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Research Article

One Forest Is Not Like Another: The Contribution of Community-Based Natural Resource Management to Reptile Conservation in Madagascar

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Abstract

Agricultural expansions and unsustainable resource extraction increasingly threaten biodiversity worldwide. Thus, knowledge on landscape management opportunities that contribute to biodiversity conservation and provide the necessary natural resources is urgently needed. Community-based natural resource management can be one solution to achieve sustainable resource provisioning and biodiversity conservation. However, evidence on the effectiveness of the management transfers to local communities for biodiversity conservation is scarce. We use a case study from south-western Madagascar to discuss the integration of community-managed forests, including forests protected by local taboos, into measures for biodiversity conservation. In particular, we compared reptile community composition and species diversity between differently managed forest sectors (used and less-used) in two community forests. We hypothesized that forest use affects biodiversity depending on the people's acceptance of management regulations. Our results show that reptile community composition was affected by forest management if the less used forest part was protected by traditional taboos. If the people did not respect regulations for resource extraction, habitat protection failed which led to a homogenization of reptile community composition across different forest parts. This highlights that the type of forest management and use and its acceptance in the rural communities determines the contribution of community forests to biodiversity conservation.

Keywords

spiny forest, production landscapes, land use, sacred forests, reptiles, dryland

Introduction

In an anthropogenic world, effective biodiversity conservation requires a human dimension for the management of ecosystems (Aymoz, Randrianjafy, Randrianjafy, & Khasa, 2013; Ferguson et al., 2014; Mangel et al., 1996; McShane et al., 2011; Sunderland, Ehringhaus, & Campbell, 2008). Agricultural expansions and resource extraction outside protected areas (PAs) are increasingly threatening biodiversity (Laurance et al., 2012). Therefore, conservation strategies should be part of management plans for production landscapes, where resource use is of paramount importance for local livelihoods. This involves trade-offs between conservation and human well-being (Gardner et al., 2013).

Fundamental information on the value of managed forests for reconciling human needs and biodiversity conservation is still lacking (Geldmann et al., 2013; Seppelt et al., 2016; Tscharntke, Klein, Kruess, Steffan-Dewenter, & Thies, 2005). In light of dramatic diversity declines and a PA network that will not be sufficient to reach global biodiversity targets, however, knowledge on

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landscape management opportunities that contribute to biodiversity conservation and provide the necessary natural resources is urgently needed.

Globally, the idea of transferring the management of resources to local communities in order to achieve sustainable resource use emerged in the 1970s (Food and Agricultural Organization, 1978). In Madagascar, community-based natural resource management has been promoted as one possibility to contribute to sustainable resource provisioning and to biodiversity conservation. The formerly centralized management of forests was shifted toward a more decentralized approach, in order to reach sustainable use of resources and counteract high rates of ecosystem degradation. This approach started to be implemented in the early 1990s (Raik, 2007).

However, apart from challenges in reaching more social justice (Dressler et al., 2010) and a call for real community participation (e.g., Gardner et al., 2013), not much evidence on the effectiveness of the management transfer for biodiversity conservation and sustainable resource provisioning has been gathered so far (Danielsen et al., 2003; Newmark & Hough, 2000). Even though the performance of decentralized conservation approaches is unknown, community management of natural resources is still advocated as an important component to counter ecosystem degradation and reach sustainable resource use, while ensuring human well-being. Newly established PAs in Madagascar directly build on the ideals of community-based natural resource management (Gardner et al., 2013).

Recent studies indicate that community forest management does not always reduce deforestation (Rasolofoson, Ferraro, Jenkins, & Jones, 2015), and thus cast doubt that the coupling of resource exploitation and biodiversity conservation can be sustainable (e.g., van Gils, Piersma, Dekinga, Spaans, & Kraan, 2006; Gardner, Jasper, Eonintsoa, Duchene, & Davies, 2016; see also recent debates on land sharing vs. sparing, reviewed in Fischer et al., 2014). However, forests whose management was transferred to communities are among the last remaining forests in Madagascar and thus might be an integral part of landscape-moderated biodiversity protection.

We use a case study from south-western Madagascar to discuss the integration of community managed forests, including forests protected by local taboos into measures for biodiversity conservation. In particular, we compared reptile community composition and species diversity between differently managed forests in two community forests. Reptiles are the dominant vertebrates in this region and respond to changes in structural properties of the habitat (e.g., Gardner et al., 2016; Nopper, Lauströer, Rödel, & Ganzhorn, 2017). In one community forest, the highly accepted local management of a part of the forest is facilitated by local taboos. In the other community forest, acceptance of novel forest management regulations by the local population was low. We hypothesized that forest management affects reptile communities depending on management acceptance.

Methods

Study Area

The study region in south-western Madagascar (Figure 1) consists of an IUCN category II PA, the Tsimanampesotse National Park, and its surrounding human-used landscape. The area is characterized by a high degree of floral endemism (Phillipson, 1996). The climate is subarid with largely unpredictable and low precipitation (<400 mm/year; Dewar & Richard, 2007; Ratovonamana, Rajeriarison, Roger, & Ganzhorn, 2011). Resource use within the PA is illegal, except for the extraction of reeds in a sustainable fashion for local house construction.

Areas surrounding the PA are exposed to human land use. Here, natural resource management has been mostly transferred to associations consisting of members from the local communities (so-called COmmunauté de BAse or COBA). The study was conducted in the community forests Tsimandikalilindraza and Mandrosoa west of Tsimanamepesotse National Park (Figure 1). Although both community forests were situated in the vicinity of the national park, we abstained from using the protected forest as a control site, because the national park in this area was established only in 2009 in the course of a national park extension and we lacked information on former forest use.

Both community forests were divided into sectors of different usage. Resource use in the utilization sectors included the extraction of construction wood for a fee.

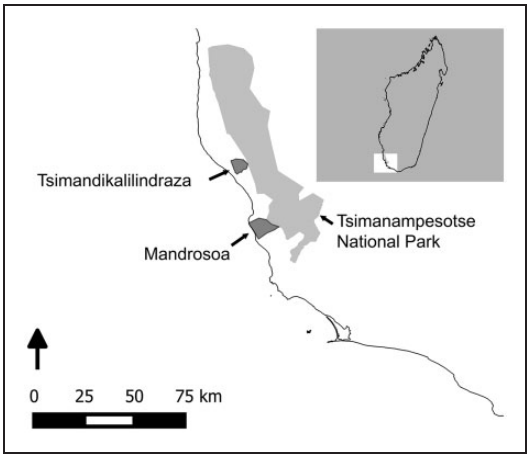


Figure 1. Study area in south-western Madagascar. The two surveyed regions in which community-based natural resource management takes place are highlighted in dark gray. They are situated adjacent to the Tsimanampesotse National Park (light gray).

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Additionally, the extraction of dead wood for cooking and the collection of plants for food and medicinal purposes were allowed. Within the community forest of Mandrosoa, one part of the forest had been protected by a ''fady'' (taboo) for at least the last 50 years (interviews with villagers). We hereafter refer to this forest sector as sacred forest. The only extractive activities were reported to be the extraction of certain tree species in marginal numbers for traditional purposes (for a ritual pole, the ''hazomanga''), as well as the collection of medicinal plants. All other extractive activities were strictly sanctioned according to traditional rules and norms. In the community forest Tsimandikalilindraza, all the forest is in use, but forest use should officially follow a rotation with a fallow cycle of a predefined forest part. According to the representatives of the COBA and our own observations, acceptance of forest protection by the community, however, was low. To control for this uncertainty in the evaluation of utilization, we conducted an objective evaluation of the forest state using structural vegetation characteristics (see below). In both regions, forest sectors of different utilization were directly adjacent to one another.

Vegetation Structure

Forest use can significantly affect structural properties of forests. Those are in turn important determinants of reptile community composition and species richness (Nopper et al., 2017). To relate possible changes in reptile assemblages to habitat structural properties, we assessed eight structural habitat parameters that have been shown to affect reptile occurrence (e.g., Catterall et al., 2004; Nopper et al., 2017) and are coupled with human land use: number of trees with a diameter at breast height above 15 cm (1) as well as between 2.5 and 15 cm (2), both used for construction purposes (predominantly houses and carts); vegetation cover in the herb (up to 30 cm above ground; 3), shrub (30 cm up to 2 m; 4), and tree stratum (above 2 m; 5); leaf litter cover (6); amount of dead wood, as the number of branches and tree stems with either a diameter above 2.5 cm up to 15 cm (7), or a diameter above 15 cm (8).

For parameters 1, 2, 7 and 8 we counted the number of stems along each 100 m transect (with 1.5 m on each side of the transect line) and used the per-transect-counts for subsequent analyses. For 3, 4, 5, and 6, we estimated cover in 5% intervals for each 10 m subsection of the 100 m line transect (again with 1.5 m on each side of the transect line) and determined median cover estimates for each transect.

Reptile Assemblages

In each of the two regions (Mandrosoa and Tsimandikalilindraza), we compared reptile diversity among forest use sectors. We installed ten 100 m line transects per forest category (used and less-used). Transects ran parallel and had a distance of at least 30 m to one another. They were each surveyed ten times in Tsimandikalilindraza between 8 and 24 April 2015 and six times in Mandrosoa between 17 July and 6 August 2015 (see Figures A1 and A2 for species accumulation curves). Surveys were conducted by a team of three persons (AR and two para-ecologists) between 08:00 and 12:00 h. Transects were surveyed alternately among differently used forest sectors (used and less-used), and only once a day. The survey team observed reptiles by walking on the transect line at a constant speed of about 8 m/min and documented the presence of all reptiles within a distance of 1.5 m on each side of the transect line. During transect walks, the team also searched for reptiles under loose bark, in tree holes, and under stones.

The reptile fauna of the region consists of approximately 50 mostly endemic species (Raselimanana, 2008), of which many are rare (Nopper et al., 2017) or restricted to certain habitat types (Glaw & Vences, 2007). The survey method used in this study is representative for reptiles occurring in dry forest on sandy and ferruginous soil (compare Ratovonamana et al., 2011 for vegetation classifications). These are diurnal species as well as those nocturnal species that use dead wood, tree holes, and stones as diurnal retreat sites. Our approach excludes the species that are adapted to the adjacent limestone habitat or exclusively retreat to burrows underground during daytime. Our survey method did not allow for the detection of burrowing and some of the nocturnal reptile species. All species encountered by a single individual only were excluded from analyses, because of a possible undersampling bias (e.g., Coddington, Agnarsson, Miller, Kuntner, & Hormiga, 2009): In Mandrosoa, these were *Furcifer verrucosus* and *Liphidium* aff. *trilineatum*; in Tsimandikalilindraza, these were *Trachylepis gravenhorstii* and *Madascincus igneocaudatus*.

Analyses

We determined if forest use (according to the a priori defined management categories) had led to different structural characteristics of the habitat, irrespective of high or low acceptance of management. For this, we used random forest classifications implemented in package *randomForest* (Liaw & Wiener, 2002) in *R* (R Core Team, 2014). Random forest is a machine learning algorithm that here classifies samples of transects according to habitat characteristics into the predefined forest categories in multiple iterations. We drew 10,000 bootstrap samples with three variables randomly selected at each node. After the computation of a confusion matrix which summarizes the results of the classifications, the variables' importance for correct classifications

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was returned. The results of the random forest classification were confirmed by Wilcoxon tests comparing habitat characteristics between used and less-used forests.

For each transect, we determined the relative abundance of each reptile species by taking the maximum number of individuals found during a single transect walk of the respective transect. We determined differences in alpha diversity among habitat types by calculating abundance-based diversity profiles with Hill numbers using functions from Chao and Jost (2015, Appendix S8). Diversity profiles based on Hill numbers do not only provide information on species richness but also take into account species abundances to varying degrees. With increasing order q, the weight of dominant species increases in the calculation of species diversity. While Hill numbers of order $q = 0$ are the number of observed species (irrespective of their abundance), Hill numbers of order $q = 1$ can be roughly termed the ''number of typical species'' and of order $q = 2$ ''the number of very abundant species'' (after Gotelli & Chao, 2013).

Differences in community composition between differently used forest sectors were determined by permutational multivariate analyses of variance (perMANOVA) implemented in function *adonis( )* in R package *vegan* (Oksanen et al., 2015). For graphical representation of group differences, we performed nonmetric multidimensional scaling with function *metaMDS( )* in R package *vegan* (Oksanen et al., 2015), using default options. For each of the latter two analyses, we built community dissimilarity matrices by calculating Bray-Curtis dissimilarities of relative abundance data.

Table 1. Differences in Habitat Structure Between Forest Parts in the Community Forest of Mandrosoa (With Sacred Forest): Variables That Contributed to the Classification According to *randomForest* Classification (i.e., Variable Importance E) Are Listed in Decreasing Order.

Character	Used forest	Sacred forest	E	W
Veg.cover >2 m	40% (30–60%)	64% (58–85%)	3.78	2.5***
Trees >15 cm	10 (2–28)	26 (17–42)	2.63	10.0**
Leaf litter cover	63% (55–70%)	70% (60–78%)	0.76	19.5*
Dead wood >15 cm	2 (0–4)	3 (0–11)	0.53	38.5
Trees >2.5 cm	69 (59–97)	74 (61–102)	0.52	39.0
Veg-cover <0.3 m	3% (0–10%)	4% (2–5%)	0.49	56.5
Veg cover <2 m	24% (15–30%)	18% (10–30%)	0.44	70.0
Dead wood >2.5 cm	15 (0–25)	20 (6–40)	0.36	33.5

Note. Results of Wilcoxon test (W; stars denote test significance) are provided. Given are respective median values and range of structural variables (in parentheses).
*$p < 0.05$; **$p < 0.01$; ***$p < 0.001$.

Table 2. Differences in Habitat Structure Between Forest Parts in the Community-Managed Forest Tsimandikalilindraza (Management Rules Not Respected).

Character	Used forest	Fallow forest	E	W
Veg.cover >2 m	43% (35–58%)	50% (35–60%)	0.52	67
Trees >15 cm	11.5 (8–18)	21 (17–39)	3.95	98***
Leaf litter cover	45% (15–65%)	55% (45–78%)	0.62	8
Dead wood >15 cm	4 (1–10)	9 (3–13)	1.63	87**
Trees >2.5 cm	25 (12–48)	33 (15–65)	0.93	68
Veg-cover <0.3 m	9% (5–23%)	5% (3–10%)	0.85	16**
Veg cover <2 m	25% (30–40%)	25% (20–40%)	0.62	31
Dead wood >2.5 cm	12 (8–19)	16.5 (9–23)	0.39	74

Note. Variable importance (E) according to *randomForest* classification as well as results of Wilcoxon tests (W; stars denote test significance) are provided. Given are respective median values and range of structural variables (in parentheses).
**$p < 0.01$; ***$p < 0.001$.

Results

Vegetation Structure

The majority of transects could be readily assigned to a respective management class in both community forests irrespective if management acceptance was low or high (Tables A1 and A2). The most important variables differing among used and less-used forests were the number of large trees (Mandrosoa and Tsimandikalilindraza), the abundance of large pieces of dead wood (Tsimandikalilindraza only) and canopy cover (Mandrosoa only; see Tables 1 and 2). Canopy cover in used and less-used forests in Tsimandikalilindraza was comparable to that of used forest in Mandrosoa. However, in Mandrosoa, the canopy cover in the used forest part was lower than in the sacred forest part (Table 1).

Reptile Assemblages

In total, we documented the presence of 12 reptile species in the community forests of Mandrosoa and 15 species in Tsimandikalilindraza (Table 3). In Mandrosoa (sacred forest vs. used forest), diversity profiles show that species richness was lower and community equitability was higher in sacred than used forest. Diversity profiles overlapped, indicating that differences in alpha diversity were not pronounced, namely the number of typical species (higher order q) did not differ between management categories (Figure 2(a)). However, we detected significant differences in assemblage compositions between used and sacred forest (perMANOVA: $F = 7.66$; $R^2 = 0.30$; $p < .001$; Figure 3(a)).

In Tsimandikalilindraza (management rules not respected), we neither detected any differences in

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alpha diversity or community equitability (Figure 2(b)) nor any changes in community composition (perMANOVA: $F = 0.72$; $R^2 = 0.04$; $p = 0.64$; Figure 3(b)) between the different forest parts, namely used and least-used forest.

Table 3. Relative Abundance of Reptiles in Two Community-Managed Forests, Separately for Used and Sacred/Less-Used Forest Parts.

	Mandrosoa		Tsimandikalilindraza		
Species	Used	Sacred	Used	Less-used	Threat status
Astrochelys radiata	12	20	5	7	CE
Chalarodon madagascariensis	7	–	95	80	LC
Dromicodryas bernieri	–	–	–	2	LC
Geckolepis typica	20	9	–	2	LC
Furcifer major	–	–	1	1	LC
Furcifer verrucosus	–	–	9	7	LC
Hemidactylus mercatorius	–	6	7	6	LC
Lygodactylus aff. *tuberosus*	15	13	4	8	LC
Mimophis mahfalensis	6	7	7	5	LC
Oplurus cyclurus	5	2	8	4	LC
Paroedura picta	–	–	2	1	LC
Phelsuma breviceps	4	–	–	–	VU
Phelsuma mutabilis			11	11	LC
Pyxis arachnoides	1	1	–	–	CE
Tracheloptychus madagascariensis	44	15	49	38	LC
Trachylepis aureopunctata	1	–	12	19	LC
Trachylepis elegans	6	–	31	30	LC

Note. Species' threat status according to the IUCN red list (International Union for Conservation of Nature, 2016) is provided.

Discussion

We documented differences in reptile community composition between used and less-used forest parts, when the less-used forest part was protected by locally respected traditional taboos. If management regulations received low acceptance, this resulted in a homogenization of reptile community composition between a presumably protected (less-used) forest part and a forest part that was used for subsistence purposes. Even though, relative differences in vegetation characteristics existed between less-used and used forest parts in both cases, absolute differences in vegetation characteristics between community forests likely influenced the relationship between forest use and reptile diversity.

If plant cover does not fall below a threshold vegetation cover of 10%–30%, species richness has been reported to remain high in an ecosystem (Andrén, 1994; Lindenmayer & Luck, 2005; Nopper et al., 2017). These areas of higher vegetation cover and species richness can thereby contribute to biodiversity conservation in the human-used landscape. In addition to this evidence, we show that the contribution of forests to biodiversity conservation depends on the type of forest use (and management acceptance). If protection status was respected, differences in vegetation structures correlated with differences in reptile community composition. In contrast, in the community forest in which forest management received low acceptance, reptile assemblages did not differ between used and less-used forest parts. Here, relative differences in vegetation structure between forest parts did not result in differences of reptile communities, that is, if management was not respected, forest use led to a homogenization of the reptile assemblages. This is likely the effect of an absolute change of specific vegetation characteristics that equally affected reptile community

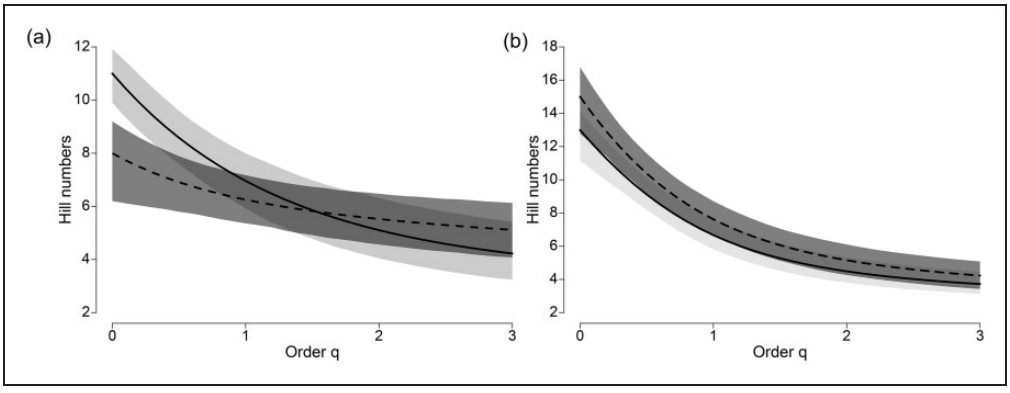


Figure 2. Diversity profiles for reptiles based on Hill numbers in used (continuous line; 95% confidence interval (CI) in light gray) and less-used forest parts (dashed line; 95% CI in dark gray) in the community-managed forests Mandrosoa (a) and Tsimandikalilindraza (b). While for Hill numbers of order $q = 0$ species are equally weighed (irrespective of their abundance), diversity is increasingly affected by dominant species with increasing order q.

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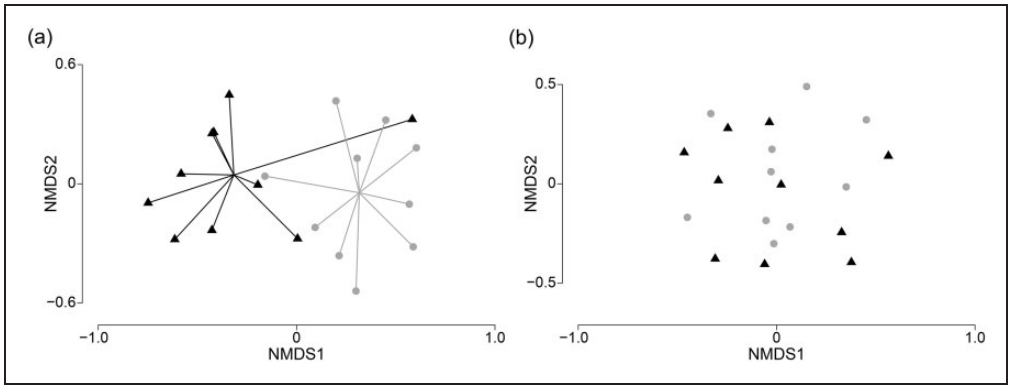


Figure 3. Nonmetric multidimensional scaling of reptile community composition. Displayed are differences in community composition between transects in used forest (triangles) and less-used forest parts (circles) in two community managed forests: (a) Mandrosoa, sacred forest as less-used site (Stress = 0.14); (b) Tsimandikalilindraza, management rules not respected by local population (Stress = 0.20).

composition in less-used and used forest parts. If management regulations received low acceptance, canopy cover and density of small trees was in median lower than in a forest part that was protected by locally accepted traditional taboos. Forest protection and with it the conservation of specific vegetation characteristics that influence reptile community composition was only effective, if regulations of forest protection were respected. Thus, if regulations for forest protection were not respected, the contribution of presumably protected forest parts to reptile conservation seemed to be comparable to that of forest parts that are used for subsistence purposes.

In our study area, the sacred forest harbored slightly less reptile species than the more intensively used forest part. This indicates a positive impact of forest use on reptile species richness. Indeed, it seems that used forests that provide the human population with ecosystem goods can be suitable habitat for many species (e.g., Gardner, 2009; Nopper et al., 2017; Riemann, Ndriantsoa, Raminosoa, Rödel, & Glos, 2015). Intermediate levels of disturbance that do not remove forests but open up forest to cover values just above 30% could thus favor reptile diversity (compare Gardner, 2009; Nopper et al., 2017). Nonetheless, the sacred forest provides habitat for a differently assembled reptile community. The combined biodiversity that both differently used forest parts provide is higher than that of each of the forest parts taken for itself. Moreover, sacred forest should also be preserved because reactions of other taxa might differ from those of reptiles and intermediate disturbance might not always lead to increased species richness (Mackey & Currie, 2001). Sacred forests have been often documented to contribute to the protection of biodiversity in human-used landscapes (e.g., Colding & Folke, 2001; Decher & Bahian, 1999; Ormsby & Bhagwat, 2010; Tengö et al., 2007).

Many rarer species might have already vanished in all surveyed forests. Forest dependent species are lost from intact to altered forests, a seemingly general phenomenon across many taxa and regions (Barlow et al., 2007; Gardner et al., 2016; Gibson et al., 2011). Used forests might therefore not possess the same conservation value as protected forests (Wright, 2010). Hence, for biodiversity conservation, any community-managed forests almost certainly cannot be a substitute to strictly PAs that prohibit human land use and have the principal goal of biodiversity conservation. This however does not make community forests less attractive for conservation incentives. They can be important complements to biodiversity conservation, increasing biodiversity in the human-used landscape for the sake of resource conservation. These ''novel ecosystems'' can provide important ecosystem services and are thus important in production landscapes, although they may provide only habitat for ''opportunistic biodiversity'' (Kueffer & Kaiser-Bunbury, 2014; Mascaro, Hughes, & Schnitzer, 2012).

Evidently, human interventions (like resource extraction) affect species in an ecosystem differently and consequently effects on animal communities through community restructuring are possible. For sustainable resource production, it is essential that the production system continuously (in the long term) provides the necessary resources. This is facilitated by high levels of original biodiversity (Hooper et al., 2005). In the study system, the role of reptiles for seed dispersal, pollination, pest control, or nutrient cycling has not yet been evaluated and also for other animal taxa, empirical evidence is scarce (e.g., Dausmann, Glos, Linsenmair, & Ganzhorn, 2008; Ganzhorn, Fietz, Rakotovao, Schwab, & Zinner, 1999; Martin, Ratsimisetra, Laloe, & Carrière, 2009). Thus, it is unknown in how far changes in the reptile community affect ecosystem functioning. However,

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theoretically negative effects of community restructuring on ecosystem functioning are possible, and as long as consequences remain unknown, dynamics of resource availability should be tightly monitored. Because consequences of community restructuring on ecosystem processes may not necessarily be visible in the short term, long-term monitoring programs should be installed. According to local communities, monitoring programs already exist at least in south-western Madagascar. They potentially provide valuable information (Danielsen et al., 2014). However, tools to evaluate current impacts of land use on ecosystems have not yet been fully implemented.

Implications for Conservation

For successful biodiversity conservation in the human-used landscape, a management on different scales is necessary, integrating local scale results into approaches on larger scales (Cunningham et al., 2013). Even though some land uses might have negative effects on the local occurrence of some species, diversity on the larger landscape scale may not suffer if land use is diversified (Gardner, Barlow, Sodhi, & Peres, 2010; Gonthier et al., 2014). The studied community forests in south-western Madagascar provided habitat for some threatened taxa (*Astrochelys radiata*, *Pyxis arachnoides*, and *Phelsuma brevipes*). Thus, community forests may play vital roles in a network of suitable habitats that contribute to a higher biodiversity within the human-used landscape (compare Hanski, 2011; Hobbs et al., 2014; Leménager, King, Elliot, Gibbons, & King, 2014; Tscharntke et al., 2005). The management of multifunctional landscapes with many ecosystems, integrating strictly PAs and areas of human land use, can allow for conservation and resource provisioning (Cunningham et al., 2013; Hobbs et al., 2014).

Up to now, the success of the transfer of forest management has been evaluated largely from a social perspective and was reported to have failed in the attempt to implement a participative community management that ensures long-term sustenance of forest resources (Ferguson et al., 2014; Pollini, Hockley, Muttenzer, & Ramamonjisoa, 2014). Certainly prevention of deforestation is most important (Gardner et al., 2016; Gibson et al., 2011; Nopper et al., 2017); however, the evidence base for the contribution of remaining used forests to sustainable resource production and especially biodiversity conservation remains scarce (Gardner et al., 2013). Our results show that forest management is reflected in changes of reptile community composition. The next step should be to understand how this relates to ecosystem functioning and sustainable resource provisioning to inform conservation and resource management.

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Supplementary material

Supplementary material for this paper can be found at the journal website: http://journals.sagepub.com/home/trc

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