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Original Articles

Differences in land cover – biodiversity relationships complicate the assignment of conservation values in human-used landscapes

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ABSTRACT

The anthropogenic conversion of natural landscapes continues to be a severe threat to biodiversity. As humans depend on these natural landscapes for sustainable resource provisioning as well, the conversion likewise affects biodiversity and people's livelihoods. To identify options for reconciling human needs and biodiversity conservation we studied the relationship of land use and reptile diversity.

In a dryland in southwestern Madagascar, we compared reptile diversity between five land-cover classes and identified trait-dependent effects of land use on reptiles with regard to substrate preference and activity phase. Thresholds of diversity declines were detected at the transition from intact forest towards land-cover types with reduced forest cover, and at the transition to habitats with a woody plant cover below 10–30%. Community equitability increased towards lower vegetation cover. With increasing habitat openness, rare species were lost and a small subset of similarly successful degradation-tolerant species remained. Land cover – biodiversity relationships varied considerably between reptile assemblages that differed in activity phases. Nocturnal reptiles were more strongly affected by habitat conversion than diurnal reptiles. Substrate preference did not determine degradation tolerance.

These findings provide important implications for conservation planning in drylands. For many species converted habitats can still be suitable. However, ignoring the fine-scale habitat requirements of animals will likely lead to imprecise conclusions on the conservation value of converted drylands.

1. Introduction

Finding solutions to prevent the loss of biodiversity remains a major issue in the political arena and in academia (Pereira et al., 2013; Pimm et al., 2014). Regardless of the important efforts to increase the number of protected areas, they still become increasingly isolated islands in biodiversity poor landscapes (Joppa et al., 2008; Watson et al., 2016). In order to preserve ecosystems and maintain ecosystem services, options to reconcile biodiversity conservation and human land use are needed. Conservation and restoration programs are called for that target on increasing connectivity between suitable habitats and on maintaining high biodiversity within the human used landscapes (Daily, 2001; Ndriantsoa et al., 2017; Villard and Metzger, 2014). For this, it is fundamental to understand the relationship between land use and biodiversity.

Many facets of biodiversity (e.g. species, genetic, phylogenetic, functional and evolutionary diversity) are affected by land use and associated habitat conversions. Among these effects, it is assumed that land use and associated habitat conversions result in decreases in species and functional diversity in a wide range of taxa and regions (Clavel et al., 2011; Ekroos et al., 2010; Hanski, 2011; Hillers et al., 2008; Irwin et al., 2010; Riemann et al., 2015, 2017). Negative effects of land conversion into agricultural utilization are aggravated by actual or future changes in climatic conditions (Hannah et al., 2008; Sinervo et al., 2010).

In Madagascar, about 85% of the fast growing human population lives in rural areas, depending ultimately on agriculture, livestock, and natural resources provided by the land they live on (Cincotta et al., 2000; Scales, 2014). The dependence on natural resources is particularly high in Madagascar's dry Southwest (Andriamparany et al., 2015; Neudert et al., 2015; Ranaivoson et al., 2015), a global and regional center of endemism (Myers et al., 2000; Olson and Dinerstein, 1998),

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which was mainly covered by spiny and dry deciduous forest until a few decades ago. Forest cover has been reduced to a fraction of its original extent. The current land cover is the result of habitat structural impoverishment that followed human activities such as deforestation, dead wood extraction, fire, and overgrazing by livestock (Brinkmann et al., 2014; Ratovonamana, 2016). Therefore, the ecosystems of southwestern Madagascar can be a prime model for gaining knowledge on options for reconciling human land use and biodiversity conservation in drylands, where the loss of natural habitat is especially high (Watson et al., 2016).

Determining the conservation value of this transformed landscape is paramount for the identification of management options to sustain biodiversity in the human-used landscape. The reduction of diversity with increasing human disturbance has been observed in a wide range of taxa and regions (e.g. Ekroos et al., 2010; Flynn et al., 2009; Irwin et al., 2010). Because biodiversity declines are often non-linear and most dramatic after thresholds of original vegetation cover are surpassed (Andren, 1994; Folke et al., 2004; Lindenmayer and Luck, 2005; Scheffer et al., 2001; Scheffer and Carpenter, 2003), we concurrently aimed at determining the thresholds in vegetation cover, necessary for the persistence of high diversity. However, also differing reactions to anthropogenic disturbances have been described (Mackey and Currie, 2001), i.e., local diversity does not necessarily decline with increasing degradation. Differences in observed species diversity may result from different capacities of communities to cope with changing conditions. These capacities should be related to specific habitat requirements of single species.

The conditions in southwestern Madagascar allow us to assess conservation implications of land use on animal diversity in dry ecosystems. We investigated the relationship between land cover and diversity of reptiles, the dominant vertebrate group in the arid and semiarid ecosystems around the globe (Pianka, 1986). Therefore, we assessed differences in reptile diversity between land-cover types along a gradient of decreasing vegetation cover, from natural to structurally impoverished habitats. Potential trait-dependent differences in the land cover – biodiversity relationship (LCB) were recovered by grouping reptiles according to substrate preference (terrestrial/arboreal) and activity phase (diurnal/nocturnal); two aspects potentially determining extinction proneness due to human impacts (Bennie et al., 2014; Theisinger and Ratianarivo, 2015; Tingley et al., 2013).

We hypothesized that in our study system (1) thresholds of vegetation cover exist after which alpha diversity of reptiles declines rapidly; that (2) LCB relationships differ between four functional reptile groups according to habitat preference and activity mode, as a consequence of species' adaptations to specific niches (Lindenmayer et al., 2005); and that (3) not only land cover but also current land use affects functional groups differently.

2. Methods

2.1. Study site

The study took place in the Tsimanampesotse National Park (TNP) and its surroundings in southwestern Madagascar (Fig. A.1 in Supplementary Material). The region is one of the driest in Madagascar, with xerophilous forest as the dominant vegetation. Annual precipitation varies between 150 and 750 mm, but rarely exceeds 400 mm (Andriatsimietry et al., 2009; Ratovonamana et al., 2011). Rain usually falls between December and April.

Natural habitats outside the protected area are predominantly affected by slash and burn agriculture, livestock grazing, charcoal production, and firewood collection. Habitat conversion is most pronounced in cultivated land near settlements, whereas protected forests were least affected. Even though officially prohibited, livestock grazing and browsing occurs regularly in protected forest (Feldt and Schlecht, 2016; Ratovonamana et al., 2013).

2.2. Data acquisition

We used 121 line transects (length = 100 m; width = 3 m) spread over an area of approximately 500 km^2 in and around TNP, covering a wide range of different land cover types that were affected by various human land-use actions (Fig. A.1 and Table B.1 in Supplementary Material). Because a final land cover classification had not yet been established in 2012, we referred to a preliminary land cover classification which was provided by our Malagasy collaborator (Y. R. Ratovonamana, pers. comm.) as well as information from the local population for transect placement. One subset of transects was established along a track that traverses TNP from East to West, another part was established within a well-protected zone of TNP and yet another part of the transects was established in the human dominated landscapes East and West of TNP. In order to be able to cover a high number of transects within a short span of time, we accounted for accessibility, spreading our survey effort over the whole region. Transect locations were chosen arbitrarily. Transects were sampled during the rainy season from the end of January to the beginning of April 2012 at least twice during the day (max. 3 times) and twice at night (max. 3 times) resulting in 592 transect walks. On all transects standardized visual encounter surveys were conducted searching for squamate reptiles (lizards and snakes). Transects were walked at a constant pace of 8 m/min, with stops to look into crevices and turn over logs and stones. We observed individuals only up to a distance of 1.5 m to either side of the transect line, which was readily observable in each habitat type. This way, we ensured that detection probability of individuals of a species was similar across habitat types. All species were determined on site according to Glaw and Vences (2007). Taxonomy follows Uetz et al. (2016) with one exception. We encountered a species, clearly belonging to the genus *Blaesodactylus*, that was morphologically distinct from any species described from this ecoregion. Herein, we refer to this morphospecies as *Blaesodactylus* sp.

With our survey method we did not sample the fossorial reptile fauna (e.g. Nagy et al., 2015). We excluded the two tortoise species, *Astrochelys radiata* and *Pyxis arachnoides*, because they are illegally extracted (Ganzhorn et al., 2014; Walker et al., 2013), which likely masks land-cover effects on their occurrence. For our analyses we used species richness and relative abundance on each transect. The latter was defined as the maximum abundance of a species on a transect recorded during one of the transect walks.

To identify the prevailing land-cover type for each transect, we used a land-cover map of 2013 (30 m resolution) by Brinkmann et al. (2014). The map was derived from supervised classification of remote sensing data. Land-cover classes were characterized following a standard approach based on vegetation structure, particularly vegetation cover and height of vegetation (White, 1983; Brinkmann et al., 2014). Brinkmann et al. (2014) provide an accuracy assessment of the remote sensing data that was used to define the distribution of the land cover classes. The original land cover classes were simplified to improve accuracy (Table 1). We corrected identified misclassification errors based on own field observations and current Google Earth images (© 2015 Google Inc.) taken during the rainy season of 2013.

2.3. Calculation of LCB relationship

To compare differences in diversity between land-cover types, we calculated diversity profiles based on Hill numbers (Chao and Jost, 2015; Hill, 1973) of order q = 0 to q = 3 using data on the relative abundance of species per transect. Hill numbers are a family of diversity measures. They are more intuitive for interpretation than many diversity measures, because they follow the doubling property (i.e., the sum of Hill numbers of two completely distinct assemblages with identical abundance distributions equals the Hill number of the combined assemblages; Chao et al., 2014). Derived diversity profiles provide information on some diversity indices of which Hill numbers are