



Plant and herbivorous insect diversity loss are greater than null model expectations due to land-use changes in agro-ecosystems



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ABSTRACT

The effect of land-use changes on plant diversity is an important issue, as plant diversity is vital to maintaining of threatened insect diversity in agro-ecosystems. Although many studies have examined patterns of a few threatened species, few studies have examined a large number of threatened species while comparing organisms of different trophic levels. Observed and expected community assembly patterns of common and threatened species should be evaluated along land-use gradients in agro-ecosystems, using species richness and red list indicator (RLI). This study examined local and regional diversity of plants and herbivorous insects among abandoned, traditional, and intensified land-use types in semi-natural grasslands and compared observed and expected values via null model analysis. This study showed that the richness and RLI of plants and herbivorous insects were maintained in the traditional land-use. However threatened plants and herbivorous insects at sites of land abandonment and intensified use were significant lower than null model expectations. Although, plant richness, rather than RLI, was responsible for maintaining richness and the RLIs of herbivorous insects, plant richness did not linearly correlate with plant RLIs. Threatened plants did not depend on plant richness, it will be vital to identify and conserve unique environments for diversity of plants and herbivores. In conclusion, we should reintroduce traditional land-use practices in intensified use and abandoned grasslands. National and local governmental support for the appropriate management of semi-natural grasslands is beneficial for biodiversity conservation at both the local (e.g., present study area) and national (Japanese) levels.

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1. Introduction

Biodiversity provides ecosystem stability and ecosystem services (Tilman et al., 2006; Cardinale et al., 2012). Historically, agricultural lands have maintained high biodiversity, especially in non-planting areas, such as around field margins and in mosaic landscapes (Tilman et al., 2001; Foley et al., 2005, 2011; Tscharntke et al., 2005; Uchida and Ushimaru, 2014), and agricultural lands occupies approximately 40% of the terrestrial area worldwide (Ramankutty and Foley, 1999; Foley et al., 2005). However, modern changes to agricultural practices have accelerated the loss of biodiversity through changes in the habitats of numerous species (Sala et al., 2000; McNeely et al., 2001; Sutcliffe et al., 2015). Consequently, biodiversity loss due to land-use changes has become a focal research subject in ecology (e.g., Benton et al., 2003; Butler et al., 2007; Newbold et al., 2015).

Traditional management practices (e.g., extensive farming) have been shown to result in high biodiversity (Tscharntke et al., 2005; Knop et al., 2006; Kleijn et al., 2011; Uchida and Ushimaru, 2014).

Although the advantages of traditional management practices in agro-ecosystems have been increasingly recognized (Pykälä, 2000; Tscharntke et al., 2005; Uchida and Ushimaru, 2014, 2015; Jakobsson et al., 2015), land-use changes, land abandonment and intensified use, accelerate the extinction of local biodiversity (Kleijn et al., 2009; Uchida and Ushimaru, 2014, 2015). Land abandonment promotes the succession of vegetation towards a secondary forest and thus causes declines in grassland species (Uematsu et al., 2010; Middleton, 2013; Uchida and Ushimaru, 2014, 2015; Nagata and Ushimaru, 2016). Intensified land-use has also been identified as a major driver of declines in biodiversity (Warren et al., 2001; Kruess and Tscharntke, 2002; Kleijn et al., 2009; Abadie et al., 2011; Uchida and Ushimaru, 2014; Shackelford et al., 2015). Thus, it is important to elucidate the unified effects of these two land-use practices on biodiversity, from small to regional scale patterns, as this would provide valuable information regarding biodiversity conservation (Huston, 1999; Newbold et al., 2015).

Declines in the diversity of producers (i.e., plants) are likely to affect primary consumers (e.g., herbivorous insects), as has been shown by studies demonstrating that plant diversity maintains arthropod diversity (Joern, 2005; Haddad et al., 2009; Pöyry et al., 2009; Scherber et al., 2010; Uchida and Ushimaru, 2014). As different types of land-use

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(land abandonment and intensified use) may cause changes in habitat quality, it is important to understand their unified effects on producers and primary consumers (Lambin and Meyfroidt, 2011; Uchida and Ushimaru, 2014). Although patterns of declines in the diversity of a small number or some of threatened species have been reported (e.g., Schtickzelle et al., 2005; Körösi et al., 2014), few studies have examined large numbers of threatened species and differences in patterns of diversity of producers and primary consumers (cf. Maes and van Dyke, 2005; Wassen et al., 2005; Bottrill et al., 2011; Monks and Burrows, 2014).

The null-model analysis were examined the evaluation of local community variances according to species pool on a regional scale (Chase and Myers, 2011; Kraft et al., 2011; Myers et al., 2013). The null-model approach examines communities on a species level across land-use gradients, including the amount of variation in the species pool, and correlates observed diversity patterns with predicted patterns based on spatial variation and the dispersal-assembly models (Hubbell, 2001; Chase and Myers, 2011; Myers et al., 2013). In addition, the Red List Index identifies conservation priorities by highlighting threatened species in regional areas (Butchart et al. 2007; Koyanagi and Furukawa, 2013), and can help elucidate observed patterns in biodiversity along land-use gradients. In this study, we combined the null-model analysis with calculated Red List Indicator (RLI: slightly modified from previously published studies) from all 47 Japanese prefectural red lists (i.e., prefectures are the first level of division of Japan), how land-use changes affect biodiversity. Using this analytical method, we hypothesized that the diversity of threatened plant and insect species will decline more than predicted by the null-model of community assembly, owing to the fact that threatened species are often replaced by species that can adapt to the new environments that result from land-use changes. Threatened/red-listed species are more susceptible to land-use changes, and will be less abundant than common species since they are often associated with unique environments (Wassen et al., 2005; Uematsu et al., 2010; Kleijn et al., 2011; Uchida and Ushimaru, 2014).

Japan is considered to be a global hotspot of biodiversity (Conservation International, 2015); however, recent land-use changes have rapidly driven loss of biodiversity in agro-ecosystems (Uchida and Ushimaru, 2014). Despite this alarming trend, and although studies in Europe have shown that 10–20% of red-listed taxa (e.g., plants and insects) are threatened by the intensified use of ecosystems (Norris, 2008; Fahrig et al., 2015), there have been few studies from East Asia regarding the shifting status of threatened species as a result of land-use changes.

In this study, we describe local to regional patterns of plant and herbivorous insect (butterflies and orthopterans) species richness and RLIs, resulting from land-use changes in agro-ecosystems. We aimed to answer the following three questions: 1) Are richness and RLIs of plants and herbivorous insects lower than expectations by null-model community assemblages as a result of land abandonment and intensified use? 2) Does richness explain RLI values for plants and herbivorous insects? 3) Do plant richness and RLI value increase the richness and the RLI value for herbivorous insects?

2. Materials and methods

2.1. Study area

The study was conducted at 31 agricultural sites (paddy terraces) in the south-eastern Hyogo Prefecture, Japan (ca. $19 \times 30 \text{ km}^2$), $34^\circ 48' - 57' \text{ N}$, $135^\circ 03' - 24' \text{ E}$ (Fig. S1). In the study area, semi-natural grasslands were maintained by periodic mowing on the levees of paddy field margins and irrigation ponds, and at the edges between paddy fields and secondary forests (Uchida and Ushimaru, 2014). Paddy terraces were categorized into three land-use types (Uchida and Ushimaru, 2014): 1) Abandoned terraces, where farmers had ceased rice cultivation and

the mowing of semi-natural grasslands 3–15 years ago (mean: 9.8 years). 2) Traditional terraces, which are paddy terraces that have been managed in traditional ways for at least 100 years. 3) Intensive terraces, characterized as land-consolidated paddy terraces, which underwent land consolidation 12–31 years ago (mean: 20 years). In most cases, the same farmer managed a single terrace. Interviews with all farmers indicated that they used little to no insecticide for paddy crops and did not apply insecticides to semi-natural grasslands.

The study area, study plots, anthropogenic disturbances, and survey methods for plants, butterflies, and orthopterans are described in Appendix S1 and in a previous study (Uchida and Ushimaru, 2014). Therefore, only a brief overview is provided below.

2.2. Study plots

We established four $5 \text{ m} \times 50 \text{ m}$ plots in semi-natural grasslands on paddy field margins at each site (i.e., 124 plots in 31 sites). The 31 sites were categorized into three land-use types (defined in Section 2.1.): abandoned sites, traditional sites, and intensive sites. A total of 8 abandoned sites, 13 traditional sites and 10 intensive sites were surveyed in 2011; while 7 abandoned sites, 12 traditional sites and 10 intensive sites were surveyed in 2012 (one abandoned and one traditional sites could not be surveyed owing to development in 2012). The number of mowing events in each plot during our survey period (April–October: plant growing season) was recorded to determine the disturbance frequency in 2011 and 2012. Mowing frequency was significantly higher in traditional sites (0–4 times per year, mean = 1.6) than in abandoned sites (0–1 times per year, mean = 0.13) and lower than intensive sites (2–6 times per year, mean = 4.1) (Uchida and Ushimaru, 2014). Distance between sites were not different among land-use types ($P > 0.1$, GLM with Wald test); the distance between abandoned and traditional site varied from 0.10 to 25.55 km (mean = 12.33 km), whereas intensive and traditional sites were separated by 0.11–33.57 km (mean = 14.18 km).

2.3. Plant survey

We established six $0.5 \text{ m} \times 0.5 \text{ m}$ box-plots at regular intervals along each plot at every study site. Each box-plot was further divided into four $0.25 \text{ m} \times 0.25 \text{ m}$ subplots (24 box-plots and 96 subplots per site). In all, we established 744 box-plots (2976 subplots) in 31 sites. We recorded all vascular plant species in each subplot in October 2011 and estimated the abundance of each plant species as the total number of subplots per site (0–96) in which the species was found. With these surveys, a total plant species list was prepared for each sites and used to calculate species richness and RLIs.

2.4. Herbivorous insect survey

The butterfly community in each study site was surveyed monthly from April to September in 2011 and 2012 (six times per year); the identity and abundance of butterfly species in each plot were recorded using the standardized transect count method (Pollard and Yates, 1994; Uchida and Ushimaru, 2014). The butterfly survey was conducted for 60 min per site (four $5 \text{ m} \times 50 \text{ m}$ plots) under sunny and warm conditions. We took great care to avoid double counting during the survey. The identity and abundance of orthopteran species were surveyed using a sweep net (42 cm diameter) with 800 sweeps per site. Orthopteran surveys were conducted twice (between mid-August and early October) for each plot in 2011 and 2012. We assessed a total of five families of butterfly and seven families of Orthoptera, which comprised the herbivorous insects in this study.

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