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A proposal of framework to obtain an integrated biodiversity indicator for agricultural ponds incorporating the simultaneous effects of multiple pressures

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ABSTRACT

One of the promising approaches to monitoring biodiversity is assessing the status of pressures driving the biodiversity state. To achieve this, we need to identify the principal pressures that cause simultaneous biodiversity loss across taxonomic groups and clarify how multiple pressures act synergistically or at least simultaneously to decrease biodiversity in the focal ecosystem. Here, we used a series of 64 ponds as a case study and we developed a framework for an integrated biodiversity indicator that took into consideration the estimated relative importance of multiple pressures. The indicator is defined as a function of the pressure(s) and is parameterized to explain a number of individual indicators of biodiversity, such as richness, abundance, and functional diversity of focal taxa. We selected aquatic macrophytes, Odonata, and benthic macroinvertebrates as the focal taxa. In addition, we focused on three types of pressure: eutrophication (represented by total phosphorus, total nitrogen, suspended solids, chlorophyll *a*, and density of cyanobacteria of pond water), habitat destruction (land-use type around the pond and pond bank protection), and invasive alien species (abundance of bluegill, largemouth bass, red swamp crayfish, and American bullfrog). We then evaluated the relationships among direct pressures and the individual biodiversity indicators and used a hierarchical Bayesian approach to calculate the integrated biodiversity indicator. Using this framework, we demonstrated that eutrophication had greater effects on the state of biodiversity of the agricultural ponds than did habitat destruction or the presence of invasive alien species. We also showed that the integrated indicator could well explain the behaviors of several individual biodiversity indicators, including total richness, endangered species richness, and functional diversity of focal taxa. These results demonstrate the advantages of the framework in providing a more practical method for assessing biodiversity, and quantifying the relative importance of the major threats to biodiversity to prioritize strategies in conservation planning and policy making.

1. Introduction

Biodiversity is a complex, multiscale, multifaceted entity (Noss, 1990), and thus establishing integrated and operational indicators for the status of biodiversity is essential for effective monitoring (Duelli and Obrist, 2003; Scholes and Biggs, 2005). One of the promising approaches is to monitor biodiversity by assessing the status of pressures driving the biodiversity state. It is often easier to quantify these pressures at large spatial scales using remote-sensing techniques than to undertake field surveys to directly assess biodiversity indicators (Green et al., 2005), which include richness and diversity of target species or taxa. However, for this approach to be feasible, at least two problems must be addressed. One is that different taxonomic groups or guilds can behave differently in response to changes, even with the same driver or pressure (Perfecto et al., 2003; Schulze et al., 2004; Kadoya et al., 2009; Heino, 2010); the other is that in real systems, multiple, rather than single, pressures of ecological change can interact synergistically to accelerate biodiversity loss (Didham et al., 2007; Brook et al., 2008; Darling and Cote, 2008). The effects of multiple pressures are not predictable from single-pressure impacts, and such a prediction would represent a major source of uncertainty in the assessment of biodiversity by using pressures as surrogates. For example, adverse effects of invasion by an introduced species on a native ecosystem can be either accelerated or suppressed, depending on the degree of landscape modification (Didham et al., 2007). Thus, to achieve a framework for biodiversity monitoring by status of pressures, we need to identify the principle pressures that cause simultaneous biodiversity loss across taxonomic groups and clarify how these multiple pressures act synergistically, or at least simultaneously, to decrease biodiversity in an ecosystem.

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Agricultural ponds and small reservoirs support diverse populations of aquatic animal and plant species (Knutson et al., 2004; Oertli et al., 2005) and are the most important habitats in terms of both local and regional biodiversity among several aquatic habitat types (i.e., lakes, ponds, ditches, streams, and rivers) in agricultural landscapes (Williams et al., 2004). In such shallow water ecosystems, pollution by eutrophication can be the most severe (Scheffer and Carpenter, 2003). The build-up of phosphorous from agricultural fertilizers, sewage effluent, and urban stormwater runoff pushes freshwater bodies, such as lakes and agricultural ponds, into an algae-dominated (eutrophic) state in which the simultaneous decrease in biodiversity indices such as the abundance, richness, and/or functional diversity of a number of taxa occurs. This is mainly because, as the algae decay, oxygen levels in the water are depleted and there is widespread die-off of other aquatic life (Scheffer, 2004). In addition, especially in closed freshwater ecosystems, including lakes and ponds, invasive alien species, such as carnivorous fish, crayfish, and aquatic plants, have become another major threat to biodiversity (Revenga et al., 2005; Dudgeon et al., 2006). Therefore, to assess the biodiversity state of agricultural ponds with respect to pressures, it is essential to understand the synergetic, or at least simultaneous, effects of the multiple pressures quantitatively.

Here, we used a series of 64 ponds as a case study and developed a framework for an integrated biodiversity indicator that takes into consideration the multiple pressures leading to biodiversity loss. The indicator is defined as a function of the pressure(s) and is parameterized to explain a number of individual indicators of biodiversity, such as richness, abundance, and functional diversity of focal taxa. We selected aquatic macrophytes, Odonata, and benthic macroinvertebrates as the focal taxa, all of which are considered to be indicators of healthy biodiversity in freshwater ecosystems (e.g., Engelhardt and Ritchie, 2001; Heino, 2002; Heino et al., 2003; Declerck et al., 2005; Ilmonen and Paasivirta, 2005; Kadoya et al., 2009; Simaika and Samways, 2009). In particular, we selected Odonata as a single group from macroinvertebrates because among aquatic organisms, odonates have been widely proposed as indicators of the ecological quality of land–water ecotones, and aquatic habitat heterogeneity (e.g., Steyler and Samways, 1995; Clark and Samways, 1996; Chovanec and Waringer, 2001; Hawking and New, 2002; Schindler et al., 2003; D'Amico et al., 2004; Kadoya et al., 2004).

On the basis of comprehensive surveys of flora and fauna at the all 64 ponds studied, we calculated eight individual indicators: total richness of (1) aquatic plants, (2) adult Odonata and (3) benthic macroinvertebrates; number of endangered species of (4) aquatic plants and (5) Odonata; and functional diversity of (6) aquatic plants, (7) larval Odonata and (8) benthic macroinvertebrates. In addition, we focused on three types of pressure: eutrophication, habitat destruction, and invasive alien species, each of which includes several candidate factors describing the pressure state. We then evaluated the relationships among direct pressures and the individual biodiversity indicators and used a hierarchical Bayesian approach to calculate the integrated biodiversity indicator. In our model we could also quantify the relative importance of the contrasting multiple pressures to the biodiversity of agricultural ponds, allowing us to compare the magnitude of effects from the individual pressures.

2. Materials and methods

2.1. Study area

This study was conducted over an area of approximately 700 km^2 in southwestern Hyogo Prefecture, Japan (34°46′N, 134°56′E) where the predominant land uses are paddy fields (36.7%), broad-leaved forests (35.3%), and urban areas (15.3%). The study area has a warm temperate climate with a mean annual temperature of 14.4 °C (minimum, 3.5 °C in January; maximum, 26.4 °C in August) and mean annual precipitation of 1198.3 mm (data provided by the Miki Climatological Observatory located within the study area at 145 m above sea level [a.s.l.]). Before 1910, many irrigation ponds had been created in Hyogo Prefecture (8395 km^2) to provide water for paddy cultivation, and over 55,000 agricultural ponds, corresponding to about 20% of all ponds in Japan, were present by the 1950s. However, by 1997 over 11,000 ponds had been lost, mainly as a result of urban or residential development (Takamura, 2007). Even where ponds have not been destroyed, their biodiversity has drastically decreased during recent decades.

We chose nine pond types to study on the basis of dominant vegetation (emergent, floating-leaved, and no apparent vegetative cover) and dominant surrounding land use (predominantly urban, rural, or forest), with 6–9 replicates for each combination for a total of 64 ponds. The selected ponds had surface areas ranging from 751 to 114,339 m^2 (mean ± SD: 11,033 ± 14,829 m^2) and maximum depths ranging from 0.2 to 5.2 m (2.1 ± 1.3 m). The elevation of the ponds averaged 58.2 ± 33.6 m a.s.l.

2.2. Census of aquatic organisms

2.2.1. Aquatic macrophytes

Native aquatic macrophytes growing throughout each pond were extensively surveyed at the approximate peak of macrophyte biomass (August–September) in 2006 or 2007 by using an inflatable boat, wading, and using a rake. We recorded all native species of emergent, floating-leaved, free-floating, and submerged aquatic macrophytes encountered. Nomenclature and classification criteria followed the method of Kadono (1994). *Chara* sp., and *Nitella* sp. were identified to the genus level, but each genus was treated as one species.

2.2.2. Adult and larval odonates

We censused the number of adult odonate species by walking the shoreline of each pond between 9:00 and 15:00 on a fine day. We caught and identified odonate species when these were not distinguishable by sight. When we could not walk the shoreline, the route census was conducted by using an inflatable boat. The census was conducted six times (early spring, spring–early summer, summer, midsummer, early autumn, and late autumn) for each pond in 2007 or 2008, except for the late autumn census, which was conducted in the preceding year (i.e., 2006 or 2007). We recorded all individuals during each census at each pond.

The survey of larvae was conducted twice (early spring and late spring in 2007 or late autumn in 2007 and early summer in 2008) in each pond to cover the emergence of species with different seasonal life cycles. We set a quadrat (0.9 m × 0.9 m, 0.45 m height) for each 60 m of shoreline length in the littoral zone and swept a D-frame dipnet (2-mm mesh) through the entire surficial sediments, including decayed aquatic plants, within the quadrat. We sorted larval odonates >5 mm in length and identified them to species level under a binocular microscope. We recorded all individuals from each survey at each pond.

2.2.3. Benthic macroinvertebrates

The surveys were conducted once for each pond during 14–24 May 2007 or 12–17 May 2008, except for three ponds that had dried just before the survey periods; these three were surveyed later in the year (29 October 2007). We collected three samples in the center of each pond using Ekman–Birge-type bottom sampler (open mouth of 150 mm × 150 mm) and from three to nine samples near the shore according to the pond's size using the D-frame dipnet