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Measuring ecosystem degradation through half a century of fish species introductions and extirpations in a large isolated lake

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ABSTRACT

The introduction of exotic species and the extirpation of native species that occurred during the past two centuries have strongly modified the structure of most plant and animal assemblages across the globe. Such a biotic change is particularly marked in isolated environments such as islands or isolated lakes. Most studies reported drastic changes between before and after human disturbances, but the dynamics of change in assemblage structure through the invasion and extirpation processes are rarely reported. Here we measured the aquatic ecosystem degradation through exotic species introduction and native species extirpation experienced by Lake Erhai (China) during the last 50 years using structural, functional and taxonomic distinctness biodiversity indices. Structural diversity (species richness) did not varied monotonically along the temporal gradient, due to an opposite trend between exotic species increase and a concomitant decline of native species richness. Functional diversity displayed unclear ascending trends driven by the introduction of exotic species having distinct functional traits than natives. Taxonomic distinctness indices exhibited an increase of the average taxonomic distinctness (Δ^+), but a decrease of the variation in taxonomic distinctness (Λ^+) through time. Structural, functional and distinctness indices providing complementary information on ecosystem degradation, we here proposed a new multifaceted degradation index integrating these three facets of biodiversity. Such an index provided an accurate representation of the faunistic changes experienced by Lake Erhai and might constitute a comprehensive way to measure ecosystem degradation through exotic fish species introductions and native fish species extirpations.

1. Introduction

The freshwater biodiversity is currently experiencing an unprecedented worldwide decline (Dudgeon et al., 2006; Olson et al., 2002; Poff, 2014). This decline is particularly marked in lakes that suffer from multiple human disturbances such as habitat fragmentation and loss, hydrologic alteration, climate change, overexploitation and pollution that are responsible from a marked extinction trend (Dudgeon et al., 2006; Sala et al., 2000). In addition to extinctions, human-induced exotic species introductions are also listed as one of the most detrimental anthropogenic activities (Dudgeon et al., 2006; Sala et al., 2000). Exotic species can threaten aquatic biodiversity and influence ecological process (Clarkson et al., 2005; Jeschke et al., 2014; Volta et al., 2013). There is thereby a pressing need to manage the use of biodiversity and resources, and to conserve and restore ecological processes. This necessitates developing appropriate metrics to measure and manage biological diversity and its changes through extinctions and invasions.

A wide variety of indices are currently available in biodiversity assessment (Gallardo et al., 2011; Lyashevska and Farnsworth, 2012; Purvis and Hector, 2000). These indices can be classified into three major categories: structural/compositional diversity metrics, taxonomic distinctness metrics, and functional diversity metrics (Lyashevska and Farnsworth, 2012). The most commonly used metrics are structural diversity indices, such as richness, Shannon index, Pielou's Eveness, or Simpson index. All these indices take into account species richness and/or evenness, but they are facing criticism for uncertain ecological interpretation, excessive data requirement, insensitiveness to disturbance, and oversimplification (Brosse et al., 2011; von Euler and Svensson, 2001).

Functional diversity is defined as the diversity of functional traits in a community weighted by their abundances (Villéger et al., 2008). Functional diversity indices consider the biological characteristics of the species, i.e., value, range, distribution, and relative

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abundance of the functional traits in a community. It was efficiently used to detect functional changes in fish assemblages across Europe due to species introductions and extirpations (Villéger et al., 2014). Functional approaches are nevertheless limited by the necessity to run precise measures on living or well-conserved specimens, which is rarely feasible after the extinction of endemic species (Cordlandwehr et al., 2013). Moreover, the functional approaches are also limited by their inability to detect anthropic disturbances when the extirpated species are replaced by functional similar non-native species (Wellnitz and Poff, 2001).

The taxonomic distinctness indices are a measurement of the taxonomic relatedness of species. They stipulate that for a given species richness, an assemblage including species from different families is more diverse than an assemblage with the same number of species from one family (Warwick and Clarke, 1998). These indices are widely used in biodiversity assessment due to their independence to sampling methods (e.g., sample size and sampling effort) and natural habitat type or complexity (Clarke and Warwick, 1999; Warwick and Clarke, 1998; Xu et al., 2012). Testing the sensitivity of taxonomic distinctness indices in discriminating anthropogenic impacts was the subject of a number of studies, but they yielded inconsistent results. Their sensitivity to human disturbances was shown in some studies (Campbell et al., 2011; Milosevic et al., 2012; Miranda et al., 2005), but not in others (Abellan et al., 2006; Costa et al., 2010; Leira et al., 2009).

Structural, functional, and distinctness indices reveal independent facets of biodiversity, and are hence complementary (Gallardo et al., 2011). Nevertheless, few studies have been devoted to a comprehensive consideration of those facets (Lyashevska and Farnsworth, 2012; Miranda et al., 2005; Strecker et al., 2011). The purpose of the present study is to analyze the response of these three facets through a gradient of disturbance strength in Lake Erhai (Yun-Gui Plateau, South China), an isolated lake system subjected to non-native species introduction and native species extirpation processes across a more than 50 years' time frame (from 1960 to 2014). We hence measured the sensitivity of compositional diversity, taxonomic distinctness, and functional diversity indices to the human mediated changes in fish assemblage composition, and evaluated the usefulness of each index as a metric to measure human disturbance estimator. Finally, we proposed a new multifaceted index integrating these three facets of biodiversity to measure the ecosystem degradation through fish species introductions and extirpations across time.

2. Methods

2.1. Study area

Lake Erhai (25°36′–25°58′ N; 100°06′–100°18′ E; 1973.7 m above the sea level) is the second largest freshwater lake from the Yun-Gui Plateau (Yunnan Province, South China; Fig. 1). It covers a surface area of c.a 249.8 km^2, for a volume of c.a. 25.3×10^8 m^3. It is 42 km long, and has an 8.4 km maximum width. Average depth is 10.5 m and maximum depth 20.9 m. This tectonic rift lake is subjected to a subtropical plateau monsoon climate, with mean annual precipitation of 1060 mm, mean annual air temperature of 15 °C (minimum: 8.5 °C; maximum: 20.1 °C), and mean annual evaporation of 1209 mm (Wang and Dou, 1998). It belongs to the upper Mekong River (Lancang-Jiang in Chinese) basin, and its water discharges into the Yangbi Jiang, an upstream tributary of this basin. Fish exchanges between this lake and the Yangbi-Jiang were blocked by a dam constructed in 1960s.

Lake Erhai, like other lakes on the Yun-Gui plateau, has a depaurate indigenous fish diversity with less than 30 species, in contrast to 60–70 species found in shallow-water lakes of the Yangtze River floodplain (Chu and Chen, 1989; Li, 1982). Lacking predator fishes and having plenty of unoccupied niches have led to a high degree of species endemism found in this lake. Historically, it harbored only 17 native species, 7 of which are endemic. Most of this native fish assemblage comes from two cyprinid genera: *Schizothorax* (4 species) and *Cyprinus* (5 species) (Du and Li, 2001; Li, 1982; Wu and Wang, 1999). These middle-sized fish species are of economic importance, and are thus exploited by local fisheries.

During the past six decades, Lake Erhai has been experiencing drastic changes in fish assemblages that were due to several human disturbances among which over-harvesting, exotic species introductions for aquaculture and eutrophication are the most obvious (Fei et al., 2012; Lin et al., 2013; Wu and Wang, 1999). Aquaculture activity started from the 1960s. Four domestic Chinese carps, i.e., the black carp (*Mylopharyngodon piceus*), the grass carp (*Ctenopharyngodon idella*), the silver carp (*Hypophthalmichthys molitrix*), and the bighead carp (*Aristichthys nobilis*), and the icefish (*Neosalanx taihuensis*) were purposefully introduced for fish farming. The others exotic fishes were introduced as fellow travelers, such as the topmouth gudgeon (*Pseudorasbora parva*), and gobies (*Rhinogobius giurinus*, and *Rhinogobius cliffordpopei*) (Du and Li, 2001; He et al., 2010; Wu and Wang, 1999). Following the establishment of those non-native species and the concomitant decline or extirpation of native species over the past 50 years, the main harvest species of Lake Erhai shifted from the middle-sized native fishes (*Schizothorax taliensis*, *Cyprinus chilia*, and *Cyprinus megalophthalmus*) to the small-sized exotic fishes (*P. parva*, *R. giurinus*, and *R. cliffordpopei*) (Du and Li, 2001; Tang et al., 2013; Wu and Wang, 1999). Currently, 11 out of the 17 native fish species are included in the China Red List of Threatened Species, with 6 listed as critically endangered.

2.2. Data sets

Historical fish assemblages of Lake Erhai as well as their changes through the last 60 years have been well documented in the literature and our data results from an extensive literature survey on the lake fish fauna since the fifties. The main data sources were Ley et al. (1963), Chu and Chen (1989), Chen et al. (1998), Zhou (2000), Du and Li (2001), Lu and Song (2003), Wang et al. (2006), He et al. (2010), Yuan et al. (2010), and Tang et al. (2013). On the basis of these published literature, we grouped data per decades giving rise to seven time periods: 1950s (1950–1959); 1960s (1960–1969), 1970s (1970–1979), 1980s (1980–1989), 1990s (1990–1999), 2000s (2000–2009), and 2010s (2010–2014). As sampling methods and efforts varied between periods, we did not consider fish abundance; only the presence or absence of the species during each period was considered, giving rise to a data matrix with 7 time periods and 39 fish species (Table 1).

The functional diversity measure in each period was based on a set of 10 biological and ecological traits related to fish morphology, trophy, life history, and habitat (Table 2). Trait assignments were compiled from the literature (scientific papers, books, and gray literature) and Fishbase (Froese and Pauly, 2014). The raw functional matrix contained continuous and categorical variables. Categorical variables were then transformed into binary variables to make all variables numerical.

2.3. Data analysis

Two taxonomic distinctness metrics were computed: the average taxonomic distinctness (Δ^+) and the variation in taxonomic distinctness (Λ^+). Δ^+ is defined as the average path length between all pairs of species through a Linnean hierarchical taxonomic tree; a sample with only multiple species within a single family will have lower taxonomic distinctness than a sample having multiple