BIOLOGICAL CONSERVATION

Do we protect biological originality in protected areas? A new index and an application to the Bonifacio Strait Natural Reserve

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ABSTRACT

Changes in biodiversity may disrupt the ecological functions performed by species assemblages. Hence, we urgently need to examine the implications of biodiversity loss not only in terms of species conservation but also in terms of sustainability of ecosystem services. The ability of protected areas to maintain local species richness has been clearly demonstrated. However, preserving goods and services provided by ecosystems requires not only the conservation of species richness but also the conservation of the most 'original' species, i.e. the ones with the highest average rarity of their attributes which are likely to perform some unique functions in ecosystems. We proposed a new conservation of biological originality (CBO) index as well as associated randomization tests to quantify the ability of protected areas to maintain viable populations for the most original species. As an application, we used long-term fisheries data collected in the Bonifacio Strait Natural Reserve (BSNR) to determine the species which benefited from the protection reinforcement in 1999. We also estimated a set of 14 ecomorphological functional traits on the 37 fish species caught in the BSNR and we obtained a functional originality value for each species. As a result, we found that functional originality was significantly protected in the fish assemblage of the BSNR: species with the most original functional trait combinations became more abundant after 1999. Our finding suggests that protecting most original species is an insurance against functional diversity erosion in the BSNR. More generally, our new index can be used to test whether protected areas may protect preferentially the most original species and whether restorative management promotes the reestablishment of the most original species with particular habitat requirements.

1. Introduction

In a natural world increasingly transformed by human activities, there is more and more evidence for declining trends in biodiversity for both terrestrial (Thomas et al., 2004) and marine realms (Roberts and Hawkins, 1999). A few major sources of ecological alterations may be identified from the long list of factors explaining these trends. For terrestrial ecosystems, changes in land use have probably had the largest effect, followed by climate change, nitrogen deposition, biotic exchange and elevated carbon dioxide concentration (Sala et al., 2000). For aquatic ecosystems, the most important

factors are certainly climatic change, biotic exchange and overfishing; the latter being the most direct human disturbance to all coastal ecosystems (Jackson et al., 2001). Land use change has also significant impact on inland, estuarine and near shore coastal waters (e.g. Ramos Miranda et al., 2005).

The causes of biodiversity loss are well established and the consequences of such dramatic declines or alterations have spurred considerable research and tremendous debate (reviewed in Hooper et al. (2005)). Indeed biodiversity should be preserved not only for aesthetic reasons and for its direct usefulness but also for its indirect benefits through services that species provide to ecosystems (Chapin et al., 2000). For instance, it has been experimentally demonstrated that, locally, species richness *per se* positively influences ecosystem functioning and some fundamental properties of ecosystems such as productivity, resistance to invasion, stability and resilience (e.g. Loreau et al., 2001). Thus, as alterations of biodiversity may disrupt ecological functions performed by species assemblages (Hughes et al., 2003), it is urgent to carefully examine the implications of biodiversity loss not only in terms of species conservation but also in terms of sustainability of ecosystem services upon which human welfare depends.

Protected areas are indisputably the primary tool for *in situ* biodiversity conservation across the world (Ravenel and Redford, 2005) with more than 100,000 sites covering nearly 11.7 per cent of the land surface of the planet and about one per cent of the marine environment (Bishop et al., 2004). However, protected areas have been set up with respect to species and habitat considerations rather than based on considerations about the functioning of ecological systems. For instance, in protected areas, biodiversity is almost exclusively assimilated to species richness (the number of species coexisting on a site) while the definition of biodiversity includes various facets of the diversity of life (Purvis and Hector, 2000). It is thus worth noting that the most widely used measure of biodiversity ignores what makes species different in an assemblage (Cousins, 1991): their relative abundances and their biological traits. Yet, these two facets of biodiversity, which are based on differences among species, are known to influence ecosystem functioning. They are generally measured by two indices: (i) evenness, which measures the relative distribution of abundance among species and is positively related to resistance against invasion (Wilsey and Polley, 2002); and (ii) functional diversity, which measures the value and range of functional traits in organism assemblages, is now widely recognized as a main driver of ecosystem processes in both terrestrial (Petchey et al., 2004) and aquatic (Waldbusser et al., 2004) environments. Beyond protecting species richness against erosion, we should thus ask the question of whether protected areas are able to maintain these other facets of biodiversity.

When considering differences among species to assess the diversity of an assemblage we can assume that the species that contributes more to the biological diversity of this assemblage is the one with the most original features, i.e. the one with the highest average rarity of its attributes (Pavoine et al., 2005). From this standpoint, losing the most original species is more likely to result in losing some unique biolog-

ical features such as life-history traits, morphological attributes or behaviours. This is obvious for some species-poor taxonomic groups (e.g. rhinoceros and kiwis; Purvis et al., 2000). In turn, the loss of some rare biological features may significantly disrupt ecosystem functioning. For instance only a few species of parrotfish, with particularly long and strong jaws, can substantially erode reef carbonate of dead corals (Bellwood and Choat, 1990). When such species are threatened by overfishing (like in Caribbean coral reefs) the bioerosion of dead corals by fish decreases. As a result, the increased spatial extent of dead coral may prevent new coral settlement on fragile or unstable foundations (Bellwood et al., 2004). The absence of species able to play the same role as eroding parrotfish thus makes the preservation of their functionally original attributes essential to the recovery of coral reef ecosystems. More generally, the degree of functional originality of species within an assemblage determines the strength and shape of the relationship between taxonomic
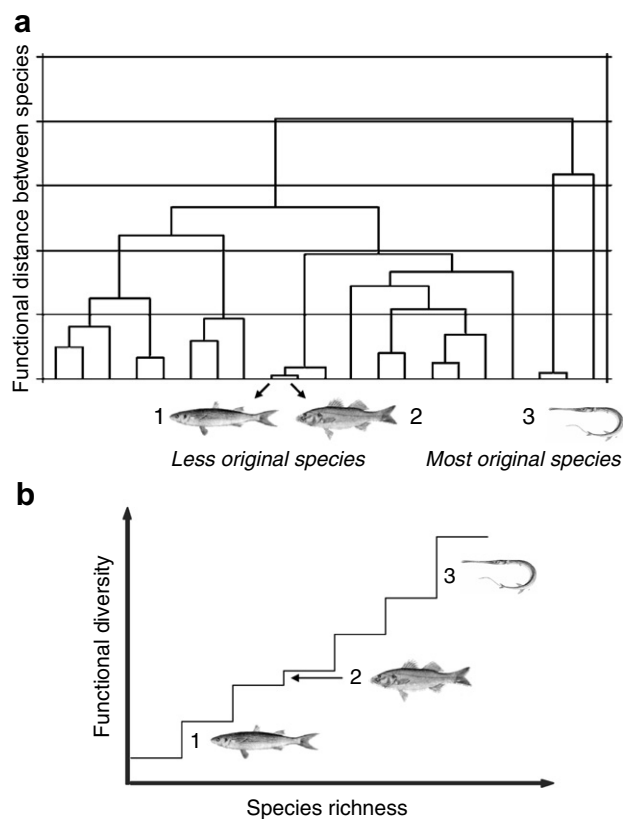
Fig. 1 – (a) Location of the most original species (3) and the less original species (1 and 2) on a dendrogram based on functional distances between species. **(b)** Species 1, 2 and 3 contribute in a different way to the relationship between functional diversity and species richness. Each species contributes to 1 on the species richness axis (whatever their originality value) while their contribution on the functional diversity axis depends on their originality compared to the species already considered. As an illustration, the loss of the most original species (3) would decrease substantially functional diversity while the loss of species 2 does not affect substantially functional diversity as a redundant species (1) is already present in the assemblage.