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Preliminary identification of the indicators of benthic biodiversity and ecosystem function for the test phase in different case studies

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Background and State of the Art

The relationship between biodiversity and ecosystem functioning has been assessed using many different kinds of variables in the literature (Bengtsson,1998). The main goal of this summary is to review these potential relationships and analyse thresholds of change of ecosystem status on different type of communities. For this purpose, scientific literature about these relationships and thresholds of change of ecosystem status in terrestrial and aquatic ecosystems has been reviewed in order to apply them in the case of benthic communities.

Biodiversity have been quantified in different ways. Several studies have quantified biodiversity as species richness (McGrady-Steed *et al.*,1997, Stachowicz *et al.*,1999, Bascompte & Rodriguez,2001, Engelhardt & Ritchie,2001, Mulder *et al.*,2001, Lyons & Schwartz,2001, Kennedy *et al.*,2002). Few articles have considered species composition (Seiderer and Newell,1999), functional diversity (Walker *et al.*,1999, Nijs & Impens,2000), diversity within functional groups (species redundancy, that is species that play similar roles in a community) (Borrvall *et al.*,2000, Nijs & Impens,2000), genetic diversity (Hughes & Stachowicz,2004), or habitat complexity (Ceccherelli *et al.*,2002, Kelaher,2003) as an alternative approach to study the biodiversity within a community. Some authors have focused their studies on the role of particular species on ecosystem functioning (Bulleri *et al.*,2002, Kraufvelin & Salovius,2004). Even, a few studies have combined some of these approaches related to any aspect of ecosystem functioning (Tilman *et al.*,1997, Loreau *et al.*,2001. Mikola *et al.*,2002, Micheli & Halpern,2005). The approach of these studies includes natural (Seiderer and Newell,1999) or manipulation experiments in the field (McGrady-Steed *et al.*,1997, Tilman *et al.*,1997, Lyons & Schwartz,2001) or in laboratory conditions (Yachi & Loreau,1999, Borrvall *et al.*,2000, Engelhardt & Ritchie,2001, Micheli & Halpern,2005). Very few studies have been done combining these two experimental approaches (Stachowicz *et al.*,2002).

Ecosystem functioning has been quantified as productivity (Huston *et al.*,2000, Mulder *et al.*, 2001, Downing & Leibold,2002, Smith & Knapp,2003), non-native species invasion resistance (Stachowicz *et al.*,1999, Lyons and Schwartz,2001, Kennedy *et al.*,2002, Klein *et al.*,2005) or stability/reliability/resilience of the community after disturbances (Naeem,1998, Walker *et al.*,1999, McCann,2000, Nijs & Impens,2000, Aoki & Mizushima,2001).

Bolger (2001) summarizes the general ideas used by studies about the relationship between biodiversity and ecosystem functioning in four global hypothesis:

- Null hypothesis: ecosystem functioning is not affected if one or more species are add or deleted.
- Diversity-stability hypothesis: there's a positive linear relationship between biodiversity and ecosystem productivity or resistance after a disturbance.
- Rivet hypothesis: there's a non-linear relationship between species richness and ecosystem functioning.
- Redundancy hypothesis: the loss of certain species can be compensated by others that play similar roles and ecosystem functioning is not affected.
- Idiosyncratic hypothesis: there's an indeterminated relationship between species composition and ecosytem functioning and it depends of which species are removed.

Problems about this question are mainly two. First, which variable reflects better the biodiversity of a community? and second which experimental design is closer to the natural conditions of a community?.

Traditionally, the concept of biodiversity has been measured as species richness. Nevertheless, alternative studies have considered that this approach is not the best one because of there are species more important than other ones for ecosystem functioning (Tilman *et al.*,1997, Engelhardt & Ritchie,2001). Even, some papers have considered that the occurrence of certain species within a community are essential for ecosystem functioning (Seiderer and Newell,1999). While other species can be functionally replaced by other ones that play similar roles within the community (Naeem,1998, Borrvall *et al.*,2000, Nijs & Impens,2000) at least at short-term (Kraufvelin & Salovius,2004). It would be interesting to know how these different approaches are related as some papers have studied (Tilman *et al.*,1997, Loreau *et al.*,2001. Mikola *et al.*,2002, Micheli & Halpern,2005).

Moreover, experimental design is not a negligible question when the relationship between biodiversity and ecosystem functioning is analysed. The vast majority of the study designs involve manipulation experiments due to large difficulties for controlling confounding factors that can affect in situ studies. But, manipulation experiments are not free from an important number of problems. For example, some studies have been based on the effect of the random removal of one or more species (Lyons & Schwartz,2001), but maybe, in nature, extinction of species after a disturbance are not produced randomly because it depends of its abundance, so uncommon or rare species will lost first (Smith & Knapp,2003).

Indeed, the study of the effect of biodiversity on different variables that define ecosystem functioning can suppose some difficulties. Community productivity seems to show different trends in relation with biodiversity in different situations. While, the majority of the literature have found a positive linear (Mulder *et al.*,2001) or non-linear (Tilman *et al.*,1997) correlation, others haven't observed any relation between them (Huston *et al*.,2000). Although few studies haven't found any relation between diversity and invasion resistance (Klein *et al.*,2005), most of them seem to show a positive correlation with biodiversity (Stachowicz *et al.*,1999, Davis *et al.*,2000, Kennedy *et al.*,2002, Stachowicz *et al.*,2002) but there's not only a unique theory that explain it in literature. For example, this positive correlation has been explained assuming that when biodiversity increases it reduces the free space available for new colonizers (Stachowicz *et al.*,1999, Stachowicz *et al.*,2002) or because it reduces amount of available resources (Davis *et al.*,2000) for invasive species.

In summary, current scientific literature has shown that, possibly, it doesn´t exist a universal principle that defines the relationship between biodiversity and ecosystem functioning. Until now, few studies based on this relationship have obtained curves that suggest thresholds of change of community (Tilman *et al*.,1997, Stachowicz *et al*.,2002).

Tilman et al.,1997 performed a field experiment in which plant species diversity (defined as number of plant species added to plots) and functional diversity (defined as number of functional groups added to plots) related with plant productivity was assessed. For this purpose, 0,1,2,4,8,16 or 32 perennial savanna-grassland species representing 0,1,2,3,4 or 5 functional groups were planted in 289 plots of 169m2 each. They described an asymptotic curves in both cases that suggested thresholds of change of community status (Fig.1, page 1300).

Stachowicz et al.,2002 recorded sessile native species (from 0 to 14 species) of a subtidal marine invertebrate community in individual 25x25 cm quadrats related with proportion of plots containing nonnative sessile species at 12 sites (10 quadrats per site). Also, they obtained a strong non-linear regression that suggested a threshold of change of community status (Fig.2,page 2580).

Further studies are needed to identify the best study approaches (selection of variables and experimental designs) to find thresholds of change of community status in the case of benthic ecosystems.

Recommended the indicators of benthic biodiversity and ecosystem function for the test phase in different case studies

The main goal of this deliverable is to derive opperational (i.e. Simple, aggreagated, generic, and applicable to monitoring/management programs) indicators of benthic community structure, status and dynamics sensitive to pressures (e.g. Nutrient inputs). Provided the experience and knowledge reported above, as well as the constraints on data availability for the various case studies, the following benthic indicators are proposed as the parsimonious and suitable to assess the link between benthic biodiversity and ecosystem function:

1. **Macrophyte cover and biomass** (units, % and g DW m^{-2}): This indicator reflects the % of the sea floor covered by macrophytes (macroalgae or seagrass), which reflect adequate light penetration and suitable oxygen conditions at the study sites and provide, in addition, and indication of the habitat available, which is an important determinant of benthic biodiversity.
2. **Benthic to Pelagic Biomass ratio** (non dimensional): This indicator reflects the partitioning of the autotrophic biomass between planktonic and benthic primary producers. This partitioning has important consequences for the cycling of carbon in ecosystems, their response and vulnerability to nutrient inputs, and affects greatly benthic biodiversity.
3. **Seagrass to Macroalgae ratio** (non dimensional, calculated from biomass or cover data): This indicator reflects the partitioning of benthic cover between seagrass and macroalgae, which are conducive to very contrasting food webs, benthic biodiversity and responses to stresses, such as nutrient inputs. The partitioning between these two types of benthic primary producers is an indicator highly sensitive to changes in nutrient inputs and disturbance to the ecosystems.
4. **Macrophyte depth limits**. (units, m). Macrophyte depth limits are critical and reliable, easy-to-determine indicators of environmental quality, largely driven by underwater light penetration as affeteced by water quality, that determine the maximum possible extent of benthic

vegetation. It has been used successfully to monitor environmental quality and changes in coastal ecosystems and is recommended here as an integrative, robust indicator of ecosystem status and quality. It is highly sensitive to increase nutrient inputs (Duarte 1991, 1995).

5. **Species richness** (units, number). The number of species of macrozoobenthos and macrophytobenthos at a particular coastal site.

6. **Crustacean to Polyquete biomass ratio** (non dimensional). This simple, but robust index can be used to detect community shifts following disturbance, such as increased eutrophication and associated adverse effects, such as hypoxia, as polyquetes are more resistant to those adverse conditions than crustaceans, which are relatively sensitive, are.

7. **Presence of invasive species.** (units, number). An increased number of invasive species may provide indications of changes in environmental conditions, which affects the vulnerability to species invasion, as well as alert to possible future changes as the invasive species alter food web interactions and resource partitioning in the ecosystem.

8. **Community stability** (units, rate of change in either of the above properties vs. time). Monitoring of either of the simple indicators reported above with time provides an opportunity to detect sudden or abrupt changes in these properties. This requires the calculations of rates of change, between consecutive sampling events, and the development of some prior statistics describing the dynamics of the undisturbed system which can be used to test unusual or abrupt changes in community stability, detected as unusually high rates of change exceeding the 95% confidence limits of the climatology. More sophisticated analyses, such as intervention analyses and others (cf. Stream 2) can then be used to model these dynamics and assess the significance of the sudden change in terms of regime shift of the ecosystem and its link to external pressures.

Applications

The indicators above are being used already to address threshold effects at a number of case studies:

- Thresholds of sustainability of fish farms (case study).
- Nutrient thresholds for shifts between pelagic and benthic primary producers.
- Thresholds of nutrient inputs for sediment communities.
- Thresholds of sediment communities to hypoxia.
- Thresholds of light penetration for seagrass communities.

The dissemination of this deliverable amongst the project participants will encourage their application to additional cases, as well as to data bases compiled within the project.

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