures or societal demands. Hence, the causal links analysed in a particular case study may cover all the supply-side or focus either towards upstream (with drivers and pressures) or downstream (ESS demand and benefits) in the causal chain (Figure 8 and Table 9).

Table 9: Potential application of steps of the supply-side assessment in AQUACROSS case studies

| AQUACROSS case studies | Aquatic realms | | | | | | | Potentially relevant steps from conceptual diagram (e.g.) |
|---|---|---|---|---|---|---|---|---|
| | Lakes | Rivers | Wetlands | Inlets & Transitional | Coastal | Shelf | Ocean | |
| CS1 Trade-offs in ecosystem-based fisheries management in the North Sea aimed at achieving Biodiversity Strategy targets | | | | | X | X | | 1, 3, 5, 7, 9, 10, 15, 19, 20 |
| CS2 Analysis of transboundary water ecosystems and green/blue infrastructures in the Intercontinental Biosphere Reserve of the Mediterranean Andalusia (Spain) - Morocco | | | X | | X | X | | 1, 5, 7, 9, 10, 15 |
| CS3 Danube River Basin - harmonising inland, coastal and marine ecosystem management to achieve aquatic biodiversity targets | | X | | X | X | | | 1, 2, 5, 6, 10, 15, 20 |
| CS4 Management and impact of Invasive Alien Species (IAS) in Lough Erne in Ireland | X | X | X | | | | | 1, 2, 6, 9, 15, 20 |
| CS5 Improving integrated management of Natura 2000 sites in the Vouga River, from catchment to coast, Portugal | X | X | X | X | X | | | 1, 2, 4, 5, 6, 8, 10, 12, 13, 20 |
| CS6 Understanding eutrophication processes and restoring good water quality in Lake Ringsjön - Rönne å Catchment in Kattegat, Sweden | X | | X | | | | | 3, 5, 7, 9, 15 |
| CS7 Biodiversity management for rivers of the Swiss Plateau | | X | | | | | | 3, 5, 7, 15, 20 |
| CS8 Ecosystem-based solutions to solve sectoral conflicts on the path to sustainable development in the Azores | X | X | | | X | X | X | 3, 4, 5, 7, 8, 9, 13, 14 |

Legend: The case studies (CS) represent different aquatic domains from freshwaters (blue) to saline environments (dark blue), and also ecosystems at the land-water interface (yellow), with some CS covering several ecosystem types. Source for case studies details: [www.aquacross.eu/](www.aquacross.eu/)

# 8 References


Albouy, C., F. Guilhaumon, S. Villéger, M. Mouchet, L. Mercier, JM Culioli, JA Tomasini, F Le Loc'h, and D. Mouillot. 2011. "Predicting Trophic Guild and Diet Overlap from Functional Traits: Statistics, Opportunities and Limitations for Marine Ecology." *Marine Ecology Progress Series* 436 (August): 17–28. doi:10.3354/meps09240.

Allgeier, Jacob E, Craig A Layman, Peter J Mumby, and Amy D Rosemond. 2014. "Consistent Nutrient Storage and Supply Mediated by Diverse Fish Communities in Coral Reef Ecosystems." *Global Change Biology* 20 (8): 2459–72. doi:10.1111/gcb.12566.

Armstrong, Claire W., Naomi S. Foley, Rob Tinch, and Sybille van den Hove. 2012. "Services from the Deep: Steps towards Valuation of Deep Sea Goods and Services." *Ecosystem Services* 2. Elsevier: 2–13. doi:10.1016/j.ecoser.2012.07.001.

Balbi, Stefano, and Carlo Giupponi. 2010. "Agent-Based Modelling of Socio-Ecosystems." *International Journal of Agent Technologies and Systems* 2 (4): 17–38. doi:10.4018/jats.2010100103.

Balvanera, Patricia, Andrea B Pfisterer, Nina Buchmann, Jing-Shen He, Tohru Nakashizuka, David Raffaelli, and B. Schmid. 2006. "Quantifying the Evidence for Biodiversity Effects on Ecosystem Functioning and Services." *Ecology Letters* 9 (10): 1146–56. doi:10.1111/j.1461-0248.2006.00963.x.

Balvanera, Patricia, Ilyas Siddique, Laura Dee, Alain Paquette, Forest Isbell, Andrew Gonzalez, Jarrett E K Byrnes, Mary I. O'Connor, Bruce A. Hungate, and John N. Griffin. 2014. "Linking Biodiversity and Ecosystem Services: Current Uncertainties and the Necessary next Steps." *BioScience* 64 (1): 49–57. doi:10.1093/biosci/bit003.

Barbier, Edward B, Sally D. Hacker, Chris; Kennedy, Evamaria W. Kock, Adrian C. Stier, and Brian R. Sillman. 2011. "The Value of Estuarine and Coastal Ecosystem Services." *Ecological Monographs* 81 (2): 169–93. doi:10.1890/10-1510.1.

Barbier, Edward B, E W Koch, B R Silliman, S D Hacker, E Wolanski, J Primavera, E F Granek, et al. 2008. "Coastal Ecosystem-Based Management with Nonlinear Ecological Functions and Values." Journal Article. *Science* 319 (5861). American Association for the Advancement of Science: 321–23. doi:10.1126/science.1150349.

Bartkowski, Bartosz, Nele Lienhoop, and Bernd Hansjürgens. 2015. "Capturing the Complexity of Biodiversity: A Critical Review of Economic Valuation Studies of Biological Diversity." *Ecological Economics* 113. Elsevier B.V.: 1–14. doi:10.1016/j.ecolecon.2015.02.023.

Belley, Rénald, and Paul V.R. Snelgrove. 2016. "Relative Contributions of Biodiversity and

Environment to 2." *Frontiers in Marine Science* 3 (November). doi:10.3389/fmars.2016.00242.

Bennett, Elena M, Wolfgang Cramer, Alpina Begossi, Georgina Cundill, Sandra Díaz, Benis N Egoh, Ilse R. Geijzendorffer, et al. 2015. "Linking Biodiversity, Ecosystem Services, and Human Well-Being: Three Challenges for Designing Research for Sustainability." *Current Opinion in Environmental Sustainability* 14: 76–85. doi:10.1016/j.cosust.2015.03.007.

Berg, Torsten, Karin Fürhaupter, Heliana Teixeira, Laura Uusitalo, and Nikolaos Zampoukas. 2015. "The Marine Strategy Framework Directive and the Ecosystem-Based Approach – Pitfalls and Solutions." *Marine Pollution Bulletin* 96 (1–2). Elsevier Ltd: 18–28. doi:10.1016/j.marpolbul.2015.04.050.

Best, R J, N C Caulk, and John J. Stachowicz. 2012. "Trait vs. Phylogenetic Diversity as Predictors of Competition and Community Composition in Herbivorous Marine Amphipods." Journal Article. Edited by J Emmett Duffy. *Ecology Letters* 16 (1): 72–80. doi:10.1111/ele.12016.

Bigelow, John, and Robert Pargetter. 1987. "Functions." *The Journal of Philosophy* 84 (4): 181. doi:10.2307/2027157.

Birk, Sebastian, Wendy Bonne, Ángel Borja, Sandra Brucet, Anne Courrat, Sandra Poikane, Angelo Solimini, Wouter Van De Bund, Nikolaos Zampoukas, and Daniel Hering. 2012. "Three Hundred Ways to Assess Europe's Surface Waters: An Almost Complete Overview of Biological Methods to Implement the Water Framework Directive." *Ecological Indicators* 18. Elsevier Ltd: 31–41. doi:10.1016/j.ecolind.2011.10.009.

Birkhofer, Klaus, Eva Diehl, Jesper Andersson, Johan Ekroos, Andrea Früh-Müller, Franziska Machnikowski, Viktoria L Mader, et al. 2015. "Ecosystem Services – Current Challenges and Opportunities for Ecological Research." Journal Article. *Frontiers in Ecology and Evolution* 2 (January). Frontiers: 413. doi:10.3389/fevo.2014.00087.

Biswas, Shekhar R, and Azim U Mallik. 2011. "Species Diversity and Functional Diversity Relationship Varies with Disturbance Intensity." Journal Article. *Ecosphere* 2 (4): art52. doi:10.1890/ES10-00206.1.

Böhnke-Henrichs, Anne, Corinne Baulcomb, Rebecca Koss, S. Salman Hussain, and Rudolf S. de Groot. 2013. "Typology and Indicators of Ecosystem Services for Marine Spatial Planning and Management." *Journal of Environmental Management* 130 (November). Elsevier Ltd: 135–45. doi:10.1016/j.jenvman.2013.08.027.

Borja, Ángel, Michael Elliott, Jesper H Andersen, Torsten Berg, Jacob Carstensen, Benjamin S. Halpern, Anna-Stiina Heiskanen, et al. 2016. "Overview of Integrative Assessment of Marine Systems: The Ecosystem Approach in Practice." *Frontiers in Marine Science* 3 (55): 68. doi:10.3389/fmars.2016.00020.

Borja, Ángel, Theo Prins, Nomiki Simboura, Jesper H Andersen, Torsten Berg, João Carlos Marques, Joao M Neto, Johnny Reker, Heliana Teixeira, and Laura Uusitalo. 2014. "Tales from a Thousand and One Ways to Integrate Marine Ecosystem Components When Assessing the Environmental Status Tales from a Thousand and One Ways to Integrate Marine Ecosystem Components When Assessing the Environmental Status." doi:10.3389/fmars.2014.00022.

Boyer, Katharyn E, Johanna S Kertesz, and John F Bruno. 2009. "Biodiversity Effects on Productivity and Stability of Marine Macroalgal Communities: The Role of Environmental Context." Journal Article. *Oikos* 118 (7). Blackwell Publishing Ltd: 1062–72. doi:10.1111/j.1600-0706.2009.17252.x.

Boyero, Luz, Bradley J Cardinale, Mikis Bastian, and Richard G Pearson. 2014. "Biotic vs. Abiotic Control of Decomposition: A Comparison of the Effects of Simulated Extinctions and Changes in Temperature." Journal Article. Edited by Francesco de Bello. *PLoS ONE* 9 (1). Public Library of Science: e87426. doi:10.1371/journal.pone.0087426.

Bracken, Matthew E S, and Natalie H N Low. 2012. "Realistic Losses of Rare Species Disproportionately Impact Higher Trophic Levels." Journal Article. *Ecology Letters* 15 (5). Blackwell Publishing Ltd: 461–67. doi:10.1111/j.1461-0248.2012.01758.x.

Bracken, Matthew E S, and Susan L Williams. 2013. "Realistic Changes in Seaweed Biodiversity Affect Multiple Ecosystem Functions on a Rocky Shore." Journal Article. *Ecology* 94 (9): 1944–54. doi:10.1890/12-2182.1.

Bruno, John F, Katharyn E Boyer, JE Duffy, Sarah C Lee, and Johanna S Kertesz. 2005. "Effects of Macroalgal Species Identity and Richness on Primary Production in Benthic Marine Communities." Journal Article. *Ecology Letters* 8 (11). Blackwell Science Ltd: 1165–74. doi:10.1111/j.1461-0248.2005.00823.x.

Bruno, John F, and Bradley J Cardinale. 2008. "Cascading Effects of Predator Richness." Journal Article. *Frontiers in Ecology and the Environment* 6 (10): 539–46. doi:10.1890/070136.

Bruno, John F, and Mary I O'Connor. 2005. "Cascading Effects of Predator Diversity and Omnivory in a Marine Food Web." Journal Article. *Ecology Letters* 8 (10). Blackwell Science Ltd: 1048–56. doi:10.1111/j.1461-0248.2005.00808.x.

Bulleri, Fabio, Andrea Cucco, Martina Dal Bello, Elena Maggi, Chiara Ravaglioli, and Lisandro Benedetti-Cecchi. 2016. "The Role of Wave-Exposure and Human Impacts in Regulating the Distribution of Alternative Habitats on NW Mediterranean Rocky Reefs." *Estuarine, Coastal and Shelf Science*, February. doi:10.1016/j.ecss.2016.02.013.

Burkhard, Benjamin, Marion Kandziora, Ying Hou, and Felix Müller. 2014. "Ecosystem Service Potentials, Flows and Demands-Concepts for Spatial Localisation, Indication and

Quantification." *Landscape Online* 34 (1): 1–32. doi:10.3097/LO.201434.

Burkhard, Benjamin, Franziska Kroll, Stoyan Nedkov, and Felix Muller. 2012. "Mapping Ecosystem Service Supply, Demand and Budgets." *Ecological Indicators* 21. Elsevier Ltd: 17–29. doi:10.1016/j.ecolind.2011.06.019.

Byrnes, Jarrett E. K., Lars Gamfeldt, Forest Isbell, Jonathan S. Lefcheck, John N. Griffin, Andy Hector, Bradley J Cardinale, David U. Hooper, Laura E. Dee, and J. Emmett Duffy. 2014. "Investigating the Relationship between Biodiversity and Ecosystem Multifunctionality: Challenges and Solutions." Edited by Robert Freckleton. *Methods in Ecology and Evolution* 5 (2): 111–24. doi:10.1111/2041-210X.12143.

Byrnes, Jarrett E., Pamela L. Reynolds, and John J. Stachowicz. 2007. "Invasions and Extinctions Reshape Coastal Marine Food Webs." Edited by David Lusseau. *PLoS ONE* 2 (3): e295. doi:10.1371/journal.pone.0000295.

Byrnes, Jarrett E., and John J. Stachowicz. 2009. "The Consequences of Consumer Diversity Loss: Different Answers from Different Experimental Designs." *Ecology* 90 (10): 2879–88. doi:10.1890/08-1073.1.

Cadotte, Marc W., Lea R. Mehrkens, and Duncan N. L. Menge. 2012. "Gauging the Impact of Meta-Analysis on Ecology." *Evolutionary Ecology* 26 (5): 1153–67. doi:10.1007/s10682-012-9585-z.

Caliman, Adriano, Luciana S Carneiro, João J F Leal, Vinicius F Farjalla, Reinaldo L Bozelli, and Francisco A Esteves. 2013. "Biodiversity Effects of Ecosystem Engineers Are Stronger on More Complex Ecosystem Processes." Journal Article. *Ecology* 94 (9): 1977–85. doi:10.1890/12-1385.1.

Campbell, Veronik, Grace Murphy, and Tamara N Romanuk. 2011. "Experimental Design and the Outcome and Interpretation of Diversity-stability Relations." Journal Article. *Oikos* 120 (3). Blackwell Publishing Ltd: 399–408. doi:10.1111/j.1600-0706.2010.18768.x.

Capps, Krista A, Carla L Atkinson, and Amanda T Rugenski. 2015. "Implications of Species Addition and Decline for Nutrient Dynamics in Fresh Waters." Journal Article. *Freshwater Science* 34 (2): 485–96. doi:10.1086/681095.

Cardinale, Bradley J. 2011. "Biodiversity Improves Water Quality through Niche Partitioning." *Nature* 472 (7341): 86–89. doi:10.1038/nature09904.

Cardinale, Bradley J., Diane S. Srivastava, J. Emmett Duffy, Justin P. Wright, Amy L. Downing, Mahesh Sankaran, Claire Jouseau, et al. 2009. "Effects of Biodiversity on the Functioning of Ecosystems: A Summary of 164 Experimental Manipulations of Species Richness." Edited by W. K. Michener. *Ecology* 90 (3): 854–854. doi:10.1890/08-1584.1.

Cardinale, Bradley J, Danuta M Bennett, Craig E Nelson, and Kevin Gross. 2009. "Does

Productivity Drive Diversity or Vice Versa? A Test of the Multivariate Productivity–Diversity Hypothesis in Streams." Journal Article. *Ecology* 90 (5): 1227–41.

Cardinale, Bradley J, J. Emmett Duffy, Andrew Gonzalez, David U. Hooper, Charles Perrings, Patrick A Venail, Anita Narwani, et al. 2012. "Biodiversity Loss and Its Impact on Humanity." *Nature* 486 (7401): 59–67. doi:10.1038/nature11148.

Cardinale, Bradley J, Kristin L. Matulich, David U. Hooper, Jarrett E K Byrnes, J. Emmett Duffy, Lars Gamfeldt, Patricia Balvanera, Mary I. O'Connor, and Andrew Gonzalez. 2011. "The Functional Role of Producer Diversity in Ecosystems." *American Journal of Botany* 98 (3): 572–92. doi:10.3732/ajb.1000364.

Cardinale, Bradley J, Karen Nelson, and Margaret A Palmer. 2000. "Linking Species Diversity to the Functioning of Ecosystems: On the Importance of Environmental Context." Journal Article. *Oikos* 91 (1). Munksgaard: 175–83. doi:10.1034/j.1600-0706.2000.910117.x.

Cardinale, Bradley J, and Margaret A. Palmer. 2002. "Disturbance Moderates Biodiversity–Ecosystem Function Relationships: Experimental Evidence from Caddisflies in Stream Mesocosms." *Ecology* 83 (7): 1915–27. doi:10.1890/0012-9658(2002)083[1915:DMBEFR]2.0.CO;2.

Cardinale, Bradley J, Diane S Srivastava, J Emmett Duffy, Justin P Wright, Amy L Downing, Mahesh Sankaran, and Claire Jouseau. 2006. "Effects of Biodiversity on the Functioning of Trophic Groups and Ecosystems." *Nature* 443 (7114): 989–92. doi:10.1038/nature05202.

Cardinale, Bradley J, J. P. Wright, M. W. Cadotte, I. T. Carroll, A. Hector, D. S. Srivastava, M. Loreau, and J. J. Weis. 2007. "Impacts of Plant Diversity on Biomass Production Increase through Time because of Species Complementarity." *Proceedings of the National Academy of Sciences* 104 (46). National Acad Sciences: 18123–28. doi:10.1073/pnas.0709069104.

Connolly, John, Thomas Bell, Thomas Bolger, Caroline Brophy, Timothee Carnus, John A Finn, Laura Kirwan, et al. 2013. "An Improved Model to Predict the Effects of Changing Biodiversity Levels on Ecosystem Function." Edited by Scott Chamberlain. *Journal of Ecology* 101 (2): 344–55. doi:10.1111/1365-2745.12052.

Cook, Geoffrey S, Pamela J Fletcher, and Christopher R Kelble. 2014. "Towards Marine Ecosystem Based Management in South Florida: Investigating the Connections among Ecosystem Pressures, States, and Services in a Complex Coastal System." Journal Article. *Ecological Indicators* 44: 26–39. doi:10.1016/j.ecolind.2013.10.026.

Cooper, Philip. 2013. "Socio-Ecological Accounting: DPSWR, a Modified DPSIR Framework, and Its Application to Marine Ecosystems." *Ecological Economics* 94. Elsevier B.V.: 106–15. doi:10.1016/j.ecolecon.2013.07.010.

Costa, Karla G., and Sérgio A. Netto. 2014. “Effects of Small-Scale Trawling on Benthic Communities of Estuarine Vegetated and Non-Vegetated Habitats.” *Biodiversity and Conservation* 23 (4): 1041–55. doi:10.1007/s10531-014-0652-3.

Costanza, Robert, Ralph d’Arge, Rudolf de Groot, Stephen Farber, Monica Grasso, Bruce Hannon, Karin Limburg, et al. 1997. “The Value of the World’s Ecosystem Services and Natural Capital.” *Nature* 387 (6630): 253–60. doi:10.1038/387253a0.

Covich, Alan P., Melanie C. Austen, Felix Bärlocher, Eric Chauvet, Bradley J Cardinale, Catherine L. Biles, Pablo Inchausti, et al. 2004. “The Role of Biodiversity in the Functioning of Freshwater and Marine Benthic Ecosystems.” *BioScience*. doi:10.1641/0006-3568(2004)054[0767:TROBIT]2.0.CO;2.

De Laender, Frederik, Carlos J Melian, Richard Bindler, Paul J Van den Brink, Michiel Daam, Helene Roussel, Jonas Juselius, Dirk Verschuren, and Colin R Janssen. 2013. “The Contribution of Intra- and Interspecific Tolerance Variability to Biodiversity Changes along Toxicity Gradients.” Journal Article. Edited by Helmut Hillebrand. *Ecology Letters* 17 (1): 72–81. doi:10.1111/ele.12210.

de Moura Queirós, Ana, Jan Geert Hiddink, Gareth Johnson, Henrique Nogueira Cabral, and Michel J. Kaiser. 2011. “Context Dependence of Marine Ecosystem Engineer Invasion Impacts on Benthic Ecosystem Functioning.” Journal Article. *Biological Invasions* 13 (5). Springer Netherlands: 1059–75. doi:10.1007/s10530-011-9948-3.

Delgado-Baquerizo, Manuel, Fernando T Maestre, Peter B Reich, Thomas C Jeffries, Juan J Gaitan, Daniel Encinar, Miguel Berdugo, Colin D Campbell, and Brajesh K Singh. 2016. “Microbial Diversity Drives Multifunctionality in Terrestrial Ecosystems.” Journal Article. *Nature Communications* 7: 10541. doi:10.1038/ncomms10541.

Dell’Anno, Antonio, Francesca Beolchini, Laura Rocchetti, Gian Marco Luna, and Roberto Danovaro. 2012. “High Bacterial Biodiversity Increases Degradation Performance of Hydrocarbons during Bioremediation of Contaminated Harbor Marine Sediments.” Journal Article. *Environmental Pollution* 167 (August): 85–92. doi:10.1016/j.envpol.2012.03.043.

Diaz, Sandra, Sebsebe Demissew, Julia Carabias, Carlos Joly, Mark Lonsdale, Neville Ash, Anne Larigauderie, et al. 2015. “The IPBES Conceptual Framework — Connecting Nature and People.” *Current Opinion in Environmental Sustainability* 14: 1–16. doi:10.1016/j.cosust.2014.11.002.

Downing, Amy L, Bryan L Brown, and Mathew A Leibold. 2014. “Multiple Diversity-Stability Mechanisms Enhance Population and Community Stability in Aquatic Food Webs.” Journal Article. *Ecology* 95 (1): 173–84.

Duffy, J. E. 2009. “Why Biodiversity Is Important to the Functioning of Real-World

Ecosystems." *Frontiers in Ecology and the Environment* 7 (8): 437–44. doi:10.1890/070195.

Duffy, JE. 2006. "Biodiversity and the Functioning of Seagrass Ecosystems." *Marine Ecology Progress Series* 311 (April): 233–50. doi:10.3354/meps311233.

Duffy, JE, Bradley J Cardinale, Kristin E France, Peter B McIntyre, Elisa Thébault, and Michel Loreau. 2007. "The Functional Role of Biodiversity in Ecosystems: Incorporating Trophic Complexity." Journal Article. *Ecology Letters* 10 (6). Blackwell Publishing Ltd: 522–38. doi:10.1111/j.1461-0248.2007.01037.x.

Duncan, Clare, Julian R Thompson, and Nathalie Pettorelli. 2015. "The Quest for a Mechanistic Understanding of Biodiversity–ecosystem Services Relationships." *Proceedings of the Royal Society B-Biological Sciences* 282 (1817). The Royal Society: 20151348. doi:10.1098/rspb.2015.1348.

Dussault, Antoine C., and Frédéric Bouchard. 2016. "A Persistence Enhancing Propensity Account of Ecological Function to Explain Ecosystem Evolution." *Synthese*, May. Springer Netherlands, 1–31. doi:10.1007/s11229-016-1065-5.

Edwards, Kyle F, Kristin M Aquilino, Rebecca J Best, Kirsten L Sellheim, and John J. Stachowicz. 2010. "Prey Diversity Is Associated with Weaker Consumer Effects in a Meta-Analysis of Benthic Marine Experiments." Journal Article. *Ecology Letters* 13 (2). Blackwell Publishing Ltd: 194–201. doi:10.1111/j.1461-0248.2009.01417.x.

EEA. 2015. "State of Europe's Seas." Luxembourg. doi:10.2800/64016.

Egoh, B, M B Dunbar, Joachim Maes, L Willemen, and E G Drakou. 2012. "Indicators for Mapping Ecosystem Services: A Review."

Ehrlich, Paul R., and Harold A. Mooney. 1983. "Extinction , Substitution , and Ecosystem Services." *BioScience* 33 (4): 248–54. doi:10.2307/1309037.

Elliott, M. 2011. "Marine Science and Management Means Tackling Exogenic Unmanaged Pressures and Endogenic Managed Pressures--a Numbered Guide." *Marine Pollution Bulletin* 62 (4): 651–55. doi:10.1016/j.marpolbul.2010.11.033.

Enquist, Brian J., Jon Norberg, Stephen P. Bonser, Cyrille Violle, Colleen T. Webb, Amanda Henderson, Lindsey L. Sloat, and Van M. Savage. 2015. "Scaling from Traits to Ecosystems: Developing a General Trait Driver Theory via Integrating Trait-Based and Metabolic Scaling Theories," February.

Entwisle, Barbara, George Malanson, Ronald R. Rindfuss, and Stephen J. Walsh. 2008. "An Agent-Based Model of Household Dynamics and Land Use Change." *Journal of Land Use Science* 3 (1): 73–93. doi:10.1080/17474230802048193.

Ericson, Jonas, Fredrik Ljunghager, and Lars Gamfeldt. 2009. "Are There Direct and Cascading Effects of Changes in Grazer and Predator Species Richness in a Model System with Heterogeneously Distributed Resources?" Journal Article. *Marine Biodiversity* 39 (1). Springer-Verlag: 71–81. doi:10.1007/s12526-009-0008-5.

Fabrègue, Mickaël, Agnès Braud, Sandra Bringay, Corinne Grac, Florence Le Ber, Danielle Levet, and Maguelonne Teisseire. 2014. "Discriminant Temporal Patterns for Linking Physico-Chemistry and Biology in Hydro-Ecosystem Assessment." *Ecological Informatics* 24 (November). Elsevier B.V.: 210–21. doi:10.1016/j.ecoinf.2014.09.003.

Farnsworth, Keith D., A. H. Adenuga, and R. S. de Groot. 2015. "The Complexity of Biodiversity: A Biological Perspective on Economic Valuation." *Ecological Economics* 120. Elsevier B.V.: 350–54. doi:10.1016/j.ecolecon.2015.10.003.

Farnsworth, Keith D., Olga Lyashevska, and Tak Fung. 2012. "Functional Complexity: The Source of Value in Biodiversity." *Ecological Complexity* 11. Elsevier B.V.: 46–52. doi:10.1016/j.ecocom.2012.02.001.

Farnsworth, Keith D., John Nelson, and Carlos Gershenson. 2013. "Living Is Information Processing: From Molecules to Global Systems." *Acta Biotheoretica* 61 (2): 203–22. doi:10.1007/s10441-013-9179-3.

Feio, M. J., T. Alves, M. Boavida, A. Medeiros, and M. A S GraçA. 2010. "Functional Indicators of Stream Health: A River-Basin Approach." *Freshwater Biology* 55 (5): 1050–65. doi:10.1111/j.1365-2427.2009.02332.x.

Ferrier, Simon, Glenn Manion, Jane Elith, and Karen Richardson. 2007. "Using Generalized Dissimilarity Modelling to Analyse and Predict Patterns of Beta Diversity in Regional Biodiversity Assessment." *Diversity and Distributions* 13 (3): 252–64. doi:10.1111/j.1472-4642.2007.00341.x.

Filip, Joanna, Barbara Bauer, Helmut Hillebrand, Anna Beniermann, Ursula Gaedke, and Stefanie D Moorthi. 2014. "Multitrophic Diversity Effects Depend on Consumer Specialization and Species-specific Growth and Grazing Rates." Journal Article. *Oikos* 123 (8). Blackwell Publishing Ltd: 912–22. doi:10.1111/oik.01219.

Finke, Deborah L, and Robert F Denno. 2005. "Predator Diversity and the Functioning of Ecosystems: The Role of Intraguild Predation in Dampening Trophic Cascades." Journal Article. *Ecology Letters* 8 (12). Blackwell Science Ltd: 1299–1306. doi:10.1111/j.1461-0248.2005.00832.x.

Finke, Deborah L, and William E Snyder. 2010. "Conserving the Benefits of Predator Biodiversity." Journal Article. *Biological Conservation* 143 (10): 2260–69. doi:10.1016/j.biocon.2010.03.022.

Fischer, Anke, and Antonia Eastwood. 2016. "Coproduction of Ecosystem Services as Human-nature interactions—An Analytical Framework." *Land Use Policy* 52 (March). Elsevier Ltd: 41–50. doi:10.1016/j.landusepol.2015.12.004.

Fisher, Brendan, R. Kerry Turner, and Paul Morling. 2009. "Defining and Classifying Ecosystem Services for Decision Making." *Ecological Economics* 68 (3). Elsevier B.V.: 643–53. doi:10.1016/j.ecolecon.2008.09.014.

Fitzpatrick, Matthew C., NJ Sanders, S Normand, J Svenning, S Ferrier, AD Gove, and RR Dunn. 2013. "Data from: Environmental and Historical Imprints on Beta Diversity: Insights from Variation in Rates of Species Turnover along Gradients. Dryad Digital Repository." *Dryad Digital Repository*. doi:10.5061/dryad.81p60.

Frainer, André, and Brendan G. McKie. 2015. "Chapter Seven – Shifts in the Diversity and Composition of Consumer Traits Constrain the Effects of Land Use on Stream Ecosystem Functioning." In *Advances in Ecological Research*, 52:169–200. doi:10.1016/bs.aecr.2015.03.002.

Frainer, André, Brendan G McKie, and Björn Malmqvist. 2013. "When Does Diversity Matter? Species Functional Diversity and Ecosystem Functioning across Habitats and Seasons in a Field Experiment." Journal Article. Edited by Guy Woodward. *Journal of Animal Ecology* 83 (2): 460–69. doi:10.1111/1365-2656.12142.

France, Kristin E, and J. Emmett Duffy. 2006. "Diversity and Dispersal Interactively Affect Predictability of Ecosystem Function." Journal Article. *Nature* 441 (7097): 1139–43. doi:10.1038/nature04729.

Fu, Hui, Jiayou Zhong, Guixiang Yuan, Leyi Ni, Ping Xie, and Te Cao. 2014. "Functional Traits Composition Predict Macrophytes Community Productivity along a Water Depth Gradient in a Freshwater Lake." Journal Article. *Ecology and Evolution* 4 (9): 1516–23. doi:10.1002/ece3.1022.

Fung, Tak, Keith D Farnsworth, David G Reid, and Axel G Rossberg. 2015. "Impact of Biodiversity Loss on Production in Complex Marine Food Webs Mitigated by Prey-Release." Journal Article 6: 6657. doi:10.1038/ncomms7657.

Gamfeldt, L, Js Lefcheck, and Jarrett E K Byrnes. 2014. "Marine Biodiversity and Ecosystem Functioning: What's Known and What's Next?" *PeerJ PrePrints*, 1–40. doi:http://dx.doi.org/10.7287/peerj.preprints.249v1.

Geijzendorffer, Ilse R., Eugenie C Regan, Henrique M. Pereira, Lluis Brotons, Neil Brummitt, Yoni Gavish, Peter Haase, et al. 2016. "Bridging the Gap between Biodiversity Data and Policy Reporting Needs: An Essential Biodiversity Variables Perspective." Edited by Marc Cadotte. *Journal of Applied Ecology* 53 (5): 1341–50. doi:10.1111/1365-2664.12417.

Giller, Paul, Helmut Hillebrand, Ulrike-G Berninger, Mark O Gessner, Stephen Hawkins, Pablo Inchausti, Cheryl Inglis, et al. 2004. "Biodiversity Effects on Ecosystem Functioning: Emerging Issues and Their Experimental Test in Aquatic Environments." Journal Article. *Oikos* 104 (3). Munksgaard: 423–36. doi:10.1111/j.0030-1299.2004.13253.x.

Glöckner, F O, J M Gasol, N McDonough, and J B Calewaert. 2012. "Marine Microbial Diversity and Its Role in Ecosystem Functioning and Environmental Change." Journal Article.

Godbold, Jasmin A. 2012. "Effects of Biodiversity-environment Conditions on the Interpretation of Biodiversity-function Relations." Book Section. In *Marine Biodiversity and Ecosystem Functioning*, 101–14. Oxford University Press. doi:10.1093/acprof:oso/9780199642250.003.0008.

Gómez, Carlos, Gonzalo Declacámara, Juan Arévalo-Torres, Julian Barbière, Julian Barbière, Ana Luisa Barbosa, and Alejandro Iglesias-campos Juan Arévalo-torres. 2016a. "The AQUACROSS Innovative Concept. Deliverable 3.1, European Union's Horizon 2020 Framework Programme for Research and Innovation Grant Agreement No. 642317."

Gómez, Carlos, Gonzalo Declacámara, and et al. 2016b. "Developing the AQUACROSS Assessment Framework." Deliverable 3.2. European Union's Horizon 2020 Framework Programme for Research and Innovation grant agreement No. 642317.

Gonçalves, Bárbara, Alexandra Marques, Amadeu Mortágua Velho Da Maia Soares, and Henrique M. Pereira. 2015. "Biodiversity Offsets: From Current Challenges to Harmonized Metrics." *Current Opinion in Environmental Sustainability* 14 (June): 61–67. doi:10.1016/j.cosust.2015.03.008.

GOOS BioEco Panel. 2016. "GOOS Biology and Ecosystems Expert Panel: April 2016."

Grabowski, Jonathan H., Robert D. Brumbaugh, Robert F. Conrad, Andrew G. Keeler, James J. Opaluch, Charles H. Peterson, Michael F. Piehler, Sean P. Powers, and Ashley R. Smyth. 2012. "Economic Valuation of Ecosystem Services Provided by Oyster Reefs." *BioScience* 62 (10): 900–909. doi:10.1525/bio.2012.62.10.10.

Graham, S Elizabeth, Jonathan M O'Brien, Teresa K Burrell, and Angus R McIntosh. 2015. "Aquatic Macrophytes Alter Productivity-Richness Relationships in Eutrophic Stream Food Webs." Journal Article. *Ecosphere* 6 (6): art89. doi:10.1890/ES14-00341.1.

Gravel, Dominique, Thomas Bell, Claire Barbera, Marine Combe, Thomas Pommier, and Nicolas Mouquet. 2012. "Phylogenetic Constraints on Ecosystem Functioning." Journal Article 3: 1117. doi:10.1038/ncomms2123.

Griffin, John N, Jarrett E K Byrnes, and Bradley J Cardinale. 2013. "Effects of Predator Richness on Prey Suppression: A Meta-Analysis." Journal Article. *Ecology* 94 (10): 2180–87. doi:10.1890/13-0179.1.

Griffin, John N, Stuart R Jenkins, Lars Gamfeldt, Douglas Jones, Stephen J Hawkins, and Richard C Thompson. 2009. "Spatial Heterogeneity Increases the Importance of Species Richness for an Ecosystem Process." Journal Article. *Oikos* 118 (9). Blackwell Publishing Ltd: 1335–42. doi:10.1111/j.1600-0706.2009.17572.x.

Gross, Kevin, and Bradley J Cardinale. 2005. "The Functional Consequences of Random vs. Ordered Species Extinctions." Journal Article. *Ecology Letters* 8 (4). Blackwell Science Ltd: 409–18. doi:10.1111/j.1461-0248.2005.00733.x.

Guisan, Antoine, and Niklaus E. Zimmermann. 2000. "Predictive Habitat Distribution Models in Ecology." *Ecological Modelling* 135 (2–3): 147–86. doi:10.1016/S0304-3800(00)00354-9.

Haines-Young, R, and M. Potschin. 2013. "Common International Classification of Ecosystem Services (CICES): Consultation on Version 4, August-December 2012." Report. *Access through Www. Cices. Eu*. European Environment Agency.

Handa, I Tanya, Rien Aerts, Frank Berendse, Matty P Berg, Andreas Bruder, Olaf Butenschoen, Eric Chauvet, et al. 2014. "Consequences of Biodiversity Loss for Litter Decomposition across Biomes." Journal Article. *Nature* 509 (7499): 218–21. doi:10.1038/nature13247.

Harrison, P.A., P.M. Berry, G. Simpson, J.R. Haslett, M. Blicharska, M. Bucur, R. Dunford, et al. 2014. "Linkages between Biodiversity Attributes and Ecosystem Services: A Systematic Review." *Ecosystem Services* 9 (September): 191–203. doi:10.1016/j.ecoser.2014.05.006.

Hattam, Caroline, Jonathan P. Atkins, Nicola Beaumont, Tobias Börger, Anne Böhnke-Henrichs, Daryl Burdon, Rudolf De Groot, et al. 2015. "Marine Ecosystem Services: Linking Indicators to Their Classification." *Ecological Indicators* 49: 61–75. doi:10.1016/j.ecolind.2014.09.026.

Hawkins, S. J. 2004. "Scaling up: The Role of Species and Habitat Patches in Functioning of Coastal Ecosystems." Journal Article. *Aquatic Conservation: Marine and Freshwater Ecosystems* 14 (3). John Wiley & Sons, Ltd.: 217–19. doi:10.1002/aqc.637.

Hector, Andy, and Robert Bagchi. 2007. "Biodiversity and Ecosystem Multifunctionality." Journal Article. *Nature* 448 (7150): 188–90. doi:10.1038/nature05947.

Hedges, Larry V, Jessica Gurevitch, and Peter S Curtis. 1999. "The Meta-Analysis of Response Ratios in Experimental Ecology." *Ecology* 80 (4): 1150. doi:10.2307/177062.

Heink, Ulrich, Jennifer Hauck, Kurt Jax, and Ulrich Sukopp. 2016. "Requirements for the Selection of Ecosystem Service Indicators – The Case of MAES Indicators." *Ecological Indicators* 61. Elsevier Ltd: 18–26. doi:10.1016/j.ecolind.2015.09.031.

Hensel, Marc J S, and Brian R Silliman. 2013. "Consumer Diversity across Kingdoms Supports

Multiple Functions in a Coastal Ecosystem." *Proceedings of the National Academy of Sciences of the United States of America* 110 (51): 20621–26. doi:10.1073/pnas.1312317110.

Hijmans, Robert J., Susan E. Cameron, Juan L. Parra, Peter G. Jones, and Andy Jarvis. 2005. "Very High Resolution Interpolated Climate Surfaces for Global Land Areas." *International Journal of Climatology* 25 (15): 1965–78. doi:10.1002/joc.1276.

Hines, Jes, Wim H van der Putten, Gerlinde B De Deyn, Cameron Wagg, Winfried Voigt, Christian Mulder, Wolfgang W Weisser, et al. 2015. "Towards an Integration of Biodiversity–Ecosystem Functioning and Food Web Theory to Evaluate Relationships between Multiple Ecosystem Services." Book Section. In *Ecosystem Services – From Biodiversity to Society, Part 1*, 53:161–99. Elsevier. doi:10.1016/bs.aecr.2015.09.001.

Hodapp, D, S Meier, F Muijsers, T H Badewien, and H Hillebrand. 2015. "Structural Equation Modeling Approach to the Diversity–Productivity Relationship of Wadden Sea Phytoplankton." Journal Article. *Marine Ecology Progress Series* 523: 31–40. doi:10.3354/meps11153.

Hooper, David U., F. S. Chapin, J. J. Ewel, A. Hector, P. Inchausti, S. Lavorel, J. H. Lawton, et al. 2005. "Effects of Biodiversity on Ecosystem Functioning: A Consensus of Current Knowledge." *Ecological Monographs* 75 (1): 3–35. doi:10.1890/04-0922.

Howe, Caroline, Helen Suich, Bhaskar Vira, and Georgina M Mace. 2014. "Creating Win-Wins from Trade-Offs? Ecosystem Services for Human Well-Being: A Meta-Analysis of Ecosystem Service Trade-Offs and Synergies in the Real World." *Global Environmental Change* 28 (September): 263–75. doi:10.1016/j.gloenvcha.2014.07.005.

Hughes, A Randall, Brian D Inouye, Marc T J Johnson, Nora Underwood, and Mark Vellend. 2008. "Ecological Consequences of Genetic Diversity." Journal Article. *Ecology Letters* 11 (6). Blackwell Publishing Ltd: 609–23. doi:10.1111/j.1461-0248.2008.01179.x.

Hughes, A Randall, and John J. Stachowicz. 2004. "Genetic Diversity Enhances the Resistance of a Seagrass Ecosystem to Disturbance." Journal Article. *Proceedings of the National Academy of Sciences* 101 (24). National Acad Sciences: 8998–9002. doi:10.1073/pnas.0402642101.

Hummel, Herman, Matt Frost, José a. Juanes, Judith Kochmann, Carlos F. Castellanos Perez Bolde, Fernando Aneiros, François Vandenbosch, et al. 2015. "A Comparison of the Degree of Implementation of Marine Biodiversity Indicators by European Countries in Relation to the Marine Strategy Framework Directive (MSFD)." *Journal of the Marine Biological Association of the United Kingdom*, no. June: 1–13. doi:10.1017/S0025315415000235.

Huston, Michael A. 1979. "A General Hypothesis of Species Diversity." *The American*

*Naturalist* 113 (1): 81–101. doi:10.1086/283366.

Irisson, J.-O., S. Mormede, and B. Raymond. 2014. "Atlasr – R Code for the Biogeographic Atlas of the Southern Ocean."

Isbell, Forest, Vincent Calcagno, Andy Hector, John Connolly, W Stanley Harpole, Peter B Reich, Michael Scherer-Lorenzen, et al. 2011. "High Plant Diversity Is Needed to Maintain Ecosystem Services." Journal Article. *Nature* 477 (7363): 199–202. doi:10.1038/nature10282.

Isbell, Forest, Dylan Craven, John Connolly, Michel Loreau, B. Schmid, Carl Beierkuhnlein, T Martijn Bezemer, et al. 2015. "Biodiversity Increases the Resistance of Ecosystem Productivity to Climate Extremes." Journal Article. *Nature* 526 (7574). Nature Publishing Group: 574–77. doi:10.1038/nature15374.

Ives, Anthony R, and Bradley J Cardinale. 2004. "Food-Web Interactions Govern the Resistance of Communities after Non-Random Extinctions." Journal Article. *Nature* 429 (6988): 174–77. doi:10.1038/nature02515.

Jabiol, Jérémy, Brendan G McKie, Andreas Bruder, Caroline Bernadet, Mark O Gessner, and Eric Chauvet. 2013. "Trophic Complexity Enhances Ecosystem Functioning in an Aquatic Detritus-Based Model System." Journal Article. Edited by Guy Woodward. *Journal of Animal Ecology* 82 (5): 1042–51. doi:10.1111/1365-2656.12079.

Jain, Meha, Dan F B Flynn, Case M Prager, Georgia M Hart, Caroline M DeVan, Farshid S Ahrestani, Matthew I Palmer, et al. 2013. "The Importance of Rare Species: A Trait-Based Assessment of Rare Species Contributions to Functional Diversity and Possible Ecosystem Function in Tall-Grass Prairies." Journal Article. *Ecology and Evolution* 4 (1): 104–12. doi:10.1002/ece3.915.

Jax, K. 2016. "Briefing Note 3.1: Ecosystem Functions: A Critical Perspective." In *Routledge Handbook of Ecosystem Services*, edited by M. Potschin, R. Haines-Young, R. Fish, and R. K. Turner, 42. Routledge.

Jax, Kurt. 2005. "Function and 'functioning' in Ecology: What Does It Mean?" *Oikos* 111 (3): 641–48. doi:10.1111/j.1600-0706.2005.13851.x.

Jesus, Fátima T, Celso Martins, and António J A Nogueira. 2014. "Changes in Life-History Parameters of Daphnia Longispina (Cladocera, Crustacea) as a Function of Water Chemistry." Journal Article. *Journal of Limnology* 73 (2). doi:10.4081/jlimnol.2014.886.

Jongman, R. H. G., C. J. F. Ter Braak, and O. F. R. van Tongeren. 1995. *Data Analysis in Community and Landscape Ecology*. Edited by R. H. G. Jongman, C. J. F. Ter Braak, and O. F. R. van Tongeren. Cambridge: Cambridge University Press. doi:10.1017/CBO9780511525575.

Jopke, Cornelius, Juergen Kreyling, Joachim Maes, and Thomas Koellner. 2015. “Interactions among Ecosystem Services across Europe: Bagplots and Cumulative Correlation Coefficients Reveal Synergies, Trade-Offs, and Regional Patterns.” *Ecological Indicators* 49 (February). Elsevier Ltd: 46–52. doi:10.1016/j.ecolind.2014.09.037.

Jørgensen, S.E. 2012. *Introduction to Systems Ecology*. Boca Raton, Florida: CRC.

Jørgensen, S.E., and S.N. Nielsen. 2013. “The Properties of the Ecological Hierarchy and Their Application as Ecological Indicators.” *Ecological Indicators* 28 (May). Elsevier Ltd: 48–53. doi:10.1016/j.ecolind.2012.04.010.

Jørgensen, S.E., Søren Nors Nielsen, and Brian D. Fath. 2016. “Recent Progress in Systems Ecology.” *Ecological Modelling* 319 (January). Elsevier B.V.: 112–18. doi:10.1016/j.ecolmodel.2015.08.007.

Jost, Lou. 2006. “Entropy and Diversity.” *Oikos* 113 (2): 363–75. doi:10.1111/j.2006.0030-1299.14714.x.

Joy, Michael K., and Russell G. Death. 2003. “Assessing Biological Integrity Using Freshwater Fish and Decapod Habitat Selection Functions.” *Environmental Management* 32 (6): 747–59. doi:10.1007/s00267-003-2057-z.

Kaiser, MJ, KR Clarke, H. Hinz, MCV Austen, PJ Somerfield, and I. Karakassis. 2006. “Global Analysis of Response and Recovery of Benthic Biota to Fishing.” *Marine Ecology Progress Series* 311 (April): 1–14. doi:10.3354/meps311001.

Kandziora, Marion, Benjamin Burkhard, and Felix Müller. 2013. “Interactions of Ecosystem Properties, Ecosystem Integrity and Ecosystem Service Indicators: A Theoretical Matrix Exercise.” *Ecological Indicators* 28. Elsevier Ltd: 54–78. doi:10.1016/j.ecolind.2012.09.006.

Karlson, Agnes M L, Francisco J A Nascimento, Johan Näslund, and Ragnar Elmgren. 2010. “Higher Diversity of Deposit-Feeding Macrofauna Enhances Phytodetritus Processing.” Journal Article. *Ecology* 91 (5): 1414–23. doi:10.1890/09-0660.1.

Kominoski, J S, T J Hoellein, C J Leroy, C M Pringle, and C M Swan. 2009. “Beyond Species Richness: Expanding Biodiversity-Ecosystem Functioning Theory in Detritus-Based Streams.” Journal Article. Edited by Alexander M Milner and Klement Tockner. *River Research and Applications* 26 (1). John Wiley & Sons, Ltd.: 67–75. doi:10.1002/rra.1285.

Kremen, Claire. 2005. “Managing Ecosystem Services: What Do We Need to Know about Their Ecology?” *Ecology Letters* 8 (5): 468–79. doi:10.1111/j.1461-0248.2005.00751.x.

Lambert, Gwladys I., Simon Jennings, Michel J. Kaiser, Thomas W. Davies, and Jan G. Hiddink. 2014. “Quantifying Recovery Rates and Resilience of Seabed Habitats Impacted by

Bottom Fishing." *Journal of Applied Ecology* 51 (5): 1326–36. doi:10.1111/1365-2664.12277.

Larsen, Trond H., Neal M. Williams, and Claire Kremen. 2005. "Extinction Order and Altered Community Structure Rapidly Disrupt Ecosystem Functioning." *Ecology Letters* 8: 538–47. doi:10.1111/j.1461-0248.2005.00749.x.

Latta, Leigh C, Michelle Baker, Todd Crowl, J Jacob Parnell, Bart Weimer, Daryll B DeWald, and Michael E Pfrender. 2010. "Species and Genotype Diversity Drive Community and Ecosystem Properties in Experimental Microcosms." Journal Article. *Evolutionary Ecology* 25 (5). Springer Netherlands: 1107–25. doi:10.1007/s10682-010-9457-3.

Laurila-Pant, Mirka, Annukka Lehikoinen, Laura Uusitalo, and Riikka Venesjärvi. 2015. "How to Value Biodiversity in Environmental Management?" *Ecological Indicators* 55 (55). Elsevier Ltd: 1–11. doi:10.1016/j.ecolind.2015.02.034.

Lavery, Paul S., Miguel Ángel Mateo, Oscar Serrano, and Mohammad Rozaimi. 2013. "Variability in the Carbon Storage of Seagrass Habitats and Its Implications for Global Estimates of Blue Carbon Ecosystem Service." *PLoS ONE* 8 (9). doi:10.1371/journal.pone.0073748.

Lee, Heera, and Sven Lautenbach. 2016. "A Quantitative Review of Relationships between Ecosystem Services." *Ecological Indicators* 66. Elsevier Ltd: 340–51. doi:10.1016/j.ecolind.2016.02.004.

Lefcheck, Jonathan S., Jarrett E K Byrnes, Forest Isbell, Lars Gamfeldt, John N. Griffin, Nico Eisenhauer, Marc J. S. Hensel, Andy Hector, Bradley J Cardinale, and J. Emmett Duffy. 2015. "Biodiversity Enhances Ecosystem Multifunctionality across Trophic Levels and Habitats." *Nature Communications* 6. Nature Publishing Group: 6936. doi:10.1038/ncomms7936.

Lillebø, Ana Isabel, F. Somma, K. Norén, J. Gonçalves, M.F. Alves, E. Ballarini, L. Bentes, et al. 2016. "Assessment of Marine Ecosystem Services Indicators: Experiences and Lessons Learned from 14 European Case Studies." *Integrated Environmental Assessment and Management* 12 (November 2015): 726–34. doi:10.1002/ieam.1782.

Liquete, Camino, Núria Cid, Denis Lanzanova, Bruna Grizzetti, and Arnaud Reynaud. 2016. "Perspectives on the Link between Ecosystem Services and Biodiversity: The Assessment of the Nursery Function." *Ecological Indicators* 63 (April). Elsevier Ltd: 249–57. doi:10.1016/j.ecolind.2015.11.058.

Liquete, Camino, Chiara Piroddi, Evangelia G Drakou, Leigh Gurney, Stelios Katsanevakis, Aymen Charef, and Benis Egoh. 2013. "Current Status and Future Prospects for the Assessment of Marine and Coastal Ecosystem Services: A Systematic Review." *PloS One* 8 (7): e67737. doi:10.1371/journal.pone.0067737.

Loreau, Michel, and Claire de Mazancourt. 2013. "Biodiversity and Ecosystem Stability: A Synthesis of Underlying Mechanisms." *Ecology Letters*, January. doi:10.1111/ele.12073.

Loreau, Michel, Nicolas Mouquet, and Robert D Holt. 2003. "Meta-Ecosystems: A Theoretical Framework for a Spatial Ecosystem Ecology." Journal Article. *Ecology Letters* 6 (8): 673–79. doi:10.1046/j.1461-0248.2003.00483.x.

MA. 2005. "Millennium Ecosystem Assessment." Island Press, Washington, D.C., USA.

Mace, Georgina M, Ken Norris, and Alastair H Fitter. 2012. "Biodiversity and Ecosystem Services: A Multilayered Relationship." *Trends in Ecology & Evolution* 27 (1): 19–26. doi:10.1016/j.tree.2011.08.006.

Mach, Megan E, Rebecca G Martone, and Kai M A Chan. 2015. "Human Impacts and Ecosystem Services: Insufficient Research for Trade-off Evaluation." Journal Article. *Ecosystem Services* 16: 112–20. doi:10.1016/j.ecoser.2015.10.018.

Maes, Joachim, Camino Liquete, Anne Teller, Markus Erhard, Maria Luisa Paracchini, José I. Barredo, Bruna Grizzetti, et al. 2016. "An Indicator Framework for Assessing Ecosystem Services in Support of the EU Biodiversity Strategy to 2020." *Ecosystem Services* 17 (February): 14–23. doi:10.1016/j.ecoser.2015.10.023.

Maes, Joachim, M.L. Paracchini, G. Zulian, M.B. Dunbar, and R. Alkemade. 2012. "Synergies and Trade-Offs between Ecosystem Service Supply, Biodiversity, and Habitat Conservation Status in Europe." *Biological Conservation* 155 (October). Elsevier Ltd: 1–12. doi:10.1016/j.biocon.2012.06.016.

Maes, Joachim, A. Teller, M. Erhard, P. Murphy, M.L. Paracchini, J.I. Barredo, B. Grizzetti, et al. 2014. "Mapping and Assessment of Ecosystems and Their Services. Indicators for Ecosystem Assessments under Action 5 of the EU Biodiversity Strategy to 2020." Publications Office of the European Union, Luxembourg. doi:10.2779/75203.

Martínez-López, Javier, Julia Martínez-Fernández, Babak Naimi, María F. Carreño, and Miguel A. Esteve. 2015. "An Open-Source Spatio-Dynamic Wetland Model of Plant Community Responses to Hydrological Pressures." *Ecological Modelling* 306 (June): 326–33. doi:10.1016/j.ecolmodel.2014.11.024.

Marzorati, Massimo, Annalisa Balloi, Francesca De Ferra, Lorenzo Corallo, Giovanna Carpani, Lieven Wittebolle, Willy Verstraete, and Daniele Daffonchio. 2010. "Bacterial Diversity and Reductive Dehalogenase Redundancy in a 1,2-Dichloroethane-Degrading Bacterial Consortium Enriched from a Contaminated Aquifer." Journal Article. *Microbial Cell Factories* 9 (1). BioMed Central Ltd: 12. doi:10.1186/1475-2859-9-12.

Massa, Sónia I, Cristina M Paulino, Ester A Serrao, Carlos M Duarte, and Sophie Arnaud-Haond. 2013. "Entangled Effects of Allelic and Clonal (Genotypic) Richness in the

Resistance and Resilience of Experimental Populations of the Seagrass Zostera Noltii to Diatom Invasion." Journal Article. *Bmc Ecology* 13 (1). BioMed Central Ltd: 39. doi:10.1186/1472-6785-13-39.

Matthiessen, Birte, Lars Gamfeldt, Per R Jonsson, and Helmut Hillebrand. 2007. "Effects of Grazer Richness and Composition on Algal Biomass in a Closed and Open Marine System." Journal Article. *Ecology* 88 (1): 178–87. doi:10.1890/0012-9658(2007)88[178:EOGRAC]2.0.CO;2.

McBride, P D, J Cusens, and L N Gillman. 2014. "Revisiting Spatial Scale in the Productivity-Species Richness Relationship: Fundamental Issues and Global Change Implications." Journal Article. *Aob Plants* 6 (0). Oxford University Press: plu057–plu057. doi:10.1093/aobpla/plu057.

McCann, Robert K, Bruce G Marcot, and Rick Ellis. 2006. "Bayesian Belief Networks: Applications in Ecology and Natural Resource Management." *Canadian Journal of Forest Research* 36 (12): 3053–62. doi:10.1139/x06-238.

McIntyre, P. B., L. E. Jones, A. S. Flecker, and M. J. Vanni. 2007. "Fish Extinctions Alter Nutrient Recycling in Tropical Freshwaters." *Proceedings of the National Academy of Sciences* 104 (11). National Acad Sciences: 4461–66. doi:10.1073/pnas.0608148104.

Messmer, Vanessa, Shane A Blowes, Geoffrey P Jones, and Philip L Munday. 2014. "Experimental Evaluation of Diversity-productivity Relationships in a Coral Reef Fish Assemblage." Journal Article. *Oecologia* 176 (1). Springer Berlin Heidelberg: 237–49. doi:10.1007/s00442-014-2992-9.

Meyer, Sebastian T., Anne Ebeling, Nico Eisenhauer, Lionel Hertzog, Helmut Hillebrand, Alexandru Milcu, Sven Pompe, et al. 2016. "Effects of Biodiversity Strengthen over Time as Ecosystem Functioning Declines at Low and Increases at High Biodiversity." *Ecosphere* 7 (12): e01619. doi:10.1002/ecs2.1619.

Mokany, Karel, Simon Ferrier, Sean R Connolly, Piers K Dunstan, Elizabeth A Fulton, Michael B Harfoot, Thomas D Harwood, et al. 2015. "Integrating Modelling of Biodiversity Composition and Ecosystem Function." *Oikos*. Blackwell Publishing Ltd, n/a-n/a. doi:10.1111/oik.02792.

Mokany, Karel, Joshua J Thomson, A Jasmyn J Lynch, Gregory J Jordan, and Simon Ferrier. 2015. "Linking Changes in Community Composition and Function under Climate Change." Journal Article. *Ecological Applications* 25 (8). Ecological Society of America: 2132–41. doi:10.1890/14-2384.1.

Mononen, L, A P Auvinen, A L Ahokumpu, M Rönkä, N Aarras, H Tolvanen, M Kamppinen, E Viirret, T Kumpula, and P Vihervaara. 2015. "National Ecosystem Service Indicators: Measures of Social-ecological Sustainability." Journal Article. *Ecological Indicators* 61:

27–37. doi:10.1016/j.ecolind.2015.03.041.

Moore, Jonathan W. 2006. “Animal Ecosystem Engineers in Streams.” Journal Article. *BioScience* 56 (3). Oxford University Press: 237. doi:10.1641/0006-3568(2006)056[0237:AEEIS]2.0.CO;2.

Moore, Tim N, and Peter G Fairweather. 2006. “Decay of Multiple Species of Seagrass Detritus Is Dominated by Species Identity, with an Important Influence of Mixing Litters.” Journal Article. *Oikos* 114 (2). Blackwell Publishing Ltd: 329–37. doi:10.1111/j.2006.0030-1299.14576.x.

Mora, Camilo, Roberto Danovaro, and Michel Loreau. 2014. “Alternative Hypotheses to Explain Why Biodiversity-Ecosystem Functioning Relationships Are Concave-up in Some Natural Ecosystems but Concave-down in Manipulative Experiments.” Journal Article. *Scientific Reports* 4. doi:10.1038/srep05427.

Mouchet, Maud A., Pénélope Lamarque, Berta Martín-López, Emilie Crouzat, Pierre Gos, Coline Byczek, and Sandra Lavorel. 2014. “An Interdisciplinary Methodological Guide for Quantifying Associations between Ecosystem Services.” *Global Environmental Change* 28 (1): 298–308. doi:10.1016/j.gloenvcha.2014.07.012.

Mouillot, David, David R Bellwood, Christopher Baraloto, Jerome Chave, Rene Galzin, Mireille Harmelin-Vivien, Michel Kulbicki, et al. 2013. “Rare Species Support Vulnerable Functions in High-Diversity Ecosystems.” *PLoS Biology* 11 (5): e1001569. doi:10.1371/journal.pbio.1001569.

Mouillot, David, Graham, Villéger, Mason, and Bellwood. 2013. “A Functional Approach Reveals Community Responses to Disturbances.” *Trends in Ecology & Evolution* 28 (3): 167–77. doi:10.1016/j.tree.2012.10.004.

Mouillot, David, Sébastien Villéger, Michael Scherer-Lorenzen, and Norman W H Mason. 2011. “Functional Structure of Biological Communities Predicts Ecosystem Multifunctionality.” Edited by Tamara Romanuk. *PLoS ONE* 6 (3): e17476. doi:10.1371/journal.pone.0017476.

Naeem, Shahid. 2008. “Species Redundancy and Ecosystem Reliability.” Journal Article. *Conservation Biology* 12 (1). Blackwell Science Inc: 39–45. doi:10.1111/j.1523-1739.1998.96379.x.

Naeem, Shahid, JE Duffy, and Erika Zavaleta. 2012. “The Functions of Biological Diversity in an Age of Extinction.” Journal Article. *Science* 336 (6087). American Association for the Advancement of Science: 1401–6. doi:10.1126/science.1215855.

Narwani, Anita, Blake Matthews, Jeremy Fox, and Patrick A Venail. 2015. “Using Phylogenetics in Community Assembly and Ecosystem Functioning Research.” Journal Article. Edited by

Charles Fox. *Functional Ecology* 29 (5): 589–91. doi:10.1111/1365-2435.12431.

Nunes-Neto, Nei, Alvaro Moreno, and Charbel N. El-Hani. 2014. "Function in Ecology: An Organizational Approach." *Biology and Philosophy* 29 (1): 123–41. doi:10.1007/s10539-013-9398-7.

O'Connor, Mary I, and John F Bruno. 2009. "Predator Richness Has No Effect in a Diverse Marine Food Web." Journal Article. *Journal of Animal Ecology* 78 (4). Blackwell Publishing Ltd: 732–40. doi:10.1111/j.1365-2656.2009.01533.x.

O'Higgins, T., A. Barbosa, A. Iglesias-Campos, J. Arévalo-Torres, J. Barbière, A. de Wever, A. Lillebø, A. Nogueira, A. SchmidtKloeiber, and R. Schinegger. 2016. "Review and Analysis of Policy Data, Information Requirements and Lessons Learnt in the Context of Aquatic Ecosystems. Deliverable 2.2." European Union's Horizon 2020 Framework Programme for Research and Innovation Grant Agreement No. 642317.

O'Higgins, T. G., and a.J. Gilbert. 2014. "Embedding Ecosystem Services into the Marine Strategy Framework Directive: Illustrated by Eutrophication in the North Sea." *Estuarine, Coastal and Shelf Science* 140 (March). Elsevier Ltd: 146–52. doi:10.1016/j.ecss.2013.10.005.

Olden, Julian D., and Donald A. Jackson. 2002. "A Comparison of Statistical Approaches for Modelling Fish Species Distributions." *Freshwater Biology* 47 (10): 1976–95. doi:10.1046/j.1365-2427.2002.00945.x.

Oliver, Tom H, Matthew S Heard, Nick J B Isaac, David B Roy, Deborah Procter, Felix Eigenbrod, Rob Freckleton, et al. 2015. "Biodiversity and Resilience of Ecosystem Functions." Journal Article. *Trends in Ecology and Evolution* 30 (11): 673–84. doi:10.1016/j.tree.2015.08.009.

Osman, Richard W. 2015. "Regional Variation in the Colonization of Experimental Substrates by Sessile Marine Invertebrates: Local vs. Regional Control of Diversity." *Journal of Experimental Marine Biology and Ecology* 473: 227–86. doi:10.1016/j.jembe.2015.08.004.

Overton, Jacob McC, Gary M. Barker, and Robbie Price. 2009. "Estimating and Conserving Patterns of Invertebrate Diversity: A Test Case of New Zealand Land Snails." *Diversity and Distributions* 15 (5): 731–41. doi:10.1111/j.1472-4642.2009.00589.x.

Pascual, Marta, Elena Pérez Miñana, and Eva Giacomello. 2016. "Integrating Knowledge on Biodiversity and Ecosystem Services: Mind-Mapping and Bayesian Network Modelling." *Ecosystem Services* 17. Elsevier: 112–22. doi:10.1016/j.ecoser.2015.12.004.

Patrício, Joana, Michael Elliott, Krysia Mazik, Konstantia-nadia Papadopoulou, and Christopher J. Smith. 2016. "DPSIR—Two Decades of Trying to Develop a Unifying

Framework for Marine Environmental Management?" *Frontiers in Marine Science* 3 (September): 1–14. doi:10.3389/fmars.2016.00177.

Patrício, Joana, Sally Little, Krysia Mazik, Konstantia-nadia Papadopoulou, Christopher J. Smith, Heliana Teixeira, Helene Hoffmann, et al. 2016. "European Marine Biodiversity Monitoring Networks: Strengths, Weaknesses, Opportunities and Threats." *Frontiers in Marine Science* 3 (September). doi:10.3389/fmars.2016.00161.

Pedrono, Miguel, Bruno Locatelli, Driss Ezzine-de-Blas, Denis Pesche, Serge Morand, and Aurélie Binot. 2015. "Impact of Climate Change on Ecosystem Services." Book Section. In *Climate Change and Agriculture Worldwide*, 251–61. Dordrecht: Springer Netherlands. doi:10.1007/978-94-017-7462-8_19.

Pereira, Henrique M., Simon Ferrier, M Walters, G. N. Geller, R. H. G. Jongman, R. J. Scholes, M. W. Bruford, et al. 2013. "Essential Biodiversity Variables." *Science* 339 (6117): 277–78. doi:10.1126/science.1229931.

Piet, Gerjan J, and Henrice M Jansen. 2005. "Evaluating Potential Indicators for an Ecosystem Approach to Fishery Management in European Waters" 65 (November): 1449–55.

Pinto, Rute, Victor N de Jonge, and João Carlos Marques. 2014. "Linking Biodiversity Indicators, Ecosystem Functioning, Provision of Services and Human Well-Being in Estuarine Systems: Application of a Conceptual Framework." *Ecological Indicators* 36: 644–55. doi:10.1016/j.ecolind.2013.09.015.

Pinto, Rute, and Joao CARLOS João Carlos Marques. 2015. "Ecosystem Services in Estuarine Systems: Implications for Management." Book Section. In *Ecosystem Services and River Basin Ecohydrology*, 319–41. Dordrecht: Springer Netherlands. doi:10.1007/978-94-017-9846-4_16.

Piroddi, Chiara, Heliana Teixeira, Christopher P. Lynam, Chris Smith, Maria C. Alvarez, Krysia Mazik, Eider Andonegi, et al. 2015. "Using Ecological Models to Assess Ecosystem Status in Support of the European Marine Strategy Framework Directive." *Ecological Indicators* 58 (November). Elsevier Ltd: 175–91. doi:10.1016/j.ecolind.2015.05.037.

Poisot, Timothée, Nicolas Mouquet, and Dominique Gravel. 2013. "Trophic Complementarity Drives the Biodiversity-Ecosystem Functioning Relationship in Food Webs." Journal Article. Edited by Frederick Adler. *Ecology Letters* 16 (7): 853–61. doi:10.1111/ele.12118.

Potschin, M., and R. Haines-Young. 2011. "Introduction to the Special Issue: Ecosystem Services." *Progress in Physical Geography* 35 (5): 571–74. doi:10.1177/0309133311422976.

Ptacnik, R., A. G. Solimini, T. Andersen, T. Tamminen, P. Brettum, L. Lepisto, E. Willen, and S.

Rekolainen. 2008. “Diversity Predicts Stability and Resource Use Efficiency in Natural Phytoplankton Communities.” *Proceedings of the National Academy of Sciences* 105 (13). National Acad Sciences: 5134–38. doi:10.1073/pnas.0708328105.

Queirós, Ana M., Jorn Bruggeman, Nicholas Stephens, Yuri Artioli, Momme Butenschön, Jeremy C Blackford, Stephen Widdicombe, J Icarus Allen, and Paul J Somerfield. 2015. “Placing Biodiversity in Ecosystem Models without Getting Lost in Translation.” *Journal of Sea Research* 98: 83–90. doi:10.1016/j.seares.2014.10.004.

Queirós, Ana M., James Asa Strong, Krysia Mazik, Jacob Carstensen, John Bruun, Paul J. Somerfield, Annette Bruhn, et al. 2016. “An Objective Framework to Test the Quality of Candidate Indicators of Good Environmental Status.” *Frontiers in Marine Science* 3 (0): 1–34. doi:10.3389/fmars.2016.00073.

Radchuk, Viktoriia, Frederik De Laender, Paul J. Van den Brink, and Volker Grimm. 2015. “Biodiversity and Ecosystem Functioning Decoupled: Invariant Ecosystem Functioning despite Non-Random Reductions in Consumer Diversity.” *Oikos*, no. June: n/a–n/a. doi:10.1111/oik.02220.

Raffaelli, David. 2004. “How Extinction Patterns Affect Ecosystems.” Journal Article. *Science* 306 (5699). American Association for the Advancement of Science: 1141–42. doi:10.1126/science.1106365.

———. 2006. “Biodiversity and Ecosystem Functioning: Issues of Scale and Trophic Complexity.” *Marine Ecology Progress Series* 311: 285–94. doi:10.3354/meps311285.

Ramus, Aaron P., and Zachary T. Long. 2015. “Producer Diversity Enhances Consumer Stability in a Benthic Marine Community.” Journal Article. Edited by Brian Silliman. *Journal of Ecology* 104 (2): n/a–n/a. doi:10.1111/1365-2745.12509.

Reich, Peter B., David Tilman, Forest Isbell, Kevin Mueller, Sarah E. Hobbie, Dan F. B. Flynn, and Nico Eisenhauer. 2012. “Impacts of Biodiversity Loss Escalate Through Time as Redundancy Fades.” Journal Article. *Science* 336 (6081). American Association for the Advancement of Science: 589–92. doi:10.1126/science.1217909.

Reiss, Julia, Jon R. Bridle, José M. Montoya, and Guy Woodward. 2009. “Emerging Horizons in Biodiversity and Ecosystem Functioning Research.” *Trends in Ecology & Evolution* 24 (9): 505–14. doi:10.1016/j.tree.2009.03.018.

Reynolds, Pamela L, and John F Bruno. 2012. “Effects of Trophic Skewing of Species Richness on Ecosystem Functioning in a Diverse Marine Community.” Journal Article. Edited by Jon Moen. *PLoS ONE* 7 (5). Public Library of Science: e36196. doi:10.1371/journal.pone.0036196.

Ricketts, Taylor H., Keri B. Watson, Insu Koh, Alicia M. Ellis, Charles C. Nicholson, Stephen

Posner, Leif L. Richardson, and Laura J. Sonter. 2016. “Disaggregating the Evidence Linking Biodiversity and Ecosystem Services.” *Nature Communications* 7 (October). Nature Publishing Group: 13106. doi:10.1038/ncomms13106.

Roger, Fabian, Anna Godhe, and Lars Gamfeldt. 2012. “Genetic Diversity and Ecosystem Functioning in the Face of Multiple Stressors.” Journal Article. Edited by Andrew Hector. *PLoS ONE* 7 (9). Public Library of Science: e45007. doi:10.1371/journal.pone.0045007.

Rossberg, Axel G., Laura Uusitalo, Torsten Berg, Anastasija Zaiko, Anne Chenuil, María C. Uyarra, Ángel Borja, and Christopher P. Lynam. 2017. “Quantitative Criteria for Choosing Targets and Indicators for Sustainable Use of Ecosystems.” *Ecological Indicators* 72. Elsevier Ltd: 215–24. doi:10.1016/j.ecolind.2016.08.005.

Rouquette, J R, H Posthumus, J Morris, T M Hess, Q L Dawson, and D J G Gowing. 2011. “Synergies and Trade-Offs in the Management of Lowland Rural Floodplains: An Ecosystem Services Approach.” *Hydrological Sciences Journal – Journal Des Sciences Hydrologiques Hydrological Sciences Journal* 56 (568): 1566–81. doi:10.1080/02626667.2011.629785.

Sanz-Lázaro, Carlos, Luca Rindi, Elena Maggi, Martina Dal Bello, and Lisandro Benedetti-Cecchi. 2015. “Effects of Grazer Diversity on Marine Microphytobenthic Biofilm: A ‘tug of War’ between Complementarity and Competition.” *Marine Ecology Progress Series* 540 (November): 145–55. doi:10.3354/meps11507.

Saunders, Manu E., and Gary W. Luck. 2016. “Limitations of the Ecosystem Services versus Disservices Dichotomy.” *Conservation Biology* 0 (0): n/a-n/a. doi:10.1111/cobi.12740.

Schaafsma, Willem, and Gerrit N. van Vark. 1979. “Classification and Discrimination Problems with Applications, Part IIa.” *Statistica Neerlandica* 33 (2): 91–126. doi:10.1111/j.1467-9574.1979.tb00666.x.

Schabhüttl, Stefanie, Peter Hingsamer, Gabriele Weigelhofer, Thomas Hein, Achim Weigert, and Maren Striebel. 2013. “Temperature and Species Richness Effects in Phytoplankton Communities.” *Oecologia* 171 (2): 527–36. doi:10.1007/s00442-012-2419-4.

Schmid, B., A. Hector, P. Saha, and M. Loreau. 2008. “Biodiversity Effects and Transgressive Overyielding.” *Journal of Plant Ecology* 1 (2): 95–102. doi:10.1093/jpe/rtn011.

Schmid, B., Andrea B. Pfisterer, and Patricia Balvanera. 2009. “Effects of Biodiversity on Ecosystem, Community, and Population Variables Reported 1974-2004.” Journal Article. Edited by W K Michener. *Ecology* 90 (3): 853–853. doi:10.1890/08-1465.1.

Schweiger, Andreas H, and Carl Beierkuhnlein. 2014. “Water Temperature and Acidity Regime Shape Dominance and Beta-Diversity Patterns in the Plant Communities of Springs.” Journal Article. *Frontiers of Biogeography* 6 (3): 1–14.

Science for Environmental Policy. 2015. “Ecosystem Services and the Environment. In-Depth Report 11 Produced for the European Commission.” Report. Bristol: DG Environment by the Science Communication Unit, UWE. doi:10.2779/162593.

Smith, Christopher J., Konstantia-Nadia Papadopoulou, Steve Barnard, Krysia Mazik, M. Elliott, Joana Patrício, Oihana Solaun, Sally Little, Natasha Bhatia, and Ángel Borja. 2016. “Managing the Marine Environment, Conceptual Models and Assessment Considerations for the European Marine Strategy Framework Directive.” *Frontiers in Marine Science* 3 (August): 1-19. doi:10.3389/fmars.2016.00144.

Solan, Martin, Bradley J Cardinale, Amy L Downing, Katharina a M Engelhardt, Jennifer L Ruesink, and Diane S Srivastava. 2004. “Extinction and Ecosystem Function in the Marine Benthos.” Journal Article. *Science* 306 (5699). American Association for the Advancement of Science: 1177-80. doi:10.1126/science.1103960.

Sousa, Lisa P, Ana I Sousa, Fátima L Alves, and Ana Isabel Lillebø. 2016. “Ecosystem Services Provided by a Complex Coastal Region: Challenges of Classification and Mapping.” *Scientific Reports* 6. Nature Publishing Group: 22782. doi:10.1038/srep22782.

Srivastava, Diane S, M. W. Cadotte, A Andrew M MacDonald, Robin G Marushia, and Nicholas Mirotchnick. 2012. “Phylogenetic Diversity and the Functioning of Ecosystems.” Journal Article. Edited by Arne Mooers. *Ecology Letters* 15 (7): 637-48. doi:10.1111/j.1461-0248.2012.01795.x.

Srivastava, Diane S, Bradley J Cardinale, Amy L Downing, JE Duffy, Claire Jouseau, Mahesh Sankaran, and Justin P Wright. 2009. “Diversity Has Stronger Top-down than Bottom-up Effects on Decomposition.” Journal Article. *Ecology* 90 (4). Ecological Society of America: 1073-83. doi:10.1890/08-0439.1.

Stachowicz, John J., R J Best, M E S Bracken, and M H Graham. 2008. “Complementarity in Marine Biodiversity Manipulations: Reconciling Divergent Evidence from Field and Mesocosm Experiments.” Journal Article. *Proceedings of the National Academy of Sciences* 105 (48). National Acad Sciences: 18842-47. doi:10.1073/pnas.0806425105.

Stachowicz, John J., John F. Bruno, and JE Duffy. 2007. “Understanding the Effects of Marine Biodiversity on Communities and Ecosystems.” *Annual Review of Ecology, Evolution, and Systematics* 38 (1): 739-66. doi:10.1146/annurev.ecolsys.38.091206.095659.

Staddon, Philip, Zoë Lindo, Peter D Crittenden, Francis Gilbert, and Andrew Gonzalez. 2010. “Connectivity, Non-Random Extinction and Ecosystem Function in Experimental Metacommunities.” *Ecology Letters* 13 (5): 543-52. doi:10.1111/j.1461-0248.2010.01450.x.

Strong, James Asa, Eider Andonegi, Kemal Can Bizsel, Roberto Danovaro, M. Elliott, Anita Franco, Esther Garces, et al. 2015. “Marine Biodiversity and Ecosystem Function

Relationships: The Potential for Practical Monitoring Applications." *Estuarine, Coastal and Shelf Science* 161: 46–64. doi:10.1016/j.ecss.2015.04.008.

Swift, M.J., A.-M.N. Izac, and M. van Noordwijk. 2004. "Biodiversity and Ecosystem Services in Agricultural Landscapes—are We Asking the Right Questions?" *Agriculture, Ecosystems & Environment* 104 (1): 113–34. doi:10.1016/j.agee.2004.01.013.

Tan, Can Ozan, Uygar Özesmi, Meryem Beklioglu, Esra Per, and Bahtiyar Kurt. 2006. "Predictive Models in Ecology: Comparison of Performances and Assessment of Applicability." *Ecological Informatics* 1 (2): 195–211. doi:10.1016/j.ecoinf.2006.03.002.

TEEB. 2010. "The Economics of Ecosystems and Biodiversity: Ecological and Economic Foundations." In: Kumar, Pushpam (Ed.) Earthscan, London, United Kingdom.

Teixeira, Heliana, Torsten Berg, Laura Uusitalo, Karin Fürhaupter, Krysia Mazik, Christopher P Lynam, Suzanna Neville, et al. 2016. "A Catalogue of Marine Biodiversity Indicators." *Frontiers in Marine Science* 3 (October): 1–16. doi:10.3389/fmars.2016.00207.

The QUINTESSENCE Consortium. 2016. "Networking Our Way to Better Ecosystem Service Provision." *Trends in Ecology & Evolution* 31 (2). Elsevier Ltd: 105–15. doi:10.1016/j.tree.2015.12.003.

Thebault, E, and M Loreau. 2011. "Food-Web Constraints on Biodiversity-Ecosystem Functioning Relationships." Journal Article. *Proceedings of the National Academy of Sciences* 100 (25): 14949–54. doi:10.1073/pnas.2434847100.

Thompson, Patrick L, T Jonathan Davies, and Andrew Gonzalez. 2015. "Ecosystem Functions across Trophic Levels Are Linked to Functional and Phylogenetic Diversity." Journal Article. Edited by Wenju Liang. *PLoS ONE* 10 (2). Public Library of Science: e0117595. doi:10.1371/journal.pone.0117595.

Tittensor, Derek P, Matt Walpole, Samantha L. L. Hill, Daniel G. Boyce, Gregory L. Britten, Neil D. Burgess, Stuart H M Butchart, et al. 2014. "A Mid-Term Analysis of Progress toward International Biodiversity Targets." *Science* 346 (6206): 241–44. doi:10.1126/science.1257484.

Tomscha, Stephanie A., and Sarah E. Gergel. 2016. "Ecosystem Service Trade-Offs and Synergies Misunderstood without Landscape History." *Ecology and Society* 21 (1): art43. doi:10.5751/ES-08345-210143.

Truchy, Amélie, David G Angeler, Ryan A Sponseller, Richard K Johnson, and Brendan G McKie. 2015. "Linking Biodiversity, Ecosystem Functioning and Services, and Ecological Resilience." Book Section. In *Ecosystem Services – From Biodiversity to Society, Part 1*, 53:55–96. Elsevier. doi:10.1016/bs.aecr.2015.09.004.

United Nations, European Commission, Food and Agricultural Organization of the United

Nations, Organization for Economic Co-operation and Development, and World Bank. 2014. *System of Environmental-Economic Accounting 2012: Experimental Ecosystem Accounting*. *White Cover Publication*. doi:ST/ESA/STAT/Ser.F/109.

van der Linden, P., A. Marchini, Marina Dolbeth, Joana Patrício, H. Veríssimo, and J. C. Marques. 2016. "The Performance of Trait-Based Indices in an Estuarine Environment." *Ecological Indicators* 61 (April 2016). doi:10.1016/j.ecolind.2015.09.039.

Vaughn, Caryn C. 2010. "Biodiversity Losses and Ecosystem Function in Freshwaters: Emerging Conclusions and Research Directions." Journal Article. *BioScience* 60 (1): 25–35. doi:10.1525/bio.2010.60.1.7.

Venail, Patrick A, R C Maclean, C N Meynard, and N Mouquet. 2010. "Dispersal Scales up the Biodiversity–Productivity Relationship in an Experimental Source-Sink Metacommunity." Journal Article. *Proceedings of the Royal Society B: Biological Sciences* 277 (1692). The Royal Society: 2339–45. doi:10.1098/rspb.2009.2104.

Venail, Patrick A, and Martha J Vives. 2013. "Positive Effects of Bacterial Diversity on Ecosystem Functioning Driven by Complementarity Effects in a Bioremediation Context." Journal Article. Edited by Hauke Smidt. *PLoS ONE* 8 (9). Public Library of Science: e72561. doi:10.1371/journal.pone.0072561.

Villa, Ferdinando, Kenneth J Bagstad, Brian Voigt, Gary W Johnson, Rosimeiry Portela, Miroslav Honzák, and David Batker. 2014. "A Methodology for Adaptable and Robust Ecosystem Services Assessment." Journal Article. Edited by Antoni Margalida. *PLoS ONE* 9 (3): e91001. doi:10.1371/journal.pone.0091001.

Villamagna, Amy M, and Paul L Angermeier. 2015. "A Methodology for Quantifying and Mapping Ecosystem Services Provided by Watersheds." Book Section. In *Ecosystem Services and River Basin Ecohydrology*, 151–80. Dordrecht: Springer Netherlands. doi:10.1007/978-94-017-9846-4_8.

Walker, Meaghan K, and Ross M Thompson. 2010. " Consequences of Realistic Patterns of Biodiversity Loss: An Experimental Test from the Intertidal Zone." Journal Article. *Marine and Freshwater Research* 61 (9). CSIRO PUBLISHING: 1015–22. doi:10.1071/MF09244.

Weis, Jerome J, Daniel S Madrigal, and Bradley J Cardinale. 2008. "Effects of Algal Diversity on the Production of Biomass in Homogeneous and Heterogeneous Nutrient Environments: A Microcosm Experiment." Journal Article. Edited by Angus Buckling. *PLoS ONE* 3 (7). Public Library of Science: e2825. doi:10.1371/journal.pone.0002825.

Williams, Byron K. 1983. "Some Observations of the Use of Discriminant Analysis in Ecology." *Ecology* 64 (5): 1283–91. doi:10.2307/1937836.

Wolf, Amelia A, and Erika S Zavaleta. 2015. "Species Traits Outweigh Nested Structure in

Driving the Effects of Realistic Biodiversity Loss on Productivity." Journal Article. *Ecology* 96 (1): 90–98. doi:10.1890/14-0131.1.

Worm, B, Edward B Barbier, N Beaumont, JE Duffy, C Folke, B S Halpern, J B C Jackson, et al. 2006. "Impacts of Biodiversity Loss on Ocean Ecosystem Services." Journal Article. *Science* 314 (5800). American Association for the Advancement of Science: 787–90. doi:10.1126/science.1132294.

Zampoukas, Nikolaos, Henna Piha, Emanuele Bigagli, Nicolas Hoepffner, Georg Hanke, and Ana Cristina Cardoso. 2013. "Marine Monitoring in the European Union: How to Fulfill the Requirements for the Marine Strategy Framework Directive in an Efficient and Integrated Way." *Marine Policy* 39 (May 1992). Elsevier: 349–51. doi:10.1016/j.marpol.2012.12.004.

Zeglin, Lydia H. 2015. "Stream Microbial Diversity in Response to Environmental Changes: Review and Synthesis of Existing Research." Journal Article. *Frontiers in Microbiology* 6 (20131760). Frontiers: 11512. doi:10.3389/fmicb.2015.00454.

Zimmerman, Emily K, and Bradley J Cardinale. 2013. "Is the Relationship between Algal Diversity and Biomass in North American Lakes Consistent with Biodiversity Experiments?" Journal Article. *Oikos* 123 (3). Blackwell Publishing Ltd: 267–78. doi:10.1111/j.1600-0706.2013.00777.x.

# 9 Annexes

There are two annexes to this Deliverable 5.1:

- **Annex I** contains the classifications, and indicator lists and sources proposed for the different steps of the supply side of the AF presented in Chapter 5: Biodiversity (BD), Ecosystem Function (EF), and Ecosystem Services (ESS). *Annex I is provided as a side document to this report in excel format.*
- **Annex II** contains a compilation of literature applying meta-analysis on BD-EF-ESS relevant topics, as introduced in Chapter 4. *Annex II is included in the present report as Table A I.1.*

## 9.1 Annex I

The detailed classifications for Biodiversity (BD), Ecosystem Function (EF), and Ecosystem Services (ESS), presented in Chapter 5, are fully provided in Annex I excel file: *D51_INDICATORS_BD_EF_ESS_vs1.xlsx*. This file contains **several supporting tables** specificying the classifications for BD, EF and ESS and providing links to additional sources of information useful for operationalising the use of these broad classifications across the different aquatic realms: from lakes, to rivers, including wetlands in fresh and saline environements, to marine inlets and transitional waters, and extending to coastal, shelf and oceanic waters. The objective was to, as much as possible, adopt existing and broadly used classifications across these different aquatic domains, and integrate them into a harmonised broader classification within AQUACROSS. This aims at promoting a better integration of the results obtained from the application of the AF concepts in AQUACROSS eight case studies.

Besides detailing each of the classifications, the supporting tables include also associated **lists of potential indicators, indices and metrics** and/or **links to additional sources** of indicators, which can be used to assess BD, EF and ESS supply and demand.

The tables are organized in order to allow their immediate use by the case studies, for selecting, adding and organizing their set of indicators across the steps of the AQUACROSS AF described in this Deliverable. The classifications proposed attempt also to provide links to other stages of the socio-ecological system as described in the AF (Chapter 5), namely upstream to the Drivers-Pressures-State, and downstream to the Benefits and Values.

> This annex is a **working document**, which may suffer further adaptations after the case studies workshop (in the following months), which will test the applicability of the proposed classifications by using real scenarios (as part of Task 5.2). Therefore, indicators lists included have not yet been assigned under the classifications proposed. This will be part of the subsequent tasks.

Below we present the structure of the excel file and explain how it can be used to support the practical implementation of some of the concepts discussed in the present deliverable.

The excel file contains 11 spreadsheets with several supporting tables, most of them present only information for consultation (**guidance** spreasheets), others allow selection and/or reporting of indicators, indices and metrics (reporting spreadsheets):

- **1 BD-EF-ESS classification** – (guidance) contains Table S1, which provides an overview of the classification proposed for the different steps of the supply side of the AF considered in WP5: Biodiversity (BD), Ecosystem Function (EF), and Ecosystem Services (ESS), the later accounting both for the supply and demand of ecosystem services. For details regarding each of the stages (i.e. **BD**, **EF** and **ESS**) and application in specific aquatic realms the other spreadsheets must be consulted.
- **2aBiodiversity(BD)** – (guidance) contains Table S2.1 BD, which details the classification proposed for Biodiversity (BD) (corresponding partially to the State of the ecosystem also relevant for WP4; see Deliverable 4.1). At the Class level, this table refers to existing classifications in use accross different aquatic realms in freshwater (FW), transitional (TW) coastal (CW) and marine (MW) waters. This classification should be used to assign BD related indicators, applied within the case studies, to higher levels in a standardized way across Project partners.
- **2bBD Class auxiliar tables** – (guidance) contains two tables:
  - Table S2.2 BD. Sources and links to details of classifications (mentioned at *Class level*) available for the different biodiversity components, previously identified at *Group level* in Table S2.1 BD (both for Biological elements and Habitats). The detailed classifications apply to different aquatic realms relevant for AQUACROSS;
  - Table S2.3 BD. Auxiliar table to S2.2BD, with detailed classifications for I. Biological elements and II. Habitats, covering different aquatic realms relevant for AQUACROSS.
- **2cBD Sources** – (guidance) contains Table S2.4 BD, which provides links to sources of indicators (e.g. databases; reviews; reports; initiatives), with indication of the aquatic ecosystems for which the sources provide indicators.
- **2dBD Lists** – (reporting) contains Table S2.5 BD, which lists indicators, indices and metrics (although non-exhaustively) available for Biodiversity and Ecosystem State assessment. This table allows:
  - selecting indicators for use in specific case studies, as well as,
  - entering new indicators (not listed) selected by the Project partners. for AQUACROSS case studies.
- **3aEcosystemFunctions(EF)** – (guidance) contains Table S3.1 EF, which details the classification proposed for Ecosystem Function (EF), identifying most relevant ecological processes and functions in aquatic ecosystems, and grouping the latter into major categories.
- **3bEF list & sources** – (guidance/reporting) contains Table S3.2 EF, which provides examples of indicators, indices and associated metrics (and links to sources) potentially useful for measuring Ecosystem Function (EF) in AQUACROSS.
- **3cEF case study** – (reporting) contains Table S3.3 EF., for entering new indicators (and indices and associated metrics) for Ecosystem Function, selected for a specific AQUACROSS case study.
- **4aEcosystemServices(ESS)** – (guidance/reporting) contains Table S4.1 ESS, which details the classification proposed for Ecosystem Services (ESS), following CICES & MAES and including Abiotic Services (see section 5.1.4); distinguishing if possible between ESS supply and ESS

demand indicators. This table allows already to report indicators selected for AQUACROSS case studies, assigning them to the proposed classification.

- **4bESS sources I** – (guidance) contains Table S4.2 ESS, which provides sources and lists indicators, indices and associated metrics potentially useful for measuring Ecosystem Services (ESS) in AQUACROSS.
- **4cESS sources II** – (guidance) contains two tables Table S4.3 ESS and Table S4.4 ESS, which provide lists of ESS indicators specific for marine and fresh water ecosytems, respectively, from MAES.

SEE ATTACHED EXCEL FILE: “D51_INDICATORS_BD_EF_ESS_vs1”

## 9.2 Annex II

A literature review on meta-analyses (Table AII.1) associated with data from the aquatic environment in the bibliographic databases JStore, PubMed, Scopus and Web of knowledge using a search key defined as (META-ANALYSIS AND (BIODIVERSITY OR “ECOSYSTEM FUNCTIONS” OR “ECOSYSTEM SERVICES”) AND AQUATIC). This held 294 unique results that were narrowed down to 108 relevant papers dealing with meta-analyses linked with the aquatic environment. Selected meta-analyses were arranged per theme and year. Themes were extracted from a causal chain linking pressures (P) to biodiversity (BD), ecosystem functions (EF), ecosystem services (ESS) and benefits:

- P-BD: From pressures to biodiversity
- P-EF: From pressures to ecosystem functions
- BD: Biodiversity
- BEF: From biodiversity to ecosystem functions
- BES: From biodiversity to ecosystem services
- BD-Benefit: From biodiversity to benefits
- ESS-Benefit: From ecosystem services to benefit
- scale: Assessment of different scales on elements of the causal chain above

This list of selected meta-analysis is aimed at supporting case-study needs when dealing with specific elements of the causal chain, by providing information about the likely relationships being considered.

Table AII.1. Literature review on meta-analysis supporting the establishment relationshsips along the causal chain linking Pressures to Benefits in aquatic ecosystems.

| Theme | Year | Reference |
| --- | --- | --- |
| **P - BD** | 2016 | Yang, W., Sun, T., & Yang, Z. (2016). Does the implementation of environmental flows improve wetland ecosystem services and biodiversity? A literature review. Restoration Ecology, 24(6), 731–742. http://doi.org/10.1111/rec.12435 |
| | 2016 | Carlson, P. E., McKie, B. G., Sandin, L., & Johnson, R. K. (2016). Strong land-use effects on the dispersal patterns of adult stream insects: implications for transfers of aquatic subsidies to terrestrial consumers. Freshwater Biology, 61(6), 848–861. http://doi.org/10.1111/fwb.12745 |
| | 2016 | Tonkin, J. D., Stoll, S., Jähnig, S. C., & Haase, P. (2016). Anthropogenic land-use stress alters community concordance at the river-riparian interface. Ecological Indicators, 65, 133–141. http://doi.org/10.1016/j.ecolind.2015.08.037 |
| | 2016 | Jackson, M. C., Loewen, C. J. G., Vinebrooke, R. D., & Chimimba, C. T. (2016). Net effects of multiple stressors in freshwater ecosystems: a meta-analysis. Global Change Biology, 22(1), 180–189. http://doi.org/10.1111/gcb.13028 |
| | 2015 | Baumgartner, S. D., & Robinson, C. T. (2015). Land-use legacy and the differential response of stream macroinvertebrates to multiple stressors studied using in situexperimental mesocosms. Freshwater Biology, 60(8), 1622–1634. http://doi.org/10.1111/fwb.12594 |
| | 2015 | Garssen, A. G., Baattrup-Pedersen, A., Voesenek, L. A. C. J., Verhoeven, J. T. A., & Soons, M. B. (2015). Riparian plant community responses to increased flooding: a meta-analysis. Global Change Biology, 21(8), 2881–2890. http://doi.org/10.1111/gcb.12921 |
| | 2015 | Janse, J. H., Kuiper, J. J., Weijters, M. J., Westerbeek, E. P., Jeuken, M. H. J. L., Bakkenes, M., et al. (2015). GLOBIO-Aquatic, a global model of human impact on the biodiversity of inland aquatic ecosystems. Environmental Science and Policy, 48, 99–114. http://doi.org/10.1016/j.envsci.2014.12.007 |
| | 2015 | Mackintosh, T. J., Davis, J. A., & Thompson, R. M. (2015). The influence of urbanisation on macroinvertebrate biodiversity in constructed stormwater wetlands. Science of the Total Environment, 536, 527–537. http://doi.org/10.1016/j.scitotenv.2015.07.066 |
| | 2015 | Tolkkinen, M., Mykrä, H., Annala, M., Markkola, A. M., Vuori, K. M., & Muotka, T. (2015). Multi-stressor impacts on fungal diversity and ecosystem functions in streams: natural vs. anthropogenic stress. Ecology, 96(3), 672–683. http://doi.org/10.6084/m9.figshare.c.3307500.v1 |
| | 2014 | Kuiper, J. J., Janse, J. H., Teurlincx, S., Verhoeven, J. T. A., & Alkemade, R. (2014). The impact of river regulation on the biodiversity intactness of floodplain wetlands. Wetlands Ecology and Management, 22(6), 647–658. http://doi.org/10.1007/s11273-014-9360-8 |

| Theme | Year | Reference |
|---|---|---|
| | 2014 | Mantyka-Pringle, C. S., Martin, T. G., Moffatt, D. B., Linke, S., & Rhodes, J. R. (2014). Understanding and predicting the combined effects of climate change and land-use change on freshwater macroinvertebrates and fish. Journal of Applied Ecology, 51(3), 572–581. http://doi.org/10.1111/1365-2664.12236 |
| | 2014 | Meli, P., Rey Benayas, J. M., Balvanera, P., & Martínez Ramos, M. (2014). Restoration enhances wetland biodiversity and ecosystem service supply, but results are context-dependent: a meta-analysis. PLoS ONE, 9(4), e93507. http://doi.org/10.1371/journal.pone.0093507 |
| | 2014 | Yoshioka, A., Miyazaki, Y., Sekizaki, Y., Suda, S.-I., Kadoya, T., & Washitani, I. (2014). A "lost biodiversity" approach to revealing major anthropogenic threats to regional freshwater ecosystems. Ecological Indicators, 36, 348–355. http://doi.org/10.1016/j.ecolind.2013.08.008 |
| | 2014 | Murphy, G. E. P., & Romanuk, T. N. (2014). A meta-analysis of declines in local species richness from human disturbances. Ecology and Evolution, 4(1), 91–103. http://doi.org/10.1002/ece3.909 |
| | 2013 | Comte, L., Buisson, L., Daufresne, M., & Grenouillet, G. (2013). Climate-induced changes in the distribution of freshwater fish: observed and predicted trends. Freshwater Biology, 58(4), 625–639. http://doi.org/10.1111/fwb.12081 |
| | 2012 | Mantyka-Pringle, C. S., Martin, T. G., & Rhodes, J. R. (2012). Interactions between climate and habitat loss effects on biodiversity: a systematic review and meta-analysis. Global Change Biology, 18(4), 1239–1252. http://doi.org/10.1111/j.1365-2486.2011.02593.x |
| | 2012 | Murphy, G. E. P., & Romanuk, T. N. (2012). A Meta-Analysis of Community Response Predictability to Anthropogenic Disturbances. The American Naturalist, 180(3), 316–327. http://doi.org/10.1086/666986 |
| | 2012 | Szivák, I., & Csabai, Z. (2012). Are there any differences between taxa groups having distinct ecological traits based on their responses to environmental factors? Aquatic Insects, 34(sup1), 173–187. http://doi.org/10.1080/01650424.2012.643052 |
| | 2012 | Webb, J. A., Wallis, E. M., & Stewardson, M. J. (2012). A systematic review of published evidence linking wetland plants to water regime components. Aquatic Botany, 103, 1–14. http://doi.org/10.1016/j.aquabot.2012.06.003 |
| | 2010 | Floeder, S., Jaschinski, S., Wells, G., & Burns, C. W. (2010). Dominance and compensatory growth in phytoplankton communities under salinity stress. Journal of Experimental Marine Biology and Ecology, 395(1-2), 223–231. http://doi.org/10.1016/j.jembe.2010.09.006 |
| | 2010 | Miller, S. W., Budy, P., & Schmidt, J. C. (2010). Quantifying Macroinvertebrate Responses to In-Stream Habitat Restoration: Applications of Meta-Analysis to River Restoration. Restoration Ecology, 18(1), 8–19. http://doi.org/10.1111/j.1526-100X.2009.00605.x |

| Theme | Year | Reference |
|---|---|---|
| | 2009 | McKie, B. G., Schindler, M., Gessner, M. O., & Malmqvist, B. (2009). Placing biodiversity and ecosystem functioning in context: environmental perturbations and the effects of species richness in a stream field experiment. Oecologia, 160(4), 757–770. http://doi.org/10.1007/s00442-009-1336-7 |
| | 2009 | Heino, J., Virkkala, R., & Toivonen, H. (2009). Climate change and freshwater biodiversity: detected patterns, future trends and adaptations in northern regions. Biological Reviews, 84(1), 39–54. http://doi.org/10.1111/j.1469-185X.2008.00060.x |
| | 2009 | Johnston, E. L., & Roberts, D. A. (2009). Contaminants reduce the richness and evenness of marine communities: A review and meta-analysis. Environmental Pollution, 157(6), 1745–1752. http://doi.org/10.1016/j.envpol.2009.02.017 |
| | 2009 | Poff, N. L. (2009). Managing for Variability to Sustain Freshwater Ecosystems. Journal of Water Resources Planning and Management, 135(1), 1–4. http://doi.org/10.1061/(ASCE)0733-9496(2009)135:1(1) |
| | 2008 | Haxton, T. J., & Findlay, C. S. (2008). Meta-analysis of the impacts of water management on aquatic communities. Canadian Journal of Fisheries and Aquatic Sciences, 65(3), 437–447. http://doi.org/10.1139/f07-175 |
| | 2008 | Petrin, Z., Englund, G., & Malmqvist, B. (2008). Contrasting effects of anthropogenic and natural acidity in streams: a meta-analysis. Proceedings of the Royal Society B-Biological Sciences, 275(1639), 1143–1148. http://doi.org/10.1098/rspb.2008.0023 |
| | 2007 | Pusceddu, A., Gambi, C., Manini, E., & Danovaro, R. (2007). Trophic state, ecosystem efficiency and biodiversity of transitional aquatic ecosystems: analysis of environmental quality based on different benthic indicators. Chemistry and Ecology, 23(6), 505–515. http://doi.org/10.1080/02757540701760494 |
| | 2005 | Dudgeon, D., Arthington, A. H., Gessner, M. O., Kawabata, Z.-I., Knowler, D. J., Lévêque, C., Naiman, R. J., Prieur-Richard, A.-H., Soto, D., Stiassny, M. L. J., & Sullivan, C. A. (2005). Freshwater biodiversity: importance, threats, status and conservation challenges. Biological Reviews, 81(02), 163–182. http://doi.org/10.1017/S1464793105006950 |
| **P - EF** | 2016 | Ferreira, V., Koricheva, J., Duarte, S., Niyogi, D. K., & Guérold, F. (2016). Effects of anthropogenic heavy metal contamination on litter decomposition in streams – A meta-analysis. Environmental Pollution, 210, 261–270. http://doi.org/10.1016/j.envpol.2015.12.060 |
| | 2015 | Tolkkinen, M., Mykrä, H., Annala, M., Markkola, A. M., Vuori, K. M., & Muotka, T. (2015). Multi-stressor impacts on fungal diversity and ecosystem functions in streams: natural vs. anthropogenic stress. Ecology, 96(3), 672–683. http://doi.org/10.6084/m9.figshare.c.3307500.v1 |
| | 2012 | Holland, A., Duivenvoorden, L. J., & Kinnear, S. H. W. (2012). Naturally acidic waterways: conceptual food webs for better management and understanding of ecological functioning. Aquatic Conservation: Marine and Freshwater Ecosystems, 22(6), 836–847. http://doi.org/10.1002/aqc.2267 |

| Theme | Year | Reference |
|---|---|---|
| | 2010 | Downing, A. L., & Leibold, M. A. (2010). Species richness facilitates ecosystem resilience in aquatic food webs. Freshwater Biology, 55(10), 2123–2137. http://doi.org/10.1111/j.1365-2427.2010.02472.x |
| | 2009 | McKie, B. G., Schindler, M., Gessner, M. O., & Malmqvist, B. (2009). Placing biodiversity and ecosystem functioning in context: environmental perturbations and the effects of species richness in a stream field experiment. Oecologia, 160(4), 757–770. http://doi.org/10.1007/s00442-009-1336-7 |
| **BD** | 2015 | Massicotte, P., Proulx, R., Cabana, G., & Rodríguez, M. A. (2015). Testing the influence of environmental heterogeneity on fish species richness in two biogeographic provinces. PeerJ, 3(2), e760. http://doi.org/10.7717/peerj.760 |
| | 2013 | Cao, M. (2013). Approaches to assessment of global biodiversity and advancements in their researches. Journal of Ecology and Rural Environment, 29(1), 8–16. |
| | 2013 | Chmara, R., Szmeja, J., & thébaut, E. (2013). Patterns of abundance and co-occurrence in aquatic plant communities. Ecological Research, 28(3), 387–395. http://doi.org/10.1007/s11284-013-1028-y |
| | 2013 | Göthe, E., FRIBERG, N., Kahlert, M., Temnerud, J., & Sandin, L. (2013). Headwater biodiversity among different levels of stream habitat hierarchy. Biodiversity and Conservation, 23(1), 63–80. http://doi.org/10.1007/s10531-013-0584-3 |
| | 2012 | Soininen, J., Passy, S., & Hillebrand, H. (2012). The relationship between species richness and evenness: a meta-analysis of studies across aquatic ecosystems. Oecologia, 169(3), 803–809. http://doi.org/10.1007/s00442-011-2236-1 |
| | 2011 | Hassall, C., Hollinshead, J., & Hull, A. (2011). Environmental correlates of plant and invertebrate species richness in ponds. Biodiversity and Conservation, 20(13), 3189–3222. http://doi.org/10.1007/s10531-011-0142-9 |
| | 2008 | Muneepeerakul, R., Bertuzzo, E., Rinaldo, A., & Rodriguez-Iturbe, I. (2008). Patterns of vegetation biodiversity: The roles of dispersal directionality and river network structure. Journal of Theoretical Biology, 252(2), 221–229. http://doi.org/10.1016/j.jtbi.2008.02.001 |
| | 2007 | Cogan, C. (2007). Marine classification, mapping, and biodiversity analysis. In Special Paper - Geological Association of Canada (pp. 129–139). |
| **BEF** | 2016 | Dalzochio, M. S., Baldin, R., Stenert, C., & Maltchik, L. (2016). How does the management of rice in natural ponds alter aquatic insect community functional structure? Marine and Freshwater Research, 67(11), 1644–1654. http://doi.org/10.1071/MF14246 |
| | 2016 | Harvey, E., Gounand, I., Ward, C. L., & Altermatt, F. (2016). Bridging ecology and conservation: from ecological networks to ecosystem function. Journal of Applied Ecology. http://doi.org/10.1111/1365-2664.12769 |

| Theme | Year | Reference |
|---|---|---|
| | 2016 | Lewandowska, A. M., Biermann, A., Borer, E. T., Cebrián-Piqueras, M. A., Declerck, S. A. J., De Meester, L., et al. (2016). The influence of balanced and imbalanced resource supply on biodiversity-functioning relationship across ecosystems. Philosophical Transactions of the Royal Society of London. Series B, Biological Sciences, 371(1694), 20150283. http://doi.org/10.1098/rstb.2015.0283 |
| | 2016 | Rakowski, C., & Cardinale, B. J. (2016). Herbivores control effects of algal species richness on community biomass and stability in a laboratory microcosm experiment. Oikos, 125(11), 1627–1635. http://doi.org/10.1111/oik.03105 |
| | 2016 | Ballari, S. A., Kuebbing, S. E., & Nunez, M. A. (2016). Potential problems of removing one invasive species at a time: a meta-analysis of the interactions between invasive vertebrates and unexpected effects of removal programs. PeerJ, 4(2). http://doi.org/10.7717/peerj.2029 |
| | 2015 | Behl, S., & Stibor, H. (2015). Prey diversity and prey identity affect herbivore performance on different time scales in a long term aquatic food-web experiment. Oikos, 124(9), 1192–1202. http://doi.org/10.1111/oik.01463 |
| | 2015 | Lefcheck, J. S., Byrnes, J. E. K., Isbell, F., Gamfeldt, L., Griffin, J. N., Eisenhauer, N., et al. (2015). Biodiversity enhances ecosystem multifunctionality across trophic levels and habitats. Nature Communications, 6, 6936. http://doi.org/10.1038/ncomms7936 |
| | 2015 | Tornroos, A., Bonsdorff, E., Bremner, J., Blomqvist, M., Josefson, A. B., Garcia, C., & Warzocha, J. (2015). Marine benthic ecological functioning over decreasing taxonomic richness. Journal of Sea Research, 98, 49–56. http://doi.org/10.1016/j.seares.2014.04.010 |
| | 2013 | Velghe, K., & Gregory-Eaves, I. (2013). Body Size Is a Significant Predictor of Congruency in Species Richness Patterns: A Meta-Analysis of Aquatic Studies. PLoS ONE, 8(2). http://doi.org/10.1371/journal.pone.0057019 |
| | 2013 | Cardinale, B. J., Gross, K., Fritschie, K., Flombaum, P., Fox, J. W., Rixen, C., et al. (2013). Biodiversity simultaneously enhances the production and stability of community biomass, but the effects are independent. Ecology, 94(8), 1697–1707. http://doi.org/10.1890/12-1334.1 |
| | 2013 | Harvey, E., Séguin, A., Nozais, C., Archambault, P., & Gravel, D. (2013). Identity effects dominate the impacts of multiple species extinctions on the functioning of complex food webs. Ecology, 94(1), 169–179. http://doi.org/10.1890/12-0414.1 |
| | 2013 | Lanari, M. de O., & Coutinho, R. (2013). Reciprocal causality between marine macroalgal diversity and productivity in an upwelling area. Oikos, 123(5), 630–640. http://doi.org/10.1111/j.1600-0706.2013.00952.x |
| | 2013 | Simões, N. R., Colares, M. A. M., Lansac-Tôha, F. A., & Bonecker, C. C. (2013). Zooplankton species richness-productivity relationship: Confronting monotonic positive and hump-shaped models from a local perspective. Austral Ecology, 38(8), 952–958. http://doi.org/10.1111/aec.12038 |

| Theme | Year | Reference |
| --- | --- | --- |
| | 2012 | Allen, D. (2012). Bottom-up biodiversity effects increase resource subsidy flux between ecosystems. Ecology, 93(10), 2165–2174. http://dx.doi.org/10.1890/11-1541.1 |
| | 2012 | Narwani, A., & Mazumder, A. (2012). Bottom-up effects of species diversity on the functioning and stability of food webs. Journal of Animal Ecology, 81(3), 701–713. http://doi.org/10.1111/j.1365-2656.2011.01949.x |
| | 2012 | Poore, A. G. B., Campbell, A. H., Coleman, R. A., Edgar, G. J., Jormalainen, V., Reynolds, P. L., et al. (2012). Global patterns in the impact of marine herbivores on benthic primary producers. Ecology Letters, 15(8), 912–922. http://doi.org/10.1111/j.1461-0248.2012.01804.x |
| | 2012 | Tornroos, A., & Bonsdorff, E. (2012). Developing the multitrait concept for functional diversity: lessons from a system rich in functions but poor in species. Ecological Applications, 22(8), 2221–2236. |
| | 2010 | Carey, M. P., & Wahl, D. H. (2010). Fish diversity as a determinant of ecosystem properties across multiple trophic levels. Oikos, 120(1), 84–94. http://doi.org/10.1111/j.1600-0706.2010.18352.x |
| | 2010 | Gessner, M. O., Swan, C. M., Dang, C. K., McKie, B. G., Bardgett, R. D., Wall, D. H., & Hättenschwiler, S. (2010). Diversity meets decomposition. Trends in Ecology and Evolution, 25(6), 372–380. http://doi.org/10.1016/j.tree.2010.01.010 |
| | 2010 | Narwani, A., & Mazumder, A. (2010). Community composition and consumer identity determine the effect of resource species diversity on rates of consumption. Ecology, 91(12), 3441–3447. http://doi.org/10.1890/10-0850.1 |
| | 2010 | Peter, H., Beier, S., Bertilsson, S., Lindström, E. S., Langenheder, S., & Tranvik, L. J. (2010). Function-specific response to depletion of microbial diversity. The ISME Journal, 5(2), 351–361. http://doi.org/10.1038/ismej.2010.119 |
| | 2010 | Hengst, A., Melton, J., & Murray, L. (2010). Estuarine Restoration of Submersed Aquatic Vegetation: The Nursery Bed Effect. Restoration Ecology, 18(4), 605–614. http://doi.org/10.1111/j.1526-100X.2010.00700.x |
| | 2009 | Poorter, H., Niinemets, U., Poorter, L., Wright, I. J., & Villar, R. (2009). Causes and consequences of variation in leaf mass per area (LMA): a meta-analysis. The New Phytologist, 182(3), 565–588. |
| | 2009 | Scherber, C., Eisenhauer, N., Weisser, W. W., Schmid, B., Voigt, W., Fischer, M., et al. (2010). Bottom-up effects of plant diversity on multitrophic interactions in a biodiversity experiment. Nature, 468(7323), 553–556. http://doi.org/10.1038/nature09492 |
| | 2009 | Schmid, B., Balvanera, P., Cardinale, B. J., Godbold, J., Pfisterer, A. B., Raffaelli, D., et al. (2009). Consequences of species loss for ecosystem functioning: meta-analyses of data from biodiversity experiments. In Biodiversity, Ecosystem Functioning, and Human Wellbeing (pp. 14–29). Oxford University Press. http://doi.org/10.1093/acprof:oso/9780199547951.003.0002 |

| Theme | Year | Reference |
| --- | --- | --- |
| | 2009 | Caliman, A., Pires, A. F., Esteves, F. A., Bozelli, R. L., & Farjalla, V. F. (2009). The prominence of and biases in biodiversity and ecosystem functioning research. Biodiversity and Conservation, 19(3), 651–664. http://doi.org/10.1007/s10531-009-9725-0 |
| | 2009 | FRIBERG, N., DYBKJAER, J. B., OLAFSSON, J. S., GISLASON, G. M., LARSEN, S. E., & LAURIDSEN, T. L. (2009). Relationships between structure and function in streams contrasting in temperature. Freshwater Biology, 54(10), 2051–2068. http://doi.org/10.1111/j.1365-2427.2009.02234.x |
| | 2008 | LECERF, A., & Chauvet, E. (2008). Diversity and functions of leaf-decaying fungi in human-altered streams. Freshwater Biology, 53(8), 1658–1672. http://doi.org/10.1111/j.1365-2427.2008.01986.x |
| | 2008 | McKie, B. G., Woodward, G., Hladyz, S., Nistorescu, M., Preda, E., Popescu, C., et al. (2008). Ecosystem functioning in stream assemblages from different regions: contrasting responses to variation in detritivore richness, evenness and density. Journal of Animal Ecology, 77(3), 495–504. http://doi.org/10.1111/j.1365-2656.2008.01357.x |
| | 2007 | Bracken, M. E. S., Bracken, B. E., & Rogers-Bennett, L. (2007). Species diversity and foundation species: Potential indicators of fisheries yields and marine ecosystem functioning. California Cooperative Oceanic Fisheries Investigations Reports, 48, 82–91. |
| | 2007 | Szabó, K. (2007). Importance of rare and abundant populations for the structure and functional potential of freshwater bacterial communities. Aquatic Microbial Ecology, 47(1), 1–10. |
| | 2006 | Schmitz, O. (2006). Predators have large effects on ecosystem properties by changing plant diversity, not plant biomass. Ecology, 87(6), 1432–1437. |
| | 2006 | Balvanera, P., Pfisterer, A. B., Buchmann, N., He, J.-S., Nakashizuka, T., Raffaelli, D., & Schmid, B. (2006). Quantifying the evidence for biodiversity effects on ecosystem functioning and services. Ecology Letters, 9(10), 1146–1156. http://doi.org/10.1111/j.1461-0248.2006.00963.x |
| | 2004 | Covich, A. P., Austen, M. C., Barlocher, F., Chauvet, E., Cardinale, B. J., Biles, C. L., et al. (2004). The Role of Biodiversity in the Functioning of Freshwater and Marine Benthic Ecosystems. BioScience, 54(8), 767–775. http://doi.org/10.1641/0006-3568(2004)054[0767:trobit]2.0.co;2?ref=search-gateway:23e3dc66023b36c1a2228c22176e1e97 |
| | 2002 | Worm, B., Lotze, H. K., Hillebrand, H., & Sommer, U. (2002). Consumer versus resource control of species diversity and ecosystem functioning. Nature, 417(6891), 848–851. http://doi.org/10.1038/nature00830 |
| **BES** | 2016 | Yang, W., Sun, T., & Yang, Z. (2016). Does the implementation of environmental flows improve wetland ecosystem services and biodiversity? A literature review. Restoration Ecology, 24(6), 731–742. http://doi.org/10.1111/rec.12435 |
| | 2014 | Balvanera, P., Siddique, I., Dee, L., Paquette, A., Isbell, F., Gonzalez, A., et al. (2014). Linking Biodiversity and Ecosystem Services: Current Uncertainties and the Necessary Next Steps. BioScience, 64(1), 49–57. http://doi.org/10.1093/biosci/bit003 |

| Theme | Year | Reference |
|---|---|---|
| | 2014 | Meli, P., Rey Benayas, J. M., Balvanera, P., & Martínez Ramos, M. (2014). Restoration enhances wetland biodiversity and ecosystem service supply, but results are context-dependent: a meta-analysis. PLoS ONE, 9(4), e93507. http://doi.org/10.1371/journal.pone.0093507 |
| | 2013 | van Oudenhoven, A. P. E., & de Groot, R. S. (2013). Protecting biodiversity and safeguarding ecosystem services provision in a changing world. International Journal of Biodiversity Science, Ecosystem Services & Management, 9(4), 277–280. http://doi.org/10.1080/21513732.2013.858441 |
| | 2012 | Polasky, S., Johnson, K., Keeler, B., Kovacs, K., Nelson, E., Pennington, D., et al. (2012). Are investments to promote biodiversity conservation and ecosystem services aligned? Oxford Review of Economic Policy, 28(1), 139–163. http://doi.org/10.1093/oxrep/grs011 |
| | 2011 | Stehle, S., Elsaesser, D., Gregoire, C., Imfeld, G., Niehaus, E., Passeport, E., et al. (2011). Pesticide risk mitigation by vegetated treatment systems: a meta-analysis. Journal of Environment Quality, 40(4), 1068–1080. http://doi.org/10.2134/jeq2010.0510 |
| | 2006 | Balvanera, P., Pfisterer, A. B., Buchmann, N., He, J.-S., Nakashizuka, T., Raffaelli, D., & Schmid, B. (2006). Quantifying the evidence for biodiversity effects on ecosystem functioning and services. Ecology Letters, 9(10), 1146–1156. http://doi.org/10.1111/j.1461-0248.2006.00963.x |
| | 2006 | Worm, B., Barbier, E. B., Beaumont, N., Duffy, J. E., Folke, C., Halpern, B. S., et al. (2006). Impacts of biodiversity loss on ocean ecosystem services. Science (New York, N.Y.), 314(5800), 787–790. http://doi.org/10.1126/science.1132294 |
| **BD - Benefit** | 2016 | Rudd, M. A., Andres, S., & Kilfoil, M. (2016). Non-use Economic Values for Little-Known Aquatic Species at Risk: Comparing Choice Experiment Results from Surveys Focused on Species, Guilds, and Ecosystems. Environmental Management, 58(3), 476–490. http://doi.org/10.1007/s00267-016-0716-0 |
| | 2016 | Whitehead, J. (2012). Ecosystems and biodiversity: Alternative perspective. In Global Problems Smart Solutions: Costs and Benefits (pp. 124–136). http://doi.org/10.1017/CBO9781139600484004 |
| | 2016 | Tyllianakis, E., & Skuras, D. (2016). The income elasticity of Willingness-To-Pay (WTP) revisited: A meta-analysis of studies for restoring Good Ecological Status (GES) of water bodies under the Water Framework Directive (WFD). Journal of Environmental Management, 182, 531–541. http://doi.org/10.1016/j.jenvman.2016.08.012 |
| | 2015 | Guareschi, S., Abellán, P., Laini, A., Green, A. J., Sánchez-Zapata, J. A., Velasco, J., & Millán, A. (2015). Cross-taxon congruence in wetlands: Assessing the value of waterbirds as surrogates of macroinvertebrate biodiversity in Mediterranean Ramsar sites. Ecological Indicators, 49, 204–215. http://doi.org/10.1016/j.ecolind.2014.10.012 |

| Theme | Year | Reference |
|---|---|---|
| | 2013 | Ghermandi, A., Ding, H., & Nunes, P. A. L. D. (2013). The social dimension of biodiversity policy in the European Union: Valuing the benefits to vulnerable communities. Environmental Science and Policy, 33, 196–208. http://doi.org/10.1016/j.envsci.2013.06.004 |
| | 2013 | Ruiz-Frau, A., Hinz, H., Edwards-Jones, G., & Kaiser, M. J. (2013). Spatially explicit economic assessment of cultural ecosystem services: Non-extractive recreational uses of the coastal environment related to marine biodiversity. Marine Policy, 38, 90–98. http://doi.org/10.1016/j.marpol.2012.05.023 |
| | 2012 | Ahtiainen, H., & Vanhatalo, J. (2012). The value of reducing eutrophication in European marine areas — A Bayesian meta-analysis. Ecological Economics, 83, 1–10. http://doi.org/10.1016/j.ecolecon.2012.08.010 |
| | 2011 | Laycock, H. F., Moran, D., Smart, J. C. R., Raffaelli, D. G., & White, P. C. L. (2011). Evaluating the effectiveness and efficiency of biodiversity conservation spending. Ecological Economics, 70(10), 1789–1796. http://doi.org/10.1016/j.ecolecon.2011.05.002 |
| | 2010 | Ghermandi, A., van den Bergh, J. C. J. M., Brander, L. M., de Groot, H. L. F., & Nunes, P. A. L. D. (2010). Values of natural and human-made wetlands: A meta-analysis. Water Resources Research, 46(12), –n/a. http://doi.org/10.1029/2010WR009071 |
| | 2010 | Hatton, M. D. (2010). Valuing biodiversity using habitat types. Australasian Journal of Environmental Management, 17(4), 235–243. |
| | 2009 | Kontoleon, A., Pascual, U., & Swanson, T. (Eds.). (2009). Valuing ecological and anthropocentric concepts of biodiversity: a choice experiments application. In Biodiversity Economics (pp. 343–368). Cambridge: Cambridge University Press. http://doi.org/10.1017/CBO9780511551079.015 |
| | 2006 | Christie, M., Hanley, N., Warren, J., Murphy, K., Wright, R., & Hyde, T. (2006). Valuing the diversity of biodiversity. Ecological Economics, 58(2), 304–317. http://doi.org/10.1016/j.ecolecon.2005.07.034 |
| | 2006 | Thompson, R., & Starzomski, B. M. (2006). What does biodiversity actually do? A review for managers and policy makers. Biodiversity and Conservation, 16(5), 1359–1378. http://doi.org/10.1007/s10531-005-6232-9 |
| **ESS - Benefit** | 2016 | Quintas-Soriano, C., Martín-Lopez, B., Santos-Martin, F., Loureiro, M., Montes, C., Benayas, J., & García-Llorente, M. (2016). Ecosystem services values in Spain: A meta-analysis. Environmental Science and Policy, 55, 186–195. http://doi.org/10.1016/j.envsci.2015.10.001 |
| | 2016 | Schmidt, S., Manceur, A. M., & Seppelt, R. (2016). Uncertainty of Monetary Valued Ecosystem Services - Value Transfer Functions for Global Mapping. PLoS ONE, 11(3), e0148524. http://doi.org/10.1371/journal.pone.0148524 |
| | 2016 | Grizzetti, B., Lanzanova, D., Liquete, C., Reynaud, A., & Cardoso, A. C. (2016). Assessing water ecosystem services for water resource management. Environmental Science and Policy, 61, 194–203. http://doi.org/10.1016/j.envsci.2016.04.008 |

| Theme | Year | Reference |
|---|---|---|
| | 2014 | Ghermandi, A., & Nunes, P. A. L. D. (2014). The benefits of coastal recreation in Europe's seas: an application of meta-analytical value transfer and GIS (pp. 313–333). Edward Elgar Publishing. http://doi.org/10.4337/9781781955161.00027 |
| | 2013 | Brander, L., Brouwer, R., & Wagtendonk, A. (2013). Economic valuation of regulating services provided by wetlands in agricultural landscapes: A meta-analysis. Ecological Engineering, 56, 89–96. http://doi.org/10.1016/j.ecoleng.2012.12.104 |
| | 2013 | Ghermandi, A., & Nunes, P. A. L. D. (2013). A global map of coastal recreation values: Results from a spatially explicit meta-analysis. Ecological Economics, 86, 1–15. http://doi.org/10.1016/j.ecolecon.2012.11.006 |
| | 2012 | de Groot, R., Brander, L., van der Ploeg, S., Costanza, R., Bernard, F., Braat, L., et al. (2012). Global estimates of the value of ecosystems and their services in monetary units. Ecosystem Services, 1(1), 50–61. http://doi.org/10.1016/j.ecoser.2012.07.005 |
| **scale** | 2016 | Vellend, M., Dornelas, M., Baeten, L., Beauséjour, R., Brown, C. D., De Frenne, P., et al. (2016). Estimates of local biodiversity change over time stand up to scrutiny. Ecology. http://doi.org/10.1002/ecy.1660 |
| | 2015 | Quesnelle, P. E., Lindsay, K. E., & Fahrig, L. (2015). Relative effects of landscape-scale wetland amount and landscape matrix quality on wetland vertebrates: a meta-analysis. Ecological Applications, 25(3), 812–825. http://doi.org/10.1890/14-0362.1 |
| | 2013 | Vellend, M., Baeten, L., Myers-Smith, I. H., Elmendorf, S. C., Beauséjour, R., Brown, C. D., et al. (2013). Global meta-analysis reveals no net change in local-scale plant biodiversity over time. Proceedings of the National Academy of Sciences of the United States of America, 110(48), 19456–19459. http://doi.org/10.1073/pnas.1312779110 |
| | 2011 | Brander, L. M., Bräuer, I., Gerdes, H., Ghermandi, A., Kuik, O., Markandya, A., et al. (2011). Using Meta-Analysis and GIS for Value Transfer and Scaling Up: Valuing Climate Change Induced Losses of European Wetlands. Environmental and Resource Economics, 52(3), 395–413. http://doi.org/10.1007/s10640-011-9535-1 |

## AQUACROSS PARTNERS

**Ecologic Institute** (ECOLOGIC) | Germany
**Leibniz Institute of Freshwater Ecology and Inland Fisheries** (FVB-IGB) | Germany
**Intergovernmental Oceanographic Commission of the United Nations Educational, Scientific and Cultural Organization** (IOC-UNESCO) | France
**Stichting Dienst Landbouwkundig Onderzoek** (IMARES) | Netherlands
**Fundación IMDEA Agua** (IMDEA) | Spain
**University of Natural Resources & Life Sciences, Institute of Hydrobiology and Aquatic Ecosystem Management** (BOKU) | Austria
**Universidade de Aveiro** (UAVR) | Portugal
**ACTeon – Innovation, Policy, Environment** (ACTeon) | France
**University of Liverpool** (ULIV) | United Kingdom
**Royal Belgian Institute of Natural Sciences** (RBINS) | Belgium
**University College Cork, National University of Ireland** (UCC) | Ireland
**Stockholm University, Stockholm Resilience Centre** (SU-SRC) | Sweden
**Danube Delta National Institute for Research & Development** (INCDDD) | Romania
**Eawag – Swiss Federal Institute of Aquatic Science and Technology** (EAWAG) | Switzerland
**International Union for Conservation of Nature** (IUCN) | Belgium
**BC3 Basque Centre for Climate Change** (BC3) | Spain


| | |
|---|---|
| **Contact** | aquacross@ecologic.eu |
| **Coordinator** | Dr. Manuel Lago, Ecologic Institute |
| **Duration** | 1 June 2015 to 30 November 2018 |
| **Website** | http://aquacross.eu/ |
| **Twitter** | @AquaBiodiv |
| **LinkedIn** | www.linkedin.com/groups/AQUACROSS-8355424/about |
| **ResearchGate** | www.researchgate.net/profile/Aquacross_Project2 |