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Reconciling conservation and development in Madagascar's rapidly-expanding protected area system

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Abstract

The creation and management of protected areas is our principal approach to conserving biodiversity worldwide. Management and governance models for these diverse institutions have become more pluralistic in recent decades, moving away from the traditional exclusionary protected area model that has proliferated historically. Indeed, most new protected areas are being established for 'multiple-use' and, therefore, permit a range of human livelihood activities to occur within their boundaries. However, we know little about how such sites can be effectively managed.

In this thesis, I use an interdisciplinary mixed-methods approach to investigate the implementation of new multiple-use protected areas in Madagascar. Madagascar is a global conservation priority characterised by high levels of endemism, and has a largely forest-dependent biota. Since most of the human population is rural and dependent on natural resources for subsistence and income to differing extents, the expanded protected area system is managed for both conservation and socioeconomic goals (poverty alleviation and development). However, these objectives may be conflicting since human resource use can be a significant driver of biodiversity loss. I begin by examining trends in new protected area establishment at the nationwide-level to generate insights into protected area categorisation, and the role of natural resources and protected areas in poverty alleviation. I then consider the impacts of forest use on biodiversity, through a literature review and empirical study of bird and reptile communities across a degradation gradient. The findings indicate that habitat change arising from forest use may impact the high-value, endemic component of the fauna most negatively. In addition, I develop a simple index to enumerate the conservation value of different species. This is then used to determine how degradation influences the conservation value of exploited habitats, as well as assessing if the index is a suitable tool that can be used to prioritise conservation investment across a portfolio of sites. Finally, I seek to understand the drivers of natural resource use by rural communities within the Ranobe PK32 protected area, and discover that both bushmeat hunting and charcoal production are fallback activities or supplements to other livelihoods.

The evidence collated in the thesis, derived from both ecological and social perspectives, suggests that managing new protected areas in Madagascar for conservation and development is overambitious, and that, at least in forest areas, management cannot be optimised towards both goals simultaneously.

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All chapters in this thesis were written by Charlie Gardner and are formatted in the style of manuscripts for publication. The publication details of published chapters are provided at the start of each chapter. Comments and editorial suggestions were made by Z.G. Davies (all chapters) and D.C. MacMillan (Chapters 3 and 4). Further contributions made by others are detailed on the cover pages or acknowledgements of each chapter.

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Introduction

1.1 Research preface

Mankind dominates Earth's ecosystems to such an extent that the impacts of human agency are the defining characteristic of our geological era, the Anthropocene (Crutzen and Steffen 2003). The repercussions of human activities now threaten most of the Earth's species and ecosystems (Ehrlich and Pringle 2008), and have precipitated the planet's sixth mass extinction (Chapin *et al*. 2000), with current extinction rates significantly higher than would be expected from the fossil record (Barnosky *et al*. 2011). The four principal drivers of biodiversity loss are overexploitation, introduced species, habitat destruction and co-extinctions (Diamond 1984), of which habitat destruction has been identified as the most important (Pimm and Raven 2000). Anthropogenic climate change now adds a fifth dimension to the extinction matrix (Thomas *et al*. 2004).

The continued erosion of species and ecosystems is of concern to mankind because biodiversity underpins the provision of critical ecosystem services and generates other values (MEA 2005). Consequently, most nations have committed to the conservation and sustainable use of biodiversity through the Convention on Biological Diversity (CBD), a multilateral treaty. In practice, a diverse suite of actions have been implemented to arrest threats and reverse their impacts on species, communities and ecosystems. These include both *ex situ* and *in situ* approaches. The former concerns the conservation of species away from their natural habitat, such as in zoological gardens, aquaria and seed banks (Fa *et al*. 2011), while the latter aims to conserve species, communities and ecosystems where they naturally occur. Globally, the principal tool for *in situ* conservation is the establishment and management of protected areas, a catch-all term which includes a vast array of different institutions with varied aims, rules and management approaches. All, however, have the principal objective of conserving biodiversity (Dudley 2008). Protected areas are the cornerstone of conservation to such an extent that they represent the world's largest ever planned land use (Chape *et al*. 2005), covering 12.9 % of the world's land surface in 2009 (Bertzky *et al*. 2012; Jenkins and Joppa 2009). Moreover, signatories to the CBD are expected to extend protected area coverage further still, to 17 % of land area by 2020 (CBD 2010), as well as ensure that they are effectively managed.

The overall objective of this thesis is to use Madagascar, a country that is rapidly expanding its protected area system in line with its CBD obligations, as a case study to examine and explore the establishment and management of new protected area models. During my research, I used an interdisciplinary approach to investigate the resource use of rural people and the impacts this has had on biodiversity in and around a single protected area, in addition to examining management trends at a system-wide level. The work is very applied in nature, thus making a contribution to on-the-ground practice and the development of protected area theory, policy and research agendas.

1.2 Conservation science: an interdisciplinary field

Conservation biology emerged as an academic discipline in the 1980s, with the aim of advancing the scientific foundations of biodiversity conservation (Meine 2010). Adopting the normative position that biodiversity is good and should be preserved (Noss 1999), it is a value-laden (Noss 2007), crisis-oriented (Soulé 1985) and problem-solving field, defined more by its goal as by the disciplines or subdisciplines of academia that it spans (Ehrenfeld 1992). Since the field is pragmatic, concerned with on-the-ground outcomes as well as traditional academic outputs and impacts, many conservation researchers operate at the "complicated interface of ... science, policy and practice" (Meine *et al*. 2006).

Emerging from the biological sciences, early conservation biology research primarily focussed on biological and ecological phenomena, such as the genetics and demographics of small populations, island biogeography and reserve design, the impacts of landscape fragmentation, and invasive species (Meine *et al*. 2006). However, as the field matured it became increasingly apparent that most conservation problems resulted from the actions of people, and that the solutions must therefore centre on changing human behaviour and mitigating its impacts. The purely biological nature of the discipline was recognised as inherently limiting (Hilborn and Ludwig 1993), and it thus evolved to embrace additional branches of academia, including social sciences as diverse as sociology, anthropology, political ecology, economics, psychology and geography (Dailly and Ehrlich 1999; Mascia *et al*. 2003). An understanding of human interactions with the environment is essential to identify the underlying drivers of biodiversity loss and implement appropriate interventions (Balmford and Cowling 2006; St. John *et al*. 2013) and governance regimes (Agrawal and Ostrom 2006). However, the role of social science in conservation should not be restricted to research designed to facilitate, inform and advance interventions, since conservation actors, institutions and practices themselves constitute legitimate topics of inquiry, an appreciation of which may help to improve the movement's efficacy and its legitimacy in broader society (Brosius 2006; Sandbrook *et al*. 2013).

Although conservation research is now widely acknowledged as an interdisciplinary endeavour, a range of institutional factors hinder its development as an interdisciplinary science (Adams 2007; Fox *et al*. 2006). Nevertheless, given the transdisciplinary nature of our environmental crisis, there remains a compelling call for the field to evolve further towards true transdisciplinarity (Max-Neef 2005). In this respect, the term conservation biology can be considered a misnomer, and alternatives such as 'conservation science' and 'biodiversity management' better reflect the holistic nature of the field (Kareiva and Marvier 2012).

Interdisciplinary research is particularly critical when it comes to protected areas, which are human constructs designed to change the behaviour of people within a defined geographical space. As such, they form part of complex social-ecological systems (Milner -Gulland 2012; Ostrom 2009), whose management influences (and is influenced by) the relationship between local people and natural resources (Geoghegan and Renard 2002; West and Brockington 2006). Since protected areas seek to regulate human activities within their boundaries, it is critical that their managers understand how and why the communities that live in and around them value and use the resources they contain (St. John *et al*. 2013). This is particularly pertinent given that protected area management and governance approaches have become increasingly pluralistic and people-centred over recent decades.

1.3 Protected areas

1.3.1 Evolving narratives

'Protected areas' of various forms, such as hunting reserves maintained for the elite or sacred/culturally important sites, have existed around the world for centuries (Dudley *et al*. 2009). However, the modern concept of the protected area emerged in the 20th century with the notion of setting aside areas of 'wild' nature for the conservation of biodiversity (Phillips 2004). Based on the belief that the actions and livelihoods of local and indigenous peoples were inimical to the conservation of biodiversity (Wells and McShane 2004), early protected areas sought to separate man from nature. Consequently, they generally involved the prohibition of extractive natural resource use (Brockington and

Schmidt-Soltau 2004) and, in some cases, the coerced displacement of local communities (Adams and Hutton 2007; Brockington and Igoe 2006; Duffy 2010; West *et al*. 2006).

Conscious of ethical and social justice concerns regarding the negative impacts of protected areas on local people (Brechin *et al*. 2002; Hutton *et al*. 2005), conservationists recognised that the hostility of neighbouring communities to protected areas threatened their legitimacy and created a suite of practical problems (Adams and Hulme 2001; Brandon and Wells 1992, although see Brockington 2004). The conservation movement responded accordingly with a new narrative in the 1970s and 1980s that sought to increase the participation of local people in conservation decision-making and improve the flow of benefits to them from protected areas (Hulme and Murphree 1999; Roe 2008). Based on the premise that protected areas would be strengthened and legitimised by achieving conservation and development simultaneously (Berkes 2004), 'community conservation' was enshrined in policy statements from a range of international congresses from the 1980s (Naughton-Treves *et al*. 2005) and manifested itself in a diversity of approaches. These include Integrated Conservation and Development Projects (ICDPs, Wells and McShane 2004), Community-Based Natural Resource Management (CBNRM, Child and Barnes 2010; Dressler *et al*. 2010) and, more recently, payment-based approaches conditional on the delivery of conservation or environmental services (Büscher and Whande 2007; Ferraro and Kiss 2002; Lele *et al*. 2010). During the 1990s, however, a growing body of documentation on the perceived failures of CBNRM and ICDPs (e.g. Brown 2003; Newmark and Hough 2000) triggered a 'back to the barriers' movement advocating a renewed policy focus on strict protected areas (reviewed in Hutton *et al*. 2005; Wilshusen *et al*. 2002).

While debates between advocates of 'fortress conservation' and proponents of community -centred approaches remained polarised in academic circles, protected area policy has continued to progress towards more inclusive models. It is now widely acknowledged that protected areas should generate benefits for wider society and that their establishment and management should contribute to poverty alleviation and rural development in developing countries (Adams *et al*. 2004; Agrawal and Redford 2006; Kaimowitz and Sheil 2007; Redford *et al*. 2008). These objectives are enshrined in the policy statements of institutions such as the Convention on Biological Diversity and the International Union for Conservation of Nature (IUCN). The centrally-managed, exclusionary form of protected area was explicitly rejected as a global model at the Vth World Parks Congress in 2003 (Dudley *et al*. 2014; IUCN 2005), and replaced with an emerging paradigm that incorporates a plurality of protected area categories and governance modes.

1.3.2 Categorisation and governance

The broad IUCN description of a protected area - "a clearly defined geographical space, recognised, dedicated and managed, through legal or other effective means, to achieve the long-term conservation of nature with associated ecosystem services and cultural values" (Dudley 2008) – is testament to the sheer variety of different institutions established around the world to conserve biodiversity *in situ*. These sites differ in their nomenclature, management objectives, permitted activities and spatial context, hence the development of the IUCN categorisation system as a descriptive framework permitting the classification and comparison of protected areas within and between national portfolios (Dudley 2008).

Since 1994, six main categories have been recognised in the IUCN system, ranging from 'strict' protected areas (generally I-IV) in which human impacts are strictly controlled, to 'multiple-use' sites seeking to maintain harmonious people-nature interactions (V) or promote sustainable resource extraction (VI) (Dudley *et al*. 2010; Table 1.1). Categories V and VI, in particular, have attracted debate because of their management emphasis on sustainable use. Gaston *et al*. (2008) state that such areas are "typically of conspicuously less value for biodiversity and are not protected areas in the strict sense", while Locke and Dearden (2005) suggest that only categories I-IV should be recognised. The IUCN, however, maintain that the categories do not imply a hierarchy in terms of "quality or importance" (Dudley 2008). Evidence for the effectiveness of different categories in preventing land-cover change is mixed; stricter categories have been found to be more effective globally (Joppa and Pfaff 2011; Scharlemann *et al*. 2010), although evidence from Latin America and Asia suggests that this is not always the case (Ferraro *et al*. 2013; Nelson and Chomitz 2009). Regardless of effectiveness, multiple-use categories have been increasingly established over the last two decades (Zimmerer *et al*. 2004), and now comprise half the global protected area estate (in terms of area) (Bertzky *et al*. 2012). This trend reflects both evolving conservation paradigms and the lack of remaining large 'wilderness' areas suitable for the creation of some strict categories (Hoekstra *et al*. 2005; Leroux *et al*. 2010).

Table 1.1 International Union for Conservation of Nature (IUCN) definitions of protected area categories (Dudley 2008).

Category	Definition
Ia: Strict nature reserve	Strictly protected areas set aside to protect biodiversity and also possibly geological/geomorphological features, where human visitation, use and impacts are strictly controlled and limited to ensure protection of the conservation values. Such protected areas can serve as indispensable reference areas for scientific research and monitoring.
Ib: Wilderness area	Usually large unmodified or slightly modified areas, retaining their natural character and influence, without permanent or significant human habitation, which are protected and managed so as to preserve their natural condition.
II: National park	Large natural or near natural areas set aside to protect large-scale ecological processes, along with the complement of species and ecosystems characteristic of the area, which also provide a foundation for environmentally and culturally compatible spiritual, scientific, educational, recreational and visitor opportunities.
III: Natural monument	Set aside to protect a specific natural monument, which can be a landform, sea mount, submarine cavern, geological feature such as a cave or even a living feature such as an ancient grove. They are generally small protected areas and often have high visitor value.
IV: Species/ habitat management area	Aim to protect particular species or habitats and management reflects this priority. Many category IV protected areas will need regular, active interventions to address the requirements of particular species or to maintain habitats but this is not a requirement of the category.
V: Protected landscape/ seascape	Where the interaction of people and nature over time has produced an area of distinct character, with significant ecological, biological, cultural and scenic value, and where safeguarding the integrity of this interaction is vital to protecting and sustaining the area and its associated nature conservation and other values.
VI: Sustainable use area	Conserve ecosystems and habitats, together with associated cultural values and traditional natural resource management systems. They are generally large, with most of the area in a natural condition, where a proportion is under sustainable natural resource management and where low-level non-industrial use of natural resources compatible with nature conservation is seen as one of the main aims of the area.

Diversification in management approaches and categories has paralleled an increasing plurality of governance models promoted by the IUCN. Indigenous and community conserved areas are now officially endorsed, as well as private protected areas and those administered by assorted shared governance arrangements (Borrini-Feyerabend *et al*. 2013). Globally, protected area governance shows a trend towards the growing participation of non-state actors (Balloffet and Martin 2007; Cobb *et al*. 2007; Dearden *et al*. 2005).

1.3.3 Coverage and effectiveness

Traditionally, discussions of protected area effectiveness have centred on two key questions (Gaston *et al*. 2008): i) how effective they are at representing the world's biodiversity; and, ii) how successful they are at buffering the biodiversity they represent from processes that threaten their viability. Historically, protected areas have tended to be created in regions with little direct competition for land-use (i.e. places of low economic value and human population density) (Joppa and Pfaff 2009; Pressey 1994, although see Loucks *et al*. 2008). As a result, not all biomes (Hoekstra *et al*. 2005), bioregions (Brooks *et al*. 2004) or species (Beresford *et al*. 2011; Rodrigues *et al*. 2004a) are equally represented within the global portfolio. For example, 12 % of a sample of over 11,000 species did not occur in protected areas, rising to 20 % for threatened species (which tend to have smaller ranges) (Rodrigues *et al*. 2004a). Gaps in protected coverage are disproportionately greatest in the tropics, mountainous areas and on islands (Rodrigues *et al*. 2004b), although existing sites still achieve better species representation than if they were evenly distributed across the planet (Rodrigues *et al*. 2004a). Rich countries tend to have a greater amount of land under protection (McDonald and Boucher 2011).

The non-random location of protected areas renders evaluations of their effectiveness problematic, since they disproportionally occur on marginal lands which face lower overall threat levels and, therefore, require the use of matching or counterfactual techniques to control for confounding variables (Andam *et al*. 2008; Joppa and Pfaff 2011; Nelson and Chomitz 2011). Most evaluations use land-use change (i.e. deforestation) as a proxy for management effectiveness and, in general, find that protected areas experience reduced habitat loss in comparison to unprotected lands (De Fries *et al*. 2005; Joppa and Pfaff 2011; Scharlemann *et al*. 2010). For example, an analysis of Costa Rica's protected area network showed that 7-9 % of forests designated since 1960 would have been deforested by 1997 in the absence of protection (Andam *et al*. 2008). However, some regional and national scale analyses show mixed results; while 82 % of the 76 studies reviewed by Geldmann *et al*. (2013) exhibited lower deforestation within protected areas, 12 % indicated that habitat loss was higher inside protected areas than outside. Moreover, even if protected areas are more effective than leaving land unprotected, few are successful at preventing habitat change in absolute terms. Indeed, deforestation remains a major problem globally within many protected areas (e.g. Allnutt *et al*. 2013; Gaveau *et al*. 2007; Leisher *et al*. 2013; Tang *et al*. 2010), including some of the world's best known World Heritage sites (Watson *et al*. 2014). Assessments of protected area effectiveness at the landscape scale are further complicated by the fact that their establishment may simply displace destructive land uses elsewhere, a phenomenon known as leakage (Ewers and Rodrigues 2008; Kindermann *et al*. 2008). Deforestation rates provide only a coarse measure of protected area effectiveness in conserving biodiversity, as they fail to account for changes in habitat quality within remaining forests. Such habitat degradation is pervasive and could be more prevalent than outright forest clearance (Peres *et al*. 2006), and may be a particular issue for multiple-use protected areas in which extractive activities are permitted (Gardner 2011; Locke and Dearden 2005).

While protected areas have been shown to be an effectual approach to conserving biodiversity in a range of contexts (Brooks *et al*. 2009; Coetzee *et al*. 2014; Geldmann *et al*. 2013; Leverington *et al*. 2010; Miteva *et al*. 2012), major gaps in our knowledge remain regarding the factors that influence their success (Cabeza 2013; Geldmann *et al*. 2013). Most assessments of effectiveness have tended to focus solely on ecological criteria, although the breadth of outcomes now expected of protected areas (e.g. contributions to equitable governance and poverty alleviation, CBD 2010; IUCN 2005) would require multiple dimensions of 'success' (e.g. social, economic, attitudinal) to be evaluated (J. Brooks *et al*. 2006; Lele *et al*. 2010). Such multidisciplinary assessments are rare (Hall *et al*. 2014). It has been reported frequently that protected areas in tropical developing countries have exacerbated poverty in local communities through the imposition of access restrictions and, in some cases, forced evictions (Adams *et al*. 2004; Brockington *et al*. 2006). However, evaluating the impact of protected areas on poverty is methodologically complex (Andam *et al*. 2010; Ferraro and Hanauer 2014a; Pullin *et al*. 2013; Wilkie *et al*. 2006), at least in part because the communities surrounding them tend to be poorer than the average population (Fisher and Christopher 2007; MEA 2005). Lacking access to markets, health and education infrastructure, and development assistance, these communities may be caught in a 'spatial poverty trap' (Redford *et al*. 2008; Scott 2006). Research using matching methods to control for these factors has shown that protected areas can actually contribute to poverty alleviation in a range of circumstances (Andam *et al*. 2010; Canavire-Baccareza and Hanauer 2012; Ferraro *et al*. 2011). For instance, although communities living around protected areas in Costa Rica and Thailand are significantly poorer than the national averages, the existence of protected areas has served to lessen their poverty over time (Andam *et al*. 2010). However, the mechanisms driving the observed poverty reductions, which may include ecosystem service provision, infrastructure development or incomes from tourism, are often unclear (Ferraro and Hanauer 2014b)

1.4 Biodiversity and conservation in Madagascar

1.4.1 A global conservation priority

Spanning 14 degrees of latitude in the western Indian Ocean, Madagascar is the fourth largest island in the world (Fig 1.1). Formerly part of the Gondwana supercontinent, it separated from Africa during the middle to late Jurassic (Coffin and Rabinowitz 1987) and was isolated from all other landmasses by the late Cretaceous, approximately 88 million years ago (Storey *et al*. 1995). As a result of this long isolation, the biota is characterised by extensive radiations derived from a small number of founder events (Karanth *et al*. 2005; Yoder *et al*. 2003) and extremely high rates of endemism at the species and higher taxonomic levels. Indeed, 100 % of indigenous non-volant mammals, 92 % of reptiles, 99 % of amphibians, 52 % of breeding birds and 84 % of plants occur nowhere else on Earth (Callmander *et al*. 2011; Goodman and Benstead 2005). The country forms one of 20 global zoogeographical regions, and is ranked second, behind Australia, in terms of evolutionary uniqueness (Holt *et al*. 2013). On the basis of its species diversity and endemism, as well as elevated levels of threat, Madagascar is recognized as a top global conservation priority (T. Brooks *et al*. 2006). A megadiversity country (Mittermeier *et al*. 1997) containing five of the Global 200 Ecoregions (Olson and Dinerstein 1998), it comprises – along with the other islands of the western Indian Ocean – one of 25 global biodiversity hotspots (Myers *et al*. 2000).

1.4.2 Human impacts: past and present

Although humans were long thought to have colonised Madagascar only 2500 years ago (Crowley 2010), recent findings suggest that some people may have been present at least 1500 years earlier (Dewar *et al*. 2013; Gommery *et al*. 2011). The precise role of people in subsequent environmental change has been hotly debated, but their arrival preceded both a major faunal extinction event and the massive loss of forest cover (Dewar and Richard 2012; Dewar 2014). All endemic species with body mass >10 kg are extinct (Burney and MacPhee 1988), as well as numerous smaller species from extant genera (Goodman and Jungers 2013). The extinct megafauna included nine genera and 17 species of lemur (Burney *et al*. 2004), pygmy hippopotamus, giant tortoises and eight species of ratite, amongst them the world's largest known bird, *Aepyornis maximus*. All but one of these species are known to have persisted into the human period (Burney *et al*.

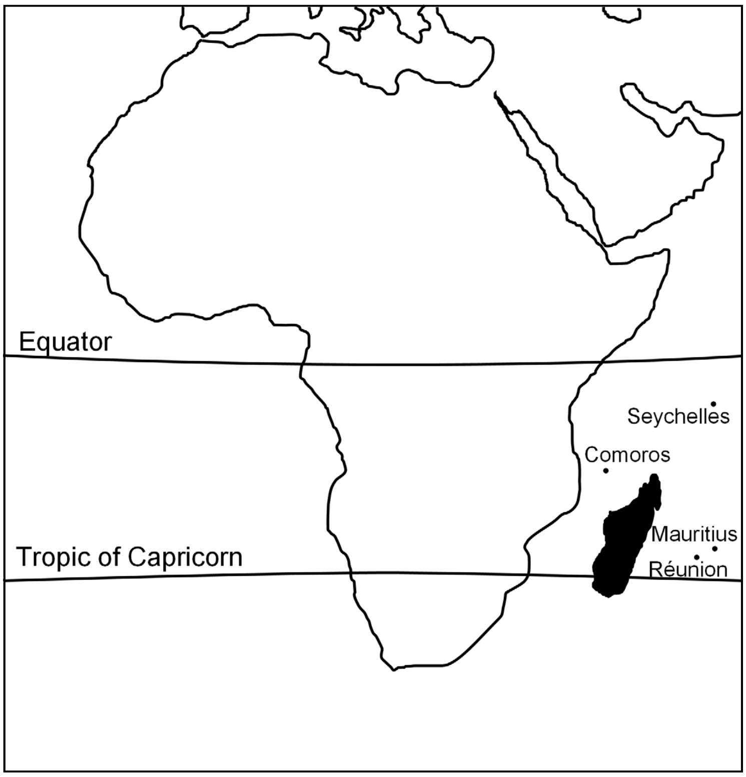

Fig 1.1 Location of Madagascar (black shading) in relation to Africa and neighbouring islands and archipelagos of the western Indian Ocean.

2004; Crowley 2010), suggesting an anthropogenic trigger for their extinction, although the precise mechanisms (e.g. direct predation, habitat transformation through fire and/or grazing and Late Holocene aridification) remain unclear (Crowley 2010; Dewar 2014; Goodman and Jungers 2013; Virah-Sawmy *et al*. 2010).

Madagascar's vegetation history is equally contentious (reviewed in Gade (2008), McConnell and Kull (2014a) and Pollini (2010)). Early European botanists (e.g. Humbert 1927; Perrier de la Bathie 1921) assumed that the island was once entirely forested and suggested that the grasslands which now cover most of the country were entirely anthropogenic in origin, a narrative which persists to this day (McConnell and Kull 2014a,b; Scales 2014a). Recent research, however, suggests that climatic change has played an important role in shaping vegetation dynamics (Bond *et al*. 2008; Virah-Sawmy 2009), and it is now recognised that Madagascar's vegetation would have comprised a mosaic of forest, grassland and shrubland prior to human colonisation (Willis *et al*. 2008). Regardless of the original extent of forest cover, deforestation has been particularly severe within contemporary times as almost 40 % of forest area was lost between 1953 and *c*. 2000. By the start of the new millennium, no more than 16 % of the island retained forest cover (Harper *et al*. 2007). Deforestation on this scale has had severe impacts on the island's endemic biodiversity, which is largely forest dependent (Goodman and Benstead 2005; Wilmé 1996), and an estimated 9 % of species have been committed to extinction by deforestation since 1950 (Allnutt *et al*. 2008).

Today Madagascar remains one of the poorest countries in the world. It has a predominantly rural population (estimated at 70 %; United Nations 2007) that, largely lacking industry, infrastructure or access to markets, remains heavily dependent on ecosystem goods and services generated by the country's forests, wetlands and shallow seas for both material and cultural wellbeing (Table 1.2). For example, 84 % and 97 % of households by Zombitse-Vohibasia and Analavelona protected areas, respectively, are dependent on natural resources for their household income and subsistence (Horning 2003). As a result, rural communities in search of a livelihood remain the principal agents of deforestation and forest degradation across the island (Fritz-Vietta *et al*. 2011), although industrial mineral extraction and agricultural 'land grabs' represent emerging threats (Cardiff and Andriamanalina 2007; Ferguson *et al*. 2014; Freudenberger 2010).

Deforestation is caused primarily by agricultural expansion using shifting cultivation or swidden techniques (Casse *et al*. 2004; Gorenflo *et al*. 2011), including *tavy* (dry rice cultivation) in humid eastern regions, and *hatsake* (primarily maize cultivation) in the dry west and south. However, the drivers of agricultural expansion are often complex and poorly understood (Scales 2014b). The dynamics of *hatsake*, for example, are influenced by export markets for maize, droughts, migration dynamics and the desire to buy cattle for participation in cultural events, amongst other factors (Blanc-Pamard 2004; Casse *et al*. 2004; Minten and Meral 2006; Razanaka *et al*. 2001; Réau 2002; Scales 2011). In addition to stores of potential agricultural land, forests provide a range of resources utilised by rural communities, including timber, fuelwood and charcoal, non-timber forest products and bushmeat (Fritz-Vietta *et al*. 2011; Jasper and Gardner 2015; Kiefer *et al*. 2010). The most important of these is charcoal, since almost 90 % of people nationwide depend on biomass for cooking fuel (Minten et al. 2012). However, relatively little is known about the drivers of natural resource use around the country, constituting a major knowledge gap which hinders the management of protected areas.

Certain cultural traits shared by the Masikoro, Mahafaly and Tandroy ethnicities of southern Madagascar have important repercussions for biodiversity management, including a propensity towards migration, the cultural primacy of cattle, and ancestor worship. Migration to *tany malalaka*, ('land where there is space', equating to the forest

Table 1.2 Examples of the contribution of ecosystem services generated by Madagascar's natural ecosystems to the material and cultural wellbeing of Malagasy populations, arranged according to the Millennium Ecosystem Assessment framework (MEA 2005).

Service	**Service sub-type**	**Examples**
Provisioning services	Food	In the north-west, wild yams (*Dioscorea* spp.) serve as important supplementary food source and 'safety net' during times of rice scarcity (Ackermann 2004). Seven endemic *Dioscorea* spp. are harvested in the Mikea region, and constitute an important food supplement; yams may also be exchanged for fish at weekly markets, and their sale permits families to invest in livestock (Cheban *et al*. 2009). In Ranomafana, harvesting of wild crayfish contributes significantly to the local economy and may constitute an important protein source for children (Jones *et al*. 2006). A range of wild vertebrate species are harvested for food throughout Madagascar, including amphibians (Jenkins *et al*. 2009), bats (Cardiff *et al*. 2009; Goodman 2006), birds, lemurs, and small mammals (Afrosoricida) (Garcia and Goodman 2003; Golden 2009; Goodman and Raselimanana 2003; Rakotondravony 2006). Bushmeat accounted for 10 % of meat consumed in a sample from western Madagascar (Randrianandrianina *et al*. 2010). For the Mikea of the southwest, the collection of wild food sources (bushmeat, yams, honey) constitutes an important component of diversified livelihood strategies (Stiles 1991; Tucker 2006).
	Fuelwood	Over 90 % of Madagascar's domestic energy is derived from fuelwood and charcoal. In cities such as Toliara and Morondava, 100 % of wood-fuel is derived from natural forests (Bertrand *et al*. 2010).
	Fibre (including construction materials)	In the west and north-west, fishing communities use wood collected from mangroves for making fish traps and canoes and the construction of houses and fences (Rasolofo 1997). Masikoro communities in the southwest use forest-derived wood for the construction of homes, fences, tools, coffins and ox-carts; the tree *Givotia madagascariensis* from the same forests is used for dugout canoe construction by neighbouring Vezo communities (Rejo-Fienena 1995).
	Biochemicals	Wild medicinal plants are used in healthcare by rural and urban populations throughout Madagascar (Lyon and Hardesty 2005; Norscia and Borbognini-Tarli 2006; Novy 1997; Randrianarivelojosia *et al*. 2003).
Regulating services	Water regulation	A sample of 20 protected areas provide hydrological regulation for 430, 000 ha of irrigated agriculture and drinking water for 17 towns (Carret and Loyer 2003). More than two thirds of Madagascar's electricity is generated through hydropower, which depends on hydrological regulation services (ADER 2008). Flood prevention as a result of watershed protection contributes significantly to the national economy (Kramer *et al*. 1997).
	Pollination	In the Androy region, loss of even the smallest forest patches results in loss of pollination services required for cultivation of food crops (Bodin *et al*. 2006).

Service	Service sub-type	Examples
	Erosion prevention	Following deforestation of the watershed, erosion and subsequent siltation had reduced Lac Alaotra to 20 % of its original size by 2000, and caused agricultural productivity to decline to 40 % of its previous levels (Bakoariniaina *et al*. 2006). Erosion of agricultural soils is estimated to cost 2.5 % of GDP per year (Carret *et al*. 2010), while sedimentation resulting from inland erosion reduces the health and productivity of marine ecosystems and economically-important fisheries (Maina *et al*. 2012; Sheridan *et al*. 2014).
Cultural services	Spiritual and religious	Forests play an important role in the cultural and spiritual lives of the Tandroy in southern Madagascar (Gardner *et al*. 2008), and are therefore maintained through informal prohibitions (*faly*); forest functions include sheltering ancestral tombs (burial forests, *ala kibory*) and spirits (*kokolampo*) (Tengö *et al*. 2007).
	Recreation and ecotourism	55 % of international visitors come to Madagascar for ecotourism (Christie and Crompton 2003); the financial value of ecotourism contributed an added value of 400 million US$ to the national economy in 2008 (Carret *et al*. 2010).
	Cultural heritage/sense of place	The Vezo culture revolves around the harvest of marine biodiversity (Astuti 1995), and is therefore dependent on the functioning and productivity of marine ecosystems. The culture of the Mikea is also tightly linked to the existence of productive forests (Tucker 2001), while Mahafale pastoralists maintain forests as an important component of their transhumant pastoralist culture (Kaufmann and Tsirahamba 2006).

frontier) is a widespread Malagasy response to resource or land scarcity (Keller 2008). This phenomenon (Malthusian extensification, as opposed to Boserupian intensification, as a reaction to declining resource availability; Bilsborrow (2002)) causes the continual expansion of agricultural land at the expense of native forests and undoubtedly underpins most of the country's deforestation (Gorenflo *et al*. 2011). Furthermore, migrants can disrupt the social bonds that support customary management of particular resources (Curran 2002; Katz 2000; Ostrom *et al*. 1999), such as taboos and sacred areas, thus triggering the breakdown of common pool resources into open access systems. As a result, it has been widely reported that migrant communities may use natural resources more destructively than residents, both in Madagascar (e.g. Kaufmann and Tsirahamba 2006; Réau 2002; Sandy 2006; Watson *et al*. 2007) and worldwide (Cassels 2005; Codjoe and Bilsborrow 2012).

Cattle are central to cultural, spiritual and economic life in southern Madagascar (Evers and van der Zwan 1998; Kaufmann 1998), but the relationship between cattle ownership and forest cover is extremely complex. On the one hand, herders appreciate forests because they provide dry season fodder, shade, and a place to shelter their herds from rustlers (Kaufmann and Tsirahamba 2006; Tsirahamba and Kaufmann 2008). However, the desire to earn money to enable the purchase of a herd is one of the principal reasons for shifting cultivation (Blanc-Pamard 2004; Fenn and Rebara 2003; Samisoa 2001). Moreover, the ritual slaughter of entire herds during the funeral of important elders represents an important drain on the region's financial resources, and may also be a significant driver behind deforestation and rural under-development.

1.4.3 Conservation history

Conservation in Madagascar has, until recently, largely concentrated on forest management, and can be historically characterised as a struggle between the state and rural peasants for control over shifting cultivation (Raik 2007). The Malagasy state has attempted to manage forests since pre-colonial times (Henkels 1999). Repressive forest policies were perpetuated during the French colonial era (1896-1961), during which period *tavy*/*hatsake* and all burning were banned (Raik 2007; Scales 2014c), and extensive areas of forest were cleared for timber extraction and state-promoted cash-cropping (Jarosz 1993, although see Gade 2008 for a critique of this analysis). The colonial French also established the country's first protected areas in 1927. State control over forest resources continued post-independence, but was seen as illegitimate by rural communities possessing *de facto* access to forests based on customary and ancestral rights (Antona *et al*. 2004; Henkels 1999). This resulted in an anarchic situation for forest management (Bertrand 1999); the government lacked the resources to exert its control over forest areas (Kull 2002a), while rural communities lacked incentives to respect or submit themselves to the law, so burnt forests as symbolic resistance (Kull 1999, 2002b). Forest loss continued unabated (Harper *et al*. 2007).

Following a period of socialist isolationism post-independence (Brown 2002), bi-lateral and multilateral donors (particularly the World Bank) began investing in Madagascar's environment during the 1980s. This led to the development of Africa's first National Environmental Action Plan in 1992 (Kull 1996, 2014; Mercier 2006). Phase I (1992-1997) of the three-phase plan focused on the creation of protected areas (particularly as ICDPs) and a raft of new institutions, including the *Association National pour la Gestion des Aires Protégées* (ANGAP), a parastatal body charged with management of the protected area portfolio (Freudenberger 2010; Pollini 2011).

Phase II (1998-2003) marked a major shift in forestry policy, embracing concepts of landscape conservation and promoting community-based management of forests outside protected areas. The GELOSE (*Gestion Locale Sécurisé*) law of 1996 allowed the transfer of limited management rights over natural resources from the state to a local community association through a time-bound, renewable contract (GoM 1996), and was followed in 2001 by a streamlined version, GCF (*Gestion Contractuel des Forêts*), specifically for forests (Antona *et al*. 2004; Bertrand *et al*. 2014; Raik and Decker 2007). Although widely promoted as a conservation tool by government agencies and conservation non-governmental organisations (NGOs), there is little evidence that these CBNRM contracts have been successful either in conserving forests or promoting community development (Hockley and Andriamarovololona 2007; Pollini *et al*. 2014). These failures have been attributed to a lack of institutional support and follow-up during the management transfer process; imbalances of objectives and inequalities of power between the state and signatory communities (Hockley and Andriamarovololona 2007; Pollini and Lassoie 2011), and; a range of incompatibilities between the processes and principles of CBNRM as practiced in Madagascar and the socio-cultural norms of rural communities (Fritz-Vietta *et al*. 2011).

1.4.4 The Durban Vision

In 2003, at the Vth World Parks Congress held in Durban, South Africa, then President of the Republic of Madagascar, Marc Ravalomanana, declared his government's intention to triple the coverage of the country's protected areas from 1.7 to 6 million hectares within five years[1] (Mittermeier *et al*. 2005; Norris 2006). Known as the 'Durban Vision', this ambitious declaration was partly influenced by World Bank analyses demonstrating the economic importance of the country's forests (Carret and Loyer 2003), but was largely brought about by the lobbying power of international environmental NGOs and funders (Corson 2011, 2014; Duffy 2006; Horning 2008).

Madagascar's first generation of protected areas, many of which were established during the colonial period prior to 1960 (with a subsequent wave of ICDPs created in the early 1990s; Virah-Sawmy *et al*. 2014), were largely created in wilderness areas with negligible human populations (Ferguson 2010). Prior to 2003, the portfolio comprised 46 sites managed by ANGAP, now rebranded Madagascar National Parks (MNP), in some cases in partnership with NGOs. With a total area of almost 1.7 million ha the network comprised three categories of protected area (Randrianandianina *et al*. 2003): i) *Réserve Naturelle Intégrale* (Strict Nature Reserve; IUCN category Ia); ii) *Parc National* (National Park; category II); and, iii) *Réserve Spéciale* (Special Reserve; category IV).

[1]The deadline was subsequently extended to 2012.

They were established with little regard to the resource requirements of adjacent communities (Durbin and Ralambo 1994), as the primary management objective of all categories was the conservation of biodiversity, alongside limited research and recreation (within categories II and IV). All extractive use of biodiversity, except for scientific purposes, was strictly prohibited according to national law (*Code des Aires Protégées*; GoM 2001).

Following the Durban Vision declaration, steering committees were established to advise on its implementation, and national-level gap analyses and prioritisation exercises were carried out to inform the location of new protected areas (Kremen *et al*. 2008; Rasoavahiny *et al*. 2008). These analyses, however, focused purely on biodiversity data and did not incorporate management-relevant variables such as cost, opportunity or threat (Corson 2012, 2014). The steering committees recognised two critical obstacles: i) MNP did not have the capacity to oversee the expansion themselves; and ii) the majority of priority sites contained significant human populations that largely depend upon natural resources for their subsistence and household income (Gardner *et al*. 2008; Virah-Sawmy *et al*. 2014). The established protected area models of IUCN categories Ia, II and IV were therefore seen as inappropriate for the majority of new protected areas and, with the support of IUCN consultants (Borrini-Feyerabend and Dudley 2005), the IUCN frameworks were used to guide the development of novel categories and governance models for the country[2]. Protected areas legislation was revised to recognise category III, V and VI protected areas, as well as to permit non-governmental actors to promote, manage and govern protected areas. Most new protected areas are proposed as category V and are, or will be, governed under some form of co-management integrating local community structures (Ferguson *et al*. 2014; Raik 2007; Virah-Sawmy *et al*. 2014). The objectives of the new Madagascar Protected Area System (known by its French acronym SAPM for *Système d'Aires Protégées de Madagascar*), which comprises the established MNP network, as well as the post-Durban generation of protected areas, are threefold: i) to conserve the whole of Madagascar's unique biodiversity; ii) to conserve Madagascar's cultural heritage; and, iii) to promote sustainable natural resource use for development and poverty alleviation (Commission SAPM 2006). Note that due to the political crisis

[2]Since the Durban vision was launched at the same Vth World Parks Congress in 2003 that marked the transition from old to new protected area models as the dominant paradigm at global policy level (Dudley *et al*. 2014), Madagascar provided an ideal test case for the implementation of new categories and governance models. The IUCN were thus heavily involved in the development of domestic policy (Corson 2014). Experiences in Madagascar, in turn, strongly influenced the development of subsequent IUCN best practice guidelines (Borrini-Feyerabend *et al*. 2013; Dudley 2008).

that left the country without a recognised legitimate government between early 2009 and 2014, the revised *Code des Aires Protégées* has yet to be ratified and passed into law; all proposed category III, V and VI protected areas await this ratification before they can gain definitive protected area status (Virah-Sawmy *et al*. 2014).

Almost a hundred new protected areas had been legally established by 2012, although official sources (a SAPM database and a 2010 interministerial decree granting blanket temporary protection to an appended list of sites) disagree on the precise number (AGRECO 2012) (Fig. 1.2). The establishment and management of these sites pose a number of challenges for the Malagasy government and its NGO partners. These include the development of robust and equitable governance mechanisms for community- or co-managed protected areas (Ferguson *et al*. 2014; Virah-Sawmy *et al*. 2014), the sustainable financing of both individual protected areas and the system as a whole (AGRECO 2012) and, most critically, satisfying the needs of multiple stakeholders (Gardner *et al*. 2013). The latter is particularly complex given the high dependence of rural communities living in and around protected areas on natural resources, and the fact that most traditional land- and resource-use has largely negative impacts on endemic biodiversity (Gardner 2009, 2011; Irwin *et al*. 2010). Thus, multiple-use protected areas within SAPM must be carefully designed and managed if they are to successfully maintain the viability of their constituent habitats and species, while simultaneously satisfying the subsistence and development needs of neighbouring populations (Gardner 2009). However, due to the time-limited nature of the Durban Vision ("an emergency conservation context", Marie *et al*. 2009), the promoters of many new protected areas rushed to establish their sites without sufficient public consultation (Corson 2012) or, in many cases, a necessary understanding of the dynamics of the local social-ecological systems (Gardner 2012). This contrasted strongly with science-based approach that underpinned the establishment of some ICDPs in the 1990s (Kremen *et al*. 1999). As a consequence, there remain enormous gaps in our knowledge regarding how to reconcile conservation and development in the country's new generation of multiple-use protected areas.

1.5 Study Site

Southern Madagascar is sub-arid and characterised by a unique vegetation type in which over 50 % of plant species are locally endemic (Phillipson 1996). As a result, it has long been recognised as a distinct biogeographical unit, although nomenclature and precise coverage have varied depending on the criteria used. Recent appellations include 'West

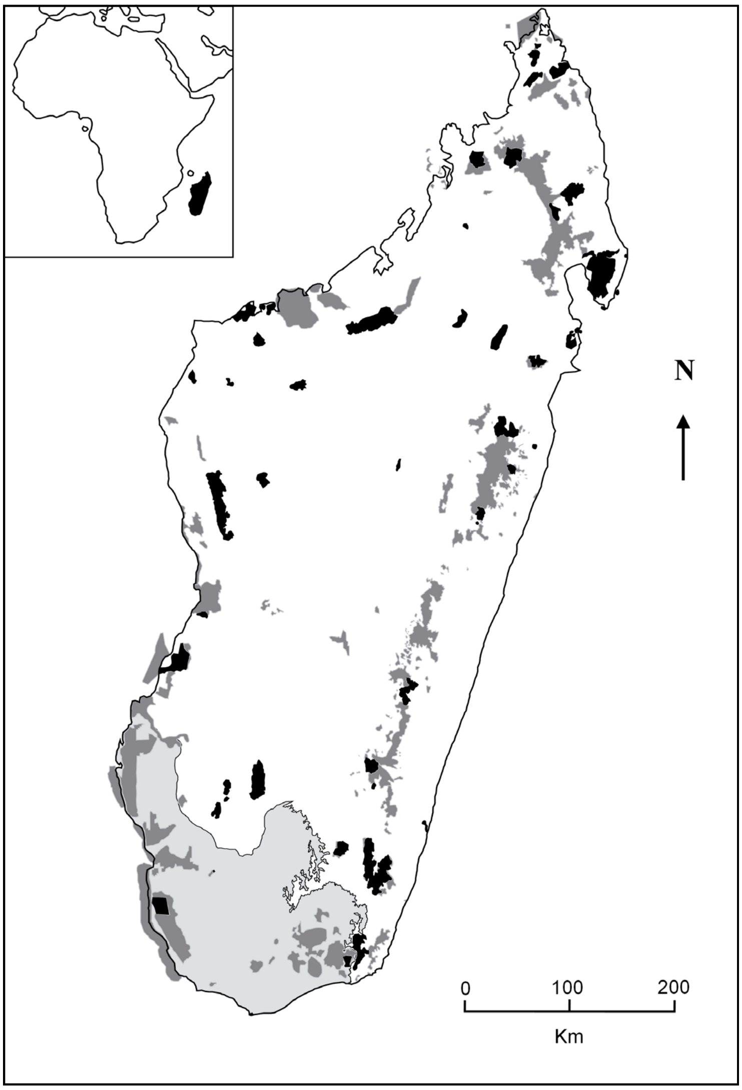

Fig 1.2 Map of Madagascar showing pre-2003 National Parks, Special Reserves and Strict Nature Reserves managed by Madagascar National Parks (black polygons), new protected areas established by 2010 (dark grey polygons), and the limits of the spiny forest ecoregion as defined by Goodman and Raherilalao (2013) (light grey shading).

Malagasy deciduous thicket' (White 1983), 'Madagascar spiny desert' (Olson and Dinerstein 1998), 'spiny forest' (Fenn 2003) and 'southwestern dry spiny forest' (Goodman and Raherilalao 2013). It is recognised as one of 200 global priority ecoregions (Olson and Dinerstein 1998), but was the least represented ecoregion within Madagascar's protected area network prior to 2003 (Fenn 2003; Seddon *et al*. 2000). In addition, the spiny forest suffered the fastest rates of deforestation in the country between 1990 and 2010 (Harper *et al*. 2007; ONE *et al*. 2013).

Ranobe PK32 is a new protected area of 148,554 ha in southwest Madagascar, promoted and co-managed by the international NGO WWF (World Wide Fund for Nature) (Virah-

Sawmy *et al*. 2014). It forms part of the South Mangoky centre of microendemism (Wilmé *et al*. 2006), and has the highest recorded faunal diversity of any protected area within the ecoregion (Gardner *et al*. 2009a,b, Chapter 5). Key species include the long-tailed ground roller (*Uratelornis chimaera*) and the sub-desert mesite (*Monias benschi*), two bird species belonging to monospecific genera of endemic families (Raherilalao and Goodman 2011; Seddon *et al*. 2000), both of which are classified as Vulnerable (IUCN 2011) and have ranges restricted to two protected areas. Other priority species which are not known to occur anywhere else include *Furcifer belalandaensis*, likely to be the world's rarest chameleon (C. Raxworthy *pers. comm*.), and the narrow-striped boky (*Mungotictis decemlineata lineata*), a mongoose-like carnivore known from only two specimens (Goodman *et al*. 2005; Hawkins *et al*. 2000) that is in the process of being elevated to full specific status (Goodman 2012).

Located within 20 km of the regional capital Toliara, Ranobe PK32[3] is a category VI multiple-use protected area containing five strict conservation areas, equating to 13.5 % of its areal extent, with the remainder comprising sustainable use zones (Fig. 1.3). It is surrounded by a population of approximately 90,000 people and serves as a regionally important source of natural resources for both rural and urban populations (WWF 2010). Despite the ecosystem goods and services that can be sustainably generated from the protected area's natural habitats, the biodiversity of the site remains highly threatened by: i) shifting cultivation (*hatsake*); and, ii) selective logging for construction wood and, in particular, charcoal production. Indeed, 98 % of households in Toliara use charcoal or fuelwood to cook, and 54 % of this urban demand is met by charcoal produced along the RN9 road, with a further 8 % derived from the Fiherenana Valley (Partage 2008). In addition, rural households harvest fish, crustaceans and aquatic plants from the lakes and rivers within Ranobe PK32, and wild yams (Cheban *et al*. 2009), fruits, medicinal plants, construction wood, bushmeat and a range of other products from forests (Chapter 7; C. Gardner, unpublished data; Rejo-Fienana 1995; WWF 2010). In addition, forests provide fodder for Zebu cattle and serve to shelter them from rustlers (Kaufmann and Tsirahamba 2006).With the protected area expected to conserve its biodiversity and continue to satisfy the needs of a large rural population, its management requires the implementation of

[3]Various attributes of the site have changed since the launch of the Ranobe PK32 protected area establishment programme in 2006, including its name, size, limits and IUCN category. Since a number of chapters in this thesis have been published as journal articles, these facts vary throughout the thesis. For instance, Chapter 3 refers to PK32-Ranobe and examines its proposal as a category V site but, subsequently (and partially as a direct result of that analysis), the designation was changed to the more appropriate category VI.

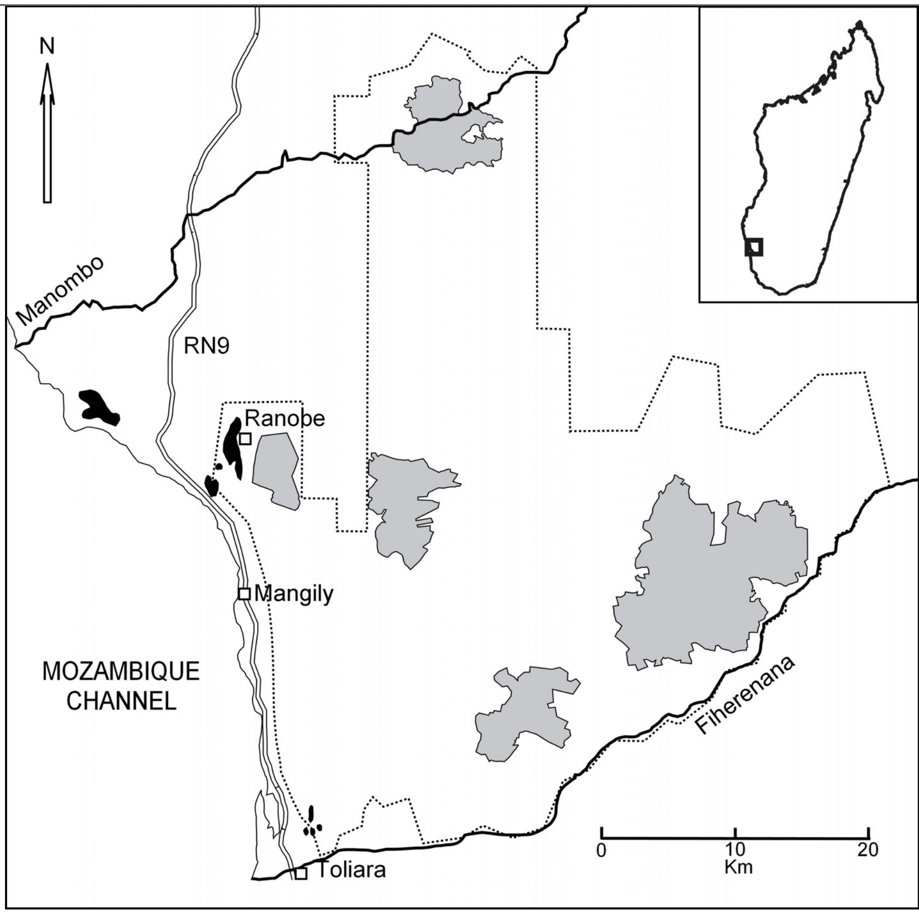

Figure 1.3 Map of Ranobe PK32 protected area (dotted line) showing five strict conservation zones (grey shading) and features of interest: rivers/wetlands, black lines/shading; towns and villages, white squares; Route Nationale 9 (RN9) road, double line. Inset shows position of site within Madagascar.

appropriate, evidence-informed strategies. However, little is known about the drivers of natural resource use by populations around the protected area or their impacts on biodiversity.

1.6 Thesis context and overview

The rapid expansion of Madagascar's protected area system has involved the implementation of novel management approaches and innovative governance arrangements in an effort to meet diverse, and potentially conflicting, ecological and social objectives. As such, this ambitious and unprecedented programme presents an ideal case study to explore and investigate the real-world establishment of new protected areas and contribute to the global knowledge-base on conservation practice and policy. Wanting to both contribute to, and learn from, this exciting initiative, my research interests and objectives were strongly influenced by three inter-related research agendas: i) from a practitioner perspective, to inform and generate a local, context-specific evidence base to support the development of management interventions in new protected areas (as well as analogous sites worldwide); ii) from a policy-relevant standpoint, to explore and learn from experiences in Madagascar, in order to contribute to the development of global best practice, and; iii) as an academic, to indentify and articulate research priorities to where they are most needed in support of effective protected area management.

Conducting research focussed on reconciling conservation and development within new protected areas necessitated an interdisciplinary, mixed-methods approach. Thus, this thesis incorporates elements of ‘traditional’ ecologically-centred conservation research, in addition to both quantitative and qualitative social science. Furthermore, two chapters focus on conservation implementation itself as a topic of study. The specific objectives are to:

Objective 1: investigate the impacts of natural resource exploitation on biodiversity within multiple-use protected areas.

Objective 2: examine the factors influencing the resource needs of local communities living adjacent to, or within, multiple-use protected areas, with a view to informing appropriate management strategies to mitigate their impacts on biodiversity without exacerbating poverty.

Objective 3: use insights from the implementation of the Durban Vision to inform conservation theory, policy and research agendas in Madagascar and worldwide.

The thesis is structured as follows:

The first two chapters explore nationwide trends in the management of new protected areas, in order to contribute to academic and policy debates about protected area categorisation and the role of protected areas in poverty alleviation. In **Chapter 2**, I use three case studies from the marine, freshwater and terrestrial realms to illustrate the strategies employed by promoters of new protected areas to simultaneously pursue conservation and poverty alleviation goals, and explore the implications for global protected area policy. Although all emphasise livelihoods-based approaches to management, the case studies differ in focus, either seeking to enhance the management of the natural resource base (in aquatic systems), or decouple its use from development by promoting alternatives. Experiences in Madagascar suggest that sustainable natural resource use can provide a foundation for poverty alleviation, but that such 'win-wins' for conservation and development may be more likely with some resources (fisheries) than with others (forests). Although little is known about the context and conditions that influence the effectiveness of different approaches, managers can collaborate with applied ecologists and social scientists to build the evidence base required.

In **Chapter 3** I use a decision-tool developed by the IUCN to assess the suitability of the IUCN management categories to Madagascar's new generation of multiple-use protected areas. I apply the tool to 10 case study sites proposed or designated as category V, representing a range of geographical and ecological contexts, but find them to be incompatible with all IUCN protected area categories. Strict protected areas (categories I-IV) are inappropriate because of the case study sites' management emphasis on extractive natural resource use, while category VI is unsuitable because the sites largely comprise cultural landscapes influenced by human action, rather than the natural landscapes required. The category V model seeks to maintain biodiverse cultural landscapes and the human livelihoods/land-uses that created them, but this differs fundamentally from the Malagasy context where managers seek to reduce, transform or mitigate human activities because they are overwhelmingly negative for endemic biodiversity. Since the IUCN categorisation system does not recognise protected areas of the type that prevail in Madagascar's expanded network, modification will be required to ensure conformity between protected area policy and practice.

An important first step in working out how to reconcile conservation and development goals within multiple-use protected areas is to understand what effects permitted livelihood activities are likely to have on the biodiversity the sites were established to conserve. Therefore, in **Chapter 4**, I carry out a review of the literature on the impacts of anthropogenic habitat change on biodiversity in Madagascar, and explore the implications of this body of knowledge for the design and management of new protected areas. The review demonstrates that habitat change can have positive or negative effects on biodiversity, depending on the species, ecosystem and type of modification, but overall may trigger turnover from specialists to generalists, and endemic to non-endemic species, thus contributing to homogenisation of the island's biota. Multiple-use protected areas must therefore be planned and managed carefully to minimise and mitigate expected impacts. The findings provide insight into: i) the spatial arrangement of strict conservation zones within protected areas; ii) the selection of conservation targets; iii) the importance of adaptive management in reconciling conservation and development; iv) the design of monitoring protocols; v) the evaluation of species viability, and; vi) the prioritisation of strict vs. multiple-use protected areas within the national network.

The literature review made it clear that multiple-use protected areas will be more effective at conserving some elements of biodiversity than others and, consequently, that some species should assume greater importance in conservation decision-making. However, we have no simple methods to ‘quantify’ or enumerate the variation in conservation value of different species. Such an index would be a helpful tool in understanding the impacts of habitat change within protected areas (Chapter 6), and could also be used to prioritise between sites. In particular, it could be an improvement on other rudimentary scoring systems that are often used by practitioners and decision-makers in site prioritisations. Therefore, in **Chapter 5**, I develop a Conservation Value Index (CVI) in which I assign scores to species based on rarity (endemism and representation in protected areas) and threat (hunting/collection pressure and degradation tolerance), and test its performance in prioritising a portfolio of 22 sites from western and southern Madagascar, based on their reptile fauna. I also use three alternative metrics (species richness, an index derived from species’ IUCN Red List status and an index based on irreplaceability) to prioritise the same portfolio of sites, comparing the results from each to a benchmark produced using the gold-standard systematic conservation planning software Zonation. The results suggest that, overall, the established generation of MNP-managed strict protected areas may be more important than the Durban Vision sites. CVI performs better than other metrics in comparison to Zonation, providing evidence that, in situations where a lack of capacity or data prevents the use of more sophisticated analyses, simple heuristic approaches may provide a useful alternative.

Having found, in Chapter 4, that almost nothing is known about the impacts of habitat degradation within the spiny forest ecoregion, I set out in **Chapter 6** to investigate how bird and reptile communities respond to different intensities of land use, from shifting cultivation and charcoal production, within Ranobe PK32 protected area. Since managers are particularly interested in rare, threatened and endemic species, rather than widespread and common ones, I use the CVI developed in Chapter 5 to generate conservation value scores for all species. I then weight species’ CVI scores by their relative abundance at each of three sites across the degradation gradient, using data obtained via pit-fall traps and transect-based refuge searches for reptiles and point counts for birds. I find that high-intensity degradation results in low species richness and conservation value in both groups, but that medium-intensity degradation provokes an increase in bird richness. However, this masks a turnover from forest specialists to generalists and it is the areas of low-intensity degradation that retain the highest conservation value. Multiple-use zones of new protected areas may not effectively conserve all of Madagascar's endemic biodiversity, casting doubt on the plausibility of managing the extended network for both conservation and social objectives.

In the next two chapters, I use social science methods to investigate the drivers of natural resource use within Ranobe PK32 and inform appropriate management strategies. In **Chapter 7**, I employ in-depth, semi-structured interviews with a small number of key informants to qualitatively explore the role of hunting and bushmeat consumption in the lives of a single rural community. In **Chapter 8**, I use a questionnaire survey to investigate an observed increase in charcoal production across the southwest region of Madagascar. The two approaches generate complementary insights into why these livelihood activities are practised; hunting is a secondary activity carried out opportunistically, while charcoal is predominantly produced as a displacement activity when farming, fishing and herding are no longer sufficiently profitable (e.g. as a result of reduced irrigation infrastructure and changing rainfall patterns). However, both are also carried out when respondents lack money, and to supplement income during the agricultural off season. These findings suggest that forest resources comprise a fallback or safety net for rural communities, and therefore that access restrictions associated with protected area establishment would exacerbate poverty. However, targeted development interventions to improve the profitability of non-forest livelihoods, particularly farming, may reduce the relative attractiveness of natural resource use and thus help protected area managers to achieve conservation goals without exacerbating poverty.

Finally, in **Chapter 9**, I synthesise the major findings of my research and discuss the insights they provide with respect to i) the reconciliation of conservation and development in Madagascar's rapidly expanding protected area system, and ii) the role of conservation science in ensuring the successful achievement of these goals. Several lines of reasoning suggest that multiple-use protected areas may be unable to simultaneously conserve biodiversity and promote its use, and therefore that conservation science is required to optimise management. Maximising the contribution of research will depend on promoting further interdisciplinarity and bridging the researcher-practitioner divide by, amongst other things, realigning research agendas with the information needs of managers. I end by suggesting priority topics for further enquiry.

References

Ackermann, K. 2004. Utilisation of wild growing yams as supplementary nutrition and its impact on the dry forest ecosystem in north-western Madagascar. *Swiss Forestry Journal* 155: 80–88.

Adams, W.M. 2007. Thinking like a human: social science and the two cultures problem. *Oryx* 41: 275–276.

Adams, W.M. and Hulme, D. 2001. Conservation and communities: changing narratives, policies and practices in African conservation. In *African Wildlife and Livelihoods: the Promise and Performance of Community Conservation*, D. Hulme and M. Murphree (Eds.), pp. 9–23. James Currey, Oxford.

Adams, W.M. and Hutton, J. 2007. People, parks and poverty: political ecology and biodiversity conservation. *Conservation and Society* 5: 147–183.

Adams, W.M., Aveling, R., Brockington, D., Dickson, B., Elliott, J., Hutton, J., Roe, D., Vira, B. and Wolmer, W. 2004. Biodiversity conservation and the eradication of poverty. *Science* 306: 1146–1149.

ADER (Agence de Développement de L'Electrification Rurale). 2008. *Statistique Officiel 2008*. Agence de Développement de L'Electrification Rurale, Antananarivo.

Agrawal, A. and Ostrom, E. 2006. Political science and conservation biology: a dialog of the deaf. *Conservation Biology* 20: 681–682.

Agrawal, A. and Redford, K. 2006. *Poverty, Development and Biodiversity Conservation: Shooting in the Dark?* Wildlife Conservation Society, New York.

AGRECO. 2012. *Analyse des Coûts et Sources de Financement du Système des Aires Protégées de Madagascar (Octobre 2010 – Janvier 2012*). AGRECO, Antananarivo.

Allnutt, T.F., Ferrier, S., Manion, G., Powell, G.V.N., Ricketts, T.H., Fisher, B.L., Harper, G.J., Irwin, M.E., Kremen, C., Labat, J.-N., *et al*. 2008. A method for quantifying biodiversity loss and its application to a 50-year record of deforestation across Madagascar. *Conservation Letters* 1: 173–181.

Allnutt, T.F., Asner, G.P., Golden, C.D. and Powell, G.V.N. 2013. Mapping recent deforestation and forest disturbance in northeastern Madagascar. *Tropical Conservation Science* 6: 1–15.

Andam, K.S., Ferraro, P.J., Pfaff, A., Sanchez-Azolfelfa, G.A. and Robalino, J.A. 2008. Measuring the effectiveness of protected area networks in reducing deforestation. *Proceedings of the National Academy of Sciences, USA* 105: 16089–16094.

Andam, K.S., Ferraro, P.J., Sims, K.R.E., Healy, A. and Holland, M.B. 2010. Protected areas reduced poverty in Costa Rica and Thailand. *Proceedings of the National Academy of Sciences, USA* 107: 9996–10001.

Antona, M., Biénabe, E.M., Salles, J.-M., Péchard, G., Aubert, S. and Ratsimbarison, R. 2004. Rights transfers in Madagascar biodiversity policies: achievements and significance. *Environment and Development Economics* 9: 825–847.

Astuti, R. 1995. *People of the Sea: Identity and Descent among the Vezo of Madagascar.* Cambridge University Press, Cambridge.

Bakoariniaina, L.N., Kusky, T. and Raharimahefa, T. 2006. Disappearing Lac Alaotra: monitoring catastrophic erosion, waterway silting, and land degradation hazards in Madagascar using Landsat imagery. *Journal of African Earth Sciences* 44: 241–252.

Balloffet, N.M. and Martin, A.S. 2007. *Governance Trends in Protected Areas: Experiences from the Parks in Peril Program in Latin America and the Caribbean*. The Nature Conservancy, Arlington.

Balmford, A. and Cowling, R.M. 2006. Fusion or failure? The future of conservation biology. *Conservation Biology* 20: 692–695.

Barnosky, A.D., Matzke, N., Tomiya, S., Wogan, G.O.U., Swartz, B., Quental, T.B., Marshall, C., McGuire, J.L., Lindsey, E.L., Maguire, K.C., *et al*. 2011. Has the Earth's sixth mass extinction already arrived? *Nature* 471: 51–57.

Beresford, A.E., Buchanan, G.M., Donald, P.F., Butchart, S.H.M., Fishpool, L.D.C. and Rondinini, C. 2011. Poor overlap between the distribution of protected areas and globally threatened birds in Africa. *Animal Conservation* 14: 99–107.

Berkes, F. 2004. Rethinking community-based conservation. *Conservation Biology* 18: 621–630.

Bertrand, A. 1999. La gestion contractuelle, pluraliste et subsidiaire des ressources renouvelables à Madagascar (1994-1998). *African Studies Quarterly* 3: 75–81.

Bertrand, A., Ramamonjisoa, B. and Montagne, P. 2010. Les filières péri-urbaines d'approvisionnement en bois énergie des grandes villes de Madagascar. In *ARINA, le Charbon de Bois à Madagascar: Entre Demande Urbaine et Gestion Durable*, P. Montagne, S. Razafimahatratra, A. Rasamindisa and R. Crehay (Eds.), pp. 23–36. CITE, Antananarivo.

Bertrand, A., Aubert, S., Montagne, P., Lohanivo, A.C. and Razafintsalama, M.H. 2014. Madagascar, politique forestière: bilan 1990–2013 et propositions. *Madagascar Conservation & Development* 9: 20–30.

Bertzky, B., Corrigan, C., Kemsey, J., Kenney, S., Ravilious, C., Besançon, C., and Burgess, N. 2012. *Protected Planet Report 2012: Tracking Progress Towards Global Targets for Protected Areas*. IUCN, Gland and UNEP-WCMC, Cambridge.

Bilsborrow, R.E. 2002. Migration, population change, and the rural environment. *Environmental Change and Security Project Report* 8: 69–94.

Blanc-Pamard, C. 2004. Un jeu foncier de front pionnier en forêt des Mikea (sud-ouest de Madagascar): pratiques, acteurs et enjeux. *Land Reforme/Réforme Agraire/Reforma Agraria* 2: 94–107.

Bodin, O., Tengö, M., Norman, A., Lundberg, J. and Elmqvist, T. 2006. The value of small size: loss of forest patches and ecological thresholds in southern Madagascar. *Ecological Applications* 16: 440–451.

Bond, W.J., Silander Jr., J.A., Ranaivonasy, J. and Ratsirarson, J. 2008. The antiquity of Madagascar's grasslands and the rise of C_4 biomes. *Journal of Biogeography* 35: 1743–1758.

Borrini-Feyerabend, G. and Dudley, N. 2005. *Elan Durban: Nouvelles Perspectives pour les Aires Protégées à Madagascar*. IUCN report to the Government of Madagascar.

Borrini-Feyerabend, G., Dudley, N., Jaeger, T., Lassen, B., Pathak Broome, N., Phillips, A. and Sandwith, T. 2013. *Governance of Protected Areas: from Understanding to Action*. IUCN, Gland.

Brandon, K.E. and Wells, M. 1992. Planning for people and parks: design dilemmas. *World Development* 20: 557–570.

Brechin, S.R., Wilshusen, P.R., Fortwangler, C.L. and West, P.C. 2002. Beyond the square wheel: toward a more comprehensive understanding of biodiversity conservation as social and political process. *Society and Natural Resources* 15: 41–64.

Brockington, D. 2004. Community conservation, inequality and injustice: myths of power in protected area management. *Conservation and Society* 2: 411–432.

Brockington, D. and Schmidt-Soltau, K. 2004. The social and environmental impacts of wilderness and development. *Oryx* 38: 140–142.

Brockington, D. and Igoe, J. 2006. Eviction for conservation: a global overview. *Conservation and Society* 4: 424–470.

Brockington, D., Igoe, J. and Schmidt-Soltau, K. 2006. Conservation, human rights, and poverty reduction. *Conservation Biology* 20: 250–252.

Brooks, J.S., Franzen, M.A., Holmes, C.M., Grote, M.N. and Mulder, M.B. 2006. Testing hypotheses for the success of different conservation strategies. *Conservation Biology* 20: 1528–1538.

Brooks, T.M., Bakarr, M.I., Boucher, T., da Fonseca, G.A.B., Hilton-Taylor, C., Hoekstra, J., Moritz, T., Olivieri, S., Parrish, J., Pressey, R.L., *et al*. 2004. Coverage provided by the global protected-area system: is it enough? *BioScience* 54: 1081–1091.

Brooks, T.M., Mittermeier, R.A., da Fonseca, G.A.B., Gerlach, J., Hoffmann, M., Lamoreux, J.F., Mittermeier, C.G., Pilgrim, J.D. and Rodrigues, A.S.L. 2006. Global biodiversity conservation priorities. *Science* 313: 58–61.

Brooks, T.M., Wright, S.J. and Sheil, D. 2009. Evaluating the success of conservation actions in safeguarding tropical forest biodiversity. *Conservation Biology* 23: 1448–1457.

Brosius, J.P. 2006. Common ground between anthropology and conservation biology. *Conservation Biology* 20: 683–685.

Brown, M. 2002. *A History of Madagascar*. Markus Wiener, Princeton.

Brown, K. 2003. Integrating conservation and development: a case of institutional misfit. *Frontiers in Ecology and the Environment* 1: 479–487.

Burney, D.A. and MacPhee, R.D.E. 1988. Mysterious island: what killed Madagascar's large native animals? *Natural History* 97: 46–55.

Burney, D.A., Burney, L.P., Godfrey, L.R., Jungers, W.L., Goodman, S.M., Wright, H.T. and Jull, A.J.T. 2004. A chronology for late prehistoric Madagascar. *Journal of Human Evolution* 47: 25–63.

Büscher, B. and Whande, W. 2007. Whims of the winds of time? Emerging trends in biodiversity conservation and protected area management. *Conservation and Society* 5: 22–43.

Cabeza, M. 2013. Knowledge gaps in protected area effectiveness. *Animal Conservation* 16: 381–382.

Callmander, M.W., Phillipson, P.B., Schatz, G.E., Andriambololonera, S., Rabarimanarivo, M., Rakotonirina, N., Raharimampionona, J., Chatelain, C., Gautier, L. and Lowry II, P.P. 2011. The endemic and non-endemic vascular flora of Madagascar updated. *Plant Ecology and Evolution* 144: 121–125.

Canavire-Bacarreza, G. and Hanauer, M.M. 2012. *Estimating the Impacts of Bolivia's Protected Areas on Poverty*. International Centre for Public Policy, Atlanta.

Cardiff, S.G. and Andriamanalina, A. 2007. Contested spatial coincidence of conservation and mining efforts in Madagascar. *Madagascar Conservation & Development* 2: 28–34.

Cardiff, S.G., Ratrimomanarivo, F.H., Rembert, G. and Goodman, S.M. 2009. Hunting, disturbance and roost persistence of bats in caves at Ankarana, northern Madagascar. *African Journal of Ecology* 47: 640–649.

Carret, J.-C. and Loyer, D. 2003. *Comment Financer Durablement le Réseau d'Aires Protégées Terrestres à Madagascar? Apport de l'Analyse Économique*. World Bank presentation at Vth World Parks Congress, Durban.

Carret, J.C., Rajaonson, B., Feno, P.J. and Brand, J. 2010. *L'Environnement à Madagascar: un Atout à Préserver, des Enjeux à Maitriser*. World Bank, Washington DC.

Casse, T., Milhøj, A., Ranaivoson, S. and Randriamanarivo, J.R. 2004. Causes of deforestation in southwestern Madagascar: what do we know? *Forest Policy and Economics* 6: 33–48.

Cassels, S., Curran, S.R. and Kramer, R. 2005. Do migrants degrade coastal environments? Migration, natural resource extraction and poverty in North Sulawesi, Indonesia. *Human Ecology* 33: 329–363.

CBD (Convention on Biological Diversity). 2010. *Decision Adopted by the Conference of the Parties to the Convention on Biological Diversity at its Tenth Meeting [Decision X/2] Nagoya, Aichi Prefecture, Japan, 18–29 October 2010*. Secretariat to the Convention on Biological Diversity, Montreal.

Chape, S., Harrison, J., Spalding, M. and Lysenko, I. 2005. Measuring the extent and effectiveness of protected areas as an indicator for meeting global biodiversity targets. *Philosophical Transactions of the Royal Society* B 360: 443–455.

Chapin, F.S., Zavaleta, E.S., Eviner, V.T., Naylor, R.L., Vitousek, P.M., Reynolds, H.L., Hooper, D.U., Lavorel, S., Sala, O.E., Hobbie, S.E., *et al*. 2000. Consequences of changing biodiversity. *Nature* 405: 234–242.

Cheban, S.A., Rejo-Fienena, F. and Tostain, S. 2009. Etude ethnobotanique des ignames (*Dioscorea* spp.) dans la forêt Mikea et le couloir d'Antseva (sud-ouest de Madagascar). *Malagasy Nature* 2: 111–126.

Child, B. and Barnes, G. 2010. The conceptual evolution and practice of community-based natural resource management in southern Africa: past, present and future. *Environmental Conservation* 37: 283–295.

Christie, I.T. and Crompton, E.D. 2003. *Republic of Madagascar: Tourism Sector Study*. World Bank, Washington DC.

Cobb, S., Ginsberg, J. and Thomsen, J. 2007. Conservation in the tropics: evolving roles for governments, international donors and non-government organisations. In *Key Topics in Conservation Biology*, D.W. Macdonald and K. Service (Eds.), pp. 145–155. Blackwell Publishing, Oxford.

Codjoe, S.N.A. and Bilsborrow, R.E. 2012. Are migrants exceptional resource degraders? A study of agricultural households in Ghana. *GeoJournal* 77: 681–694.

Coetzee, B.W.T., Gaston, K.J. and Chown, S.L. 2014. Local scale comparisons of biodiversity as a test for global protected area ecological performance: a meta-analysis. *PLoS ONE* 9: e105824

Coffin, M.F. and Rabinowitz, P.D. 1987. Reconstruction of Madagascar and Africa: evidence from the Davie Fracture Zone and Western Somali Basin. *Journal of Geophysical Research* 92: 9385–9406.

Commission SAPM 2006. *Système d'Aires Protégées Malagasy: Orientations Générales sur les Catégories et les Types de Gouvernance*. SAPM Commission, Antananarivo.

Corson, C. 2011. Territorialization, enclosure and neoliberalism: non-state influence in struggles over Madagascar's forests. *Journal of Peasant Studies* 38: 703–726.

Corson, C. 2012. From rhetoric to practice: how high-profile politics impeded community consultation in Madagascar's new protected areas. *Society & Natural Resources* 25: 336–351.

Corson, C. 2014. Conservation politics in Madagascar: the expansion of protected areas. In *Conservation and Environmental Management in Madagascar*, I.R. Scales (Ed.), pp. 193–215. Routledge, Abingdon.

Crowley, B.E. 2010. A refined chronology of prehistoric Madagascar and the demise of the megafauna. *Quaternary Science Reviews* 29: 2591–2603.

Crutzen, P.J. and Steffen, W. 2003. How long have we been in the Anthropocene era? *Climatic Change* 61: 251–257.

Curran, S. 2002. Migration, social capital, and the environment: considering migrant selectivity and networks in relation to coastal ecosystems. *Population and Development Review* 28: 89–125.

Daily, G.C., and Ehrlich, P.R. 1999. Managing Earth's ecosystems: an interdisciplinary challenge. *Ecosystems* 2: 277–280.

Dearden, P., Bennett, M. and Johnston, J. 2005. Trends in global protected area governance, 1992–2002. *Environmental Management* 36: 89–100.

DeFries, R., Hansen, A., Newton, A.C. and Hansen, M.C. 2005. Increasing isolation of protected areas in tropical forests over the past twenty years. *Ecological Applications* 15: 19–26.

Dewar, R.E. 2014. Early human settlers and their impact on Madagascar's landscapes. In *Conservation and Environmental Management in Madagascar*, I.R. Scales (Ed.), pp. 44–64. Routledge, Abingdon.

Dewar, R.E. and Richard, A.F. 2012. Madagascar: a history of arrivals, what happened, and will happen next. *Annual Review of Anthropology* 41: 495–517.

Dewar, R.E., Radimilahy, C., Wright, H.T., Jacobs, Z., Kelly, G.O. and Berna, F. 2013. Stone tools and foraging in Madagascar challenge Holocene extinction models. *Proceedings of the National Academy of Sciences, USA* 110: 12583–12588.

Diamond, J.M. 1984. 'Normal' extinction of isolated populations. In *Extinctions*, M.H. Nitecki (Ed.), pp. 191–246. University of Chicago Press, Chicago.

Dressler, W., Buscher, B., Schoon, M., Brockington, D., Hayes, T., Kull, C.A., McCarthy, J. and Shrestha, K. 2010. From hope to crisis and back again? A critical history of the global CBNRM narrative. *Environmental Conservation* 37: 5–15.

Dudley, N. (Ed.) 2008. *Guidelines for Applying Protected Area Management Categories*. IUCN, Gland.

Dudley, N., Higgins-Zogib, L. and Mansourian, S. 2009. The links between protected areas, faiths, and sacred natural sites. *Conservation Biology* 23: 568–577.

Dudley, N., Parrish, J.D., Redford, K.H. and Stolton, S. 2010. The revised IUCN protected area management categories: the debate and ways forward. *Oryx* 44: 485–490.

Dudley, N., Groves, C., Redford, K.H. and Stolton, S. 2014. Where now for protected areas? Setting the stage for the 2014 World Parks Congress. *Oryx*: doi:10.1017/S0030605314000519.

Duffy, R. 2006. Non-governmental organizations and governance states: the impact of transnational environmental management networks in Madagascar. *Environmental Politics* 15: 731–749.

Duffy, R. 2010. *Nature Crime: How We're Getting Conservation Wrong*. Yale University Press, New Haven and London.

Durbin, J.C. and Ralambo, J.A. 1994. The role of local people in the successful maintenance of protected areas in Madagascar. *Environmental Conservation* 21: 115–120.

Ehrenfeld, D.W. 1992. Conservation biology: its origins and definition. *Science* 255: 1625–1626.

Ehrlich, P.R. and Pringle, R.M. 2008. Where does biodiversity go from here? A grim business-as-usual forecast and a hopeful portfolio of partial solutions. *Proceedings of the National Academy of Sciences, USA* 105: 11579–11586.

Evers, S. and van der Zwan, N. 1998. *Madagascar: the Zebu as Guide through Past and Present*. Africa Museum, Berg en Dal.

Ewers, R.M. and Rodrigues, A.S.L. 2008. Estimates of reserve effectiveness are confounded by leakage. *Trends in Ecology and Evolution* 23: 113–116.

Fa, J.E, Funk, S.M. and O'Connell, D. 2011. *Zoo Conservation Biology*. Cambridge University Press, Cambridge.

Fenn, M.D. 2003. The spiny forest ecoregion. In *The Natural History of Madagascar*, S.M. Goodman and J.P. Benstead (Eds.), pp. 1525–1529. University of Chicago Press, Chicago.

Fenn, M. and Rebara, F. 2003. Present migration tendencies and their impacts in Madagascar's spiny forest ecoregion. *Nomadic Peoples* 7: 123–137.

Ferguson, B. 2010. Madagascar. In *REDD, Forest Governance and Rural Livelihoods: the Emerging Agenda*, O. Springate-Baginski and E. Wollenberg (Eds.), pp. 135–171. CIFOR, Bogor.

Ferguson, B., Gardner, C.J., Andriamarovololona, M.M., Healy, T., Muttenzer, F., Smith, S., Hockley, N. and Gingembre, M. 2014. Governing ancestral land in Madagascar: have policy reforms contributed to social justice? In *Governance for Justice and Environmental Sustainability: Lessons across Natural Resource Sectors in Sub-Saharan Africa*, M. Sowman and R. Wynberg (Eds.) pp. 63–93. Routledge, London.

Ferraro, P.J. and Kiss, A. 2002. Direct payments to conserve biodiversity. *Science* 298: 1718–1719.

Ferraro, P.J. and Hanauer, M.M. 2014a. Advances in measuring the environmental and social impacts of environmental programs. Annual *Review of Environment and Resources* 39: 495–517.

Ferraro, P.J. and Hanauer, M.M. 2014b. Quantifying causal mechanisms to determine how protected areas affect poverty through changes in ecosystem services and infrastructure. *Proceedings of the National Academy of Sciences, USA* 111: 4332–4337.

Ferraro, P.J., Hanauer, M.M. and Sims, K.R.E. 2011. Conditions associated with protected area success in conservation and poverty reduction. *Proceedings of the National Academy of Sciences, USA* 108: 13913–13918.

Ferraro, P., Hanauer, M.M., Miteva, D.A., Canavire-Bacarreza, J.V., Pattanayak, S.K. and Sims, K.R.E. 2013. More strictly protected areas are not necessarily more protective: evidence from Bolivia, Costa Rica, Indonesia and Thailand. *Environmental Research Letters* 8: 025011.

Fisher, B. and Christopher, T. 2007. Poverty and biodiversity: measuring the overlap of human poverty and the biodiversity hotspots. *Ecological Economics* 62: 93–101.

Fox, H.E., Christian, C., Nordby, J.C., Pergams, O.R.W., Peterson, G.D. and Pyke, C.R. 2006. Perceived barriers to integrating social science and conservation. *Conservation Biology* 20: 1817–1820.

Freudenberger K. 2010. *Paradise Lost? Lessons from 25 years of USAID Environment Programs in Madagascar*. International Resources Group, Washington DC.

Fritz-Vietta, N.V.M., Ferguson, H.B., Stoll-Kleemann, S. and Ganzhorn, J.U. 2011. Conservation in a biodiversity hotspot: insights from cultural and community perspectives in Madagascar. In *Biodiversity Hotspots: Distribution and Protection of Conservation Priority Areas*, F.E. Zachos and J.C. Habel (Eds.), pp. 209–233. Springer, Heidelberg and Berlin.

Gade, D.W. 2008. Considering Madagascar's deforestation story in paradigmatic and reflexive terms. In *Greening the Great Red Island: Madagascar in Nature and Culture*, J.C. Kaufmann (Ed.), pp. 93–111. Africa Institute of South Africa, Pretoria.

Garcia, G. and Goodman, S.M. 2003. Hunting of protected animals in the Parc National d'Ankarafantsika, north-western Madagascar. *Oryx* 37: 115–118.

Gardner, C.J. 2009. A review of the impacts of anthropogenic habitat change on terrestrial biodiversity in Madagascar: implications for the design and management of new protected areas. *Malagasy Nature* 2: 2–29.

Gardner, C.J. 2011. IUCN management categories fail to represent new, multiple-use protected areas in Madagascar. *Oryx* 45: 336–346.

Gardner, C.J. 2012. Social learning and the researcher-practitioner divide. *Oryx* 46: 313–314.

Gardner, C.J., Ferguson, B., Rebara, F. and Ratsifandrihamanana, A.N. 2008. Integrating traditional values and management regimes into Madagascar's expanded protected area system: the case of Ankodida. In *Protected Landscapes and Cultural and Spiritual Values*, J-M. Mallarach (Ed.), pp. 92–103. IUCN, GTZ and Obra Social de Caixa Catalunya. Kasparek Verlag, Heidelberg.

Gardner, C.J., Kidney, D. and Thomas, H. 2009a. First comprehensive avifaunal survey of PK32-Ranobe, a new protected area in south-western Madagascar. *Phelsuma* 17: 20–39.

Gardner, C.J., Fanning, E., Thomas, H. and Kidney, D. 2009b. The lemur diversity of the Fiherenana-Manombo Complex, southwest Madagascar. *Madagascar Conservation and Development* 4: 38–43.

Gardner, C.J., Nicoll, M.E., Mbohoahy, T., Oleson, K.L.L., Ratsifandrihamanana, A.N., Ratsirarson, J., René de Roland, L.-A., Virah-Sawmy, M., Zafindrasilivonona, B. and Davies, Z.G. 2013. Protected areas for conservation and poverty alleviation: experiences from Madagascar. *Journal of Applied Ecology* 50: 1289–1294.

Gaston, K.J., Jackson, S.F., Cantu-Salazar, L. and Cruz-Pinon, G. 2008. The ecological performance of protected areas. *Annual Review of Ecology, Evolution, and Systematics* 39: 93–113.

Gaveau, D.L.A., Wandono, H. and Setiabudi, F. 2007. Three decades of deforestation in southwest Sumatra: have protected areas halted forest loss and logging, and promoted regrowth? *Biological Conservation* 134: 495–504.

Geldmann, J., Barnes, M., Coad, L., Craigie, I.D., Hockings, M. and Burgess, N. 2013. Effectiveness of terrestrial protected areas in reducing habitat loss and population declines. *Biological Conservation* 161: 230–238.

Geoghegan, T. and Renard, Y. 2002. Beyond community involvement in protected area planning and management: lessons from the insular Caribbean. *Parks* 12: 16–27.

Golden, C.D. 2009. Bushmeat hunting and use in the Makira Forest, north-eastern Madagascar: a conservation and livelihoods issue. *Oryx* 43: 386–392.

GoM (Government of Madagascar). 1996. *Loi n° 96-125 Relative à la Gestion Locale des Ressources Naturelles Renouvelables*. Repoblikan'i Madagasikara, Antananarivo.

GoM (Government of Madagascar). 2001. *Loi N° 2001/05 Portant Code de Gestion des Aires Protégées*. Repoblikan'i Madagasikara, Antananarivo.

Gommery, D., Ramanivosoa, B., Faure, M., Guérin, C., Kerloc'h, P. Sénégas, F. and Randrianantenaina, H. 2011. Les plus anciennes traces d'activités anthropiques de Madagascar sur des ossements d'hippopotames subfossiles d'Anjohibe (Province de Mahajanga). *Comptes Rendus Palevol* 10: 271–278.

Goodman, S.M. 2006. Hunting of microchiroptera in south-western Madagascar. *Oryx* 40: 225–228.

Goodman, S.M. 2012. *Les Carnivora de Madagascar*. Association Vahatra, Antananarivo.

Goodman, S.M. and Raselimanana, A. 2003. Hunting of wild animals by Sakalava of the Menabe region: a field report from Kirindy-Mite. *Lemur News* 8: 4–6.

Goodman, S.M. and Benstead, J.P. 2005. Updated estimates of biotic diversity and endemism for Madagascar. *Oryx* 39: 73–77.

Goodman, S.M. and Jungers, W.L. 2013. *Les Animaux et Ecosystèmes de l'Holocène Disparus de Madagascar*. Association Vahatra, Antananarivo.

Goodman, S.M. and Raherilalao, M.-J. 2013. *Atlas of Selected Land Vertebrates of Madagascar*. Association Vahatra, Antananarivo.

Goodman, S.M., Thomas, H. and Kidney, D. 2005. The rediscovery of *Mungotictis decemlineata lineata* Pocock, 1915 (Carnivora: Eupleridae) in southwestern Madagascar: insights into its taxonomic status and distribution. *Small Carnivore Conservation* 33: 1–5.

Gorenflo, L.J., Corson, C., Chomitz, K.M., Harper, G., Honzák, M. and Özler, B. 2011. Exploring the association between people and deforestation in Madagascar. In *Human Population: its Influences on Biological Diversity*, R.P. Cincotta and L.J. Gorenflo (Eds.), pp.197–221. Springer, Berlin.

Hall, J.M., Burgess, N.D., Rantala, S., Vihemäki, H., Jambiya, G., Gereau, R.E., Makonda, F., Njilima, F., Sumbi, P. and Kizaji, A. 2014. Ecological and social outcomes of a new protected area in Tanzania. *Conservation Biology*: doi: 10.1111/cobi.12335.

Harper, G.J., Steininger, M.K., Tucker, C.J., Juhn, D. and Hawkins, F. 2007. Fifty years of deforestation and forest fragmentation in Madagascar. *Environmental Conservation* 34: 325–333.

Hawkins, A.F.A., Hawkins, C.E. and Jenkins, P.D. 2000. *Mungotictis decemlineata lineata* (Carnivora: Herpestidae), a mysterious Malagasy mongoose. *Journal of Natural History* 34: 305–310.

Henkels, D.M. 1999. Une vue de près du droit de l'environnement malgache. *African Studies Quarterly* 3: 39–59.

Hilborn, R. and Ludwig, D. 1993. The limits of applied ecology. *Ecological Applications* 3: 550–552.

Hockley, N.J. and Andriamarovololona, M.M. 2007. *The Economics of Community Forest Management in Madagascar: is there a Free Lunch?* USAID Madagascar, Antananarivo.

Hoekstra, J.M., Boucher, T.M., Ricketts, T.H. and Roberts, C. 2005. Confronting a biome crisis: global disparities of habitat loss and protection. *Ecology Letters* 8: 23–29.

Holt, B.G., Lessard, J.-P., Borregaard, M.K., Fritz, S.A., Araújo, M.B., Dimitrov, D., Fabre, P.-H., Graham, C.H., Graves, G.R., Jønsson, K.A., *et al*. 2013. An update of Wallace's zoogeographic regions of the world. *Science* 339: 74–78.

Horning, N.R. 2003. How rules affect conservation outcomes. In *The Natural History of Madagascar*, S.M. Goodman and J.P. Benstead (Eds.), pp. 146–153. University of Chicago Press, Chicago.

Horning, N.R. 2008. Strong support for weak performance: donor competition in Madagascar. *African Affairs* 107: 405–431.

Hulme, D. and Murphree, M. 1999. Communities, wildlife and the 'new conservation' in Africa. *Journal of International Development* 11: 277–285.

Humbert, H. 1927. La destruction d'une flore indigène par le feu: principaux aspects de la végétation malgache. *Mémoires de l'Académie Malgache* 5: 1–78.

Hutton, J., Adams, W.M. and Murombedzi, J.C. 2005. Back to the barriers? Changing narratives in biodiversity conservation. *Forum for Development Studies* 2: 341–370.

Irwin, M.T., Wright, P.C., Birkinshaw, C., Fisher, B., Gardner, C.J., Glos, J., Goodman, S.M., Loiselle, P., Rabeson, P., Raharison, J.-L., *et al*. 2010. Patterns of species change in anthropogenically disturbed habitats of Madagascar. *Biological Conservation* 142: 2351–2362.

IUCN (International Union for Conservation of Nature). 2005. *Benefits Beyond Boundaries: Proceedings of the Vth IUCN World Parks Congress*. IUCN, Gland.

IUCN 2011. *IUCN Red List of Threatened Species*. Accessible at www.iucnredlist.org.

Jarosz, L. 1993. Defining and explaining tropical deforestation: shifting cultivation and population growth in Colonial Madagascar (1896-1940). *Economic Geography* 69: 366–379.

Jasper, L.D. and Gardner, C.J. 2015. *Life Amongst the Thorns: Biodiversity and Conservation of Madagascar's Spiny Forest*. John Beaufoy, Oxford.

Jenkins, C.N. and Joppa, L. 2009. Expansion of the global terrestrial protected area system. *Biological Conservation* 142: 2166–2174.

Jenkins, R.K.B., Rabearivelo, A., Chan, C.T., Andre, W.M., Randrianavelona, R. and Randrianantoandro, J.C. 2009. The harvest of endemic amphibians for food in eastern Madagascar. *Tropical Conservation Science* 2: 25–33.

Jones, J.P.G., Andriahajaina, F.B., Ranambinintsoa, E.H., Hockley, N.J. and Ravoahangimalala, O. 2006. The economic importance of freshwater crayfish harvesting in Madagascar and the potential of community-based conservation to improve management. *Oryx* 40: 168–175.

Joppa, L.N. and Pfaff, A. 2009. High and far: biases in the location of protected areas. *PLoS ONE* 4: e8273

Joppa, L.N. and Pfaff, A. 2011. Global protected area impacts. *Proceedings of the Royal Society* B 278: 1633–1638.

Kaimowitz, D. and Sheil, D. 2007. Conserving what and for whom? Why conservation should help meet basic human needs in the tropics. *Biotropica* 39: 567–574.

Karanth, K.P., Delefosse, T., Rakotosamimanana, B., Parsons, T.J. and Yoder, A. 2005. Ancient DNA from giant extinct lemurs confirms single origin of Malagasy primates. *Proceedings of the National Academy of Sciences, USA* 102: 5090–5095.

Kareiva, P. and Marvier, M. 2012. What is conservation science? *BioScience* 62: 962–969.

Katz, E. 2000. Social capital and natural capital: a comparative analysis of land tenure and natural resource management in Guatemala. *Land Economics* 76: 114–132.

Kaufmann, J.C. 1998. The cactus was our kin: pastoralism in the spiny desert of southern Madagascar. In *Changing Nomads in a Changing World*, J. Ginat and A.M. Khazanov (Eds), pp. 124–142. Sussex Academic, Brighton.

Kaufmann, J.C. and Tsirahamba, S. 2006. Forests and thorns: conditions of change affecting Mahafale pastoralists in southwestern Madagascar. *Conservation and Society* 4: 231–261.

Keller, E. 2008. The banana plant and the moon: conservation and the Malagasy ethos of life in Masoala, Madagascar. *American Ethnologist* 35: 650–664.

Kiefer, I., Lopez, P., Ramiarison, C., Barthlott, W. and Ibish, P.L. 2010. Development, biodiversity conservation and global change in Madagascar. In *Interdependence of Biodiversity and Development Under Global Change*, P.L. Ibisch, A. Vega E. and T.M. Hermann (Eds.), pp. 58–81. Secretariat of the Convention on Biological Diversity, Montreal.

Kindermann, G., Obersteiner, M., Sohngen, B., Sathaye, J., Andrasko, K., Rametsteiner, E., Schlamadinger, B., Wunder, S. and Beach, R. 2008. Global cost estimates of reducing carbon emissions through avoided deforestation. *Proceedings of the National Academy of Sciences, USA* 105: 10302–10307.

Kramer, R.A., Richter, D.D., Pattanayak, S. and Sharma, N.P. 1997. Ecological and economic analysis of watershed protection in eastern Madagascar. *Journal of Environmental Management* 49: 277–295.

Kremen, C., Razafimahatratra, V., Guillery, R.P., Rakotomalala, J., Weiss, A. and Ratsisompatrivo, J.-S. 1999. Designing the Masoala National Park in Madagascar based on biological and socioeconomic data. *Conservation Biology* 13: 1055–1068.

Kremen, C., Cameron, A., Moilanen, A. Phillips, S.J., Thomas, C.D., Beentje, H., Dransfield, J., Fisher, B.L., Glaw, F., Good, T.C., *et al*. 2008. Aligning conservation priorities across taxa in Madagascar with high-resolution planning tools. *Science* 320: 222 –226.

Kull, C.A. 1996. The evolution of conservation efforts in Madagascar. *International Environmental Affairs* 8: 50–86.

Kull, C.A. 1999. Observations on repressive environmental policies and landscape burning strategies in Madagascar. *African Studies Quarterly* 3: 1–7.

Kull, C.A. 2002a. Empowering pyromaniacs in Madagascar: ideology and legitimacy in community-based natural resource management. *Development and Change* 33: 57–78.

Kull, C.A. 2002b. Madagascar aflame: landscape burning as peasant protest, resistance, or a resource management tool? *Political Geography* 21: 927–953.

Kull, C.A. 2014. The roots, persistence, and character of Madagascar's conservation boom. In *Conservation and Environmental Management in Madagascar*, I.R. Scales (Ed.), pp. 146–171. Routledge, Abingdon.

Leisher, C., Touval, J., Hess, S.M., Boucher, T.M. and Reymondin, L. 2013. Land and forest degradation inside protected areas in Latin America. *Diversity* 5: 779–795.

Lele, S., Wilshusen, P., Brockington, D., Seidler, R. and Bawa, K. 2010. Beyond exclusion: alternative approaches to biodiversity conservation in the developing tropics. *Current Opinion in Environmental Sustainability* 2: 94–100.

Leroux, S.J., Krawchuk, M.A., Schmiegelow, F., Cumming, S.G., Lisgo, K., Anderson, L.G. and Petkova, M. 2010. Global protected areas and IUCN designations: do the categories match the conditions? *Biological Conservation* 143: 609–616.

Leverington, F., Lemos Costa, K., Pavese, H., Lisle, A. and Hockings, M. 2010. A global analysis of protected area management effectiveness. *Environmental Management* 46: 685–698.

Locke, H. and Dearden, P. 2005. Rethinking protected area categories and the new paradigm. *Environmental Conservation* 32: 1–10.

Loucks, C., Ricketts, T.H., Naidoo, R., Lamoreux, J. and Hoekstra, J. 2008. Explaining the global pattern or protected area coverage: relative importance of vertebrate biodiversity, human activities and agricultural suitability. *Journal of Biogeography* 35: 1337–1348.

Lyon, L.M. and Hardesty, L.H. 2005. Traditional healing in the contemporary life of the Antanosy people of Madagascar. *Ethnobotany Research & Applications* 3: 287–294.

Maina, J., de Moel, H., Vermaaqt, J.E., Bruggemann, J.H., Guillaume, M.M.M., Grove, C.A., Madin, J.S., Mertz-Kraus, R. and Zinke, J. 2012. Linking coral river runoff proxies with climate variability, variability, hydrology and land-use in Madagascar catchments. *Marine Pollution Bulletin* 64: 2047–2059.

Marie, C.N., Sibelet, N., Dulcire, M., Rafalimaro, M., Danthu, P. and Carrière, S.M. 2009. Taking into account local practices and indigenous knowledge in an emergency conservation context in Madagascar. *Biodiversity and Conservation* 18: 2759–2777.

Mascia, M.B., Brosius, J.P., Dobson, T.A., Forbes, B.C., Horowitz, L., McKean, M.A. and Turner, N.J. 2003. Conservation and the social sciences. *Conservation Biology* 17: 649–650.

Max-Neef, M.A. 2005. Foundations of transdisciplinarity. *Ecological Economics* 53: 5–16.

McConnell, W.J. and Kull, C.A. 2014a. Deforestation in Madagascar: debates over the island's forest cover and challenges of measuring forest change. In *Conservation and Environmental Management in Madagascar*, I.R. Scales (Ed.), pp. 65–104. Routledge, Abingdon.

McConnell, W.J. and Kull, C.A. 2014b. Protecting lemurs: Madagascar's forests. *Science* 344: 358.

McDonald, R.I. and Boucher, T.M. 2011. Global development and the future of the protected area strategy. *Biological Conservation* 144: 383–392.

MEA (Millennium Ecosystem Assessment). 2005. *Ecosystems and Human Well-being: Synthesis*. Island Press, Washington DC.

Meine, C. 2010. Conservation biology: past and present. In *Conservation Biology for All*, N.S. Sodhi and P.R. Ehrlich (Eds.), pp. 7–26. Oxford University Press, Oxford.

Meine, C., Soulé, M. and Noss, R. 2006. "A mission-driven discipline": the growth of conservation biology. *Conservation Biology* 20: 631–651.

Mercier, J.-R. 2006. The preparation of the National Environmental Action Plan (NEAP): was it a false start? *Madagascar Conservation & Development* 1: 50–54.

Milner-Gulland, E.J. 2012. Interactions between human behaviour and ecological systems. *Philosophical Transactions of the Royal Society, B* 367: 270–278.

Minten, B. and Méral, P. 2006. *Commerce et Environnement: Impacts de l'Exportation de Maïs sur les Forêts d'Epineux du Sud-ouest Malgache*. WWF MWIOPO, Antananarivo.

Minten, B., Sander, K. and Stifel, D. 2012. Forest management and economic rents: evidence from the charcoal trade in Madagascar. *Energy for Sustainable Development* 17: 106–115.

Miteva, D.A., Pattanayak, S.K. and Ferraro, P.J. 2012. Evaluation of biodiversity policy instruments: what works and what doesn't? *Oxford Review of Economic Policy* 28: 69–92.

Mittermeier, R.A., Robles-Gil, P. and Mittermeier, C.G. (Eds.) 1997. *Megadiversity: Earth's Biologically Wealthiest Nations*. CEMEX, Monterrey.

Mittermeier, R.A., Hawkins, A.F.A., Rajaobelina, S. and Langrand, O. 2005 Wilderness conservation in a biodiversity hotspot. *International Journal of Wilderness* 11: 42–45.

Myers, N., Mittermeier, R.A., Mittermeier, C.G., da Fonseca, G.A.B. and Kent, J. 2000. Biodiversity hotspots for conservation priorities. *Nature* 403: 853–858.

Naughton-Treves, L., Buck, M. and Brandon, K. 2005. The role of protected areas in conserving biodiversity and sustaining local livelihoods. *Annual Review of Environment and Resources* 30: 219–252.

Nelson, A. and Chomitz, K.M. 2009. *Protected Area Effectiveness in Reducing Tropical Deforestation: a Global Analysis of the Impact of Protection Status*. The World Bank, Washington DC.

Nelson, A. and Chomitz, K.M. 2011. Effectiveness of strict vs. multiple use protected areas in reducing tropical forest fires: a global analysis using matching methods. *PLoS ONE* 6: e22722.

Newmark, W.D. and Hough, J.L. 2000. Conserving wildlife in Africa: integrated conservation and development projects and beyond. *BioScience* 50: 585–592.

Norris, S. 2006. Madagascar defiant. *BioScience* 56: 960–965.

Norscia, I. and Borbognini-Tarli, S.M. 2006. Ethnobotanical reputation of plant species from two forests of Madagascar: a preliminary investigation. *South African Journal of Botany* 72: 656–660.

Noss, R.F. 1999. Is there a special conservation biology? *Ecography* 22: 113–122.

Noss, R.F. 2007. Values are a good thing in conservation biology. *Conservation Biology* 21: 18–20.

Novy, J.W. 1997. Medicinal plants of the eastern region of Madagascar. *Journal of Ethnopharmacology* 55: 119–126.

Olson, D. and Dinerstein, E. 1998. The Global 200: a representation approach to conserving the world's most biologically valuable ecoregions. *Conservation Biology* 12: 502–515.

ONE (Office National pour l'Environnement), DGF (Direction Générale des Forêts), FTM (Foiben-Taosarintanin'i Madagascar), MNP (Madagascar National Parks) and Conservation International. 2013. *Evolution de la Couverture des Forêts Naturelles à Madagascar 2005-2010*. Office National pour l'Environnement, Antananarivo

Ostrom, E. 2009. A general framework for analyzing sustainability of social-ecological systems. *Science* 325 : 419–422.

Ostrom, E., Burger, J., Field, C.B., Norgaard, R.B. and Policansky, D. 1999. Sustainability – revisiting the commons: local lessons, global challenges. *Science* 284: 278–282.

Partage 2008. *Analyse de l'Offre et de la Demande en Energie Domestique au Niveau de la Commune de Toliara*. Partage, Antananarivo.

Peres, C.A., Barlow, J. and Laurance, W.F. 2006. Detecting anthropogenic disturbance in tropical forests. *Trends in Ecology and Evolution* 21: 227–229.

Perrier de la Bâthie, H. 1921. La végétation malgache. *Annales du Musée Colonial de Marseille* 9: 1–266.

Phillips, A. 2004. The history of the international system of protected area management categories. *Parks* 14: 4–14.

Phillipson, P. 1996. Endemism and non-endemism in the flora of south-west Madagascar. In *Biogéographie de Madagascar*, W. Lourenço (Ed.), pp. 125–136. Editions ORSTOM, Paris.

Pimm, S.L. and Raven, P. 2000. Biodiversity: extinction by numbers. *Nature* 403: 843–845.

Pollini, J. 2010. Environmental degradation narratives in Madagascar: from colonial hegemonies to humanist revisionism. *Geoforum* 41: 711–722.

Pollini, J. 2011. The difficult reconciliation of conservation and development objectives: the case of the Malagasy environmental action plan. *Human Organisation* 70: 74–87.

Pollini, J. and Lassoie, J.P. 2011. Trapping farmer communities within global environmental regimes: the case of the GELOSE legislation in Madagascar. *Society and Natural Resources* 24: 814–830.

Pollini, J., Hockley, N., Muttenzer, F.D. and Ramamonjisoa, B.S. 2014. The transfer of natural resource management rights to local communities. In *Conservation and Environmental Management in Madagascar*, I.R. Scales (Ed.), pp. 172–192. Routledge, Abingdon.

Pressey, R.L. 1994. *Ad hoc* reservations: forward or backward steps in developing representative reserve systems? *Conservation Biology* 8: 662–668.

Pullin, A.S., Bangpan, M., Dalrymple, S., Dickson, K., Haddaway, N.R., Healey, J.R., Hauari, H., Hockley, N., Jones, J.P.G., Knight, T. *et al*. 2013. Human well-being impacts of terrestrial protected areas. *Environmental Evidence* 2: 19.

Raherilalao, M.-J. and Goodman, S.M. 2011. *Histoire Naturelle des Familles et Sous-familles Endémiques d'Oiseaux de Madagascar*. Association Vahatra, Antananarivo.

Raik, D. 2007. Forest management in Madagascar: an historical overview. *Madagascar Conservation & Development* 2: 5–10.

Raik, D.B. and Decker, D.J. 2007. A multisector framework for assessing community-based forest management: lessons from Madagascar. *Ecology and Society* 12: 14.

Rakotondravony, H.A. 2006. Communautés locales et gibiers dans la région de Daraina, extrême nord-est de Madagascar. *Madagascar Conservation & Development* 1: 19–21.

Randrianandianina, B.N., Andriamahaly, L.R., Harisoa, F.M. and Nicoll, M.E. 2003. The role of protected areas in the management of the island's biodiversity. In *The Natural History of Madagascar*, S.M. Goodman and J.P. Benstead (Eds.), pp. 1423–1432. University of Chicago Press, Chicago.

Randrianandrianina, F.H., Racey, P.A. and Jenkins, R.K.B. 2010. Hunting and consumption of mammals and birds by people in urban areas of western Madagascar. *Oryx* 44: 411–415.

Randrianarivelojosia, M., Rasidimanana, V.T., Rabarison, H., Cheplogoi, P.K., Ratsimbason, M., Mulholland, D.A. and Mauclère, P. 2003. Plants traditionally prescribed to treat *tazo* (malaria) in the eastern region of Madagascar. *Malaria Journal* 2003 2: 25.

Rasoavahiny, L., Andrianarisata, M., Razafimpahanana, A. and Ratsifandrihamanana, A.N. 2008. Conducting an ecological gap analysis for the new Madagascar protected area system. *Parks* 17: 12–21.

Rasolofo, M.V. 1997. Use of mangroves by traditional fishermen in Madagascar. *Mangroves and Salt Marshes* 1: 243–253.

Razanaka, S., Grouzis, P., Milleville, P., Moizo, B. and Aubry, C. (Eds.). 2001. *Sociétés Paysannes, Transitions Agraires et Dynamiques Ecologiques dans le Sud-ouest de Madagascar*. CNRE-IRD, Antananarivo.

Réau, B. 2002. Burning for zebu: the complexity of deforestation issues in western Madagascar. *Norwegian Journal of Geography* 56: 219–229.

Redford, K.H., Levy, M.A., Sanderson, E.W. and de Sherbinin, A. 2008. What is the role for conservation organisations in poverty alleviation in the world's wild places? *Oryx* 42: 516–528.

Rejo-Fienena, F. 1995. *Etude Phytosociologique de la Végétation de la Région de Tuléar (Madagascar) et Gestion des Ressources Végétales par les Populations Locales (Cas du P.K. 32)*. PhD Thesis, Muséum National d'Histoire Naturelle, Paris.

Rodrigues, A.S.L., Andelman, S.J., Bakarr, M.I., Boitani, L., Brooks, T.M., Cowling, R.M., Fishpool, L.D.C., da Fonseca, G.A.B., Gaston, K.J., Hoffmann, M., *et al*. 2004a. Effectiveness of the global protected area network in representing species diversity. *Nature* 428: 640–643.

Rodrigues, A.S.L., Akçakaya, H.R., Andelman, S.J., Bakarr, M.I., Boitani, L., Brooks, T.M., Chanson, J.S., Fishpool, L.D.C., da Fonseca, G.A.B., Gaston, K.J., *et al*. 2004b. Global gap analysis: priority regions for expanding the global protected-area network. *BioScience* 54: 1092–1100.

Roe, D. 2008. The origins and evolution of the conservation-poverty debate: a review of key literature, events and policy processes. *Oryx* 42: 491–503.

Samisoa 2001. Migrations et déforestation sur le plateau calcaire de Belomotse-Vineta. In *Sociétés Paysannes, Transitions Agraires et Dynamiques Ecologiques dans le Sud-ouest de Madagascar*. S. Razanaka, M. Grouzis, P. Milleville, B. Moizo and C. Aubry (Eds), pp. 53–62. CNRE-IRD, Antananarivo.

Sandbrook, C., Adams, W.M., Büscher, B. and Vira, B. 2013. Social research and biodiversity conservation. *Conservation Biology* 27: 1487–1490.

Sandy, C. 2006. Real and imagined landscapes: land use and conservation in the Menabe. *Conservation and Society* 4: 304–324.

Scales, I.R. 2011. Farming at the forest frontier: land use and landscape change in western Madagascar, 1896–2005. *Environment and History* 17: 499–524.

Scales, I.R. 2014a. The future of conservation and development in Madagascar: time for a new paradigm? *Madagascar Conservation & Development* 9: 5–12.

Scales, I.R. 2014b. The drivers of deforestation and the complexity of land use in Madagascar. In *Conservation and Environmental Management in Madagascar*, I.R. Scales (Ed.), pp. 105–125. Routledge, Abingdon.

Scales, I.R. 2014c. A brief history of the state and the politics of natural resource use in Madagascar. In *Conservation and Environmental Management in Madagascar*, I.R. Scales (Ed.), pp. 129–145. Routledge, Abingdon.

Scharlemann, J.P.W., Kapos, V., Campbell, A., Lysenko, I., Burgess, N.D., Hansen, M.C., Gibbs, H.K., Dickson, B. and Miles, L. 2010. Securing tropical forest carbon: the contribution of protected areas to REDD. *Oryx* 44: 352–357.

Scott, L. 2006. *Chronic Poverty and the Environment: a Vulnerability Perspective*. Chronic Poverty Research Centre, Manchester.

Seddon, N., Tobias, J., Yount, J., Ramanampamonjy, J., Butchart, S. and Randrianizahana, H. 2000. Conservation issues and priorities in the Mikea Forest of south -west Madagascar. *Oryx* 34: 287–304.

Sheridan, C., Baele, J.M., Kushmaro, A., Frejaville, Y. and Eeckhaut, I. 2014. Terrestrial runoff influences white syndrome prevalence in SW Madagascar. *Marine Environmental Research*: doi: 10.1016/j.marenvres.2014.08.003

Soulé, M. 1985. What is conservation biology? *BioScience* 35: 727–734.

Stiles, D. 1991. Tubers and tenrecs: the Mikea of southwestern Madagascar. *Ethnology* 30: 251–263.

St. John, F.A.V., Keane, A. and Milner-Gulland, E.J. 2013. Effective conservation depends upon understanding human behaviour. In *Key Topics in Conservation Biology 2*, D.W. Macdonald and K.J. Willis (Eds.), pp. 344–361.Wiley-Blackwell, Chichester.

Storey M., Mahoney, J.J., Saunders, A.D., Duncan, R.A., Kelley, S.P. and Cowin, M.F. 1995. Timing of hot spot-related volcanism and the breakup of Madagascar and India. *Science* 267: 852–855.

Tang, L., Shao, G., Piao, Z., Dai, L., Jenkins, M.A., Wang, S., Wu, G., Wu, J. and Zhao, J. 2010. Forest degradation deepens around and within protected areas in East Asia. *Biological Conservation* 143: 1295–1298.

Tengö, M., Johansson, K., Rakotondrasoa, F., Lundberg, J., Andriamaherilala, J.-A., Rakotoarisoa, J.-A. and Elmqvist, T. 2007. Taboos and forest governance: Informal protection of hot spot dry forest in southern Madagascar. *Ambio* 36: 683–691.

Thomas, C.D., Cameron, A., Green, R.E., Bakkenes, M., Beaumont, L.J., Collingham, Y.C., Erasmus, B.F.N., de Siqueira, M.F., Grainger, A., Hannah, L., *et al*. 2004. Extinction risk from climate change. *Nature* 427: 145–148.

Tsirahamba, S. and Kaufmann, J.C. 2008. Absence of shade: deforestation in Mahafale forest-pastures. In *Greening the Great Red Island: Madagascar in Nature and Culture*, J. C. Kaufmann (Ed.), pp. 179–196. Africa Institute of South Africa, Pretoria.

Tucker, B. 2001. *The Behavioural Ecology and Economics of Variation, Risk, and Diversification among Mikea Forager-Farmers of Madagascar*. PhD Thesis, University of North Carolina, Chapel Hill.

Tucker, B. 2006. A future discounting explanation for the persistence of a mixed foraging -horticulture strategy among the Mikea of Madagascar. In *Behavioural Ecology and the Transition to Agriculture*, D.J. Kennett and B. Winterhalder (Eds.), pp. 22–40. University of Californis Press, Berkeley.

United Nations. 2007. *World Urbanisation Prospects: The 2007 Revision*. Population Division of the Department of Economic and Social Affairs of the United Nations Secretariat. Available from http://esa.un.org/unup.

Virah-Sawmy, M. 2009. Ecosystem management in Madagascar during global change. *Conservation Letters* 2: 163–170.

Virah-Sawmy, M., Willis, K.J. and Gillson, L. 2010. Evidence for drought and forest declines during the recent megafaunal extinctions in Madagascar. *Journal of Biogeaography* 37: 506–519.

Virah-Sawmy, M., Gardner, C.J. and Ratsifandrihamanana, A.N. 2014. The Durban Vision in practice: experiences in the participatory governance of Madagascar's new protected areas. In *Conservation and Environmental Management in Madagascar*, I.R. Scales (Ed.), pp. 216–251. Routledge, Abingdon.

Watson, R.T., René de Roland, R.-A., Rabearivony, J. and Thorstrom, R. 2007. Community-based wetland conservation protects endangered species in Madagascar: lessons from science and conservation. *Banwa* 4: 83–97.

Watson, J.E.M., Dudley, N., Segan, D.B. and Hockings, M. 2014. The performance and potential of protected areas. Nature 515: 67–73.

Wells, M.P., and McShane, T.O. 2004. Integrating protected area management with local needs and aspirations. *Ambio* 33: 513–519.

West, P. and Brockington, D. 2006. An anthropological perspective on some unexpected consequences of protected areas. *Conservation biology* 20: 609–616.

West, P., Igoe, J. and Brockington, D. 2006. Parks and peoples: the social impact of protected areas. *Annual Review of Anthropology* 35: 251–277.

White, F. 1983. *The Vegetation of Africa*. UNESCO, Paris.

Willis, K.J., Gillson, L. and Virah-Sawmy, M. 2008. Nature or nurture: the ambiguity of C_4 grasslands in Madagascar. *Journal of Biogeography* 35: 1741–1742.

Wilkie, D.S., Morelli, G.A., Demmer, J., Starkey, M., Telfer, P. and Steil, M. 2006. Parks and people: assessing the human welfare effects of establishing protected areas for biodiversity conservation. *Conservation Biology* 20: 247–249.

Wilmé, L. 1996. Composition and characteristics of bird communities in Madagascar. In *Biogéographie de Madagascar*, W. Lourenço (Ed.), pp. 349–362. Editions ORSTOM, Paris.

Wilmé, L., Goodman, S.M. and Ganzhorn, J.U. 2006. Biogeographic evolution of Madagascar's microendemic biota. *Science* 312: 1063–1065.

Wilshusen, P.R., Brechin, S.R., Fortwangler, C.L. and West, P.C. 2002. Reinventing a square wheel: critique of a resurgent "protection paradigm" in international biodiversity conservation. *Society and Natural Resources* 15: 17–40.

WWF (World Wide Fund for Nature). 2010. *Plan d'Aménagement et de Gestion de PK32 -Ranobe*. WWF Madagascar and Western Indian Ocean Programme Office, Antananarivo.

Yoder, A.D., Burns, M.M., Zehr, S., Delefosse, T., Veron, G., Goodman, S.M. and Flynn, J.J. 2003. Single origin of Malagasy Carnivora from an African ancestor. *Nature* 421: 734–737.

Zimmerer, K.S., Galt, R.E. and Buck, M.V. 2004. Globalization and multi-spatial trends in the coverage of protected-area conservation (1980-2000). *Ambio* 33: 520–529.

Chapter 2

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Protected areas for conservation and poverty alleviation: experiences from Madagascar

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Summary

1. Poverty alleviation is increasingly promoted as an objective of protected areas, but the literature remains largely theoretical and little is known regarding how it can be achieved synergistically with conservation goals.

2. We use three case studies to illustrate management approaches adopted in Madagascar's new generation of multiple-use protected areas. These differ from strictly protected sites by: i) having fewer access restrictions; ii) involving rural communities and other stakeholders in their governance, and; iii) supporting community development initiatives.

3. Approaches vary, with managers seeking to improve natural resource use (fisheries) in aquatic environments, but reduce forest resource use through the promotion of alternative livelihoods. However, the impacts of such interventions are not adequately evaluated.

4. *Synthesis and applications.* Protected area managers lack a sufficient evidence-base for decision-making. Collaboration with applied researchers is required if protected areas are to conserve biodiversity effectively and contribute to poverty alleviation. Managers, in turn, must seek to share experiences to develop best practice.

2.1 Moving beyond the 'conservation-poverty' debate towards on-the-ground implementation

Biodiversity conservation and poverty alleviation are two of the world's major challenges, and the search for synergies in the pursuit of both agendas is enshrined in their respective global policy frameworks – the Convention on Biological Diversity and the Millennium Development Goals. The 'conservation-poverty debate' has featured prominently in conservation discourses since the 1980s (Roe 2008), focusing primarily on issues such as the impact of conservation activities (particularly protected areas) on affected local communities, the role of conservation organisations in poverty alleviation, and the complex inter-relationships between biodiversity, ecosystem service provision and poverty. Much of the debate, however, has been theoretical in nature, and while it is widely acknowledged that conservationists should seek to reduce, or at least not aggravate, poverty through their actions, the literature remains sparse when it comes to illustrations of how poverty alleviation is pursued successfully in real-world conservation management. This comes at a time when there has been a substantive shift towards multiple-use protected areas, away from traditional strict reserves (Zimmerer *et al*. 2004). Indeed, 44 % of the world's protected area estate now comprises IUCN categories V and VI, which are characterised by their emphasis on sustainable extractive resource use by local communities (Jenkins and Joppa 2009). The paucity of guidelines for protected area managers tasked with achieving these twin goals is a manifestation of the researcher-practitioner divide; a well-known phenomenon to which practitioners contribute by both failing to share their experiences in open fora, and being unable to attract applied researchers to address knowledge gaps.

Here we present our experiences of actively pursuing biodiversity conservation and poverty alleviation in a rapidly expanding protected area system, using three instructive case studies. We outline the types of management interventions employed and explore the theoretical implications of our findings. Finally, we discuss priority actions required to stimulate and improve collaboration between applied researchers and managers, with the aim of instigating evidence-based protected area management.

2.2 Reconciling conservation, natural resource use and poverty alleviation in Madagascar's new multiple-use protected areas

Improving synergies between conservation and poverty alleviation is particularly important in Madagascar because not only is it amongst the world's poorest countries, it is also a leading global conservation priority (Brooks *et al*. 2006). Since 2003, the country has begun to triple the coverage of its protected area system – a process known as the Durban Vision. While the nation's first generation of protected areas, comprising 46 strictly protected sites (IUCN category Ia, II and IV) managed by the parastatal Madagascar National Parks, were principally established for biodiversity conservation, scientific research and recreation (Randrianandianina *et al*. 2003), the objectives of the expanded protected area system have been extended to incorporate maintaining the country's cultural heritage and promoting the sustainable use of natural resources for poverty alleviation and development. Almost 100 new protected areas have now been established within the Durban Vision framework, many in land- and seascapes containing large human populations that are heavily dependent on natural resources for subsistence and generating household income. Recognising this reliance, most new protected areas are designated as IUCN category V and VI multiple-use sites, in which sustainable extraction (of, for example, fuel and construction wood, non-timber forest products and bushmeat) is permitted according to a zoning plan, and are co-managed via agreements between NGOs and local community structures (Gardner 2011).

Protected areas with multiple objectives pose a huge challenge for site managers, who need to account for the interests of local communities by facilitating rural development and poverty alleviation, whilst ensuring the viability of fragile ecosystems and species. Working towards such goals has necessitated the development of new models of protected area management. Building on approaches such as integrated conservation-development projects and community-based natural resource management, the management of Madagascar's new generation of protected areas differs markedly from that of the state-managed network of strictly-protected sites. The major differences include: i) fewer access restrictions, as illustrated by the shift from strict to multiple-use protected area categories; ii) greater community participation in protected area governance, through the establishment of co-management structures and the empowerment of local users' associations; iii) an increased focus on community development activities within protected area management plans; iv) a new emphasis on the evaluation and mitigation of negative social impacts of protected area creation, with a novel (for Madagascar) legal requirement to develop a social safeguards plan, and; v) greater involvement with a diverse array of stakeholders across larger spatial scales, such as regional authorities and the private sector. The following three brief case studies (Fig. 2.1; Table 2.1), from the terrestrial, freshwater and marine realms, help illustrate the range of management approaches adopted within Madagascar's new generation of protected areas. All of them are designated as IUCN category V, defined as "a protected area where the interaction of people and nature over time has produced an area of distinct character with significant ecological, biological, cultural and scenic value: and where safeguarding the integrity of this interaction is vital to protecting and sustaining the area and its associated nature conservation and other values".

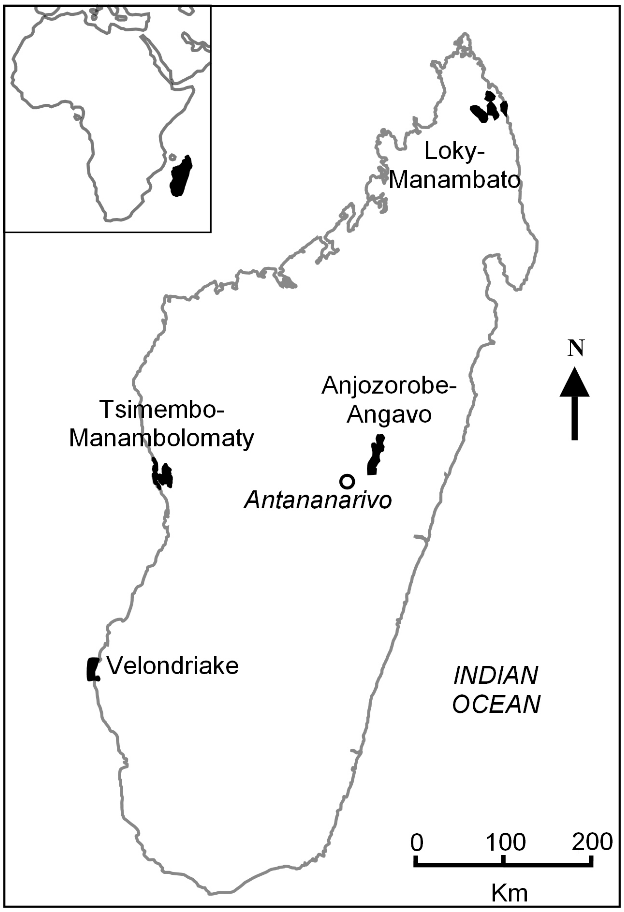

Figure 2.1 Location of the case study multiple-use protected areas in Madagascar. Inset indicates the position of Madagascar in relation to Africa.

Table 2.1 Characteristics of the case study multiple-use protected areas in Madagascar. % Pop. = percentage of population benefiting from conservation-livelihood activities.

	Anjozorobe-Angavo	**Loky-Manambato**	**Tsimembo-Manambolomaty**	**Velondriake**
Year established	2005	2005	2008	2008
Area (ha)	52,200	250,000	62,745	67,782
Human population	30,000	59,000	12,609	7260
% Pop.	20	64	75	71
Key ecosystems	Humid forest	Humid forest; deciduous dry forest; littoral forest	Freshwater wetlands; deciduous dry forest; mangroves	Coral reefs, seagrass beds; mangroves
Key species (*IUCN Red List status*)	Indri *Indri indri* (Gmelin, 1788) (*EN*); diademed sifaka *Propithecus diadema* Bennett 1832 (*EN*); Madagascar serpent eagle *Eutriorchis astur* (Sharp, 1875) (*EN*)	Golden-crowned sifaka *Propithecus tattersalli* Simons, 1988 (*EN*); Daraina sportive lemur *Lepilemur milanoii* Louis *et al*., 2006 (*DD*); white-breasted mesite *Mesitornis variegatus* Geoffroy Saint-Hilaire, 1838 (*VU*)	Madagascar fish eagle *Haliaeetus vociferoides* Des Murs, 1845 (*CR*); Madagascar teal *Anas bernieri* (Hartlaub, 1860) (*EN*); Decken's sifaka *Propithecus deckenii* A. Grandidier, 1867 (*VU*); Madagascar side-necked turtle *Erymnochelys madagascariensis* (Grandidier, 1867) (*CR*)	Five marine turtles, 18 shark species and 54 coral species on IUCN Red List

2.2.1 Anjozorobe-Angavo and Loky-Manambato

The ethos in these two protected areas, which are co-managed by the Malagasy NGO Fanamby and local community institutions, is centred on engendering innovative partnerships between communities and the private sector in order to promote development and reduce pressures on biodiversity. At Anjozorobe-Angavo, Fanamby have created Saha Forest Lodge, which is run by a professional tourism operator under an agreement with the neighbouring village. The terms of the relationship set out a land rental contract, as well as mutually determined local employment and market gardening production quotas for the hotel. At both sites, Fanamby have been exploring other entrepreneurial opportunities through organic and fair trade certification, having created a commercial venture, Sahanala, to broker markets and provide technical support to producers. Starting with ginger and red rice from Anjozorobe-Angavo and vanilla from Loky-Manambato, the enterprise has since expanded into producing essential oils and additional high value crops adapted to local growing conditions and community interests. In 2010, a deal was negotiated with Air Madagascar to provide passengers with organic-labelled cashew nuts grown by producer cooperatives associated with Loky-Manambato.

2.2.2 Tsimembo-Manambolomaty

This wetland and dry forest complex is co-managed by The Peregrine Fund (TPF) and local communities, with a focus on empowering traditional users to manage their resources more sustainably. Historically, the fishing season and permitted activities have been decided by a *tompondrano*, a local keeper of the lakes, which helped to maintain healthy fish stocks and protect the surrounding forests. An influx of migrants during the 1990s, however, resulted in the abuse of traditional rules and lead to overfishing and forest degradation that threatened local livelihoods (Watson and Rabarisoa 2000). Seeking to reinvigorate traditional practices and strengthen the capacity and power of resident communities to manage their resources, TPF and regional ministry representatives initiated the legal transfer of management rights from the state to two community users' associations, which formalised the traditional rules that existed prior to the influx of migrants. This provided the communities the legal power to ensure respect for their customs, which are vigorously enforced through the payment of fines in the traditional form (the payment of zebu cattle and rum).

The re-establishment of traditional fishing rules at Manambolomaty, such as restrictions on fishing within spawning grounds and respecting the fishing season defined by the *tompondrano*, is believed to have stabilised lake fish stocks. Total annual revenues from fishing, based on market prices for dried fish, were estimated at US$ 1,562/fisher/year in 1995, approximately 750 % of mean national income at the time (Watson and Rabarisoa 2000). Sales of fish to wholesale buyers are taxed by the site's two communes and represent an estimated 56 % of revenue (Rabearivony *et al*. 2008). Local incomes from fishing are thought to have increased as a result of community-management: although little is known about the distribution of such income within the community, its impact is illustrated by the growth of commercial activity in the village of Soatana between 2000 and 2004, during which time the number of small groceries in the village grew from one to seven. Both community management associations possess bank accounts in which income from fines, the sale of fishing and trading permits, and association membership is deposited. In turn, this finance is used to buy rice for subsidised resale to association members during the annual rice shortage season, as well as for local development micro-projects.

2.2.3 Velondriake

Velondriake is now one of the largest community-managed marine protected areas in the Indian Ocean, but grew from a single trial closure of the local economically important octopus (*Octopus cyanea*) fishery in 2004. The perceived success of the initial closure led 23 neighbouring villages to participate in the model, followed in 2006 by the creation of the formal Velondriake Management Association to govern closures (Harris 2007). The model has since spread across the nation and region. Temporary closures capitalise on the rapid growth of octopus and broad participation in the fishery: they are coordinated across the protected area, and a partnership with a seafood export company provides a guaranteed buyer when closures are opened. Preliminary evaluation of the closures over the past eight years indicate that catch per unit effort (CPUE) effects are significant and that most village's 'investment' (in terms of foregone catch during the closures) is recouped within a short period after re-opening if the closures are well-managed (K. Oleson, unpublished data). Additional management zones created following the success of the octopus management include permanent reef reserves closed to all fishing, temporary mangrove reserves, and areas for the development of aquaculture (sea cucumbers and algae) and ecotourism, while the protected area's managers have also implemented social programmes including education and population, health and environment outreach.

2.3 Generic lessons to be learnt from the Malagasy case studies

The case study protected areas share a number of characteristics. They are all: i) managed for multiple uses, so natural resource extraction is therefore permitted over much of their spatial extent; ii) either managed or co-managed by local communities and an NGO, and; iii) support initiatives with the aim of improving livelihoods through the legal or technical empowerment of local resource users. Where they differ is the way in which biodiversity is exploited in order to support local economic growth: the management of Tsimembo-Manambolomaty is concerned with enhancing the productivity and sustainability of an economically-important natural resource base, while within Anjozorobe-Angavo and Loky-Manambato the emphasis is on reducing local dependence on natural resources through the development of alternative income sources. Velondriake, meanwhile, employs both approaches, improving the management of the octopus fishery while instigating alternative livelihoods to lessen reliance on it and other fisheries resources. Notably, two of the case studies involve partnerships with the private sector aimed at adding value to local production.

Experiences from Madagascar's new generation of protected areas can feed into, and inform, the long-standing debates around the role of sustainable natural resource use in both poverty alleviation and conservation. While advocates believe that it can generate positive incentives for conservation among local communities (Rosser and Leader-Williams 2010), a dependence on economically-marginal natural resources may form a 'poverty trap', preventing users from escaping hardship (Angelsen and Wunder 2003). Indeed, Sayer (2009) suggests that significant improvements in livelihoods tend to stem only from new opportunities generated by external investments, markets and new infrastructure, rather than marginal improvements to existing livelihoods, and that "one should not focus on what the poor are doing now but on what they might do in the future in growing economies". While they may not provide a basis for development, however, it is clear that natural resources provide a critical safety net preventing many rural communities from slipping further into destitution (Kaimowitz and Sheil 2007).

For managers of these new, multiple-use protected areas seeking to reconcile conservation with the needs of local populations, the choice of which development alternatives to promote is, of course, context specific. It is noteworthy that, among our case studies, improved management of natural resources has been the objective within aquatic ecosystems, whereas the target in terrestrial protected areas has been to diminish people's use of the forest. Freshwater and marine resources are generally more rapidly renewable than trees and, critically, aquatic ecosystems cannot be "owned" and converted into productive anthropogenic systems as easily as terrestrial areas can. While the interests of users and conservationists can be closely aligned in aquatic environments – both benefit from healthy, productive ecosystems – this may be harder to achieve in forests.

If, in many tropical terrestrial environments, the use of natural resources from functioning ecosystems cannot lift people out of poverty yet acts as a critical safety net, then how can biodiversity conservation contribute meaningfully to poverty alleviation? Historically, traditional land use in Madagascar has been a hotly debated, but significant, driver of both massive deforestation and the extinction of the endemic megafauna (Dewar and Richard 2012). This has occurred without lifting rural people out of poverty, and the island remains one of the poorest nations in the world. If the country's natural capital is being depleted without an accompanying reduction in the destitution experienced by the population, it follows that it must be replaced with alternative forms of capital if poverty alleviation is to be achieved successfully. Boserup's (1965) theory of agricultural development suggests a mechanism – induced innovation – for how the required changes may occur. The basic premise is that the availability of natural resources permitting a subsistence lifestyle hinders technical advancement or intensification, but, that their absence provokes the innovation required for economic development. We believe that the evolution of land use systems will occur in any scenario; when resources run out and users must innovate in response, or if the global community, particularly the conservation movement, is prepared to provide financial and technological expertise to support the transition before they do so. Nonetheless, development strategies alone are insufficient because beneficiaries may invest their increased wealth in the continued unsustainable over-exploitation of ecosystems. Therefore poverty alleviation actions must be accompanied by robust rules, including access restrictions, if protected areas are to contribute to both conservation and development goals. Any legitimate losses or opportunity costs incurred as a result of such actions, however, must be fully and fairly compensated, and the critical importance of natural resources to rural populations as a safety net in times of hardship must be recognised. It is anticipated that the multiple-use nature of Madagascar's new generation of protected areas will allow them serve as safety nets as required, while more sustainable and productive forms of resource use are stimulated and bought to fruition.

Our first-hand experience in the establishment and management of multiple-use protected areas in Madagascar highlights the need for increased alignment with the applied research community if the combined pursuit of conservation and poverty alleviation is to have a solid foundation in evidence. The paucity of empirical quantitative and qualitative data presented in the case studies, even after ten years of the Durban Vision, draws attention to a glaring weakness of these new protected area initiatives: they do not sufficiently monitor their ecological, cultural and socio-economic impacts, either in the short- or long-term. If we fail to evaluate the outcomes of our actions, then we will not be able to maximise their effectiveness in terms of conserving biodiversity or alleviating poverty, or optimise our interventions through an adaptive management cycle. However, the design and implementation of robust monitoring programmes requires applied research capacity that may not be available to managers.

There is a clear need to improve the contribution of conservation science to the practice of protected area management since, while much research takes place within protected areas, the majority is of limited practical value in real world contexts. Given that protected areas are the predominant conservation strategy worldwide, it is amazing how little we know about how to manage them realistically. As our case studies have illustrated, approaches may focus on enhancing the management of the natural resource base, or attempting to decouple its use from development, but we know little about what works in which contexts. Local, rather than larger scale, analyses are desperately needed to inform decision-making, with more scientists adopting an applied 'problem-solving' angle to their work. This can only be achieved by actively engaging with protected area managers in order to identify and implement appropriate research agendas; academic institutions, publishers and funders all have a role to play in changing the incentive structure to encourage them to do so (Gibbons *et al*. 2011).

Protected area managers spend their time putting out fires, literally or figuratively, and have restricted time to peruse the academic literature for solutions to their challenges (Pullin *et al*. 2004). If we are to build a strong evidence base for protected area management and develop best practice, we must encourage practitioners to share their experiences, particularly their mistakes, be it through journal publications or other social learning fora. Currently, this is hampered both by institutional disincentives (practitioners are rarely rewarded for publication) and the priorities of academic journals, which favour "blue-skies" research over local case studies (Hulme 2011). However, the recent creation of fora such as *Conservation Evidence* and the Practitioner's Perspective rubric in the *Journal of Applied Ecology* testify that this need is increasingly being recognised.

As conservationists from ecological backgrounds, we also need to improve our ability to dialogue with local communities (Sayer 2009) to ensure that our strategies are as appropriate as possible. In this respect we need greater constructive collaboration with social scientists, particularly our critics, and to systematically make use of their tools and approaches in the planning of protected area management. While the Velondriake and Tsimembo-Manambolomaty case studies have demonstrated potential win-win scenarios for poverty alleviation and conservation, it is clear that the interests of conservationists and resource users will not be the same in general, and that trade-offs will be the norm. In such cases, explicit, participatory mechanisms through which both sides can debate their case and reach a resolution must be instigated (McShane *et al*. 2010). However, these honest negotiations must be informed by sound information regarding the likely ecological and social impacts of the management options being explored, in turn requiring the implementation of targeted applied research programmes. As Brockington *et al*. (2006) state, "the ultimate challenge facing conservationists today is not only to reconcile errors of the past but also to determine how to shape human interactions with nature in landscapes of which people are a part". The increased engagement of the applied research community in protected area management is critical if this challenge is to be met, both in Madagascar and globally.

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References

Angelsen, A. and Wunder, S. 2003. *Exploring the Poverty-Forestry Link: Key Concepts, Issues and Research Implications*. Center for International Forestry Research, Bogor.

Boserup, E. 1965. *The Conditions of Agricultural Growth: the Economics of Agrarian Change Under Population Pressure*. George Allen & Unwin Ltd., London.

Brockington, D., Igoe, J. and Schmidt-Soltau, K. 2006. Conservation, human rights, and poverty reduction. *Conservation Biology* 20: 250–252.

Brooks, T.M., Mittermeier, R.A., da Fonseca, G.A.B., Gerlach, J., Hoffmann, M., Lamoreux, J.F., Mittermeier, C.G., Pilgrim, J.D. and Rodrigues, A.S.L. 2006. Global biodiversity conservation priorities. *Science* 313: 58–61.

Dewar, R.E. and Richard, A.F. 2012. Madagascar: a history of arrivals, what happened, and what will happen next. *Annual Review of Anthropology* 41: 495–517.

Gardner, C.J. 2011. IUCN management categories fail to represent new, multiple-use protected areas in Madagascar. *Oryx* 45: 336–346.

Gibbons, D.W., Wilson, J.D. and Green, R.E. 2011. Using conservation science to solve conservation problems. *Journal of Applied Ecology* 48: 505–508.

Harris, A. 2007. “To live with the sea”: development of the Velondriake community-managed protected area network, southwest Madagascar. *Madagascar Conservation & Development* 2: 43–49.

Hulme, P.E. 2011. Practitioner’s perspectives: introducing a different voice in applied ecology. *Journal of Applied Ecology* 48: 1–2.

Jenkins, C.N. and Joppa, L. 2009. Expansion of the global terrestrial protected area system. *Biological Conservation* 142: 2166–2174.

Kaimowitz, D. and Sheil, D. 2007. Conserving what and for whom? Why conservation should help meet basic human needs in the tropics. *Biotropica* 39: 567–574.

McShane, T.O., Hirsh, P.D., Trung, T.C., Songorwa, A.N., Kinzig, A., Monteferri, B., Mutekanga, D., Thang, H.V., Dammert, J.L., Pulgar-Vidal, M., *et al*. 2010. Hard choices: making trade-offs between biodiversity conservation and human well-being. *Biological Conservation* 144: 966–972.

Pullin, A.S., Knight, T.M., Stone, D.A. and Charman, K. 2004. Do conservation managers use scientific evidence to support their decision-making? *Biological Conservation* 119: 245–252.

Rabearivony, J., Fanameha, E., Mampiandra, J. and Thorstrom, R. 2008. Taboos and social contracts: tools for ecosystem management – lessons from the Manambolomaty Lakes RAMSAR site, western Madagascar. *Madagascar Conservation & Development* 3: 7–16.

Randrianandianina, B.N., Andriamahaly, L.R., Harisoa, F.M. and Nicoll, M.E. 2003. The role of protected areas in the management of the island’s biodiversity. In *The Natural History of Madagascar*, S.M. Goodman and J.P. Benstead (Eds.), pp. 1423–1432. University of Chicago Press, Chicago.

Roe, D. 2008. The origins and evolution of the conservation-poverty debate: a review of key literature, events and policy processes. *Oryx* 42: 491–503.

Rosser, A.M. and Leader-Williams, N. 2010. Protection or use: a case of nuanced trade-offs? In *Trade-offs in Conservation: Deciding What to Save*, N. Leader-Williams, W.M. Adams and R.J. Smith (Eds.), pp. 135–156. Blackwell Publishing, Oxford.

Sayer, J. 2009. Can conservation and development really be integrated? *Madagascar Conservation & Development* 4: 9–12.

Watson, R.T. and Rabarisoa, R. 2000. Sakalava fishermen and Madagascar fish eagles: enhancing traditional conservation rules to control resource abuse that threatens a key breeding area for an endangered eagle. *Ostrich* 71: 2–10.

Zimmerer, K.S., Galt, R.E. and Buck, M.V. 2004. Globalization and multi-spatial trends in the coverage of protected-area conservation (1980-2000). *Ambio* 33: 520–529.

Chapter 3

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IUCN management categories fail to represent new, multiple-use protected areas in Madagascar

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Abstract

The IUCN protected area management category system provides an internationally-recognised, unifying framework for the description and classification of the world's diverse protected areas. It includes six main categories, of which category V has attracted debate because of its emphasis on the role of harmonious people-nature interactions in maintaining biodiversity within cultural landscapes. Madagascar's new generation of protected areas comprises sites mainly proposed as category V, with the joint management objectives of biodiversity conservation and the promotion of natural resource use for rural development. Here, I use a classification decision tool to investigate the categorisation of 10 new protected areas proposed as category V, and find that these sites fail to meet the criteria for any of the management classes. I suggest that category V is inappropriate for these new protected areas because their associated people-nature interactions are largely negative for biodiversity. I further argue that management of these new protected areas differs fundamentally from management of category V protected areas in Europe, and recommend the modification of the management category system to account for such distinctions.

3.1 Introduction

Protected areas form the central pillar of conservation strategies worldwide and covered at least 12.9 % of the world's land surface by 2009 (Jenkins and Joppa 2009). These areas are highly diverse in terms of their nomenclature, scale, spatial context, governance models, management objectives and approaches, and great variation therefore exists in protected areas both within and between countries. Attempts to apply a descriptive framework to this array of approaches date back to 1933 (Phillips 2004) and culminated in the *Guidelines for Protected Area Management Categories* (IUCN 1994). Revised and updated in 2008 (Dudley 2008a), the International Union for Conservation of Nature (IUCN) category system is now recognised by governments (Dillon 2004) and conservation institutions as a unifying framework for the description, definition and comparison of the world's protected areas, and its use in endorsed and encouraged by the Programme of Work on Protected Areas of the Convention on Biological Diversity (SCBD 2004). The system classifies protected areas into six main categories (Dudley 2008a; Dudley *et al*. 2010), based on their primary management objective (Table 3.1).

Of the six possible designations within the IUCN system, it is category V that has attracted much attention and debate (Locke and Dearden 2005; Mallarach *et al*. 2008; Martino 2005). The establishment of the World Commission on Protected Areas Category V Task Force, and the publication of a number of outputs intended to clarify and promote the approach, are testament to this controversy. Uniquely, category V focuses specifically on areas in which there has been a historical interaction between people and nature (Phillips 2002), which has produced the landscape characteristics that are the objects of the conservation intervention. The primary objective is to "protect and sustain important landscapes/seascapes and the associated nature conservation and other values created by interactions with humans through traditional management practices". Contrary to other categories, where the emphasis in management is placed on protecting what is seen as natural, category V "puts people at the heart of the operation - and indeed requires them to be there" (Phillips 2002, p. 5). This idea is further developed in the IUCN guidelines (Dudley 2008a, p. 21). Among the distinguishing features listed for category V "a balanced interaction between people and nature that has endured over time and still has integrity" is stated as an essential characteristic. The core management philosophy (Phillips 2002, p. 10) is to "maintain the harmonious interaction of people and nature".

Table 3.1 International Union for Conservation of Nature (IUCN) definitions of protected area categories (Dudley 2008a).

Category	Definition
Ia: Strict nature reserve	Strictly protected areas set aside to protect biodiversity and also possibly geological/geomorphological features, where human visitation, use and impacts are strictly controlled and limited to ensure protection of the conservation values. Such protected areas can serve as indispensable reference areas for scientific research and monitoring.
Ib: Wilderness area	Usually large unmodified or slightly modified areas, retaining their natural character and influence, without permanent or significant human habitation, which are protected and managed so as to preserve their natural condition.
II: National park	Large natural or near natural areas set aside to protect large-scale ecological processes, along with the complement of species and ecosystems characteristic of the area, which also provide a foundation for environmentally and culturally compatible spiritual, scientific, educational, recreational and visitor opportunities.
III: Natural monument	Set aside to protect a specific natural monument, which can be a landform, sea mount, submarine cavern, geological feature such as a cave or even a living feature such as an ancient grove. They are generally small protected areas and often have high visitor value.
IV: Species/habitat management area	Aim to protect particular species or habitats and management reflects this priority. Many category IV protected areas will need regular, active interventions to address the requirements of particular species or to maintain habitats but this is not a requirement of the category.
V: Protected landscape/seascape	Where the interaction of people and nature over time has produced an area of distinct character, with significant ecological, biological, cultural and scenic value, and where safeguarding the integrity of this interaction is vital to protecting and sustaining the area and its associated nature conservation and other values.
VI: Sustainable use area	Conserve ecosystems and habitats, together with associated cultural values and traditional natural resource management systems. They are generally large, with most of the area in a natural condition, where a proportion is under sustainable natural resource management and where low-level non-industrial use of natural resources compatible with nature conservation is seen as one of the main aims of the area.

The protected landscapes approach has been better established in Europe than elsewhere because of the continent's long history of settlement, the lack of remaining large natural areas, and the existence of many cultural landscapes with significant natural values (Phillips 2002). As a category that reflects the increasingly dominant conservation paradigm (Büscher and Whande 2007) of integrating local people into conservation initiatives and encouraging sustainable use rather than strict preservation (Locke and

Dearden 2005; Naughton-Treves *et al*. 2005; Wells and McShane 2004), the category V model is seen as "an approach whose time has come" (Phillips 2002, p. 13), and its application and management principles have been strongly promoted for adoption globally. To some extent, this mirrors the fact that most large wilderness or natural areas have already been incorporated into protected areas (Leroux *et al*. 2010), and the remaining landscapes available for new protected area creation often include substantial human populations whose needs must be incorporated into appropriate management objectives (Hutton and Leader-Williams 2003; Mallarach *et al*. 2008).

Here I seek to contribute to the process of reviewing the IUCN category system (recommended by Dudley *et al*. (2004, 2010) to ensure that protected areas can adapt and respond to global challenges) by examining the application of the designations within the context of Madagascar's rapidly expanding protected area system. I first provide a brief history of protected areas in Madagascar, before using a categorisation decision-making tool to explore the suitability of each management category for 10 case study sites of new, multiple-use protected areas. Finally, I discuss the applicability of the existing category V model to new protected areas in Madagascar and provide recommendations for the modification of the IUCN system to ensure that protected areas of this type are adequately represented.

3.2 The Madagascar protected area system

Madagascar is considered a top conservation priority, harbouring high levels of endemism at species and higher taxonomic levels (Myers *et al*. 2000). Prior to 2003 Madagascar's protected area network consisted of 46 areas managed by the parastatal Madagascar National Parks (MNP, previously ANGAP), in some cases in partnership with NGOs. With a total area of almost 1.7 million ha the network comprised three categories of protected area (Randrianandianina *et al*. 2003): (i) *Réserve Naturelle Intégrale* (Strict Nature Reserve; IUCN category Ia); (ii) Parc *National* (National Park; category II), and; (iii) *Réserve Spéciale* (Special Reserve; category IV). These areas were established with little regard to the resource requirements of adjacent communities (Durbin and Ralambo 1994), as their primary management objective was the conservation of biodiversity, alongside limited research and recreation (within categories II and IV); all extractive use of natural resources was strictly prohibited according to national law (*Code des Aires Protégées*; GoM 2001). At the 2003 Vth World Parks Congress, the government of Madagascar declared its intention to increase the nation's protected area coverage to

6 million ha (the Durban Vision; Ravalomanana, 2003). The goals of the new Madagascar Protected Area System (SAPM), which comprises the established MNP network as well as the post-Durban generation of new protected areas, are threefold: (i) to conserve the whole of Madagascar's unique biodiversity; (ii) to conserve Madagascar's cultural heritage, and; (iii) to promote sustainable natural resource use for development and poverty alleviation (Commission SAPM 2006).

Steering committees established to advise on the implementation of the Durban Vision recognised that few isolated natural habitats remained and that substantial human populations depended upon most remaining natural areas for their subsistence and household income (Gardner *et al*. 2008). The established protected area models of categories Ia, II and IV were therefore seen as inappropriate for the majority of new protected areas and, with the support of IUCN consultants (Borrini-Feyerabend and Dudley 2005), the IUCN category system was used to guide the development of new categories and governance structures for the country (although these were adapted to the Malagasy context). Madagascar's protected areas legislation was revised to recognise category III, V and VI protected areas within SAPM, as well as to permit non-state bodies to promote, manage and govern new protected areas. Indeed, most new protected areas are proposed as category V and are, or will be, governed under some form of co-management (Raik 2007). Note that due to the political crisis that has left the country without a recognised legitimate government since early 2009, the revised *Code des Aires Protégées* has yet to be ratified and passed into law; all proposed category III, V and VI protected areas await this ratification before they can gain definitive protected area status (N. Ratsifandrihamanana *pers. comm*.)

3.3 Methods

Ten newly-established or proposed category V protected areas in Madagascar were identified for inclusion in this study, based on there being sufficient information available to permit analysis (Table 3.2, Fig. 3.1). The case study sites span marine, freshwater and terrestrial realms, and include all major terrestrial habitat types (humid, dry-deciduous and spiny forests) represented within Madagascar's new category V protected areas. I then applied Dudley's (2008b) categorisation decision-tool to assess the applicability of each management category to each protected area. This decision-tool was included in the final draft (Dudley 2008b) but not in the published guidelines (Dudley 2008a) because it

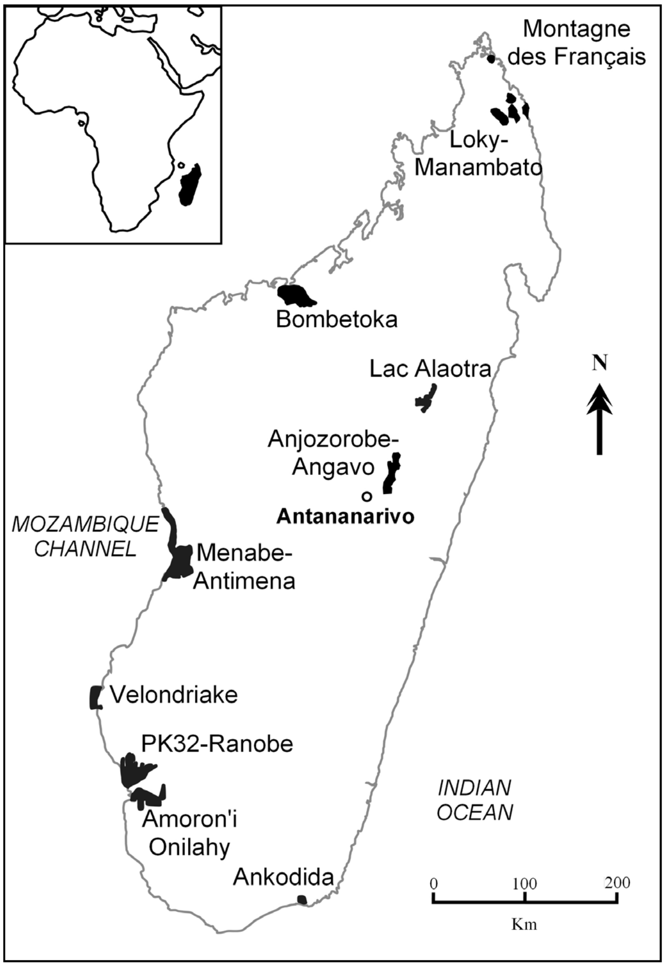

Figure 3.1 Map of Madagascar showing the 10 case study International Union for Conservation of Nature (IUCN) category V protected areas used to evaluate the suitability of IUCN protected area management category classifications to Madagascar's new generation of multiple-use protected areas. The inset shows the location of Madagascar relative to Africa.

had not been sufficiently tested (N. Dudley *pers. comm.*). Nevertheless, it remains the only available tool with which to objectively deliberate categorisation decision-making.

The tool presents a range of protected area characteristics ('key issues') and a series of states for each characteristic ('questions') (Table 3.3). For each key issue, managers are asked to select the question(s) that most accurately describes the state of each characteristic for the protected area under scrutiny. The decision-tool contains an internal scoring system that assesses the compatibility of each characteristic-state with every protected area category as either 'particularly compatible' (assigned a value of 1), 'not incompatible' (assigned 0), 'tends to be incompatible' and 'never normally suitable' (both assigned -1). Assigning these values allows the suitability of each protected area category

Table 3.2 Summary of the 10 case study International Union for Conservation of Nature (IUCN) category V protected areas used to evaluate the suitability of IUCN protected area management category classifications to Madagascar's new generation of multiple-use protected areas.

Protected area	Principal ecosystems	Size (ha)	Proposed IUCN category	Protected status (June 2010)
Amoron'i Onilahy	Spiny forest	163,000	V (including category III zones)	Temporary
Anjozorobe-Angavo	Humid forest	52,200	V	Temporary
Ankodida	Spiny and transitional forest	10,744	V (including category III zones)	Temporary
Bombetoka	Freshwater wetlands, mangroves, dry forest	46,000	V	Temporary
Lac Alaotra	Freshwater wetlands	42,478	V	Temporary
Loky-Manambato	Transitional humid/ dry forest	70,837	V	Temporary
Menabe Antimena	Dry forest, mangroves	125,000	V (including category III zones)	Temporary
Montagne des Français	Dry forest	6,092	V	Temporary
PK32-Ranobe	Spiny forest, freshwater wetlands	151,000	V	Temporary
Velondriake	Marine and coastal	c. 80,000	V (including zones of multiple categories)	Proposed

to be evaluated on the basis of the cumulative score for a particular site, with high positive scores indicating greater suitability. For this analysis, I completed the decision-tool questionnaire for each case study protected area on the basis of available information in the peer-reviewed and grey literature, and validated the accuracy of the outcome by asking experts with relevant management experience of each site to verify that the selected statements reflected on-the-ground reality. Finally, I used the decision-tool results to highlight two principal characteristics relevant to management, common to all sites, which influence categorisation decision-making.

3.4 Results

The cumulative scores for each of the 10 case study sites demonstrate the substantial variation in the suitability of each IUCN management category (Table 3.4). For all but one site, the scoring system indicates that category V is the most suitable designation. Closer analysis, however, reveals incompatibilities between the case study protected areas and each of the IUCN management categories (Table 3.5). Principally, the traditional

Table 3.3 The International Union for Conservation of Nature (IUCN) decision-tool developed to aid the categorisation of protected areas (Dudley 2008b), showing results from 10 case study new, multiple-use category V protected areas in Madagascar: AO, Amoron'i Onilahy; AA, Anjozorobe-Angavo; Ank, Ankodida; Bom, Bombetoka; LA, Lac Alaotra; LM, Loky-Manambato; MA, Menabe Antimena; MdF, Montagne des Français; PR, PK32-Ranobe; Vel, Velondriake. The compatibility of each question to each category is indicated by symbols: ✓, particularly compatible; —, not incompatible; ✗, tends to be incompatible; ☒, never normally suitable.

Key issue	Question	Scoring system to assess the compatibility of the protected area with the categories							Selected question(s), per key issue, for each protected area									
		Ia	Ib	II	III	IV	V	VI	AO	AA	Ank	Bom	LA	LM	MA	MdF	PR	Vel
Naturalness	Entire area in more-or-less natural state	✓	✓	✓	✓	—	☒	✗										
	Most of area in more-or-less natural state	—	—	✓	✓	—	✗	✓		X								
	<50 % of area in more-or-less natural state	✗	✗	—	—	—	—	☒	X		X	X	X	X	X	X	X	
	Entire area resulting from people-nature interaction over time	✗	✗	—	—	—	✓	✗										X
	Area requiring management to maintain biodiversity	☒	☒	—	—	✓	—	—										
Scale	Site large enough to conserve an ecosystem	✓	✓	✓	—	—	—	—	X	X		X	X	X	X	X	X	X
	Site not large enough to conserve an ecosystem	—	—	☒	—	—	—	—			X							
	Site designated to conserve specific feature	—	—	—	✓	—	—	—										
Connected-ness	Connected with other protected areas or similar habitats	—	—	✓	—	—	—	—	X	X		X					X	X
	Unconnected with other protected areas or similar habitats	—	—	✗	—	—	—	—			X		X	X	X	X		

Key issue	Question	Scoring system to assess the compatibility of the protected area with the categories							Selected question(s), per key issue, for each protected area									
		Ia	Ib	II	III	IV	V	VI	AO	AA	Ank	Bom	LA	LM	MA	MdF	PR	Vel
Biodiversity	Many species requiring natural conditions	✓	✓	✓	—	—	☒	—	X	X	X	X	X	X	X		X	
	Most species able to live in human-modified areas	—	—	—	—	✓	✓	—			X	X				X	X	X
	Key species need active management intervention to survive	☒	☒	—	—	✓	—	—										
	Some wild species routinely used in extractive manner	☒	☒	✗	—	—	✓	✓	X	X	X	X	X	X	X	X	X	X
Regeneration	Ecosystem capable of regeneration	—	—	—	—	✓	✓	✓		X		X		X				X
	Ecosystem difficult to regenerate to original quality	✓	✓	✓	—	—	✗	—	X		X		X		X	X	X	
Environmental services	Providing environmental services (e.g. water, soil)	—	—	—	—	—	—	—	X	X	X	X	X	X	X	X	X	X
	Not providing environmental services	—	—	—	—	—	—	—										
Social values (livelihoods, economic etc.)	Providing few socio-economic values	✓	✓	✓	—	—	✗	✗										
	Providing non-extractive socio-economic values (e.g. tourism)	—	—	✓	✓	—	—	—	X	X	X	X	X	X	X	X	X	X
	Providing extractive renewable resources	☒	☒	✗	—	—	✓	✓	X	X	X	X	X	X	X	X	X	X
	Providing extractive mineral resources	☒	☒	✗	✗	✗	—	—	X			X		X	X		X	
Traditional occupancy	Comprising traditional settlement/migration routes	✗	✗	✗	✗	—	✓	✓	X	X	X	X	X	X	X	X	X	X
	Empty of traditional settlements/migration routes	✓	✓	✓	—	—	—	—										

Key issue	Question	Scoring system to assess the compatibility of the protected area with the categories							Selected question(s), per key issue, for each protected area									
		Ia	Ib	II	III	IV	V	VI	AO	AA	Ank	Bom	LA	LM	MA	MdF	PR	Vel
User needs and wants	Users wish to practice resource extraction	✖	✖	—	—	—	✓	✓	X	X	X	X	X	X	X	X	X	X
	No users wishing to extract resources	✓	✓	—	—	—	✖	☒										
Tourism	Many tourists expected to use the site	☒	☒	✓	—	—	✓	—	X	X		X	X	X	X	X	X	X
	Few if any tourists expected to use the site	✓	✓	—	—	—	—	—			X							
Sacred and cultural values	Sacred and culturally valuable sites that are not regularly visited	✓	✓	—	—	—	—	—	X			X	X	X	X	X	X	X
	Sacred and culturally valuable sites that are regularly visited	✖	✖	—	✓	—	—	—		X	X		X					
	Without sacred or culturally valuable sites	—	—	—	—	—	—	—										
People-nature interaction	Historically present	✖	✖	—	✓	—	✓	—	X	X	X		X	X	X	X	X	X
	Historically absent	✓	✓	✓	—	—	✖	✖				X						
	Mostly negative with respect to desired biodiversity	✓	✓	✓	✓	—	☒	✖	X	X	X	X	X	X	X	X	X	X
	Mixed results with respect to desired biodiversity	—	—	—	—	—	—	—										
	Mostly positive with respect to desired biodiversity	—	—	—	—	—	✓	✓										
	Very positive results with respect to desired biodiversity	✖	✖	✖	✖	✓	✓	✓										

Table 3.4: Results of categorisation decision-tool analyses for new protected areas in Madagascar, showing cumulative scores by International Union for Conservation of Nature (IUCN) management category. High, positive scores indicate that a category is suitable for that protected area.

Protected area	**IUCN category**					
	Ia	**II**	**III**	**IV**	**V**	**VI**
Amoron'i Onilahy	-3	3	1	-1	3	1
Anjozorobe-Angavo	-4	4	4	1	4	5
Ankodida	-3	-1	3	0	3	2
Bombetoka	-2	3	0	1	4	2
Lac Alaotra	-4	2	3	0	3	2
Loky-Manambato	-4	0	1	0	5	3
Menabe Antimena	-3	1	1	-1	3	2
Montagne des Français	-3	1	2	1	5	2
PK32-Ranobe	-3	2	1	0	4	2
Velondriake	-4	2	2	2	8	3
Mean	**-3.3**	**1.7**	**1.8**	**0.3**	**4.2**	**2.4**

human occupancy and management emphasis on sustainable resource extraction renders each of the protected areas unsuitable for designation as categories I, II or III, and their focus on landscapes rather than specific habitats or species requiring management is mismatched with category IV. The two highest-scoring categories, V and VI, have objectives that are congruent with large-scale natural resource use, but there are two particular protected area attributes, naturalness and people-nature interactions, that are critical to the designation of these two categories.

3.4.1 Naturalness

The key difference between categories V and VI concerns the degree of human modification of the landscape. Category V is suited to cultural landscapes shaped by human influence over time, whereas category VI guidelines suggest that two-thirds of a

Table 3.5 Principal sources of incompatibility between the 10 case study protected areas in Madagascar and International Union for Conservation of Nature (IUCN) management categories.

Category	Incompatibility with case study protected areas
Ia (Strict nature reserve)	Established in least human-impacted areas, and strictly controls human visitation and use to ensure protection of conservation values. Case study areas are established in cultural landscapes and permit a range of human uses.
Ib (Wilderness area)	Established in large, unmodified landscapes without significant habitation and managed to retain natural condition. Case study areas are generally too small and too modified to qualify as wilderness.
II (National park)	Established in large, natural areas primarily to ensure conservation at ecosystem scale, with limited human use apart from recreation. Case study areas are smaller scale and permit a range of human uses.
III (Natural monument)	Generally small and established to protect specific natural features or culturally important natural sites. Management of case study areas is focused on conservation of landscapes or seascapes rather than specific features.
IV (Species/ habitat management area)	Established to protect specific habitat or species and usually requires active management. Management of case study areas is focused on conservation of landscapes or seascapes rather than specific habitats or species, and does not include active species or habitat-focused interventions.
V (Protected landscape/ seascape)	Established to maintain people-nature interactions that enhance conservation value in cultural landscapes. In the case study areas, people-nature interactions generally diminish conservation value (in terms of viability of endemic species and communities).
VI (Sustainable use area)	Established in predominantly natural areas and permits low-impact resource use. Case study areas are established in predominantly cultural landscapes and permit human uses that have a greater impact on natural habitats.

protected area should be composed of natural or unmodified areas (defined as "those that still retain a complete or almost complete complement of species native to the area, within a more-or-less naturally functioning ecosystem"; Dudley 2008a, p. 12). The seven terrestrial case study protected areas all fail to meet this criterion, having significant areas of deforested land and little undegraded forest within their boundaries. Indeed, for PK32-Ranobe, Amoron'i Onilahy and Ankodida, the limits of the protected areas include deliberate, and significant, areas of deforested land to permit development interventions within buffer zones of low conservation value. The ecosystems of Lac Alaotra have been altered by marsh drainage and burning, sedimentation and the introduction of invasive plants and fish (Andrianandrasana *et al.* 2005; Ranarijaona 2007), while Velondriake has been overfished to the point where trophic dynamics have been transformed and provoked phase shifts from coral to algal cover (Harris 2007). Beyond these sites, it is debatable whether any area of Madagascar possesses a "complete complement of species native to the area", given the relatively recent (< 2,000 years) extinction of the island's mammal, bird and reptile megafauna (Crowley 2010). All potential conservation areas could therefore be termed cultural landscapes.

3.4.2 People-nature interaction

Human influence on Madagascar's ecosystems has been largely negative for biodiversity (Gardner 2009; Irwin *et al*. 2010), with two major impacts apparent: (i) the extinction of the megafauna (Crowley 2010), and; (ii) the loss of forest cover. Although the extent of original forest cover is disputed (Virah-Sawmy 2009), up to 84 % is thought to have been lost since human colonisation through anthropogenic deforestation (Harper *et al*. 2007). Deforested landscapes largely comprise species-poor grasslands and bushlands, with little value for endemic biodiversity (Lowry *et al*. 1997). Since the majority of the island's biota is restricted to forests (Goodman and Benstead 2005), the overall effect of human land-use over the last two millennia has been detrimental for Madagascar's endemic species. Two examples from the case study protected areas, Lac Alaotra and Southern spiny forest, serve to illustrate the issues.

Lac Alaotra

This protected area is the largest body of freshwater in Madagascar and has been a Ramsar site since 2003. The lake and its associated marshes include the entire global range of the Critically Endangered Alaotra gentle lemur *Hapalemur alaotrensis*, as well as the only known breeding area of the now extinct Alaotra little grebe *Tachybaptus rufolavatus* and (until recently) the Critically Endangered Madagascar pochard *Aythya innotata*. The human population of the Alaotra watershed increased five-fold from 1960 to 2003 (Bakoariniaina *et al*. 2006) and, as a consequence, the forested hills of the watershed have mostly been cleared. Subsequent erosion and siltation resulted in acidification and a reduction in lake area to 20 % of its original size by 2000 (Bakoariniaina *et al.* 2006). Most of the lake's marshes have been converted to rice cultivation and remain threatened by burning (Copsey *et al*. 2009), introduced plant and fish species have altered aquatic vegetation dynamics, and overfishing remains a serious problem (Andrianandrasana *et al*. 2005). This range of pressures on the ecosystem has had extreme knock-on impacts on the locally-endemic biodiversity: *T. rufolavatus* has not been recorded since 1982 (IUCN 2010; Wilmé 1994) and *A. innotata* has not been recorded since 1991 (it was presumed extinct until it was rediscovered recently elsewhere; René de Roland *et al*. 2007). The population of *H. alaotrensis* has declined from an estimated 10,710 individuals in 1994 to 2,480 in 2002 (Ralainasolo 2004).

Southern spiny forest

The spiny forest ecoregion of southern Madagascar was, prior to the Durban Vision (and the establishment of PK32-Ranobe, Amoron'i Onilahy and Ankodida amongst the case studies), the least represented major forest type within the country's protected area system (Fenn 2003a). People-nature interactions within the ecoregion take various forms, including pastoralism, timber and non-timber forest product extraction, charcoal production, and slash-and-burn agriculture (*hatsake*, Gardner *et al*. 2008; Seddon *et al*. 2000). Although the *hatsake* may have been sustainable at low human population densities and under certain social institutional conditions (Elmqvist *et al*. 2007), changing macroeconomic conditions (Casse *et al*. 2004; Minten *et al*. 2006), population growth and rising migration (Kaufmann and Tsirahamba 2006; Rabesahala Horning 2003) have led to the region suffering the fastest rates of forest loss anywhere in the country since 1990 (Harper *et al*. 2007). In the only existing study on the impacts of forest loss on biodiversity within the ecoregion, Scott *et al*. (2006) found that species richness of lizards, small mammals and birds declined by 50, 40 and 26 % respectively, and species turnover resulted in community composition shifts from habitat specialists to generalists. All three of the case study protected areas within the region are primarily threatened by *hatsake* and charcoal production. The interaction between people and nature has been negative for the ecological and biological values of the sites, as well as for certain environmentally-favourable cultural values, such as the preservation of culturally and spiritually important forest areas (Bodin *et al*. 2006; Fenn 2003b; Gardner *et al*. 2008). Other cultural values, however, such as the opportunity to derive a livelihood from ancestral lands (Keller 2008), are enhanced by the interaction.

3.5 Discussion

The analysis suggests that the 10 case study protected areas do not fall neatly into any of the six IUCN management categories, despite being proposed as category V. Of the two designations compatible with large-scale natural resource extraction, category VI is unsuitable because of the degree of human modification of the land- and seascapes in question, while the negative impact of land and resource use on biodiversity violates the key principle of category V. The choice between categories V and VI, at least for some of the terrestrial case studies, is complicated by the deliberate inclusion of degraded and deforested ecosystems within the protected area boundaries. They have been incorporated into protected areas because: (i) they are seen as valuable for the promotion of economic development to reduce dependence on unsustainable resource use, and; (ii) they often occur in a landscape-scale mosaic alongside higher quality habitat, making their exclusion from protected areas complex or impossible spatially. Although the exclusion of these degraded zones would lead to the protected areas more closely aligning with the criteria for category VI, categorisation must be based on site-specific realities and thus category V remains the most suitable based on the decision-tool analysis.

With regard to human impacts on biodiversity, I do not suggest that there are no examples of harmonious people-nature relationships in Madagascar of the type envisaged in the IUCN definition of category V, but rather that the case study protected areas are not typified by such interactions. Possible examples of sustainable interactions include the sclerophyllous scrub and alti-montane prairies of Andringitra, which are at least partly maintained by cattle grazing and fires (Rabetaliana and Schachenmann 1999), the fire-maintained Tapia woodlands of the central highlands (Kull 2004), forest management by Mahafaly pastoralists (Kaufmann and Tsirahamba 2006), and the suppression of the invasive endemic vine *Sarcostemma viminale* (Asclepiadaceae) by cattle grazing within Beza Mahafaly Special Reserve (Sussman and Rakotozafy 1994).

The key difference between category V in Madagascar and the model as conceived and implemented in Europe concerns the role of people-nature interactions within present and future protected area management. In the IUCN classification model, these relationships are seen as intrinsic to the landscape and essential for the maintenance of conservation values (Phillips 2002). In Madagascar, however, such interactions are largely negative for the maintenance of conservation value and, if left unchecked, could result in the near complete loss of natural habitats and their associated high species diversity. The very justification for protected area creation, therefore, is threatened by the impact of local communities on the landscape. Rather than maintaining existing people-nature interactions, management of category V areas in Madagascar is focused on modifying and reducing the type and intensity of natural resource use to promote long-term sustainability (Gardner *et al*. 2008; Harris 2007); harmonious people-nature relationships are a desired future state to be fostered, rather than an existing dynamic to be maintained. This distinction is not purely semantic, having real and important management ramifications. In the European model of category V, a balance between humans and nature has been reached and the challenge is to maintain traditional land use in the face of more destructive modern practices (Phillips 2002). In Madagascar, this is not the case and the objective is to adapt these relationships between people and their environment into more benign forms before further biodiversity is lost. This fundamental difference in approach is particularly important given that category V has been promoted by the IUCN for adoption worldwide; emphasising the maintenance of traditional livelihood practices will not result in conservation gains where such activities are detrimental to biodiversity. Unfortunately, this distinct difference diminishes the utility of the category system as a framework for the description and comparison of protected areas.

Given the incompatibilities between Malagasy protected areas proposed as category V and each of the six main IUCN categories, how should these sites be categorised? The protected areas need to incorporate the livelihood needs of local communities in management decision-making and, consequently, certain activities that have the potential to negatively affect the conservation values of the site if not managed sustainably. Such protected areas are likely to become more common in an ever-modernising world, where remnant natural ecosystems are increasingly small and fragmented, and growing numbers of rural people depend on the resources from these habitats for their well-being (Mallarach *et al*. 2008). My analysis suggests that protected areas established in such a context, where human land and resource use may diminish the viability of species and ecosystems but must nevertheless be accepted in the initial stages of protected area establishment, are not adequately represented by the IUCN protected area management categories. By not recognising the fundamental differences between category V protected areas of the type proposed in Madagascar and those prevalent in Europe, the system fails to acknowledge the former for what they are; valuable areas for the conservation of biodiversity that are threatened, rather than maintained, by human agency. Pigeon-holing these sites into other categories may result in conservation scientists, practitioners and policy-makers overlooking the critical role played by this emerging type of protected area in maintaining biodiversity.

I therefore suggest that the IUCN category system should be modified to formally distinguish the intrinsically distinct nature of such protected areas. Two possibilities present themselves: (i) that the definition of category V is relaxed so as to reduce the emphasis on people-nature interactions that are positive for biodiversity, and to include all protected areas in which human-biodiversity relationships of any type are dominant features of the landscape (guidelines could simply require the potential for sustainable interaction to be restored or fostered), or; (ii) that an additional category or subcategory is created and defined, so as to specifically account for the types of protected area in which human-nature interactions must be transformed, rather than maintained, to meet protected area management objectives. Of these two options, the first would meet the goal of recognising new protected areas of the type prevalent in Madagascar, but would fail nonetheless to make the distinction between them and category V protected areas as managed in Europe. The second proposal, however, would specifically acknowledge the unique management objectives and approach of Malagasy-type multiple-use protected areas, which is a necessary conceptual step if we are to successfully employ the protected area approach to conserve biodiversity in human-dominated landscapes.

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References

Andrianandrasana, H.T., Randriamahefasoa, J., Durbin, J., Lewis, R.E. and Ratsimbazafy, J.H. 2005. Participatory ecological monitoring of the Alaotra wetlands in Madagascar. *Biodiversity and Conservation* 14: 2757–2774.

Bakoariniaina, L.N., Kusky, T. and Raharimahefa, T. 2006. Disappearing Lac Alaotra: Monitoring catastrophic erosion, waterway silting, and land degradation hazards in Madagascar using Landsat imagery. *Journal of African Earth Sciences* 44: 241–252.

Bodin, O., Tengö, M., Norman, A., Lundberg, J. and Elmqvist, T. 2006. The value of small size: loss of forest patches and ecological thresholds in southern Madagascar. *Ecological Applications* 16: 440–451.

Borrini-Feyerabend, G. and Dudley, N. 2005. *Elan Durban: Nouvelles Perspectives Pour les Aires Protégées à Madagascar*. IUCN report to the Government of Madagascar.

Büscher, B. and Whande, W. 2007. Whims of the winds of time? Emerging trends in biodiversity conservation and protected area management. *Conservation and Society* 5: 22 –43.

Casse, T., Milhøj, A., Ranaivoson, S. and Randriamanarivo, J.R. 2004. Causes of deforestation in southwestern Madagascar: what do we know? *Forest Policy and Economics* 6: 33–48.

Commission SAPM 2006. *Système d'Aires Protégées Malagasy: Orientations Générales sur les Catégories et les Types de Gouvernance*. SAPM Commission, Antananarivo.

Copsey, J.A., Jones, J.P.G., Andrianandrasana, H., Rajaonarison, L.H. and Fa, J.E. 2009. Burning to fish: local explanations for wetland burning in Lac Alaotra, Madagascar. *Oryx* 43: 403–406.

Crowley, B.E. 2010. A refined chronology of prehistoric Madagascar and the demise of the megafauna. *Quaternary Science Reviews* 29: 2591–2603.

Dillon, B. 2004. The use of the categories in national and international legislation and policy. *Parks* 14: 15–22.

Dudley, N., Harrison, J. and Rosabal, P. 2004. The future development of the categories system. *Parks* 14: 72–81.

Dudley, N. (Ed.). 2008a. *Guidelines for Applying Protected Area Management Categories*. IUCN, Gland.

Dudley, N. (Ed.). 2008b. *Guidelines for Applying Protected Area Management Categories – Final draft of revised guidelines*, July 2008. IUCN, Gland.

Dudley, N., Parrish, J.D., Redford, K.H. and Stolton, S. 2010. The revised IUCN protected area management categories: the debate and ways forward. *Oryx* 44: 485–490.

Durbin, J.C. and Ralambo, J.A. 1994. The role of local people in the successful maintenance of protected areas in Madagascar. *Environmental Conservation* 21: 115–120.

Elmqvist, T., Pyykönen, M., Tengö, M., Rakotondrasoa, F., Rabakonandrianina, E. and Radimilahy, C. 2007. Patterns of loss and regeneration of tropical dry forest in Madagascar: the social institutional context. *PLoS ONE* 2: e402.

Fenn, M.D. 2003a. The spiny forest ecoregion. In *The Natural History of Madagascar*, S.M. Goodman and J.P. Benstead (Eds.), pp. 1525–1529. University of Chicago Press, Chicago.

Fenn, M. D. 2003b. Learning conservation strategies. A case study of the Parc National d'Andohahela. In *The Natural History of Madagascar*, S.M. Goodman and J.P. Benstead (Eds.), pp. 1494–1501. University of Chicago Press, Chicago.

Gardner, C.J. 2009. A review of the impacts of anthropogenic habitat change on terrestrial biodiversity in Madagascar: implications for the design and management of new protected areas. *Malagasy Nature* 2: 2–29.

Gardner, C.J., Ferguson, B., Rebara, F. and Ratsifandrihamanana, A.N. 2008. Integrating traditional values and management regimes into Madagascar's expanded protected area system: the case of Ankodida. In *Protected Landscapes and Cultural and Spiritual Values*, J.-M. Mallarach (Ed.), pp. 92–103. IUCN, GTZ and Obra Social de Caixa Catalunya. Kasparek Verlag, Heidelberg.

GoM (Government of Madagascar) 2001. *Loi N° 2001/05 portant Code de Gestion des Aires Protégées*. Repoblikan'i Madagasikara, Antananarivo.

Goodman, S.M. and Benstead, J.P. 2005. Updated estimates of biotic diversity and endemism for Madagascar. *Oryx* 39: 73–77.

Harper, G.J., Steininger, M.K., Tucker, C.J., Juhn, D. and Hawkins, F. 2007. Fifty years of deforestation and forest fragmentation in Madagascar. *Environmental Conservation* 34: 325–333.

Harris, A. 2007. "To live with the sea": development of the Velondriake community-managed protected area network, southwest Madagascar. *Madagascar Conservation & Development* 2: 43–49.

Hutton, J.M. and Leader-Williams, N. 2003. Sustainable use and incentive driven conservation: realigning human and conservation interests. *Oryx* 37: 215–226.

Irwin, M.T., Wright, P.C., Birkinshaw, C., Fisher, B., Gardner, C.J., Glos, J., Goodman, S.M., Loiselle, P., Rabeson, P., Raharison, J.-L., *et al*. 2010. Patterns of species change in anthropogenically disturbed forests of Madagascar. *Biological Conservation* 143: 2351–2362.

IUCN 1994. *Guidelines for Protected Area Management Categories*. IUCN, Gland.

IUCN 2010. News release: Wetland aliens cause bird extinction. http://www.iucnredlist.org/news/wetland-aliens-cause-bird-extinction [accessed 29.05.10]

Jenkins, C.N. and Joppa, L. 2009. Expansion of the global terrestrial protected area system. *Biological Conservation* 142: 2166–2174.

Kaufmann, J.C. and Tsirahamba, S. 2006. Forests and thorns: conditions of change affecting Mahafale pastoralists in southwestern Madagascar. *Conservation and Society* 4: 231–261.

Keller, E. 2008. The banana plant and the moon: conservation and the Malagasy ethos of life in Masoala, Madagascar. *American Ethnologist* 35: 650–664.

Kull, C.A. 2004. *Isle of Fire: the Political Ecology of Landscape Burning in Madagascar*. University of Chicago Press, Chicago.

Leroux, S.J., Krawchuk, M.A., Schmiegelow, F., Cumming, S.G., Lisgo, K., Anderson, L.G. and Petkova, M. 2010. Global protected areas and IUCN designations: do the categories match the conditions? *Biological Conservation* 143: 609–616.

Locke, H. and Dearden, P. 2005. Rethinking protected area categories and the new paradigm. *Environmental Conservation* 32: 1–10.

Lowry II, P.P., Schatz, G.E. and Phillipson, P.B. 1997. The classification of natural and anthropogenic vegetation in Madagascar. In *Natural Change and Human Impact in Madagascar*, S.M. Goodman and B.D. Patterson (Eds.), pp. 92–123. Smithsonian Institution Press, Washington DC.

Mallarach, J.M., Morrison, J., Kothari, A., Sarmiento, F., Atauri, J.-A. and Wishitemi, B. 2008. In defence of protected landscapes: a reply to some criticisms of category V protected areas and suggestions for improvement. In *Defining Protected Areas: an International Conference in Almeria, Spain*, N. Dudley and S. Stolton (Eds.), pp. 30–36. IUCN, Gland.

Martino, D. 2005. Unleashing the wild: response to Locke and Dearden's 'rethinking protected area categories'. *Environmental Conservation* 32: 195–196.

Minten, B., Randrianarison, L., Méral, P. and Swinnen, J. 2006. *Trade Liberalization, Rural Poverty and the Environment: the Case of Madagascar*. WWF Madagascar and Western Indian Ocean Programme Office, Antananarivo.

Myers, N., Mittermeier, R.A., Mittermeier, C.G., da Fonseca, G.A.B. and Kent, J. 2000. Biodiversity hotspots for conservation priorities. *Nature* 403: 853–858.

Naughton-Treves, L., Buck Holland, M. and Brandon, K. 2005. The role of protected areas in conserving biodiversity and sustaining local livelihoods. *Annual Review of Environment and Resources* 30: 219–252.

Phillips, A. 2002. *Management Guidelines for IUCN Category V Protected Areas: Protected Landscapes/Seascapes*. IUCN, Gland and Cambridge.

Phillips, A. 2004. The history of the international system of protected area management categories. *Parks* 14: 4–14.

Rabesahala Horning, N. 2003. How rules affect conservation outcomes. In *The Natural History of Madagascar*, S.M. Goodman and J.P. Benstead (Eds.), pp. 146–153. University of Chicago Press, Chicago.

Rabetaliana, H. and Schachenmann, P. 1999. Coordinating traditional values, scientific research and practical management to enhance conservation and development objectives in the Andringitra Mountains, Madagascar; lessons learned! *African Studies Quarterly* 3: 60–67.

Raik, D. 2007. Forest management in Madagascar: an historical overview. *Madagascar Conservation & Development* 2: 5–10.

Ralainasolo, F.B. 2004. Influence des effets anthropiques sur la dynamique de population de *Hapalemur griseus alaotrensis* ou “Bandro” dans son habitat naturel. *Lemur News* 9: 32–35.

Ranarijaona, H.L.T. 2007. Concept de modèle écologique pour la zone humide Alaotra. *Madagascar Conservation & Development* 2: 35–42.

Randrianandianina, B.N., Andriamahaly, L.R., Harisoa, F.M. and Nicoll, M.E. 2003. The role of protected areas in the management of the island’s biodiversity. In *The Natural History of Madagascar*, S.M. Goodman and J.P. Benstead (Eds.), pp. 1423–1432. University of Chicago Press, Chicago.

Ravalomanana, M. 2003. *Durban Vision Declaration*. Speech presented at the Vth World Parks Congress, Durban, 07–19 September 2003.

René de Roland, L.-A., Sam, T.S., Rakotondratsima, M.P.H. and Thorstrom, R. 2007. Rediscovery of the Madagascar pochard *Aythya innotata* in northern Madagascar. *Bulletin of the African Bird Club* 14: 171–174.

SCBD (Secretariat of the Convention on Biological Diversity) 2004. *Decisions Adopted by the Conference of the Parties to the Convention on Biological Diversity at its Seventh Meeting*. UNEP/CBD/CoP/7/21, SCBD, Montreal.

Scott, D.M., Brown, D., Mahood, S., Denton, B., Silburn, A. and Rakotondraparary, F. 2006. The impacts of forest clearance on lizard, small mammal and bird communities in the arid spiny forest, southern Madagascar. *Biological Conservation* 127: 72–87.

Seddon, N., Tobias, J., Yount, J.W., Ramanampamonjy, J.R., Butchart, S. and Randrianizahana, H. 2000. Conservation issues and priorities in the Mikea Forest of south -west Madagascar. *Oryx* 34: 287–304.

Sussman, R.W. and Rakotozafy, A. 1994. Plant diversity and structural analysis of a tropical dry forest in southwestern Madagascar. *Biotropica* 26: 241–254.

Virah-Sawmy, M. 2009. Ecosystem management in Madagascar during global change. *Conservation Letters* 2: 163–170.

Wells, M.P. and McShane, T.O. 2004. Integrating protected area management with local needs and aspirations. *Ambio* 33: 513–519.

Wilmé, L. 1994. Status, distribution and conservation of two Madagascar bird species endemic to Lake Alaotra: Delacour's grebe *Tachybaptus rufolavatus* and Madagascar pochard *Aythya innotata*. *Biological Conservation* 69: 15–21.

Chapter 4

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A review of the impacts of anthropogenic habitat change on terrestrial biodiversity in Madagascar: implications for the design and management of new protected areas

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Abstract

Madagascar's recently established protected areas seek to contribute to both conservation and development objectives. They comprise mainly multiple-use sites in which a range of human activities are permitted, hence the impacts of forest use on biodiversity must be understood if such protected areas are to be designed and managed effectively. Here a review is conducted of the literature on the effects of habitat change on Madagascar's terrestrial biodiversity, and the associated range of responses of different taxonomic groups are analysed. Habitat change may lead to increases or decreases in species richness or abundance in the short-term, but the use of measures of species richness alone may mask a turnover of taxa from specialists to generalists and from endemic to non-endemic. Dry forest species and communities may be less sensitive to habitat change than those of humid forests. Biodiversity impacts appear to follow a gradient of management intensity, with selective logging and edge effects having less influence on faunal communities than secondary forests and plantations. Priorities for future research are suggested, and the implications of existing research for protected area management (including zoning, the choice of management objectives, target viability analyses and

monitoring) are discussed. Although new protected areas provide complementary conservation services to the existing network of strict protected areas, the latter may be essential for the long-term maintenance of high priority endemic taxa.

4.1 Introduction

Read almost any article on biodiversity or conservation in Madagascar and the introductory paragraphs will highlight the extraordinary rates of deforestation suffered throughout the island in recent decades. While deforestation is one of the principal causes of terrestrial biodiversity loss in Madagascar, there has been comparatively little focus on the parallel process of anthropogenic habitat modification or degradation (i.e., changes to forest structure other than outright destruction). An understanding of the impacts of natural and anthropogenic habitat alteration on Madagascar's biodiversity is important given recent shifts in the country's conservation politic, particularly with regards to the creation and management of new protected areas. The first generation of protected areas in Madagascar are managed by the para-statal Madagascar National Parks (formerly ANGAP) in accordance with IUCN Categories I, II and IV, which represent the strictest categories in terms of permitted uses (Dudley and Phillips 2006; IUCN 1994). These protected areas are managed principally for biodiversity conservation, alongside research and recreation in some categories (Randrianandianina *et al*. 2003), and they are stringently regulated within national legislation (the *Code de Gestion des Aires Protégées* or COAP; GoM 2001). The COAP forbids, among other activities, the destruction or collection of plants in all protected area categories (Article 41). In addition, Article 11 states that the "valorisation of biodiversity will be achieved primarily through research and ecotourism" (author's translation from the original French) and, by implication, not through the extractive use of natural resources (GoM 2001).

Following the launch of the 'Durban Vision' (Ravalomanana 2003), which entails the tripling of national protected area coverage from 1.7 to 6 million ha by 2012 (Mittermeier *et al*. 2005; Norris 2006), a new generation of protected areas is being established to form the *Système d'Aires Protégées de Madagascar* (Madagascar Protected Area System or SAPM), alongside the existing Madagascar National Parks network. The three principal objectives of SAPM are to:

i) Conserve the whole of Madagascar's unique biodiversity (ecosystems, species, genetic diversity);

ii) Conserve Madagascar's cultural heritage;

iii) Promote sustainable use for development and poverty alleviation (Commission SAPM (2006), author's translation from the original French).

In accordance with SAPM's multiple objectives, most of the new Durban Vision protected areas that will be established by actors other than Madagascar National Parks are managed as IUCN categories III, V and VI, and contain significant areas of sustainable use zones in various forms (up to 75 % of their surface area; Pollini 2007). Such protected areas, many of which are co-managed by local community associations (Raik 2007), seek not only to conserve Madagascar's unique biodiversity but also to maintain and enhance the material wellbeing of the human communities surrounding them (see Gardner *et al*. 2008), or at least to avoid/mitigate any negative impacts (GoM 2006). The multiple objectives of both SAPM and individual protected areas are seen as vital to long-term protected area sustainability, given the extent to which rural communities throughout Madagascar depend upon forest products and services for their survival. For example, in the southwest, 97 % of households near Analavelona and 84 % of households near Zombitse depend upon forest products for their subsistence and household income (Rabesahala Horning 2003). As such new, multiple-use protected areas will continue to suffer habitat change as a result of activities including selective logging for timber and fuelwood/charcoal, the collection of non-timber forest products and livestock grazing.

At first glance, the objectives of conserving biodiversity and promoting rural development within multiple-use protected areas could be considered as potentially conflicting, since such development often depends on the extractive use of natural resources which necessarily entails some habitat modification. If potential discord between management objectives is to be minimised, so that protected areas contribute to the goals of diverse stakeholders, it is vital that the impacts of human-induced habitat alteration on Malagasy biodiversity are understood by conservation practitioners. While much research has been carried out on the effects of habitat degradation on Malagasy species and communities (arising both from subsistence practices such as those authorised within SAPM, as well as industrial logging and deforestation, which are not permitted), there remains no synthesis of this information. The principal objective of this paper, therefore, is to review the existing literature on the effects of anthropogenic habitat change on biodiversity in Madagascar, in order to explore its implications for the design and management of new protected areas. The review thereby aims to help bridge the research-implementation gap that is prevalent in conservation biology (Knight *et al.* 2008).

4.2 Methods

Madagascar was first colonised by people approximately 2300 years before present (BP). Multiple lines of evidence suggest that their impact on natural environments was large, and includes the decline and eventual disappearance of the megafauna from around 1700 BP, followed by the extensive conversion of habitats (through fire) starting in the southwest and spreading to other coasts and the highlands (Burney *et al*. 2004). Anthropogenic habitat change therefore has a long history on the island. It is defined herein as the modification of natural habitat structure arising from a suite of human activities including selective logging, fire, the creation of paths, and the grazing of livestock. Also included within the scope of this review are other types of altered or artificial habitat, such as secondary forests regenerating after clearance and plantation forests, but papers investigating the impacts of outright forest clearance (e.g. Benstead *et al*. 2003; Scott *et al*. 2006) or research carried out within agricultural landscapes (e.g. Martin *et al.* 2009) are not considered. The influence of habitat fragmentation on biodiversity has been reviewed elsewhere (e.g., Ganzhorn *et al*. 2003; Goodman and Raherilalao 2003). However, edge effects can be seen as functionally analogous to the types of habitat change listed above and are thus incorporated into the review.

I used a searchable database of publications on Madagascar's biodiversity ('NOE 4D'), comprising 2852 articles and spanning the period 1658–2008, to search for relevant research. The database was compiled by Lucienne Wilmé in 2008 in a project funded by WWF. The database was searched using the terms 'impact', 'effect', 'degradation' and 'habitat' to identify relevant papers, with the reference lists of these works also examined in order to identify additional research of interest. Given the paucity of available information on species and community responses to habitat change in Madagascar, I sought to make full use of available evidence through knowledge transfer. Therefore, the search was not limited to the impacts of activities that may be permitted within new, multiple-use protected areas, since illicit forest uses also trigger habitat changes that may be analogous to modifications arising from authorised activities. I did not limit the review to research using appropriate counterfactuals (i.e. temporal or spatial controls following the before-after/control-impact framework; Stewart-Oaten *et al*. (1986)) because few papers (n = 17) satisfied this criterion. Studies included in the review vary in their degree of robustness and quality and thus consist of a broad range of evidence. Likewise, they were conducted across a continuous gradient of habitat degradation.

Diversity in the type of investigation undertaken precluded the use of formal meta-analytical techniques or other forms of quantitative analysis. Trends in the research were qualitatively analysed by focal taxonomic group, type of habitat modification and geographic distribution (for the latter, articles were grouped using Cornet's (1974) bioclimatic model because it was not always possible to categorise the material based on a more detailed classification system such as Moat and Smith (2007), due to a lack of detail reported in the papers). Much of the literature reviewed was published prior to recent revisions and nomenclatural changes of focal taxa; the original names from the cited publications are reproduced here, and updated names are provided in cases where the identity of the taxon is unambiguous.

4.2.1 Analysis of impacts

For research papers which generated explicit conclusions regarding the effects of habitat change on individual species, taxonomic groups or assemblages, I classified the impacts into the three categories defined below:

Positive impact: the focal species/community demonstrated an increase in either abundance/density or species richness within modified habitats or along a degradation gradient, in comparison to undegraded or less degraded areas.

Negative impact: the focal species/community demonstrated a decrease in either abundance/density or species richness within modified habitats or along a degradation gradient, in comparison to undegraded or less degraded areas.

Neutral or mixed impact: based on outcomes where there was: (i) no measurable change in either abundance/density or species richness between modified habitats and a counterfactual; (ii) an increase in abundance, but this was accompanied by a decrease in species richness (or vice versa), or; (iii) an approximately equal number of species found to benefit from habitat change as found to be negatively affected.

4.3 Results

Fifty two papers were found investigating the impacts of human-induced habitat modification on Madagascar's terrestrial biodiversity. In terms of focal taxa, lemurs, birds, small mammals, reptiles and amphibians have received the most research effort (Fig. 4.1), with comparatively little material examining bats, invertebrates and vegetation

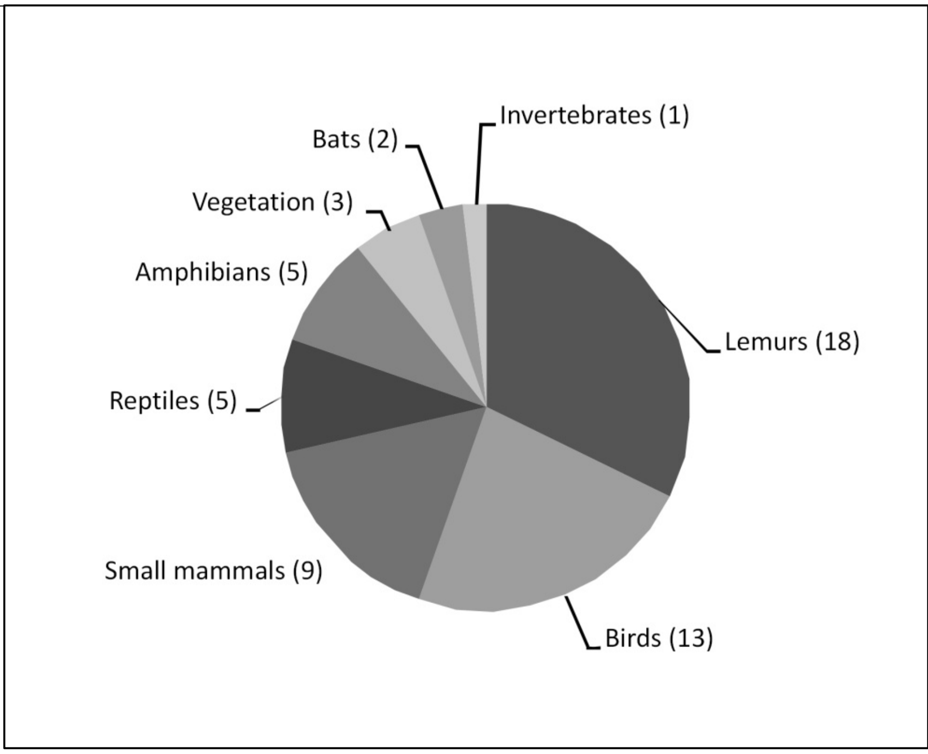

Fig. 4.1 Taxonomic focus of 52 studies investigating the impacts of anthropogenic habitat change on terrestrial biodiversity in Madagascar. Small mammals comprises Rodentia and Afrosoricida. Some studies focused on more than one taxonomic group.

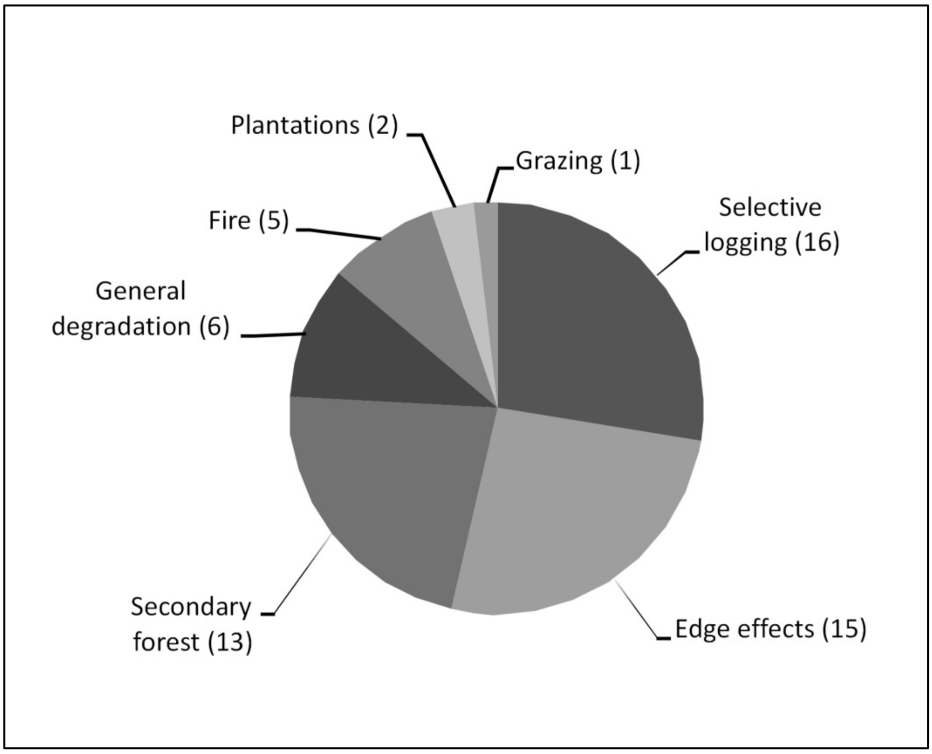

Fig. 4.2 Types of anthropogenic habitat modification or modified habitats investigated by a sample 52 studies in Madagascar. Some studies investigated multiple phenomena.

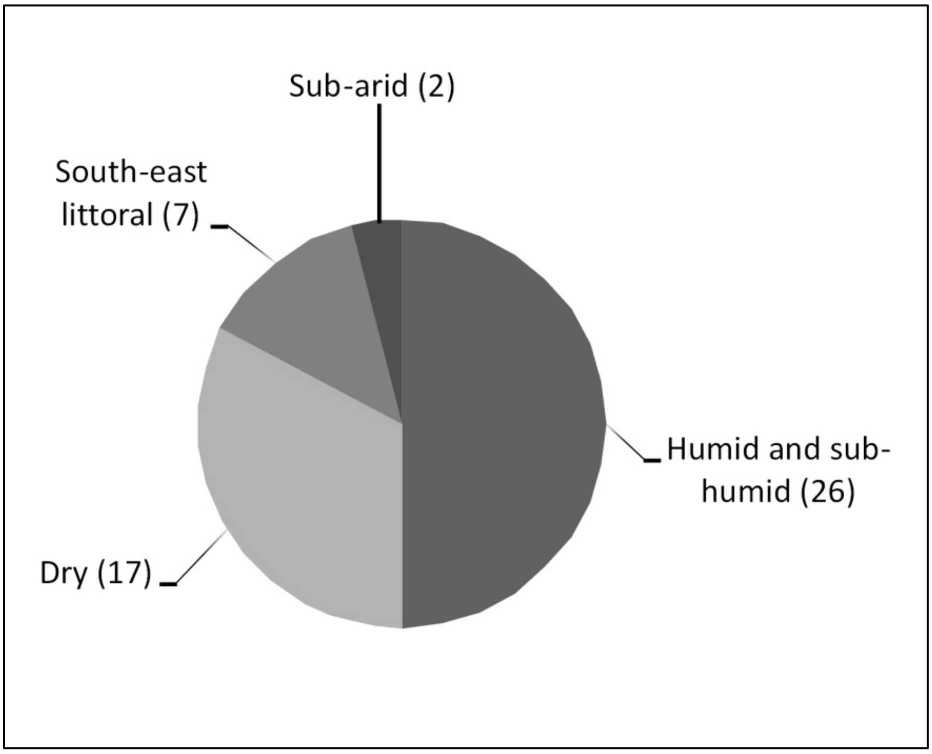

Fig. 4.3 Geographic distribution of 52 studies investigating the impacts of anthropogenic habitat change on Madagascar's terrestrial biodiversity, grouped by bioclimatic region (Cornet 1974). The southeast littoral forests are part of the humid region but are separated as an entity to highlight the research focus they have received.

(although a number of faunal studies also measure alterations to vegetation structure). The majority of reviewed papers were focused on assessing the impacts of selective logging, edge effects and secondary forests (Fig. 4.2).

Grouping the research by Cornet's (1974) bioclimatic regions (but treating the humid and sub-humid regions together because the specific location of some studies does not permit them to be accurately assigned to either region, and separating out the southeast littoral forests as a discrete entity because they have received so much attention in their own right), we see that the eastern humid and sub-humid regions, western dry region and the southeast littoral forests comprise the majority of published work (Fig. 4.3 and Fig. 4.4).

4.3.1 Birds: community impacts

In Ankarafantsika, Pons and Wendenburg (2005) observed that species richness and abundance, based on point count data, were greater in secondary regrowth a few years after a fire than in unburnt natural forest. Secondary forests contained all the same species

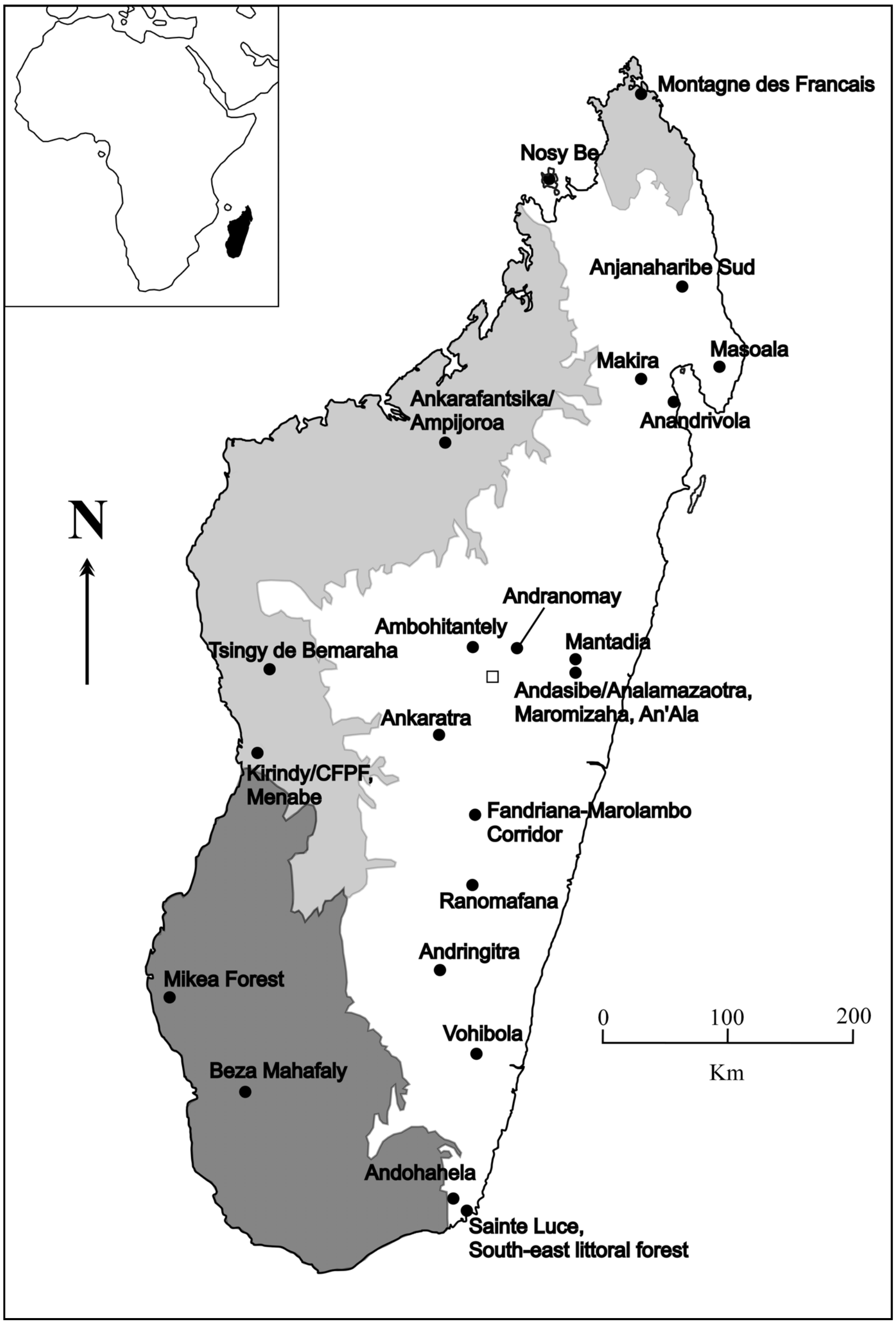

Fig. 4.4 Map of Madagascar's bioclimatic regions (following Cornet (1974)) showing locations mentioned in text and the capital Antananarivo (white square). White, humid and sub-humid regions; light grey, dry region; dark grey, sub-arid region.

present in unburnt forest, as well as an additional ten understory species. By creating an index of conservation value based on endemism, abundance and threatened status, these authors calculated that the conservation value of secondary forests was greater than that of unburnt forests. Andrianarimisa (1992) also recorded an increase in species richness in secondary and selectively logged forests compared to non-degraded forests, noting that these habitats occur in a mosaic pattern on the landscape scale, thereby increasing structural complexity and providing a greater number of niches for birds. Woog *et al*. (2006) found a higher species richness in forest edges than in the interior or agricultural matrix habitats within the Maromizaha forest, due to the presence of species typical of open habitats at the forest edge. These authors, however, do not appear to have controlled for survey effort between habitat types.

Watson *et al*. (2004a) working in the southeast littoral forest, observed the forest interior to be significantly richer than the forest edge and secondary matrix habitats (primarily *Erica* scrub), although total bird abundance between interior and edge was similar. Of the 45 species recorded only in forest, 68 % were found to be edge-sensitive or intolerant. However, a second study found no significant change in bird communities following habitat change. In the selectively logged dry forest of Kirindy/CFPF, Hawkins and Wilmé (1996) did not detect any significant change in the number of species or individuals between logged and unlogged forest areas, speculating that this may be due to the absence of rarer species from transects or the general absence of disturbance-sensitive guilds from the Kirindy/CFPF forest. Pons *et al*. (2003a) also recorded similar species diversity and abundance in both forest and matrix habitats immediately following a fire in the dry forest of Ankarafantsika.

Finally, Goodman *et al*. (1996) compared the bird communities of a mature plantation of indigenous *Weinmannia bojeriana* on the Ankaratra Massif, with humid forests from the same altitudinal band in Anjanaharibe-Sud and Andohahela. They observed the bird communities to be “more-or-less similar” between the three sites in terms of species richness, although the Andohahela community included a greater proportion of forest-dwelling species. The plantation community, however, possessed a lower proportion of endemic species (43.2 %, compared to 59.0 % at Anjanaharibe-Sud and 50.0 % at Andohahela), and was particularly poor in species endemic at higher taxonomic levels; no members of the Brachypteraciidae, Mesitornithidae, Couinae or Philepittinae were recorded, and only two forest-dwelling species of endemic genera were observed (compared to 16 at Anjanaharibe-Sud and eight at Andohahela). The Ankaratra plantation was also depauperate in birds restricted to higher elevational zones, being populated mainly by altitudinal generalists.

4.3.2 Birds: impacts on foraging guilds

Several researchers have demonstrated that the sensitivity of birds to habitat degradation is related to a given species' foraging behaviour, and that the relative abundance of foraging guilds therefore differs in habitats of varying quality or structure. In Ankarafantsika, Pons and Wendenburg (2005) found that both degraded and non-degraded forests are dominated by canopy and understory insectivores, followed by frugivores and aerial feeders. Furthermore, after such zones were converted to savannah, the bird community was dominated by granivores and aerial insectivores. At the same site, Andrianarimisa (1992) observed that granivores, frugivores, understory insectivores and aerial insectivores reach higher abundance in degraded forests, while terrestrial insectivores decrease in abundance.

In the southeast littoral forest, Watson *et al*. (2004a) found all frugivore species to be edge-sensitive, as well as 88 % of canopy insectivores, 46 % of terrestrial insectivores and 25 % of understory insectivores. Granivorous species and raptors, on the other hand, were found to be more abundant within the non-forest matrix or at the forest edge. Watson *et al*. (2004b) reported similar results in an investigation of bird responses to habitat fragmentation, observing that canopy insectivores and large canopy frugivores are the most sensitive guilds to reduction in fragment size. Goodman *et al*. (1996) noted the relative scarcity of insectivores and the relative over-representation of raptors within plantation forests at Ankaratra. Finally, Langrand and Wilmé (1997) observed a decrease in understory and canopy insectivores and nectarivores from the smallest patches at Ambohitantely. These findings correspond closely to those of Gray *et al*. (2007) in a global review of foraging guild responses to habitat disturbance, indicating that the Malagasy avifauna does not respond differently to that of other tropical regions.

4.3.3 Birds: endemic and non-endemic species

Wilmé (1996) states that "the tolerance of the endemic forest avifauna to forest degradation is proportional to its degree of taxonomic endemism", finding that of 32 endemic forest bird genera, 27 do not occur in secondary forests or anthropogenic grasslands (although some may occur in lightly degraded habitats). The literature reviewed here, much of it published after Wilmé's (1996) review, finds mixed support for this statement, although there is a clear geographical and taxonomic bias in the research conducted to date.

Four studies directly investigated the response of individual endemic species or guilds to habitat modification, all focused on a suite of medium-sized, dry forest, terrestrial insectivores that are endemic at higher taxonomic levels. Hawkins (1993, 1994) found that *Mesitornis variegata* in Menabe benefited from the higher density of the shrub layer following logging, which provided greater cover from predation and heat, and an increase in potential nest sites. Likewise, Seddon and Tobias (2007) observed that *Uratelornis chimaera* nesting density is greater in lower stature or degraded habitats in the Mikea Forest, and that pairs will preferentially nest alongside a path or clearing if available in their territory. This is in contrast to the sympatric *Monias benschi*, which avoided more degraded habitats (Seddon *et al*. 2003). In studies of *Coua* spp. in Kirindy/CFPF (Chouteau 2004) and Ankarafantsika (Chouteau *et al.* 2004), responses were found to vary according to species and type of habitat change. In Kirindy/CFPF, *C. coquereli* density increased but *C. gigas* density decreased significantly in selectively logged forest; the former species preferred denser understory vegetation which probably enhanced prey availability and cover from predators (Chouteau 2004). In Ankarafantsika, however, *C. coquereli* was less abundant in once-burnt forest than in unburnt areas, in contrast to *C. ruficeps*, which was more abundant in once-burnt forest and occurred in twice-burnt areas (Chouteau *et al*. 2004). Hawkins and Wilmé (1996) also found *C. coquereli* to be more abundant in logged areas in Kirindy/CFPF, where *Schetba rufa* was the only species with a significant negative response to logging.

Of the 12 species belonging to endemic or near-endemic families and sub-families in Pons and Wendenburg's (2005) study in Ankarafantsika, two were more common in unburnt forests (*Coua ruficeps* (in contrast to Chouteau *et al*. 2004) and *Vanga curvirostris*), one was more common in regenerating forests (*Leptopterus chabert*) and the rest showed no significant change. Also in Ankarafantsika, Andrianarimisa (1993, cited in Pons *et al*. 2003a) found that non-endemic birds attained higher relative abundance in degraded than in intact forests.

Notably less research has been carried out on endemic birds in humid forests than in the dry forests. In a study of fragmentation in Ambohitantely, Langrand and Wilmé (1997) found five species to be significant edge avoiders (*Atelornis pittoides, Phyllastrephus* [=*Bernieria*] *madagascariensis, Newtonia ampichroa, Calicalicus madagascariensis* and *Foudia omissa*); of these all are endemic, two are members of endemic families and two belong to a near-endemic family. While Watson *et al*. (2004a) did not analyse their data by endemism, they do note that *Foudia omissa* is an edge-tolerant species in littoral forest fragments, in contrast to the situation at Ambohitantely where this species is an edge-avoider (Langrand 1994; Langrand and Wilmé 1997). These results therefore indicate that the same species may respond differently to edge effects in different habitats.

4.3.4 Reptiles and amphibians

Jenkins *et al*. (2003) measured chameleon density in low-disturbance (selectively logged) and high-disturbance (post-fire regeneration) forests at Andranomay, as well as in riparian areas. Of the four *Calumma* and two *Brookesia* spp. present at the study site, only *B. minima* was absent from forests regenerating after fire, while three of the four *Calumma* species were significantly more abundant in low-disturbance compared to high-disturbance forest. Metcalf *et al*. (2005) measured the edge effect from forest paths on chameleon abundance in Ankarafantsika, and found both *Furcifer oustaleti* and *F. rhinoceratus* at significantly greater densities with increasing proximity to paths. This result is similar to that of Jenkins *et al*. (1999), who found *C. brevicornis* and *B. nasus* to occur at higher densities along well-established forest paths than in randomly placed transects within the forest interior. The only non genus-specific reptile study was carried out in Kirindy/CFPF, where Bloxam *et al*. (1996) found selective logging had no significant effects on either assemblage composition or species abundances.

Amongst research into frog communities, Vallan *et al*. (2004) found that selective logging had no significant impact on amphibian abundance or species richness in An'Ala forest, but noted a marked shift in community composition from specialist to generalist species; terrestrial species (Mantellinae) were scarcer, and arboreal species (Boophinae and Cophylinae) were more abundant in logged compared to unlogged forests. A similar result was found by Andreone (1994) in Ranomafana, where frog communities were surveyed in undisturbed forests and a range of different degraded habitats (such as roadsides and banana plantations). This author found a higher proportion of arboreal species and a lower proportion of terrestrial species in degraded forests compared to non-degraded areas, with no indication of a difference in species richness. Of the 40 species recorded, 27.5 % were restricted to non-degraded forests, 22.5 % were limited to degraded areas, and 50 % occurred in both habitat types. In a study of fragmentation at Ambohitantely, Vallan (2000) also found differences in disturbance-sensitivity (in this case edge effects) related to life-history; species which do not live in water (most microhylids, *Mantidactylus malagasius*, *M. aglavei*) were absent in small fragments due to the greater relative influence of edge effects, while stream-dwelling species were not affected.

Vallan (2002) studied frog communities in relatively undisturbed forest, secondary forest, *Eucalyptus* plantations and rice-fields in Andasibe, and found significant differences in species richness between habitat types; secondary forests harboured only 54 %, plantations 46 % and rice-fields 12 % of the total number of species recorded in relatively undisturbed forests. Habitat preferences were found to reflect taxonomy, with the Hyperoliidae and Raninae [= Ptychadenidae] occurring only in rice fields, while Mantellinae, Microhylidae and the genus *Boophis* were more species rich in relatively undisturbed forests.

Reptile and amphibian communities in littoral forest fragments in southeast Madagascar have been observed to display strong seasonal differences in edge sensitivity (Lehtinen *et al*. 2003). All frogs were found to be significant edge-avoiders during the dry season, but during the rainy season a mix of responses were recorded, with one species (*Mantidactylus boulengeri*) being an edge-avoider, one being an interior-avoider (*Heterixalus boettgeri*) and the others showing non-significant differences in distribution. Among reptiles, three species (*Mabuya* [=*Trachylepis*] *elegans*, *Phelsuma lineata* and *Geckolepis maculata*) were interior-avoiders and none were edge-avoiders during the wet season. The authors also found a significant correlation between the strength of edge-avoidance and extinction risk (estimated on the basis of absences from sampled fragments), although this was non-significant when only reptiles were considered.

4.3.5 Small mammals: community impacts

Based on knowledge of the species' ecological requirements, Ganzhorn *et al*. (1996) hypothesised that selective logging would favour *Mus musculus*, *Suncus madagascariensis*, and *Geogale aurita*, but would reduce habitat quality of the spiny tenrecs (Tenrecinae) *Tenrec ecaudatus*, *Setifer setosus*, and *Echinops telfairi* at Kirindy/ CFPF. Nonetheless, trapping results showed no significant effect of logging on either *S. setosus* or *T. ecaudatus*. Similarly, Ganzhorn *et al*. (1990) found that selective logging had no impact on the density of *T. ecaudatus*, but did slightly impair habitat quality for *E. telfairi*. All of the spiny tenrec species mentioned above are known to be tolerant of habitat degradation and anthropogenic environments (Nicoll and Rathbun 1990; Ganzhorn *et al*. 2003).

In Kirindy/CFPF, the rodent *Macrotarsomys bastardi* was not captured outside of relatively intact forests, but *Eliurus* spp. occurred in all forest types (including secondary forests dominated by the invasive *Ziziphus mauritiana*) within 50 m of the ecotone with relatively undisturbed forest (Ganzhorn 2003). *Eliurus webbi* was not captured outside of the least disturbed littoral forest fragments (Ramanamanjato and Ganzhorn 2001).

Amongst studies carried out within humid forest ecosystems, Lehtonen *et al*. (2001) investigated the density of rodents across a gradient of habitat degradation from intact to heavily-logged forest and secondary scrub. Although abundance of endemic rodents was higher in relatively undisturbed than in secondary forest, no species were most abundant in undisturbed habitats. Both *Nesomys rufus* and *N. audeberti* were absent from secondary scrub, although the former was most abundant in selectively-logged forest while the latter was most abundant in heavily-logged forest. All three *Eliurus* spp. occurred in heavily-logged forest. Low-intensity selective logging was found to have little effect on the native rodent fauna, but high-intensity logging was associated with the loss of endemic species.

Stephenson (1993) compared small mammal richness and abundance between one secondary forest and three relatively undisturbed forest sites in Analamazaotra. Four species were only recorded in the least disturbed area, where species richness was greatest but total abundance lowest, and two species were only recorded from secondary forest. The lowest species richness was recorded from a site heavily impacted by tourism. Stephenson (1995) also investigated small mammal abundance in relatively undisturbed and logged secondary forest at Anandrivola. Endemic species richness was higher in relatively undisturbed rather than secondary forest, as was the abundance of two species (*Microgale talazaci* and *E. webbi*). Three endemic species (*Oryzorictes hova*, *M. pulla* [=*parvula*] and *Nesomys rufus*) were not recorded in secondary forest, while *Hemicentetes semispinosus* and the introduced *Rattus rattus* and *Suncus murinus* were not trapped in relatively undisturbed forest.

Rasolonandrasana and Goodman (2006) compared small mammal faunas in an area of recently burned forest with an adjacent control site in the upper elevational sclerophyllous forest of Andringitra. They found that *Eliurus minor* and *Nesomys rufus* did not recolonise burned patches after three years, but these species were at the limit of their altitudinal range and therefore occurred at low densities at this elevation. In contrast, *M.*

longicaudata and *M. fotsifotsy* were more abundant in the burnt space, while no significant differences were found between the abundance of *M. cowani*, *M. dobsoni*, and *Monticolomys koopmani* in burnt and unburnt patches. When introduced rodents were excluded, the burnt area had a lower total rodent density than the control site. Finally, Goodman *et al*. (1996) investigated small mammal presence and density in a plantation forest of native trees at Ankaratra, and compared the observed fauna to that of other sites. They found no significant differences in the species richness and abundance of Lipotyphla [=Afrosoricida] between the plantation and natural forests in Anjanaharibe-Sud, although relative density of endemic rodents was significantly lower within plantations than in natural forests.

4.3.6 Small mammals: effects of introduced species

Compared to other taxonomic groups, research into the tolerance of endemic small mammals to habitat degradation is confounded by the presence of introduced invasive species (e.g. *Rattus rattus* and *Mus musculus*) which, although they are capable of penetrating relatively undisturbed forest, occur at highest density in degraded and anthropogenic habitats. The interactions between endemic and introduced small mammals have been discussed elsewhere (e.g. Goodman 1995) and are beyond the scope of this paper. In brief, the diets of introduced and endemic species do overlap to varying extent (Goodman and Sterling 1996), but there appears to be no firm evidence of a reduction in endemic small mammal density or species richness as a result of competition with introduced species. Ramanamanjato and Ganzhorn (2001) failed to find evidence of competitive exclusion within littoral forest fragments, but suggest that *R. rattus* may outcompete native species within degraded or marginal habitats that may otherwise have supported endemic species.

4.3.7 Lemurs

In a series of papers investigating the influence of edge effects on lemur communities in the Vohibola III Classified Forest, Lehman and colleagues observed a high degree of interspecific variation in edge-tolerance. Lehman (2007) found *Eulemur fulvus rufus* [= *E. rufus*] to occur at lower densities at the edge compared to interior of forests, while *Lepilemur mustelinus* showed no significant spatial variation in density. The author suggests that this pattern can be explained by the distribution of food; edge effects have a greater impact on the distribution of fruit resources than on leaf resources, with the result that folivorous species demonstrate greater edge tolerance than frugivores. Lehman *et al*.

(2006a) surveyed eight species along transects at varying distances from the forest edge, collecting sufficient data to analyse the distributions of four species. Of these, three species (*E. rubriventer*, *Avahi laniger*, and *Microcebus rufus*) demonstrated positive edge effects, while one species exhibited a neutral edge effect (*Hapalemur griseus*). For a further three species (*Propithecus diadema edwardsi* [= *P. edwardsi*], *L. microdon*, and *E. f. rufus* [= *E. rufus*]), insufficient observations were made to provide statistically significant results, but descriptive evidence indicates that these species are more likely to show a neutral than a negative edge effect. It was predicted that *E. rubriventer*, being a frugivorous species, would exhibit a negative edge effect. However, this hypothesis was not supported by the data, possibly because the study was conducted at a time of low fruit set when the species behaves more as a folivore/frugivore. In a second study in the same forest, Lehman *et al*. (2006b) found three species showed neutral edge effects (*Avahi laniger*, *E. rubriventer,* and *H. griseus*), two species demonstrated positive edge effects (*P. d. edwardsi*, and *M. rufus*), and one species exhibited a negative edge effect (*Cheirogaleus major*). These findings are explained largely by the distribution of food resources; the three species showing neutral edge effects are folivorous or frugivorous/ folivorous during the survey period, while the two edge-preferring species have varied diets.

Ganzhorn (1995) measured the effects of low-intensity selective logging on primary production, leaf chemistry, and lemur populations in the forest of Kirindy/CFPF, and found that the greater light penetration in logged areas results in enhanced leaf quality (which compensates for reduced leaf biomass) and increased fruit production in tree species favoured by frugivorous lemurs. For all lemur species present at the study site (*Eulemur fulvus*, *Propithecus verreauxi*, *Cheirogaleus medius*, *Lepilemur mustelinus*, *Microcebus* spp., *Mirza coquereli*, and *Phaner furcifer*), observed densities were higher in logged than in unlogged forest, although these relationships were only significant for three species. The author stresses that the results should be interpreted with caution, and for some species may represent shifts in home ranges to take advantage of seasonal leaf flush/fruiting rather than genuine increases in population density. While low-level forest disturbance may be beneficial for lemurs, encounter rates of most of the lemurs of the dry deciduous forest decreased with higher intensity logging.

Merenlender *et al*. (1998) monitored the demographics of *Varecia variegata rubra* [= *V. rubra*] and *Eulemur fulvus albifrons* [= *E. albifrons*] in both undisturbed and selectively logged forest on the Masoala Peninsula; they found no significant differences in population density, group size or female fecundity between forest types in either species, but did observe a female-biased adult sex ratio of *E. f. albifrons* in selectively logged forest. This observation is explained by an increase in male dispersal in disturbed forests due to insufficient food resources, indicating that these areas may function as population sinks for this species. In contrast, studies of *Varecia* elsewhere have shown *V. v. variegata* to be susceptible to forest degradation. Balko and Underwood (2005) found the species to be absent from logged areas in Ranomafana for over a decade post-harvest, while White *et al*. (1995) also noted that the species was sensitive to forest conditions.

Two studies have looked at the distribution and density of lemur faunas in forest patches across a landscape. Rasolofoson *et al*. (2007) surveyed lemur populations at 12 sites within the Makira forest and observed 14 species; they discovered that the relative densities of 11 species were negatively correlated with the density of cut stumps, although the surveyed forests were heavily hunted and disturbed, so it was not possible to disentangle the effects of selective logging from other factors. Lehman *et al*. (2006c) surveyed seven sites within the Fandriana-Marolambo Corridor and found lemur diversity to be significantly negatively correlated with both altitude and agricultural intensity. The relationships between lemur diversity and both hunting pressure and logging intensity were also found to be negatively correlated, although these results were statistically insignificant.

Andrianasolo *et al*. (2006) investigated the association between nocturnal lemur density and structural vegetation parameters across a gradient of degradation in the southeast littoral forests. *Microcebus* spp. was the least specialised in terms of microhabitat use, whereas *Cheirogaleus* spp. and *Avahi laniger* were the most. *Avahi laniger* was observed to select areas with a high density of large trees, and the authors suggest that this preference is due to the mode of locomotion of this species rather than due to the distribution of food resources.

In Menabe, Ganzhorn and Schmid (1998) observed that *M. murinus* in secondary forest had reduced body mass and survivorship, and occurred at lower density, compared to relatively undisturbed forest, while females were less likely to enter into daily torpor and hibernation in secondary forests. The authors suggested that both the higher ambient temperatures and the relative scarcity of large trees with tree holes in secondary forests inhibited energy-saving torpor which, coupled with decreased food abundance, reduced the ability of the species to survive the dry season. These findings were corroborated by Ganzhorn (2003), who observed that although *M. murinus* were found in all relatively undisturbed and secondary forest formations in Menabe, capture rates decreased in secondary forests of increasing size, indicating that these habitats were suboptimal for this species. In the southeast littoral forests, however, Ramanamanjato and Ganzhorn (2001) failed to capture *M. murinus* in secondary formations over three years of trapping, even though the species is known to utilise plantations elsewhere in the region. Within relatively undisturbed forests, capture rates for *M. murinus* were higher in areas with greater canopy cover.

The habitat utilisation of two sympatric *Microcebus* species (*M. murinus* and *M. ravelobensis*) was investigated in two forest areas in Ankarafantsika by Rendigs *et al*. (2003). They found both species in a non-degraded forest with a high density of large trees, but only *M. ravelobensis* in the more degraded area. The degraded area contained fewer large trees with tree holes, which were thought to be a critical resource for *M. murinus* but not for *M. ravelobensis*.

Arrigo-Nelson and Wright (2004) examined the distributions of *Hapalemur* spp. in relation to the distribution of bamboo and human disturbance. They found that small-culm bamboo occurred at greatest density at forest edges and within light gaps, and thus *H. griseus* and *H. aureus* are likely to use forest edges. This edge-tolerance may have adverse effects on the species, exposing them to greater hunting pressure.

The presence of *Eulemur collaris* in littoral forest fragments has been examined in relation to patch size and habitat characteristics (Ralison *et al*. 2006). The lemur was present in only four out of 10 patches, and did not occur in patches smaller than 220 ha in area. Occupied patches were characterised by greater canopy height, canopy cover, tree diameter, and density of pandan trees (*Pandanus* spp., Pandanaceae) (i.e. were less degraded) than patches in which the species was absent. This study, however, was unable to disentangle the effects of habitat degradation, patch size, and edge effects. In constrast, *E. m. macaco* is known to survive in both degraded and well-established secondary forests on Nosy Be and the adjacent mainland, as well as areas where it can feed on cultivated fruit trees, which support greater population densities than relatively undisturbed forest (Bayart and Simmen 2005). Cultivated fruit trees have also been observed to attract other species when adjacent to natural forest (e.g. *E. coronatus* and *Daubentonia madagascariensis* feeding in mango trees (*Mangifera indica*, Anacardiaceae) on Montagne des Français (*pers. obs*.).

The use of plantation forests by lemurs has been investigated at Analamazaotra (primarily *Eucalyptus*) and Ampijoroa (mixed-species plantations) (Ganzhorn 1987). At Analamazaotra, three species regularly used and foraged within mature *Eucalyptus* plantations with dense undergrowth (*Microcebus rufus*, *Cheirogaleus major* and *Eulemur f. fulvus* [= *E. fulvus*]). Three small folivorous species (*Avahi laniger*, *Hapalemur griseus,* and *Lepilemur mustelinus*) were observed to occasionally forage within these plantations, while *Indri indri* made only occasional use of plantations to cross between areas of natural forest. No lemur species were ever recorded within young *Eucalyptus* plantations lacking substantial understory vegetation. At Ampijoroa, three species were either resident or regularly observed within mixed-species plantations contiguous with natural forest (*Propithecus verreauxi*, *L. mustelinus* and *E. fulvus*). This mixed-species plantation was used to a much greater extent by lemurs than mature *Eucalyptus* plantations in the east. No lemur was recorded within a mature plantation of the indigenous *Weinmannia bojeriana* in Ankaratra (Goodman *et al*. 1996).

4.3.8 Bats

Compared to non-volant mammal species, relatively little research has been carried out into Malagasy chiropterans (Goodman *et al*. 2003), in particular with regards to habitat utilisation (Kofoky *et al*. 2007). Kofoky *et al*. (2007) used trapping and acoustic sampling to investigate bat species composition, abundance, and activity in three habitats (forest interior, clearings, and forest edge) within and around Tsingy de Bemaraha National Park. Results differed according to the method employed; while four species (*Triaenops rufus* [=*T. menamena*], *T. furculus*, *Miniopterus manavi*, and *Myotis goudoti*) were strongly associated with the forest interior based on trapping data, acoustic sampling revealed *T. rufus* and *M. manavi* at the forest edge. Overall, bat activity differed significantly between habitats, being lowest within the forest interior and highest at the interface of forest and agricultural land. As trapping within the forest interior was conducted along trails adjacent to cave roosts, the authors hypothesised that species trapped in the forest were likely using these trails as thoroughfares leading to the forest edge where most foraging takes place. If this is the case, the findings suggest that while forests are important in providing shelter around cave roosts, forest edge constitutes a more important foraging habitat for microchiropterans.

Similar results were found in the humid forests of Mantadia and Analamazaotra, with the highest taxonomic richness of any site being recorded within relatively intact forest areas, but the greatest activity levels occurring within *Eucalyptus* plantations and agricultural land (Randrianandrianina *et al.* 2006). Four species were captured solely within intact forest, two species within agricultural land only, and just *Rousettus madagascariensis* in plantations (although overall capture rates were low). Based on acoustic sampling, *Myotis goudoti* was the only species to be strongly associated with relatively intact forest. A matrix of non-forest land surrounding protected humid forests contributes greatly to chiropteran diversity by providing roosts and foraging areas for species that do not use forests (Randrianandrianina *et al.* 2006), as was demonstrated for western dry forests (Kofoky *et al*. 2007).

The results from the above studies are corroborated by Goodman *et al*. (2005) who, based on surveys throughout the dry regions of Madagascar, discovered that no more than five out of 27 recorded species are strictly dependent on large expanses of intact forest. In addition to the species studied above, others known to use anthropogenic habitats and non -indigenous food plants include the fruit bats *Eidolon dupreneanum* (Ratrimomanarivo 2007) and *Pteropus rufus* (Hutcheon 2003). One striking example of a species potentially benefiting from forest degradation is the sucker-footed bat *Myzopoda aurita* of the endemic family Myzopodidae. This species roosts in the travellers' tree *Ravenala madagascariensis* (Strelitziaceae), and has benefited from the elevated densities of the plant within secondary and degraded habitats (Eger and Mitchell 2003).

4.3.9 Invertebrates

Olson and Andriamiadana (1996) compared the leaf litter invertebrate faunas of unlogged, selectively logged and edge habitats (along logging trails) in the dry forests of Kirindy/CFPF. Edge habitats were found to support a lower number of individuals and total biomass, but species richness and community composition did not differ significantly between forest edge and unlogged forest blocks. Selectively logged sites demonstrated similar species richness and composition to unlogged sites.

4.3.10 Vegetation

Cadotte *et al*. (2002) surveyed the vegetation of three littoral forest fragments in Sainte Luce and one fragment in Lokaro; the Lokaro fragment is closer to the sea, more isolated, and more heavily impacted by local communities. As a consequence, the Lokaro forest area contained significantly fewer species and families, lower tree density and Shannon-Wiener diversity, and was largely dominated by one tree species, *Tambourissa purpurea* (Monomiaceae). This study, however, did not directly measure degree of anthropogenic disturbance between plots, and so was unable to disentangle the effects of edaphic factors, isolation, and degradation on the observed patterns.

Sussman and Rakotozafy (1994) investigated forest structure and composition in three areas at Beza Mahafaly, comprising gallery forest on humid soils, spiny thicket on dry soils (both of which were fenced and therefore not subject to grazing), and an unfenced patch of spiny thicket on dry soils. Tree density was lower in the grazed area than in the fenced spiny thicket area, but higher than in the gallery forest. The grazed forest also had a greater proportion of indigenous trees than either of the fenced areas, although indigenous tree species richness was reduced. In seedling plots established in the same forest patches, no appreciable differences were found in the proportion of saplings and juveniles of middle and upper stratum trees, indicating that grazing was not having negative impacts on the regeneration of these species (although grazing intensity was not quantified). Grasses, however, were proportionally more common within the grazed area, and herbs less common, presumably as a result of preferential grazing of herbs by livestock. The authors note that the indigenous vine *Sarcostemma viminale* (Asclepediaceae) behaves as an invasive within this forest and was controlled by grazing prior to the establishment of the reserve; conservation grazing is therefore under consideration as a management tool in order to control this species (M. Nicoll, *pers. comm.*).

Brown and Gurevitch (2004), working in Ranomafana, investigated differences in vegetation structure and composition between unlogged forest, forest selectively logged 50 years previously, and forest clear-cut and subsequently abandoned years 150 previously. They found that logging significantly decreased species diversity and increased invasion by exotic plants, primarily *Psidium cattleianum* (Myrtaceae). The presence and abundance of native plants were significantly different between logged and unlogged areas, but no such trends were apparent between the two types of previously logged stand. Once established, *P. cattleianum* formed dense monospecific stands which prevented the establishment of native species, such that selectively logged forests may never recover their original structure or composition once invaded. Invasive exotic plants are also known to prevent regeneration within degraded forests in other regions of Madagascar (e.g. *Ziziphus mauritiana* in Menabe (Ganzhorn 2003), *Opuntia* spp. in the south (Middleton 1999), and *Acacia* spp. in the central highlands and eastern escarpment (Kull *et al*. 2008)).

A number of studies have examined vegetation responses to forest degradation, in terms of structure (Table 4.1). It was not possible to formally assess variation as a result of different impacts or within bioclimatic regions, due to the wide range of methodologies and broad diversity of variables measures/estimated. However, it appears that vegetation responses tend to be relatively uniform, with degradation leading to a reduction in mean canopy height and cover, tree density, tree diameter, tree species richness and leaf litter cover, and an increase in the density of herb and shrub layers. In general, vegetation structural complexity is higher in non-degraded forests.

4.3.11 Overall effects on biodiversity

There was a notable level of variation in the impacts of habitat change on Madagascar's terrestrial biodiversity in terms of species abundance/density and assemblage composition, both between and within different taxonomic groups. Of the studies in which the data generated explicit conclusions, 10 papers found that habitat change had a positive impact on biodiversity, 14 demonstrated a neutral or mixed response, and 20 demonstrated a negative response. Differences in methodology, focal species, ecosystems, and types of anthropogenic impacts between these studies make it difficult to draw out general conclusions. In some cases, species responses were highly idiosyncratic, including different responses to the same type of habitat degradation between members of the same genus (*Coua*; Chouteau 2004; Chouteau *et al*. 2004), and the same species (*Foudia omissa*) in different regions (Langrand and Wilmé 1997; Watson *et al*. 2004a). These findings suggest that, given the diversity of Madagascar's biota, ecosystems and social-ecological interactions, it is probably unrealistic to search for rules-of-thumb to explain and predict the impacts of habitat change across the country.

Table 4.1 Summary of vegetation responses to a range of anthropogenic impacts on forests in Madagascar. DBH = diameter at breast height

Source	Bio-climatic region	Type of impact	Vegetation response
Ganzhorn *et al*. (1990)	Dry	Selective logging	Increase in trails, logging debris and density of herb-layer, decrease in canopy tree size, number of species, depth of litter layer and regeneration of woody species. Predicted to lead in long-term to decline in tree species diversity.
Chouteau (2004)	Dry	Selective logging	Understory vegetation denser and canopy more sparse in logged areas.
Hawkins (1993)	Dry	Selective logging	Increase in understory vegetation in logged areas.
Hawkins and Wilmé (1996)	Dry	Selective logging	Leaf litter depth and large tree density decrease, herb layer and shrub layer increase in logged areas.
Bloxam *et al*. (1996)	Dry	Selective logging	Reduction in canopy cover.
Ganzhorn *et al.* (1996)	Dry	Selective logging	Increase in herb layer, woody understory and rotting wood, reduction in tree density.
Ganzhorn (1995)	Dry	Selective logging	Increased light availability to remaining trees increases foliage quality and fruit production.
Ganzhorn and Schmid (1998)	Dry	Selective logging	Tree densities significantly lower in secondary forest for all size classes. Tree species richness and diversity significantly higher in relatively undisturbed forest.
Vallan *et al*. (2004)	Dry	Selective logging	Forest gaps colonized by heliophilous vegetation.
Lehtonen *et al*. (2001)	Dry	Selective logging	Herbaceous cover increased and canopy cover decreased with increasing degradation.
Balko and Underwood (2005)	Dry	Selective logging	Reduction in tree height and diameter, and increase in tree dispersion, in most degraded site compared to least degraded.
Rasolonandrasana and Goodman (2006)	High Mountai n	Fire	Only 1 of 7 forest tree species survived fire, and regeneration of other species was low. Thicket species survive better and dominate regenerating cohort.
Chouteau *et al*. (2004)	Dry	Fire	Increase in understory vegetation and decrease in canopy cover following fire.
Pons and Wendenburg (2005)	Dry	Fire	Increase in density of understory vegetation.
Jenkins *et al*. (2003)	Dry	Fire	Bamboo and ground fern density significantly lower in regenerating forest.

Source	Bio-climatic region	Type of impact	Vegetation response
Lehman (2007)	Humid	Edge effects	Trees at forest edge had significantly lower height and diameter than in forest interior, and lower stem density.
Lehman *et al.* (2006a)	Humid	Edge effects	Significant negative edge effects on the density and size of lemur food trees.
Lehman *et al.* (2006c)	Humid	Edge effects	Tree height and diameter lower at forest edge.
Watson *et al.* (2004a)	South eastern littoral	Edge effects	Forest edge has significantly greater shrub cover, but reduced canopy cover and leaf litter cover.
Arrigo-Nelson and Wright (2004)	Humid	Edge effects	Small-culm bamboo is more abundant in forest edges and gaps.
Vallan (2002)	Humid	Secondary forest	Undisturbed forest has greater vegetation structural complexity.
Stephenson (1995)	Humid	Secondary forest	Light penetration, woody stem density and tree fern density higher in secondary forest; woody species number, herbaceous species number, herbaceous stem density, percentage herbaceous cover, fallen log dispersion, percentage exposed rock and liana stem density higher in relatively undisturbed forest.
Andrianasolo *et al.* (2006)	South eastern littoral	General degradation	Density of small (DBH < 10cm) and especially large (DBH > 10cm) trees is reduced in more degraded patches.
Ralison *et al.* (2006)	South eastern littoral	General degradation	Average tree height and diameter decrease with increasing degradation.
Seddon and Tobias (2007)	Sub-arid	General degradation	Reduction in canopy height and leaf litter depth in degraded areas.
Metcalf *et al.* (2005)	Dry	Paths	Thin opening of canopy above paths.

4.3.12 Impacts by forest ecosystem type

Although it is not possible to perform a formal meta-analysis of the reviewed papers, the data appear to indicate that there may be differences in the degree of disturbance sensitivity of faunal taxonomic groups between humid and dry forest ecosystems (Table 4.2). In total, 61 % of studies carried out within humid forests demonstrated negative impacts on biodiversity associated with habitat change, while 19 % of studies carried out in dry forests showed habitat change to have a negative impact.

While these findings appear to indicate that dry forest taxa are less disturbance-sensitive than those in humid forest, the results should be interpreted with caution and a number of caveats must be stressed. Firstly, the humid forest sample largely comprised lemur research (generally demonstrating negative impacts), while the dry forest sample was biased by research into birds (demonstrating positive and neutral impacts), and the avifauna is clearly less forest-dependent that the order Primata. Secondly, it is important to note that the extent and intensity of habitat degradation has not been controlled for, and much of the research from dry forests has been carried out at Kirindy/CFPF where modifications are restricted to low-intensity logging.

If faunal species and communities of dry and humid forests do respond differently to habitat change, this may reflect variation in the degree of heterogeneity between the ecosystems, and thus the diversity of habitat specialisation amongst their constituent species. The literature suggests two possible mechanisms for such differences. Seddon and Tobias (2007) note that because of the "sparse, deciduous nature" of the Mikea forest, an increase in light within and around forest gaps and edges makes little difference to the understory. Dry forests may therefore be less susceptible to the impacts of habitat disturbance because natural light penetration is greater compared to the darker humid forests, meaning that modification has less of an influence on prevailing abiotic conditions. The second hypothesis stems from the high climatic variability of

Table 4.2 Summary of species or community responses of a range of vertebrate groups to anthropogenic degradation of humid and dry forests. Positive and negative impacts refer to increases or decreases, respectively, in either richness or abundance.

Taxonomic group	**Humid ecosystems**			**Dry ecosystems**		
	Positive impact	**Neutral/mixed impact**	**Negative impact**	**Positive impact**	**Neutral/mixed impact**	**Negative impact**
Birds	1	0	1	4	4	0
Amphibians	0	2	1	0	0	0
Reptiles	0	1	1	1	1	0
Herpetofauna	0	0	1	0	0	0
Small mammals	0	1	3	0	2	1
Lemurs	3	1	7	1	0	2
Total	**4**	**5**	**14**	**6**	**7**	**3**

Madagascar's dry regions compared to the humid east, in terms of larger inter-annual variation in rainfall (Dewar and Richard 2007) and greater intra-annual variation in temperature (Donque 1972), which dry forest species may be adapted to.

4.3.13 Effects on community composition

While the reviewed body of research fails to provide an equivocal answer to the question of whether anthropogenic habitat degradation has negative impacts on overall biodiversity, there is strong evidence that degradation leads to community turnover from specialist to generalist species, and/or endemic to non-endemic species. This has been observed in bird (Andrianarimisa 1992; Goodman *et al*. 1996; Watson *et al*. 2004a; Woog *et al*. 2006), amphibian (Andreone 1994; Vallan 2002; Vallan *et al*. 2004) and small mammal communities (Goodman 1995; Lehtonen *et al*. 2001; Rasolonandrasana and Goodman 2006; Stephenson 1993, 1995).

The relative increase of widespread habitat generalists in degraded habitat, at the expense of forest specialists, is a concern for conservationists as it results in the homogenisation of biodiversity (McKinny and Lockwood 1999). This is particularly pertinent in Madagascar where the majority of the endemic biota are forest specialists (Goodman and Benstead 2005), and the level of habitat specialisation and taxonomic level of endemism are correlated for some taxonomic groups (e.g. birds; Wilmé 1996). Habitat degradation is therefore expected to impact most heavily on the island's endemic taxa, which represent the most important and valuable component of the biota.

4.3.14 Source-sink dynamics and population viability

Much of the research presented here suggests that modified or degraded habitats can support levels of species diversity, and population densities of particular species, that can match or even exceed that of undisturbed forests, at least in the short term. However, very little research has investigated the long-term viability of biodiversity in degraded forest areas. It could be that these habitats are sub-optimal for some taxa and thus act as population sinks, being maintained only through immigration from adjacent source areas (Pulliam 1988). For example, Jenkins *et al*. (2003) found four *Calumma* spp. at their post-fire regeneration study site, but speculated that chameleon abundance in these areas is probably maintained by dispersal from contiguous low-disturbance forest, or through recruitment from eggs laid before the fire, and that such secondary forests would probably not support viable chameleon populations in the long term. Likewise, several researchers have reported that *Microcebus* spp. tolerate degraded habitats (Andrianasolo *et al*. 2006; Ganzhorn 1995; Lehman *et al*. 2006a, b), giving the impression that such areas offer suitable habitat for these species. However, Ganzhorn and Schmid (1998) discovered that while *M. murinus* occurred in secondary forests in Menabe, it did so at lower densities than in relatively undisturbed forest and had lower body mass and reduced survivorship; in addition, it was less likely to enter into energy-saving torpor due to the higher ambient temperatures of secondary habitats. Although not explicitly stated by the authors, it is possible that the *Microcebus* population within the secondary forests studied functions as a sink. Finally, while Pons and Wendenburg (2005) found an increase in species richness in regenerating forests compared to relatively undisturbed forests, they suggested that this may “conceal potential demographic problems for certain species” (i.e. *in situ* recruitment may not offset mortality).

4.3.15 Types of habitat change

As one might expect, the papers examined here indicate a positive relationship between the extent/intensity of habitat modification and severity of the impact on either species richness/abundance. Selective logging appears to have the least influence on biodiversity, with many studies reporting either no impact or a positive impact (Bloxam *et al*. 1996; Ganzhorn 1995; Ganzhorn *et al*. 1996; Hawkins 1993; Hawkins and Wilmé 1996; Merenlender *et al*. 1998; Vallan *et al*. 2004). Jenkins *et al*. (2003) discovered that while selective logging did negatively affect chameleon densities, it was less damaging than fire, a finding also observed by Lehtonen *et al*. (2001) for rodent communities. Research into edge effects has provided contradictory results for bird communities (Watson *et al*. (2004a) recorded a decrease in species richness, while Woog *et al*. (2006) found the reverse), and mixed results for herpetofauna (Lehtinen *et al*. 2003) and lemurs (Lehman *et al*. 2006a, 2006b).

Regenerating or secondary forests have been shown to be more species rich than relatively undisturbed forests for birds (Andrianarimisa 1992; Pons and Wendenberg 2005), but less diverse for amphibians (Vallan 2002) and endemic small mammals (Stephenson 1995). Rasolonandrasana and Goodman (2006) found no significant change in small mammal species richness in high mountain sclerophyllous forests regenerating after fire, in comparison to unburnt areas. Although little research has been carried out into the biodiversity value of plantations (representing the greatest degree of human modification), Ganzhorn (1987) found them to be sub-optimal for all lemur species, irrespective of whether the trees were native or exotic, and Vallan (2002) reported that *Eucalyptus* plantations support only 46 % of the number of frog species recorded in natural forests.

4.4 Discussion

The research reviewed herein has focused on the impacts of a range of human actions on Madagascar's biodiversity. The types of anthropogenic habitat change investigated arise both from activities that may be authorised within new multiple-use protected areas, such as grazing and non-timber forest product collection, and others that are not permitted, including industrial selective logging and fire. The latter were still deemed important because the outcomes of such studies are likely to be closely analogous to effects of small -scale selective logging for timber and charcoal production. The insights generated from such research were thus considered directly relevant to the management of multiple-use protected areas.

4.4.1 Future Research Priorities

Assessing different types of habitat modification

The impacts of fire, selective logging, paths, and edge effects on biodiversity have all been the subject of some research in Madagascar (Figure 4.2), in one or more taxonomic groups and across various natural ecosystems. However, certain types of forest use or modification, have been overlooked to date. Perhaps the most important of these is the grazing and browsing of livestock, which would be expected to have some effect on forest structure and composition, as has been demonstrated on other continents (e.g. Enright and Miller 2007; Evans *et al*. 2006; Mata-González *et al*. 2007). Livestock primarily graze and browse vegetation within reach of the ground, clearly influencing patterns of regeneration. If some species are unpalatable to livestock, they might be expected to increase in dominance; this is thought to be the case within the PK32-Ranobe protected area where the spiny *Didierea madagascariensis* (Didiereaceae) appears to be more dominant in heavily grazed areas (P.-J. Rakotomalaza, *pers. comm*.). Domestic livestock may also act as seed predators. For example, in Tsimanampetsotsa National Park, livestock predate the seeds of the southern endemic genera *Lemuropisum* (Fabaceae) and *Androya* (Scrophulariaceae), thereby inhibiting reproduction (MNP 2008).

There are a number of reasons why an understanding of the impacts of grazing is important for the management of new, multiple-use protected areas. Firstly, grazing is prevalent in most protected areas within the dry regions of Madagascar, and it provides a strong incentive for forest conservation by pastoralist communities, as standing forests serve to shelter cattle from rustlers (Kaufmann and Tsirahamba 2006; Rabesahala Horning 2003, although see Réau 2002). In addition, the rearing of cattle is culturally important to the peoples of the dry regions of the island (Evers and van der Zwan 1998). Hence, grazing cannot be restricted within co-managed protected areas without: i) removing incentives for forest conservation; and, ii) risking conflict between protected area authorities and local communities and management structures, leading to the erosion of local support for the protected areas (Gardner *et al*. 2008).

The second major understudied use of forests is the collection of firewood, even though this is amongst the most common and important ways that forests are exploited by local communities throughout Madagascar. While collectors normally target deadwood, and therefore may not impact forest structure, rotting wood provides an important habitat for a number of endemic vertebrate taxa (Glaw and Vences 2007; Stephenson 2003), as well as economically important bee species (Bodin *et al*. 2006). Large scale removal of deadwood from forests could also be expected to disrupt the essential processes of decomposition and nutrient cycling (Golley 1977).

Quantification of habitat modification intensity

Our knowledge of the impacts of forest degradation is hampered by variation in the methods and scope of the studies conducted, most of which fail to explicitly quantify habitat change, instead classifying the degree of alteration using qualitative terms such as ‘low-impact’ and ‘high-impact’. In future, researchers should quantify the intensity of human-induced modification through the measurement of a standardised set of micro- and macro-habitat variables, such as those employed by Stephenson (1995).

Widening the biogeographical focus

Almost two thirds (63.5 %) of reviewed publications were based on studies carried out in the humid and sub-humid regions, including the southeast littoral forests (Figure 4.3), with a further 32.7 % conducted in the dry region. To a large extent this may reflect the existence of forest areas in which conditions or management regimes can provide natural experiments into particular phenomena (e.g. selective logging in Kirindy/CFPF, fragmentation in Ambohitantely and the southeast littoral forests). In contrast, the sub-arid region (spiny forest) of the south is under-represented in terms of existing research.

The lack of information is a concern given that they: i) currently suffer the fastest rates of forest loss in the country (MEFT *et al*. 2009); ii) possess high rates of local endemism for some taxonomic groups (Phillipson 1996; Stattersfield *et al*. 1998); and, iii) are deemed to be particularly susceptible to degradation due to their low capacity for regeneration (Rioux Paquette 2008; Seddon *et al*. 2000; Soarimalala and Raherilalao 2008, although see Elmqvist *et al*. 2007). In addition, little research has been carried out in high altitude areas (two papers), although the biodiversity of the high mountain domain is threatened primarily by climate change rather than human activity (Raxworthy *et al*. 2008).

Improving taxonomic representation

While birds, reptiles, amphibians, small mammals, lemurs, and to a lesser extent bats have received some research attention, there is a paucity of information in relation to other taxonomic groups. Amongst the reptiles, for instance, studies have focused on chameleons (probably reflecting the relative ease of surveying chameleons at night), meaning that little is known about the disturbance sensitivity of other lizards, snakes, or chelonians. The latter is particularly surprising given that all four of Madagascar's indigenous species of terrestrial chelonian are now classified as Critically Endangered (IUCN 2008). Although habitat degradation is not among the main threats suffered by any of these species (the two primary drivers of decline are hunting for domestic consumption, particularly in the case of *Astrochelys radiata* (Leuteritz *et al*. 2005; O'Brien *et al*. 2003) and collection for the international pet trade (Pedrono and Smith 2003; Walker *et al*. 2004)), a deeper understanding of species responses would allow conservation practitioners to advance and implement appropriate management plans.

Bird communities have been well studied in both the dry and humid forest ecosystems of Madagascar, although research centred on the impacts of habitat degradation on particular species is limited to the country's dry regions, and knowledge of the responses of endemic terrestrial, understory, and canopy insectivores of the humid forests is limited. Only one study was found investigating the effects of forest modification on invertebrates (Olson and Andrimiadana 1996). Further research into the disturbance response of invertebrate communities in a range of ecosystems would provide insight into the mechanisms by which habitat degradation influences certain vertebrate species (e.g. Afrosoricida and insectivorous birds). Moreover, understanding the differential impacts of degradation on invertebrate functional groups, such as pollinators and detritivores, is a crucial step in appreciating potential cascade effects and ecosystem decay (defined as the sequential loss of species and erosion of ecosystem function triggered by habitat fragmentation or other disturbances; Laurance *et al*. 2002).

The need for long-term research

The reviewed research consists almost entirely of short-term responses of select taxonomic groups to forest degradation, although few papers reported the elapsed time following disturbance (range 0-45 years, mean = 11.1 years, median = 8 years, n = 9). Therefore, little is known about the long-term effects on ecological processes and ecosystem function resulting from the loss or reduction in density of certain species, or shifts in community composition. In some cases, this may be relatively easy to predict; for example, many lemur species are known to be important seed dispersers (Birkinshaw 2001; Bollen *et al*. 2004a, b; Dew and Wright 1998; Ganzhorn *et al*. 1999; Lahann 2007; Spehn and Ganzhorn 2000), so their disappearance from forests (through hunting or habitat degradation) is expected to lead to a decline in plant species adapted to lemur dispersal (Ganzhorn *et al*. 1999). Other cascade effects, including the loss of insect pollinators dependent on tree holes/deadwood for reproduction, or disequilibrium resulting from declines in predators or competitors, are likely to occur within forests modified by human agency. Further research is needed to identify such impacts, and inform their mitigation, if Madagascar's new generation of multiple-use protected areas are to successfully conserve the country's unique biodiversity.

Most of the studies comprise temporal 'snapshots' of species responses to habitat change, which may not be sufficient to accurately detect emerging population trends as a result of degradation, and may even give the misleading impression that such habitats remain suitable in the longer-term. A number of authors have suggested that while a species may persist in a degraded habitat for some time after disturbance has occurred, local populations are maintained via source-sink dynamics and immigration from non-degraded areas (e.g. Ganzhorn and Schmid 1998; Jenkins *et al*. 2003; Merenlender *et al*. 1998; Pons and Wendenburg 2005). It is essential that long-term research is carried out into the dynamics of disturbance-sensitive taxa (especially those of high conservation priority) to determine whether or not species are subject to extinction debt (Kuussaari *et al.* 2009), if we are to understand the factors influencing their long-term viability and manage protected areas accordingly.

Reconciling resource use and conservation

New multiple-use protected areas, with the twin management objectives of conserving biodiversity and contributing to sustainable rural development, must attempt to simultaneously maximise their value for two different groups of stakeholders. It is not possible, however, to plan protected area management to achieve these duals aims without a detailed appreciation of how forest use influences both conservation and socio-economic values. While the research reviewed here provides an insight into the impacts of habitat change on biodiversity, we have little idea of how it may affect the generation of ecosystem goods and services of local economic or cultural importance. Studies addressing the potential synergies and trade-offs between conservation and forest use are required, therefore, to provide an evidence-base to support management/policy decision-making.

4.4.2 Implications for conservation and protected area management

Protected area design and zoning

Most of the newly-established or proposed Durban Vision protected areas are composed of large expanses of sustainable use zone, with relatively small sections designated for strict conservation. For example, the new protected area of Ankodida has a total surface area of 10,744 ha, of which 2019 ha (18.8 %) is designated as a priority conservation zone, with the remaining 8725 ha composed of zones for activities such as charcoal production and wood exploitation (WWF 2008). The zoning of new protected areas, particularly those that are co-managed by local community associations, must account for human requirements in terms of access to natural resources or culturally important sites (GoM 2006; SAPM 2007). Nevertheless, it remains essential that protected areas be designed so as to maximise the long-term viability of biodiversity.

Taxa that are highly sensitive to degradation may not maintain viable populations in forest areas zoned for extractive use, with the result that undisturbed habitat patches may function as ecological islands within an unsuitable matrix. Without adequate knowledge of the ecological requirements of target species, protected area managers may assume that the species will persist in the long-term throughout the forest mosaic. Research from the fields of island biogeography and meta-population dynamics has long been used to inform protected area design (Whittaker 1998; Whittaker *et al*. 2005), in order to maintain species' and ecosystem viability within human-dominated landscapes, and such theories should equally be applied to the creation of new, multiple-use protected areas in

Madagascar. While the zoning of protected areas is often constrained by other priorities (e.g. social), it is essential that the spatial distribution of conservation, sustainable use, and restoration zones is designed to maximise persistence of disturbance-sensitive conservation targets.

Protected area management objectives

In some circumstances, habitat modification can lead to comparable or higher levels of biodiversity than that in equivalent unmodified habitats, when species richness is used as a metric. However, this can mask a shift in community composition from forest specialists to generalists. All species are not equal in terms of conservation importance, and in Madagascar most endemic species are restricted to forests and show a greater degree of habitat specialisation than non-endemic taxa (Goodman and Benstead 2005; Wilmé, 1996). As species richness is a poor proxy of conservation value (Barlow *et al*. 2007; Gardner *et al*. 2007a), protected area managers must decide whether their objective is to conserve overall levels of biodiversity, or whether they intend to provide effective protection for certain target species, taxonomic groups or ecosystems of conservation importance. Any changes to forest habitats within protected areas as a result of human use will influence a whole range of species; for some the outcome will be positive, but for many it will be negative, and it is the taxa of greatest conservation value that are most likely to experience damaging impacts. Protected area managers must therefore pursue the conservation of species that are of notable importance (e.g. globally threatened species, local endemics or species under-represented within the protected area system). The use of target-driven management planning systems by protected areas in Madagascar should allow managers to focus objectives and actions in this way.

Reconciling natural resource use and conservation

Much of the literature reviewed herein reflects the well-established pattern in ecology that low or intermediate levels of forest disturbance can promote species richness (the ‘intermediate disturbance hypothesis’; Connell 1978). While this implies that Madagascar’s new generation of multiple-use protected areas may be effective in protecting biodiversity if disturbance can be limited to ‘intermediate levels’ (but please see ‘Effects on community composition’ above), protected area managers are currently not in a position to identify the appropriate levels of natural resource use that optimise prevailing conditions for biodiversity. Identification of the modification threshold over which activities become detrimental to species or communities is critical to reconciling forest use and conservation.

Protected area management worldwide is constrained by a lack of funding (Balmford and Whitten 2003; James *et al*. 1999), particularly so in developing countries (Balmford *et al*. 2003; Bruner et *al*. 2004). Given the scarcity of available financial resources and the high costs associated with establishing new protected areas, it is probable that research budgets will not be sufficient to provide the evidence-base necessary to underpin fully-informed management decision-making. Without knowledge of the types and intensity of habitat alterations that are acceptable, in terms of having no significant negative impacts on biodiversity, the precautionary principal (Principal 15 of the Convention on Biological Diversity; CBD 1992) suggests that such forest use should be prohibited or strictly controlled. Adherence to this principal is not, however, an option for the managers of new protected areas in Madagascar, because such sites must contribute to national development goals and the safeguarding of local livelihoods. Consequently, the only option is to carry out regular monitoring to evaluate the impacts of forest use and to alter management regimes accordingly, a process referred to as adaptive management (Plummer and Fennell 2009; Prato 2009). Whether true adaptive management is possible within the context of new, co-managed protected areas in Madagascar remains to be seen, as the requirements of local communities may constrain available management options. For instance, if the extraction of fuel-wood within a sustainable use zone is found to be having negative impacts on biodiversity, it may not be possible to reduce permitted extraction rates without diminishing the support of the protected area by local communities.

Monitoring and evaluation

Ecological monitoring is vital for protected area managers to evaluate the effectiveness of their management and, if necessary, adapt and change; this is particularly important for protected areas in which the harvesting of natural resources by local communities is permitted (Kremen *et al*. 1998). Nevertheless, it is expensive and places a great strain on protected area management budgets, so must be done in a manner that effectively detect population trends at minimal cost (Danielsen *et al*. 2005; Hockley *et al*. 2005). The choice of taxa for monitoring must not be based simply on considerations of feasibility, but must explicitly target those that are sensitive to disturbance and will therefore act as effective indicators. Similarly, logistical ease should not be the primary factor behind the selection of areas in which monitoring is carried out. As forest can be difficult to work in, monitoring is normally carried out along existing path systems, yet paths represent modified habitats in their own right and so should not be used to conduct research into biodiversity within undisturbed habitats (Jenkins *et al*. 1999; Metcalf *et al*. 2005).

Evaluating viability and extinction risk

Most new and existing protected areas in Madagascar use the target-based 5S or Miradi management planning systems, whereby conservation strategies are developed based on an assessment of threats and of the viability of designated conservation targets (i.e. species or habitats). An understanding of how the long-term population viability of species is impacted by habitat degradation and disturbance is thus vital. Frequently, the extent of remaining habitat is used as a proxy of population size, but this measure is not necessarily a reliable surrogate. A forest block of 10,000 ha, for example, may provide 10,000 ha of suitable habitat for a species that is tolerant of edge effects and moderate degradation, but substantially less for a species dependent on intact forest, if this is restricted to a smaller core.

The vulnerability of species to habitat degradation is also important in evaluating global extinction risk, yet this is not amongst the criteria utilised in determining threat status by the IUCN (IUCN 2008). Raxworthy and Nussbaum (2000) highlight the problems with IUCN Red List classifications for Malagasy herpetofauna, which reflect a historical bias towards the conservation of chelonians and boid snakes. To illustrate, Madagascar's three boa species are classified as globally Vulnerable (IUCN 2008), yet they all are tolerant of habitat degradation and even survive in agricultural land and villages; because the loss of forests alone does not result in their extirpation, they appear to be at much lower risk of extinction than species which are dependent on relatively undisturbed forest (Raxworthy and Nussbaum 2000)[1]. Another example of this point is provided by two sympatric bird species restricted to the Mikea forest of southwest Madagascar, *Monias benschi* and *Uratelornis chimaera*, which are both classified as globally Vulnerable (IUCN 2008) based on rates of habitat loss and associated population decline. The global population of *M. benschi* is estimated at over 100,000 individuals (Tobias and Seddon 2002), while that of *U. chimaera* is estimated at around 20,000 (Seddon and Tobias 2007). While *M. benschi* is considerably more abundant within relatively undisturbed habitat, the Mikea forest is increasingly degraded over much of its extent (Seddon *et al*. 2000), creating

[1]Following publication of this chapter, and in recognition of the argument cited here, the three boa species were downgraded to Least Concern in the 2013 IUCN Red List assessment of Malagasy reptiles.

conditions more favourable to *U. chimaera*. Despite having a much smaller population size and similar extent of occurrence, Seddon and Tobias (2007) suggest that *U. chimaera* may be less vulnerable to extinction than *M. benschi*, due to its tolerance of forest degradation.

Prioritisation of conservation areas

Research conducted into the conservation value of strict and multiple-use protected areas in eastern Africa has shown that while species richness did not decrease in less well-protected areas, these sites did harbour distinctly different communities (Gardner *et al*. 2007b). The conclusion from these studies is that both strict protected areas and sites permitting human resource use are required to ensure the conservation of the full complement of the biota of a landscape. This mirrors findings in other continental areas, which also suggest that maintaining a mosaic of pristine and disturbed habitats across a landscape is necessary to ensure maximum biodiversity is conserved (e.g. European birds, Pons *et al*. 2003b; African birds, Borghesio 2008; North American snakes, Todd and Andrews 2008).

However, the Malagasy fauna differs from continental faunas since a much higher proportion of the taxa are specialist species that are adapted to natural forest environments. While multiple-use protected areas on continents support a high diversity of species not found within forests, this is less significant for conservation in Madagascar, where the majority of important species (i.e. endemics) are forest-dependent (Goodman and Benstead 2005; Wilmé 1996). This presents a dilemma for Madagascar, where, with limited funds, the government and conservationists are in the process of establishing a large number of new multiple-use protected areas, while simultaneously maintaining the well-established network of strict protected areas. If human forest use in the new protected areas is not well managed and optimised to meet biodiversity targets, then we are likely to see the gradual, successive loss of disturbance-sensitive species and a homogenisation of the fauna across the sites, which would only serve to increase the importance of the Madagascar National Parks network of strictly-protected areas. It is therefore imperative that the Durban Vision does not draw conservation funds and attention away from strict protected areas, which must continue to be promoted and managed as the 'crown jewels' of the national protected area system. The Madagascar Biodiversity Fund (*Fondation pour les Aires Protégées et la Biodiversité de Madagascar*) has been established as a sustainable financing mechanism for Madagascar's protected areas (Klug *et al*. 2003), requiring site managers to demonstrate successful biodiversity conservation and the contribution to biodiversity representation within SAPM (M. Nicoll, *pers. comm*.). Such mechanisms should help ensure that conservation attention remains focused on the most important protected areas.

4.5 Conclusion

The Durban Vision, which entails the tripling of Madagascar's protected area coverage, will help slow the loss of forests and is undoubtedly a great boost to the conservation of the country's unique biodiversity. Much of the expanded protected area system, however, will be used to satisfy local subsistence needs, and the biodiversity therein will be under pressure from anthropogenic habitat modification. Conservation practitioners, policy-makers, and funders cannot therefore assume that the expansion of the protected area system alone will be sufficient to ensure the conservation of high-value, disturbance-sensitive taxa. Careful consideration must be given to the design and management of new, multiple-use protected areas if they are to succeed in conserving biodiversity while simultaneously contributing to national development objectives.

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References

Andreone, F. 1994. The amphibians of Ranomafana rainforest, Madagascar – preliminary community analysis and conservation considerations. *Oryx* 28: 207–214.

Andrianarimisa, A. 1992. Analyse fonctionnelle de l'écosystème forestier du massif d'Ankarafantsika et de ses stades de dégradation par l'étude du peuplement ornithologique. *Working Group on Birds in the Madagascar Region Newsletter* 2: 4–6.

Andrianarimisa, A. 1993. Variation structurale de la communauté avienne d'une forêt dense sèche semi-caducifoliée dans l'Ouest malgache. In *Proceedings of the Eighth Pan-African Ornithological Congress*, R.T. Wilson (Ed.), pp. 187–191. Koninklijk Museum voor Midden-Afrika, Tervuren.

Andrianasolo, T.H., Andrianjazalahatra, T.L., Rakotondranary, S.J., Ramarokoto, R.E.A.F., Randria, G., Rüdel, N., Schüller, J. and Ganzhorn, J.U. 2006. Habitat utilisation of nocturnal lemurs in evergreen littoral forests of different degrees of degradation. In *Proceedings of the German-Malagasy Research Cooperation in Life and Earth Sciences*, C. Schwitzer, S. Brandt, O. Ramilijaona, M. Rakotomalala Razanahoera, D. Ackermand, T. Razakamanana and J.U. Ganzhorn (Eds.), pp 151–159. University of Hamburg, Hamburg.

Arrigo-Nelson, S.J. and Wright, P.C. 2004. Survey results from Ranomafana National Park: new evidence for the effects of habitat preference and disturbance on the distribution of *Hapalemur*. *Folia Primatologica* 75: 331–334.

Balko, E.A. and Underwood, H.B. 2005. Effects of forest structure and composition in food availability for *Varecia variegata* at Ranomafana National Park, Madagascar. *American Journal of Primatology* 66: 45–70.

Balmford, A. and Whitten, T. 2003. Who should pay for tropical conservation, and how should the costs be met? *Oryx* 37: 2338–250.

Balmford, A., Gaston, K.J., Blyth, S., James, A. and Kapos, V. 2003. Global variation in terrestrial conservation costs, conservation benefits, and unmet conservation needs. *Proceedings of the National Academy of Sciences, USA* 100: 1046–1050.

Barlow, J., Gardner, T.A., Araujo, I.S., Avila-Pires, T.C., Bonaldo, A.B., Costa, J.E., Esposito, M.C., Ferreira, L.V., Hawes, J., Hernandez, M.I.M., *et al*. 2007. Quantifying the biodiversity value of tropical primary, secondary, and plantation forests. *Proceedings of the National Academy of Sciences, USA* 104: 18555–18560.

Bayart, F. and Simmen, B. 2005. Demography, range use, and behavior in black lemurs (*Eulemur macaco macaco*) at Ampasikely, northwest Madagascar. *American Journal of Primatology* 67: 299–312.

Benstead, J.P., Douglas, M.M. and Pringle, C.M. 2003. Relationships of stream invertebrate communities to deforestation in eastern Madagascar. *Ecological Applications* 13: 1473–1490.

Birkinshaw, C. 2001. Fruit characteristics of species dispersed by the Black Lemur (*Eulemur macaco*) in the Lokobe Forest, Madagascar. *Biotropica* 33: 478–486.

Bloxam, Q.M.C., Behler, J.L., Rakotovao, E.R., Randriamahazo, H.J.A.R., Hayes, K.T., Tonge, S.J. and Ganzhorn, J.U. 1996. Effects of logging on the reptile fauna of the Kirindy Forest with special emphasis on the flat-tailed tortoise (*Pyxis planicauda*). In *Ecology and Economy of a Tropical Dry Forest in Madagascar*, J. U. Ganzhorn and J.-P. Sorg (Eds.), pp. 189–201. *Primate Report* 46-1.

Bodin, O., Tengö, M., Norman, A., Lundberg, J. and Elmqvist, T. 2006. The value of small size: loss of forest patches and ecological thresholds in southern Madagascar. *Ecological Applications* 16: 440–451.

Bollen, A., van Elsacker, L. and Ganzhorn, J.U. 2004a. Relations between fruits and disperser assemblages in a Malagasy littoral forest: a community-level approach. *Journal of Tropical Ecology* 20: 599–612.

Bollen, A., van Elsacker, L. and Ganzhorn, J.U. 2004b. Tree dispersal strategies in the littoral forest of Sainte Luce (SE-Madagascar). *Oecologia* 139: 604–616.

Borghesio, L. 2008. Effects of human subsistence activities on forest birds in northern Kenya. *Conservation Biology* 22: 384–394.

Brown, K.A. and Gurevitch, J. 2004. Long-term impacts of logging on forest diversity in Madagascar. *Proceedings of the National Academy of Sciences, USA* 101: 6045–6049.

Bruner, A.G., Gullison, R.E. and Balmford, A. 2004. Financial costs and shortfalls of managing and expanding protected-area systems in developing countries. *BioScience* 54: 1119–1126.

Burney, D.A., Burney, L.P., Godfrey, L.R., Jungers, W.L., Goodman, S.M., Wright, H.T. and Jull, A.J.T. 2004. A chronology for late prehistoric Madagascar. *Journal of Human Evolution* 47: 25–63.

Cadotte, M.W., Franck, R., Reza, L. and Lovett-Doust, J. 2002. Tree and shrub diversity and abundance in fragmented littoral forest of southeastern Madagascar. *Biodiversity and Conservation* 11: 1417–1436.

CBD (Convention on Biological Diversity). 1992. http://www.cbd.int (Accessed 06.05.09).

Chouteau, P. 2004. The impacts of logging on the microhabitats used by two species of couas in the western forest of Madagascar. *Compte Rendues Biologies* 327: 1157–1170.

Chouteau, P., Fenosoa, R. and Rakotoarimanana, V. 2004. Habitat selection and density of couas in Madagascar: implication for their conservation. *Compte Rendues Biologies* 327: 37–50.

Commission SAPM 2006. *Système d'Aires Protégées Malagasy: Orientations Générales sur les Catégories et les Types de Gouvernance*. SAPM Commission, Antananarivo.

Connell, J.H. 1978. Diversity in tropical rain forests and coral reefs. *Science* 199: 1302–1310.

Cornet, A. 1974. *Essai de Cartographie Bioclimatique à Madagascar*. Orstom, Paris.

Danielsen, F., Burgess, N.D. and Balmford, A. 2005. Monitoring matters: examining the potential of locally-based approaches. *Biodiversity and Conservation* 14: 2507–2542.

Dew, J.L. and Wright, P. 1998. Frugivory and seed dispersal by four species of primates in Madagascar's eastern rain forest. *Biotropica* 30: 425–437.

Dewar, R.E. and Richard, A.F. 2007. Evolution in the hypervariable environment of Madagascar. *Proceedings of the National Academy of Sciences, USA* 104: 13723–13727.

Donque, G. 1972. The climatology of Madagascar. In *Biogeography and Ecology of Madagascar*, R. Battistini and G. Richard-Vindard (Eds.), pp. 87–144. G.W Junk, The Hague.

Dudley, N. and Phillips, A. 2006. *Forests and Protected Areas: Guidance on the Use of the IUCN Protected Area Management Categories*. IUCN, Gland.

Eger, J.L. and Mitchell, L. 2003. Chiroptera, bats. In *The Natural History of Madagascar*, S.M. Goodman and J.P. Benstead (Eds.), pp. 1287–1298. University of Chicago Press, Chicago.

Elmqvist, T., Pyykönen, M., Tengö, M., Rakotondrasoa, F., Rabakonandrianina, E. and Radimilahy, C. 2007. Patterns of loss and regeneration of tropical dry forest in Madagascar: the social institutional context. *PLoS ONE* 2: e402.

Enright, N.J. and Miller, B.P. 2007. Livestock grazing impacts on desert vegetation, Khirthar National Park, Pakistan. *Rangeland Ecology & Management* 60: 680–684.

Evans, D.M., Redpath, S.M., Evans, S.A., Elston, D.A., Gardner, C.J., Dennis, P. and Pakeman, R.J. 2006. Low intensity, mixed livestock grazing improves the breeding abundance of a common insectivorous passerine. *Biology Letters* 2: 636–638.

Evers, S. and van der Zwan, N. 1998. *Madagascar: the Zebu as Guide through Past and Present*. Africa Museum, Berg en Dal.

Ganzhorn J.U. 1987. A possible role of plantations for primate conservation in Madagascar. *American Journal of Primatology* 12: 205–215.

Ganzhorn, J.U. 1995. Low-level forest disturbances effects on primary production, leaf chemistry, and lemur populations. *Ecology* 76: 2084–2096.

Ganzhorn, J.U. 2003. Effects of introduced *Rattus rattus* on endemic small mammals in dry deciduous forest fragments of western Madagascar. *Animal Conservation* 6: 147–157.

Ganzhorn, J.U. and Schmid, J. 1998. Different population dynamics of *Microcebus murinus* in primary and secondary deciduous dry forests of Madagascar. *International Journal of Primatology* 19: 785–796.

Ganzhorn, J.U., Ganzhorn, A.W., Abraham, J.-P., Andriamanarivo, L. and Ramananjatovo, A. 1990. The impact of selective logging on forest structure and tenrec populations in western Madagascar. *Oecologia* 84: 126–133.

Ganzhorn, J.U., Sommer, S., Abraham, J.-P., Ade, M., Raharivololona, B.M., Rakotovao, E.R., Rakotondrasoa, C. and Randriamarosoa, R. 1996. Mammals of the Kirindy Forest with special emphasis on *Hypogeomys antimena* and the effects of logging on the small mammal fauna. In *Ecology and Economy of a Tropical Dry Forest in Madagascar*, J.U. Ganzhorn and J.-P. Sorg (Eds.), pp 215–232. *Primate Report* 46-1.

Ganzhorn, J.U., Fietz, J., Rakotovao, E., Schwab, D. and Zinner, D. 1999. Lemurs and the regeneration of dry deciduous forest in Madagascar. *Conservation Biology* 13: 794–804.

Ganzhorn, J.U., Goodman, S.M. and Dehgan, A. 2003. Effects of forest fragmentation on small mammals and lemurs. In *The Natural History of Madagascar*, S.M. Goodman and J.P. Benstead (Eds.), pp. 1228–1234. University of Chicago Press, Chicago.

Gardner, C.J., Ferguson, B., Rebara, F. and Ratsifandrihamanana, A.N. 2008. Integrating traditional values and management regimes into Madagascar's expanded protected area system: the case of Ankodida. In *Protected Landscapes and Cultural and Spiritual Values*, J.-M. Mallarach (Ed.), pp. 92–103. IUCN, GTZ and Obra Social de Caixa Catalunya, Kasparek Verlag, Heidelberg.

Gardner, T.A., Barlow, J. and Peres, C.A. 2007a. Paradox, presumption and pitfalls in conservation biology: the importance of habitat change for amphibians and reptiles. *Biological Conservation* 138: 166–179.

Gardner, T.A., Caro, T., Fitzherbert, E.B., Banda, T. and Lalbhai, P. 2007b. Conservation value of multiple-use areas in East Africa. *Conservation Biology* 21: 1516–1525.

Glaw, F. and Vences, M. 2007. *A Field Guide to the Amphibians and Reptiles of Madagascar*. Third Edition. Vences & Glaw Verlag, Cologne.

Golley, F.B. 1977. Insects as regulators of forest nutrient cycling. *Tropical Ecology* 18: 116–123.

GoM (Government of Madagascar). 2001. *Loi N° 2001/05 portant Code de Gestion des Aires Protégées*. Repoblikan'i Madagasikara, Antananarivo.

GoM (Government of Madagascar). 2006. *Guide Pour la Réalisation d'une Etude d'Impact Environnemental et Social Pour les Projets de Création des Nouvelles Aires Protégées*. Ministère de l'Environnement, Office National pour l'Environnement and Commission SAPM, Antananarivo.

Goodman, S.M. 1995. *Rattus* on Madagascar and the dilemma of protecting endemic rodent fauna. *Conservation Biology* 9: 450–453.

Goodman, S.M. and Sterling, E.J. 1996. The utilization of *Canarium* (Burseraceae) seeds by vertebrates in the Réserve Naturelle Intégrale d'Andringitra, Madagascar. In *A Floral and Faunal Inventory of the Réserve Naturelle Intégrale d'Andringitra, Madagascar: With Reference to Altitudinal Variation*, S.M. Goodman (Ed.), pp. 83–89. *Fieldiana: Zoology,* new series 85.

Goodman, S.M. and Raherilalao, M.J. 2003. Effects of forest fragmentation on bird communities. In *The Natural History of Madagascar*, S.M. Goodman and J.P. Benstead (Eds.), pp 1064–1066. University of Chicago Press, Chicago.

Goodman, S.M. and Benstead, J.P. 2005. Updated estimates of biotic diversity and endemism for Madagascar. *Oryx* 39: 73–77.

Goodman, S.M., Rakotondravony, D., Schatz, G. and Wilmé, L. 1996. Species richness of forest-dwelling birds, rodents and insectivores in a planted forest of native trees: a test case from the Ankaratra, Madagascar. *Ecotropica* 2: 109–120.

Goodman, S.M., Ganzhorn, J.U. and Rakotondravony, D. 2003. Introduction to the mammals. In *The Natural History of Madagascar*, S.M. Goodman and J.P. Benstead (Eds.), pp. 1159–1186. University of Chicago Press, Chicago.

Goodman, S.M., Andriafidison, D., Andrianaivoarivelo, R., Cardiff, S.G., Ifticene, I., Jenkins, R.K.B., Kofoky, A., Mbohoahy, T. Rakotondravony, D., Ranivo, J., *et al*. 2005. The distribution of bats in the dry regions of Madagascar. *Animal Conservation* 8: 153–165.

Gray, M.A., Baldauf, S.L., Mayhew, P.J. and Hill, J.K. 2007. The response of avian feeding guilds to tropical forest disturbance. *Conservation Biology* 21: 133–141.

Hawkins, A.F.A. 1993. Relationships among vegetation structure and White-breasted Mesite *Mesitornis variegata* presence and foraging site choice. In *Proceedings of the Eighth Pan-African Ornithological Congress*, R.T. Wilson (Ed.), pp. 155–163. Koninklijk Museum voor Midden-Afrika, Tervuren.

Hawkins, A.F.A. 1994. Conservation status and regional population estimates of the White-breasted Mesite *Mesitornis variegata*, a rare Malagasy endemic. *Bird Conservation International* 4: 279–303.

Hawkins, A.F.A. and Wilmé, L. 1996. Effects of logging on forest birds. In *Ecology and Economy of a Tropical Dry Forest in Madagascar*, J.U. Ganzhorn and J.-P. Sorg (Eds.), pp. 203–213. *Primate Report* 46-1.

Hockley, N.J., Jones, J.P.G., Andriahajaina, F.B., Manica, A., Ranambitsoa, E.H. and Randriamboahary, J.A. 2005. When should communities and conservationists monitor exploited resources? *Biodiversity and Conservation* 14: 2795–2806.

Horning, N. 2003. How rules affect conservation outcomes. In *The Natural History of Madagascar*, S.M. Goodman and J.P. Benstead (Eds.), pp. 146–153. University of Chicago Press, Chicago.

Hutcheon, J.M. 2003. Frugivory by Malagasy bats. In *The Natural History of Madagascar*, S.M. Goodman and J.P. Benstead (Eds.), pp. 1205–1207. University of Chicago Press, Chicago.

IUCN (International Union for Conservation of Nature) 1994. *Guidelines for Protected Area Management Categories*. IUCN, Cambridge and Gland.

IUCN (International Union for Conservation of Nature) 2008. *IUCN Red List of Threatened Species*. IUCN, Gland.

James, A.N., Gaston, K.J. and Balmford, A. 1999. Balancing the Earth's accounts. *Nature* 401: 323–324.

Jenkins, R.K.B., Brady, L.D., Huston, K., Kauffmann, J.L.D., Rabearivony, J., Raveloson, G. and Rowcliffe, J.M. 1999. The population status of chameleons within Ranomafana National Park, and recommendations for future monitoring. *Oryx* 33: 38–47.

Jenkins, R.K.B., Brady, L.D., Bisoa, M., Rabearivony, J. and Griffiths, R.A. 2003. Forest disturbance and river proximity influence chameleon abundance in Madagascar. *Biological Conservation* 109: 407–415.

Kaufmann, J.C. and Tsirahamba, S. 2006. Forests and thorns: conditions of change affecting Mahafale pastoralists in southwestern Madagascar. *Conservation and Society* 4: 231–261.

Klug, U., Moye, M. and Carr-Dirick, B. 2003. *Establishing Environmental Funds for Protected Areas in Francophone Africa: Challenges and Opportunities. The Experience of the Sangha Tri-National Initiative and proposed Madagascar Foundation for Protected Areas and Biodiversity*. Paper presented at the Vth World Parks Congress: Sustainable Finance Stream, Durban, South Africa, September 2003.

Knight, A.T., Cowling, R.M., Rouget, M., Balmford, A., Lombard, A.T. and Campbell, B.M. 2008. Knowing but not doing: selecting priority conservation areas and the research -implementation gap. *Conservation Biology* 22: 610–617.

Kofoky, A., Andriafidison, D., Ratrimomanarivo, F., Razafimanahaka, H.J., Rakotondravony, D. Racey, P.A. and Jenkins, R.K.B. 2007. Habitat use, roost selection and conservation of bats in Tsingy de Bemaraha National Park, Madagascar. *Biodiversity and Conservation* 16: 1039–1053.

Kremen, C., Raymond, I. and Lance, K. 1998. An interdisciplinary tool for monitoring conservation impacts in Madagascar. *Conservation Biology* 12: 549–563.

Kull, C.A., Tassin, J., Rambeloarisoa, G. and Sarrailh, J.-M. 2008. Invasive Australian acacias on western Indian Ocean islands: a historical and ecological perspective. *African Journal of Ecology* 46: 684–689.

Kuussaari, M., Bommarco, R., Heikkinen, R.K., Helm, A., Krauss, J., Lindborg, R., Öckinger, E., Pärtel, M., Pino, J., Rodà, F., *et al*. 2009. Extinction debt: a challenge for biodiversity conservation. *Trends in Ecology & Evolution* 24: 564–571.

Lahann, P. 2007. Feeding ecology and seed dispersal of sympatric cheirogaleid lemurs (*Microcebus murinus*, *Cheirogaleus medius*, *Cheirogaleus major*) in the littoral rainforest of south-east Madagascar. *Journal of Zoology* 271: 88–98.

Langrand, O. 1994. The effects of forest fragmentation on forest dependent birds in Madagascar. *Working Group on Birds in the Madagascar Region Newsletter* 3: 8–10.

Langrand, O. and Wilmé, L. 1997. Effects of forest fragmentation on extinction patterns of the endemic avifauna on the central high plateau of Madagascar. In *Natural Change and Human Impact in Madagascar*, S.M. Goodman and B.D. Patterson (Eds.), pp. 280–305. Smithsonian Institution Press, Washington DC.

Laurance, W.F., Lovejoy, T.E., Vasconcelos, H.L., Bruna, E.M., Didham, R.K., Stouffer, P.C., Gascon, C., Bieregaard, R.O., Laurance, S.G. and Sampaio, E. 2002. Ecosystem decay of Amazonian forest fragments: a 22-year investigation. *Conservation Biology* 16: 605–618.

Lehman, S.M. 2007. Spatial variations in *Eulemur fulvus rufus* and *Lepilemur mustelinus* densities in Madagascar. *Folia Primatologica* 78: 46–55.

Lehman, S.M., Rajaonson, A. and Day, S. 2006a. Edge effects and their influence on lemur density and distribution in southeast Madagascar. *American Journal of Physical Anthropology* 129: 232–241.

Lehman, S.M., Rajaonson, A. and Day, S. 2006b. Lemur responses to edge effects in the Vohibola III Classified Forest, Madagascar. *American Journal of Primatology* 68: 293–299.

Lehman, S.M., Ratsimbazafy, J., Rajaonson, A. and Day, S. 2006c. Ecological correlates to lemur community structure in southeast Madagascar. *International Journal of Primatology* 27: 1023–1040.

Lehtinen, R.M., Ramanamanjato, J.-B. and Raveloarison, J.G. 2003. Edge effects and extinction proneness in a herpetofauna from Madagascar. *Biodiversity and Conservation* 12: 1357–1370.

Lehtonen, J.T., Mustonen, O., Ramiarinjanahary, H., Niemelä, J. and Rita, H. 2001. Habitat use by endemic and introduced rodents along a gradient of forest disturbance in Madagascar. *Biodiversity and Conservation* 10: 1185–1202.

Leuteritz, T.E.J., Lamb, T. and Limberaza, J.C. 2005. Distribution, status, and conservation of radiated tortoises (*Geochelone radiata*) in Madagascar. *Biological Conservation* 124: 451–461.

Martin, E.A., Ratsimisetra, L., Laloë, F. and Carrière, S. M. 2009. Conservation value for birds of traditionally managed isolated trees in an agricultural landscape of Madagascar. *Biodiversity and Conservation* 18: 2719–2742.

Mata-González, R., Figueroa-Sandoval, B., Clemente, F. and Manzano, M. 2007. Vegetation changes after livestock grazing exclusion and shrub control in the southern Chihuahuan desert. *Western North American Naturalist* 67: 63–70.

McKinny, M.L. and Lockwood, J.L. 1999. Biotic homogenization: a few winners replacing many losers in the next mass extinction. *Trends in Ecology and Evolution* 14: 450–453.

MEFT (Ministère de l'Environnement, des Forêts et du Tourisme), USAID (United States Agency for International Development) and Conservation International. 2009. *Evolution de la Couverture de Forêts Naturelles à Madagascar, 1990–2000–2005*. MEFT, Antananarivo.

Merenlender, A., Kremen, C., Rakotondratsima, M. and Weiss, A. 1998. Monitoring impacts of natural resource extraction on lemurs of the Masoala Peninsula, Madagascar. *Conservation Ecology* 2: http://consecol.org/vol2/iss2/art5/

Metcalf, J., Bayly, N., Bisoa, M., Rabearivony, J. and Stewart-Jones, A. 2005. Edge effects from paths on two chameleon species in Madagascar. *African Journal of Herpetology* 54: 99–102.

Middleton, K. 1999. Who killed 'Malagasy Cactus'? Science, environment and colonialism in southern Madagascar (1924–1930). *Journal of Southern African Studies* 25: 215–248.

Mittermeier R.A., Hawkins, A.F.A., Rajaobelina, S. and Langrand, O. 2005. Wilderness conservation in a biodiversity hotspot. *International Journal of Wilderness* 11: 42–45.

MNP (Madagascar National Parks). 2008. *Plan d'Aménagement et de Gestion: Park National Tsimanampesotse*. Madagascar National Parks, Antananarivo.

Moat, J. and Smith, P. 2007. *Atlas of the Vegetation of Madagascar*. Kew Publishing, Kew.

Nicoll, M.E. and Rathbun, G.B. 1990. *African Insectivora and Elephant-shrews: an Action Plan for their Conservation*. IUCN/SSC Insectivore, Tree-shrew and Elephant-shrew Specialist Group, IUCN, Gland.

Norris, S. 2006. Madagascar defiant. *BioScience* 56: 960–965.

O'Brien, S., Emahalala, E.R., Beard, V., Rakotondrainy, R.M., Reid, A., Raharisoa, V. and Coulson, T. 2003. Decline of the radiated tortoise *Geochelone radiata* due to overexploitation. *Oryx* 37: 338–343.

Olson, D.M. and Andriamiadana, A. 1996. The effects of selective logging on the leaf litter invertebrate community of a tropical dry forest in western Madagascar. In *Ecology and Economy of a Tropical Dry Forest in Madagascar*, J.U. Ganzhorn and J.-P. Sorg (Eds.), pp 175–187. *Primate Report* 46-1.

Pedrono, M. and Smith, L.L. 2003. Testudinae, land tortoises. In *The Natural History of Madagascar*, S.M. Goodman and J.P. Benstead (Eds.), pp. 951–956. University of Chicago Press, Chicago.

Phillipson, P.B. 1996. Endemism and non-endemism in the flora of south-west Madagascar. In *Biogéographie de Madagascar*, W.R. Lourenço (Ed.), pp 125–136. Editions ORSTOM, Paris.

Plummer, R. and Fennell, D.A. 2009. Managing protected areas for sustainable tourism: prospects for adaptive co-management. *Journal of Sustainable Tourism* 17: 149–168.

Pollini, J. 2007. *Slash-and-burn Cultivation and Deforestation in the Malagasy Rain Forests: Representations and Realities*. PhD Thesis, Cornell University, Ithaca.

Pons, P. and Wendenburg, C. 2005. The impact of fire and forest conversion into savanna on the bird communities of West Madagascan dry forests. *Animal Conservation* 8: 183–193.

Pons, P., Rakotobearison, G. and Wendenburg, C. 2003a. Immediate effects of a fire on birds and vegetation at Ankarafantsika Strict Nature Reserve, NW Madagascar. *Ostrich* 74: 146–148.

Pons, P., Lambert, B., Rigolot, E. and Prodon, R. 2003b. The effects of grassland management using fire on habitat occupancy and conservation of birds in a mosaic landscape. *Biodiversity and Conservation* 12: 1843–1860.

Prato, T. 2009. Fuzzy adaptive management of social and ecological carrying capacities for protected areas. *Journal of Environmental Management* 90: 2551–2557.

Pulliam, H.R. 1988. Sources, sinks and population regulation. *American Naturalist* 132: 652–661.

Raherilalao, M.J. 2001. Effets de la fragmentation de la forêt sur les oiseaux autour du Parc National de Ranomafana (Madagascar). *Revue d'Ecologie (Terre et Vie)* 56: 389–406.

Raik, D. 2007. Forest management in Madagascar: an historical overview. *Madagascar Conservation & Development* 2: 5–10.

Ralison, J.M., Ramanamanjato, J.-B. and Rakotomalala Razanahoera, M. 2006. Influence de la qualité de la forêt sur *Eulemur collaris* (E. Geoffroy, 1812) dans les formations littorales de Mandena et de Sainte Luce à Tolagnaro, Madagascar, en 1998–1999. In *Proceedings of the German-Malagasy Research Cooperation in Life and Earth Sciences*, C. Schwitzer, S. Brandt, O. Ramilijaona, M. Rakotomalala Razanahoera, D. Ackermand, T. Razakamanana and J.U. Ganzhorn (Eds.), pp 161–167. University of Hamburg, Hamburg.

Ramanamanjato, J.-B. and Ganzhorn, J.U. 2001. Effects of forest fragmentation, introduced *Rattus rattus* and the role or exotic tree plantations and secondary vegetation for the conservation of an endemic rodent and a small lemur in littoral forests of southeastern Madagascar. *Animal Conservation* 4: 175–183.

Randrianandianina, B.N., Andriamahaly, L.R., Harisoa, F.M. and Nicoll, M.E. 2003. The role of protected areas in the management of the island's biodiversity. In *The Natural History of Madagascar*, S.M. Goodman and J.P. Benstead (Eds.), pp. 1423–1432. University of Chicago Press, Chicago.

Randrianandrianina, F.H., Andriafidison, D., Kofoky, A.F., Ramilijaona, O., Ratrimomanarivo, F., Racey, P.A. and Jenkins, R.K.B. 2006. Habitat use and conservation of bats in rainforest and adjacent human-modified habitats in eastern Madagascar. *Acta Chiropterologica* 8: 429–437.

Rasolofoson, D., Rakotondratsimba, G., Rakotonirainy, O., Rakotozafy, L.M.A., Ratsimbazafy, J.H., Rabetafika, L. and Randrianarison, R.M. 2007. Influences des pressions anthropiques sur les lémuriens d'Anantaka, dans la partie Est du Plateau de Makira, Maroantsetra, Madagascar. *Madagascar Conservation & Development* 2: 21–27.

Rasolonandrasana, B.P.N. and Goodman, S.M. 2006. The influence of fire on mountain sclerophyllous forests and their small-mammal communities in Madagascar. In *Land Use Change and Mountain Biodiversity*, E.M. Spehn, M. Liberman and C. Körner (Eds.), pp 77–87. Taylor & Francis, Boca Ranton.

Ratrimomanarivo, F.H. 2007. Etude du régime alimentaire d'*Eidolon dupreanum* (*Chiroptera: Pteropodidae*) dans la région anthropisée des hautes terres du centre de Madagascar. *Revue d'Ecologie (Terre et Vie)* 62: 229–244.

Ravalomanana, M. 2003. Durban Vision declaration. Speech presented at the Vth World parks Congress, Durban, South Africa, 07–19 September 2003.

Raxworthy, C.J. and Nussbaum, R.A., 2000. Extinction and extinction vulnerability of amphibians and reptiles in Madagascar. *Amphibian and Reptile Conservation* 2: 15–23.

Raxworthy, C.J., Pearson, R.G., Rabibisoa, N., Rakotondrazafy, A.M., Ramanamanjato, J.-B., Raselimanana, A.P., Wu, S., Nussbaum, R.A. and Stone, D.A. 2008. Extinction vulnerability of tropical montane endemism from warming and upslope displacement: a preliminary appraisal for the highest massif in Madagascar. *Global Change Biology* 14: 1 –18.

Réau, B. 2002. Burning for zebu: the complexity of deforestation issues in western Madagascar. *Norwegian Journal of Geography* 56: 219–229.

Rendigs, A., Radespiel, U., Wrogemann, D. and Zimmermann, E. 2003. Relationship between microhabitat structure and distribution of mouse lemurs (*Microcebus* spp.) in northwest Madagascar. *International Journal of Primatology* 24: 47–64.

Rioux Paquette, S. 2008. *La Génétique au Service de la Conservation d'une Espèce Menacée Endémique à Madagascar, la Tortue Radiée* Astrochelys radiata. PhD thesis, Université de Montréal, Montréal.

Scott, D.M., Brown, D., Mahood, S., Denton, B., Silburn, A. and Rakotondraparary, F. 2006. The impacts of forest clearance on lizard, small mammal and bird communities in the arid spiny forest, southern Madagascar. *Biological Conservation* 127: 72–87.

Seddon, N. and Tobias, J.A. 2007. Population size and habitat associations of the long-tailed ground-roller *Uratelornis chimaera*. *Bird Conservation International* 17: 1–12.

Seddon, N., Tobias, J., Yount, J.W., Ramanampamonjy, J.R., Butchart, S. and Randrianizahana, H. 2000. Conservation issues and priorities in the Mikea Forest of south -west Madagascar. *Oryx* 34: 287–304.

Seddon, N., Tobias, J. and Butchart, S.H.M. 2003. Group living, breeding behaviour and territoriality in the Subdesert Mesite *Monias benschi*. *Ibis* 145: 277–294.

Soarimalala, V. and Raherilalao, M.J. 2008. Pressions et menaces dans la région forestière sèche malgache. In *Les Forêts Sèches de Madagascar*, S.M. Goodman and L. Wilmé (Eds.), pp. 157–161. *Malagasy Nature* 1.

Spehn, S.E. and Ganzhorn, J.U. 2000. Influence of seed dispersal by brown lemurs on removal rates of three *Grewia* species (Tiliaceae) in the dry deciduous forest of Madagascar. *Ecotropica* 6: 13–21.

Stattersfield, A., Crosby, M., Long, A. and Wege, D. 1998. *Global Directory of Endemic Bird Areas*. BirdLife International, Cambridge.

Stephenson, P.J. 1993. The impacts of tourism on nature reserves in Madagascar: Périnet, a case-study. *Environmental Conservation* 20: 262–265.

Stephenson, P.J. 1995. Small mammal microhabitat use in lowland rain forest of north-east Madagascar. *Acta Theriologica* 40: 425–438.

Stephenson, P.J. 2003. Lipotyphla (Ex Insectivora): *Geogale aurita*, Large-eared tenrec. In *The Natural History of Madagascar*, S.M. Goodman and J.P. Benstead (Eds.), pp. 1265–1267. University of Chicago Press, Chicago.

Stewart-Oaten, A., Murdoch, W.W. and Parker, K.R.. 1986. Environmental Impact Assessment: "pseudoreplication" in time? *Ecology* 67: 929–940.

Sussman, R.W. and Rakotozafy, A. 1994. Plant diversity and structural analysis of a tropical dry forest in southwestern Madagascar. *Biotropica* 26: 241–254.

SAPM (Système d'Aires Protégées de Madagascar) 2007. *Cadre Fonctionnel de Procédures: Création et Gestion d'Aires Protégées; Mesures de Sauvegardes; Mécanisme de Financement*. Système d'Aires Protégées de Madagascar, Antananarivo.

Tobias, J.A. and Seddon, N. 2002. Estimating population size in the subdesert mesite (*Monias benschi*): new methods and implications for conservation. *Biological Conservation* 108: 199–212.

Todd, B.D. and Andrews, K.M. 2008. Response of a reptile guild to forest harvesting. *Conservation Biology* 22: 753–761.

Vallan, D. 2000. Influence of forest fragmentation on amphibian diversity in the nature reserve of Ambohitantely, highland Madagascar. *Biological Conservation* 96: 31–43.

Vallan, D. 2002. Effects of anthropogenic environmental changes on amphibian diversity in the rain forests of eastern Madagascar. *Journal of Tropical Ecology* 18: 725–742.

Vallan, D., Andreone, F., Raherisoa, V.H. and Dolch, R. 2004. Does selective wood exploitation affect amphibian diversity? The case of An'Ala, a tropical rainforest in eastern Madagascar. *Oryx* 38: 410–417.

Walker, R.C.J., Rix, C. and Woods-Ballard, A. 2004. The legal and illegal trade in the Madagascar spider tortoise *Pyxis arachnoides* for the exotic pet trade. *The Herpetological Bulletin* 90: 2–9.

Watson, J.E.M., Whittaker, R.J. and Dawson, T.P. 2004a. Habitat structure and proximity to forest edge affect the abundance and distribution of forest-dependent birds in tropical coastal forests of southeastern Madagascar. *Biological Conservation* 120: 311–327.

Watson, J.E.M., Whittaker, R.J. and Dawson, T. . 2004b. Avifaunal responses to habitat fragmentation in the threatened littoral forests of south-eastern Madagascar. *Journal of Biogeography* 31: 1791–1807.

White, F.J., Overdorff, D.J., Balko, E.A. and Wright, P.C. 1995. Distribution of ruffed lemurs (*Varecia variegata*) in Ranomafana National Park, Madagascar. *Folia Primatologica* 64: 124–131.

Whittaker, R.J. 1998. *Island Biogeography: Ecology, Evolution, and Conservation*. Oxford University Press, Oxford.

Whittaker, R.J., Araújo, M.B., Jepson, P., Ladle, R.J., Watson, J.E.M. and Willis, K.J. 2005. Conservation biogeography: assessment and prospect. *Diversity and Distributions* 11: 3–23.

Wilmé, L. 1996. Composition and characteristics of bird communities in Madagascar. In *Biogéographie de Madagascar*, W. R. Lourenço (Ed.), pp. 349–362. Editions ORSTOM, Paris.

Woog, F., Ramanitra, N. and Andrianarivelosoa, S.R. 2006. Effects of deforestation on eastern Malagasy bird communities. In *Proceedings of the German-Malagasy Research Cooperation in Life and Earth Sciences*, C. Schwitzer, S. Brandt, O. Ramilijaona, M. Rakotomalala Razanahoera, D. Ackermand, T. Razakamanana and J.U. Ganzhorn (Eds.), pp. 203–214. University of Hamburg, Hamburg.

WWF 2008. *Plan de Gestion d'Ankodida*. WWF Madagascar and Western Indian Ocean Programme Office, Antananarivo.

Chapter 5

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Comparing methods for prioritising between existing protected areas: a case study using Madagascar's dry forest reptiles

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Abstract

There are insufficient resources available to manage the world's existing protected area portfolio effectively, so the most important sites should be prioritised in investment decision-making. Sophisticated conservation planning and assessment tools have been developed and used to identify locations for new protected areas. However, decision-makers in many countries lack the institutional support and necessary capacity to use the associated software, as well as being hindered by data availability. As such, simple heuristic approaches such as species richness or number of threatened species are generally adopted to inform prioritisation decisions. Using the reptile fauna of

Madagascar's dry forests as a case study, we evaluate the performance of four site prioritisation protocols used to rank the conservation value of 22 protected and unprotected sites. We compare the results to a benchmark produced by the widely-used, gold-standard systematic conservation planning software Zonation. The four indices scored sites on the basis of: i) species richness; ii) an index based on species' Red List status; iii) irreplaceability (the foundation of systematic conservation planning); and, iv) a novel conservation value index (*CVI*), which incorporates species-level information on endemism, representation in the protected area system, tolerance of habitat degradation and hunting/collection pressure. Rankings produced by the four protocols were positively correlated to the results of Zonation, but *CVI* and irreplaceability performed better than species richness and the Red List index. Given the constraints of data availability and capacity experienced by decision-makers in the developing world, our findings suggest that simple metrics using readily-available data can represent a useful alternative to more sophisticated analyses, particularly when they integrate species-specific information related to extinction risk.

5.1 Introduction

Conservation is severely under-resourced globally (James *et al*. 1999; Waldron *et al*. 2014), but particularly in tropical developing countries where biodiversity is concentrated (Balmford *et al*. 2003; Bruner *et al*. 2004), so interventions must be prioritised to ensure maximum impact. The principal strategy for conserving biodiversity is the establishment of protected areas, which now cover almost 13 % of the world's land surface (Bertzky *et al*. 2012; Jenkins and Joppa 2009). However, protected areas vary greatly in the value of their constituent biodiversity (Le Saout *et al*. 2013; Rodrigues *et al*. 2004), having largely been established in landscapes where opportunity costs have been lowest (Barr *et al*. 2013; Joppa and Pfaff 2011; Pressey 1994), rather than the most important areas for conservation (Beresford *et al*. 2011; Brooks *et al*. 2004a).

While protected areas can be effective in reducing the pressures that threaten biodiversity (Brooks *et al*. 2009; Joppa and Pfaff 2011; Nelson and Chomitz 2011), this is dependent on investment in active management (Bruner *et al*. 2001). Some sites, 'paper parks', lack any management at all (Brandon *et al*. 1998). Ongoing deforestation and forest degradation in protected areas worldwide (e.g. Allnutt *et al*. 2013; Gaveau *et al*. 2009; Tang *et al*. 2010) suggests that there are more protected areas than can be resourced and managed effectively in some countries. Therefore, protected area portfolios should be subjected to triage (Bottrill *et al*. 2008), because conservation goals will be best achieved if available investment is targeted preferentially towards the most important sites (Fuller *et al*. 2010).

Decisions regarding the selection of priority areas from within existing protected area networks should be evidence-based. However, such choices are almost always made by state-mandated protected area management agencies, non-governmental organisations and conservation funding bodies (henceforth decision-makers), actors who do not tend to make systematic use of scientific tools and approaches in decision-making (Cook *et al*. 2010; Pullin and Knight 2005; Pullin *et al*. 2004). Indeed, their priorities may be influenced by the (often implicit) values held by individuals and organisations (Game *et al*. 2013; Marris 2007; Pullin *et al*. 2013; Wilson 2008), rather than rational, explicit methods, which may lead to suboptimal results.

A range of metrics and approaches can be used to provide an evidence-base for protected area prioritisation and triage, including the sophisticated and gold-standard software tools developed for systematic conservation planning and assessments (i.e. to inform the design of protected area portfolios where the representation and persistence of biodiversity are maximised for least cost) (Margules and Pressey 2000). However, the uptake and use of such tools by decision-makers is limited by the need for specific training, data availability (e.g. cost layer information) and institutional support (Bottrill and Pressey 2012; Gaston *et al*. 2008). As a result, prioritisation decisions continue to be carried out based on simple measures such as richness of threatened species, or without any evidence base at all (e.g. Schwitzer *et al*. 2013, 2014).

Given that outputs from conservation planning tools are rarely applied in the context of prioritisation across existing protected area portfolios, it is important to assess the performance of the different metrics that can be, or are, used in such circumstances. In this paper, we evaluate four simple indices, including a novel Conservation Value Index (*CVI*), that could be used to identify priority sites for reptile conservation from a network of 22 designated and proposed protected areas in the dry regions of Madagascar. We benchmark our findings against the site ranking produced using the widely-used systematic planning software, Zonation, and consider the relative strengths and weaknesses of each approach, given the constraints of data availability that may hinder conservation decision-making.

5.2 Materials and methods

5.2.1 Study region and taxa

Madagascar is one of the world's top conservation priorities (Brooks *et al*. 2006; Holt *et al*. 2013). Since 2003, it has been implementing its 'Durban Vision', an ambitious programme to extend the coverage of its protected area system from 1.7 million ha to 6 million ha (Corson 2014; Raik 2007). Prior to this, all Malagasy protected areas were managed by the para-statal Madagascar National Parks (MNP) and were designated as IUCN category I, II or IV, with the primary objective of conserving biodiversity (Randrianandianina *et al*. 2003). The new generation of sites, however, is composed mainly of category III, V and VI multiple-use protected areas, managed for multiple socio -economic objectives as well as the maintenance of biodiversity, and administered by a range of actors including local community associations, non-governmental organisations (NGOs) and decentralised state authorities (Gardner 2011; Gardner *et al*. 2013; Virah-Sawmy *et al*. 2014). Together with the existing MNP protected areas, they form the Madagascar Protected Area System (SAPM).

The location of new protected areas has been partially informed by gap analyses and systematic conservation planning (Kremen *et al*. 2008; Rasoavahiny *et al*. 2008), based purely on biodiversity data without the inclusion of cost information (Corson 2014), with the aim of maximising the representation of endemic biodiversity within SAPM. Over 500 priority sites were identified, of which 93 have been granted definitive or temporary protected status (AGRECO 2012). The organisation(s) responsible for each protected area, which are primarily Malagasy and international NGOs, are expected to independently source the necessary funds to ensure its long-term management. However, the Madagascar Foundation for Protected Areas and Biodiversity has been created to meet emergency shortfalls for the 'best' protected areas, which are characterised as such based on factors including a site's contribution to biodiversity representation within SAPM (M. Nicoll, *pers. comm.*). Thus, protected areas are essentially competing for funds from the same, limited, pool of financial support. In their management plans, individual protected areas promote their importance on the basis of species richness, as well as lists of threatened and locally endemic species.

The freely available and comprehensive reptile inventory data from a range of sites in the contiguous dry regions of Madagascar make this taxonomic group an ideal case study for assessing the performance of prioritisation tools. The country's reptile fauna is diverse, comprising almost 400 species, with 92% of these being endemic (Goodman and Benstead 2005; Jenkins *et al*. 2014; Nagy *et al*. 2012). Many of the species are highly range restricted or micro-endemic (Jenkins *et al*. 2014; Vences *et al*. 2009), and the majority are forest-dependent (Glaw and Vences 2007; Jenkins *et al*. 2014; Raxworthy 2003): given historical deforestation trends (Harper *et al*. 2007), such species may depend on the effective maintenance of protected areas for their long-term survival. The dry regions of Madagascar are composed of two 'Global 200' priority ecoregions: the Madagascar dry forests in the west, and the Madagascar spiny desert of the south and southwest (Olson and Dinerstein 1998). Although northern Madagascar also harbours dry forests, they are isolated from the western areas by a large band of humid forest and are excluded from this analysis.

5.2.2 Biodiversity survey data

We compiled a database of all known reptile inventories in western and southern Madagascar including both protected and currently unprotected sites (Fig. 5.1; Appendix 1, Table A1.1), and supplemented these data with our own survey records (C. Raxworthy, unpublished data). All inventories used standard protocols including pit-fall traps and refuge searches (see references in the Supplementary Material, Appendix 1). The database included inventories of 22 sites, comprising eight national parks, seven new protected areas and seven hitherto unprotected sites which have been identified as potential sites for protected area designation (Fig. 5.1). Species taxonomy follows Glaw and Vences (2007) and subsequent revisions wherever the specific identity of split taxa is unambiguous (Cadle and Ineich 2008; Köhler *et al*. 2009; Nagy *et al*. 2010; Raxworthy *et al*. 2007). The database was cleaned by removing all records of species no longer considered valid (i.e. subsequently synonymised; n = 2), records that have not been described/identified to species level (n = 10), probable misidentifications (n = 5) and introduced species (n = 1).

5.2.3 Simple site prioritisation indices

Sites were ranked on the basis of four simple prioritisation protocols: i) species richness (a measure often used by decision-makers); ii) an index derived from species Red List status (IUCN 2012); iii) an irreplaceability index (the foundation of systematic conservation planning); and, iv) a conservation value index (*CVI*) in which species were scored on the basis of four attributes reflecting rarity and threat. For protocols 2, 3 and 4, scores were assigned to individual species and the site score (SS) was then calculated as the cumulative score of all species occurring there.

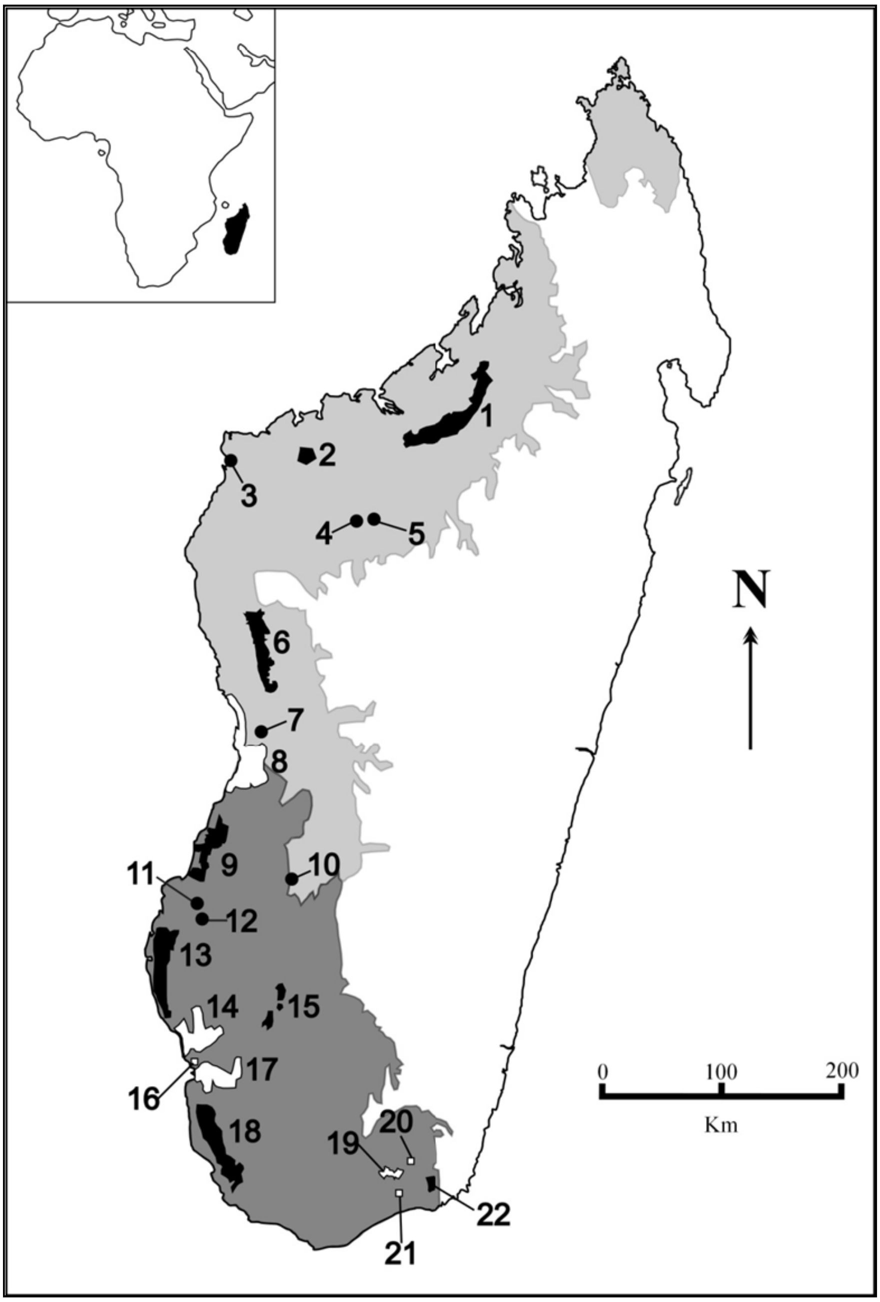

Figure 5.1 Map of Madagascar showing location of protected areas and unprotected sites used in prioritisation. National parks are indicated by black polygons, new protected areas by white polygons/squares, and unprotected sites by black circles: 1, Ankarafantsika; 2, Namoroka; 3, Andranomanintsy; 4, Kelifely; 5, Ankara; 6, Tsingy de Bemaraha; 7, Masoarivo; 8, Menabe Antimena; 9, Kirindy Mite; 10, Makay; 11, Berento; 12, Nosy-Ambositra; 13, Mikea; 14, Ranobe PK32; 15, Zombitse-Vohibasia; 16, Tsinjoriake; 17, Amoron'i Onilahy; 18, Tsimanampesotse; 19, Nord Ifotaka; 20, Anadabolava-Betsimalaho; 21, Behara-Tranomaro, and; 22, Andohahela Parcel 2. The dry bioclimatic region is shown in light grey, and the sub-arid region in dark grey (following Cornet (1974)). The inset shows the position of Madagascar relative to mainland Africa.

Protocol 1 – Richness (SR)

$SS_{SR} = r$ (where r is the species richness of a site)

Protocol 2 – Red List Index (RL)

Scores were assigned to species on the basis of their Red List rankings from the 2011 Global Reptile Assessment for Madagascar (IUCN 2012), as follows: 5, Critically Endangered (CR); 4, Endangered (EN); 3, Vulnerable (VU); 2, Near Threatened (NT); 1, Least Concern (LC); 0, Data Deficient (DD) or Not Evaluated (NE).

$SS_{RL} = \Sigma RL_{species}$

Protocol 3 – Irreplaceability (IR)

Using a method based on Brugière and Scholte (2013), where each species is weighted by the inverse of the number of protected areas in which it was recorded.

$IR_{species} = 1/n$ (where n is the number of protected areas within the sample at which a species occurs)

$SS_{IR} = \Sigma\ IR_{species}$

Protocol 4 – Conservation Value Index (CVI)

Scores were assigned to each species based on four attributes that reflect relative rarity and threat, and thus extinction risk. Rarity was assessed using geographical scale of endemism (E) and representation in protected areas within the study sample (R); for both these attributes, 'rarer' species score higher than widespread and well-represented species. Threat was based on hunting and collection pressure (C) and degradation tolerance (T), because these factors have a significant influence on the long-term viability of Madagascar's reptiles (Jenkins *et al*. 2014; Raxworthy 2003). The relative tolerance of species to habitat modification is particularly critical (Daily 2001; Fischer *et al*. 2006; Gibbons *et al*. 2000; Jenkins *et al*. 2014) as most species in Madagascar are forest dependent (Raxworthy 2003), and forest loss and degradation outside of protected areas show no sign of reducing (Harper *et al*. 2007; ONE *et al*. 2013). Degradation-tolerant species may maintain viable populations in transformed environments outside protected areas (Gardner 2009; Gardner *et al*. 2009; Harris and Pimm 2004) and so assumed lower conservation priority within the *CVI*. Hunting for domestic consumption and collection for the global pet trade affect comparatively few species, but represent the primary extinction threat to those that are targeted (Jenkins *et al*. 2014; Raxworthy and Nussbaum 2000; Walker and Rafeliarisoa 2012).

We assigned scores to each species on a scale of 1-5 for each attribute (Table 5.1). For *E*, we visually estimated range thresholds using distribution maps in Glaw and Vences (2007). Watershed-based biogeographical models (Wilmé *et al*. 2006) were not used to identify micro-endemic species because they are not a good proxy for local endemism in reptiles (Pearson and Raxworthy 2009). Instead we used 10,000 km^2 (approximately 2% of Madagascar's land surface) as the threshold range size to distinguish between micro-endemics and endemic species restricted to a single bioclimatic region. *R* was scored on the basis of occurrences in protected areas within this study. *C* threat values were determined using CITES (Convention on International Trade in Endangered Species) listings and the literature on reptile declines in Madagascar. Scores for *T* were assigned on a review of the literature; species for which no degradation tolerance data were available were assumed to be degradation intolerant on the basis of the precautionary principle.

Table 5.1 Attributes and scoring criteria used in Conservation Value Index (*CVI*) and Zonation assessments (*E*, *C* and *T* only). PA = protected area.

	Rarity factors		Threat factors	
Score	**Scale of endemism (*E*)**	**Representation in sample PAs (*R*)**	**Hunting and collection pressure (*C*)**	**Degradation tolerance (*T*)**
1	Indigenous, non-endemic species	Recorded in 12-15 PAs (n > 75%)	No known threat	Tolerant of modified or artificial habitats
2	Widespread endemic, occurring in dry and humid regions	Recorded in 8-11 PAs (45 > n < 75%)	N/A	N/A
3	Endemic to dry regions	Recorded in 4-7 PAs (20 > n < 45%)	Known threat (CITES Appendix I and II), but not likely to cause local extirpations	Tolerant of edge effects, medium-intensity degradation or secondary growth.
4	Endemic to one bioclimatic region[a]	Recorded in 2-3 PAs (10 > n < 20%)	N/A	N/A
5	Local endemic, range size estimated as < 10,000 km^2	Recorded in only 1 PA (n < 10%)	Threat known to have caused local extirpations or severe population declines	Intolerant of low-intensity degradation

[a] Following Cornet (1974)

Since rarity and threat are likely to interact in influencing species viability, summed rarity and threat values were multiplied to produce a *CVI* score (range: 4-100).

$CVI_{species} = (E+R) \times (C+T)$

$SS_{CVI} = \Sigma\, CVI_{species}$

To test the sensitivity of *CVI* to variation in the weighting of individual attribute scores for species, we performed sensitivity analyses in which the relative weighting of each attribute was doubled.

5.2.4 Zonation

In order to produce a definitive benchmark against which to compare the site prioritisation protocols, we ran an assessment using a gold-standard systematic conservation planning software Zonation v3.1 (Moilanen *et al*. 2012). Zonation is a spatial conservation prioritisation framework which is based on species distributions defined using grid cells (Moilanen 2007). The underlying meta-algorithm starts from the full landscape and proceeds by iterative removal of cells (sites), at each step eliminating those which result in the smallest marginal loss in conservation value. The most important cells in the landscape are thus retained until last. Subsequently, Zonation produces a hierarchical ranking of conservation priority for each cell over the entire landscape (Moilanen *et al*. 2012). We therefore: i) converted the presence-absence data for each species into a raster grid format to identify the distribution of each species across the landscape; and, ii) used the Zonation additive-benefit function removal rule which bases selection on a cell's weighted summed occurrence value over all species (Moilanen 2007). With this cell removal rule, species occurrences are considered additive, so the cell that has the lowest value summed across all species will be removed at each step (Moilanen *et al*. 2012). The result is that species rich cells tend to have a higher value than cells with a high occurrence value for one or a few species. Species were weighted on the basis of endemism (*E*), hunting and collection pressure (*C*) and degradation tolerance (*T*) scores from *CVI*, according to the formula: weighting = (E) x (C + T). Representation scores were not included in the weighting as these data form the basis of the Zonation algorithm.

5.2.5 Comparison and assessment of site prioritisation indices

We evaluated the performance of each of the four prioritisation protocols by comparing the resultant site rankings with those produced by Zonation, using Spearman's rank correlations.

5.3 Results

The final dataset contained 134 species distributed across 12 families: Boidae (3), Chamaeleonidae (14), Crocodylidae (1), Gekkonidae (36), Gerrhosauridae (8), Iguanidae (6), Lamprophiidae (34), Pelomedusidae (2), Podocnemididae (1), Scincidae (24), Testudinidae (3), and Typhlopidae (2).

5.3.1 Species rankings

The three non-*Richness* site prioritisation protocols produced species rankings that are broadly similar, but important differences emerged for certain species (Table 5.2). For example, two tortoises (*Astrochelys radiata* and *Pyxis arachnoides*) ranked in the top 10 % of species using *CVI* and *Red List Index*, but featured in the lower 35 % of species using *Irreplaceability*. Using *CVI*, the 15 highest ranked species include four members of the order Testudines (tortoises and turtles), seven species in the family Chamaeleonidae (chameleons) and four species in the family Gekkonidae (geckos) (Appendix 1, Table A1.2).

5.3.2 Site rankings

Site species richness ranged from 17 in Kelifely to 72 in Ranobe PK32 (mean = 35.3). The site rankings produced by the four prioritisation protocols are all strongly positively correlated with the output of Zonation (Table 5.3, Fig. 5.2). In ascending order, the weakest correlation was between Zonation and *Richness* (r_s= 0.711, p < 0.01, n = 22), followed by Zonation and *Red List Index* (r_s= 0.861, p < 0.01, n = 22), Zonation and *Irreplaceability* (r_s= 0.920, p < 0.01, n = 22), and Zonation and *CVI* (r_s= 0.927, p < 0.01, n = 22). Inter-protocol variability was greater for lower, rather than higher, ranked sites (the highest ranked site being number 1; Fig. 5.2). Sensitivity analyses indicated that *CVI* is relatively robust to changes in individual species attribute weightings, with correlation coefficients ranging from 0.916 to 0.932 when each of the four attribute scores were doubled (Appendix 1, Table A1.3)'.

Table 5.2 Rank of the highest and lowest scoring 20 reptile species (n = 134) from the dry regions of Madagascar according to the conservation value index (*CVI*), and compared with equivalent scores and ranks generated by the red list (*RL*) and irreplaceability (*IR*) protocols.

Highest scoring 20 species according to *CVI*	*CVI* score	*CVI* rank	*RL* score	*RL* rank	*IR* score	*IR* rank
Brookesia bonsi	80	=1	5	=1	1	=1
Brookesia decaryi	80	=1	4	=7	1	=1
Brookesia exarmata	80	=1	4	=7	1	=1
Brookesia perarmata	80	=1	4	=7	1	=1
Furcifer belalandaensis	80	=1	5	=1	1	=1
Pyxis planicauda	80	=1	5	=1	1	=1
Erymnochelys madagascariensis	80	=1	5	=1	0.5	=37
Furcifer nicosiai	72	=8	4	=7	1	=1
Phelsuma breviceps	72	=8	3	=19	0.33	=59
Uroplatus henkeli	72	=8	3	=19	1	=1
Furcifer rhinoceratus	72	=8	3	=19	1	=1
Astrochelys radiata	70	=12	5	=1	0.17	=92
Pyxis arachnoides	70	=12	5	=1	0.17	=92
Phelsuma borai	64	=14	0	=122	0.5	=37
Uroplatus guentheri	64	=14	4	=7	0.33	=59
Ebenavia maintimainty	60	=16	4	=7	1	=1
Lygodactylus klemmeri	60	=16	2	=37	1	=1
Paragehyra petiti	60	=16	3	=19	1	=1
Pygomeles petteri	60	=16	4	=7	1	=1
Sirenoscincus yamagishii	60	=16	4	=7	1	=1
Lowest scoring 20 species according to *CVI*	***CVI* score**	***CVI* rank**	***RL* score**	***RL* rank**	***IR* score**	***IR* rank**
Zonosaurus laticaudatus	16	=110	1	=45	0.08	=123
Oplurus cyclurus	16	=110	1	=45	0.09	=117
Oplurus cuvieri	16	=110	1	=45	0.2	=85
Langaha madagascariensis	16	=110	1	=45	0.13	=104
Leioheterodon madagascariensis	16	=110	1	=45	0.13	=104
Lygodactylus tuberosus	14	120	1	=45	0.25	=75
Lygodactylus tolampyae	12	=121	1	=45	0.08	=123
Madagascarophis colubrinus	12	=121	1	=45	0.08	=123
Dromicodryas quadrilineatus	12	=121	1	=45	0.5	=37
Thamnosophis lateralis	12	=121	1	=45	0.33	=59
Furcifer verrucosus	10	=125	0	=122	0.09	=117
Paroedura picta	10	=125	1	=45	0.09	=117

Lowest scoring 20 species according to *CVI*	*CVI* score	*CVI* rank	*RL* score	*RL* rank	*IR* score	*IR* rank
Furcifer lateralis	8	=127	1	=45	0.1	=114
Furcifer oustaleti	8	=127	1	=45	0.09	=117
Chalarodon madagascariensis	8	=127	1	=45	0.09	=117
Trachylepis elegans	6	=130	1	=45	0.07	=131
Trachylepis gravenhorstii	6	=130	1	=45	0.08	=123
Dromicodryas bernieri	6	=130	1	=45	0.07	=131
Mimophis mahfalensis	6	=130	1	=45	0.07	=131
Hemidactylus mercatorius	4	134	1	=45	0.08	=128

Using *CVI* to compare sites of different protected status suggests that national parks (mean *CVI* = 1130.0, S.E. = 214.7, n = 8) are of greater conservation value for the reptile fauna than new protected areas (mean *CVI* = 942.3, S.E. = 203.4, n = 7) or unprotected sites (mean *CVI* = 517.4, S.E. = 38.5, n =7). This finding was consistent across all the prioritisation protocols.

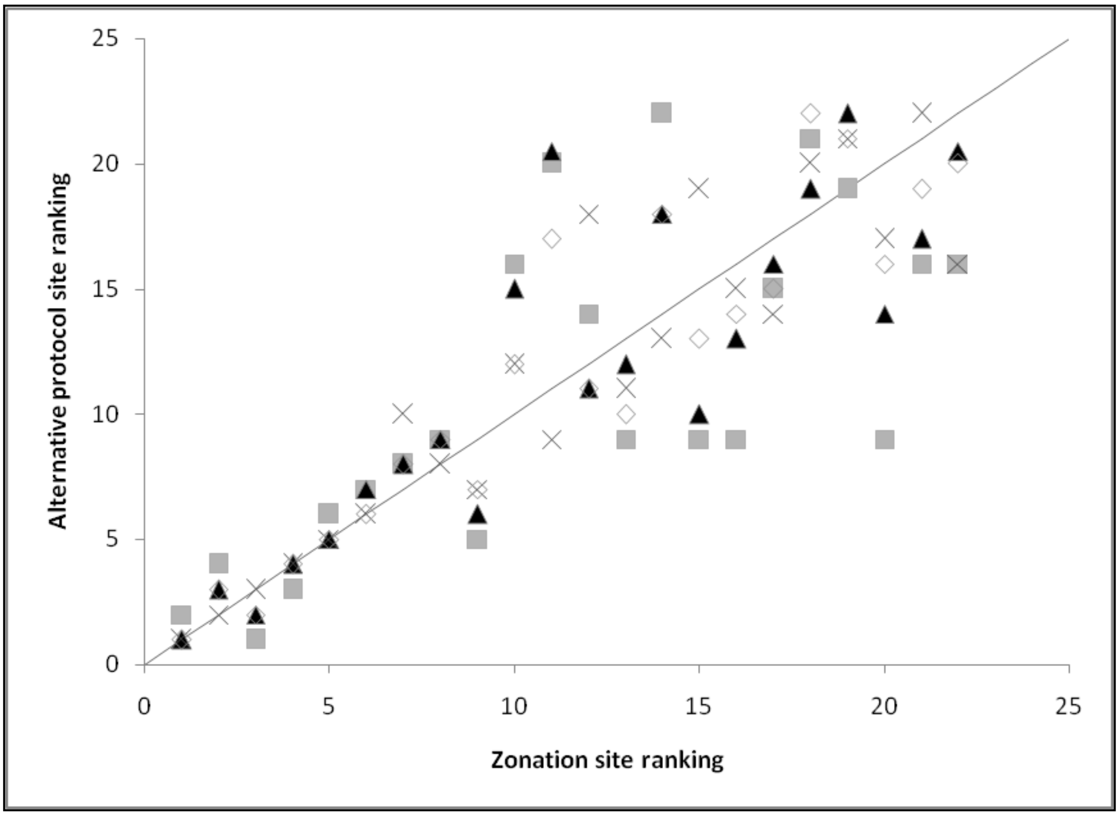

Figure 5.2 Correlation of site rankings produced by Zonation and four simple protocols: grey squares, richness; black triangles, red list index; crosses, irreplaceability; white diamonds, conservation value index (*CVI*). Solid line represents x = y.

Table 5.3 Site status, scores and ranks of 22 sites in the dry regions of Madagascar, prioritised for conservation value using Zonation (*Z*) and four alternative prioritisation protocols: *SR*, species richness; *RL*, red list index; *IR*, irreplaceability; *CVI*, conservation value index. NP, national park; NPA, new protected area; U, unprotected.

Site	Site status	*Z* rank	*SR* rank	*RL* rank	*IR* rank	*CVI* rank	*SR* score	*RL* score	*IR* score	*CVI* score
Tsingy de Bemaraha	NP	1	2	1	1	1	62	96	26.8	2054
Ankarafantsika	NP	2	4	3	2	3	53	83	20.0	1720
Ranobe PK32	NPA	3	1	2	3	2	72	91	15.7	1966
Mikea	NP	4	3	4	4	4	57	73	11.8	1544
Tsimanampesotse	NP	5	6	5	5	5	51	63	10.2	1342
Menabe Antimena	NPA	6	7	7	6	6	42	55	8.7	1130
Tsinjoriake	NPA	7	8	8	10	8	34	46	6.5	884
Namoroka	NP	8	=9	9	8	9	30	38	7.0	708
Amoron'i Onilahy	NPA	9	5	6	7	7	52	59	7.8	1124
Nosy-Ambositra	U	10	=16	15	12	12	24	31	5.2	586
Ankara	U	11	20	=20	9	17	19	23	6.7	508
Anadabolava-Betsimalaho	NPA	12	14	11	18	11	29	36	3.5	622
Andranomanitsy	U	13	=9	12	11	10	30	34	6.2	688
Kelifely	U	14	22	18	13	18	17	25	5.1	460
Andohahela P2	NP	15	=9	10	19	13	30	37	3.5	572
Zombitse-Vohibasia	NP	16	=9	13	15	14	30	33	3.8	572
Masoarivo	U	17	15	16	14	15	26	30	4.5	556
Berento	U	18	21	19	20	22	18	24	3.1	386
Behara-Tranomaro	NPA	19	19	22	21	21	23	22	2.6	422
Kirindy Mite	NP	20	=9	14	17	16	30	32	3.5	528
Nord Ifotaka	NPA	21	=16	17	22	19	24	28	2.5	448
Makay	U	22	=16	=20	16	20	24	23	3.6	438

5.4 Discussion

We used a range of prioritisation protocols to produce a ranking of Madagascar's dry forest sites for reptile conservation. Our results indicate that, as a group, the established generation of national parks are more valuable for reptile conservation than both the Durban Vision generation of new protected areas and hitherto unprotected sites; national parks comprise four of the five highest ranking sites for all protocols apart from species richness, and only one national park is in the bottom-ranked 30 %. Two unprotected sites, Nosy-Ambositra and Ankara, rank amongst the top 50 % of sites using Zonation, and therefore warrant consideration for future protected area establishment. In general, however, the analyses suggest that conservation funding would be best invested in the existing national park system and the Durban Vision generation of new protected areas in order to maximise the conservation of reptile biodiversity, rather than designating and managing further new protected areas.

The site rankings produced by all prioritisation protocols were strongly correlated with the outputs of Zonation because they are partially driven by species richness. Since each individual species score is positive, sites scores will increase with greater numbers of recorded species. Nevertheless, variation in the performance of different protocols when compared to the Zonation benchmark provides insight into the suitability of each for use in the prioritisation of protected areas for investment. Conservation assessments are intended to inform decisions, rather than provide definitive prescriptions (Knight *et al*. 2006; Pullin *et al*. 2013). Ideally, in any prioritisation assessment, decision-makers should use a systematic approach such as Zonation whenever they have the capacity to do so, and incorporate data reflecting species value if they are available. This is why we included additional data reflecting species threat status into our *CVI* protocol and benchmark assessment, thus ensuring the evidence-based was as robust as possible. It would have been best practice to have integrated non-biodiversity data, such as cost information (Cawardine *et al*. 2010; Joseph *et al*. 2009) and the relative effectiveness (Nelson and Chomitz 2011) of the different protected area models employed in SAPM, into the assessment. However, such data are unavailable for Madagascar, hence they were not used in the prioritisation exercise that informed the location of new protected areas within the Durban Vision expansion (Corson 2014). In situations where the capacity for such systematic assessments is lacking, decision-makers should seek to develop or 'borrow' the necessary expertise by collaborating with research institutions (Knight *et al*. 2011; Smith *et al*. 2009).

However, in situations where systematic conservation assessment software cannot be used, simple indices can provide a transparent, repeatable evidence-base to inform prioritisation decision-making, thus representing an improvement on non-systematic approaches. The simplest such index is species richness, but this metric performed relatively poorly in our analysis, and would have identified Ranobe PK32 as the most important site for reptile conservation in our sample. All other protocols consistently rank Tsingy de Bemaraha as the most valuable site, despite it harbouring only 86 % as many species as Ranobe PK32, because 28 % of its reptiles are locally endemic (Bora *et al*. 2010). Furthermore, species richness is not an accurate indicator of conservation value (Barlow *et al*. 2007; Rey Benayas and de la Montaña 2003) because all species are not equal. While value can be assigned to species according to a range of criteria (e.g. genotypic (Diniz 2004) or phenotypic (Owens and Bennett 2000) distinctiveness, public preferences (Smith *et al*. 2012), or ecological function (Scheiner 2012)), we differentiated between species using parameters that reflect extinction risk as this is the most urgent issue facing conservationists (Brooks *et al*. 2004b). Understanding the threats faced by species is critical to estimating their vulnerability and thus dependence on conservation interventions (Raxworthy and Nussbaum 2000), yet systematic conservation planning assessments rarely incorporate such data.

The richness of threatened species is often used to inform prioritisations (e.g. Schwitzer *et al*. 2013, 2014), and the strong (0.86) positive correlation between the *Red List Index* and Zonation suggests that this metric may be a useful proxy measure if the necessary data are available. However the *Red List Index* failed to account for data deficient and unevaluated species, and thus almost 10 % of species in our sample were given no score. Additionally, since the use of such an index is dependent on the availability of full, up-to-date Red List assessments, its utility will be limited for many taxonomic groups and geographical regions, given that only 2.75 % of described species had been evaluated by 2010 (Pullin *et al*. 2013).

The concept of irreplaceability is a key metric in systematic conservation assessments, and the *Irreplaceability* index performed well in comparison to Zonation. However, measures of irreplaceability alone may not adequately reflect conservation value because some species may be widespread and occur in a number of protected areas, yet remain highly threatened. For example, the tortoises *Astrochelys radiata* and *Pyxis arachnoides* were ranked low in terms of irreplaceability as both species are present in six protected areas, but this prioritisation protocol failed to account for the rapid, range-wide declines in population density (Leuteritz *et al*. 2005; Walker *et al*. 2013) that have seen both species classed as Critically Endangered (van Dijk *et al*. 2013). In addition, care is needed when dealing with species that are commonly found outside the sites being considered.

For example, the gecko *Phelsuma modesta* was recorded at only one site and therefore ranked high in terms of irreplaceability, although its abundance in heavily modified, non-forest habitats (e.g. urban areas (Gardner and Jasper 2009; Glaw and Vences 2007)) demonstrate that it is not dependent on the effective management of protected areas for its survival. Similar problems may arise if species occur within the study region, but only at the periphery of their range (e.g. if reptiles widespread in humid eastern Madagascar occurred at sites on the edge of the dry forest). However, in the case of the current case study, this issue is mitigated by the extremely high rates of species turnover between the humid and dry regions of Madagascar (Glaw and Vences 2007; Jenkins *et al*. 2014). The problem of species that appear scarce in a dataset but, in reality, are not, will afflict any richness or complementarity-based analysis. In such cases, one might consider excluding these species, or using an explicitly target-based approach to measuring irreplaceability that sets lower targets for species deemed of minimal conservation importance by planners. Nonetheless, this risks introducing an element of subjectivity into the prioritisation exercise unless the species can be identified systematically by, for instance, using *CVI*.

The strongest correlation between site rankings was produced for Zonation and *CVI*, suggesting that the latter index could be used to inform protected area prioritisation in situations where more complex analyses are not feasible. The index incorporated measures of rarity (a proxy for irreplaceability) and threat (a proxy for Red List status). As it only used readily-available inventory data, which was compiled into a database of species presence, and published literature to assign attribute scores using a simple scoring system, it can be easily adopted by decision-makers without the need for specific training. Although *CVI* performed well in prioritising the forests of Madagascar's dry regions for reptile conservation, additional case studies are needed to further examine its utility and performance. In particular, the approach may be most appropriate for dealing with a small number of pre-identified sites (e.g. prioritising across an existing protected area portfolio), rather than for carrying out a conservation assessment which might seek to prioritise among many (i.e. hundreds or even thousands) of localities to optimise the establishment of new protected areas.

The prioritisation of existing protected areas for investment is important if we are to maximise the effectiveness of protected area networks for biodiversity conservation. While sophisticated analytical tools can and should be used to inform such decisions, decision-makers often lack the capacity to use them. Instead, they frequently rely on non-transparent, subjective processes or simple measures such as species richness or the number of threatened species. It is therefore important to understand how such metrics can perform and in what circumstances they should be used. Our analysis suggests that some simple indices can provide a transparent framework to support evidence-based decision-making by practitioners, although their performance is variable. Our *CVI*, which incorporates measures of rarity and threat for individual species, appears to provide a useful alternative to more sophisticated, gold-standard systematic conservation planning tools, and emphasises the benefits of integrating species-specific data into conservation assessments.

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References

AGRECO. 2012. *Analyse des Coûts et Sources de Financement du Système des Aires Protégées de Madagascar (Octobre 2010 – Janvier 2012)*. AGRECO, Antananarivo.

Allnutt, T.F., Asner, G.P., Golden, C.D. and Powell, G.V.N. 2013. Mapping recent deforestation and forest disturbance in northeastern Madagascar. *Tropical Conservation Science* 6: 1–15.

Balmford, A., Gaston, K.J., Blyth, S., James, A. and Kapos, V. 2003. Global variation in terrestrial conservation costs, conservation benefits, and unmet conservation needs. *Proceedings of the National Academy of Sciences, USA* 100: 1046–1050.

Barlow, J., Gardner, T.A., Araujo, I.S., Avila-Pires, T.C., Bonaldo, A.B., Costa, J.E., Esposito, M.C., Ferreira, L.V., Hawes, J., Hernandez, M.I.M., *et al*. 2007. Quantifying the biodiversity value of tropical primary, secondary, and plantation forests. *Proceedings of the National Academy of Sciences, USA* 104: 18555–18560.

Barr, L.M., Pressey, R.L., Fuller, R.A., Segan, D.B., McDonald-Madden and Possingham, H.P. 2011. A new way to measure the world's protected area coverage. *PLoS ONE* 6: e24707

Beresford, A.E., Buchanan, G.M., Donald, P.F., Butchart, S.H.M., Fishpool, L.D.C. and Rondini, C. 2011. Poor overlap between the distribution of protected areas and globally threatened birds in Africa. *Animal Conservation* 14: 99–107.

Bertzky, B., Corrigan, C., Kemsey, J., Kenney, S., Ravilious, C., Besançon, C. and Burgess, N. 2012. *Protected Planet Report 2012: Tracking Progress Towards Global Targets for Protected areas*. IUCN, Gland and UNEP-WCMC, Cambridge.

Bora, P., Randrianantoandro, J.C., Randrianavelona, R., Hantalalaina, E.F., Andriantsimanarilafy, R.R., Rakotondravony, D., Ramilijaona, O.R., Vences, M., Jenkins, R.K.B., Glaw, F., *et al*. 2010. Amphibians and reptiles of the Tsingy de Bemaraha Plateau, western Madagascar: checklist, biogeography and conservation. *Herpetological Conservation and Biology* 5: 111–125.

Bottrill, M.C. and Pressey, R.L. 2012 The effectiveness and evaluation of conservation planning. *Conservation Letters* 5: 407–420.

Bottrill, M.C., Joseph, L.N., Cawardine, J., Bode, M., Cook, C., Game, E.T., Grantham, H., Kark, S., Linke, S., McDonald-Madden, E., *et al*. 2008. Is conservation triage just smart decision making? *Trends in Ecology and Evolution* 23: 649–654.

Brandon, K., Sanderson, S. and Redford, K. 1998. *Parks in Peril: People, Politics, and Protected Areas*. Island Press, Washington DC.

Brooks, T.M., Bakarr, M.I., Boucher, T., da Fonseca, G.A.B., Hilton-Taylor, C., Hoekstra, J.M., Moritz, T., Olivieri, S., Parrish, J., Pressey, R.L., *et al*. 2004a. Cosverage provided by the global protected-area system: is it enough? *BioScience* 54: 1081–1091.

Brooks, T., da Fonseca, G.A.B. and Rodrigues, A.S.L. 2004b. Species, data, and conservation planning. *Conservation Biology* 18: 1682–1688.

Brooks, T.M., Mittermeier, R.A., da Fonseca, G.A.B., Gerlach, J., Hoffmann, M., Lamoreux, J.F., Mittermeier, C.G., Pilgrim, J.D. and Rodrigues, A.S.L. 2006. Global biodiversity conservation priorities. *Science* 313: 58–61.

Brooks, T.M., Wright, S.J. and Sheil, D. 2009. Evaluating the success of conservation actions in safeguarding tropical forest biodiversity. *Conservation Biology* 23: 1448–1457.

Brugière, D. and Scholte, P. 2013. Biodiversity gap analysis of the protected area system in poorly-documented Chad. *Journal for Nature Conservation* 21: 286–293.

Bruner, A.G., Gullison, R.E., Rice, R.E. and da Fonseca, G.A.B. 2001. Effectiveness of parks in protecting tropical biodiversity. *Science* 291: 125–128.

Bruner, A.G., Gullison, R.E. and Balmford, A. 2004. Financial costs and shortfalls of managing and expanding protected-area systems in developing countries. *BioScience* 54: 1119–1126.

Cadle, J.E. and Ineich, I. 2008. Nomenclatural status of the Malagasy snake genus *Bibilava* Glaw, Nagy, Franzen, and Vences, 2007: Resurrection of *Thamnosophis* Jan and designation of a lectotype for *Leptophis lateralis* Duméril, and Bibron, and Duméril (Serpentes: Colubridae). *Herpetological Review* 39: 285–288.

Cawardine, J., Wilson, K.A., Hajkowicz, S.A., Smith, R.J., Klein, C.J., Watts, M. and Possingham, H.P. 2010. Conservation planning when costs are uncertain. *Conservation Biology* 24: 1529–1537.

Cook, C.N., Hockings, M. and Carter, R.W. 2010. Conservation in the dark? The information used to support management decisions. *Frontiers in Ecology and the Environment* 8: 181–186.

Cornet, A. 1974. *Essai de Cartographie Bioclimatique à Madagascar*. Orstrom, Paris.

Corson, C. 2014. Conservation politics in Madagascar: the expansion of protected areas. In *Conservation and Environmental Management in Madagascar*, I.R. Scales (Ed.), pp. 193–215. Routledge, London and New York.

Daily, G.C. 2001. Ecological forecasts. *Nature* 411: 245–245.

Diniz, J.A.F. 2004. Phylogenetic diversity and conservation priorities under distinct models of phenotypic evolution. *Conservation Biology* 18: 698–704.

Fischer, J., Lindenmayer, D.B. and Manning, A.D. 2006. Biodiversity, ecosystem function, and resilience: ten guiding principles for commodity production landscapes. *Frontiers in Ecology and the Environment* 4: 80–86.

Fuller, R.A., McDonald-Madden, E., Wilson, K.A., Cawardine, J., Grantham, H.S., Watson, J.E.M., Klein, C.J., Green, D.C. and Possingham, H.P. 2010. Replacing underperforming protected areas achieves better conservation outcomes. *Nature* 466: 365 –367.

Game, E.T., Kareiva, P. and Possingham, H.P. 2013. Six common mistakes in conservation priority setting. *Conservation Biology* 27: 480–485.

Gardner, C.J. 2009. A review of the impacts of anthropogenic habitat change on terrestrial biodiversity in Madagascar: implications for the design and management of new protected areas. *Malagasy Nature* 2: 2–29.

Gardner, C.J. 2011. IUCN management categories fail to represent new, multiple-use protected areas in Madagascar. *Oryx* 45: 336–346.

Gardner, C. and Jasper, L. 2009. The urban herpetofauna of Toliara, southwest Madagascar. *Herpetology Notes* 2: 239–242.

Gardner, C.J., Nicoll, M.E., Mbohoahy, T., Olesen, K.L.L., Ratsifandrihamanana, A.N., Ratsirarson, J., René de Roland, L.-A., Virah-Sawmy, M., Zafindrasilivonona, B. and Davies, Z.G. 2013. Protected areas for conservation and poverty alleviation: experiences from Madagascar. *Journal of Applied Ecology* 50: 1289–1294.

Gardner, T.A., Barlow, J., Chazdon, R., Ewers, R.M., Harvey, C.A., Peres, C.A. and Sodhi, N.S. 2009. Prospects for tropical forest biodiversity in a human-modified world. *Ecology Letters* 12: 561–582.

Gaston, K.J., Jackson, S.F., Nagy, A., Cantú-Salazar, L. and Johnson, M. 2008. Protected areas in Europe: principle and practice. *Annals of the New York Academy of Sciences* 1134: 97–119.

Gaveau, D.L.A., Epting, J., Lyne, O., Linkie, M., Kumara, I., Kanninen, M. and Leader-Williams, N. 2009. Evaluating whether protected areas reduce tropical deforestation in Sumatra. *Journal of Biogeography* 36: 2165–2175.

Gibbons, J.W., Scott, D.E., Ryan, T.J., Buhlmann, K.A., Tuberville, T.D., Metts, B.S., Greene, J.L., Mills, T., Leiden, Y., Poppy, S., *et al*. 2000. The global decline of reptiles, déjà vu amphibians. *BioScience* 50: 653–666.

Glaw, F. and Vences, M. 2007. *A Field Guide to the Amphibians and Reptiles of Madagascar*. Third Edition. Vences & Glaw Verlag, Köln.

Goodman, S.M. and Benstead, J.P. 2005. Updated estimates of biotic diversity and endemism for Madagascar. *Oryx* 39: 73–77.

Harper, G.J., Steininger, M.K., Tucker, C.J., Juhn, D. and Hawkins, F. 2007. Fifty years of deforestation and forest fragmentation in Madagascar. *Environmental Conservation* 34: 325–333.

Harris, G.M. and Pimm, S.L. 2004. Bird species' tolerance of secondary forest habitats and its effects on extinction. *Conservation Biology* 18: 1607–1616.

Holt, B.G., Lessard, J.P., Borregaard, M.K., Fritz, S.A., Araújo, M.B., Dimitrov, D., Fabre, P.-H., Graham, C.H., Graves, G.R., Jønsson, K.A., *et al*. 2013. An update of Wallace's zoogeographic regions of the world. *Science* 339: 74–78.

IUCN (International Union for Conservation of Nature). 2012. IUCN Red List of Threatened Species http://www.iucnredlist.org/. Accessed 19 March 2013.

James, A.N., Gaston, K.J. and Balmford, A. 1999. Balancing the Earth's accounts. *Nature* 401: 323–324.

Jenkins, C.N. and Joppa, L. 2009. Expansion of the global terrestrial protected area system. *Biological Conservation* 142: 2166–2174.

Jenkins, R.K.B., Tognelli, M.F., Bowles, P., Cox, N., Brown, J.L., Chan, L., Andreone, F., Andriamazava, A., Andriantsimanarilafy, R.R., Anjeriniaina, M., *et al*. 2014. Extinction risks and the conservation of Madagascar's reptiles. *PLoS ONE* 9: e100173.

Joppa, L.N. and Pfaff, A. 2011. Global protected area impacts. *Proceedings of the Royal Society B, Biological Sciences* 278: 1633–1638.

Joseph, L.N., Maloney, R.F. and Possingham, H.P. 2009. Optimal allocation of resources among threatened species: a project prioritisation protocol. *Conservation Biology* 23: 328 –338.

Knight, A.T., Cowling, R.M. and Campbell, B.M. 2006. An operational model for implementing conservation action. *Conservation Biology* 20: 408–419.

Knight, A.T., Cowling, R.M., Boshoff, A.F., Wilson, S.L. and Pierce, S.M. 2011. Walking in step: lessons for linking spatial prioritisations to implementation strategies. *Biological Conservation* 111: 202–211.

Köhler, G., Diethert, H.H., Nussbaum, R. and Raxworthy, C.J. 2009. A revision of the fish scale geckos, genus *Geckolepis* Grandidier (Squamata, Gekkonidae) from Madagascar and the Comoros. *Herpetologica* 65: 419–435.

Kremen, C., Cameron, A., Moilanen, A. Phillips, S.J., Thomas, C.D., Beentje, H., Dransfield, J., Fisher, B.L., Glaw, F., Good, T.C., *et al*. 2008. Aligning conservation priorities across taxa in Madagascar with high-resolution planning tools. *Science* 320: 222 –226.

Le Saout, S., Hoffmann, M., Shi, Y., Hughes, A., Bernard, C., Brooks, T.M., Bertzky, B., Butchart, S.H.M., Stuart, S.N., Badman, T., *et al*. 2013. Protected areas and effective biodiversity conservation. *Science* 342: 803–805.

Leuteritz, T.E.J., Lamb, T. and Limberaza, J.C. 2005. Distribution, status, and conservation of radiated tortoises (*Geochelone radiata*) in Madagascar. *Biological Conservation* 124: 451–461.

Marris, E. 2007. What to let go. *Nature* 450: 152–155.

Margules, C.R. and Pressey, R.L. 2000. Systematic conservation planning. *Nature* 405: 243–253.

Moilanen, A. 2007. Landscape zonation, benefit functions and target based planning: unifying reserve selection strategies. *Biological Conservation* 134: 571–579.

Moilanen, A., Meller, L., Leppänen, J., Pouzols, F.M., Arponen, A. and Kujala, H. 2012. *Zonation v3.1: Spatial Conservation Planning Framework and Software. User manual*. University of Helsinki, Helsinki.

Nagy, Z.T., Glaw, F. and Vences, M. 2010. Systematics of the snake genera *Stenophis* and *Lycodryas* from Madagascar and the Comoros. *Zoologica Scripta* 39: 426–435.

Nagy, Z.T., Sonet, G., Glaw, F. and Vences, M. 2012. First large-scale DNA barcoding assessment of reptiles in the biodiversity hotspot of Madagascar, based on newly designed COI primers. *PLoS ONE* 7: e34506.

Nelson, A. and Chomitz, K.M. 2011. Effectiveness of strict vs. multiple use protected areas in reducing tropical forest fires: a global analysis using matching methods. *PLoS ONE* 6: e22722.

Olson, D. and Dinerstein, E. 1998. The Global 200: a representation approach to conserving the world's biologically valuable ecoregions. *Conservation Biology* 12: 502–515.

ONE (Office National pour l'Environnement), DGF (Direction Générale des Forêts), FTM (Foiben-Taosarintanin'i Madagascar), MNP (Madagascar National Parks) and Conservation International. 2013. *Evolution de la Couverture des Forêts Naturelles à Madagascar 2005-2010*. Office National pour l'Environnement, Antananarivo.

Owens, I.P.F. and Bennett, P.M. 2000. Quantifying biodiversity: a phenotypic perspective. *Conservation Biology* 14: 1014–1022.

Pearson, P.G. and Raxworthy, C.J. 2009. The evolution of local endemism in Madagascar: watershed vs. climatic gradient hypotheses evaluated by null biogeographic models. *Evolution* 63: 959–967.

Pressey, R.L. 1994. Ad hoc reservations: forward or backward steps in developing representative reserve systems? *Conservation Biology* 8: 662–668.

Pullin, A.S. and Knight, T.M. 2005. Assessing conservation management's evidence base: a survey of management-plan compilers in the United Kingdom and Australia. *Conservation Biology* 19: 1989–1996.

Pullin, A.S., Knight, T.M., Stone, D.A. and Charman, K. 2004. Do conservation managers use scientific evidence to support their decision-making? *Biological Conservation* 119: 245–252.

Pullin, A.S., Sutherland, W., Gardner, T., Kapos, V. and Fa, J.E. 2013. Conservation priorities: identifying need, taking action and evaluating success. In *Key Topics in Conservation Biology 2*, D.W. Macdonald and K.J. Willis (Eds.), pp. 3–22. Wiley-Blackwell, Oxford.

Raik, D. 2007. Forest management in Madagascar: an historical overview. *Madagascar Conservation & Development* 2: 5–10.

Randrianandianina, B.N., Andriamahaly, L.R., Harisoa, F.M. and Nicoll, M.E. 2003. The role of protected areas in the management of the island's biodiversity. In *The Natural History of Madagascar*, S.M. Goodman and J.P. Benstead (Eds.), pp. 1423–1432. The University of Chicago Press, Chicago.

Rasoavahiny, L., Andrianarisata, M., Razafimpahanana, A. and Ratsifandrihamanana, A.N. 2008. Conducting an ecological gap analysis for the new Madagascar protected area system. *Parks* 17: 12–21.

Raxworthy, C.J. 2003. Introduction to the reptiles. In *The Natural History of Madagascar*, S.M. Goodman and J.P. Benstead (Eds.), pp. 934–949. The University of Chicago Press, Chicago.

Raxworthy, C.J. and Nussbaum, R.A., 2000. Extinction and extinction vulnerability of amphibians and reptiles in Madagascar. *Amphibian and Reptile Conservation* 2: 15–23.

Raxworthy, C.J., Ingram, C.M., Rabibisoa, N. and Pearson, R.G. 2007. Applications of ecological niche modelling for species delimitation: a review and empirical evaluation using day geckos (*Phelsuma*) from Madagascar. *Systematic Biology* 56: 907–923.

Rey Benayas, J.M. and de la Montaña, E. 2003. Identifying areas of high-value vertebrate diversity for strengthening conservation. *Biological Conservation* 114: 357–370.

Rodrigues, A.S.L., Andelman, S.J., Bakarr, M.I., Boitani, L., Brooks, T.M., Cowling, R.M., Fishpool, L.D.C., da Fonseca, G.A.B., Gaston, K.J., Hoffmann, M., *et al*. 2004. Effectiveness of the global protected area network in representing species diversity. *Nature* 428: 640–643.

Scheiner, S.M. 2012. A metric of biodiversity that integrates abundance, phylogeny, and function. *Oikos* 121: 1191–1202.

Schwitzer, C., Mittermeier, R.A., Davies, N., Johnson, S., Ratsimbazafy, J., Razafindramanana, J., Louis Jr., E.E. and Rajaobelina, S. 2013. *Lemurs of Madagascar: a Strategy for their Conservation 2013–2016*. IUCN, Gland.

Schwitzer, C., Mittermeier, R.A., Johnson, S.E., Donati, G., Irwin, M., Peacock, H., Ratsimbazafy, J., Razafindramanana, J., Louis Jr., E.E., Chikhi, L., *et al*. 2014. Averting lemur extinctions amid Madagascar's political crisis. *Science* 343: 842–843.

Smith, R.J., Veríssimo, D., Leader-Williams, N., Cowling, R.M. and Knight, A.T. 2009. Let the locals lead. *Nature* 462: 280–281.

Smith, R.J., Veríssimo, D., Isaac, N.J.B. and Jones, K.E. 2012. Identifying Cinderella species: uncovering mammals with conservation flagship appeal. *Conservation Letters* 5: 205-212.

Tang, L., Shao, G., Piao, Z., Dai, L., Jenkins, M.A., Wang, S., Wu, G., Wu, J. and Zhao, J. 2010. Forest degradation deepens around and within protected areas in East Asia. *Biological Conservation* 143: 1295–1298.

van Dijk, P.P., Leuteritz, T.E.J., Rhodin, A.G.J., Mittermeier, R.A. and Randriamahazo, H. 2013. Turtles on the brink in Madagascar: an IUCN red listing assessment workshop. In *Turtles on the Brink in Madagascar: Proceedings of Two Workshops on the Status, Conservation, and Biology of Malagasy Tortoises and Freshwater Turtles*, C.M. Castellano, A.G.J. Rhodin, M. Ogle, R.A. Mittermeier, H. Randriamahazo, R. Hudson and R.E. Lewis (Eds.). *Chelonian Research Monographs* 6: 33–36.

Vences, M., Wollenberg, K.C., Vieites, D.R. and Lees, D.C. 2009. Madagascar as a model region of species diversification. *Trends in Ecology and Evolution* 24: 456–465.

Virah-Sawmy, M., Gardner, C.J. and Ratsifandrihamanana, A.N. 2014. The Durban Vision in practice: experiences in the participatory governance of Madagascar's new protected areas. In *Conservation and Environmental Management in Madagascar*, I.R. Scales (Ed.), pp. 216–251. Routledge, London and New York.

Waldron, A., Mooers, A.O., Miller, D.C., Nibbelink, N., Redding, D., Kuhn, T.S., Roberts, J.T. and Gittleman, J.L. 2014. Targeting global conservation funding to limit immediate biodiversity declines. *Proceedings of the National Academy of Sciences, USA*. doi: 10.1073/pnas.1221370110

Walker, R.C.J. and Rafeliarisoa, T.H. 2012. Distribution of radiated tortoise (*Astrochelys radiata*) bush meat poaching effort. *Chelonian Conservation and Biology* 11: 223–226.

Walker, R.C.J., Gardner, C.J., Rafeliarisoa, T.H., Smith, I., Razafimanatsoa, R. and Louis Jr., E.E. 2013. Conservation of the Madagascar spider tortoise (*Pyxis arachnoides*) amid changing land use policy: assessing the spatial coincidence of relict populations with protected areas and mining concessions. In *Turtles on the Brink in Madagascar: Proceedings of Two Workshops on the Status, Conservation, and Biology of Malagasy Tortoises and Freshwater Turtles*, C.M. Castellano, A.G.J. Rhodin, M. Ogle, R.A. Mittermeier, H. Randriamahazo, R. Huson and R.E. Lewis (Eds.). *Chelonian Research Monographs* 6: 135–145.

Wilmé, L., Goodman, S.M. and Ganzhorn, J.U. 2006. Biogeographic evolution of Madagascar's microendemic biota. *Science* 312: 1063–1065.

Wilson, R.S. 2008. Balancing emotion and cognition: a case for decision-aiding in conservation efforts. *Conservation Biology* 22: 1452–1460.

Chapter 6

The impact of natural resource use on bird and reptile communities within multiple-use protected areas: evidence from Madagascar

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Abstract

Multiple-use protected areas, in which sustainable levels of extractive livelihood activities are permitted, play an increasingly important role in the global protected area estate, and are expected to rise in prevalence in future. However, we know little about their effectiveness at conserving biodiversity. We surveyed bird and reptile communities in three areas across a forest disturbance gradient resulting from legal and illegal activities (charcoal production and shifting cultivation respectively) within a new, multiple-use protected area in Madagascar's sub-arid spiny forest. We scored individual species using a Conservation Value Index (CVI; a simple metric based on rarity (endemism and representation in the protected area network), threat and distinctiveness) and estimated the total conservation value of each treatment by calculating the sum of frequency-weighted CVI scores across all present species. Bird and reptile community responses to forest disturbance were idiosyncratic. Bird species richness was greatest in the moderate-disturbance treatment, but the low-disturbance treatment had the superior conservation value due to higher frequencies of locally-endemic species. Reptile species richness was

the same in low- and moderate-disturbance treatments, but the conservation value of the latter was greater. The high-disturbance areas had lowest richness and conservation value for both groups. For birds, increasing disturbance levels were accompanied by community turnover from high-value to low-value species, a pattern highlighted by CVI that is masked by assessing species richness alone. Although some endemic species appear to be resilient to degradation, multiple-use protected areas in Madagascar may lose biodiversity since most endemic species are forest-dependent. Stricter protected area models may be more appropriate in areas where much of the high-value biodiversity is sensitive to habitat degradation.

6.1 Introduction

The impacts of human activity now threaten most of the Earth's species and ecosystems (Ehrlich and Pringle 2008) and have precipitated the planet's sixth mass extinction (Chapin *et al*. 2000). Our primary strategy to stem this biodiversity loss is the creation and management of protected areas, which cover over 12 % of the world's land area (Bertzky *et al*. 2012; Jenkins and Joppa 2009) and constitute the largest planned land use in history (Chape *et al*. 2005). All protected areas are spaces "recognised, dedicated and managed... to achieve the long-term conservation of nature with associated ecosystem services and cultural values" (Dudley 2008), but they vary greatly in management objective and approach. These differences form the basis for the World Conservation Union's (IUCN) protected area categorisation system (Dudley 2008; Dudley *et al*. 2010). For simplicity's sake the categories are often divided into 'strict' protected areas (generally categories I-IV), which seek to isolate nature from human processes that threaten it, and 'multiple-use' sites, which promote conservation through the sustainable extractive use of natural resources (category VI) or traditional land uses that sustain biodiversity (category V).

Recent decades have seen the number of multiple-use protected areas grow significantly in many parts of the world (Bertzky *et al*. 2012). Although some strict sites have been downgraded (Mascia *et al*. 2014), this has been driven primarily by the predominance of multiple-use categories amongst new protected areas (Zimmerer *et al*. 2004). The trend can largely be attributed to: i) the lack of remaining 'wilderness' areas, with a low human footprint, suitable for the creation of strict categories (Hoekstra *et al*. 2005; Leroux *et al*. 2010); and, ii) a paradigm shift in conservation, reflecting concern for the effects of exclusionary approaches on human wellbeing (Adams and Hutton 2007; Miller 2014; Roe

2008), and the suggestion that sustainable use may be a more effective long-term conservation strategy than strict protection (Hutton and Leader-Williams 2003; Rosser and Leader-Williams 2010). As a result, only 45 % of the world's protected areas are assigned to categories I-IV (Jenkins and Joppa 2009), and category VI sites expanded from 14 to 32 % of the world's protected area estate between 1990 and 2010 (Bertzky *et al*. 2012). This trend is expected to become even more pronounced in the future (McDonald and Boucher 2011).

Signatories to the Convention of Biological Diversity are expected to increase the coverage of terrestrial protected areas to 17 % of their national territory by 2020 and ensure that they are "effectively managed" (CBD 2010). Thus, if new protected areas are expected to largely comprise multiple-use categories, it is important to know whether they are likely to be effective at achieving their objective – the long-term conservation of nature – in the face of authorised human impacts. This is particularly apposite given the suggestion that multiple-use sites are less important for biodiversity (Gaston *et al*. 2008) and should not be classified as protected areas at all (Locke and Dearden 2005).

The effectiveness of protected areas depends on both their coverage (i.e. ensuring that maximum biodiversity is represented within them) and their success in buffering the biodiversity from the processes that threaten its viability (Gaston *et al*. 2008; Margules and Pressey 2000). Evaluations of effectiveness have generally focussed on the former, meaning that we know little about the success of protected areas in maintaining their condition over time (Cabeza 2013; Geldmann *et al*. 2013). This knowledge gap is particularly acute with regards to multiple-use categories. Global studies comparing across categories have found stricter protected areas to be more effective at slowing deforestation in some regions (Joppa and Pfaff 2011; Scharlemann *et al*. 2010), whereas multiple-use sites demonstrate greater success in other countries (Ferraro *et al*. 2013; Nelson and Chomitz 2011). However, the use of remote sensed data within such analyses only allows us to quantify vegetation cover, therefore providing little insight into the ecological integrity of remaining natural vegetation and faunal communities beneath the canopy (Peres *et al*. 2006). Less conspicuous changes to forest structure and composition (i.e. forest degradation) can stem from activities such as non-industrial selective logging, fuelwood collection, livestock grazing and the harvesting of non-timber forest products (NTFPs). Typically, these are precisely the types of activity that are sanctioned within category V and VI protected areas (Dudley 2008) as they sustain the livelihoods of hundreds of millions of people worldwide (Vedeld *et al*. 2007; Vira and Kontoleon 2010). Indeed, conservationists still have a very limited understanding of species and community responses to habitat change, and our knowledge is largely derived from a small number of sites (Barlow *et al*. 2007; T. Gardner *et al*. 2009, 2010). Furthermore, few researchers have investigated the impacts of subsistence activities on biodiversity (Borghesio 2008; Brown *et al*. 2013; Kumar and Shahabuddin 2006).

Madagascar is an example of a biodiversity-rich tropical developing country that is expanding its protected area system through the creation of new multiple-use sites. The island is a global conservation priority, boasting an unparalleled combination of species diversity and endemism (Brooks *et al*. 2006; Myers *et al*. 2000), with the majority of its endemic biota being forest dependent (Goodman and Benstead 2005). However, less than 16 % of the country retained forest cover by 2000 (Harper *et al*. 2007; McConnell and Kull 2014). Since 2003, Madagascar has been in the process of tripling the coverage of its protected area system, from 1.7 to over 6 million ha, in response to lobbying from international conservation organisations and funders (Corson 2014; Duffy 2006). Known as the 'Durban Vision' after the location of the fifth World Parks Congress at which it was launched, this ambitious programme has necessitated modifications to the country's conception of protected areas and their governance. Previously, all protected areas were governed by the State, managed by the para-statal Madagascar National Parks, and comprised only strict categories (I, II and IV; Randrianandianina *et al*. 2003). Most of the new protected areas established as part of the Durban Vision are co-managed by non-governmental organisations (NGOs) and local communities, and are proposed or designated as categories V and VI (AGRECO 2012; Gardner 2011; Virah-Sawmy *et al*. 2014), with zoned areas where subsistence and low-level commercial natural resource use activities are permitted (e.g. Gardner *et al*. 2008; Virah-Sawmy *et al*. 2014; WWF 2010).

The goals of the expanded Madagascar Protected Area System (SAPM) are to conserve the country's unique biodiversity and its cultural heritage, as well as promoting the sustainable use of natural resources for poverty alleviation and development (Commission SAPM 2006). The simultaneous achievement of these goals is particularly complex because most forms of traditional land and resource use in Madagascar have negative impacts on biodiversity (Gardner 2009, 2011; Irwin *et al*. 2010). Planning the management of new multiple-use protected areas requires an understanding of species and community responses to habitat degradation arising from permitted resource use, yet our knowledge of the influence this has on biodiversity is patchy for the country as a whole (Irwin *et al*. 2010) and entirely lacking for the globally-important spiny forest ecoregion (Gardner 2009). Moreover, existing studies in Madagascar have mirrored research from elsewhere (Burivalova *et al*. 2014) by quantifying assemblage-level change via species richness (e.g. Scott *et al*. 2006; Vallan 2002; Watson *et al*. 2004), a measure criticised because it can mask community turnover from specialist to generalist species (Barlow *et al*. 2007; Gardner *et al*. 2010). Here we investigate bird and reptile community responses to habitat change in a new protected area in the spiny forest ecoregion to ascertain the impacts of permitted and illegal livelihood activities (charcoal production and shifting cultivation respectively) on the conservation value of the vertebrate fauna. To overcome the issues associated with species richness as a metric, we use a Conservation Value Index (CVI; Chapter 5) to examine the influence of habitat degradation on the two taxonomic assemblages.

6.2 Methods

6.2.1 Study site

Madagascar's spiny desert (or spiny forest), is a global priority ecoregion (Olson and Dinerstein 1998) and Endemic Bird Area (Stattersfield *et al*. 1998) with extremely high rates of local floral endemism (Phillipson 1996). Between 1990 and 2010 it suffered the fastest rates of deforestation of any ecoregion in the country (Harper *et al*. 2007; ONE *et al*. 2013) and, prior to 2003, it was the least represented ecoregion within the country's protected area network (Fenn 2003; Seddon *et al*. 2000).

Ranobe PK32 is a new protected area that received temporary protected status within the Durban Vision framework in 2008, and is co-managed by local community associations, the regional Forest Service and the international NGO WWF (Virah-Sawmy *et al*. 2014). Lying north of the regional capital Toliara between the Fiherenana and Manombo rivers (Fig. 6.1), it is the richest landscape in the ecoregion in terms of its bird, reptile and lemur fauna (Gardner *et al*. 2009a,b; Chapter 5). However, the area is inhabited by approximately 90,000 people (WWF 2010), many of whom depend on natural resources from within and around the protected area for their subsistence and household income (Gardner and Davies 2014; Chapter 8). Ranobe PK32 is thus proposed as a category VI protected area in which subsistence and low-level commercial livelihood activities (such as timber cutting, fuelwood collection and charcoal production, grazing and the harvesting of NTFPs) are permitted in sustainable use zones which cover 86.5 % of the protected area's 148,554 ha (WWF 2010).

Charcoal is primarily produced in the western part of the protected area, due to the presence of the Route Nationale 9 (RN9) road that facilitates transportation. The industry is driven by the close proximity of Toliara, a city of approximately 200,000 people in which 98 % of households use wood or charcoal for cooking; demand from the city tripled between 2000 and 2007, and is largely met by anarchic charcoal production along the RN9 (Partage 2008). Since the region lacks fuelwood plantations, charcoal is produced entirely from natural forests (Bertrand *et al*. 2010). Charcoal producers select only hardwood trees (Mana *et al*. 2001; Rejo-Fienana 1995), thus causing forest degradation rather than outright deforestation (Casse *et al*. 2004).

We conducted our study in the vicinity of Ranobe, a complex of three villages with a total population of approximately 2000 people (Gardner and Davies 2014), where the surrounding forests had been subjected to both charcoal production and shifting cultivation within recent years. We selected three areas within 3 km of the main village

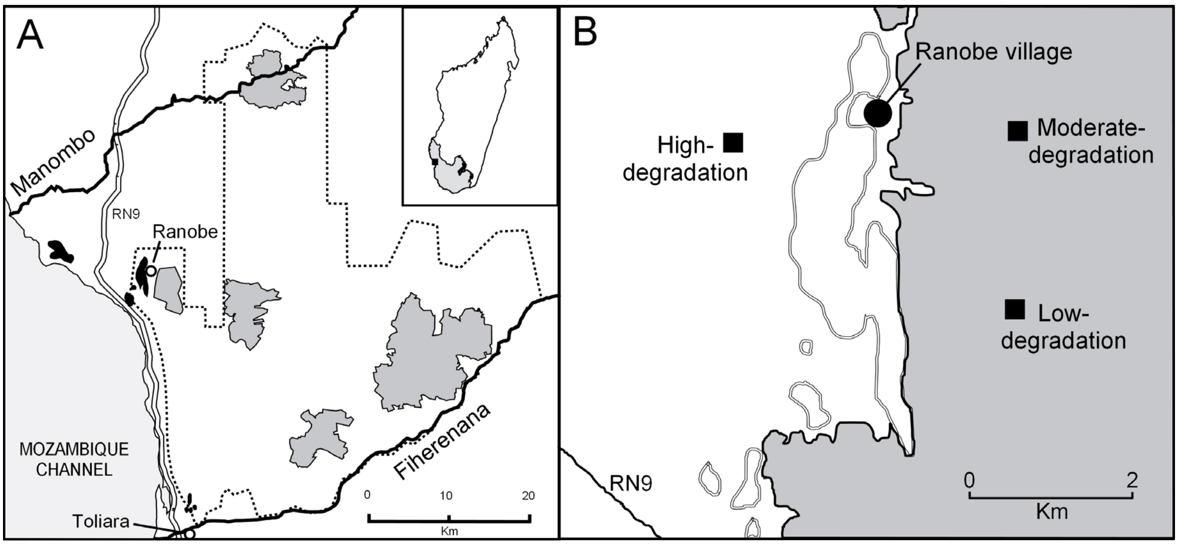

Figure 6.1 Map of: A) Ranobe PK32 protected area (dotted line) showing location of five strict conservation zones (grey shading), wetlands and rivers (black shading/ lines) and Ranobe village; and, B) location of three vegetation treatments used to survey bird and reptile communities across a gradient of degradation (forest cover, grey shading; wetlands, double line). Inset shows location of Ranobe PK32 within Madagascar (black square) and limits of spiny forest ecoregion following Goodman and Raherilalao (2013) (grey shading).

which, until recently, were part of a contiguous and relatively homogeneous forest block. Subsequently, the three areas have suffered varying levels of disturbance that are indicative of the habitat degradation gradient found across the whole landscape: i) a forest area showing minimal impacts of human activity (low-disturbance, hereafter *Low*); ii) a forest area subject to intensive charcoal production (moderate-disturbance, *Mod*); and, iii) an area regenerating following shifting cultivation (high-disturbance, *High*). While *Low* and *Mod* retained a complex three-dimensional structure and can be termed forest, *High* represented an open area dominated by shrubs, with only scattered trees (Fig. 6.1, Table 6.1). As there were no areas of forest near Ranobe that had not been subject to any human disturbance, it was not possible to include a control site representing intact habitat. Birds and reptiles were surveyed between January and March 2010 in the rainy season, when both groups are most active (Glaw and Vences 2007; Safford and Hawkins 2013).

Table 6.1 Disturbance history and vegetation description of three habitat treatments used to investigate the impacts of degradation on birds and reptiles at Ranobe, southwest Madagascar.

Treatment	Disturbance history	Habitat description
Low disturbance (*Low*)	Low level charcoal production since 2007	Relatively closed canopy dominated by *Didierea madagascariensis* and hardwood trees, with no understory shrub layer. Some charcoal production resulting in small openings, but canopy generally unbroken. Thick leaf litter layer.
Moderate disturbance (*Mod*)	Intensive charcoal production since 1995	Broken canopy dominated by *Didierea madagascariensis*, with hardwood trees largely absent. Small openings are frequent and possess a dense shrub layer of regenerating stumps. Characterised by piles of dead branches and bark left over from charcoal production. Thin leaf litter layer.
High disturbance (*High*)	Forest cleared for shifting cultivation in 2001, regenerating naturally since 2004/5	Dense shrub layer (height of 1-2 m) of regenerating stumps dominated by *Cedrelopsis grevei* and *Fernandoa madagascariensis*, with no litter layer. Relict individual trees and small forest patches (< 1ha) occur within a mosaic pattern.

6.2.2 Bird survey protocol

We established 48 census stations within each area and used the point count method (Bibby *et al*. 1998) to estimate bird relative abundance. Access to the forest interior was hindered by the impenetrable nature of the vegetation at *Low* and *Mod*, so census stations were placed on a stratified random grid along existing ox-cart tracks. We positioned all stations at a perpendicular distance of 75 m from a track (following Jones *et al*. 1995) to minimise the influence of edge effects, and at least 150 m apart to minimise the risk of double counting.

We surveyed each census station for 15 minutes (following a settling period of four minutes after arrival), during which we recorded all visual and auditory contacts within 50 m of the census station. To reduce time-of-day and weather-related effects, surveys were limited to between 06.00 and 08.00 and were not conducted on rainy or windy days. The majority of bird contacts in spiny forest (> 85 % at *Low* and *Mod*) were auditory due to the dense vegetation, thus making it difficult to generate reliable distance estimates for bird contacts and, as such, we did not employ distance sampling methods. However, the non-visual nature of most contacts reduces the likelihood of a detectability bias arising from surveying in forests of varying degradation levels (Bibby and Buckland 1987). As most records were auditory, we could not accurately count the number of individuals for social species, and thus we recorded the presence of groups not individuals. We did not include contacts with juvenile birds in our data analysis to reduce seasonality effects. Point count observations yielded both relative frequency (defined as the proportion of counts in which a given species was recorded) and relative abundance (mean number of contacts of a given species per count) data.

6.2.3 Reptile survey protocol

We calculated the relative abundance of reptiles based on capture in pitfall traps and area constrained refuge searches (transects), because observation and capture-based methods permit the sampling of different components of the reptile fauna (Raselimanana 2008; Raxworthy 1988). For pitfall trapping we followed a standard protocol widely used in Madagascar (D'Cruze *et al*. 2007; Raselimanana 2008). The traps consisted of plastic buckets (270 mm deep, 290 mm internal diameter at top, 220 mm internal diameter at base) placed 10 m apart and buried in the ground with the rim level with the surface. Drainage holes were drilled in the bottom of each bucket and the handles were removed. Buckets were connected by a drift fence 500 mm high, passing directly over the centre of each bucket, constructed from a sheet of plastic supported by wooden stakes. The lower 50 mm of the fence was buried in the soil and covered with leaf litter to prevent animals passing underneath. Within each treatment we established three trap lines (each of 10 or 11 buckets), placed randomly, but at least 150 m apart. Traps were constructed in the morning and left open for 13 nights, equating to 403 trap nights in total per treatment and were checked at 07.00 and 16.00 each day. All captured animals were marked on the hind leg or ventral surface with nail polish, and released at the site of capture. Recaptured individuals were excluded from the data analysis.

We also established 38 transects along which we conducted active refuge searches. Each transect consisted of a 50 m rope erected adjacent to forest tracks based on a stratified random grid. Each transect was at least 150 m apart, ran perpendicular to a track and started 10 m into the forest to reduce the influence of edge effects. We established each transect 24 hours prior to surveying to minimise disturbance effects. During surveys, two observers moved slowly along each transect and searched for reptiles within 2 m of the central line, scanning the trunks and branches of trees, searching within tree holes, under bark, in the leaf litter and under/within dead branches. All reptiles initially observed within 2 m of the central line were recorded. Transects were walked from 08.00-10.00 (n = 22/site) and 15.00-17.00 (n = 16/site); we did not survey during periods of rain or thick cloud cover to minimise variation in weather-related detectability, which reduced the number of appropriate afternoon survey periods. Juveniles were omitted from the dataset to minimise any bias that might be associated with the effects of the breeding season. Transects and pitfalls generated density and capture rate data, respectively: we pooled the data and used total contacts for further analyses (not including rarefaction).

6.2.4 Data analysis

In order to compare species richness between treatments and estimate the completeness of our sampling, we generated individual-based observed species richness rarefaction curves and associated 95 % confidence intervals using EstimateS v.9.0 (Colwell 2013). For reptiles, we combined the two datasets by assigning species to one or other method on the basis of substrate use, following a protocol adapted from Bicknell *et al*. (*In Review*), whenever a species was recorded by both methods. Thus all arboreal species were assigned to transects and all terrestrial and fossorial lizards were assigned to pitfall traps. Remaining terrestrial species (snakes and a tortoise) were assigned to the method by which they were most frequently recorded. We used chi-squared contingency tables to test for homogeneity of observed species relative frequency (birds) or total contacts (reptiles) across treatments.

Conservation Value Index

All species are not equal, and may differ in their value to conservationists on the basis of extinction risk, endemism, taxonomic distinctiveness or other attributes (Humphries *et al*. 1995). We therefore used a Conservation Value Index (CVI) adapted from Gardner *et al*. (Chapter 5) to quantify the conservation value of individual species. The CVI scores from individual species were combined to assess the impacts of natural resource use, and subsequent habitat degradation, on the conservation value of spiny forest habitats for birds and reptiles.

For the CVI we assigned scores to each individual species based on four attributes that reflect rarity, distinctiveness and threat. We use different combinations of attributes for the two taxonomic groups because the variation in conservation value within each group is driven by different factors. We scored rarity using geographical scale of endemism (*G*) and representation within SAPM (*R*), distinctiveness by taxonomic level of endemism (*E*), and threat on the basis of hunting and collection pressure (*C*) and degradation tolerance (*T*). We did not use *E* for reptiles due to the fact that all species are endemic and there are no endemic families, so variation in the attribute is limited. Similarly, we did not use *C* for birds because most species in the Ranobe area are subject to comparable hunting pressure (Gardner and Davies 2014).

Introduced species were removed from the dataset and scores assigned to indigenous taxa on a scale of 1-5 for each attribute (Table 6.2). For *G* we used different scoring systems for reptiles and birds because species distributions of the two taxonomic groups are best explained by different biogeographical models (Pearson and Raxworthy 2009). For birds we used distribution maps from Safford and Hawkins (2013) and followed Stattersfield *et al*. (1998) to classify microendemic species, whereas for reptiles we visually estimated range criteria using maps in Glaw and Vences (2007) and adopted 10,000 km^2 as the threshold for microendemic species (following Chapter 5). *E* was assigned on the basis of taxonomy in Safford and Hawkins (2013), *R* scores were assigned on the basis of occurrence in 14 (birds) or 15 (reptiles) protected areas in the dry regions of Madagascar derived from the literature (Table A2.1, Appendix 2), and values for *C* were based on occurrence in CITES (Convention on International Trade in Endangered Species)

Table 6.2 Scoring criteria for Conservation Value Index (CVI), used to quantify the conservation value of individual bird and reptile species at Ranobe, southwest Madagascar. EBA = Endemic Bird Area (Stattersfield *et al*. 1998), PA = protected area. Bioclimatic regions are defined by Cornet (1974).

Taxonomic group	Score	Geographic scale of endemism (*G*)	Taxonomic level of endemism (*E*)	Representation in sample PAs (*R*)	Hunting/collection pressure (*C*)	Degradation tolerance (*T*)
Birds	**1**	Indigenous, non-endemic species	Indigenous, non-endemic species	Recorded in 12-14 PAs (n > 85%)	N/A	Tolerant of modified or artificial habitats
	2	Endemic to western Indian Ocean	Endemic species	Recorded in 8-11 PAs (55 > n < 85%)	N/A	N/A
	3	Widespread Madagascar endemic	Endemic genus	Recorded in 4-7 PAs (30 > n < 50%)	N/A	Tolerant of edge effects, medium -intensity degradation or secondary growth.
	4	Endemic to dry regions of Madagascar	Endemic subfamily	Recorded in 2-3 PAs (10 > n < 20%)	N/A	N/A
	5	EBA species	Endemic family	Recorded in only 1 PA (n < 10%)	N/A	Intolerant of low-intensity degradation
Reptiles	**1**	Indigenous, non-endemic species	N/A	Recorded in 12-15 PAs (n > 75%)	No known threat	Tolerant of modified or artificial habitats
	2	Widespread endemic, occurring in dry and humid regions	N/A	Recorded in 8-11 PAs (45 > n < 75%)	N/A	N/A
	3	Endemic to dry regions	N/A	Recorded in 4-7 PAs (20 > n < 45%)	Known threat (CITES Appendix I and II), but not known to cause local extirpations	Tolerant of edge effects, medium -intensity degradation or secondary growth.
	4	Endemic to one bioclimatic region	N/A	Recorded in 2-3 PAs (10 > n < 20%)	N/A	N/A
	5	Local endemic, range size estimated as < 10,000 km^2	N/A	Recorded in only 1 PA (n < 10%)	Threat known to have caused local extirpations or severe population declines	Intolerant of low-intensity degradation

appendices and the literature on reptile declines in Madagascar. *T* was attributed following the methods outlined in Gardner *et al*. (Chapter 5) for reptiles, and were based on the literature (Safford and Hawkins 2013; Wilmé 1996) for birds. Species for which no degradation tolerance data were available were scored as intolerant on the basis of the precautionary principle.

The individual species CVI scores were calculated, producing a value in the range of 4-100, using the following formulae for reptiles and birds:

$CVI_{reptiles} = (G+R) \times (C+T)$

$CVI_{birds} = (G+E) \times (R+T)$

The conservation value of a site can be considered a function of: i) the value of the species occurring there; and, ii) their abundance, because an area with a large population of a valuable species is more important than one with a small population. To understand the relative conservation value of each habitat treatment, we therefore wanted a metric that combined the CVI of each species with their relative abundance. However, simply weighting the CVI score by the relative frequency would heavily bias common species at the expense of rarer ones which are recorded only infrequently. We thus gave each species weightings standardised to the treatment where it was most frequent (e.g., a species with relative frequency of 0.36, 0.18 and 0.12 across each of the three treatments would be given weightings of 1, 0.5 and 0.33 respectively). In each treatment the CVI was then multiplied by the weighting to produce a frequency-weighted CVI score for each species, before these were summed to produce a conservation value score for each treatment.

6.3 Results

6.3.1 Degradation impacts on birds

We recorded 2385 bird contacts, comprising 53 species, in point counts across all the treatments. Rarefaction curves approach an asymptote in all treatments, indicating that bird communities were sufficiently sampled (Fig. A2.1, Appendix 2). Although observed richness was highest in the moderate-degradation treatment (*Low* – 36 spp.; *Mod* – 43 spp.; *High* – 37 spp.), rarefaction curves show no significant differences in richness since the 95 % confidence intervals overlap (Fig. A2.1, Appendix 2). Total richness is estimated at 42.0 (*Low*), 46.8 (*Mod*) and 39.7 (*High*) species in the three treatments. Twenty-four species (45.3 %) were recorded in all treatments, one species (1.9 %) was restricted to *Low*, five species (9.4 %) were restricted to *Mod*, and seven (13.2 %) species were restricted to *High*: seventeen species (32.1 %) were recorded only in forest habitats (*Low* and *Mod*).

Observed patterns of species relative frequency differed significantly for 22 species (41.5 %) across the three treatments. Three of these species (*Cuculus rochii*, *Hypsipetes madagascariensis* and *Dicrurus forficatus*) were observed more frequently in the low-degradation treatment, one species (*Ploceus sakalava*) in the moderate-degradation treatment, and six species (*Turnix nigricollis*, *Oena capensis*, *Agapornis canus*, *Cisticola cherina*, *Acridotheres tristis* and *Foudia madagascariensis*) in the high-degradation treatment. A further 12 species were recorded less frequently in the high-degradation treatment than in forest habitat (*Low* or *Mod*) (Table A2.2).

Patterns of species endemism varied across the degradation gradient (Fig. 6.2). While the proportion of endemic species was approximately equal in all treatments, the high-degradation treatment contained a lower proportion of regionally-endemic birds (defined as restricted to Madagascar and the islands of the western Indian Ocean) and a higher

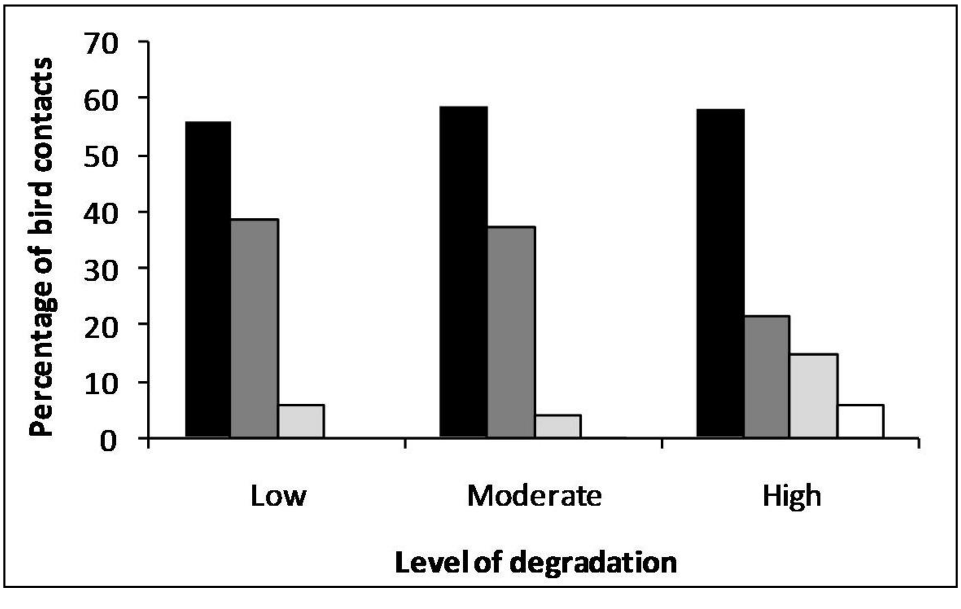

Figure 6.2 Endemism status of birds at Ranobe expressed as a percentage of contacts from 48 point counts at three sites across a gradient of degradation. Black, Madagascar endemic; dark-grey, regional endemic; light-grey, indigenous non-endemic; white, introduced. Regional endemic species are defined as restricted to Madagascar and the western Indian Ocean islands (Comoros, Mascarene and Seychelles archipelagos).

proportion of non-endemic species. The vast majority (97.9 %) of contacts with introduced species (*Acridotheres tristis*) occurred in the high-degradation treatment.

6.3.2 Degradation impacts on reptiles

We recorded 661 reptile contacts comprising 32 species, 27 of which were recorded at *Low* and *Mod*, and 15 species at *High*. Twenty-two species were observed during transects, and 27 were captured in pitfall traps (Table A2.3, Appendix 2). Twelve species (37.5 %) were recorded in all treatments, 17 species (53.1 %) were only recorded in forest habitats, and one species (*Lygodactylus tuberosus*) was recorded only in the high-disturbance site. Rarefaction curves indicate that *Low* and *Mod* had significantly higher species richness than *High*, as there is no overlap between confidence intervals (Fig. A2.2, Appendix 2). Total richness is estimated at 30.5 (*Low*), 34.2 (*Mod*) and 19.1 (*High)* species in the three treatments.

Observed patterns of reptile abundance, based on total contacts, were significantly heterogeneous for 11 species (34.4 %). Three species were recorded more frequently in the low-degradation treatment (*Chalarodon madagascariensis*, *Lygodactylus verticillatus* and *Oplurus cyclurus*), two species in the moderate-degradation treatment (*Madascincus* cf. *igneocaudatus* and *Tracheloptychus petersi*), and three species in the high-degradation treatment (*Lygodactylus tuberosus*, *Paroedura picta* and *Typhlops arenarius*). A further three species (*Geckolepis* c.f. *polypelis*, *Phelsuma mutabilis* and *Trachylepis elegans*) were recorded more frequently in the two forest areas than in the high-degradation treatment.

Forest disturbance affected distinct components of the reptile community differently, depending on their foraging substrate (Table A2.4, Appendix 2). Terrestrial species decreased in frequency (capture rate and/or density) with increasing disturbance, while arboreal species demonstrated reduced frequency at *Mod* and reduced richness at *High* compared to the less degraded site. Fossorial and litter dwelling species reached peak frequency under conditions of moderate-intensity disturbance.

6.3.3 Conservation value of species and habitat treatments

The CVI allowed us to weight species on the basis of their conservation value. The six highest scoring bird species were locally-endemic forest specialists (Table 6.3), while the highest scoring reptile was the heavily harvested (and thus Critically Endangered) tortoise

Table 6.3 Bird species recorded at Ranobe showing attributes used in Conservation Value Index (CVI) and frequency-weighted CVI scores for three sites across a gradient of degradation: Low, Mod and High indicate low-, moderate- and high-degradation treatments. CVI attributes: *G* – geographic scale of endemism, *E* – taxonomic level of endemism, *R* – representation in sample protected areas, *T* – degradation tolerance. Asterisks indicate species endemic to the spiny forest Endemic Bird Area (Stattersfield *et al*. 1998).

Species	**CVI attribute scores**				**CVI score**	**Frequency-weighted CVI**		
	G	**E**	**R**	**T**		**Low**	**Mod**	**High**
* *Monias benschi*	5	5	4	5	**90**	90	22.5	22.5
* *Xenopirostris xenopirostris*	5	5	3	5	**80**	0	11.4	80
* *Coua cursor*	5	4	3	5	**72**	72	20.6	30.9
* *Uratelornis chimaera*	5	5	4	3	**70**	0	0	0
* *Thamnornis chloropetoides*	5	5	2	5	**70**	70	60.0	0
* *Newtonia archboldi*	5	5	2	5	**70**	70	47.6	22.4
Coua ruficeps olivaceiceps	4	4	2	5	**56**	40.0	56	0
Calicalicus madagascariensis	3	5	2	5	**56**	56	56	0
Artamella viridis	3	5	2	5	**56**	32.0	56	0
Vanga curvirostris	3	5	1	5	**48**	48	29.2	4.2
Coua cristata	3	4	1	5	**42**	42	36.6	25.7
Falco zoniventris	3	2	3	5	**40**	0	40	0
Falculea palliata	4	5	1	3	**36**	36	36	0
Leptosomus discolor	2	5	2	3	**35**	0	0	0
* *Nesillas lantzii*	5	2	2	3	**35**	35	0	11.7
Newtonia brunneicauda	3	5	1	3	**32**	32	30.7	14
Leptopterus chabert	3	5	1	3	**32**	19.2	16	32
Aviceda madagascariensis	3	2	3	3	**30**	0	30	0
Neomixis striatigula	3	3	2	3	**30**	21.5	30	10.8
Cuculus rochii	3	2	2	3	**25**	25	11.7	5
Polyboroides radiatus	3	2	1	3	**20**	20	10	0
Buteo brachypterus	3	2	1	3	**20**	0	20	0
Mirafra hova	3	2	3	1	**20**	0	0	20
Copsychus albospecularis	3	2	1	3	**20**	20	19.3	14.3
Treron australis	2	1	2	3	**15**	0	0	15
Nectarinia notata	2	1	2	3	**15**	0	15	3.75
Ploceus sakalava	4	1	2	1	**15**	0.7	15	5.0
Accipiter francesiae	2	1	1	3	**12**	0	12	0
Turnix nigricollis	2	1	1	3	**12**	2.1	0	12

Species	**CVI attribute scores**				**CVI score**	**Frequency-weighted CVI**		
	G	**E**	**R**	**T**		**Low**	**Mod**	**High**
Nesoenas picturata	2	1	1	3	**12**	9.7	12	2.9
Coracopsis vasa	2	1	1	3	**12**	12	12	0
Coracopsis nigra	2	1	1	3	**12**	5.0	12	0
Phedina borbonica	2	1	3	1	**12**	0	12	12
Hirundo rustica	1	1	5	1	**12**	0	0	0
Hypsipetes madagascariensis	2	1	1	3	**12**	12	2.6	6.8
Terpsiphone mutata	2	1	1	3	**12**	9.7	12	4.6
Neomixis tenella	3	3	1	1	**12**	12	11.7	9.3
Cisticola cherina	2	1	3	1	**12**	0	0	12
Nectarinia souimanga	2	1	1	3	**12**	11.5	12	8.8
Dicrurus forficatus	2	1	1	3	**12**	12	9	9.5
Falco peregrinus	1	1	4	1	**10**	0	0	10
Agapornis canus	3	2	1	1	**10**	3.3	3.3	10
Tachymarptis melba	1	1	4	1	**10**	0	10	0
Eurystomus glaucurus	1	1	2	3	**10**	0	10	0
Upupa marginata	3	2	1	1	**10**	10	6.4	8.6
Falco newtoni	2	1	2	1	**9**	4.1	3.3	9
Caprimulgus madagascariensis	2	1	2	1	**9**	9	0	0
Falco concolor	1	1	3	1	**8**	0	0	8
Foudia madagascariensis	3	1	1	1	**8**	0.2	0.8	8
Milvus migrans	1	1	2	1	**6**	0	0	0
Oena capensis	1	1	2	1	**6**	3.4	1.4	6
Centropus toulou	2	1	1	1	**6**	4.3	4.3	6
Apus barbatus	1	1	2	1	**6**	3	6	0
Merops superciliosus	1	1	2	1	**6**	3.7	3.3	6
Corvus albus	1	1	2	1	**6**	0	0	6
Numida meleagris	1	1	2	1	**6**	0	0	6
Total conservation value of treatment						**856.4**	**825.7**	**478.6**

Pyxis arachnoides (Table 6.4). The relative conservation value of each treatment varied for the two taxonomic groups. Total bird conservation value was highest in *Low*, while total reptile conservation value was highest in *Mod*, although in both cases the differences between the two forest areas were small (Table 6.5). The high-degradation treatment had the lowest conservation value for both taxa.

6.4. Discussion

We have provided the first data on responses of spiny forest fauna to forest degradation, since hitherto the only available information concerned community change following deforestation (Scott *et al*. 2006). We found that while forest clearance greatly altered species composition and reduced species richness in both reptiles and birds, moderate-level forest degradation resulting from charcoal production provoked idiosyncratic responses that varied between groups. Reptile communities were little affected by degradation and experienced minimal community turnover between the low- and moderate-disturbance treatments, as 23 of the 27 species recorded in each area occurred in both. Conservation value of reptiles was in fact greatest at the moderate-disturbance site, perhaps reflecting an increase in microhabitat heterogeneity or structural complexity (MacArthur and MacArthur 1961; Tews *et al*. 2004). Bird communities were more responsive to habitat degradation, undergoing community turnover. This was reflected in the greater prevalence of birds adapted to open areas, and a decrease in the frequency of certain high-value, locally-endemic species such as *Monias benschi*, *Coua cursor* and *Newtonia archboldi*, with increasing degradation intensity.

Wilmé (1996) suggests that "the tolerance of [Madagascar's] endemic forest avifauna to forest degradation is proportional to its degree of taxonomic endemism". However, we recorded seven members of endemic genera (*Coua cursor*, *Monias benschi*, *Neomixis striatigula*, *Newtonia brunneicauda*, *N. archboldi*, *Vanga curvirostris* and *Xenopirostris xenopirostris*) previously thought to occur only in undisturbed or slightly disturbed habitats, within a largely deforested habitat in our high-disturbance treatment. These findings lend some support to the hypothesis that faunal species of Madagascar's dry and spiny forests may be more tolerant of degradation than those same or congeneric species in the country's humid east and north (Gardner 2009). This may arise due to the more 'gentle' habitat modifications occurring in dry forests compared to rainforests (Irwin *et al*. 2010): for example, the increased light penetration in forest gaps is thought to make little difference to the understory in the spiny forest, because the sparse, deciduous nature of the canopy already allows illumination at ground level (Seddon and Tobias 2007). However, while tropical dry forests are thought to be more resilient than humid forests in terms of regeneration capacity (Lebrija-Trejos *et al*. 2008), little is known about the relative disturbance sensitivity of their respective faunas. Such research should be considered a priority since it has important repercussions for the implementation of multiple-use protected areas in different bioclimatic contexts.

Table 6.4 Reptile species recorded at Ranobe showing attributes used in Conservation Value Index (CVI) score and relative frequency-weighted CVI scores for three sites across a gradient of degradation: Low, Mod and High indicate low-, moderate- and high-degradation treatments. CVI attributes: *G* – geographic scale of endemism, *R* – representation in sample protected areas, *C* – collection/hunting threat, *T* – degradation tolerance. Locally-endemic species are indicated by an asterisk.

Species	**CVI attribute scores**				**CVI score**	**Frequency-weighted CVI**		
	G	**R**	**C**	**T**		**Low**	**Mod**	**High**
Pyxis arachnoides	4	3	5	5	**70**	70	0	0
* *Voeltzkowia petiti*	5	4	1	5	**54**	14.7	54	0
* *Tracheloptychus petersi*	5	4	1	5	**54**	22.1	54	2.5
Geckolepis polylepis	4	4	1	5	**48**	48	32.8	0
Paroedura androyensis	4	3	1	5	**42**	14	42	0
* *Pygomeles braconnieri*	5	4	1	3	**36**	14.4	36	0
Voeltzkowia rubrocauda	3	3	1	5	**36**	36	0	10.3
* *Zonosaurus quadrilineatus*	5	4	1	3	**36**	36	32	8
Ithycyphus oursi	3	3	1	5	**36**	0	36	0
* *Liophidium chabaudi*	5	4	1	3	**36**	36	28.8	21.6
Madascincus cf. *igneocaudatus*	3	2	1	5	**30**	12	30	0
Madagascarophis ocellatus	4	3	1	3	**28**	28	0	0
Blaesodactylus sakalava	3	1	1	5	**24**	24	16	0
Zonosaurus karsteni	3	3	1	3	**24**	24	24	0
Madagascarophis meridionalis	3	3	1	3	**24**	0	24	0
Trachylepis aureopunctata	3	2	1	3	**20**	6.7	20	3.3
Heteroliodon occipitalis	3	2	1	3	**20**	10	20	0
Leioheterodon geayi	3	2	1	3	**20**	20	0	0
Typhlops arenarius	3	2	1	3	**20**	0	6.2	20
Typhlops decorsei	3	2	1	3	**20**	0	20	0
Lygodactylus verticillatus	4	4	1	1	**16**	16	4	0
Phelsuma mutabilis	3	1	3	1	**16**	16	10.3	2.3
Amphiglossus ornaticeps	2	2	1	3	**16**	9.6	16	0
Oplurus cyclurus	2	2	1	3	**16**	16	6.5	0.73
Lygodactylus tuberosus	4	3	1	1	**14**	0	0	14
Paroedura picta	3	2	1	1	**10**	3.8	1.9	10
Furcifer verrucosus	3	2	1	1	**10**	10	2.9	0
Chalarodon madagascariensis	2	2	1	1	**8**	8	4	3.0

Species	CVI attribute scores				CVI score	Frequency-weighted CVI		
	G	R	C	T		Low	Mod	High
Trachylepis elegans	2	1	1	1	**6**	4.9	6	2.9
Dromicodryas bernieri	2	1	1	1	**6**	6	6	6
Mimophis mahfalensis	2	1	1	1	**6**	4.5	6	5.3
Hemidactylus mercatorius	1	1	1	1	**4**	4	3.4	2.9
Total conservation value of treatment						**514.7**	**542.8**	**112.7**

The finding that moderate levels of degradation provoked an increase in richness of birds, and maintained richness in reptiles, is consistent with Connell's (1978) 'intermediate disturbance hypothesis', and reflects a pattern widely reported from other tropical environments, at least for some guilds (Burivalova *et al*. 2014; Child *et al*. 2009; Gray *et al*. 2007; Martin and Blackburn 2010; Pons and Wendenberg 2005). However, all species are not equal, and the greater richness may often mask a turnover from range-restricted specialists to widespread generalists (Canaday 1997; Christian *et al*. 2009; Holbech 2005; Petit and Petit 2003; Scott *et al*. 2006). The latter are of less importance to conservationists precisely because they adapt well to anthropogenic disturbance and thus do not require conservation actions, such as protected areas, to maintain them (Harris and Pimm 2004; T. Gardner *et al*. 2009). The use of species richness as a measure of conservation value has been widely criticised for this reason (Barlow *et al*. 2007; DeClercke *et al*. 2010; Fermon *et al*. 2005; Norris *et al*. 2010), but remains persistent (e.g., studies reviewed by Burivalova *et al*. 2014). Our use of the CVI provides further evidence of the inadequacies of richness in prioritising between sites or habitats, as the use of richness would indicate that forests degraded by charcoal production are more valuable for bird conservation in the spiny forest than less degraded habitats. Of course, the CVI does not represent a definitive quantification of conservation value, but is a useful heuristic tool to help conservationists prioritise action to where it is most needed, i.e. high-value species.

Although the use of CVI provides novel insights into the impacts of habitat change on the conservation value of spiny forest bird and reptile assemblages, our results must be interpreted with caution. We carried out surveying during the rainy season when both groups are most active, and surveyed each site sequentially for logistical reasons. However, biases may have arisen due to changes in species detectability related to the

Table 6.5 Observed and estimated species richness and Conservation Value Index (CVI) score for birds and reptiles at three sites across a gradient of disturbance at Ranobe, southwest Madagascar.

	Low disturbance	**Moderate disturbance**	**High disturbance**
Observed bird richness	36	43	37
Estimated bird richness	42.0	46.8	39.7
Bird CVI	856.4	825.7	478.6
Observed reptile richness	27	27	15
Estimated reptile richness	30.5	34.2	19.1
Reptile CVI	514.7	542.8	112.7

advancing breeding season. In addition, the entry of new cohorts may have increased population size as surveying progressed. We minimised the latter problem by excluding all records of juveniles from the analysis, although it would have been preferable to repeat data collection over multiple years, or to survey each site simultaneously using multiple teams. Nonetheless, the latter approach has a number of drawbacks, including the extensive training needed to minimise the biases associated with potential differences in the bird detection abilities and/or identification skills of the various research assistants.

Although our observations appear to suggest that the majority of bird and reptile species in Ranobe are somewhat resilient to moderate or high levels of disturbance, the presence of a species does not necessarily equate to its viability. It should not be assumed that local populations in disturbed areas will persist in the long-term because there are likely to be time lags associated with the impacts arising from perturbation, meaning that the degraded habitats at Ranobe may be carrying an ‘extinction debt’ (Kuussaari *et al*. 2009; Tilman *et al*. 1994). This is particularly true given that the habitat modifications that are the focus of this study are relatively recent (range: 3-15 years across the treatments). In addition, the persistence of some species within degraded habitats may be the result of source-sink dynamics, with populations maintained only by immigration from nearby areas of higher quality habitat (Hylander and Ehrlén 2013; Pulliam 1988; Tilman *et al*. 1994). The degraded habitats at Ranobe may therefore experience future local extinctions, even without further modification, and we may have over-estimated the value of these areas for bird and reptile diversity (Barlow *et al*. 2007; Sekercioğlu *et al*. 2007). The scale of extinction debt can be influenced by habitat quantity, quality, or connectivity (Hylander and Ehrlén 2013). As such, when destructive activities such as charcoal production cannot be prevented within the 'sustainable use zones' of multiple-use protected areas, both the structural and functional connectivity between high-quality habitat patches should be maximised in order to maintain biodiversity and mitigate the negative impacts associated with resource exploitation.

The suggestion that Madagascar's new protected areas may suffer the continued erosion of biodiversity has important ramifications for the objectives and management of multiple -use sites worldwide. In a multi-taxon assessment across a continuum of protection levels in East Africa, Gardner *et al*. (2007) found that multiple-use protected areas provide significant and complementary conservation services to strictly-protected sites, maintaining species richness but conserving significantly different faunal communities to those occurring in national parks. Thus a spectrum of protected area categories may be appropriate to conserve the full complement of biodiversity in continental regions, if these possess a range of faunal assemblages adapted to a continuum of habitat types from dense forests to wooded savannahs and grasslands (Borghesio 2008; Gardner *et al*. 2007; Pons *et al*. 2003).

Madagascar, however, differs from continents in that the vast majority of the endemic biota is forest-dependent (Goodman and Benstead 2005), while non-forest areas typically contain floristically- and faunistically-impoverished assemblages characterised by non-endemic species of low conservation value (Irwin *et al*. 2010; Koechlin *et al*. 1974; Lowry II *et al.* 1997). In this context, multiple-use sites essentially conserve the same communities as strict protected areas, but may do so less successfully than the latter. Thus, while multiple-use categories may be the only politically, ethically and logistically feasible option for many of Madagascar's new generation of protected areas, given the socioeconomic importance to rural communities of remaining forest resources, it should not be assumed that they will be successful in maintaining the biodiversity they were established to conserve. Given that range restricted habitat specialists are disproportionately likely to go extinct in modified habitats (Posa and Sodhi 2006; Scales and Marsden 2008), and are of greatest conservation interest worldwide, careful attention must be paid to the choice of protected area models in different contexts; in regions where the majority of priority species are disturbance-sensitive, strict protected areas may be a more appropriate model if they can be managed effectively.

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References

Adams, W.M. and Hutton, J. 2007. People, parks and poverty: political ecology and biodiversity conservation. *Conservation and Society* 5: 147–183.

AGRECO 2012. *Analyse des Coûts et Sources de Financement du Système des Aires Protégées de Madagascar (Octobre 2010 – Janvier 2012)*. AGRECO, Antananarivo.

Barlow, J., Gardner, T.A., Araujo, I.S., Avila-Pires, T.C., Bonaldo, A.B., Costa, J.E., Esposito, M.C., Ferreira, L.V., Hawes, J., Hernandez, M.I.M., *et al*. 2007. Quantifying the biodiversity value of tropical primary, secondary, and plantation forests. *Proceedings of the National Academy of Sciences*, USA 104: 18555–18560.

Bertrand, A., Ramamonjisoa, B. and Montagne, P. 2010. Les filières péri-urbaines d'approvisionnement en bois énergie des grandes villes de Madagascar. In *ARINA, le Charbon de Bois à Madagascar: entre Demande Urbaine et Gestion Durable*, P. Montagne, S. Razafimahatratra, A. Rasamindisa and R. Crehay (Eds.), pp. 23–36. CITE, Antananarivo.

Bertzky, B., Corrigan, C., Kemsey, J., Kenney, S., Ravilious, C., Besançon, C. and Burgess, N. 2012. *Protected Planet Report 2012: Tracking Progress Towards Global Targets for Protected Areas*. IUCN, Gland and UNEP-WCMC, Cambridge.

Bibby, C.J. and Buckland, S.T. 1987. Bias of bird census results due to detectability varying with habitat. *Acta Oecologica* 8: 103–112.

Bibby, C., Jones, M. and Marsden, S. 1998. *Expedition Field Techniques: Bird Surveys*. Royal Geographical Society, London.

Bicknell, J.E., Struebig, M.J. and Davies, Z.G. *In Review*. Reconciling timber extraction with biodiversity conservation in tropical forests using Reduced-Impact Logging.

Borghesio, L. 2008. Effects of human subsistence activities on forest birds in northern Kenya. *Conservation Biology* 22: 384–394.

Brooks, T.M., Mittermeier, R.A., da Fonseca, G.A.B., Gerlach, J., Hoffmann, M., Lamoreux, J.F., Mittermeier, C.G., Pilgrim, J.D. and Rodrigues, A.S.L. 2006. Global biodiversity conservation priorities. *Science* 313: 58–61.

Brown, K.A., Johnson, S.E., Parks, K.E., Holmes, S.M., Ivoandry, T., Abram, N.K., Delmore, K.E., Ludovic, R., Andriamaharoa, H.E., Wyman, T. M., *et al*. 2013. Use of provisioning ecosystem services drives loss of functional traits across land use intensification gradients in tropical forests in Madagascar. *Biological Conservation* 161: 118–127.

Burivalova, Z., Şekercioğlu, C.H. and Koh, L.P. 2014. Thresholds of logging intensity to maintain tropical forest biodiversity. *Current Biology* 24: 1893–1898.

Cabeza, M. 2013. Knowledge gaps in protected area effectiveness. *Animal Conservation* 16: 381–382.

Canaday, C. 1997. Loss of insectivorous birds along a gradient of human impact in Amazonia. *Biological Conservation* 77: 63–77.

Casse, T., Milhøj, A., Ranaivoson, S. and Randriamanarivo, J.R. 2004. Causes of deforestation in southwestern Madagascar: what do we know? *Forest Policy and Economics* 6: 33–48.

CBD (Convention on Biological Diversity), 2010. *Decision Adopted by the Conference of the Parties to the Convention on Biological Diversity at Its Tenth Meeting [Decision X/2] Nagoya, Aichi Prefecture, Japan, 18–29 October 2010*. Secretariat to the Convention on Biological Diversity, Montreal.

Chape, S., Harrison, J., Spalding, M. and Lysenko, I. 2005. Measuring the extent and effectiveness of protected areas as an indicator for meeting global biodiversity targets. *Philosophical Transactions of the Royal Society* B 360: 443–455.

Chapin, F.S., Zavaleta, E.S., Eviner, V.T., Naylor, R.L., Vitousek, P.M., Reynolds, H.L., Hooper, D.U., Lavorel, S., Sala, O.E., Hobbie, S.E., *et al*. 2000. Consequences of changing biodiversity. *Nature* 405: 234–242.

Child, M.F., Cumming, G.S. and Amano, T. 2009. Assessing the broad-scale impact of agriculturally transformed and protected area landscapes on avian taxonomic and functional richness. *Biological Conservation* 142: 2593–2601.

Christian, K., Isabelle, L.V., Frédéric, J. and Vincent, D. 2009. More species, fewer specialists: 100 years of changes in community composition in an island biogeographical study. *Diversity and Distributions* 15: 641–648.

Colwell, R.K., 2013. *EstimateS: Statistical estimation of species richness and shared species from samples*. Version 9.0.

Commission SAPM 2006. *Système d'Aires Protégées Malagasy: Orientations Générales sur les Catégories et les Types de Gouvernance*. SAPM Commission, Antananarivo.

Connell, J.H. 1978. Diversity in tropical rainforests and coral reefs. *Science* 199: 1302–1310.

Cornet, A. 1974. *Essai de Cartographie Bioclimatique à Madagascar*. Editions ORSTOM, Paris.

Corson, C. 2014. Conservation politics in Madagascar: the expansion of protected areas. In *Conservation and Environmental Management in Madagascar*, I. Scales (Ed.), pp. 193–215. Routledge, London.

D'Cruze, N., Sable, J., Green, K., Dawson, J., Gardner, C., Robinson, J., Starkie, G., Vences, M. and Glaw, F. 2007 The first comprehensive survey of amphibians and reptiles at Montagne des Français, Madagascar. *Herpetological Conservation and Biology* 2: 87–99.

DeClerck, F.A.J., Chazdon, R., Holl, K.D., Milder, J.C., Finegan, B., Martinez-Salinas, A., Imbach, P., Canet, L. and Ramos, Z. 2010. Biodiversity conservation in human-modified landscapes of Mesoamerica: past, present and future. *Biological Conservation* 143: 2301–2313.

Dudley, N. (Ed.) 2008. *Guidelines for Applying Protected Area Management Categories*. IUCN, Gland.

Dudley, N., Parrish, J.D., Redford, K.H. and Stolton, S. 2010. The revised IUCN protected area management categories: the debate and ways forward. *Oryx* 44: 485–490.

Duffy, R. 2006. Non-governmental organizations and governance states: the impact of transnational environmental management networks in Madagascar. *Environmental Politics* 15: 731–749.

Ehrlich, P.R. and Pringle, R.M. 2008. Where does biodiversity go from here? A grim business-as-usual forecast and a hopeful portfolio of partial solutions. *Proceedings of the National Academy of Sciences, USA* 105: 11579–11586.

Fenn, M.D. 2003. The spiny forest ecoregion. In *The Natural History of Madagascar*, S.M. Goodman and J.P. Benstead (Eds.), pp. 1525–1529. University of Chicago Press, Chicago.

Fermon, H., Waltert, M., Vane-Wright, R.I. and Muhlenberg, M. 2005. Forest use and vertical stratification in fruit-feeding butterflies of Sulawesi, Indonesia: impacts for conservation. *Biodiversity and Conservation* 14: 333–350.

Ferraro, P.J., Hanauer, M.H., Miteva, D.A., Canavire-Bacarreza, G.J., Pattanayak, S.K. and Sims, K.R. 2013. More strictly protected areas are not necessarily more protective: evidence from Bolivia, Costa Rica, Indonesia and Thailand. *Environmental Research Letters* 8: 025011.

Gardner, C.J. 2009. A review of the impacts of anthropogenic habitat change on terrestrial biodiversity in Madagascar: Implications for the design and management of new protected areas. *Malagasy Nature* 2: 2–29.

Gardner, C.J. 2011. IUCN management categories fail to represent new, multiple-use protected areas in Madagascar. *Oryx* 45: 336–346.

Gardner, C.J. and Davies, Z.G. 2014. Rural bushmeat consumption within multiple-use protected areas: qualitative evidence from southwest Madagascar. *Human Ecology* 42: 21–34.

Gardner, C. J., Ferguson, B., Rebara, F. and Ratsifandrihamanana, A. N. 2008. Integrating traditional values and management regimes into Madagascar's expanded protected area system: the case of Ankodida. In *Protected Landscapes and Cultural and Spiritual Values*, J-M. Mallarach (Ed.), pp. 92–103. IUCN, GTZ and Obra Social de Caixa Catalunya. Kasparek Verlag, Heidelberg.

Gardner, C.J., Kidney, D. and Thomas, H. 2009a. First comprehensive avifaunal survey of PK32-Ranobe, a new protected area in south-western Madagascar. *Phelsuma* 17: 20–39.

Gardner, C.J., Fanning, E., Thomas, H. and Kidney, D. 2009b. The lemur diversity of the Fiherenana-Manombo Complex, southwest Madagascar. *Madagascar Conservation & Development* 4: 38–43.

Gardner, T.A., Caro, T., Fitzherbert, E.B., Banda, T. and Lalbhai, P. 2007. Conservation value of multiple-use areas in East Africa. *Conservation Biology* 21: 1516–1525.

Gardner, T.A., Barlow, J., Chazdon, R., Ewers, R.M., Harvey, C.A., Peres, C.A. and Sodhi, N.S. 2009. Prospects for tropical forest biodiversity in a human-modified world. *Ecology Letters* 12: 561–582.

Gardner, T.A., Barlow, J., Sodhi, N.S. and Peres, C.A. 2010. A multi-region assessment of tropical forest biodiversity in a human-modified world. *Biological Conservation* 143: 2293–2300.

Gaston, K.J., Jackson, S.F., Cantu-Salazar, L. and Cruz-Pinon, G. 2008. The ecological performance of protected areas. *Annual Review of Ecology, Evolution, and Systematics*: 39: 93–113.

Geldmann, J., Barnes, M., Coad, L., Craigie, I.D., Hockings, M. and Burgess, N. 2013. Effectiveness of terrestrial protected areas in reducing habitat loss and population declines. *Biological Conservation* 161: 230–238.

Glaw, F. and Vences, M. 2007. *A Field Guide to the Amphibians and Reptiles of Madagascar*, 3rd Edition. Vences and Glaw Verlags GbR, Köln.

Goodman, S.M. and Benstead, J.P. 2005. Updated estimates of biotic diversity and endemism for Madagascar. *Oryx* 39: 73–77.

Goodman, S.M. and Raherilalao, M.-J. 2013. *Atlas of Selected Land Vertebrates of Madagascar*. Association Vahatra, Antananarivo.

Gray, M.A., Baldauf, S.L., Mayhew, P.J. and Hill, J.K. 2007. The response of avian feeding guilds to tropical forest disturbance. *Conservation Biology* 21: 133–141.

Harper, G.J., Steininger, M.K., Tucker, C.J., Juhn, D. and Hawkins, F. 2007. Fifty years of deforestation and forest fragmentation in Madagascar. *Environmental Conservation* 34: 325–333.

Harris, G.M. and Pimm, S.L. 2004. Bird species' tolerance of secondary forest habitats and its effects on extinction. *Conservation Biology* 18: 1607–1616.

Hoekstra, J.M., Boucher, T.M., Ricketts, T.H. and Roberts, C. 2005. Confronting a biome crisis: global disparities of habitat loss and protection. *Ecology Letters* 8: 23–29.

Holbech, L.H. 2005. The implications of selective logging and forest fragmentation for the conservation of avian diversity in evergreen forests of south-west Ghana. *Bird Conservation International* 15: 27–52.

Humphries, C.J., Williams, P.H. and Vane-Wright, R.I. 1995. Measuring biodiversity value for conservation. *Annual Review of Ecology and Systematics*: 26: 93–111.

Hutton, J. and Leader-Williams, N. 2003. Sustainable use and incentive-driven conservation: realigning human and conservation interests. *Oryx* 37: 215–226.

Hylander, K. and Ehrlén, J. 2013. The mechanisms causing extinction debts. *Trends in Ecology and Evolution* 28: 341–346.

Irwin, M.T., Wright, P.C., Birkinshaw, C., Fisher, B., Gardner, C.J., Glos, J., Goodman, S.M., Loiselle, P., Rabeson, P., Raharison, J.-L., *et al*. 2010. Patterns of species change in anthropogenically disturbed habitats of Madagascar. *Biological Conservation* 142: 2351–2362.

Jenkins, C.N. and Joppa, L. 2009. Expansion of the global terrestrial protected area system. *Biological Conservation* 142: 2166–2174.

Jones, M.J., Linsley, M.D. and Marsden, S.J. 1995. Population sizes, status and habitat associations of the restricted-range bird species of Sumba, Indonesia. *Bird Conservation International* 5: 21–52.

Joppa, L.N. and Pfaff, A. 2011. Global protected area impacts. *Proceedings of the Royal Society* B 278: 1633–1638.

Koechlin, J., Guillaumet, J.-L. and Morat, P. 1974. *Flore et Végétation de Madagascar*. J. Cramer Verlag, Vaduz.

Kumar, R. and Shahabuddin, G. 2006. Consequences of rural biomass extraction for bird communities in an Indian tropical dry forest and the role of vegetation structure. *Conservation and Society* 4: 562–591.

Kuussaari, M., Bommarco, R., Heikkinen, R.K., Helm, A., Krauss, J., Lindborg, R., Öckinger, E., Pärtel, M., Pino, J., Rodà, F., *et al*. 2009. Extinction debt: a challenge for biodiversity conservation. *Trends in Ecology and Evolution* 24: 564–571.

Lebrija-Trejos, E., Bongers, F., Pérez-Garcia, E.A. and Meave, J.A. 2008. Successional change and resilience of a very dry tropical deciduous forest following shifting agriculture. *Biotropica* 40: 422–431.

Leroux, S.J., Krawchuk, M.A., Schmiegelow, F., Cumming, S.G., Lisgo, K., Anderson, L.G. and Petkova, M. 2010. Global protected areas and IUCN designations: do the categories match the conditions? *Biological Conservation* 143: 609–616.

Locke, H. and Dearden, P. 2005. Rethinking protected area categories and the new paradigm. *Environmental Conservation* 32: 1–10.

Lowry, P.P. II, Schatz, G.E. and Phillipson, P.B. 1997. The classification of natural and anthropogenic vegetation in Madagascar. In *Natural Change and Human Impact in Madagascar*, S.M. Goodman and B.D. Patterson (Eds.), pp. 93–123. Smithsonian Institution Press, Washington and London.

MacArthur, R.H. and MacArthur, J.W. 1961. On bird diversity. *Ecology* 42: 594–598.

Mana, P., Rajaonarivelo, S. and Milleville, P. 2001. Production de charbon de bois dans deux situations forestières de la région de Tuléar. In *Sociétés Paysannes, Transitions Agraires et Dynamiques Ecologiques dans le Sud-ouest de Madagascar*, S. Razanaka, M. Grouzis, P. Milleville, B. Moizo and C. Aubry (Eds.), pp. 199–210. CNRE-IRD, Antananarivo.

Margules, C.R. and Pressey, R.L. 2000. Systematic conservation planning. *Nature* 405: 243–253.

Martin, T.E. and Blackburn, G.A. 2010. Impacts of tropical forest disturbance upon avifauna on a small island with high endemism: implications for conservation. *Conservation and Society* 8: 127–139.

Mascia, M.B., Pailler, S., Krithivasan, R., Roshchanka, V., Burns, D., Mlotha, M.J., Murray, D.R. and Peng, N. 2014. Protected area downgrading, downsizing and degazettement (PADDD) in Africa, Asia and Latin America and the Caribbean, 1900–2010. *Biological Conservation* 169: 355–361.

McConnell, W.J. and Kull, C.A. 2014. Deforestation in Madagascar: debates over the island's forest cover and challenges of measuring forest change. In *Conservation and Environmental Management in Madagascar*, I.R. Scales (Ed.), pp. 65–104. Routledge, Abingdon.

McDonald, R.I. and Boucher, T.M. 2011. Global development and the future of the protected area strategy. *Biological Conservation* 144: 383–392.

Miller, D.C. 2014. Explaining global patterns of international aid for linked biodiversity conservation and development. *World Development* 59: 341–359.

Myers, N., Mittermeier, R.A., Mittermeier, C.G., da Fonseca, G.A.B. and Kent, J. 2000. Biodiversity hotspots for conservation priorities. *Nature* 403: 853–858.

Nelson, A. and Chomitz, K.M. 2011. Effectiveness of strict vs. multiple use protected areas in reducing tropical forest fires: a global analysis using matching methods. *PLoS ONE* 6: e22722.

Norris, K., Asase, A., Collen, B., Gockowski, J., Mason, J., Phalan, B. and Wade, A., 2010. Biodiversity in a forest-agriculture mosaic – the changing face of West African rainforests. *Biological Conservation* 143: 2341–2350.

Olson, D.M. and Dinerstein, E. 1998. The Global 200: a representation approach to conserving the earth's most biologically valuable ecoregions. *Conservation Biology* 12: 502–515.

ONE (Office National pour l'Environnement), DGF/MEF, Conservation International, FTM and Madagascar National Parks 2013. *Evolution de la Couverture de Forêts Naturelles à Madagascar 2005-2010*. Office National pour l'Environnement, Antananarivo.

Partage 2008. *Analyse de l'Offre et de la Demande en Energie Domestique au Niveau de la Commune de Toliara*. Partage, Antananarivo.

Pearson, R.G. and Raxworthy, C.J. 2009. The evolution of local endemism in Madagascar: watershed versus climatic gradient hypotheses evaluated by null biogeographic models. *Evolution* 63: 959–967.

Peres, C.A., Barlow, J. and Laurance, W.F. 2006. Detecting anthropogenic disturbance in tropical forests. *Trends in Ecology and Evolution* 21: 227–229.

Petit, L.J. and Petit, D.R. 2003. Evaluating the importance of human-modified lands for Neotropical bird conservation. *Conservation Biology* 17: 687–694.

Phillipson, P.B., 1996. Endemism and non-endemism in the flora of south-west Madagascar. In *Biogéographie de Madagascar*, W.R. Lourenço (Ed.), pp. 125–136. Editions ORSTOM, Paris.

Pons, P. and Wendenbeurg, C. 2005. The impact of fire and forest conversion into savanna on the bird communities of West Madagascan dry forests. *Animal Conservation* 8: 183–193.

Pons, P., Lambert, B., Rigolot, E. and Prodon, R. 2003. The effects of grassland management using fire on habitat occupancy and conservation of birds in a mosaic landscape. *Biodiversity and Conservation* 12: 1843–1860.

Posa, M.R.C. and Sodhi, N.S. 2006. Effects of anthropogenic land use on forest birds and butterflies in Subic Bay, Philippines. *Biological Conservation* 129: 256–270.

Pulliam, H.R. 1988. Sources, sinks, and population regulation. *The American Naturalist* 132: 652–661.

Randrianandianina, B.N., Andriamahaly, L.R., Harisoa, F.M. and Nicoll, M.E. 2003. The role of protected areas in the management of the island's biodiversity. In *The Natural History of Madagascar*, S.M. Goodman and J.P. Benstead (Eds.), pp. 1423–1432. The University of Chicago Press, Chicago.

Raselimanana, A.P. 2008. Herpétofaune des forêts sèches malgaches. In *Les Forêts Sèches de Madagascar*, S.M. Goodman and L. Wilmé (Eds.). *Malagasy Nature* 1: 46–75.

Raxworthy, C.J. 1988. Reptiles, rainforest and conservation in Madagascar. *Biological Conservation* 43: 181 – 211.

Rejo-Fienena, F. 1995. *Etude Phytosociologique de la Végétation de la Région de Tuléar (Madagascar) et Gestion des Ressources Végétales par les Populations Locales (Cas du P.K. 32)*. PhD Thesis, Muséum National d'Histoire Naturelle, Paris.

Roe, D. 2008. The origins and evolution of the conservation-poverty debate: a review of key literature, events and policy processes. *Oryx* 42: 491–503.

Rosser, A.M. and Leader-Williams, N. 2010. Protection or use: a case of nuanced trade-offs? In *Trade-offs in Conservation: Deciding What to Save*, N. Leader-Williams, W.M. Adams and R.J. Smith (Eds.), pp 135–156. Wiley-Blackwell, Oxford.

Safford, R. and Hawkins, F. (Eds.) 2013. *Birds of Africa, vol. VIII: the Malagasy Region*. Christopher Helm, London.

Scales, B.R. and Marsden, S.J. 2008. Biodiversity in small-scale tropical agroforests: a review of species richness and abundance shifts and the factors influencing them. *Environmental Conservation* 35: 160–172.

Scharlemann, J.P.W., Kapos, V., Campbell, A., Lysenko, I., Burgess, N.D., Hansen, M.C., Gibbs, H.K., Dickson, B. and Miles, L. 2010. Securing tropical forest carbon: the contribution of protected areas to REDD. *Oryx* 44: 352–357.

Scott., D.M., Brown, D., Mahood, S., Denton, B., Silburn, A. and Rakotondraparany. 2006. The impacts of forest clearance on lizard, small mammal and bird communities in the arid spiny forest, southern Madagascar. *Biological Conservation* 127: 72–87.

Seddon, N. and Tobias, J.A. 2007. Population size and habitat associations of the long-tailed ground-roller *Uratelornis chimaera*. *Bird Conservation International* 17: 1–12.

Seddon, N., Tobias, J., Yount, J.W., Ramanampamonjy, J.R., Butchart, S. and Randrianizahana, H. 2000. Conservation issues and priorities in the Mikea Forest of south -west Madagascar. *Oryx* 34: 287–304.

Sekercioğlu, C.H., Loarie, S.R., Oviedo Brenes, F., Ehrlich, P.R. and Daily, G.C. 2007. Persistence of forest birds in the Costa Rican agricultural countryside. *Conservation Biology* 21: 482–494.

Stattersfield, A.J., Crosby, M.J., Long, A.J. and Wege, D.C. 1998. *Endemic Bird Areas of the World: Priorities for Biodiversity Conservation*. BirdLife International, Cambridge.

Tews, J., Brose, U., Grimm, V., Tielbörger, K., Wichmann, M.C., Schwager, M. and Jeltsch, F. 2004. Animal species diversity driven by habitat heterogeneity/diversity: the importance of keystone structures. *Journal of Biogeography* 31: 79–92.

Tilman, D., May, R.M., Lehman, C.L. and Nowak, M.A. 1994. Habitat destruction and the extinction debt. *Nature* 371: 65–66.

Vallan, D. 2002. Effects of anthropogenic environmental changes on amphibian diversity in the rain forests of eastern Madagascar. *Journal of Tropical Ecology* 18: 725–742.

Vedeld, P., Angelsen, A. Bojö, J., Sjaastad, E. and Berg, G.K. 2007. Forest environmental incomes and the rural poor. *Forest Policy and Economics* 9: 869–879.

Vira, B. and Kontoleon, A. 2010. Dependence of the poor on biodiversity: which poor, what biodiversity? In *Linking Biodiversity Conservation and Poverty Alleviation: a State of Knowledge Review*, Roe, D. (Ed.), pp. 13–28. Secretariat of the Convention on Biological Diversity, Montreal.

Virah-Sawmy, M., Gardner, C.J. and Ratsifandrihamanana, A.N. 2014. The Durban Vision in practice: experiences in participatory governance of Madagascar's new protected areas. In *Conservation and Environmental Management in Madagascar*, I. Scales (Ed.), pp. 216–252. Routledge, London.

Watson, J.E.M., Whittaker, R.J. and Dawson, T.P. 2004. Habitat structure and proximity to forest edge affect the abundance and distribution of forest-dependent birds in tropical coastal forests of southeastern Madagascar. *Biological Conservation* 120: 311–327.

Wilmé, L., 1996. Composition and characteristics of bird communities in Madagascar. In *Biogéographie de Madagascar*, W.R. Lourenço (Ed.), pp 349–362. Editions ORSTOM, Paris.

WWF. 2010. *Plan d'Aménagement et de Gestion de PK32-Ranobe*. WWF Madagascar and Western Indian Ocean Programme Office, Antananarivo.

Zimmerer, K.S., Galt, R.E. and Buck, M.V. 2004. Globalization and multi-spatial trends in the coverage of protected-area conservation (1980–2000). *Ambio* 33: 520–529.

Chapter 7

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Rural bushmeat consumption within multiple-use protected areas: qualitative evidence from southwest Madagascar

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Abstract

Ensuring the sustainability of bushmeat consumption is critical for both biodiversity conservation and poverty alleviation in tropical developing countries, yet we know little about the role of hunting and bushmeat consumption in the daily lives of rural communities. We provide the first detailed, qualitative examination of bushmeat hunting activities conducted by a rural community within one of Madagascar's new, multiple-use protected areas, in order to inform appropriate management strategies. Results suggest that most species are eaten, but that few are favoured above domestic meat. Hunting is generally a secondary pursuit, carried out opportunistically during the course of other activities, although its importance does increase in times of food stress. Management focused on increasing domestic meat availability and directing hunting effort away from sensitive species may improve the sustainability of hunting, but development interventions to reduce forest dependence may be required to promote conservation and poverty alleviation simultaneously.

7.1 Introduction

The hunting and consumption of bushmeat (meat derived from wild animals) is a growing concern for conservationists in tropical developing countries because it can have major impacts both on targeted species (Fa and Brown 2009) and ecosystem dynamics (Stoner *et al.* 2007). Bushmeat is also critical to the food security and income of millions of people (Milner-Gulland *et al.* 2003) and is particularly important for the rural poor (Brashares *et al.* 2011; de Merode *et al*. 2004) and, therefore, must be sustainably managed (Fa *et al.* 2003). With biodiversity conservation and poverty alleviation being two of society's greatest challenges in the 21st Century, it is essential to develop strategies to reduce or mitigate the effects of bushmeat hunting and consumption without these interventions impacting upon the people who rely on natural resources.

The management of bushmeat use should be evidence-based, but existing research is largely focused on the sustainability of harvesting and the factors influencing commercial bushmeat trade. While these studies have provided valuable insights into how demand could be reduced, investigations into the motivations and drivers of both hunter behaviour and rural consumption are under-represented in the literature (Kümpel *et al.* 2009; Pailler *et al.* 2009), hindering our ability to tackle the proximal causes of bushmeat hunting in and around protected areas. Even when rural hunters and bushmeat consumers are targeted, such research tends to take place in areas where hunting is driven by commerce (e.g. Kümpel *et al.* 2010; Pangau-Adam *et al.* 2012), so we know little about the factors influencing hunting and bushmeat consumption amongst communities who are likely to be depending on it for subsistence.

The gaps in our knowledge are particularly critical in Madagascar, one of the world's poorest nations and top conservation priorities (Brooks *et al.* 2006) where, since 2003, the government have been in the process of tripling the extent of the protected area system. The objectives of the expanded protected area network are to conserve the country's biodiversity and cultural heritage, while promoting the sustainable use of natural resources for poverty alleviation and development (Gardner 2011). As such, the new generation sites are mainly designated as IUCN category V and VI, in which sustainable resource extraction is permitted (Gardner 2011), and many are co-managed by local communities (Raik 2007). Meeting the dual objectives of conservation and poverty alleviation is a complex task for protected area managers as most resource use has negative impacts on the island's biodiversity (Gardner 2009; Irwin *et al.* 2010). This situation is further complicated because the need to rapidly establish these new protected areas ("an emergency conservation context"; Marie *et al*. 2009) has necessitated the implementation of conservation interventions with incomplete information on the social and ecological systems that will determine their success or otherwise (Gardner 2012). In addition, most of the new sites are being established in landscapes containing sizeable human populations that, to varying extents, depend on natural resources for subsistence and household income (Fritz-Vietta *et al*. 2011).

Bushmeat consumption has been historically under-recognised as a threat to animal populations in Madagascar (Goodman 2006), although recent surveys have revealed the extent of the practice and provided the first evidence regarding the rates at which various species are harvested for food (Golden 2009; Jenkins *et al.* 2011; Randrianandrianina *et al.* 2010; Razafimanahaka *et al.* 2012). Nevertheless, such quantitative analyses tell us little about the role of hunting and bushmeat consumption in the daily lives of the communities living within/adjacent to the country's new multiple-use protected areas, yet this information is vital to site managers if they are to develop successful interventions that will minimise the ecological effects of exploitation, without jeopardising the food security of local populations. The richness of qualitative data is equally important if we are to contextualise and understand why hunting is carried out and, thus, be able to plan appropriate evidence-based mitigation strategies. The collection of such data is, however, hampered by the sensitivity of bushmeat consumption as a topic, particularly within and around protected areas (Jenkins *et al.* 2011; Razafimanahaka *et al.* 2012), so qualitative studies on the motivations of hunters are rarely published Here, we present the results of a series of in-depth, semi-structured interviews with community members designed to overcome such methodological constraints and underpin the development of management strategies for a new, multiple-use protected area in southwest Madagascar. Specifically, we pose the questions: i) who does the hunting; ii) why do they hunt; iii) what species do they hunt, and; iv) how do they hunt them?

7.2 Methods

7.2.1 Study Site

Ranobe PK32 is a new protected area co-managed by WWF and local communities that received temporary protected status in 2008 (Fig. 7.1). It forms part of the spiny forest ecoregion (Fenn 2003), which suffered the highest rates of forest loss in the country between 2000 and 2005 (Harper *et al.* 2007), and was the least protected major forest

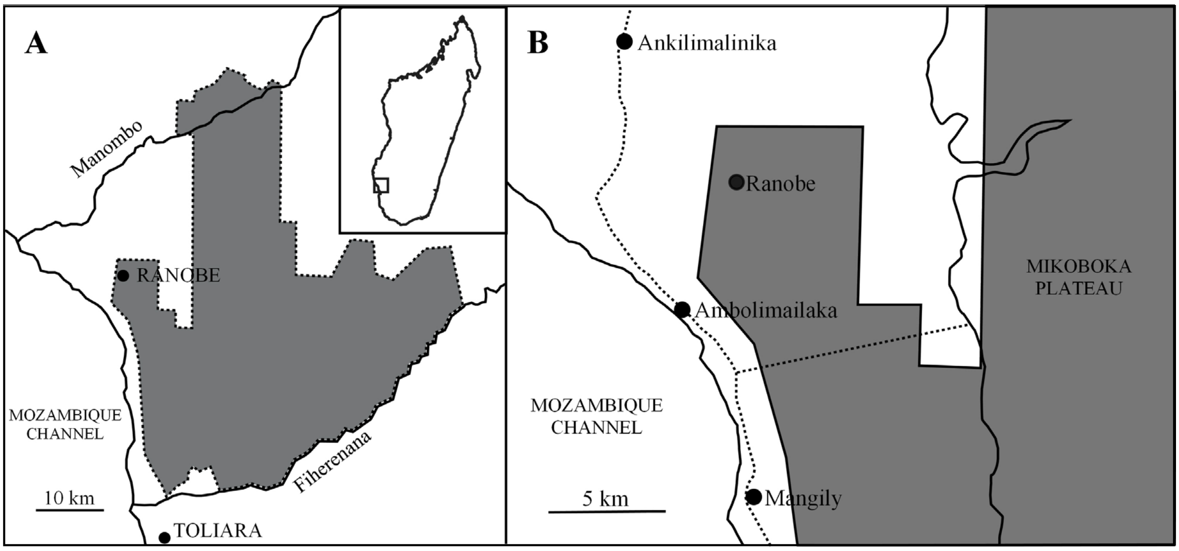

Figure 7.1 Map showing location of Ranobe village in relation to: A) Ranobe PK32 protected area (shaded grey), and; B) local villages and features mentioned in text (roads, dotted line; coastline and escarpment, solid line; Ranobe PK32 protected area, shaded grey). Inset shows location within Madagascar.

ecosystem in Madagascar prior to 2003 (Seddon *et al*. 2000). Ranobe PK32 was established primarily to conserve the habitat of two locally endemic bird species, the long -tailed ground roller (*Uratelornis chimaera*) and subdesert mesite (*Monias benschi*), both of which represent monospecific genera within endemic families that are restricted to southwest Madagascar (Raherilalao and Goodman 2011). Moreover, the protected area is recognised as the richest within the ecoregion, with more recorded species of bird, mammal, reptile and amphibian than any other site (Gardner *et al*. 2009a, b; Thomas *et al*. 2006). It contains five 'core zones' in which natural resource extraction is strictly controlled (20, 046 ha), with the remainder (128, 508 ha) zoned for sustainable use by local communities.

Ranobe is a complex of three villages with a total population of approximately 2000 people, lying 40 km north of the regional capital Toliara in southwest Madagascar (Atsimo-Andrefana region). Situated on the edge of a freshwater lake and marsh, the village is bordered to the east by forest and to the west by agricultural lands converted from forest over the last two decades. The people, mainly of the Masikoro ethnic group, are primarily cultivators and herders. Rice and sugar cane are grown around the lake, and corn, manioc and pulses on drier lands. These products are traded in the weekly markets at Ankilimalinika and Ambolimailaka (both approximately 5 km away), which are visited by most villagers. Cattle are kept principally for cultural reasons and to provide agricultural labour, while goats and chickens are reared for food and trade. Many people supplement their income through the commercial exploitation of forest wood for building materials or charcoal production (both of which are sold in Toliara), or through the trade of aquatic plants (*vondro* (*Typha angustifolia*) and *bararata* (*Phragmites mauritianum*)) from the marshes which are used for house construction. As well as medicinal plants and honey, forests and wetlands provide a range of goods used to supplement the diet including edible fruits and tubers (Dioscoreaceae; Cheban *et al*. 2009), fish and bushmeat. The forests of Ranobe are managed by the local community association, VOI Ezaka, under the terms of a GELOSE management transfer, through which limited rights are transferred from the State to the local community, according to a time-limited, renewable contract (Antona *et al*. 2004).

7.2.2 Data Collection

Bushmeat hunting and consumption can be a sensitive issue, particularly within protected areas, and the reticence of people to discuss their practices with outsiders can hinder the collection of viable or complete data (Jenkins *et al*. 2011; Razafimanahaka *et al*. 2012). This research was thus focussed on six key informants from Ranobe village with whom CJG (first author) had established a trusting, amicable relationship over the course of fieldwork carried out as part of a parallel research project. The informants had all worked closely with the author for at least two weeks prior to data collection, during which time local livelihoods, forest biodiversity and resource use, including hunting, were continuously discussed to provide a soft entry point to later interviews.

In-depth, semi-structured interviews were conducted by CJG during February and March 2010, and carried out in a mixture of French and Malagasy. The setting was informal, located at a camp at the edge of Ranobe village used for research. The interviews took place following communal mealtimes, with each one lasting two to six hours (mean = 4.3 hours; median = 4.5 hours), spread out over one to three sessions on consecutive days to avoid participant fatigue. Interviews were structured into two sections. Firstly, the participants were asked about their own livelihood practices, and hunting by villagers in general, in order to gather data on who hunts, when and why. Direct questions pertaining to the personal hunting practices of individual informants were avoided due to the sensitive nature of the topic. Secondly, they were asked to report which species of mammal, bird, reptile and amphibian were hunted and how they were captured, with the help of illustrated field guides (Garbutt 2007; Glaw and Vences 2007; Sinclair and

Langrand 1998); these books were familiar to all participants, having previously been used to converse about the fauna of the area. Informants were asked to leaf through the books to find species they recognise from Ranobe, and were only prompted for a response in cases where species known to occur in the area were overlooked. In such cases, the participant was requested to look at the picture again to confirm whether the species had genuinely never been encountered. When identification of an animal was unclear (e.g. if a species not known to occur in the region was noted), corroborating evidence was sought by asking the informant to describe its appearance, behaviour or autoecology. For every recognised species, it was enquired whether the animal was consumed and, if not, whether it was taboo (*faly*; see Jones *et al*. 2008) or simply not favoured for some reason. If eaten, participants were asked how widely it was hunted, who it was hunted by, when it was hunted, how it was prepared and eaten, and whether it was traded.

Although our sample size is small, the qualitative, in-depth nature of the research presented here supplements existing research on bushmeat hunting and consumption in Madagascar. While previous studies have generated information on the species targeted and, in some cases, the rates at which they are exploited (e.g. Garcia and Goodman 2003; Golden 2009; Goodman 2006; Goodman and Raselimanana 2003; Goodman *et al*. 2004, 2008; Jenkins *et al*. 2009, 2011; Rakotondravony 2006; Randriamanalina *et al*. 2000; Randrianandrianina *et al*. 2010), we provide novel information regarding who hunts, how, and why they do so, in addition to a detailed description of the range of species hunted and consumed. While a larger sample size of informants would have been desirable, this would have necessitated interviewing villagers with whom a trusting relationship had not already been established. We therefore decided *a priori* to restrict our interviews to participants we knew and from whom we could expect honest and unguarded responses, thus limiting the potential for bias.

7.3 Results

The participants ranged in age from 18 to 50 and all stated that farming constituted their primary livelihood activity; for two of them this was supplemented by trade in aquatic plants, while another cut construction wood in the forest to sell in the nearby tourist resort of Mangily (they were not asked whether they produced charcoal directly, as the practice is illegal within the Ranobe management transfer). Informants generally displayed a good knowledge of the fauna occurring in Ranobe lake and forest, and were able to name and describe all but the smallest and most cryptic species occurring in area. They only occasionally claimed to recognise species that are not known to occur in the region, and in no instance was a species known to be present in or around Ranobe unidentified by all six informants. Their knowledge of the forest, its fauna, and how to hunt it had been acquired since childhood, when going into the forest with their fathers, or from other boys while herding livestock.

7.3.1 Hunting activity in Ranobe

All informants agreed that not everybody in Ranobe hunts animals for food. Hunting is carried out almost exclusively by men, primarily those young in age or with livelihoods based around forest use. They stated that only a few individuals (no more than four) in Ranobe village hunt regularly as a means to generate an income (hereafter referred to as specialist hunters), and that these men all have little land and are therefore unable to derive a livelihood through farming. For the rest of the population, hunting is *ad hoc*, either being carried out opportunistically during the course of other activities (e.g. when cutting construction wood, searching for edible yams or medicinal plants, travelling to other areas), or as and when required. As one participant reported, encountering an edible animal "is like finding money on the ground when going to market – you just pick it up". The population of Ranobe principally consists of farmers, who have little time or reason to enter the forest regularly. However, men going into the forest will always take a catapult and a knife, and are thus well prepared to capture animals to supplement their diet. In the same way, people do not visit the lake and marshes specifically to hunt or catch birds, but go to fish or gather aquatic plants, and will collect any birds or eggs that they come across fortuitously. Egg collecting in the marshes was the single hunting-related activity reported to be carried out by women.

When asked about their preferences for meat, all informants stated that their favourites were beef and chicken, but that certain wild species were equally or almost as good. They frequently stated that all meat is the same, "meat is just meat, all meat is good", when questioned about the relative quality of meat from wild species. Nonetheless, this view was often contradicted as particular species were said to be very good or unpalatable. The fat content of wild meats appeared to be an important determinant of taste predilection, with several mammal species preferentially eaten at times of year when they are fattest (Table 7.1). All participants bought beef at the weekly markets in Ankilimalinika and Ambolimailaka whenever they had money (pork and goat are more rarely bought, while chicken is produced at home), and thus only ate domestic meat for one day a week.

Table 7.1 Information on the hunting, consumption and cultural significance of mammals recognised by informants from Ranobe village, southwest Madagascar.

English name	Scientific name	Used as bushmeat?	Informants' comments
Bush pig	*Potamochoerus larvatus*	Yes	A favoured meat of all participants. Common in the area and considered a big problem for farmers because it can destroy a whole field of maize or manioc in a single night. Although wire snares can be laid in areas it frequents, it is normally hunted with dogs and spears. Some people in the village are specialist bush pig hunters and have trained dog teams; when fresh tracks are found, such individuals are called, the dogs track the animal via its scent for up to three hours, until it is located, tires and backs itself against a tree, whereupon the dogs surround it and continue to bite its legs until the hunter(s) arrives to spear it. Dogs are often killed, predominantly inexperienced ones. The animals can provide up to 15 kg of meat, meaning that it is never entirely consumed at home and portions of meat (~1 kg) are sold in the village and, occasionally, at market for 2000-3000 Ar (US$0.97-1.45).
Common tenrec	*Tenrec ecaudatus*	Yes	A favoured species due to its large size and fatty meat; consistently elicited the most excited reactions from informants. Pursued purposefully by all members of the community, even those who do not generally hunt. Sought during the rainy season, especially in March, because it hibernates underground from April-November. During this time, some people spend all day in the forest or fields specifically searching for common tenrecs, potentially catching 50-100 in a single day (including up to 30 young per female). Can be followed, with or without dogs, to its burrow/refuge by following the pig-like spoor it leaves in the sand when foraging, and is then dug out or speared. Mainly hunted for sale in the village, rather than for domestic consumption, and excess are salted and taken to the weekly markets. Adults are traded for 1000-2000 Ar (US$0.48-0.97) depending on size, and young 200 Ar (US$0.10). Large males can cost over 3000 Ar in the dry season (US$1.45), because it is both extremely fatty and very difficult to find during hibernation. All informants stated that the species is abundant in years with heavy rains, and that it had become extremely rare in the previous two years due to drought. Prices cited above are from 2008, because no common tenrec had been for available for sale since that time. One participant suggested that in periods of drought the species is bitten by a mite- or tick-like parasite and dies, thereby proposing this as the mechanism for their rarity in such years.
Lesser hedgehog	*Echinops telfairi* *Setifer setosus*	Yes	Informants could readily distinguish these species and describe differences in their behaviour; the lesser is more arboreal and abundant in forests, whereas the greater is said to be encountered only occasionally and occurs

English name	Scientific name	Used as bushmeat?	Informants' comments
tenrec, Greater hedgehog tenrec			primarily around the village. Both species hibernate during the dry season and are nocturnal, residing in holes in the ground or trees, and cut stumps and branches left over from charcoal production. Up to four individuals of the lesser hedgehog tenrec can be found hibernating in the same hole. Detected by the smell of their urine and often discovered haphazardly during firewood collection and charcoal production. Targeted hunting trips with dogs can result in three or four individuals being caught, but this figure can be as great as 20 If the hunter is lucky. Both species are primarily hunted during their hibernation in the dry season, because they are very fatty and the viscera are clean, allowing all except the skin and bones to be eaten. Rarely sold because several are required to feed a family. However, when an excess number have been acquired they can be traded in the village, where they are valued at 200 Ar (US$0.10). They are never taken to market. One informant reported that some villagers have a *faly* against eating hedgehog tenrecs during the rainy season, because it is thought to interfere with their *ody gasy* (a type of traditional charm) against being struck by lightning.
Madagascar straw-coloured fruit bat, Madagascar flying fox	*Eidolon dupreneanum* *Pteropus rufus*	Yes	Only three informants were able to differentiate between these species. Both come to the village and adjacent forests during the rainy season to feed on fruiting trees including *mandresy* (*Ficus* cf. *pumila*), *varo* (*Cordia* cf. *mairai*) and *sakoa* (*Poupartia caffra*). They are hunted while foraging at night, either by hitting them with long poles, throwing sticks or using catapults. They are also trapped by placing the spiny seeds of *farehitsy* (*Uncarina* spp.) in tree branches, which can pierce the wings of a bat and become entangled in their fur, and so prevent the individual from flying away. The animals are cooked directly on hot coals following evisceration, and are never sold because it is unusual to catch sufficient numbers.
Micro-chiroptera	Multiple genera	No	Participants did not distinguish between microchiropteran species. These bats are hunted by children as they emerge from their roost in the village school, and occasionally around the village by adults in the evening. In both cases the animals are swatted with *bararata* poles, but it is not considered to be worth the effort as only a few will be caught.
Grey mouse lemur, Grey-brown mouse lemur	*Microcebus murinus* *M. griseorufus*	Yes	Informants did not distinguish between these species. They are nocturnal and hunted during the day by searching for their refugia in tree holes, which are easily detected by smell and the characteristic black staining beneath the entrance. The whole tree may be cut down to extract a single animal if the hole is not easily accessible. Mouse lemurs are also often found in the hollow, dead branches of *sogno* (*Didierea madagascariensis*), and may construct nests of leaves in the canopy when no suitable tree holes are available. Usually hunted in the dry season because they are fatter during this period, particularly on the tail, and up to 30 individuals can be found roosting together. Males will also be taken if they are found during the rainy season, despite the fact that they are not fat, but pregnant

English name	Scientific name	Used as bushmeat?	Informants' comments
			females or those with infants are never killed. When encountered in the daytime when active, mouse lemurs are caught by placing the untreated latex of *famanta* (*Euphorbia* spp.) or *folotse* (*Folotsia grandifolia*) on a long pole and poking the individual, so the stick adheres to the fur of the animal. In the dry season, some people go to the forest especially to hunt them, and may return with over 50 in a day. In such circumstances, these supplies are eaten at home (one person can eat three in a meal), with the surplus sold in the village for 200 Ar (US$0.10).
Sportive lemur	*Lepilemur* cf. *hubbardorum*	Yes	forest degradation, rather than as a direct consequence of hunting and that it is still relatively common on the Mikoboka Plateau approximately 10 km east. The animal is thought to be intelligent and difficult to find because it places its latrines far from the nest. However, when found, it is hunted either by knocking it out of the tree with sticks (when roosting in the open), or via extraction from its roost hole by hand.
Fat-tailed dwarf lemur	*Cheirogaleus medius*	Yes	Identified by only two informants, it is thought to sleep in nests in the canopy and become fat in winter, particularly on the tail. It is hunted by poking the nest with a pole until the individual wakes, before catapulting it.
Ring-tailed lemur, Verreaux's sifaka	*Lemur catta* *Propithecus verreauxi*	Yes	Both diurnal lemur species were said to have formerly been present in the area, but had disappeared at around the same time at least ten, and perhaps twenty, years ago (although a single troop of Verreaux's sifaka had been seen in 2009, a fact known to four informants). It was uniformly stated that they thought forest degradation, as a result of charcoal production, was the probable reason for their disappearance, and that populations remained on the Mikoboka Plateau. The species were formerly hunted by groups of people with dogs, which would chase the lemurs through the forest until they became tired, whereupon they would be shot with catapults or sticks until the whole troop was captured.
Fosa	*Cryptoprocta ferox*	No	The meat of the fosa is avoided by most people because the animal resembles a dog, but is consumed by very few. However, it is occasionally persecuted because it comes to the village at night to take chickens and young lambs.
Small Indian civet	*Viverricula indica*	Yes	The civet is known as *telofory* ('three anuses') in recognition of its musk glands, and is said to have extremely smelly flatulence. Nonetheless, it is hunted with dogs and commonly eaten because the meat is very fatty.
Small Afrotheria and rodents	Multiple genera	No	Never eaten because they are considered to be "like rats" and were not eaten by the ancestors.

Consumption of domestic meat is highly seasonal because people only have money for a few months (April-August) after harvest at the end of the rainy season (Fig. 7.2).

The participants stated that they would only consider going into the forest, with hunting normally being a secondary purpose for their trip, once their funds had run low. That said, three informants stated that they, and villagers in general, would also go hunting specifically to find meat before major celebrations, such as Christmas, New Year and Independence Day (26 June). During the rainy season (November to March) people are too busy working their fields to seek out and capture bushmeat (Fig. 7.2). Apart from occasional forest users and specialists hunters, a third group of bushmeat hunters comprises boys and young men aged 10-18, who spend considerable amounts of time in and around forests when herding livestock, and often catapult, grill and eat birds *in situ* "to pass the time". None of our participants mentioned any ceremonial or ritual aspects to hunting or bushmeat consumption.

7.3.2 Bushmeat preparation, consumption and commerce

According to the participants, bushmeat is hunted for sustenance in the home, but the form of consumption depends on the quantity obtained. The majority of hunted items, including most birds and all mammals apart from the bush pig (*Potamochoerus larvatus*) and common tenrec (*Tenrec ecaudatus*), are very small and must be caught in bulk to make a family meal (Table 7.1 and 7.2). Most species are therefore normally eaten as snacks, cooked on an open fire and eaten alone or with boiled manioc; feathers and fur are

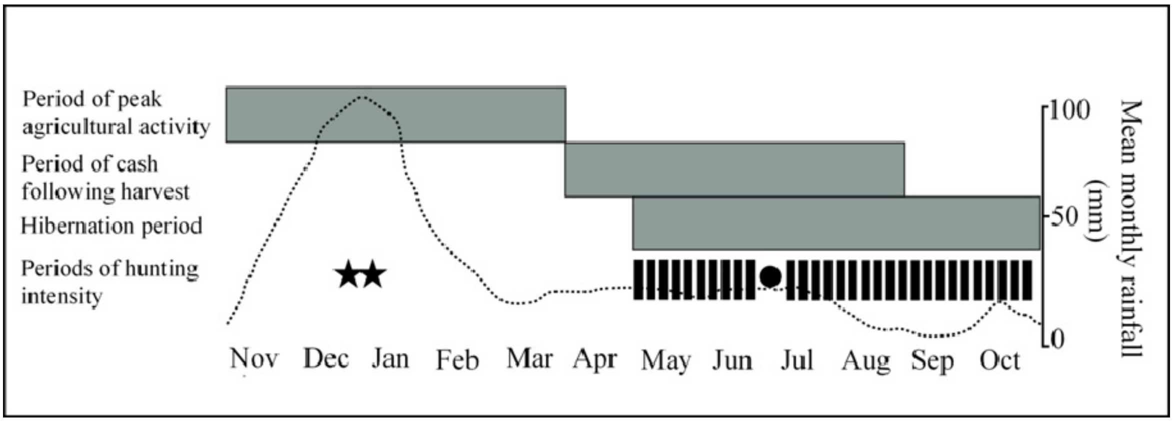

Figure 7.2 Seasonal calendar for Ranobe, southwest Madagascar, in relation to bushmeat hunting and factors which restrict or promote harvesting. Black dotted line, mean rainfall data are from Service de la Météorologie, Toliara, for 2005 to 2008. Black symbols indicate peaks of hunting intensity: stars, Christmas and New Year; circles, Independence Day; bars, hibernation.

burnt off in the fire (feathers are sometimes plucked), then the animal is eviscerated and rubbed inside and out with salt, and grilled directly on the hot coals. This may take place either in the forest where the animal is captured or back at the village. If sufficient animals are caught, however, they will be made into *laoka*, the dish that accompanies a carbohydrate staple. In this case the meat is cleaned and may be grilled, fried or cooked in a sauce, and accompanied by rice or manioc.

Only if large quantities of animals are caught is any excess sold, normally within the village, with prices ranging from 100-200 Ar (US$0.05-0.10) for most birds and small mammals through to 3000 Ar (US$1.45) for an adult helmeted guineafowl (*Numida meleagris*) or substantial piece (~1 kg) of bush pig meat (Table 7.1 and 7.2). With the exception of live birds that can be raised in captivity (white-faced duck (*Dendrocygna viduata*) and young helmeted guineafowl), bushmeat is rarely sold in the weekly markets of Ambolimailaka or Ankilimalinika. For the few specialist hunters, animals are caught specifically for sale within the village, and orders may be taken for particular species (mouse lemurs (*Microcebus* spp.) and lesser hedgehog tenrec (*Echinops telfairi*)). In such cases, if any meat cannot be sold, it will be grilled and traded in the Ambolimailaka or Ankilimalinika market.

7.3.3 Hunting of birds, mammals and reptiles

The majority of wetland and forest birds recognised by informants from Ranobe are hunted and eaten (Table 7.2). A variety of hunting techniques are employed, most of which are adapted to specific target species, although two main methods prevail: (i) trapping or snaring, and; (ii) shooting with catapults. Although some men in the village do possess home-made rifles, these are kept primarily for security and are rarely used for hunting because of the unavailability of ammunition. Bullets can occasionally be procured from foreigners in Mangily, in which case guns may be used for the hunting of wetland birds, such as flamingos (*Phoenicopterus ruber*) and large herons (*Ardea* spp.). Almost all men and boys own a catapult, which are made from rubber bought in Ambolimailaka market, with a pouch of leather salvaged from old bags or shoes. The geology of Ranobe means that there are no small rocks available, so ammunition consists of small balls of clay collected from the rice paddies and baked in the sun, unripe fruits of *lamonty* (*Flacourtia ramoutchi*), or the seeds of *sakoa*.

Table 7.2 Information on the hunting, consumption and cultural significance of bird species occurring around Ranobe village, southwest Madagascar, provided by informants during a series of semi-structured interviews.

English name	Scientific name	Used as bushmeat?	Informants' comments
Little grebe	*Tachybaptus ruficollis*	Yes	Difficult to shoot as dives when approached, but can be trapped at edge of reeds.
Small herons	Multiple genera	Yes	Trapped in reed beds, and eggs collected by women collecting *bararata*.
Large herons	*Ardea* spp.	Yes	Rarely eaten as difficult to trap, but can be shot with guns.
Greater flamingo	*Phoenicopterus ruber*	Yes	Occasionally occurs at Andranomanintsy lake, 5 km south of Ranobe, and is shot with guns. Meat is good.
Hammerkop	*Scopus umbretta*	No	One informant stated that species is not *faly* but people do not like to eat it, all others claimed it is *faly*.
White-faced duck; Red-billed teal; Knob-billed duck	*Dendrocygna viduata*; *Anas erythrorhyncha*; *Sarkidiornis melanotos*	Yes	Good meat. Occasionally trapped at edge of reeds, and nests and nestlings are collected by *bararata* collectors/fishermen if found. Are hunted by foreigners during the rainy season, who bring guns and plastic canoes. *D. viduata* is sometimes caught with fishing nets by Tanosy migrants and sold in Ambolimailaka for 3000 Ar.
Common moorhen; Purple gallinule	*Gallinula chloropus*; *Porphyrio porphyrio*	Yes	Trapped along small paths in reeds. Meat is good, but gallinule is very aggressive when caught. Red-knobbed coot (*Fulica cristata*) is not trapped as it remains in open water.
White-throated rail	*Dryolimnas cuvieri*	Yes	Trapped in reeds, and eggs are collected by *bararata* harvesters.
Madagascar crested ibis	*Lophotibis cristata*	Yes	Meat is better than chicken, and bird is larger. Does not like to fly and tires quickly, so can be chased and then catapulted or hit with throwing sticks. Alternatively it can be snared using a noose at the nest. Occasionally sold in the village for 2000 Ar. All informants stated that the bird used to occur near the village but is now rare, although it can still be found in pristine forest to the east. Disappearance is said to be due to degradation from charcoal production, rather than as a consequence of hunting.
Yellow-billed kite	*Milvus aegyptius*	No	Never eaten as it is *faly* because it eats dead things. Although it takes chickens, it is not persecuted.
Other raptors	Multiple genera	Yes	Although raptor meat is not as good as that of other birds, they are occasionally catapulted and eaten.

English name	Scientific name	Used as bushmeat?	Informants' comments
Helmeted guineafowl	*Numida meleagris*	Yes	As good as chicken. Drinks at the lake in the rainy season and is caught at the water's edge. Can be shot with guns, but is usually trapped using corn or scraps of *babo* (*Dioscorea* sp.), including at sites where *babo* has been dug. Excess animals are sold in the village for 2000-3000 Ar. Juveniles are occasionally sold in Ambolimailaka market for 200 Ar, where they are bought to be raised.
Madagascar partridge	*Margaroperdix Madagarensis*	Yes	Occurs in weedy fields, where it is trapped. Does not fly far, so can be chased until tired and then catapulted. If a nest is discovered and the eggs are not taken the finder's father is said to die; if the eggs are taken the mother dies.
Madagascar buttonquail	*Turnix nigricollis*	Yes	Highly favoured because meat is fatty and, unlike other birds, the bones can be crunched and eaten. It is often trapped by groups of 3-5 boys, by erecting a line of nooses and herding the flock is towards the traps. Alternatively, a small barrier of leaf litter can be made, which birds will not cross, and used to funnel the birds towards traps.
Subdesert mesite	*Monias benschi*	Yes	Widely eaten, although there is disagreement about the quality of the meat. Can be attracted by imitating its call or chased until it flies into a tree, and then catapulted. Can also be taken by hand on the nest.
Madagascar sandgrouse	*Pterocles personatus*	Yes	Shy and difficult to hunt, but is occasionally shot with guns (especially by foreigners) or trapped in foraging areas. Does not come to the lake for water. *Faly* for one informant.
Madagascar green pigeon	*Treron australis*	Yes	Good meat. It is hunted with catapults at fruiting *nonoky* (*Ficus* sp.) trees, where people will go specifically to hunt.
Madagascar turtle dove	*Streptopelia picturata*	Yes	Drinks at lake edge every morning, where traps are set. *Bararata* fences can be erected to herd birds towards traps. Can catch 5-20 in a morning and the excess are sold in the village for 200 Ar.
Namaqua dove	*Oena capensis*	Yes	Caught at lake edge in the mornings, particularly by children. Up to 20 traps are erected as species moves in flocks. Alternatively, the resin of *nonoky* is spread on branches which are 'planted' at the lake edge, catching individuals when they perch.
Vasa parrots	*Coracopsis* spp.	Yes	Easily catapulted as perch on high, exposed branches. Meat is not good but is eaten. Common crop pests of maize, but are not persecuted.
Grey-headed lovebird	*Agapornis cana*	Yes	Is trapped at lake edge where it drinks, and can be caught with *folotse* resin on branches.

English name	Scientific name	Used as bushmeat?	Informants' comments
Madagascar coucal	*Centropus toulou*	No	Not *faly*, but never eaten because meat is distasteful and is said to have a soporific effect.
Green-capped coua Running coua Giant coua	*Coua ruficeps olivaceiceps* *C. cursor* *C. gigas*	Yes	All terrestrial couas have good, fatty meat. They can be trapped by setting nooses along small forest paths, or chased on foot or with dogs until they fly into a tree, and then catapulted.
Crested coua	*Coua cristata*	Yes	Good meat, easily catapulted as perches in high branches.
Owls	Multiple genera	No	Strictly *faly* as they are "birds of the dead" and "sorcerer's birds".
Madagascar nightjar	*Caprimulgus madagascariensis*	Yes	Easily caught by hand or catapulted when roosting on forest floor during the day.
Madagascar kingfisher	*Alcedo vintsioides*	Yes	Nests in village wells, and occasionally catapulted on waterside perches.
Long-tailed ground roller	*Uratelornis chimaera*	Yes	Good meat. Difficult to find but easy to catch. Rarely flies and is stupid, so can be surrounded by people and catapulted. Can also follow tracks to the nest, which is often along a track. One participant stated that the birds have become rare due to hunting.
Madagascar cuckoo-roller	*Leptosomus discolor*	Yes	Large bird with good meat, easily catapulted.
Madagascar hoopoe	*Upupa marginata*	No	Meat is strong-smelling and distasteful, so rarely eaten.
Sakalava weaver	*Ploceus sakalava*	Yes	Nesting colonies in villages are never targeted, but otherwise is catapulted and eaten by children
Madagascar fody	*Foudia madagascariensis*	Yes	A major crop pest of rice. Can be trapped in rice fields in wire cages, where up to 50 can be caught at one time, or by placing *folotse* resin on branches.
All other passerines	Multiple genera	Yes	All small birds are catapulted and eaten by young boys.
Crested drongo	*Dicrurus forficatus*	No	Considered the king of the birds and is strictly *faly*.
Pied crow	*Corvus albus*	No	Not *faly* but never eaten, since it consumes dead animals including dogs.
Indian mynah	*Acridotheres tristis*	No	Strictly *faly*, although most informants could not give a reason why. One informant stated that it had been brought to the area by nuns. Local beliefs, originating from the Tandroy, state that anyone hunting the bird will receive the same injury that they give it. All informants agreed that bird had arrived within last 10-12 years.

Bird trapping is generally an activity for boys, who have the time to carry it out. All traps use a basic noose which tightens around the neck or foot of a bird, the twine of which can be made from *satra* palms (*Hyphaene coriacea*), the tail-hair of cattle, plastic strips taken from rice sacks, or sisal (*Agave* spp.) string purchased in the market. Nooses for marsh birds are set on paths through the reeds, and suspended from bent-over reed stems strengthened with sticks. For terrestrial birds, nooses can either be attached to a bent-over sapling or suspended from a frame made of branches. Traps are not placed randomly in the forest, but erected in locations where birds are known to frequent and/or tracks are plentiful. Species which drink at the periphery of the lake (grey-headed lovebird (*Agapornis cana*), Madagascar turtle dove (*Streptopelia picturata*), Namaqua dove (*Oena capensis*), and are caught at the water's edge by erecting *bararata* fences which guide birds towards the traps (Table 7.2). Other species are actively herded towards traps, particularly reluctant fliers such as the Madagascar buttonquail (*Turnix nigricollis*). This species walks only over clear ground, and individuals can therefore be diverted to the traps by building ridges of leaf litter. Finally, some species can be attracted to traps using bait; this applies particularly to couas (*Coua* spp.) and helmeted guineafowl, for which the skin of *babo* yams (*Dioscorea* spp.) is used, as well as maize for the latter of the two. The only form of bird trapping not to employ a noose is the use of natural plant glues (from *nonoky* (*Ficus* spp.), *famanta* and *folotse*) collected from the forest, which are spread on perches.

As is the case for birds, most mammal species known from Ranobe are eaten. This is in contrast to the situation for reptiles, which are all avoided with the exception of two terrestrial and one aquatic chelonian species; the spider tortoise (*Pyxis arachnoides*), radiated tortoise (*Astrochelys radiata*) and yellow-bellied mud turtle (*Pelusios castanoides*), whose heart is said to be poisonous. The radiated tortoise is no longer present near Ranobe, with informants agreeing that it had not been seen for 10-15 years, although it is said to still occur on the Mikoboka Plateau approximately 10 km to the east. All the participants believed that hunting was probably responsible for its disappearance. The spider tortoise is said to be common and is collected whenever it is seen, no matter what size, although it is *faly* for some people. A fourth chelonian species, the African helmeted turtle (*Pelomedusa subrufa*), is not eaten because it has an unpleasant, muddy flavour. No other reptiles or amphibians are eaten; they are said to be bad animals and were not eaten by the ancestors.

7.4 Discussion

The research presented here expanded our current knowledge of bushmeat hunting and consumption in Madagascar by using an approach that differed markedly from the majority of bushmeat-related studies carried out worldwide. Rather than generating quantitative data from large numbers of respondents, we wanted to produce rich, qualitative information via in-depth conversations with knowledgeable informants. Since this type of data collection requires established, trusted relationships as the subject matter is sensitive, as well a substantial time investment, the sample size is necessarily small. Nevertheless, the insights derived from such detailed semi-structured interviews are complementary to quantitative assessments and provide knowledge pertinent to understanding the role of bushmeat hunting in local livelihoods, thus supporting the development of appropriate management strategies. By asking people directly about all species known to occur in an area, we have demonstrated that the range of species eaten within a single community is much higher than suggested by previous quantitative studies of consumption. For example, in a household survey of eight rural settlements, Randrianandrianina *et al*. (2010) reported that only six mammal and five bird species were eaten, while our findings suggest that all but a handful of birds may be harvested for food. If our results are typical of other regions of Madagascar, birds are probably under-represented in questionnaire data focused on mealtime diets, as they are perceived as snacks that are eaten outside the home. Bird remains (e.g. bones, feathers) are also less likely to be found around camps and villages for the same reason yet, hitherto, this comprised the only data on the consumption of birds by rural communities (Goodman and Raselimanana 2003; Goodman *et al*. 2004).

In addition, our study has provided a deeper understanding of the role bushmeat hunting and consumption plays in the Ranobe community. This information can be used to inform the development of management interventions designed to reduce the ecological impacts of exploitation without negatively affecting the well-being of the user communities. We are now aware that hunting is a secondary activity for most people, carried out opportunistically to supplement the diet rather than for trade. This reflects the findings of other studies conducted in Madagascar, where bushmeat comprised only a small proportion of the diet (Jenkins *et al*. 2011; Rakotondravony 2006; Randrianandrianina *et al*. 2010). Bushmeat appears to play a role as a 'safety net' for the people of Ranobe; its importance increases during times of scarcity when there is no money to buy domestic meat, but diminishes during times of agricultural labour because hunting carries a high opportunity cost (Fig. 7.2). This has been noted elsewhere in Madagascar (Favre 1996; Golden *et al*. 2011; Goodman 2006; Goodman and Raselimanana 2003), but is not always the case as illustrated by Randriamanalina *et al.* (2000), who found that bushmeat hunting was a commercially-motivated activity in the southeast of the country. Additionally, we know that the majority of species are eaten regardless of size, but that few are particularly favoured and domestic meat is preferred (also see Jenkins *et al*. 2011; Randrianandrianina *et al*. 2010), as well as the full range of techniques used to procure bushmeat, most of which are highly selective.

The traditional management response to bushmeat hunting within protected areas, both in Madagascar and globally, is to impose an outright ban. However, such an interdiction would be inappropriate given the poverty alleviation objectives of Madagascar's expanded protected area system, and unethical given the importance of bushmeat to human nutrition in times of scarcity (Favre 1996; Golden *et al*. 2011; Goodman 2006). Such bans are also difficult and expensive to enforce (Leader-Williams and Albon 1988), and are unrealistic in heavily-populated landscapes such as Ranobe PK32 where human populations are widely dispersed and there are multiple entry points into the forest.

Furthermore, an outright ban on hunting may be unnecessary if hunting can be directed towards species capable of withstanding substantial off-take. Human consumption is believed to have contributed to the extinction of Madagascar's megafauna (Burney *et al*. 2004; Crowley 2010; Dewar 1997; Godfrey and Irwin 2007), and the successive defaunation of the largest-bodied, slowest-reproducing species has continued through to the present day. Indeed, many apparently intact forest areas are devoid of diurnal lemurs as a result of recent hunting pressure (Goodman and Raselimanana 2003; Goodman *et al*. 2004). These "empty forests" (Redford 1992) include Ranobe, where three diurnal lemurs (ring-tailed lemur *Lemur catta*, red-fronted brown lemur *Eulemur rufifrons* and Verreaux's sifaka *Propithecus verreauxi*) and the large-bodied radiated tortoise occurred until the 1990s (Domergue 1983; Gardner *et al*. 2009b; Nicoll and Langrand 1989; Raxworthy 1995) but no longer exist. If hunting has acted as an 'extinction filter' (Balmford 1996) contributing to the loss of the most sensitive species, it may be that the harvesting of the remaining small-bodied species represents an example of post-depletion sustainability (i.e. the species are capable of withstanding substantial off-take; Cowlishaw *et al*. 2005). Of the two most highly-prized animals mentioned in Ranobe (Table 7.1), the bush pig is introduced and the common tenrec has amongst the highest reproductive capacities of any mammal, suggesting that they could be harvested with minimal conservation impact. Beyond these, many regularly hunted species such as hedgehog tenrecs, mouse lemurs and most birds remain common in Ranobe's forests (C. Gardner, unpublished data), as they do in heavily exploited areas elsewhere in western Madagascar (Goodman and Raselimanana 2003). Although this suggests that current hunting practices in Ranobe may be sustainable for some species, quantitative population data and extractions rates would be required to verify whether this is the case (Golden 2009).

With the most sensitive species already extirpated, directing hunting effort towards those animals that can be hunted sustainably in the long-term should ensure that bushmeat can remain a food security safety net. Although environmental education and awareness-raising campaigns are often recommended and implemented as bushmeat reduction strategies (e.g. Breuer and Mavinga 2010), increasing knowledge in this way does not necessarily lead to modification of human behaviour (McKenzie-Mohr *et al*. 2012; Thompson 2008). Alternative approaches to alter consumption patterns include social marketing (defined as the use of commercial marketing techniques to achieve positive social change; Butler *et al*. 2007), which has proved effective in fostering sustained behaviour change (McKenzie-Mohr 2000; Schultz 2011) and, thus, has been suggested as a potential tool to decrease bushmeat demand (Drury 2009; Wilkie and Carpenter 1999). Encouraging more selective hunting in areas such as Ranobe should be achievable as it is targeted (i.e. using specifically designed traps and snares to catch particular species), rather than passive and generalised as is the case in eastern Madagascar (Golden 2009) and Africa more widely (Kümpel *et al*. 2009).

In recent years a number of authors have highlighted the potential importance of informal institutions such as sacred forests and taboos (*faly*) in the conservation of habitats and species in Madagascar (e.g. Bodin *et al*. 2006; Jones *et al*. 2008; Lingard *et al*. 2003; Loudon *et al*. 2006; Tengö *et al*. 2007; Vargas *et al*. 2002). Our findings suggest that few of the *faly* observed in Ranobe will serve this purpose, as they largely relate to widespread, non-endemic, unthreatened species (Tables 7.1 and 7.2). With the exception of the Madagascar spider tortoise (*Pyxis arachnoides*), none of the region's species of conservation concern (threatened or locally-endemic species) were reported to be subject to any cultural restrictions. Our informants distinguished between species that were *faly* and others that were simply not considered edible, with three bird species and a freshwater turtle deemed to be unpalatable. Similarly, all rodents, small tenrecs, amphibians and non-chelonian reptiles are not eaten because they are not regarded as food and were not eaten by the ancestors. This distinction between inedible and taboo species was raised by all participants, but does not appear to have been recorded in previous studies. For instance, Jones *et al*. (2008) list snakes, geckos and dwarf chameleons among the *faly* reported around Ranomafana National Park, but these reptiles were not recognised as food items in Ranobe.

Development activities aimed at promoting animal husbandry and increasing the availability of meat from farmed species are often advocated as a way to offset demand for protein derived from wild sources (Foerster *et al*. 2012; Mbete *et al*. 2011). This may appear to be a potential solution in areas such as Ranobe, where villagers prefer the taste of domestically reared meat and bushmeat has no particular cultural significance. However, cattle are kept primarily for cultural reasons, transport and agricultural labour, rather than meat production (Evers and van der Zwan 1998; Kaufmann 1998), so the promotion of livestock rearing will not inevitably result in the desired outcome. In addition, as noted by Bowen-Jones *et al*. (2003), the costs associated with such meat production may still render bushmeat economically favourable within rural communities.

Moreover, the impacts of livestock rearing on the environment and biodiversity in Madagascar have been little explored, and the relationship between pastoralism and the maintenance of forest cover is complex (Klein *et al*. 2008). The direct conversion of forest to pastureland is rare (Casse *et al*. 2004), with forests generally cleared for slash-and-burn agriculture and converted to pasture following abandonment. Livestock rearing may provide both incentives and disincentives for forest clearing. For example, Mahafale pastoralists in the southwest maintain forests which act as a dry season grazing reserve and place to hide cattle from rustlers (Kaufmann and Tsirahamba 2006; Tsirahamba and Kaufmann 2008), yet the primary cause of deforestation in western Madagascar is slash-and-burn agriculture (Scales 2011), carried out primarily by Tandroy migrants to generate cash with which to purchase cattle (Réau 2002). Another issue that further exacerbates the situation is that high stocking densities lead to forest degradation through a reduction in species richness and changes to vegetation structure (Ratovonamana and Kiefer 2009).

Our findings indicate that bushmeat hunting is an opportunistic activity, conducted whenever the forest is visited for other reasons. This corroborates observations from

Ankarafantsika National Park, where hunting is carried out by seasonal collectors of the palm *Raphia farinfera* (Garcia and Goodman 2003), and Mikea National Park, where people enter the forest to make outrigger canoes from the tree *Givotia madagascariensis* (Goodman *et al*. 2004). If this trend is typical across the forested regions of Madagascar, encouraging the exploitation of alternative meat sources is unlikely to serve as a 'silver bullet' to stem bushmeat hunting, because people entering forests to procure non-meat resources will continue to opportunistically collect any edible animals they encounter as it is "like finding money on the ground".

Most of Madagascar's new generation of IUCN category V and VI multiple-use protected areas are zoned to permit the sustainable extraction of forest, freshwater and/or marine resources (Gardner 2011). In terrestrial sites, activities such as livestock grazing and the collection of forest products (e. g. medicinal and edible plants, wild fibres and wood resources for fuel and construction) remain permissible. As these actions are associated with bushmeat hunting, it is likely that exploitation will persist throughout the country's new generation of protected areas. It therefore follows that the development of interventions designed to reduce rural dependence on forest resources will contribute to decreasing harvesting rates. Such strategies may also be more effective for poverty alleviation than a continuation of natural resource-based livelihoods (Sayer 2009), because the exploitation of forests generally serves only as a safety net for rural communities, rather than providing a route out of poverty (Wunder 2001; Angelsen and Wunder 2003). Consequently, the development of alternative livelihoods and improved revenue-generating activities that are not forest oriented may be essential if multiple-use protected areas are to achieve their twin goals of successful conservation and poverty alleviation (Gardner *et al*. 2013).

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References

Angelsen, A. and Wunder, S. 2003. *Exploring the Poverty-Forestry Link: Key Concepts, Issues and Research Implications*. Center for International Forestry Research, Bogor.

Antona, M., Biénabe, E.M., Salles, J.-M., Péchard, G., Aubert, S. and Ratsimbarison, R. 2004. Rights transfers in Madagascar biodiversity policies: achievements and significance. *Environment and Development Economics* 9: 825–847.

Balmford, A. 1996. Extinction filters and current resilience: the significance of past selection pressures for conservation biology. *Trends in Ecology and Evolution* 11: 193–196.

Bodin, O., Tengö, M., Norman, A., Lundberg, J. and Elmqvist, T. 2006. The value of small size: loss of forest patches and ecological thresholds in southern Madagascar. *Ecological Applications* 16: 440–451.

Bowen-Jones, E., Brown, D. and Robinson, E.J.Z. 2003. Economic commodity or environmental crisis? An interdisciplinary approach to analysing the bushmeat trade in Central and West Africa. *Area* 35: 390–402.

Brashares, J.S., Golden, C.D., Weinbaum, K.Z., Barrett, C.B. and Okello, G.V. 2011. Economic and geographic drivers of wildlife consumption in rural Africa. *Proceedings of the National Academy of Sciences, USA* 108: 13931–13936.

Breuer, T. and Mavinga, F.B. 2010. Education for the conservation of great apes and other wildlife in northern Congo – the importance of nature clubs. *American Journal of Primatology* 72: 454–461.

Brooks, T.M., Mittermeier, R.A., da Fonseca, G.A.B., Gerlach, J., Hoffmann, M., Lamoreux, J.F., Mittermeier, C.G., Pilgrim, J.D. and Rodrigues, A.S.L. 2006. Global biodiversity conservation priorities. *Science* 313: 58–61.

Burney, D.A., Burney, L.P., Godfrey, L.R., Jungers, W.L., Goodman, S.M., Wright, H.T. and Jull, A.J.T. 2004. A chronology for late prehistoric Madagascar. *Journal of Human Evolution* 47: 25–63.

Butler, B., Tyrell, M., Feinberg, G., VanManen, S., Wiseman, L. and Wallinger, S. 2007. Understanding and reaching family forest owners: lessons from social marketing research. *Journal of Forestry* 106: 348–357.

Casse, T., Milhøj, A., Ranaivoson, S. and Randriamanarivo, J.R. 2004. Causes of deforestation in southwestern Madagascar: what do we know? *Forest Policy and Economics* 6: 33–48.

Cheban, S.A., Rejo-Fienena, F. and Tostain, S. 2009. Etude ethnobotanique des ignames (*Dioscorea* spp.) dans la Forêt Mikea et le Couloir d'Antseva (sud-ouest de Madagascar). *Malagasy Nature* 2: 111–126.

Cowlishaw, G., Mendelson, S. and Rowcliffe, J.M. 2005. Evidence for post-depletion sustainability in a mature bushmeat market. *Journal of Applied Ecology* 42: 460–468.

Crowley, B.E. 2010. A refined chronology of prehistoric Madagascar and the demise of the megafauna. *Quaternary Science Reviews* 29: 2591–2603.

de Merode, E., Homewood, K. and Cowlishaw, G. 2004. The value of bushmeat and other wild foods to rural households living in extreme poverty in Democratic Republic of Congo. *Biological Conservation* 118: 573–581.

Dewar, R.E. 1997. Were people responsible for the extinction of Madagascar's subfossils, and how will we ever know? In *Natural Change and Human Impact in Madagascar*, S.M. Goodman and B.D. Patterson (Eds.), pp. 364–377. Smithsonian Institution Press, Washington DC.

Domergue, C.A. 1983. Note préliminaire en vue de la mise en réserve de la forêt du point kilomètre 32 au nord de Tuléar, Madagascar. *Bulletin de l'Académie Malgache* 61: 105–114.

Drury, R. 2009. Reducing urban demand for wild animals in Vietnam: examining the potential of wildlife farming as a conservation tool. *Conservation Letters* 2: 263–270.

Evers, S. and van der Zwan, N. 1998. *Madagascar: the Zebu as Guide through Past and Present*. Africa Museum, Berg en Dal.

Fa, J.E. and Brown, D. 2009. Impacts of hunting on mammals in African tropical moist forests: a review and synthesis. *Mammal Review* 39: 231–264.

Fa, J.E., Currie, D. and Meeuwig, J. 2003. Bushmeat and food security in the Congo Basin: linkages between wildlife and people's future. *Environmental Conservation* 30: 71 –78.

Favre, J.-C. 1996 Traditional utilization of the forest. In *Ecology and Economy of a Tropical Dry Forest in Madagascar*, J.U. Ganzhorn and J.P. Sorg (Eds.), pp. 33–40. *Primate Report* 46-1.

Fenn, M.D. 2003. The spiny forest ecoregion. In *The Natural History of Madagascar*, S.M. Goodman and J.P. Benstead (Eds.), pp. 1525–1529. University of Chicago Press, Chicago.

Foerster, S., Wilkie, D.S., Morelli, G.A., Demmer, J., Starkey, M., Telfer, P., Steil, M. and Lewbel, A. 2012. Correlates of bushmeat hunting among remote rural households in Gabon, Central Africa. *Conservation Biology* 26: 335–344.

Fritz-Vietta, N.V.M., Ferguson, H.B., Stoll-Kleemann, S. and Ganzhorn, J.U. 2011. Conservation in a biodiversity hotspot: insights from cultural and community perspectives in Madagascar. In *Biodiversity Hotspots: Distribution and Protection of Conservation Priority Areas*, F.E. Zachos and J.C. Habel (Eds.), pp. 209–233. Springer, Heidelberg and Berlin.

Garbutt, N. 2007. *Mammals of Madagascar: a Complete Guide*. A&C Black, London.

Garcia, G. and Goodman, S.M. 2003. Hunting of protected animals in the Parc National d'Ankarafantsika, north-western Madagascar. *Oryx* 37: 115–118.

Gardner, C.J. 2009. A review of the impacts of anthropogenic habitat change on terrestrial biodiversity in Madagascar: implications for the design and management of new protected areas. *Malagasy Nature* 2: 2–29.

Gardner, C.J. 2011. IUCN management categories fail to represent new, multiple-use protected areas in Madagascar. *Oryx* 45: 336–346.

Gardner, C.J. 2012. Social learning and the researcher-practitioner divide. *Oryx* 46: 313–314.

Gardner, C.J., Kidney, D. and Thomas, H. 2009a. First comprehensive avifaunal survey of PK32-Ranobe, a new protected area in south-western Madagascar. *Phelsuma* 17: 20–39.

Gardner, C.J., Fanning, E., Thomas, H. and Kidney, D. 2009b. The lemur diversity of the Fiherenana-Manombo Complex, southwest Madagascar. *Madagascar Conservation & Development* 4: 38–43.

Gardner, C.J., Nicoll, M.E., Mbohoahy, T., Olesen, K.L.L., Ratsifandrihamanana, A.N., Ratsirarson, J., René de Roland, L.-A., Virah-Sawmy, M., Zafindrasilivonona, B. and Davies, Z.G. 2013. Protected areas for conservation and poverty alleviation: experiences from Madagascar. *Journal of Applied Ecology* 50: 1289–1294.

Glaw, F. and Vences, M. 2007. *A Field Guide to the Amphibians and Reptiles of Madagascar.* 3rd Edition. Vences and Glaw Verlags GbR, Köln.

Godfrey, L.R. and Irwin, M.T. 2007. The evolution of extinction risk: past and present anthropogenic impacts on the primate communities of Madagascar. *Folia Primatologica* 78: 405–419.

Golden, C.D. 2009. Bushmeat hunting and use in the Makira Forest, north-eastern Madagascar: a conservation and livelihoods issue. *Oryx* 43: 386–392.

Golden, C.D., Fernald, L.C.H., Brashares, J.S., Rasolofoniaina, B.J.R. and Kremen, C. 2011. Benefits of wildlife consumption to child nutrition in a biodiversity hotspot. *Proceedings of the National Academy of Sciences, USA* 108: 19653–19656.

Goodman, S.M. 2006. Hunting of microchiroptera in south-western Madagascar. *Oryx* 40: 225–228.

Goodman, S.M. and Raselimanana, A.P. 2003. Hunting of wild animals by Sakalava of the Menabe region: a field report from Kirindy-Mite. *Lemur News* 8: 4–6.

Goodman, S.M., Soarimalala, V. and Ganzhorn, J.U. 2004. La chasse aux animaux sauvages dans la forêt de Mikea. In *Inventaire Floristique et Faunistique de la Forêt de Mikea: Paysage Ecologique et Diversité Biologique d'une Préoccupation Majeure pour la Conservation*, S.M. Goodman and A.P. Raselimanana (Eds.), pp. 95–100. *Recherches pour le Développement, Séries Sciences Biologiques* 21.

Goodman, S.M., Ratrimomanarivo, F.H., Ranivo, J. and Cardiff, S.G. 2008. The hunting of microchiropteran bats in different portions of Madagascar. *African Bat Conservation News* 16: 4–7.

Harper, G.J., Steininger, M.K., Tucker, C.J., Juhn, D. and Hawkins, F. 2007. Fifty years of deforestation and forest fragmentation in Madagascar. *Environmental Conservation* 34: 325–333.

Irwin, M.T., Wright, P.C., Birkinshaw, C., Fisher, B., Gardner, C.J., Glos, J., Goodman, S.M., Loiselle, P., Rabeson, P., Raharison, J.-L., *et al*. 2010. Patterns of species change in anthropogenically disturbed habitats of Madagascar. *Biological Conservation* 142: 2351–2362.

Jenkins, R.K.B., Rabearivelo, A., Chan, C.T., Andre, W.M., Randrianavelona, R. and Randrianantoandro, J.C. 2009. The harvest of endemic amphibians for food in eastern Madagascar. *Tropical Conservation Science* 2: 25–33.

Jenkins, R.K.B., Keane, A., Rakotoarivelo, A.R., Rakotomboavonjy, V., Randrianandrianina, F.H., Razafimanahaka, H.J., Ralaiarimalala, S.R. and Jones, J.P.G. 2011. Analysis of patterns of bushmeat consumption reveals extensive exploitation of protected species in eastern Madagascar. *PLoS ONE* 6: e27570.

Jones, J.P.G., Andriamarovololona, M.M. and Hockley, N. 2008. The importance of taboos and social norms to conservation in Madagascar. *Conservation Biology* 22: 976–986.

Kaufmann, J.C. 1998. The cactus was our kin: pastoralism in the spiny desert of southern Madagascar. In *Changing Nomads in a Changing World*, J. Ginat and A.M. Khazanov (Eds.), pp. 124–142. Sussex Academic, Brighton.

Kaufmann, J.C. and Tsirahamba, S. 2006. Forests and thorns: conditions of change affecting Mahafale pastoralists in southwestern Madagascar. *Conservation and Society* 4: 231–261.

Klein, J., Réau, B. and Edwards, M. 2008. Zebu landscapes: conservation and cattle in Madagascar. In *Greening the Great Red Island: Madagascar in Nature and Culture*, J.C. Kaufmann (Ed.), pp. 157–178. Africa Institute of South Africa, Pretoria.

Kümpel, N.F., Rowcliffe, J.M., Cowlishaw, G. and Milner-Gulland, E.J. 2009. Trapper profiles and strategies: insights into sustainability from hunter behaviour. *Animal Conservation* 12: 531–539.

Kümpel, N.F, Milner-Gulland, E.J., Cowlishaw, G. and Rowcliffe, J.M. 2010. Incentives for hunting: the role of bushmeat in the household economy in rural Equatorial Guinea. *Human Ecology* 38: 251–264.

Leader-Williams, N. and Albon, S.D. 1988. Allocation of resources to conservation. *Nature* 336: 533–535.

Lingard, M., Raharison, N., Rabakonandrianina, E., Rakotoarisoa, J.-A. and Elmqvist 2003. The role of local taboos in conservation and management of species: the radiated tortoise in southern Madagascar. *Conservation and Society* 1: 223–246.

Loudon, J.E., Sauther, M.L., Fish, K.D., Hunter-Ishikawa, M. and Ibrahim, Y.J. 2006. One reserve, three primates: applying a holistic approach to understand the interconnections among ring-tailed lemurs (*Lemur catta*), Verreaux's sifaka (*Propithecus verreauxi*) and humans (*Homo sapiens*) at Beza Mahafaly Special Reserve, Madagascar. *Ecological and Environmental Anthropology* 2: 54–74.

Marie, C.N., Sibelet, N., Dulcire, M., Rafalimaro, M., Danthu, P. and Carrière, S.M. 2009. Taking into account local practices and indigenous knowledge in an emergency conservation context in Madagascar. *Biodiversity and Conservation* 18: 2759–2777.

Mbete, R.A., Banga-Mboko, H., Racey, P., Mfoukou-Ntsakala, A., Nganga, I., Vermeulen, C., Doucet, J.-L., Hornick, J.-L. and Leroy, P. 2011. Household bushmeat consumption in Brazzaville, the Republic of the Congo. *Tropical Conservation Science* 4: 187–202.

McKenzie-Mohr, D. 2000. Fostering sustainable behaviour through community-based social marketing. *American Psychologist* 25: 531–537.

McKenzie-Mohr, D., Lee, N., Schultz, P.W. and Kotler, P. 2012. *Social Marketing to Protect the Environment: What Works*. Sage, Thousand Oaks.

Milner-Gulland, E.J., Bennett, E.L. and the SCB 2002 Annual Meeting Wild Meat Group 2003. Wild meat: the bigger picture. *Trends in Ecology and Evolution* 18: 351–357.

Nicoll, M.E. and Langrand, O. 1989. *Madagascar: Revue de la Conservation et des Aires Protégées*. WWF, Gland.

Pailler, S., Wagner, J.E., McPeak, J.G. and Floyd, D.W. 2009. Identifying conservation opportunities among Malinké bushmeat hunters of Guinea, West Africa. *Human Ecology* 37: 761–774.

Pangau-Adam, M., Noske, R. and Muehlenberg, M. 2012. Wildmeat or bushmeat? Subsistence hunting and commercial harvesting in Papua (West New Guinea), Indonesia. *Human Ecology* 40: 611–621.

Raherilalao, M.J. and Goodman, S.M. 2011. *Histoire Naturelle des Familles et Sous-familles Endémiques d'Oiseaux de Madagascar*. Association Vahatra, Antananarivo.

Raik, D. 2007. Forest management in Madagascar: an historical overview. *Madagascar Conservation & Development* 2: 5–10.

Rakotondravony, H.A. 2006. Communautés locales et gibiers dans la région de Daraina, Extrême Nord-est de Madagascar. *Madagascar Conservation & Development* 1: 19–21.

Randriamanalina, M.H., Rafararano, L., Babary, L. and Laha, R. 2000. Rapport des enquêtes sur les chasses dans les Fokontany d'Ivondro, d'Erara et d'Etsilesy. *Lemur News* 5: 11–14.

Randrianandrianina, F.H., Racey, P.A. and Jenkins, R.K.B. 2010. Hunting and consumption of mammals and birds by people in urban areas of western Madagascar. *Oryx* 44: 411–415.

Ratovonamana, R.Y. and Kiefer, I. 2009. Impact of grazing on the dry ecosystems of southern Madagascar. Unpublished report, University of Hamburg, Hamburg and WWF, Antananarivo.

Raxworthy, C. 1995. Field survey of amphibians, reptiles and small mammals from the Toliara region, Madagascar, October 5–30 1995. Unpublished field report, University of Michigan, USA.

Razafimanahaka, J.H., Jenkins, R.K.B., Andriafidison, D., Randrianandrianina, F., Rakotomboavonjy, V., Keane, A. and Jones, J.P.G. 2012. Novel approach for quantifying illegal bushmeat consumption reveals high consumption of protected species in Madagascar. *Oryx* 46: 584–592.

Réau, B. 2002. Burning for zebu: the complexity of deforestation issues in western Madagascar. *Norsk Geografisk Tidsskrift – Norwegian Journal of Geography* 56: 219–229.

Redford, K.H. 1992. The empty forest. *Bioscience* 42: 412–422.

Sayer, J. 2009. Can conservation and development really be integrated? *Madagascar Conservation & Development* 4: 9–12.

Scales, I. 2011. Farming at the forest frontier: land use and landscape change in western Madagascar, 1896–2005. *Environment and History* 17: 499–524.

Schultz, P.W. 2011. Conservation means behavior. *Conservation Biology* 25: 1080–1083.

Seddon, N., Tobias, J., Yount, J., Ramanampamonjy, J., Butchart, S. and Randrianizahana, H. 2000. Conservation issues and priorities in the Mikea Forest of south -west Madagascar. *Oryx* 34: 287–304.

Sinclair, I. and Langrand, O. 1998. *Birds of the Indian Ocean Islands*. Struik, Cape Town.

Stoner, K.E., Vulinec, K., Wright, S.J. and Peres, C.A. 2007. Hunting and plant community dynamics in tropical forests: a synthesis and future directions. *Biotropica* 39: 385–392.

Tengö, M., Johansson, K., Rakotondrasoa, F., Lundberg, J., Andriamaherilala, J.-A., Rakotoarisoa, J.-A. and Elmqvist, T. 2007. Taboos and forest governance: informal protection of hot spot dry forest in southern Madagascar. *Ambio* 36: 683–691.

Thomas, H., Kidney, D., Rubio, P. and Fanning, E. 2006. *The Southern Mikea: A Biodiversity Survey*. Society for Environmental Exploration, London, and Institut Halieutique et des Sciences Marines, Toliara.

Thompson, M.H. 2008. Fostering sustainable behaviours in community-based co-managed fisheries. *Marine Policy* 32: 413–420.

Tsirahamba, S. and Kaufmann, J.C. 2008. Absence of shade: deforestation in Mahafale forest-pastures. In *Greening the Great Red Island: Madagascar in Nature and Culture*, J.C. Kaufmann (Ed.), pp. 179–196. Africa Institute of South Africa, Pretoria.

Vargas, A., Jiménez,I., Palomares, F. and Palacios, M.J. 2002. Distribution, status, and conservation needs of the golden-crowned sifaka (*Propithecus tattersalli*). *Biological Conservation* 108: 325–334.

Wilkie, D.S. and Carpenter, J.F. 1999. Bushmeat hunting in the Congo Basin: an assessment of impacts and options for mitigation. *Biodiversity and Conservation* 8: 927–955.

Wunder, S. 2001. Poverty alleviation and tropical forests: what scope for synergies? *World Development* 29: 1817–1833.

Chapter 8

In Press

Oryx

Changing livelihoods and protected area management: a case study of charcoal production in southwest Madagascar

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Abstract

Protected areas are usually conceived and managed as static entities, although this approach is increasingly being viewed as unrealistic due to climate change and ecosystem dynamics. The way in which people use land and/or natural resources within and around protected areas can shift and evolve temporally, but remains an under-acknowledged challenge for protected area managers. Here we investigate the factors driving a rapid rise in charcoal production within a new, multiple-use protected area in Madagascar, in order to inform appropriate management responses. We conducted a questionnaire survey of 208 charcoal producers to ascertain the mix of livelihood activities they practiced in 2010/11 and five years previously. Respondents had diversified their livelihood activities over time, although cultivation and pastoralism had decreased as primary sources of revenue. Reasons for the growing reliance on charcoal production include the reduced viability of alternative livelihoods (primarily farming), as a result of changing rainfall patterns and the loss of irrigation infrastructure, as well as a growing need for cash to support themselves and their family. Our results suggest that charcoal production is not a desirable activity, but a ‘safety net’ when times are difficult. Conservation efforts to ameliorate underlying factors driving livelihood change, such as dam restoration, could

reduce the prevalence of charcoal production, but simultaneous action to cut demand is also required. We recommend that mechanisms to detect, understand and respond to social change are systematically integrated into protected area management planning, alongside traditional biodiversity monitoring.

8.1 Introduction

Covering over 12% of the world's land surface, protected areas constitute our principal approach to biodiversity conservation and comprise the largest planned land use globally (Jenkins and Joppa 2009). Described as "clearly defined geographical space[s], recognized, dedicated and managed, through legal or other effective means, to achieve the long-term conservation of nature with associated ecosystem services and cultural values" (Dudley 2008), the very concept of a protected area assumes that it will be preserved and/or managed in perpetuity, remaining a permanent fixture in the landscape. As such, they have been conceived and managed historically as static features that should persist unchanged through time (Bengtsson *et al*. 2003; Folke *et al*. 2005; Mascia and Pailler 2012).

Over recent years, the 'steady state' paradigm of protected area management has been increasingly viewed as inadequate, given the fact that ecosystems are inherently dynamic and that climate change will lead to the migration of species and habitats beyond protected area boundaries (Hannah 2008). Similarly, there is a growing awareness that protected areas are components of complex social-ecological systems (Milner-Gulland 2012; Ostrom 2009), with the resource use patterns of rural communities living within and around protected areas evolving through time (Aung *et al*. 2004; Geoghegan and Renard 2002; Newton 2011; Venter *et al*. 2008). Indeed, temporal shifts in land use and livelihoods should be seen as the rule rather than the exception (Folke 2003). However, while there is a large literature on protected areas as agents of social change (e.g. Ghimire and Pimbert 1997; Schmitz *et al*. 2012; West *et al*. 2006), there has been little research, policy or practical focus on livelihood dynamics as a management challenge for protected area managers. For example, none of the relevant publications within the International Union for the Conservation of Nature's (IUCN) Best Practice Protected Area Guidelines series (Dudley 2008; Phillips 2002; Thomas and Middleton 2003) provide any explicit instructions or recommendations regarding how to detect or manage sites being influenced by shifting livelihoods.

Resource use by local communities is a key threat to the viability of many protected areas worldwide (Gaston *et al*. 2008; Naughton-Treves *et al*. 2005). Yet, in order to design and implement appropriate evidence-based interventions, protected area managers must understand the factors that influence livelihood decision-making (St John *et al*. 2013). This is particularly important for protected areas with management objectives that include poverty alleviation or rural development alongside biodiversity conservation. However, the success rates associated with both integrated conservation and development projects (ICDPs) and community-based natural resource management (CBNRM) have been generally low. This is because, at least in part, managers have failed to sufficiently appreciate, understand and integrate local livelihood strategies and resource use patterns into their planning (Brown 2003; Dressler *et al*. 2010; Newmark and Hough 2000; Wells and McShane 2004).

Here, we investigate the drivers of rapid livelihood change, in the form of increased charcoal production, threatening biodiversity in southwest Madagascar. In 2009, staff of the international non-governmental organization (NGO) WWF observed a major increase in the amount of charcoal being produced inside a new multiple-use protected area, Ranobe PK32, and transported into a nearby city. In order to develop suitable protected area management strategies, it was necessary to understand the shift towards this livelihood; we needed to know what income generating activities people had been doing previously and why they had switched occupation.

8.2 Study system

Worldwide almost 3 billion people depend on biomass such as fuel-wood and charcoal for cooking (IEA 2010). Charcoal is produced by the slow pyrolysis (heating in the absence of oxygen) of wood and is a favoured cooking fuel of urban communities because it has a higher energy density than fuelwood and is therefore easy to transport (Arnold *et al*. 2003). However, inefficiencies in the conversion process mean that charcoal use consumes greater quantities of wood than the use of fuelwood (Bouwer and Falcão 2004).

The dynamics and impacts of the charcoal industry in southern Madagascar have been little researched, despite charcoal and fuel-wood constituting the primary domestic fuel for most of the population (Minten *et al*. 2012). Charcoal producers in the region use only hardwood trees (Mana *et al*. 2001), thus causing forest degradation rather than outright deforestation (Casse *et al*. 2004); this is also the case in sub-Saharan Africa (Ahrends *et* *al*. 2010). Degradation of southern Madagascar's spiny forest, a Global 200 priority ecoregion (Olson and Dinerstein 1998), triggers bird community turnover from endemic forest specialists to non-endemic generalists, thereby reducing its conservation value (Gardner, unpublished data). Additionally, the debris (e.g. leaves, small branches) left by charcoal production increases the standing fuel load of the forest and leaves it more vulnerable to fire (Gardner, unpublished data).

The average Malagasy family uses approximately 500 kg of charcoal per year (Meyers *et al*. 2006). In Toliara, the capital city of Atsimo Andrefana region (Fig. 8.1), less than 2 % of households regularly use electricity or gas to cook, and demand for charcoal in the city tripled between 2000 and 2007; 54 % of this demand was met by charcoal produced along the Route Nationale 9 (RN9) (Partage 2008), the sole road connecting Toliara with towns to the north.

Ranobe PK32 is a new protected area created in 2008, and extended in 2010, as part of Madagascar's 2003 'Durban Vision' initiative to triple the size of the protected area network. The establishment of the protected area has been led and funded by WWF, but it is governed by a co-management structure comprising WWF, regional authorities and local community representatives (Virah-Sawmy *et al*. 2014). It is managed for biodiversity conservation and the sustainable use of natural resources for poverty alleviation and development, in line with the objectives of the country's expanded protected area system (SAPM). Consequently, Ranobe PK32 is proposed for designation as an IUCN category VI multiple-use protected landscape (Gardner 2011). The protected area has the greatest species richness of lemurs and birds of any site in the spiny forest ecoregion (Gardner *et al*. 2009a; 2009b).

The majority of the human population around Ranobe PK32 live along the Fiherenana Valley, RN9 or in the agricultural plains south of the Manombo River, which were previously irrigated by a system of canals emanating from a colonial-era dam (Fig. 8.1). Livelihoods vary according to location and ethnicity; coastal villages are principally made up of Vezo fishers, while inland areas are inhabited largely by Masikoro agro-pastoralists (WWF 2010). The eastern part of the protected area lies on a Tertiary limestone plateau and is threatened by slash-and-burn maize cultivation (*hatsake*), but the unconsolidated sands of the coastal plain to the west are of poor quality for agriculture. Instead, the forests here are threatened by the production of charcoal for the urban market in Toliara,

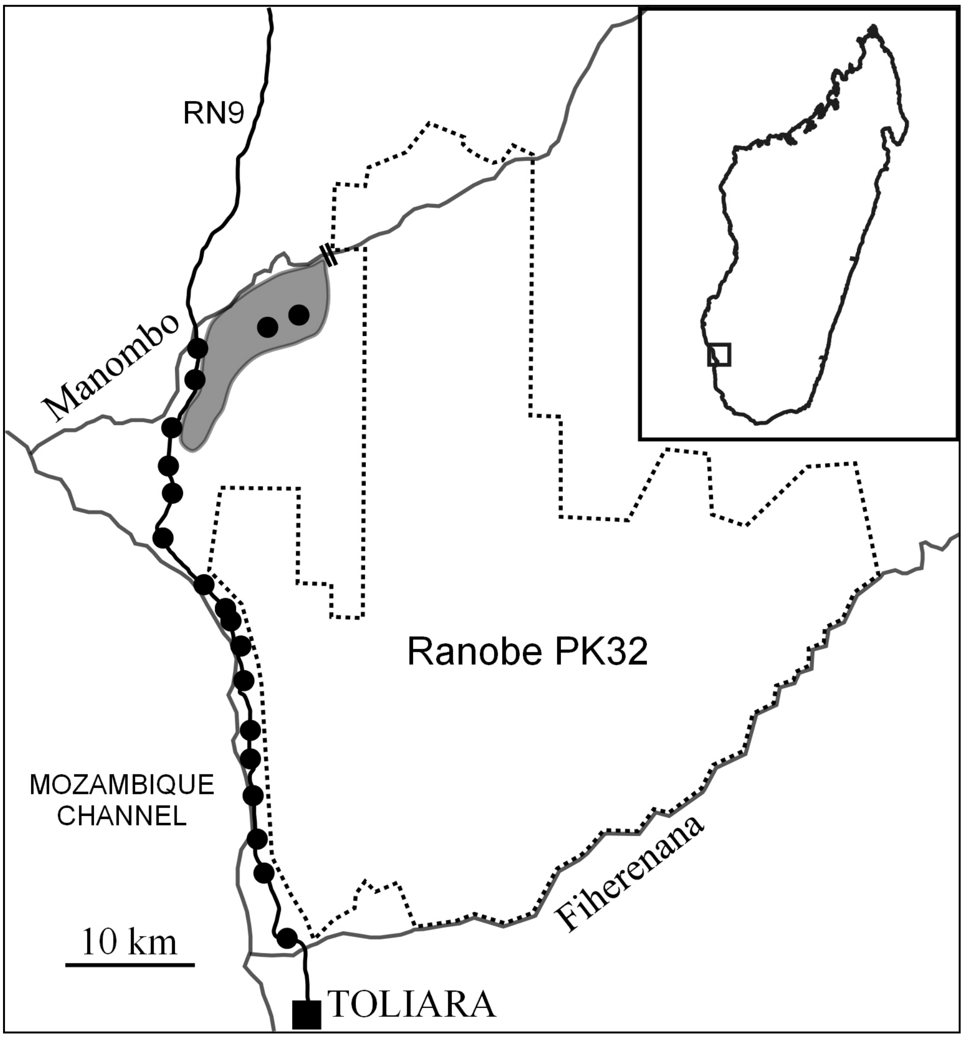

Figure 8.1 Map of Ranobe PK32 protected area (dotted outline) showing the Route Nationale 9 (RN9) road (dashed line) and villages where questionnaires were undertaken with charcoal producers (black circles). The two rivers (Manombo and Fiherenana) are indicated by grey lines, the Manombo River dam is shown as a double bar, and the extent of a formerly irrigated area is shaded in light grey. Inset shows location of the study region within Madagascar.

facilitated by the proximity of the RN9 (Seddon *et al*. 2000; Virah-Sawmy *et al*. 2014). Charcoal production is allowed within the protected area according to a zoning plan (i.e. outside core conservation areas), but all charcoal producers must obtain a permit from the State's Forest Service.

8.3 Methods

We delivered a questionnaire face-to-face with charcoal producers resident in villages along the RN9 between December 2010 and June 2011. As our aim was to understand the livelihood dynamics of people currently involved in charcoal production (rather than quantify the prevalence of charcoal producers in the population), non-probabilistic snowball sampling (Newing 2011) was used to identify potential respondents. Since we were investigating change in livelihoods, charcoal producers were only considered eligible if they had been independently earning a living for at least five years (i.e. they were not at school or otherwise dependent on their parents for a minimum of five years prior to interview). We also employed opportunistic sampling when producers were encountered transporting charcoal by ox-cart along the RN9, between villages.

On arrival in each village we met with the *Chef de Fokontany* (the head of the *Fokontany*, which is the smallest administrative unit, equivalent to a village or small cluster of villages) to explain the purpose of our research and asked him to suggest suitable participants. The homes of these individuals were visited in turn, with each questionnaire respondent asked to suggest additional possible participants within the village. The questionnaire was delivered in the local dialect of Malagasy by the second author (FULG), in a location chosen by the respondent (generally outside the home or in a public space), and all participants were assured that their responses would be anonymous and confidential.

The questionnaire consisted of both closed- and open-ended questions, structured into four short sections (Appendix 3, File A3.1). The survey was developed with colleagues and piloted on 14 initial respondents; this led to the modification of the questionnaire and, therefore, the data collected were discarded.

The first section collected basic socio-demographic information (e.g., age, level of education, ethnicity, village of current residence). Any participants found to be living away from their natal village (migrants) were asked about the factors that had contributed to them leaving their previous home and their choice of destination via an open-ended question. In order to detect any shifts in the relative importance of individual livelihood activities for household income, respondents were asked to rate activities on a three point ordinal scale both at the present time (2010/11) and five years previously (2005/6): (1) an activity that is never carried out; (2) an activity that is conducted infrequently (e.g., one day a week or for two months a year) and/or is of secondary importance to other sources of household revenue during the year (henceforth minor livelihoods), or; (3) an activity that is carried out often (e.g., three days a week or for six months of the year) and/or is an important source of revenue for their household (henceforth major livelihoods). A five-year period was selected as it was sufficient to capture the increase in charcoal production that had been observed anecdotally by WWF staff, without being so long as to diminish the viability of recall data (Golden *et al*. 2013). Individuals who indicated that the mix of livelihood activities they practice had changed over time were asked an open-ended question as to why this was the case in section three, which was facilitated by prompts from the interviewer from a list of potentially relevant drivers (e.g., lack of rain, not enough fish to catch, problems in life or family requiring money). The precise nature of any 'family problems' was not enquired about, as our pilot revealed that this made respondents uncomfortable. The final section focused on charcoal production specifically, with open-ended questions used to explore when during the year this activity is carried out, for how long and, if for just part of the year, why it was seasonal. In addition, open-ended questions were asked regarding where charcoal is produced and how far away from home this location is, and what difficulties the individual faces when producing charcoal.

Responses to open-ended questions were coded and grouped by response-type. Quantitative analyses of temporal shifts in livelihood activities, and differences in livelihoods between residents and migrants, were tested using chi-squared analyses in SPSS (version 20.0; IBM). A paired *t*-test was used to ascertain whether the number of livelihoods practiced by individual respondents had increased over time.

8.4 Results

A total of 208 questionnaires were completed in full by charcoal producers resident in villages along the RN9, representing 16 % of the estimated population of charcoal producers in the study area (WWF, unpublished data). The age range of study participants, who were all male, was 20 to 70 (median = 39.5, IQR = 33-47; mean = 39.7, SE = 0.76), and 60.6 % of individuals were currently living away from their natal village (Table 8.1). Of these, 15.1 % (n = 19) had migrated from Toliara, 57.1 % (n = 72) from elsewhere in Atsimo Andrefana region, and 27.8 % (n = 35) from the far south of Madagascar (primarily Androy region). Respondents who had migrated cited a number of factors that influenced their decision to translocate and which underpinned their selection of settling area (Table 8.2), all of which related to the need to earn money and the opportunities (or lack of) for doing so.

Only 7.2 % (n = 15) of individuals who participated in the questionnaire produced charcoal as their sole revenue generating activity. In addition to charcoal production, 63.9 % (n = 133) of respondents also engaged in sedentary cultivation, whilst 35.6 % (n = 74)

Table 8.1 Summary sociodemographic data from 19 villages along the Route Nationale 9, south-west Madagascar, where a questionnaire survey of charcoal producers was administered. Education data are presented in terms of: i) percentage of respondents that received some formal schooling; ii) range of years schooled; and, iii) median years schooled. Villages are ordered in increasing distance from Toliara.

Village	Number of respondents	Age		Mean no. children	Education			% migrants	Ethnicity				
		Range	Median		i	ii	iii		Masikoro	Vezo	Tanalana	Tandroy	Other
Belalanda	4	37-53	45	6.9	50	5-7	2.5	75.0	50	50	0	0	0
Tsongoritelo	4	29-52	40.5	4.8	50	3-6	1.5	50.0	0	75	25	0	0
Beravy	5	37-45	38	4.0	60	5-13	5	80.0	40	20	20	20	0
Ambalaboy	5	33-66	50	3.6	80	3-9	4	80.0	20	0	20	40	20
Tsivanoe	10	32-53	39.5	5.2	50	4-7	2	40.0	0	60	30	10	0
Mangily	10	20-60	44	3.8	80	1-9	2	90.0	20	50	10	20	0
Amboaboake	15	28-58	40	5.1	67	3-9	3	66.7	6.7	53.3	26.7	13.3	0
Madiorano	15	29-59	41	5.3	73	2-9	4	73.3	26.7	33.3	13.3	26.7	0
Betsibaroke	10	35-45	39.5	5.3	60	5-9	5	80.0	30	40	30	0	0
Ambolimailaka	20	30-54	40	4.5	70	2-9	4	90.0	30	10	25	30	5
Andrevo Haut	10	20-43	27	2.9	90	3-14	5.5	30.0	60	10	0	20	10
Antapoake	5	20-57	24	1.2	60	6-10	6	60.0	40	20	0	40	0
Ankatrakatraka	9	20-60	30	3.4	67	2-10	2	100	0	0	0	100	0
Ankilimalinika	15	20-70	25	3.9	67	2-13	5	33.3	66.7	0	6.7	13.3	13.3
Benetse	10	32-64	40	6.6	100	2-13	7	10.0	100	0	0	0	0
Saririake	6	38-52	42.5	5.3	33	1-9	0	16.7	83.3	0	0	16.7	0
Tsianisiha	9	22-51	35	5.0	100	2-9	5	44.4	100	0	0	0	0

Village	Number of respondents	Age		Mean no. children	Education			% migrants	Ethnicity				
		Range	Median		i	ii	iii		Masikoro	Vezo	Tanalana	Tandroy	Other
Berave Antsoity	16	24-66	44	5.5	88	2-10	5.5	56.3	75	6.3	12.5	0	6.3
Tsiafanoka	30	20-64	40	6.5	80	1-13	5	60.0	73.3	6.7	3.3	10	6.7
All villages	**208**	**20-70**	**39.5**	**4.9**	**73.1**	**1-14**	**5**	**60.6**	**46.6**	**19.7**	**12.0**	**17.8**	**3.9**

Table 8.2 Reasons cited by migrant charcoal producers (n = 126) encountered along Route Nationale 9 for migrating and selecting their current place of residence. Some participants provided multiple responses, therefore totals exceed 100 %.

Reason for migrating	% of respondents	Reason for selecting current location	% of respondents
Family reasons	28.6	Family lives here	46.0
Lack of work/activities	27.8	Availability of land	18.8
Insecurity	16.7	Existence of good forest	18.3
Extreme poverty/famine	11.9	Good fishing	7.1
Lack of cultivable land	9.5	Existence of agricultural infrastructure	5.4
Drought/lack of rain	6.4	To trade	4.0
Loss of agricultural infrastructure	4.0	To work for foreigners	3.2
Disappearance of forest	3.2	Looking for work	1.6
Fleeing life of crime	1.6	To herd livestock	0.8
Lack of fish	0.8	Secure location without cattle rustlers	0.8

reared livestock and 20.2 % (n = 42) fished or harvested other marine resources. Almost a third (32.7 %, n = 68) were additionally involved in other livelihood activities including timber harvesting, shop keeping and making reed houses. Three activities increased significantly over the five year study period from 2005/6; charcoal production (562.2 % rise; chi-squared = 247.6, $p < 0.01$, d.f. = 2), livestock rearing (224.2 % rise; chi-squared = 20.8, $p < 0.01$, d.f. = 2) and timber harvesting (300.0 % rise; chi-squared = 5.2, $p < 0.05$, d.f. = 2). An 11.3 % decrease in sedentary cultivation over the same timeframe was not significant. People producing charcoal in 2010/11 had diversified their livelihoods through time, from a mean of 1.4 to 2.5 activities/person ($t = 18.4$, $p < 0.01$, d.f. = 207), with the vast majority (92.8 %, n = 193) reporting changes in the mix of livelihood activities they practiced over the five years.

Amongst major livelihoods, the number of participants engaging in charcoal production (565.6 % rise, from 15.4 % in 2005/6 to 87.0 % in 2010/11; chi-squared = 213.6, $p < 0.01$, d.f. = 2) and shifting cultivation (from 0.0 % in 2005/6 to 1.9 % in 2010/11; chi-squared = 4.0, $p < 0.05$, d.f. = 2) rose significantly over time. Conversely, sedentary cultivation (12.8 % drop, from 71.1 % in 2005/6 to 62.0 % in 2010/11; chi-squared = 4.1, $p < 0.05$, d.f. = 2) and livestock rearing (79.1 % decline, from 11.5 % in 2005/6 to 2.4 % in 2010/11; chi-squared = 4.5, $p < 0.05$, d.f. = 2) declined significantly in the five years. Although livestock rearing decreased as a major livelihood, it showed significant increases as a minor livelihood activity (772.1 % growth, from 4.3 % in 2005/6 to 33.2 % in 2010/11; chi-squared = 56.0, $p < 0.01$, d.f. = 2). Charcoal production was the only other activity to grow as a minor livelihood (541.7 % rise, from 2.4 % in 2005/6 to 13.0 % in 2010/11; chi-squared = 16.4, $p < 0.01$, d.f. = 2).

Variation in these trends was further examined for migrants and residents separately (Table 8.3). Few differences were apparent between the two groups, although, between 2005/6 and 2010/11, there were significant declines in the proportion of migrants engaging in sedentary cultivation and rearing livestock as major livelihoods that were not reflected in resident populations (chi-squared = 6.6, $p < 0.01$, d.f. = 2 and chi-squared = 20.8, $p < 0.01$, d.f. = 2 respectively).

Respondents cited a range of factors as having driven a shift in livelihood activities over the period investigated (Fig. 8.2), reporting a mean of 2.7 factors per individual. We grouped proximal causes into those that refer to the needs of the respondent and his household (which we term endogenous factors) and those that affected his ability to meet those needs (exogenous factors). Exogenous factors diminished the viability of some livelihood activities, particularly sedentary cultivation. Over two-thirds of participants (n = 134) stated that their shift to charcoal production was a consequence of lack of rain or a change in the rainy season, while 30.3 % (n = 63) cited the loss of irrigation infrastructure (dam and associated canals) south of the Manombo River. Similarly, 15.4 % (n = 32) of respondents referred to a decrease in fish/marine resources, and 7.7 % (n = 16) to growing rates of cattle theft, as factors that contributed to their adoption of charcoal production. Endogenous factors accounted for almost half (47.5 %) of the responses provided, with participants reporting rising living costs, having more children, and family problems as driving their increased needs for cash, and hence a shift in livelihoods.

Over half of our study participants (52.4 %, n = 109) produce charcoal for only part of the year, during the agricultural off-season (generally March to August) when they are not occupied in their fields. Seven per cent (n = 14) produce charcoal whenever they need

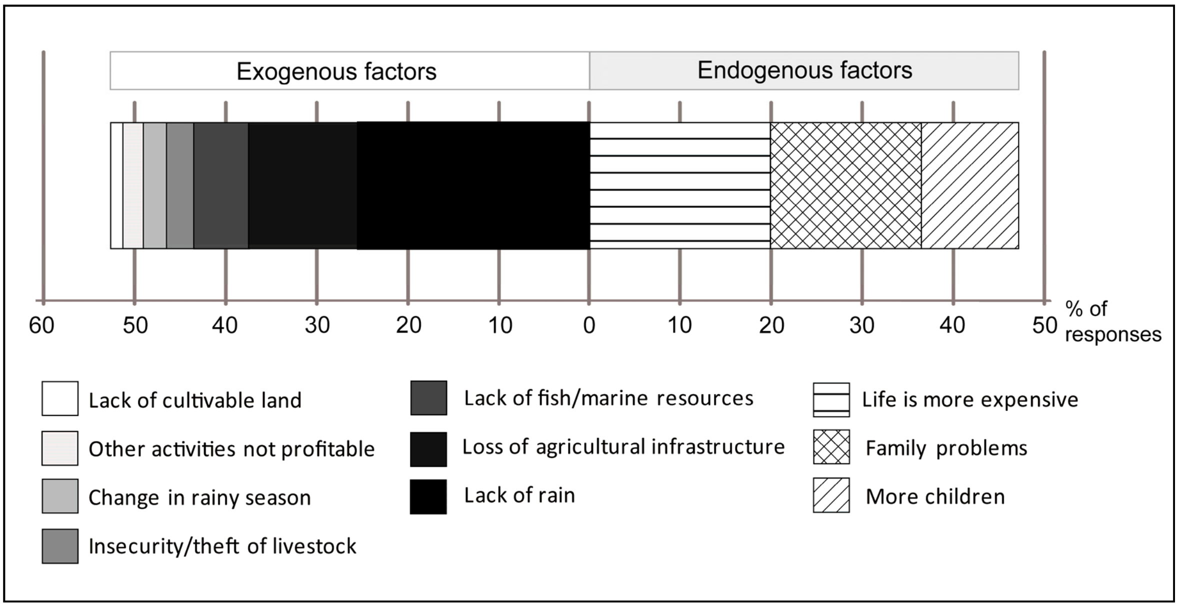

Figure 8.2 Drivers of livelihood change between 2005/6 and 2010/11 cited by charcoal producers encountered along the Route Nationale 9, who had altered their revenue generating activities over the five year period (n = 193). Answers total greater than 100 % because survey participants provided a mean of 2.7 responses each.

money (including for celebrations such as Christmas and Independence Day, and emergencies), whereas the remaining 40.9 % (n = 85) do so throughout the year.

Most participants stated that their lives were harder (58.7 %, n = 122) or just as hard (27.4 %, n = 57) in 2010/11 as they had been five years previously, while 13.9 % (n = 29) stated that their lives had improved. When asked about the difficulties experienced with charcoal production as a livelihood, respondents provided a range of answers (mean = 2.3 responses/person) reflecting the physical hardship, medical problems, shame and low revenues associated with the activity (Table 8.4).

8.5 Discussion

Following the observation that charcoal production had proliferated in southwest Madagascar and was causing increased forest degradation, particularly in and around the Ranobe PK32 protected area, we sought to understand the reasons why people where taking up this livelihood. Our findings suggest that charcoal production is a means of earning money when other preferred options are no longer viable or sufficiently productive. Fewer than one in five participants in our study had been producing charcoal in 2005/6, but a range of issues had either diminished the feasibility of pursuing

Table 8.3 Percentage of charcoal producers along Route Nationale 9 (n = 208), separated into residents and migrants, carrying out different revenue generating activities in 2005/6 and 2010/11. Arrows show an increase or decrease in each activity, with asterisks indicating the significance of the trend using chi-squared contingency tables: ** = $p < 0.01$, * = $p < 0.05$. NTFP = Non-timber forest products. Participants tended to engage in multiple activities, so totals exceed 100 %.

Activity	RESIDENTS (n = 82)						MIGRANTS (n = 126)					
	Major livelihood			**Minor livelihood**			**Major livelihood**			**Minor livelihood**		
	2005/6	**2010/11**	**Change**	**2005/6**	**2010/11**	**Change**	**2005/6**	**2010/11**	**Change**	**2005/6**	**2010/11**	**Change**
Charcoal Production	11.0	87.8	↑**	0	12.2	↑**	18.2	86.6	↑**	4.0	13.4	↑**
Sedentary cultivation	72.0	72.0	—	0	1.2	↑	70.6	55.6	↓**	1.6	2.4	↑
Shifting cultiva-tion	0	2.4	↑	0	1.2	↑	0.8	1.6	↑	0	0.8	↑
Livestock	0	3.7	↑	11.0	35.4	↑**	19.0	1.6	↓**	0	31.7	↑**
Fishing	17.1	18.3	↑	1.2	0	↓	15.1	18.3	↑	0	3.2	↑
Timber	4.9	7.3	↑	0	1.2	↑	0	4.0	↑	0.8	2.4	↑
NTFP collection	3.7	0	↓	0	0	—	0.8	0	↓	0.8	1.6	↑
Trade/ shop-keeping	2.4	6.1	↑	1.2	3.7	↑	7.1	7.9	↑	2.4	2.4	—
Salaried work	1.2	0	↓	0	0	—	4.8	7.9	↑	0	0	—
Reed houses	1.2	4.9	↑	0	0	—	1.6	3.2	↑	0	1.6	↑
Other	3.7	3.7	—	0	1.2	↑	0	0	—	0	0	—

Table 8.4 Factors stated by respondents in a questionnaire survey along the Route Nationale 9 (n = 208) as contributing to the difficulty of their lives as charcoal producers. Figures total more than 100 due to multiple responses per participant. KASTI are community agents of the State's Forest Service.

Problems with charcoal production as a livelihood activity	% of respondents
Work is very tiring	53.4
The forest is further away than it was previously	39.4
Lack of tools (e.g. axes)	31.7
Lack of large trees	17.3
Insecurity; have to guard kiln at night	16.8
It is shameful work (prisoners' work)	14.0
Lack of transport (ox-cart)	13.0
Work provokes respiratory illness	12.5
Have to pay KASTI in Ranobe for 'permit'	7.7
Have to stay in the forest for several weeks	6.7
It is not profitable; the price of charcoal has diminished	6.7
The whole family has to help with the work	5.8
Lack of suitable trees; forced to use stumps/roots	4.3
It is dangerous in the forest	1.9

alternative revenue streams, such as sedentary cultivation (due to exogenous factors such as lack of infrastructure or drought), or had given them a need for supplemental income (driven by endogenous pressures including family problems and growing family size). The participants had not simply abandoned their livelihood activities from five years previously, rather they have diversified their sources of revenue in order to meet their increased monetary needs. In addition, charcoal production is a 'gap-filler' during the agricultural off season, and a relatively rapid way of generating cash when needed at short notice. As such, it can be characterised as a fallback activity or safety net (Sunderlin *et al.* 2005).

This situation corresponds with many others described globally, where the rural poor turn to forest resource use in the absence of more favourable options for income generation (Vira and Kontoleon 2010) and as "employment of last resort" (Angelsen and Wunder 2003). It occurs because the chronically poor tend to live disproportionately in remote rural areas neighbouring forests (Hulme and Shepherd 2003), and because forests are generally easy to access with few physical or technical barriers to prevent their exploitation (Sunderlin *et al*. 2005). For example, analyses of illegal logging in Indonesia have found that participation in the industry grew as a consequence of declining returns from agriculture (Angelsen and Resosudarmo 1999) and the rising need for cash within rural village communities (Yonariza and Webb 2007); households with fewer options to generate income were more likely to participate in logging (Byron and Arnold 1999). In Madagascar specifically, other 'safety net' livelihood activities include bushmeat hunting (Gardner and Davies 2014; Goodman 2006), wild yam collection (Ackermann 2003) and the use of forest products more broadly (Favre 1996). Forests also provide a reserve of potential agricultural land, albeit of poor quality in general, that may be converted to shifting cultivation, primarily by migrants fleeing drought or seeking cash to invest in cattle (Réau 2002; Scales 2014).

Given the importance of charcoal production as a safety net, reducing the practice through rule enforcement (either by reducing the granting of permits or cracking down on charcoal produced without a permit) would probably exacerbate poverty. This would be incompatible with the objectives of the Madagascar protected area system (SAPM), which include both the conservation of biodiversity and the sustainable use of natural resources for poverty alleviation and local development (Gardner *et al*. 2013). Additionally, it would run counter to worldwide calls for conservation to, at very least, avoid worsening poverty among affected communities (Adams *et al*. 2004; Kaimowitz and Sheil 2007). Enforcement is also hampered by rampant corruption in natural resource extraction sectors (Randriamalala and Liu 2010) and a lack of political will. Indeed, attempts by the regional administration of Atsimo Andrefana to further regulate the charcoal sector in 2007 led to civil unrest in Toliara city and were rapidly withdrawn (Bertrand *et al*. 2010). Instigating a reversal in the trend towards increased participation in charcoal production within Ranobe PK32 will thus depend partly on reducing its attractiveness as a livelihood relative to potential alternatives. Theoretically, this can be achieved by tackling the underlying factors that push charcoal producers into the industry, and our findings suggest several potential interventions worth exploring.

Endogenous and exogenous factors were cited in approximately equal frequency as drivers of livelihood change. The primary exogenous factors raised by our respondents centred on the diminishing productivity of agriculture, due to a lack of rain and the loss of irrigation infrastructure in the north of the study area. Farmers have been forced to seek alternative sources of income due to the Manombo dam no longer being operational, and our survey suggests that restoring the dam and associated canals, a task which is currently being conducted by the African Development Bank, could lead to approximately 30% of questionnaire participants abandoning charcoal production to return to farming. Likewise, the establishment of climate-wise agricultural adaptation programmes, including the development and popularization of improved farming techniques and drought-resistant crops, would help mitigate the problems of low/unpredictable rainfall cited by the respondents. Undoubtedly, this will become increasingly pertinent in the future, with climate change expected to have a substantial negative impact on agricultural production in southern Madagascar (Thornton *et al*. 2011). Without such action, other 'safety net' forest uses such as shifting cultivation may become more prevalent, in addition to charcoal production. Similarly, interventions aimed at decreasing cattle theft by improving rural security, and improving the sustainability of marine resources through fisheries management, would help restore the viability of pastoralism and fishing as alternatives to charcoal production.

Although the interventions listed above ('distraction' activities, Milner-Gulland and Rowcliffe 2007) could diminish the relative attractiveness of charcoal production as a livelihood, higher income arising from development gains could be invested in further charcoal production or other natural resource exploitation by beneficiaries. The interventions, therefore, must be accompanied by the enforcement of regulations or be made conditional on reductions in environmentally-damaging activities (Sievanen *et al*. 2005; St John *et al*. 2013). Moreover, it is highly like that a decline in charcoal production in Ranobe PK32 would be offset by leakage (Ewers and Rodrigues 2008) so long as demand from the city remains constant, as the activity would just be displaced elsewhere. Decreasing production at a regional scale would thus require a drop in demand for charcoal from natural forests either directly (e.g. through the popularization of fuel-efficient stoves, which has been widely promoted worldwide but with variable success

(Anenberg *et al*. 2013)), and/or indirectly through the introduction of an alternative supply (e.g. from fuelwood plantations or biomass briquettes). In this regard, it is worth noting that many of our respondents expressed a desire for woodlots or small plantations of fast-growing tree species to be established near their villages, although this was beyond the scope of our questionnaire.

Endogenous drivers of increased charcoal production were also frequently raised by our study participants, including problems within their family that required cash, and their growing number of children. While we did not systematically probe into what constituted 'family problems' in the final version of the questionnaire, findings from the pilot exercise indicated that this generally referred to either expensive medical emergencies or a death in the family or community. Funerals in southern Madagascar may consist of extremely lavish ceremonies, involving the construction of expensive tombs and the slaughter of many zebu cattle (Casse *et al*. 2005). Family members are expected to contribute cattle as gifts, and such expenses, often required at short-notice, may make up a large proportion of a household's annual expenditure. While land-owners may sell a land parcel to generate the necessary funds (Blanc-Pamard 2004), charcoal production offers an opportunity to raise money relatively quickly for those lacking alternative assets. Considering that growing family size was cited as a factor pushing over a quarter of our sample towards charcoal production, the provision of family planning services could contribute to decreasing household expenditures and thus the pressure to practice environmentally destructive activities such as charcoal production to generate cash (Allendorf and Allendorf 2012; Harris *et al*. 2012). .

Approximately two thirds of the charcoal producers we surveyed were migrants to the southern RN9 area. Comparing the two social groups, we found no evidence that residents and migrants engage in a different mix of livelihoods, or that the recent growth in charcoal production has been driven by newcomers. Malagasy societies are dynamic, and migration to the forest frontier is a typical response to resource scarcity (Keller 2008). As such, numerous authors have remarked that migrant communities tend to engage in less sustainable resource-use practices than residents (e.g. Andriamalala and Gardner 2010; Horning 2003; Kaufmann and Tsirahamba 2006; Réau 2002). However, our findings do not support this observation. Furthermore, Bertrand *et al*. (2010) state that charcoal production in the Toliara region is tightly linked to migration dynamics because it is primarily produced as a secondary output from shifting cultivation; our data provide evidence to the contrary, since less than 2% of our sample were also engaged in slash-and -burn agriculture.

Our research demonstrates that livelihood change around protected areas can be rapid, involve large numbers of people, and have multiple underlying causes. Given that it can have severe impacts on biodiversity, we suggest that protected area management planning should systematically include mechanisms to: i) detect; ii) understand; and, iii) mitigate/ adapt to livelihood change in order to minimise its potentially negative effects. The detection of change requires the implementation of a monitoring plan, which is already recognized as a fundamental component of a management plan (Phillips 2002; Thomas and Middleton 2003). Nonetheless, such monitoring systems in protected areas typically focus on biodiversity (e.g. densities of indicator/important species) and, therefore, expose only the outcome of changing resource use once it has occurred. Instead, more attention needs to be given to examining the socio-economic conditions being experienced within local communities, as well as understanding the factors that motivate shifts in behaviour, so appropriate management responses can be developed. Moreover, both protected area managers working across a region, and the funding bodies supporting their conservation efforts, need to be sufficiently flexible in order to rapidly implement new management strategies in response to substantive livelihood change.

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References

Ackermann, K. 2003. The role of dry forests in Madagascar as a safety net in the rural livelihood system. Paper presented at the International Conference on Rural Livelihoods, Forests and Biodiversity, 19-23 May 2003, Bonn, Germany.

Adams, W.M., Aveling, R., Brockington, D., Dickson, B., Elliott, J., Hutton, J., Roe, D., Vira, B. and Wolmer, W. 2004. Biodiversity conservation and the eradication of poverty. *Science* 306: 1146–1149.

Ahrends, A., Burgess, N.D., Milledge, S.A.H., Bulling, M.T., Fisher, B., Smart, J.C.R., Clarke, G.P., Mhoro, B.E. and Lewis, S.L. 2010. Predictable waves of sequential forest degradation and biodiversity loss spreading from an African city. *Proceedings of the National Academy of Science, USA* 107: 14556–14561.

Allendorf, T.D. and Allendorf, K. 2012. What every conservation biologist should know about human population. *Conservation Biology* 26: 953–955.

Andriamalala, G. and Gardner, C.J. 2010. L'utilisation du *dina* comme outil de gouvernance des ressources naturelles: leçons tirés de Velondriake, sud-ouest de Madagascar. *Tropical Conservation Science* 3: 447–472.

Anenberg, S.C., Balakrishnan, K., Jetter, J., Masera, O., Mehta, S., Moss, J. and Ramanathan, V. 2013. Cleaner cooking solutions to achieve health, climate, and economic cobenefits. *Environmental Science and Technology* 47: 3944–3952.

Angelsen, A. and Resosudarmo, D. 1999. *Krismon, Farmers, and Forests: the Effects of the Economic Crisis on Farmers' Livelihoods and Forest Use in the Outer Islands of Indonesia*. Centre for International Forestry Research, Bogor.

Angelsen, A. and Wunder, S. 2003. *Exploring the Poverty-Forestry Link: Key Concepts, Issues and Research Implications*. Centre for International Forestry Research, Bogor.

Arnold, M., Köhlin, G., Persson, R. and Shepherd, G. 2003. *Fuelwood Revisited: What has Changed in the Last Decade?* Centre for International Forestry Research, Bogor.

Aung, M., Swe, K.K., Oo, T., Moe, K.K., Leimgruber, P., Allendorf, T., Duncan, C. and Wemmer, C. 2004. The environmental history of Chatthin Wildlife Sanctuary, a protected area in Myanmar (Burma). *Journal of Environmental Management* 72: 205–216.

Bengtsson, J., Angelstam, P., Elmqvist, T., Emanuelsson, U., Folke, C., Ihse, M., Moberg, F. and Nyström, M. 2003. Reserves, resilience and dynamic landscapes. *Ambio* 32: 389–396.

Bertrand, A., Ramamonjisoa, B. and Montagne, P. 2010. Les filières péri-urbaines d'approvisionnement en bois énergie des grandes villes de Madagascar. In *ARINA, le Charbon de Bois à Madagascar: Entre Demande Urbaine et Gestion Durable*, P. Montagne, S. Razafimahatratra, A. Rasamindisa and R. Crehay (Eds.), pp. 23–36. CITE, Antananarivo.

Blanc-Pamard, C. 2004. Un jeu foncier de front pionnier en forêt des Mikea (sud-ouest de Madagascar): pratiques, acteurs et enjeux. *Land Reforme/Réforme Agraire/Reforma Agraria* 2: 94–107.

Brown, K. 2003. Integrating conservation and development: a case of institutional misfit. *Frontiers in Ecology and the Environment* 1: 479–487.

Bouwer, R. and Falcão, P.N. 2004. Wood fuel consumption in Maputo, Mozambique. *Biomass and Bioenergy* 27: 233–245.

Byron, N. and Arnold, M. 1999. What futures for the people of the tropical forests? *World Development* 27: 789–805.

Casse, T., Milhøj, A., Ranaivoson, S. and Randriamanarivo, J.R. 2004. Causes of deforestation in southwestern Madagascar: what do we know? *Forest Policy and Economics* 6: 33–48.

Casse, T., Nielsen, U., Ranaivoson, S. and Randrianamarivo, J.R. 2005 Farmer strategies and forest conservation: a case study from south-western Madagascar. *International Journal of Social Economics* 32: 704–716.

Dressler, W., Buscher, B., Schoon, M., Brockington, D., Hayes, T., Kull, C.A., McCarthy, J. and Shrestha, K. 2010. From hope to crisis and back again? A critical history of the global CBNRM narrative. *Environmental Conservation* 37: 5–15.

Dudley, N. 2008 *Guidelines for Applying Protected Area Management Categories*. IUCN, Gland.

Ewers, R.M. and Rodrigues, A.S.L. 2008. Estimates of reserve effectiveness are confounded by leakage. *Trends in Ecology and Evolution* 23: 113–116.

Favre, J.-C. 1996. Traditional utilization of the forest. In *Ecology and Economy of a Tropical Dry Forest in Madagascar*, J.U. Ganzhorn and J.P. Sorg (Eds.), pp. 33–40. *Primate Report* 46-1.

Folke, C. 2003. Reserves and resilience – from single equilibrium to complex systems. *Ambio* 32: 379–379.

Folke, C., Hahn, T., Olsson, P. and Norberg, J. 2005. Adaptive governance of social-ecological systems. *Annual Review of Environment and Resources* 30: 441–473.

Gardner, C.J. 2011. IUCN management categories fail to represent new, multiple-use protected areas in Madagascar. *Oryx* 45: 336–346.

Gardner, C.J. and Davies, Z.G. 2014. Rural bushmeat consumption within multiple-use protected areas: qualitative evidence from southwest Madagascar. *Human Ecology* 42: 21–34.

Gardner, C.J., Kidney, D. and Thomas, H. 2009a. First comprehensive avifaunal survey of PK32-Ranobe, a new protected area in south-western Madagascar. *Phelsuma* 17: 20–39.

Gardner, C.J., Fanning, E., Thomas, H. and Kidney, D. 2009b. The lemur diversity of the Fiherenana-Manombo Complex, southwest Madagascar. *Madagascar Conservation &* Development 4: 38–43.

Gardner, C.J., Nicoll, M.E., Mbohoahy, T., Olesen, K.L.L., Ratsifandrihamanana, A.N., Ratsirarson, J., René de Roland, L.-A., Virah-Sawmy, M., Zafindrasilivonona, B. and Davies, Z.G. 2013. Protected areas for conservation and poverty alleviation: experiences from Madagascar. *Journal of Applied Ecology* 50: 1289–1294.

Gaston, K.J., Jackson, S.F., Cantú-Salazar, L. and Cruz-Piñón, G. 2008. The ecological performance of protected areas. *Annual Review of Ecology, Evolution and Systematics* 39: 93–113.

Geoghegan, T. and Renard, Y. 2002. Beyond community involvement in protected area planning and management: lessons from the insular Caribbean. *Parks* 12: 16–27.

Ghimire, K.B. and Pimbert, M.P. (Eds.) (1997) *Social Change and Conservation*. Earthscan, London.

Golden, C.D., Wrangham, R.W. and Brashares, J.S. 2013. Assessing the accuracy of interviewed recall for rare, highly seasonal events: the case of wildlife consumption in Madagascar. *Animal Conservation* 16: 597–603.

Goodman, S.M. 2006. Hunting of microchiroptera in south-western Madagascar. *Oryx* 40: 225–228.

Hannah, L. 2008. Protected areas and climate change. *Annals of the New York Academy of Sciences* 1134: 201–212.

Harris, A., Mohan, V., Flanagan, M. and Hill, R. 2012. Integrating family planning service provision into community-based marine conservation. *Oryx* 46: 179–186.

Horning, N.R. 2003. How rules affect conservation outcomes. In *The Natural History of Madagascar*, S.M. Goodman and J.P. Benstead (Eds.), pp. 146–153. University of Chicago Press, Chicago.

Hulme, D. and Shepherd, A. 2003. Conceptualizing chronic poverty. *World Development* 31: 403–424.

IEA (International Energy Agency). 2010. *World Energy Outlook 2010*. International Energy Agency, Paris.

Jenkins, C.N. and Joppa, L. 2009. Expansion of the global terrestrial protected area system. *Biological Conservation* 142: 2166–2174.

Kaimowitz, D. and Sheil, D. 2007. Conserving what and for whom? Why conservation should help meet basic human needs in the tropics. *Biotropica* 39: 567–574.

Kaufmann, J.C. and Tsirahamba, S. 2006. Forests and thorns: conditions of change affecting Mahafale pastoralists in southwestern Madagascar. *Conservation and Society* 4: 231–261.

Keller, E. 2008. The banana plant and the moon: conservation and the Malagasy ethos of life in Masoala, Madagascar. *American Ethnologist* 35: 650–664.

Mana, P., Rajaonarivelo, S. and Milleville, P. 2001. Production de charbon de bois dans deux situations forestières de la région de Tuléar. In *Sociétés Paysannes, Transitions Agraires et Dynamiques Ecologiques dans le Sud-ouest de Madagascar*, S. Razanaka, M. Grouzis, P. Milleville, B. Moizo and C. Aubry (Eds.), pp. 199–210. CNRE-IRD, Antananarivo.

Mascia, M.B. and Pailler, S. 2012. Protected area downgrading, downsizing, and degazettement (PADDD) and its conservation implications. *Conservation Letters* 4: 9–20.

Meyers, D., Ramamonjisoa, B., Sève, J., Rajafindramanga, M. and Burren, C. 2006. *Etude sur la Consommation et la Production en Produits Ligneux*. Jariala and USAID, Antananarivo.

Milner-Gulland, E.J. 2012. Interactions between human behaviour and ecological systems. *Philosophical Transactions of the Royal Society, B* 367: 270–278.

Milner-Gulland, E.J. and Rowcliffe, J.M. 2007. *Conservation and Sustainable Use: a Handbook of Techniques*. Oxford University Press, Oxford.

Minten, B., Sander, K. and Stifel, D. 2012. Forest management and economic rents: evidence from the charcoal trade in Madagascar. *Energy for Sustainable Development* 17: 106–115.

Naughton-Treves, L., Buck, M. and Brandon, K. 2005. The role of protected areas in conserving biodiversity and sustaining local livelihoods. *Annual Review of Environment and Resources* 30: 219–252.

Newing, H.S. 2011. *Conducting Research in Conservation: a Social Science Perspective*. Routledge, Abingdon.

Newmark, W.D. and Hough, J.L. 2000. Conserving wildlife in Africa: integrated conservation and development projects and beyond. *BioScience* 50: 585–592.

Newton, A.C. 2011. Social-ecological resilience and biodiversity conservation in a 900-year-old protected area. *Ecology and Society* 16: 13.

Olson, D. and Dinerstein, E. 1998. The Global 200: a representation approach to conserving the world's most biologically valuable ecoregions. *Conservation Biology* 12: 502–515.

Ostrom, E. 2009. A general framework for analyzing sustainability of social-ecological systems. *Science* 325 : 419–422.

Partage. 2008. *Analyse de l'Offre et de la Demande en Energie Domestique au Niveau de la Commune de Toliara*. Partage, Antananarivo.

Phillips, A. 2002. *Management Guidelines for IUCN Category V Protected Areas: Protected Landscapes/Seascapes*. IUCN, Gland.

Randriamalala, H. and Liu, Z. 2010. Rosewood of Madagascar: between democracy and conservation. *Madagascar Conservation & Development* 5: 11–22.

Réau, B. 2002. Burning for zebu: the complexity of deforestation issues in western Madagascar. *Norwegian Journal of Geography* 56: 219–229.

Scales, I.R. 2014. The drivers of deforestation and the complexity of land use in Madagascar. In *Conservation and Environmental Management in Madagascar*, I.R. Scales (Ed.), pp. 105–125. Routledge, London and New York.

Schmitz, M.F., Matos, D.G.G., De Aranzabal, I., Ruiz-Labourdette, D. and Pineda, F.D. 2012. Effects of a protected area on land-use dynamics and socioeconomic development of local populations. *Biological Conservation* 149: 122–135.

Seddon, N., Tobias, J., Yount, J., Ramanampamonjy, J., Butchart, S. and Randrianizahana, H. 2000. Conservation issues and priorities in the Mikea Forest of south -west Madagascar. *Oryx* 34: 287–304.

Sievanen, L., Crawford, B., Pollnac, R. and Lowe, C. 2005. Weeding through assumptions of alternative livelihood approaches in ICM: seaweed farming in the Philippines and Indonesia. *Ocean and Coastal Management* 48: 297–313.

St. John, F.A.V., Keane, A. and Milner-Gulland, E.J. 2013. Effective conservation depends upon understanding human behaviour. In *Key Topics in Conservation Biology 2*, D.W. Macdonald and K.J. Willis (Eds.), pp. 344–361.Wiley-Blackwell, Chichester.

Sunderlin, W.D., Angelsen, A., Belcher, B., Burgers, P., Nasi, R., Santoso, L. and Wunder, S. 2005. Livelihoods, forests, and conservation in developing countries: an overview. *World Development* 33: 1383–1402.

Thomas, L. and Middleton, J. 2003. *Guidelines for Management Planning of Protected Areas*. IUCN, Gland.

Thornton, P.K., Jones, P.G., Ericksen, P.J. and Challinor, A.J. 2011. Agriculture and food systems in sub-Saharan Africa in a 4°C+ world. *Philosophical Transactions of the Royal Society A* 369: 117–136.

Venter, F.J., Naiman, R.J., Biggs, H.C. and Pienaar, D.J. 2008. The evolution of conservation management philosophy: science, environmental change and social adjustments in Kruger National Park. *Ecosystems* 11: 173–192.

Vira, B. and Kontoleon, A. 2010. Dependence of the poor on biodiversity: which poor, what biodiversity? In *Linking Biodiversity Conservation and Poverty Alleviation: a State of Knowledge Review*, D. Roe (Ed.), pp. 13–28. Secretariat of the Convention on Biological Diversity, Montreal.

Virah-Sawmy, M., Gardner, C.J. and Ratsifandrihamanana, A.N. 2014. The Durban Vision in practice: experiences in the participatory governance of Madagascar's new protected areas. In *Conservation and Environmental Management in Madagascar*, I.R. Scales (Ed.), pp. 216–251. Routledge, London and New York.

Wells, M.P. and McShane, T.O. 2004. Integrating protected area management with local needs and aspirations. *Ambio* 33: 513–519.

West, P., Igoe, J. and Brockington, D. 2006. Parks and peoples: the social impact of protected areas. *Annual Review of Anthropology* 35: 251–277.

WWF. 2010. *Plan de Gestion pour la Nouvelle Aire Protégée Ranobe PK32*. WWF, Antananarivo.

Yonariza and Webb, E.L. 2007. Rural household participation in illegal timber felling in a protected area of West Sumatra, Indonesia. *Environmental Conservation* 34: 73–82.

Chapter 9

Discussion

This thesis has taken an interdisciplinary approach to a complex practical problem - how to integrate conservation and development in Madagascar's rapidly-expanding protected area system. I have investigated what drives rural people to use natural resources from within new protected areas, and explored what happens to biodiversity when such use occurs. In addition, I have examined the establishment of new protected areas at a nationwide level to generate insights into important policy questions, including how we classify management approaches and their influence in poverty alleviation. Finally, I developed a simple but versatile index to enumerate the conservation value of species, which can be used to refine and add depth to site prioritisations and evaluations of community-level differences between habitats. Through the seven chapters I have made a range of contributions to current knowledge, enhancing the theory and practice of protected areas in Madagascar and worldwide. Below, I discuss the key findings in relation to two major themes concerning; i) whether conservation and development really can be reconciled in Madagascar's multiple-use protected areas, and; ii) the role of conservation science in achieving these disparate goals.

9.1 Reconciling conservation and development in new protected areas: can we have our cake and eat it?

Madagascar's Durban Vision has been a groundbreaking conservation opportunity. The lobbying of conservationists and funders bore huge dividends, the required political will was stimulated, and the necessary legal frameworks were created to expand the protected area system to cover most of the remaining natural habitat (Corson 2014). However, the vision was also extremely ambitious, seeking not only to conserve the country's unique biodiversity and cultural heritage, but also to promote the sustainable use of natural resources from within the expanded protected area system to help alleviate poverty and improve the development potential of rural communities living within and around its constituent sites. Given that human impacts are the principal driver of biodiversity loss, both in Madagascar and around the world, these goals can be considered divergent at best, and directly conflicting at worst. Therefore, can Madagascar's biodiversity be conserved and used simultaneously? This thesis provides a range of insights into the feasibility and suitability of the Durban Vision programme, its multiple objectives, and models of implementation.

The principal objective of Madagascar's expanded protected area system, as it must be for all protected areas according to the IUCN definition, is to ensure the conservation of biodiversity. However, three of the chapters in this thesis cast doubt on the ability of multiple-use models to do so, at least in the Malagasy context. Chapter 3 investigated the appropriateness of multiple-use protected area categories to Madagascar on the basis of the people-nature interactions that characterise many new sites, and found that traditional land- and resource-use typically had negative effects that diminished the biodiversity that the protected areas were established to conserve. A literature review of the impacts of anthropogenic habitat change (Chapter 4) lent weight to this suggestion, finding that habitat changes resulting from forest use generally had negative impacts on biodiversity and, in particular, the more highly-specialised forest-dwelling endemics. In Chapter 6, I found that human livelihood activities permitted in new protected areas, such as charcoal production, cause habitat degradation that triggers changes in the composition of faunal communities. Once again, it is the endemic, habitat specialist component of the fauna that is disproportionally vulnerable and thus the overall conservation value of these sites is reduced. Examining the evidence across these chapters, it appears that multiple-use protected areas, in which the extractive use of natural resources is permitted, may be sub-optimal from a pure biodiversity conservation point of view and potentially unable to conserve the full complement of Madagascar's endemic biota into the future. This conclusion may be specific to Madagascar and other islands/ecosystems in which most species are forest specialists, since multiple-use protected areas in continents may conserve complementary communities (i.e. species adapted to more open habitats) to those maintained by stricter sites (Gardner *et al*. 2007).

On the other hand, the establishment of protected areas is a political decision that must trade-off the needs of other stakeholders and competing land-uses (DeFries *et al*. 2004). Most remaining forests, wetlands and shallow seas in Madagascar are surrounded by human populations that depend, to varying extents and in differing ways, on the natural resources these areas provide for their subsistence and income. In Ranobe PK32, for example, hunting provides a supplement to people's diets which increases in importance during times of scarcity, while charcoal production represents a gap-filler during the agricultural off-season and safety net when other, non-forest activities are insufficiently productive (Chapters 7 and 8). Both help prevent local communities from slipping further into destitution. Thus, the establishment of strict protected areas in inhabited landscapes would result in diminished access to critical resources that would exacerbate poverty, an outcome that is unethical, politically unfeasible, and unacceptable to conservationists given worldwide calls, and policies, for protected areas to contribute to poverty alleviation. If protected areas are to be established in landscapes that are important to both conservation and the wellbeing of local people, multiple-use models may be the only available option.

The Madagascar National Parks sub-network may be more valuable for the conservation of biodiversity than the Durban Vision generation of new protected areas, at least if the reptiles of western and southern Madagascar are indicative (Chapter 5). Furthermore, the relative importance of the strict protected areas may increase, if they are effectively managed, because multiple-use sites may harbour an extinction debt (Kuussaari *et al*. 2009) that will see them fail to maintain populations of sensitive species as their habitats are further modified (see Chapters 4 and 6). Given that the promoters of new protected areas lack the necessary funds to ensure the effective management of their sites in the long-term (AGRECO 2012), and the fact that conservation goals can be best achieved by investing in the most important protected areas (Fuller *et al*. 2010), triage arguments (Bottrill *et al*. 2008) would suggest that scarce resources are preferentially invested in the MNP network. However, there is little evidence that Madagascar's strict protected areas are effective at preventing deforestation (Allnutt *et al*. 2013; Whitehurst *et al*. 2009), logging (Patel 2007; Randriamalala and Liu 2010) or hunting (Garcia and Goodman 2003; Golden *et al*. 2014). Furthermore, the pre-2003 network was insufficient to ensure coverage of all species (Chapter 5; Jenkins *et al*. 2014; Rasoavahiny *et al*. 2008), meaning that new protected areas had to be established if all of Madagascar's endemic biota was to be conserved, in line with SAPM objectives.

If, therefore, new protected areas should be created and follow multiple-use models, how can they be effectively managed to meet their divergent goals? The objectives of the expanded protected area system specify a mechanism – the sustainable use of natural resources. However, the existing literature calls into question the underlying premise and viability of such an approach. It is widely reported that a dependence on low-value natural resources may prevent people from escaping poverty (Angelsen and Wunder 2003;

Levang *et al*. 2005; Vira and Kontoleon 2010), while Sayer (2009) cautions that development is rarely achieved through marginal improvements in existing livelihoods, and that conservationists "should not focus on what the poor are doing now but on what they might do in future in growing economies". Consequently, an understanding of how to capitalise on economic development opportunities is important in planning protected area management strategies. As illustrated by the case studies in Chapter 2, this depends partly on the available resource base.

Resources such as fish and octopus are rapidly renewable, and highly responsive to management. Thus, the interests of conservationists and user communities are closely aligned, both benefitting from healthy, productive ecosystems. The key natural resources in terrestrial ecosystems, however, are trees and the soil itself (for cultivation), which are difficult to manage sustainably and profitably because they are slow to replenish and exploited destructively (Gardner *et al*. 2013; Pollini *et al*. 2014). Thus, while protected area managers in freshwater and marine ecosystems have concentrated on improved fisheries management to underpin development, the managers of terrestrial sites have sought to decouple development from natural resource use by promoting improved productivity and/or profitability of non-forest based activities, thereby reducing the relative attractiveness of destructive practices. Chapter 7 provides evidence that efforts to improve the revenues generated outside of forests, for example through the development of agricultural infrastructure, would reduce the attractiveness of livelihoods incompatible with conservation goals. Likewise, Chapter 8 suggests that development interventions aimed at reducing forest dependence would also serve to reduce the prevalence of hunting, which is carried out as a secondary activity alongside other types of forest use.

Overall, multiple lines of reasoning indicate that the continued, unsustainable use of natural resources of low economic value will neither conserve biodiversity nor lead to rural development, at least in forest areas. Accordingly, it is unlikely to form a robust basis for the management of Madagascar's new protected areas in the long-term. However, because such use is so vital to rural communities in the present, it must persist while appropriate development strategies for these sites are designed and implemented. Thus, the management focus on sustainable resource use should perhaps best be considered as a temporal bridge, a safety net for rural communities to use until development actions designed to reduce dependence on forest products are implemented and come to fruition. However, this vision for the new protected areas is not a policy goal and does not appear to have been articulated in SAPM documentation. Rather, like protected areas worldwide, new protected areas in Madagascar have been conceived as static entities (Bengtsson *et al*. 2003; Folke *et al*. 2005; Mascia and Pailler 2012). The notion of individual protected areas evolving their management approaches through time in response to social change they themselves catalyse is not current, but may become a widespread in Madagascar as the Durban Vision protected areas mature. The complexity of the challenge facing managers is illustrated by the fact that protected areas of the type that characterise the new generation, in which activities that diminish their conservation values are nevertheless permitted (at least at early stages of implementation), are not even recognised by the IUCN's category system (Gardner 2011).

The successful management of the expanded protected area system in the long-term will depend on reconciling the divergent needs of varied stakeholders. It is a complex and high -stakes responsibility, with the wellbeing of millions of people likely to be affected and the fates of thousands of species on the line. Achieving it will require both enormous investment and a robust evidence base derived from multiple fields of enquiry. This, in turn, will require conservation science to enhance its contribution to protected area management.

9.2 Protected areas and conservation science: bridging the researcher-practitioner divide

Biodiversity conservation requires a robust evidence base if it is to be effective (Adams and Sandbrook 2013; Pulin and Knight 2009; Sutherland *et al*. 2004). However, many practitioners continue to be guided by anecdote, experience and intuition rather than empirical evidence (Cook *et al*. 2010, 2012; Laurance *et al*. 2012; Pullin and Knight 2005; Pullin *et al*. 2004). Although the field of conservation science developed in order to provide a scientific foundation for biodiversity conservation (Meine *et al*. 2006), the majority of conservation research is of only limited use to practitioners (Knight *et al*. 2008; Milner-Gulland *et al*. 2010, 2012; Whitten *et al*. 2001), and we know little about how to manage protected areas effectively (Cabeza 2013; Geldmann *et al*. 2013). This thesis has generated a range of insights into the role of science in protected area-based conservation.

9.2.1 Interdisciplinary research is indispensible

Although conservation science has been recognised as an interdisciplinary endeavour for many years (Dailly and Ehrlich 1999; Mascia *et al*. 2003), most research published in peer-reviewed conservation journals continues to be focused of ecological rather than social problems and phenomena (Fazey *et al*. 2005). This is despite the fact that our principal conservation strategy, protected areas, are human constructs created as part of complex social-ecological systems (Milner-Gulland 2012; Ostrom 2009). The success of protected areas therefore depends both on their acceptance by other stakeholders (Borrini-Feyerabend 2002; Western 2001, although see Brockington 2004 for a counter-narrative) and their ability to buffer the species and ecosystems they harbour from the human activities and impacts that threaten them (Gaston *et al*. 2008a). This thesis has highlighted the need for protected area-related research to adopt social scientific tools and approaches, and focus on the human dimensions of management.

Understanding human-nature interactions is fundamental to protected area management, since the outcomes of the relationship are a key determinant in deciding appropriate management categories (Chapter 3) and in developing poverty alleviation and development strategies (Chapter 2). Thus the specific approach of any protected area should not be determined until managers have sufficient understanding of the social context. However, the creation of protected areas tends to be decided from the top-down, and promoters may often have pre-conceived ideas of the models and strategies they intend to implement before conducting the necessary research (Corson 2014; Marie *et al*. 2009). Although this tendency was known to contribute to the failings of integrated conservation and development projects a decade ago (Brown 2003; Wells and Mcshane 2004), it remains persistent.

While social scientific research is necessary for planning broad-brush management approaches, it is equally important in the development of specific actions on the ground. In particular, an understanding of the role played by forest-based livelihood activities in the lives of rural people can be used to design interventions to reduce the prevalence of destructive activities without imposing costs on the resource users in question. The research in Chapter 8, for example, suggested that restoration of irrigation infrastructure would permit around 30 % of sampled charcoal producers to return to their favoured livelihood of farming, an outcome that represents a clear win-win for both conservationists and the communities involved. Such research is fundamental if the multiple goals of new protected areas are to be met, and is likely to become ever more important in future as rural communities adapt their livelihood portfolios to climate change (Bradley *et al*. 2012; Watson 2014). Lastly, the expansion of protected area monitoring protocols to incorporate social systems (e.g, indicators of livelihoods and/or natural resource use trends) could allow managers to detect and react to emerging threats at an early stage, rather than waiting until their impacts become apparent (cf. Caro *et al*. 2013). Thus, both research and monitoring should expand to incorporate social systems.

9.2.2 Management is hindered by a researcher-practitioner divide

This thesis has taken an academic approach to a practical problem of managing protected areas. It is therefore no surprise that the researcher-practitioner divide emerged as a theme throughout the research, although it manifested itself in various ways. We know little about the factors that influence the success or otherwise of protected area management strategies in different contexts. Providing managers with the evidence base they need to develop appropriate and effective strategies will require a greater emphasis on problem-solving research, and in particular a focus on local rather than global scale analyses (Gardner *et al*. 2013). Academic researchers, unfortunately, have different agendas and institutional incentives to protected area managers, and thus limited interest in producing the types of research that the latter need. Researchers are traditionally not rewarded for their contribution to conservation results (Arlettaz *et al*. 2010; Chapron and Arlettaz 2008), rather they are judged on the academic impact of their work (although funding bodies are increasingly encouraging research with societal impact, for example the Research Excellent Framework carried out by the Higher Education Funding Council for England in 2014 was the first to assess the impact of research on 'the economy, society, culture, public policy or services, health, the environment or quality of life, beyond academia' (REF 2012). For research to be of high academic impact, however, it must demonstrate sufficient novelty or be global, rather than local, in scope (Cook *et al*. 2013; Hulme 2011; Meffe *et al*. 2006). Furthermore, researchers may be wary of investigating the messy complexities of practical conservation problems which do not fit into robust experimental designs, and more interested in developing and testing hypotheses that push the boundaries of knowledge than investigating real-world problems (Knight *et al*. 2008; Laurance *et al*. 2012; Putz and Zuidema 2008). Managers, on the other hand, must balance their information requirements with the need to act despite uncertainty (Cook *et al*. 2013; Knight *et al*. 2010; Soulé 1985), and may therefore favour simple, rapid methods to generate viable data over elegant, sophisticated and innovative methodological designs and protocols (if they use an evidence base at all).

In addition to aspects of scale and novelty, a further difference between practitioners and academic researchers lies in the types of question they tend to ask (Braunisch *et al*. 2012). Most research on bushmeat, for example, has been quantitative and focused on calculating extraction and/or consumption rates. While such studies inform us of the urgency of the problem and may be important in persuading stakeholders to act, they do not necessarily tell us much about how to reverse the observed trends. Understanding the drivers of hunting and bushmeat consumption, as with all forms of resource use, is essential for the development of management strategies, yet this is rarely the objective of published bushmeat research (Kümpel *et al*. 2009; Pailler *et al*. 2009). A similar phenomenon is widespread in conservation social science, which is characterised by a divide between research *on* conservation and research *for* conservation (Sandbrook *et al*. 2013). Protected areas are fascinating social and political phenomena, often pitting powerful global forces against powerless rural resource users (Brockington 2004), and are therefore the focus of much anthropology and political ecology research (Agrawal and Ostrom 2006; Brosius 2006; West and Brockington 2006). While studies of the political processes and social impacts of conservation interventions can stimulate important self-reflection by conservationists and contribute to improved practices, particularly if they are critical in approach, they are not designed to inform effective management. As highlighted in Chapter 8, social changes triggered by protected area creation have been the subject of much research (e.g. Brockington and Igoe 2006; Ghimire and Pimbert 1997; Mascia and Claus 2008; Miller *et al*. 2012; Schmitz *et al*. 2012; West *et al*. 2006), but evolving livelihoods themselves can have a great impact on protected area effectiveness and this has attracted far less research attention.

The researcher-practitioner divide is not a one-way street and managers themselves perpetuate it in various ways. For example, they fail to sufficiently share their experiences or engage researchers, and thus miss the opportunity to promote practitioner-relevant research agendas amongst those in a position to contribute to them (Fazey *et al*. 2006; Gardner 2014; Gardner *et al*. 2013; Sunderland *et al*. 2009). Moreover, practitioners do not always, or even often, incorporate research findings into their decision-making (Cook *et al*. 2010; Gossa *et al*. 2014; Matzek *et al*. 2014; Pullin and Knight 2005), at least in part because they cannot access it (less than 5% of conservation research is open-access;

Fuller *et al*. 2014) and because it is not available in synthesised, easily understandable form (Walsh *et al*. 2014).

Beyond research findings, researchers also develop tools and techniques of great potential utility to practitioners. However, as in the case of systematic conservation planning software, these may be effectively beyond the reach of managers and decision-makers because they lack the necessary capacity and expertise (Bottrill and Pressey 2012; Gaston *et al*. 2008b; Smith *et al*. 2006). As a result, high-profile prioritisation exercises are often performed without a systematic basis or using only simple, heuristic methods (e.g. Schwitzer *et al*. 2013, 2014). This aspect of the divide can be bridged by promoting greater collaboration between researchers and practitioners (Knight *et al*. 2011; Smith *et al*. 2009), or by encouraging practitioners to adopt systematic approaches by developing more user-friendly methods, such as the conservation value index (CVI) tested in Chapter 5.

Academic researchers, conservation practitioners, research institutions, NGOs, funders and publishers all have a role to play in bridging the researcher-practitioner divide and ensuring that conservation science contributes as much as possible to the maintenance of biodiversity. An important first step is for those at the forefront of conservation practice to collaborate with academics in order to develop appropriate research agendas and ensure that their information needs are known (Sutherland *et al*. 2009). Academic researchers have made great advances in this regard over recent years, contributing to several fields of conservation, but the science of protected area management lags behind.

9.2.3 Reconciling conservation and development in new protected areas: a research agenda

An increasing appreciation of the researcher-practitioner divide and its impacts on biodiversity has spurred a desire to more closely align research agendas with the needs of practitioners and policy-makers in recent years. The principal outputs of this surge of activity have been a range of high-profile research prioritisations, known as the "100 questions exercises" (Cooke *et al*. 2010), which seek to generate lists of the research questions which, if answered, would make the greatest contributions to conserving biodiversity (Sutherland *et al*. 2011). They have been carried out for specific countries (e.g. the UK, Sutherland *et al*. 2006; USA, Fleischman *et al*. 2011; Canada, Rudd *et al*. 2011; Australia, Morton *et al*. 2009), for particular biomes (e.g. Parsons *et al*. 2014) or economic sectors (Pretty *et al*. 2010), and to improve the science-policy interface (Sutherland *et al*. 2012). The most influential of the prioritisation exercises has been Sutherland *et al*. (2009), which generated a list of 100 questions most important to the conservation of global biodiversity. Astonishingly, however, the list included only four questions about protected areas, and not a single question about how they can be most effectively managed, even though they constitute our principal conservation strategy. The findings of this thesis suggest a number of research areas that may provide fruitful topics for investigation while contributing to the successful management of multiple-use protected areas in Madagascar and worldwide.

Evaluating effectiveness of different interventions

Successful protected area management requires an evidence-base, but we know little about the effectiveness of different management strategies and the conditions that affect their performance (Gardner *et al*. 2013; Geldmann *et al*. 2013). To better contextualise the size and importance of this knowledge gap, I propose a thought experiment.

Imagine we are an organisation charged with conserving a patch of forest in a tropical developing country. The forest is biodiversity rich, but is also important to the human communities surrounding it, who destructively extract a variety of forest products and clear farmland around the edge. A range of approaches and interventions are open to us as managers. In terms of legal structures, we could lobby to install a strict protected area and invest in rule enforcement, or create a multiple-use site that could be co-managed by local communities. Alternatively, we could forego the protected area approach altogether and advocate recognition of local customary land tenure, in the hope that security in the form of land ownership encourages sustainable management (Deacon 1999; FAO 2002; Sunderlin 2005). In terms of actions, we could promote forest-based livelihoods programmes to improve returns from non-timber forest products, invest in 'off-site' activities (e.g. agricultural improvements) to reduce people's dependence on forests altogether, or implement an incentive-based programme (such as direct payments) encouraging resource users to forego destructive activities. We could also invest in more indirect conservation strategies, such as family planning and education, in a bid to promote development and diminish future demand for forest resources. The key question for protected area managers is straightforward – 'what works, and in what contexts'? Unfortunately, we know little about the costs and effectiveness of different conservation actions (Wilson *et al*. 2007).

A range of research approaches can contribute to filling this critical knowledge gap. Firstly, the increasing power of remote detection and sophistication of statistical analyses (e.g. matching techniques) permits global analyses that can answer these questions at a broad scale (e.g. Ferraro *et al*. 2011; Nelson and Chomitz 2011). In addition, managers themselves possess a store of valuable information on what works, and when. Although anecdotal in nature, this experiential knowledge can be a valuable information source if we expand our definitions of what constitutes 'evidence' and find ways to synthesise and communicate it (Adams and Sandbrook 2013; Fazey *et al*. 2006; Gardner 2014). Most importantly, however, protected area managers must adopt a more experimental approach to implementation, and systematically incorporate mechanisms to empirically evaluate the outcomes of their actions (Ferraro and Pattanayak 2006; Geldmann *et al*. 2013; Sutherland *et al*. 2013). This will require a change in mindset, and we must make efforts to learn from, rather than hide, failure (Cressey 2009; Sayer 2009).

Minimising impacts and maximising ecological persistence

Some human uses of natural resources have greater impacts on biodiversity than others (Barlow *et al*. 2007; Gardner 2009; Gardner *et al*. 2009). Thus, research is necessary to identify and design systems of resource use that ensure the sustainable generation of goods and services while minimising negative effects on biodiversity. Firstly, studies to inform the optimal spatial configuration of strict protected zones within multiple-use landscapes are necessary to ensure metapopulation persistence of key species and maintenance of evolutionary processes (Carroll *et al*. 2004; Chazdon *et al*. 2009). In addition, the impacts of harvesting may be minimised through the implementation of reduced-impact extraction systems. For such arrangements to be effective, we need to know which species are most and least resilient to high levels of offtake, and whether there are particular seasons/life-history stages where the impacts of harvesting can be minimised. Furthermore, there may be variation in impacts with regards to extraction method (e.g. selective or non-selective logging/trapping). Impacts on species may be non-linear, thus it is important to identify thresholds if sustainable extraction quotas are to be set.

Meeting the needs of multiple stakeholders

Ensuring the long-term viability of protected areas requires that they generate benefits to wider society, whether or not the objectives of individual sites explicitly include social goals such as development or poverty alleviation. As demonstrated in Chapters 7 and 8, research to understand the drivers of livelihood decision-making can inform the development of socially and ecologically appropriate strategies, and should be carried out to investigate all local livelihood practices that, if not managed, may threaten protected area integrity.

Given that protected areas are threatened as much, in the long term, by processes occurring in their surrounding landscapes as by activities occurring within their boundaries (DeFries *et al*. 2005), managers should seek to implement actions that will diminish future pressures on their sites. This will require the development of sustainable land management programmes at a landscape scale, to ensure that local populations are able to meet their future needs without increasing dependence on natural resources. The required research must go well beyond traditional conservation science, incorporating elements of agronomy, development studies, socio-cultural systems and resource economics. In addition, it should be explicitly solutions-oriented and stakeholder-led or, at very least, highly participatory to ensure that recommendations are culturally and contextually appropriate.

Negotiating trade-offs and prioritising interventions

As unpalatable as the idea may be, we cannot conserve all of biodiversity given available resources and the competing needs of other stakeholders. Therefore, we must rethink our objectives and prioritise our interventions, and such processes must be informed by sound science. Mechanisms to negotiate trade-offs, both internally and with other stakeholders, must be further refined and mainstreamed (Hirsch *et al*. 2010; McShane *et al*. 2010).

Madagascar's expanded protected area system has multiple objectives, but management cannot be simultaneously optimised towards disparate goals. While biodiversity and socio -economic targets may require that different types of intervention be prioritised, the goal of resilience to climate change may suggest another form of management altogether in future (Groves *et al*. 2012; Hannah 2008; Olson *et al*. 2009). Therefore, mechanisms and methods to help us reflect upon and elucidate our values as conservation actors, organisations and funders will be required if we are to systematically target our interventions towards robustly negotiated outcomes rather than vague notions of 'conserving biodiversity'.

Finally, we must ensure that we integrate the tools and approaches of conservation science as much as possible into the messy real-life world of conservation decision-making. It could be argued that the Durban Vision was too ambitious, since some legally-established protected areas receive no management due to insufficient resources. For instance, Ranobe PK32 has effectively been a paper park since 2013 due to the capricious nature of NGO funding cycles. Given substantial variation in the conservation value of Madagascar's protected area estate (Chapter 5), conservation goals may have been better met by preferentially investing available resources in the most important sites, rather than seeking to create as many new protected areas as possible (Fuller *et al*. 2010). Conservationists must accept the reality of triage (Bottrill *et al*. 2008) and ensure that scientific tools, rather than personal or institutional values, are used to help navigate the difficult decisions that this will entail. High-profile, real-world decisions that determine which of Madagascar's protected areas will receive funding are still being made without an adequate evidence base (Schwitzer *et al*. 2013, 2014), even when the necessary data and tools are readily available. That such decision-making occurs highlights the failure of conservation practice to make best use of the tools and knowledge available, and of conservation science to integrate itself into the mainstream of conservation action.

9.3 Conclusions

Madagascar's biodiversity is immensely valuable, and so global society has invested hundreds of millions of dollars to ensure its conservation (Horning 2008). Over the last decade, these efforts have focused on tripling the coverage of the protected area system, but this has necessitated the development and implementation of new forms of protected area because the traditional, strict model would have been inappropriate in heavily-populated landscapes. Most new sites have therefore been established as multiple-use protected areas that aim to conserve biodiversity and promote its use for poverty alleviation. My research has shown that meeting both objectives is ambitious, because most forms of natural resource use negatively affect endemic biodiversity. However, understanding the drivers of local livelihood strategies can help inform the design of ecologically- and socially-appropriate conservation strategies. Conservation science has an essential role to play if the twin goals are to be met and conservation and development reconciled.

References

Adams, W.A. and Sandbrook, C. 2013. Conservation, evidence and policy. *Oryx* 47: 329–335.

Agrawal, A. and Ostrom, E. 2006. Political science and conservation biology: a dialog of the deaf. *Conservation Biology* 20: 681–682.

AGRECO. 2012. *Analyse des Coûts et Sources de Financement du Système des Aires Protégées de Madagascar (Octobre 2010 – Janvier 2012)*. AGRECO, Antananarivo.

Allnutt, T.F., Asner, G.P., Golden, C.D. and Powell, G.V.N. 2013. Mapping recent deforestation and forest disturbance in northeastern Madagascar. *Tropical Conservation Science* 6: 1–15.

Angelsen, A. and Wunder, S. 2003. *Exploring the Poverty-Forestry Link: Key Concepts, Issues and Research Implications*. Center for International Forestry Research, Bogor.

Arlettaz, R., Schaub, M., Fournier, J., Reichlin, T.S., Sierro, A., Watson, J.E.M. and Braunisch, V. 2010. From publications to public actions: when conservation biologists bridge the gap between research and implementation. *BioScience* 60: 835–842.

Barlow, J., Gardner, T.A., Araujo, I.S., Avila-Pires, T.C., Bonaldo, A.B., Costa, J.E., Esposito, M.C., Ferreira, L.V., Hawes, J., Hernandez, M.I.M., *et al*. 2007. Quantifying the biodiversity value of tropical primary, secondary, and plantation forests. *Proceedings of the National Academy of Sciences, USA* 104: 18555–18560.

Bengtsson, J., Angelstam, P., Elmqvist, T., Emanuelsson, U., Folke, C., Ihse, M., Moberg, F. and Nyström, M. 2003. Reserves, resilience and dynamic landscapes. *Ambio* 32: 389–396.

Borrini-Feyerabend, G., Banuri, T., Farvar, T., Miller, K. And Phillips, A. 2002. Indigenous and local communities and protected areas: rethinking the relationship. *Parks* 12: 5–15.

Bottrill, M.C. and Pressey, R.L. 2012 The effectiveness and evaluation of conservation planning. *Conservation Letters* 5: 407–420.

Bottrill, M.C., Joseph, L.N., Cawardine, J., Bode, M., Cook, C., Game, E.T., Grantham, H., Kark, S., Linke, S., McDonald-Madden, E., *et al*. 2008. Is conservation triage just smart decision making? *Trends in Ecology and Evolution* 23: 649–654.

Bradley, B.A., Estes, L.D., Hole, D.G., Holness, S., Oppenheimer, M., Turner, W.R., Beukes, H., Schulze, R.E., Tadross, M.A. and Wilcove, D.S. 2012. Predicting how adaptation to climate change could affect ecological conservation: secondary impacts of shifting agricultural suitability. *Diversity and Distributions* 18: 425–437.

Braunisch, V. Home, R., Pellet, J. and Arlettaz, R. 2012. Conservation science relevant to conservation action: a research agenda identified and prioritized by practitioners. *Biological Conservation* 153: 201–210.

Brockington, D. 2004. Community conservation, inequality and injustice: myths of power in protected area management. *Conservation and Society* 2: 411–432.

Brockington, D. and Igoe, J. 2006. Eviction for conservation: a global overview. *Conservation and Society* 4: 424–470.

Brosius, J.P. 2006. Common ground between anthropology and conservation biology. *Conservation Biology* 20: 683–685.

Brown, K. 2003. Three challenges for real people-centred conservation. *Global Ecology & Biogeography* 12: 89–92.

Cabeza, M. 2013. Knowledge gaps in protected area effectiveness. *Animal Conservation* 16: 381–382.

Caro, T., Elisa, M., Gara, J., Kadomo, D., Martin, A., Mushi, D. And Timbuka, C. 2013. Integrating research with management: the case of Katavi National Park, Tanzania. *African Zoology* 48: 1–12.

Carroll, C., Noss, R.F., Paquet, P.C. and Schumaker, N.H. 2004. Extinction debt of protected areas in developing landscapes. *Conservation Biology* 18: 1110 –1120.

Chapron, G. and Arlettaz, R. 2008. Conservation: academics should 'conserve or perish'. *Nature* 451: 127–127.

Chazdon, R.L., Peres, C.A., Dent, D., Sheil, D., Lugo, A.E., Lamb, D., Stork, N.E. and Miller, S.E. 2009. The potential for species conservation in tropical secondary forests. *Conservation Biology* 23: 1406–1417.

Cook, C.N., Hockings, M. and Carter, R.W. 2010. Conservation in the dark? The information used to support management decisions. *Frontiers in Ecology and the Environment* 8: 181–186.

Cook, C.N., Carter, R.W., Fuller, R.A. and Hockings, M. 2012. Managers consider multiple lines of evidence important for biodiversity management decisions. *Journal of Environmental Management* 113: 341–346.

Cook, C.N., Mascia, M.B., Schwartz, M.W., Possingham, H.P. and Fuller, R.A. 2013. Achieving conservation science that bridges the knowledge-action boundary. *Conservation Biology* 27: 669–678.

Cooke, S.J., Danylchuk, A.J., Kaiser, M.J. and Rudd, M.A. 2010. Is there a need for a '100 questions exercise' to enhance fisheries and aquatic conservation, policy, management and research? Lessons from a global 100 questions exercise on conservation of biodiversity. *Journal of Fish Biology* 76: 2261–2286.

Corson, C. 2014. Conservation politics in Madagascar: the expansion of protected areas. In *Conservation and Environmental Management in Madagascar*, I.R. Scales (Ed.), pp. 193–215. Routledge, London and New York.

Cressey, D. 2009. Failure is certainly an option. *Nature*: doi:10.1038/news.2009.511

Daily, G.C., and Ehrlich, P.R. 1999. Managing Earth's ecosystems: an interdisciplinary challenge. *Ecosystems* 2: 277–280.

Deacon, R. 1999. Deforestation and ownership: evidence from historical accounts and contemporary data. *Land Economics* 75: 341–359.

DeFries, R., Foley, J.A. and Asner, G.P. 2004. Land-use choices: balancing human needs and ecosystem function. *Frontiers in Ecology and the Environment* 2: 249–257.

DeFries, R., Hansen, A., Newton, A.C. and Hansen, M.C. 2005. Increasing isolation of protected areas in tropical forests over the past twenty years. *Ecological Applications* 15: 19–26.

FAO (Food and Agriculture Organisation of the United Nations). 2002. *Land Tenure and Rural Development*. FAO, Rome.

Fazey, I., Fischer, J. and Lindenmayer, D.B. 2005. What do conservation biologists publish? *Biological Conservation* 124: 63–173.

Fazey, I., Fazey, J.A., Salisbury, J.G., Lindenmayer, D.B. and Dovers, S. 2006. The nature and role of experiential knowledge for environmental conservation. *Environmental Conservation* 33: 1–10.

Ferraro, P.J. and Pattanayak, S.K. 2006. Money for nothing? A call for empirical evaluation of biodiversity conservation investments. *PLoS Biology* 4: e105.

Ferraro, P.J., Hanauer, M.M. and Sims, K.R.E. 2011. Conditions associated with protected area success in conservation and poverty reduction. *Proceedings of the National Academy of Sciences, USA* 108: 13913–13918.

Fleishman, E., Blockstein, D., Hall, J.A., Mascia, M., Rudd, M.A., Scott, J.M., Sutherland, W.J., Baldwin, T., Bartuska, A.M., Brown, A.G., *et al*. 2011. America's top 40 priorities for policy-relevant conservation science. *BioScience* 61: 290–300.

Folke, C. 2003. Reserves and resilience – from single equilibrium to complex systems. *Ambio* 32: 379–379.

Fuller, R.A., McDonald-Madden, E., Wilson, K.A., Cawardine, J., Grantham, H.S., Watson, J.E.M., Klein, C.J., Green, D.C. and Possingham, H.P. 2010. Replacing underperforming protected areas achieves better conservation outcomes. *Nature* 466: 365 –367.

Fuller, R.A., Lee, J.R. and Watson, J.E.M. 2014. Achieving open access to conservation science. *Conservation Biology*: doi: 10.1111/cobi.12346.

Garcia, G. and Goodman, S.M. 2003. Hunting of protected animals in the Parc National d'Ankarafantsika, north-western Madagascar. *Oryx* 37: 115–118.

Gardner, C.J. 2009. A review of the impacts of anthropogenic habitat change on terrestrial biodiversity in Madagascar: implications for the design and management of new protected areas. *Malagasy Nature* 2: 2–29.

Gardner, C.J. 2011. IUCN management categories fail to represent new, multiple-use protected areas in Madagascar. *Oryx* 45: 336–346.

Gardner, C.J. 2014. Publishing for conservation and development? *Madagascar Conservation & Development* 9: 3–4.

Gardner, C.J., Nicoll, M.E., Mbohoahy, T., Olesen, K.L.L., Ratsifandrihamanana, A.N., Ratsirarson, J., René de Roland, L.-A., Virah-Sawmy, M., Zafindrasilivonona, B. and Davies, Z.G. 2013. Protected areas for conservation and poverty alleviation: experiences from Madagascar. *Journal of Applied Ecology* 50: 1289–1294.

Gardner, T.A., Caro, T., Fitzherbert, E.B., Banda, T. and Lalbhai, P. 2007. Conservation value of multiple-use areas in East Africa. *Conservation Biology* 21: 1516–1525.

Gardner, T.A., Barlow, J., Chazdon, R., Ewers, R.M., Harvey, C.A., Peres, C.A. and Sodhi, N.S. 2009. Prospects for tropical forest biodiversity in a human-modified world. *Ecology Letters* 12: 561–582.

Gaston, K.J., Jackson, S.F., Cantu-Salazar, L. and Cruz-Pinon, G. 2008a. The ecological performance of protected areas. *Annual Review of Ecology, Evolution, and Systematics* 39: 93–113.

Gaston, K.J., Jackson, S.F., Nagy, A., Cantú-Salazar, L. and Johnson, M. 2008b. Protected areas in Europe: principle and practice. *Annals of the New York Academy of Sciences* 1134: 97–119.

Geldmann, J., Barnes, M., Coad, L., Craigie, I.D., Hockings, M. and Burgess, N. 2013. Effectiveness of terrestrial protected areas in reducing habitat loss and population declines. *Biological Conservation* 161: 230–238.

Ghimire, K.B. and Pimbert, M.P. (Eds.) 1997. *Social Change and Conservation*. Earthscan, London.

Golden, C.D., Bonds, M.H., Brashares, J.S., Raolofoniaina, B.J.R. and Kremen, C. 2014. Economic valuation of subsistence harvest of wildlife in Madagascar. *Conservation Biology* 28: 234–243.

Gossa, C., Fisher, M. and Milner-Gulland, E.J. 2014. The research-implementation gap: how practitioners and researchers from developing countries perceive the role of peer-reviewed literature in conservation science. *Oryx*: doi:10.1017/S0030605313001634.

Groves, C.R., Game, E.T., Anderson, M.G., Cross, M., Enquist, C., Ferdaña, Z., Girvets, E., Gondor, A., Hall, K.R., Higgins, J., *et al*. 2012. Incorporating climate change into systematic conservation planning. *Biodiversity and Conservation* 21: 1651–1671.

Hannah, L. 2008. Protected areas and climate change. *Annals of the New York Academy of Sciences* 1134: 201–212.

Hirsch, P.D., Adams, W.M., Brosius, P.J., Zia, A., Bariola, N. and Dammert, J.L. 2011. Acknowledging conservation trade-offs and embracing complexity. *Conservation Biology* 25: 259–264.

Horning, N.R. 2008. Strong support for weak performance: donor competition in Madagascar. *African Affairs* 107: 405–431.

Hulme, P.E. 2011. Practitioner's perspectives: introducing a different voice in applied ecology. *Journal of Applied Ecology* 48: 1–2.

Jenkins, R.K.B., Tognelli, M.F., Bowles, P., Cox, N., Brown, J.L., Chan, L., Andreone, F., Andriamazava, A., Andriantsimanarilafy, R.R., Anjeriniaina, M., *et al*. 2014. Extinction risks and the conservation of Madagascar's reptiles. *PLoS ONE* 9: e100173.

Knight, A.T., Cowling, R.M., Rouget, M., Balmford, A., Lombard, A.T. and Campbell, B.M. 2008. Knowing but not doing: selecting priority conservation areas and the research -implementation gap. *Conservation Biology* 22: 610–617.

Knight, A.T., Bode, M., Fuller, R.A., Grantham, H.S., Possingham, H.P., Watson, J.E.M. and Wilson, K.A. 2010. Barometer of life: more action, not more data. *Science* 329: 141–141.

Knight, A.T., Cowling, R.M., Boshoff, A.F., Wilson, S.L., Pierce, S.M. 2011. Walking in step: lessons for linking spatial prioritisations to implementation strategies. *Biological Conservation* 111: 202–211.

Kümpel, N.F., Rowcliffe, J.M., Cowlishaw, G. and Milner-Gulland, E.J. 2009. Trapper profiles and strategies: insights into sustainability from hunter behaviour. *Animal Conservation* 12: 531–539.

Kuussaari, M., Bommarco, R., Heikkinen, R.K., Helm, A., Krauss, J., Lindborg, R., Öckinger, E., Pärtel, M., Pino, J., Rodà, F., *et al*. 2009. Extinction debt: a challenge for biodiversity conservation. *Trends in Ecology & Evolution* 24: 564–571.

Laurance, W.F., Koster, H., Grooten, M., Anderson, A.B., Zuidema, P.A., Zwick, S., Zagt, R.J., Lynam, A.J., Linkie, M. and Anten, N.P.R. 2012. Making conservation research more relevant for conservation practitioners. *Biological Conservation* 153: 164–168.

Levang, P., Dounias, E. and Sitorus, S. 2005. Out of the forest, out of poverty? *Forests, Trees and Livelihoods* 15: 211–235.

Marie, C.N., Sibelet, N., Dulcire, M., Rafalimaro, M., Danthu, P. and Carrière, S.M. 2009. Taking into account local practices and indigenous knowledge in an emergency conservation context in Madagascar. *Biodiversity and Conservation* 18: 2759–2777.

Mascia, M.B. and Claus, C.A. 2008. A property rights approach to understanding human displacement from protected areas: the case of marine protected areas. *Conservation Biology* 23: 16–23.

Mascia, M.B. and Pailler, S. 2012. Protected area downgrading, downsizing, and degazettement (PADDD) and its conservation implications. *Conservation Letters* 4: 9–20.

Mascia, M.B., Brosius, J.P., Dobson, T.A., Forbes, B.C., Horowitz, L., McKean, M.A. and Turner, N.J. 2003. Conservation and the social sciences. *Conservation Biology* 17: 649–650.

Matzek, V., Covino, J., Funk, J.L. and Saunders, M. 2014. Closing the knowing-doing gap in invasive plant management: accessibility and interdisciplinarity of scientific research. *Conservation Letters* **7:** 208–215.

McShane, T.O., Hirsch, P.D., Trung, T.C., Songorwa, A.N., Kinzig, A., Monteferri, B., Mutekanga, D., Thang, H.V., Dammert, J.L., Pulgar-Vidal, M., *et al*. 2011. Hard choices: making trade-offs between biodiversity conservation and human well-being. *Biological Conservation* 144: 966–972.

Meffe, G.K., Ehrenfeld, D. and Noss, R.F. 2006. Conservation biology at twenty. *Conservation Biology* 20: 595–596.

Meine, C., Soulé, M. and Noss, R.F. 2006. “A mission-driven discipline”: the growth of conservation biology. *Conservation Biology* 20: 631–651.

Miller, B.W., Caplow, S.C. and Leslie, P.W. 2012. Feedbacks between conservation and social-ecological systems. *Conservation Biology* 26: 218–227.

Milner-Gulland, E.J. 2012. Interactions between human behaviour and ecological systems. *Philosophical Transactions of the Royal Society, B* 367: 270–278.

Milner-Gulland, E.J., Fisher, M., Browne, S., Redford, K.H., Spencer, M. and Sutherland, W.J. 2010. Do we need to develop a more relevant conservation literature? *Oryx* 44: 1–2.

Milner-Gulland, E.J., Barlow, J., Cadotte, M.W., Hulmer, P.E., Kerby, G. and Whittingham, M.J. 2012. Ensuring applied ecology has impact. *Journal of Applied Ecology* 49: 1–5.

Morton, S.R., Hoegh-Guldberg, O., Lindenmayer, D.B., Harriss Olson, M., Hughes, L., McCulloch, M.T., McIntyre, S., Nix, H.A., Prober, S.M., Saunders, D.A., *et al*. 2009. The big ecological questions inhibiting effective environmental management in Australia. *Austral Ecology* 34: 1–9.

Nelson, A. and Chomitz, K.M. 2011. Effectiveness of strict vs. multiple use protected areas in reducing tropical forest fires: a global analysis using matching methods. *PLoS ONE* 6: e22722.

Olson, D., O'Connell, M., Fang, Y.-C. and Burger, J. 2009. Managing for climate change within protected area landscapes. *Natural Areas Journal* 29: 394–399.

Ostrom, E. 2009. A general framework for analyzing sustainability of social-ecological systems. *Science* 325: 419–422.

Pailler, S., Wagner, J.E., McPeak, J.G. and Floyd, D.W. 2009. Identifying conservation opportunities among Malinké bushmeat hunters of Guinea, West Africa. *Human Ecology* 37: 761–774.

Parsons, E.C.M., Favaro, B., Aguirre, A.A., Bauer, A.L., Blight, L.K., Cigliano, J.A., Coleman, M.A., Côté, I.M., Draheim, M., Fletcher, S., *et al*. 2014. Seventy-one important questions for the conservation of marine biodiversity. *Conservation Biology* 28: 1206–1214.

Patel, E.R. 2007. Logging of rare rosewood and palisandre (*Dalbergia* spp.) within Marojejy National Park, Madagascar. *Madagascar Conservation & Development* 2: 11–16.

Pollini, J., Hockley, N., Muttenzer, F.D. and Ramamonjisoa, B.S. 2014. The transfer of natural resource management rights to local communities. In *Conservation and Environmental Management in Madagascar*, I.R. Scales (Ed.), pp. 172–192. Routledge, Abingdon.

Pretty, J., Sutherland, W.J., Ashby, J., Auburn, J., Baulcombe, D., Bell, M., Bentley, J., Bickersteth, S., Brown, K., Burke, J., *et al*. 2001. The top 100 questions of importance to the future of global agriculture. *International Journal of Agricultural Sustainability* 8: 219–236.

Pullin, A.S. and Knight, T.M. 2005. Assessing conservation management's evidence base: a survey of management-plan compilers in the United Kingdom and Australia. *Conservation Biology* 19: 1989–1996.

Pullin, A. and Knight, T.M. 2009. Doing more good than harm – building an evidence-base for conservation and environmental management. *Biological Conservation* 142: 931–934.

Pullin, A.S., Knight, T.M., Stone, D.A. and Charman, K. 2004. Do conservation managers use scientific evidence to support their decision-making? *Biological Conservation* 119: 245–252.

Putz, F.E. and Zuidema, P.A. 2008. Contributions of ecologists to tropical forest conservation. In *Tropical Forest Community Ecology*, W.P. Carson and S.A. Schnitzer (Eds.), pp. 474–489. Wiley-Blackwell, Oxford.

Randriamalala, H. and Liu, Z. 2010. Rosewood of Madagascar: between democracy and conservation. *Madagascar Conservation & Development* 5: 11–22.

Rasoavahiny, L., Andrianarisata, M., Razafimpahanana, A. and Ratsifandrihamanana, A.N. 2008. Conducting an ecological gap analysis for the new Madagascar protected area system. *Parks* 17: 12–21.

REF(Research Excellence Framework). 2014. *Assessment Framework and Guidance on Submissions*. Higher Education Funding Council for England, Bristol and London.

Rudd, M.A., Beazley, K.F., Cooke, S.J., Fleishman, E., Lane, D.E., Mascia, M.B., Roth, R., Tabor, G., Bakker, J.A., Bellefontaine, T., *et al*. 2010. Generation of priority research questions to inform conservation policy and management at a national level. *Conservation Biology* 25: 476–484.

Sandbrook, C., Adams, W.M., Büscher, B. and Vira, B. 2013. Social research and biodiversity conservation. *Conservation Biology* 27: 1487–1490.

Sayer, J. 2009. Can conservation and development really be integrated? *Madagascar Conservation & Development* 4: 9–12.

Schmitz, M.F., Matos, D.G.G., De Aranzabal, I., Ruiz-Labourdette, D. and Pineda, F.D. 2012. Effects of a protected area on land-use dynamics and socioeconomic development of local populations. *Biological Conservation* 149: 122–135.

Schwitzer, C., Mittermeier, R.A., Davies, N., Johnson, S., Ratsimbazafy, J., Razafindramanana, J., Louis Jr., E.E. and Rajaobelina, S. 2013. *Lemurs of Madagascar: a Strategy for their Conservation 2013–2016*. IUCN, Gland.

Schwitzer, C., Mittermeier, R.A., Johnson, S.E., Donati, G., Irwin, M., Peacock, H., Ratsimbazafy, J., Razafindramanana, J., Louis Jr., E.E., Chikhi, L., *et al*. 2014. Averting lemur extinctions amid Madagascar's political crisis. *Science* 343: 842–843.

Smith, R.J., Goodman, P.S. and Matthews, W.S. 2006. Systematic conservation planning: a review of perceived limitations and an illustration of the benefits, using a case study from Maputaland, South Africa. *Oryx* 40: 400–410.

Smith, R.J., Verissimo, D., Leader-Williams, N., Cowling, R.M. and Knight, A.T. 2009. Let the locals lead. *Nature* 462: 280–281.

Soulé, M. 1985. What is conservation biology? *BioScience* 35: 727–734.

Sunderland, T., Sunderland-Groves, J., Shanley, P. and Campbell, B. 2009. Bridging the gap: how can information access and exchange between conservation biologists and field practitioners be improved for better conservation outcomes? *Biotropica* 41: 549–554.

Sunderlin, W.D., Angelsen, A., Belcher, B., Burgers, P., Nasi, R., Santoso, L. and Wunder, S. 2005. Livelihoods, forests, and conservation in developing countries: an overview. *World Development* 33: 1383–1402.

Sutherland, W.J., Pullin, A.S., Dolman, P.M. and Knight, T.M. 2004. The need for evidence-based conservation. *Trends in Ecology and Evolution* 19: 305–308.

Sutherland, W.J., Armstrong-Brown, S., Armsworth, P.R., Brereton, T., Brickland, J., Campbell, C.D., Chamberlain, D.E., Cooke, A.I., Dulvy, N.K., Dusic, N.R., *et al*. 2006. The identification of 100 ecological questions of high policy relevance in the UK. *Journal of Applied Ecology* 43: 617–627.

Sutherland, W.J., Adams, W.M., Aronson, R.B., Aveling, R., Blackburn, T.M., Broad, S., Ceballos, G., Côté, I.M., Cowling, R.M., Da Fonseca, G.A.B., *et al*. 2009. One hundred questions of importance to the conservation of global biological diversity. *Conservation Biology* 23: 557–567.

Sutherland, W.J., Fleishman, E., Mascia, M.B., Pretty, J. and Rudd, M.A. 2011. Methods for collaboratively identifying research priorities and emerging issues in science and policy. *Methods in Ecology and Evolution* 2: 238–247.

Sutherland, W.J., Bellingan, L., Bellingham, J.R., Blackstock, J.J., Bloomfield, R.M., Bravo, M., Cadman, V.M., Cleevely, D.D., Clements, A., Cohen, A.S., *et al*. 2012. A collaboratively-derived science-policy research agenda. *PLoS ONE* 7: e31824.

Sutherland, W.J., Mitchell, R., Walsh, J., Amano, T., Ausden, M., Beebee, T.J.C., Bullock, D., Daniels, M., Deutsch, J., Griffiths, R.A., *et al*. 2013. Conservation practice could benefit from routine testing and publication of management outcomes. *Conservation Evidence* 10: 1–3.

Vira, B. and Kontoleon, A. 2010. Dependence of the poor on biodiversity: which poor, what biodiversity? In *Linking Biodiversity Conservation and Poverty Alleviation: a State of Knowledge Review*, D. Roe (Ed.), pp. 13–28. Secretariat of the Convention on Biological Diversity, Montreal.

Walsh, J.C., Dicks, L.V. and Sutherland, W.J. 2014. The effect of scientific evidence on conservation practitioners' management decisions. *Conservation Biology*: doi: 10.1111/ cobi.12370.

Watson, J.E.M. 2014. Human responses to climate change will seriously impacts biodiversity conservation: it's time we start planning for them. *Conservation Letters* 7: 1–2.

Wells, M.P., and McShane, T.O. 2004. Integrating protected area management with local needs and aspirations. *Ambio* 33: 513–519.

West, P. and Brockington, D. 2006. An anthropological perspective on some unexpected consequences of protected areas. *Conservation Biology* 20: 609–616.

West, P., Igoe, J. and Brockington, D. 2006. Parks and peoples: the social impact of protected areas. *Annual Review of Anthropology* 35: 251–277.

Western, D. 2001. Taking the broad view of conservation: a response to Adams and Hulme. *Oryx* 35: 201–203.

Whitehurst, A.S., Sexton, J.O. and Dollar, L. 2009. Land cover change in western Madagascar's dry deciduous forests: a comparison of forest changes in and around Kirindy Mite National Park. *Oryx* 43: 275–283.

Whitten, T., Holmes, D. and MacKinnon, K. 2001. Conservation biology: a displacement behaviour for academia? *Conservation Biology* 15: 1–3.

Wilson, K.A., Underwood, E.C., Morrison, S.A., Klausmeyer, K.R., Murdoch, W.W., Reyers, B., Wardell-Johnson, G., Marquet, P.A., Rundel, P.W., McBride, M.F., *et al*. 2007. Conserving biodiversity efficiently: what to do, where, and when. PLoS Biology 5: 1850–1861.

Appendix 1

Supplementary materials for Chapter 5

Table A1.1 Sources of reptile inventory data for the 22 sites in the dry regions of Madagascar used as a case study system to compare four different site prioritisation protocols and Zonation. NP, national park; NPA, new protected area; U, unprotected.

Site no.	Site name	Status	Data sources	Number of survey replicates	Survey duration (days)
1	Ankarafantsika	NP	Ramanamanjato and Rabibisoa 2002; Raselimanana 2008	5	38
2	Namoroka	NP	Raselimanana 2008	2	11
3	Andranomanintsy	U	Raselimanana 2008	1	6
4	Kelifely	U	Rakotondravony and Goodman 2011	1	7
5	Ankara	U	Rakotondravony and Goodman 2011	1	7
6	Tsingy de Bemaraha	NP	ANGAP 2003; Raselimanana 2008; Bora *et al*. 2010	12	136
7	Masoarivo	U	Raselimanana 2008	2	13
8	Menabe Antimena	NPA	Bloxam *et al*. 1996[a]; Raselimanana 2008	2	14
9	Kirindy Mite	NP	Raselimanana 2008	4	28
10	Makay	U	Rakotondravony and Goodman 2011	2	16
11	Berento	U	Rakotondravony and Goodman 2011	1	7
12	Nosy-Ambositra	U	Rakotondravony and Goodman 2011	1	8
13	Mikea	NP	Raselimanana 2004; Raselimanana 2008	6	40
14	Ranobe PK32	NPA	D'Cruze and Sabel 2005; Thomas *et al*. 2006	3	578
15	Zombitse-Vohibasia	NP	Raxworthy *et al*. 1994; Goodman *et al*. 1997	5	36
16	Tsinjoriake	NPA	Raxworthy 1995	2	16
17	Amoron'i Onilahy	NPA	D'Cruze *et al*. 2009	5	378
18	Tsimanampetsotsa	NP	Goodman *et al*. 2002; Raselimanana 2008	3	22
19	Nord Ifotaka	NPA	Raselimanana 2008	1	6
20	Anadabolava-Betsimalaho	NPA	Raselimanana 2008	1	7

Site no.	Site name	Status	Data sources	Number of survey replicates	Survey duration (days)
21	Behara-Tranomaro	NPA	Raselimanana 2008	1	7
22	Andohahela Parcel 2	NP	Nussbaum *et al*. 1999	1	8

[a] Survey duration not known and therefore not accounted for in relevant column

Table A1.2 Attribute scores assigned to 134 reptile species found across 22 sites in the dry regions of Madagascar, used to calculate the conservation value index (*CVI*). E, Endemism; R, Representation, C, Hunting and collection, T, Degradation tolerance.

Family and species	*E*	*R*	*E + R*	*C*	*T*	*C+T*	*CVI* score
GEKKONIDAE							
Blaesodactylus sakalava	3	1	***4***	1	5	***6***	**24**
Ebenavia maintimainty	5	5	***10***	1	5	***6***	**60**
Geckolepis maculata	2	4	***6***	1	3	***4***	**24**
Geckolepis polylepis	4	4	***8***	1	5	***6***	**48**
Geckolepis typica	3	1	***4***	1	5	***6***	**24**
Hemidactylus mercatorius	1	1	***2***	1	1	***2***	**4**
Lygodactylus heterurus	3	5	***8***	1	1	***2***	**16**
Lygodactylus klemmeri	5	5	***10***	1	5	***6***	**60**
Lygodactylus tolampyae	2	1	***3***	1	3	***4***	**12**
Lygodactylus tuberosus	4	3	***7***	1	1	***2***	**14**
Lygodactylus verticillatus	4	4	***8***	1	1	***2***	**16**
Matoatoa brevipes	4	4	***8***	1	5	***6***	**48**
Paragehyra petiti	5	5	***10***	1	5	***6***	**60**
Paroedura androyensis	4	3	***7***	1	5	***6***	**42**
Paroedura bastardi	3	1	***4***	1	3	***4***	**16**
Paroedura homalorhina	4	5	***9***	1	5	***6***	**54**
Paroedura karstophila	4	4	***8***	1	5	***6***	**48**
Paroedura maingoka	5	5	***10***	1	3	***4***	**40**
Paroedura oviceps	3	4	***7***	1	5	***6***	**42**
Paroedura picta	3	2	***5***	1	1	***2***	**10**
Paroedura stumpffi	3	4	***7***	1	3	***4***	**28**
Paroedura tanjaka	4	4	***8***	1	3	***4***	**32**
Paroedura vahiny	3	3	***6***	1	5	***6***	**36**
Paroedura vazimba	4	5	***9***	1	5	***6***	**54**
Phelsuma abbotti	1	4	***5***	3	1	***4***	**20**

Family and species	*E*	*R*	*E + R*	*C*	*T*	*C+T*	*CVI* score
Phelsuma borai	4	4	***8***	3	5	***8***	**64**
Phelsuma breviceps	5	4	***9***	3	5	***8***	**72**
Phelsuma dubia	1	5	***6***	3	1	***4***	**24**
Phelsuma kochi	4	3	***7***	3	3	***6***	**42**
Phelsuma lineata	2	5	***7***	3	3	***6***	**42**
Phelsuma modesta	4	5	***9***	3	1	***4***	**36**
Phelsuma mutabilis	3	1	***4***	3	1	***4***	**16**
Phelsuma standingi	4	3	***7***	3	3	***6***	**42**
Uroplatus ebenaui	3	4	***7***	3	5	***8***	**56**
Uroplatus guentheri	4	4	***8***	3	5	***8***	**64**
Uroplatus henkeli	4	5	***9***	3	5	***8***	**72**
SCINCIDAE							
Amphiglossus andranovahensis	4	4	***8***	1	5	***6***	**48**
Amphiglossus ornaticeps	2	2	***4***	1	3	***4***	**16**
Amphiglossus reticulatus	3	4	***7***	1	3	***4***	**28**
Amphiglossus splendidus	2	5	***7***	1	5	***6***	**42**
Androngo trivittatus	4	4	***8***	1	3	***4***	**32**
Cryptoblepharus boutonii	1	4	***5***	1	3	***4***	**20**
Madascincus igneocaudatus	3	2	***5***	1	5	***6***	**30**
Madascincus intermedius	2	3	***5***	1	5	***6***	**30**
Pygomeles braconnieri	5	4	***9***	1	3	***4***	**36**
Pygomeles petteri	5	5	***10***	1	5	***6***	**60**
Sirenoscincus yamagishii	5	5	***10***	1	5	***6***	**60**
Trachylepis aureopunctata	3	2	***5***	1	3	***4***	**20**
Trachylepis dumasi	3	2	***5***	1	5	***6***	**30**
Trachylepis elegans	2	1	***3***	1	1	***2***	**6**
Trachylepis gravenhorstii	2	1	***3***	1	1	***2***	**6**
Trachylepis tandrefana	4	4	***8***	1	3	***4***	**32**
Trachylepis vato	2	3	***5***	1	3	***4***	**20**
Trachylepis vezo	5	4	***9***	1	5	***6***	**54**
Trachylepis volamenaloha	5	5	***10***	1	5	***6***	**60**
Voeltzkowia fierinensis	4	4	***8***	1	5	***6***	**48**
Voeltzkowia lineata	4	3	***7***	1	3	***4***	**28**
Voeltzkowia mira	5	5	***10***	1	5	***6***	**60**
Voeltzkowia petiti	5	4	***9***	1	5	***6***	**54**
Voeltzkowia rubrocaudata	3	3	***6***	1	5	***6***	**36**
GERRHOSAURIDAE							
Tracheloptychus madagascariensis	4	2	***6***	1	3	***4***	**24**
Tracheloptychus petersi	5	4	***9***	1	5	***6***	**54**
Zonosaurus bemaraha	5	5	***10***	1	5	***6***	**60**

Family and species	***E***	***R***	***E + R***	***C***	***T***	***C+T***	***CVI* score**
Zonosaurus karsteni	3	3	***6***	1	3	***4***	**24**
Zonosaurus laticaudatus	3	1	***4***	1	3	***4***	**16**
Zonosaurus maramaintso	5	5	***10***	1	5	***6***	**60**
Zonosaurus quadrilineatus	5	4	***9***	1	3	***4***	**36**
Zonosaurus trilineatus	4	4	***8***	1	3	***4***	**32**
CHAMAELEONIDAE							
Brookesia bonsi	5	5	***10***	3	5	***8***	**80**
Brookesia brygooi	3	3	***6***	3	3	***6***	**36**
Brookesia decaryi	5	5	***10***	3	5	***8***	**80**
Brookesia exarmata	5	5	***10***	3	5	***8***	**80**
Brookesia perarmata	5	5	***10***	3	5	***8***	**80**
Furcifer angeli	4	4	***8***	3	3	***6***	**48**
Furcifer antimena	5	4	***9***	3	3	***6***	**54**
Furcifer belalandaensis	5	5	***10***	3	5	***8***	**80**
Furcifer labordi	3	3	***6***	3	5	***8***	**48**
Furcifer lateralis	2	2	***4***	1	1	***2***	**8**
Furcifer nicosiai	4	5	***9***	3	5	***8***	**72**
Furcifer oustaleti	2	2	***4***	1	1	***2***	**8**
Furcifer rhinoceratus	4	5	***9***	3	5	***8***	**72**
Furcifer verrucosus	3	2	***5***	1	1	***2***	**10**
IGUANIDAE							
Chalarodon madagascariensis	2	2	***4***	1	1	***2***	**8**
Oplurus cuvieri	1	3	***4***	1	3	***4***	**16**
Oplurus cyclurus	2	2	***4***	1	3	***4***	**16**
Oplurus fierinensis	5	4	***9***	1	3	***4***	**36**
Oplurus quadrimaculatus	3	3	***6***	1	3	***4***	**24**
Oplurus saxicola	4	3	***7***	1	3	***4***	**28**
LAMPROPHIIDAE							
Alluaudina bellyi	3	5	***8***	1	5	***6***	**48**
Compsophis albiventris	3	5	***8***	1	5	***6***	**48**
Dromicodryas bernieri	2	1	***3***	1	1	***2***	**6**
Dromicodryas quadrilineatus	2	4	***6***	1	1	***2***	**12**
Heteroliodon lava	4	5	***9***	1	5	***6***	**54**
Heteroliodon occipitalis	3	2	***5***	1	3	***4***	**20**
Ithycyphus miniatus	3	4	***7***	1	5	***6***	**42**
Ithycyphus oursi	3	3	***6***	1	5	***6***	**36**
Langaha alluaudi	3	4	***7***	1	5	***6***	**42**
Langaha madagascariensis	2	2	***4***	1	3	***4***	**16**
Leioheterodon geayi	3	2	***5***	1	3	***4***	**20**
Leioheterodon madagascariensis	2	2	***4***	1	3	***4***	**16**

Family and species	*E*	*R*	*E + R*	*C*	*T*	*C+T*	*CVI* score
Leioheterodon modesta	3	2	***5***	1	3	***4***	**20**
Liophidium apperti	4	3	***7***	1	5	***6***	**42**
Liophidium chabaudi	5	4	***9***	1	3	***4***	**36**
Liophidium maintikibo	5	5	***10***	1	5	***6***	**60**
Liophidium therezieni	3	5	***8***	1	5	***6***	**48**
Liophidium torquatum	2	3	***5***	1	3	***4***	**20**
Liophidium trilineatum	4	4	***8***	1	5	***6***	**48**
Liophidium vaillanti	3	3	***6***	1	3	***4***	**24**
Lycodryas citrinus	4	4	***8***	1	5	***6***	**48**
Lycodryas granuliceps	3	5	***8***	1	3	***4***	**32**
Lycodryas inornatus	4	5	***9***	1	5	***6***	**54**
Lycodryas pseudogranuliceps	3	3	***6***	1	5	***6***	**36**
Madagascarophis colubrinus	2	1	***3***	1	3	***4***	**12**
Madagascarophis meridionalis	3	3	***6***	1	3	***4***	**24**
Madagascarophis ocellatus	4	3	***7***	1	3	***4***	**28**
Mimophis mahfalensis	2	1	***3***	1	1	***2***	**6**
Phisalixella tulearensis	3	4	***7***	1	3	***4***	**28**
Phisalixella variabilis	5	4	***9***	1	5	***6***	**54**
Pseudoxyrhopus kely	3	5	***8***	1	5	***6***	**48**
Pseudoxyrhopus quinquelineatus	2	3	***5***	1	5	***6***	**30**
Thamnosophis lateralis	2	4	***6***	1	1	***2***	**12**
Thamnosophis mavotenda	5	5	***10***	1	5	***6***	**60**
BOIDAE							
Acrantophis dumerili	3	2	***5***	3	1	***4***	**20**
Acrantophis madagascariensis	2	4	***6***	3	1	***4***	**24**
Sanzinia madagascariensis	2	3	***5***	3	1	***4***	**20**
TYPHLOPIDAE							
Typhlops arenarius	3	2	***5***	1	3	***4***	**20**
Typhlops decorsei	3	3	***6***	1	3	***4***	**24**
PODOCNEMIDIDAE							
Erymnochelys madagascariensis	4	4	***8***	5	5	***10***	**80**
PELOMEDISUDAE							
Pelomedusa subrufa	1	2	***3***	3	3	***6***	**18**
Pelusios castanoides	1	3	***4***	3	3	***6***	**24**
TESTUDINIDAE							
Astrochelys radiata	4	3	***7***	5	5	***10***	**70**
Pyxis arachnoides	4	3	***7***	5	5	***10***	**70**
Pyxis planicauda	5	5	***10***	3	5	***8***	**80**
CROCODYLIDAE							
Crocodylus niloticus	1	3	***4***	3	3	***6***	**24**

Table A1.3 The site prioritisation rankings according to the conservation value index (*CVI*) protocol, following sensitivity analyses where each of individual species attribute scores were doubled.

Site	Original *CVI*	Endemism (*E*) x 2	Representation (*R*) x 2	Hunting and collection (*C*)	Degradation tolerance (*T*)
Tsingy de Bemaraha	1	1	1	1	1
Ranobe PK32	2	2	2	2	2
Ankarafantsika	3	3	3	3	3
Mikea	4	4	4	4	4
Tsimanampetsotsa	5	5	5	5	5
Menabe Antimena	6	6	6	7	6
Amoron'i Onilahy	7	7	7	6	7
Tsinjoriake	8	8	8	8	8
Namoroka	9	9	9	9	9
Andranomanintsy	10	10	10	10	10
Anadabolava-Betsimalaho	11	11	11	11	11
Nosy-Ambositra	12	13	12	12	13
Andohahela P2	13	12	15	14	12
Zombitse-Vohibasia	14	14	13	13	14
Masoarivo	15	15	14	15	15
Kirindy Mite	16	16	17	16	16
Ankara	17	17	16	17	17
Kelifely	18	19	18	18	18
Nord Ifotaka	19	18	20	19	19
Makay	20	21	19	20	20
Behara-Tranomaro	21	20	21	21	21
Berento	22	22	22	22	22
Correlation with Zonation (Spearman's rank)	**0.927**	**0.916**	**0.932**	**0.922**	**0.921**

References for Table A1.1

ANGAP (Association Nationale pour la Gestion des Aires Protégées) 2003. *Plan de Gestion de la Conservation du Complexe d'Aires Protégées Tsingy de Bemaraha*. ANGAP, Antananarivo.

Bloxam, Q.M.C, Behler, J.L., Rakotovao, E.R., Randriamahazo, H.J.A.R., Hayes, K. T., Tonge, S.J. and Ganzhorn, J.U. 1996. Effects of logging on the reptile fauna of the Kirindy forest with special emphasis on the Flat-tailed tortoise (*Pyxis planicauda*). In *Ecology and Economy of a Tropical Dry Forest in Madagascar*, J.U. Ganzhorn and J.-P. Sorg (Eds.), pp. 189–201. *Primate Report* 46.

Bora, P., Randrianantoandro, J.C., Randrianavelona, R., Hantalalaina, E.F., Andriantsimanarilafy, R.R., Rakotondravony, D., Ramilijaona, O.R., Vences, M., Jenkins, R.K.B, Glaw, F., *et al*. 2010. Amphibians and reptiles of the Tsingy de Bemaraha Plateau, western Madagascar: checklist, biogeography and conservation. *Herpetological Conservation and Biol*ogy 5: 111–125.

D'Cruze, N.C. and Sabel, J.A. 2005. The reptiles of the Southern Mikea Forest, Madagascar. *Herpetological Bulletin* 93: 2–8.

D'Cruze, N., Olsonn, A., Henson, D., Kumar, S. and Emmett, D. 2009. The amphibians and reptiles of the lower Onilahy river valley, a temporary protected area in southern Madagascar. *Herpetological Conservation and Biology* 4: 62–79.

Goodman, S.M., Ramanamanjato, J.-B. and Raselimanana, A.P. 1997. Les amphibiens et les reptiles. In *Inventaire Biologique Forêts de Vohibasia et Isoky-Vohimena*, O. Langrand and S.M. Goodman (Eds.), pp. 110–130. *Recherches pour le Développement, Série Sciences Biologiques* 12.

Goodman, S.M., Raherilalao, M.J., Rakotondravony, D., Raselimanana, A., Razakarivony, H.V. and Soarimalala, V. 2002. Inventaire des vertébrés du Parc National de Tsimanampetsotsa (Toliara). *Akon'ny Ala* 28 : 1–36.

Nussbaum, R.A., Raxworthy, C.J., Raselimanana, A.P. and Ramanamanjato, J.-B. 1999. Amphibians and reptiles of the Réserve Naturelle Intégrale d'Andohahela, Madagascar. In *A Floral and Faunal Inventory of the Réserve Naturelle Intégrale d'Andohahela, Madagascar: with Reference to Altitudinal Variation*, S.M. Goodman (Ed.). *Fieldiana: Zoology*, new series 94: 155–173.

Rakotondravony, H.A and Goodman, S.M. 2011. Rapid herpetofaunal surveys within five isolated forests on sedimentary rock in western Madagascar. *Herpetological Conservation and Biology* 6 : 297–311.

Ramanamanjato, J.-B. and Rabibisoa, N. 2002. Evaluation rapide de la diversité biologique des reptiles et amphibiens de la Réserve Naturelle Intégrale d'Ankarafantsika. In *Une Evaluation Biologique de la Réserve Naturelle Intégrale d'Ankarafantsika, Madagascar*, L.E. Alonso, T.S. Schulenberg, S. Radilofe and O. Missa (Eds.), pp. 98–193. *RAP Bulletin of Biological Assessment* 23.

Raselimanana, A.P. 2004. l'Herpétofaune de la forêt de Mikea. In *Inventaire Floristique et Faunistique de la Forêt des Mikea: Paysage Ecologique et Diversité Biologique d'une Préoccupation Majeure pour la Conservation*, A.P. Raselimanana and S.M. Goodman (Eds.), pp. 37–52. *Recherches pour le Développement, Série Sciences Biologiques* 21.

Raselimanana, A.P. 2008. Herpétofaune des forêts sèches malgaches. In *Les Forêts Sèches de Madagascar*, S.M. Goodman and L. Wilmé (Eds.), pp. 46–75. *Malagasy Nature* 1.

Raxworthy, C.J. 1995. *Field survey of amphibians, reptiles and small mammals from the Toliara region, Madagascar, Oct. 5–30 1995*. Unpublished Report, University of Michigan.

Raxworthy, C.J., Ramanamanjato, J.-B. and Raselimanana, A. 1994. Les reptiles et les amphibiens. In *Inventaire Biologique de la Forêt de Zombitse*, S.M. Goodman and O. Langrand (Eds.), pp. 41–57. *Recherches pour le Développement, Série Sciences Biologiques No. Spécial*.

Thomas, H., Kidney, D., Rubio, P. and Fanning, E. 2006. *The Southern Mikea: a Biodiversity Survey*. Frontier, London.

Appendix 2

Supplementary materials for Chapter 6

Table A2.1 Protected areas in western and southern Madagascar used to assign representation scores for birds (n = 14) and reptiles (n = 15) in a Conservation Value Index (CVI). NPA = new protected area.

Protected area	Status	Bird data sources	Reptile data sources
Ankarafantsika	National Park	Raherilalao and Wilmé 2008; Schulenberg and Randrianasolo 2002	Ramanamanjato and Rabibisoa 2002; Raselimanana 2008
Namoroka	National Park	Raherilalao and Wilmé 2008	Raselimanana 2008
Tsingy de Bemaraha	National Park	Raherilalao and Wilmé 2008	ANGAP 2003; Raselimanana 2008; Bora *et al*. 2010
Menabe Antimena	NPA	Raherilalao and Wilmé 2008	Bloxam *et al*. 1996; Raselimanana 2008
Kirindy Mite	National Park	Raherilalao and Wilmé 2008	Raselimanana 2008
Mikea	National Park	Raherilalao *et al*. 2004	Raselimanana 2004, 2008
Ranobe PK32	NPA	Gardner *et al*. 2009	D'Cruze and Sabel 2005; Thomas *et al*. 2006
Zombitse-Vohibasia	National Park	Goodman *et al*. 1994; Langrand and Goodman 1997	Raxworthy *et al*. 1994; Goodman *et al*. 1997
Tsinjoriake	NPA	-	Raxworthy 1995
Amoron'i Onilahy	NPA	Emmett *et al*. 2003	D'Cruze *et al*. 2009
Tsimanampesotse	National Park	Goodman *et al*. 2002	Goodman *et al*. 2002; Raselimanana 2008
Nord Ifotaka	NPA	Raherilalao and Wilmé 2008	Raselimanana 2008
Anadabolava-Betsimalaho	NPA	Raherilalao and Wilmé 2008	Raselimanana 2008
Behara-Tranomaro	NPA	Raherilalao and Wilmé 2008	Raselimanana 2008
Andohahela Parcel 2	National Park	Hawkins and Goodman 1999	Nussbaum *et al*. 1999

Table A2.2 Relative frequency (RF) and relative abundance of bird species recorded in 48 point counts at three sites in Ranobe across a gradient of habitat degradation (represented by Low, Moderate and High). Chi-squared tests were used to test for heterogeneity of observed patters of relative frequency. * $p < 0.05$, ** $p < 0.01$.

Species	Low		Moderate		High		χ^2
	RF	RA	RF	RA	RF	RA	
Aviceda madagascariensis			0.04	0.04			-
Polyboroides radiatus	0.04	0.04	0.02	0.02			-
Accipiter francesiae			0.08	0.08			-
Buteo brachypterus			0.04	0.04			-
Falco newtoni	0.10	0.10	0.08	0.08	0.23	0.23	4.99
Falco zoniventris			0.04	0.04			-
Falco concolor					0.17	0.17	-
Falco peregrinus					0.02	0.02	-
Numida meleagris					0.13	0.13	-
Monias benschi	0.17	0.17	0.04	0.04	0.04	0.04	-
Turnix nigricollis	0.10	0.10			0.60	0.65	55.52**
Nesoenas picturata	0.35	0.44	0.44	0.48	0.10	0.10	13.79**
Oena capensis	0.50	0.69	0.21	0.29	0.88	1.77	43.02**
Treron australis					0.04	0.04	
Coracopsis vasa	0.17	0.17	0.17	0.17			9.00*
Coracopsis nigra	0.10	0.10	0.25	0.25			14.54**
Agapornis canus	0.13	0.13	0.13	0.13	0.38	0.40	12.13**
Cuculus rochii	0.31	0.35	0.15	0.19	0.06	0.06	10.84 **
Coua cursor	0.15	0.17	0.04	0.04	0.06	0.06	-
Coua ruficeps olivaceiceps	0.21	0.23	0.29	0.29			15.60**
Coua cristata	0.65	1.15	0.56	0.67	0.40	0.46	6.25*
Centropus toulou	0.46	0.52	0.46	0.63	0.65	0.81	4.46
Caprimulgus madagascariensis	0.02	0.02					-
Tachymarptis melba			0.02	0.02			-
Apus barbatus	0.02	0.02	0.04	0.04			-
Merops superciliosus	0.23	0.23	0.21	0.23	0.38	0.42	4.01
Eurystomus glaucurus			0.02	0.02			-
Upupa marginata	0.29	0.33	0.19	0.29	0.25	0.27	1.43
Mirafra hova					0.13	0.15	-
Phedina borbonica			0.02	0.02	0.02	0.02	-
Thamnornis chloropetoides	0.15	0.15	0.13	0.13			-
Hypsipetes madagascariensis	0.48	0.56	0.10	0.10	0.27	0.27	16.64**
Copsychus albospecularis	0.58	0.75	0.56	0.69	0.42	0.46	3.17

Species	Low		Moderate		High		χ^2
	RF	RA	RF	RA	RF	RA	
Nesillas lantzii	0.06	0.06			0.02	0.02	-
Newtonia brunneicauda	1.00	2.29	0.96	2.25	0.44	0.65	58.64**
Newtonia archboldi	0.52	0.75	0.35	0.48	0.17	0.17	13.30**
Cisticola cherina					0.88	1.54	118.59**
Neomixis tenella	0.83	1.08	0.81	1.13	0.65	0.71	5.62
Neomixis striatigula	0.58	0.83	0.81	1.23	0.29	0.33	26.58**
Terpsiphone mutata	0.52	0.60	0.65	0.79	0.25	0.31	15.77**
Nectarinia souimanga	0.96	2.13	1.00	2.17	0.73	0.98	21.88**
Nectarinia notata			0.08	0.10	0.02	0.02	-
Calicalicus madagascariensis	0.02	0.02	0.02	0.02			-
Vanga curvirostris	0.48	0.54	0.29	0.29	0.04	0.04	23.42**
Xenopirostris xenopirostris			0.02	0.02	0.15	0.15	-
Falculea palliata	0.19	0.19	0.19	0.23			10.29 **
Artamella viridis	0.08	0.08	0.15	0.15			-
Leptopterus chabert	0.13	0.13	0.10	0.10	0.21	0.21	2.34
Dicrurus forficatus	0.92	1.67	0.69	0.92	0.73	0.94	8.28*
Acridotheres tristis			0.02	0.02	0.58	0.98	65.37**
Corvus albus					0.04	0.04	-
Ploceus sakalava	0.02	0.02	0.44	0.44	0.15	0.17	27.29**
Foudia madagascariensis	0.02	0.02	0.10	0.10	1.00	3.60	120.71**

Table A2.3 Reptile capture rates and estimated densities, generated from pitfall trap and transect data respectively. Chi-squared tests were performed on the total contacts for each species, which pooled both methods. * $p < 0.05$, ** $p < 0.01$.

Species	Low			Moderate			High			χ^2
	Capture rate	Density/ ha	Total contacts	Capture rate	Density/ ha	Total contacts	Capture rate	Density/ ha	Total contacts	
Scincidae										
Trachylepis elegans	0.037	19.73	30	0.050	21.05	36	0.020	13.16	18	6.52*
Trachylepis aureopunctatus	0.002	1.32	2	0.010	2.63	6	0.002		1	-
Madascincus cf. *igneocaudatus*	0.015		6	0.037		15				16.29**
Amphiglossus ornaticeps	0.005	1.32	3	0.012		5				-
Voeltzkowia petiti	0.007		3	0.027		11				-
Voeltzkowia rubrocauda	0.017		7				0.005		2	-
Pygomeles braconnieri	0.005		2	0.012		5				-
Gerrhosauridae										
Tracheloptychus petersi	0.005	9.21	9	0.022	14.47	20	0.002		1	21.06**
Zonosaurus quadrilineatus	0.015	3.95	9	0.010	5.26	8	0.002	1.32	2	4.53
Zonosaurus karsteni	0.002		1	0.002		1				-
Iguanidae										
Chalarodon madagascariensis	0.132	32.90	78	0.040	30.26	39	0.047	13.16	29	27.54**
Oplurus cyclurus	0.002	27.63	22	0.002	10.52	9	0.002		1	21.06**
Gekkonidae										
Geckolepis cf. *polylepis*		25.0	19	0.012	10.52	13				17.69**
Blaesodactylus sakalava		3.95	3	0.002	1.32	2				-
Hemidactylus mercatorius		7.89	6		7.90	6	0.002	5.26	5	0.33

Species	Low			Moderate			High			χ^2
	Capture rate	Density/ ha	Total contacts	Capture rate	Density/ ha	Total contacts	Capture rate	Density/ ha	Total contacts	
Lygodactylus tuberosus							0.002	82.90	64	128**
Lygodactylus verticillatus		15.79	12		3.95	3				15.6**
Paroedura androyensis	0.002		1	0.005	1.32	3				-
Paroedura picta	0.020		8	0.007	1.32	4	0.052		21	14.36**
Phelsuma mutabilis		18.42	14		11.84	9		2.63	2	8.72*
Chamaeleonidae										
Furcifer verrucosus	0.007	5.26	7	0.002	1.32	2				-
Lamprophiidae										
Dromicodryas bernieri		1.32	1		1.32	1		1.32	1	-
Heteroliodon occipitalis	0.005		2	0.010		4				-
Ithycyphus oursi					1.32	1				-
Liophidium chabaudi	0.012		5	0.010		4	0.007		3	-
Leioheterodon geayi	0.005		2							-
Madagascarophis meridionalis				0.002		1				-
Madagascarophis ocellatus	0.002	2.63	3							-
Mimophis mahfalensis	0.002	6.57	6	0.002	9.21	8	0.002	7.90	7	0.29
Typhlopidae										
Typhlops arenarius				0.010		4	0.032		13	15.65**
Typhlops decorsei				0.002		1				-
Testudinidae										
Pyxis arachnoides					1.32	1				-

Table A2.4 Subdivision of reptile community from Ranobe by primary foraging substrate, with total number of contacts (using pooled transect and pitfall trap data) at three sites across a gradient of degradation (represented by Low, Mod and High). ** $p < 0.01$

Primary substrate use (number of species)	Total contacts			Chi-square
	Low	Mod	High	
Terrestrial (14)	150	130	80	21.7**
Leaf litter/ fossorial (9)	28	49	18	15.8**
Arboreal (9)	84	45	72	11.9**

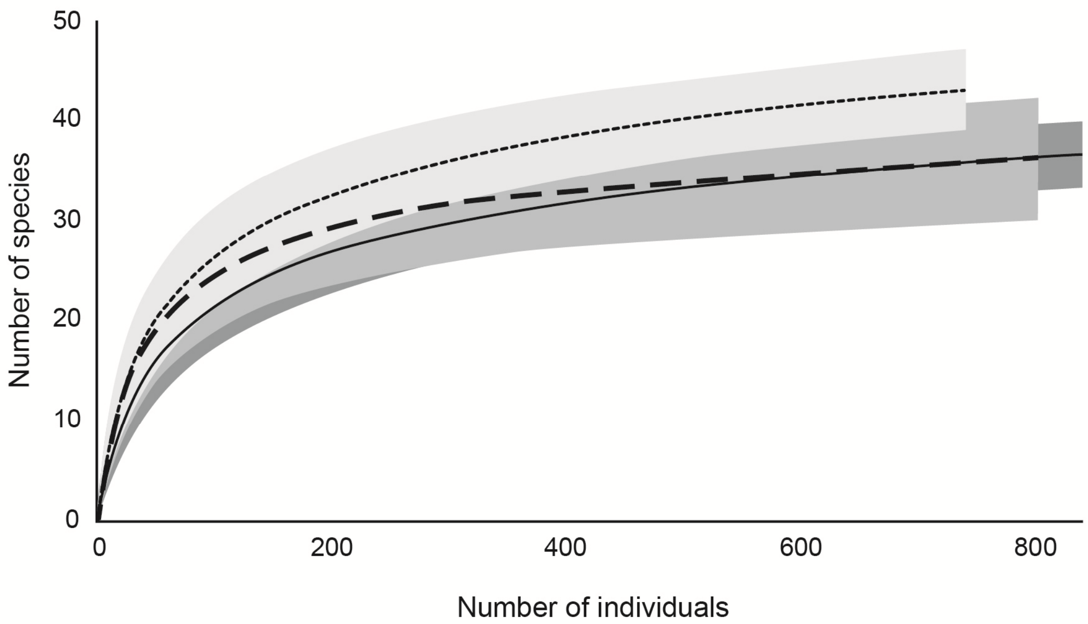

Fig. A2.1 Individual-based rarefaction curves (black lines) and 95 % confidence intervals (grey shading) for birds surveyed in three treatments across a gradient of degradation at Ranobe, southwest Madagascar. Dashed line and mid-grey shading, low-intensity degradation; dotted line and light-grey shading, medium-intensity degradation; solid line and dark-grey shading, high-intensity degradation.

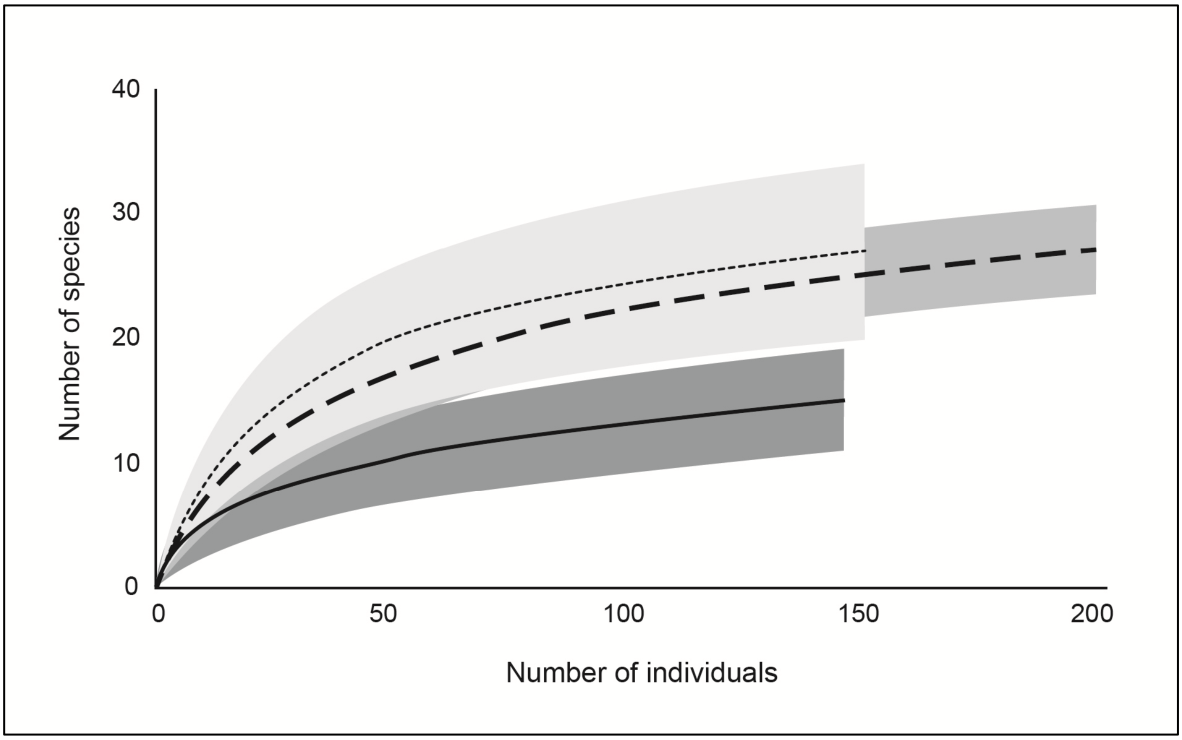

Fig. A2.2 Individual-based rarefaction curves (black lines) and 95 % confidence intervals (grey shading) for reptiles surveyed at three sites across a gradient of degradation at Ranobe, southwest Madagascar. Dashed line and mid-grey shading, low-intensity degradation; dotted line and light-grey shading, medium-intensity degradation; solid line and dark-grey shading, high-intensity degradation.

References

ANGAP (Association Nationale pour la Gestion des Aires Protégées) 2003. *Plan de Gestion de la Conservation du Complexe d'Aires Protégées Tsingy de Bemaraha*. ANGAP, Antananarivo.

Bloxam, Q.M.C, Behler, J.L., Rakotovao, E.R., Randriamahazo, H.J.A.R., Hayes, K. T., Tonge, S.J. and Ganzhorn, J.U. 1996. Effects of logging on the reptile fauna of the Kirindy forest with special emphasis on the Flat-tailed tortoise (*Pyxis planicauda*). In *Ecology and Economy of a Tropical Dry Forest in Madagascar*, J.U. Ganzhorn and J.-P. Sorg (Eds.), pp. 189–201. *Primate Report* 46.

Bora, P., Randrianantoandro, J.C., Randrianavelona, R., Hantalalaina, E.F., Andriantsimanarilafy, R.R., Rakotondravony, D., Ramilijaona, O.R., Vences, M., Jenkins, R.K.B, Glaw, F., *et al*. 2010. Amphibians and reptiles of the Tsingy de Bemaraha Plateau, western Madagascar: checklist, biogeography and conservation. *Herpetological Conservation and Biol*ogy 5: 111–125.

D'Cruze, N.C. and Sabel, J.A. 2005. The reptiles of the Southern Mikea Forest, Madagascar. *Herpetological Bulletin* 93: 2–8.

D'Cruze, N., Olsonn, A., Henson, D., Kumar, S. and Emmett, D. 2009. The amphibians and reptiles of the lower Onilahy river valley, a temporary protected area in southern Madagascar. *Herpetological Conservation and Biology* 4: 62–79.

Gardner, C.J., Kidney, D. and Thomas, H. 2009. First comprehensive avifaunal survey of PK32-Ranobe, a new protected area in south-western Madagascar. *Phelsuma* 17: 20–39.

Goodman, S.M., Langrand, O. and Razafimahaimodison, J.-C. 1994. Les oiseaux. In *Inventaire Biologique de la Forêt de Zombitse*, S.M. Goodman and O. Langrand (Eds.), pp. 73–84. *Recherches pour le Développement, Série Sciences Biologiques*, N° spécial.

Goodman, S.M., Ramanamanjato, J.-B. and Raselimanana, A.P. 1997. Les amphibiens et les reptiles. In *Inventaire Biologique Forêts de Vohibasia et Isoky-Vohimena*, O. Langrand and S.M. Goodman (Eds.), pp. 110–130. *Recherches pour le Développement, Série Sciences Biologiques* 12.

Goodman, S.M., Raherilalao, M.J., Rakotondravony, D., Raselimanana, A., Razakarivony, H.V. and Soarimalala, V. 2002. Inventaire des vertébrés du Parc National de Tsimanampetsotsa (Toliara). *Akon'ny Ala* 28: 1–36.

Hawkins, A.F.A. and Goodman, S.M. 1999. Bird community variation with elevation and habitat in parcels 1 and 2 of the Réserve Naturelle Intégrale d'Andohahela, Madagascar. In *A Floral and Faunal Inventory of the Réserve Naturelle Intégrale d'Andohahela, Madagascar: with Reference to Altitudinal Variation*, S.M. Goodman (Ed.), pp. 175–186. *Fieldiana: Zoology*, new series 94.

Langrand, O. and Goodman, S.M. 1997. Les oiseaux. In *Inventaire Biologique Forêt de Vohibasia et d'Isoky-Vohimena*, O. Langrand and S.M. Goodman (Eds.), pp. 131–143. *Recherches pour le Développement, Série Sciences Biologiques* 12.

Nussbaum, R.A., Raxworthy, C.J., Raselimanana, A.P. and Ramanamanjato, J.-B. 1999. Amphibians and reptiles of the Réserve Naturelle Intégrale d'Andohahela, Madagascar. In *A Floral and Faunal Inventory of the Réserve Naturelle Intégrale d'Andohahela, Madagascar: with Reference to Altitudinal Variation*, S.M. Goodman (Ed.), pp. 155–173. *Fieldiana: Zoology*, new series 94.

Raherilalao, M.-J. and Wilmé, L. 2008. L'avifaune des forêts sèches malgaches. In *Les Forêts Sèches de Madagascar*, S.M. Goodman and L. Wilmé (Eds.). *Malagasy Nature* 1: 76–105.

Raherilalao, M.-J., Goodman, S.M. and Willard, D.E. 2004. Diversité de l'avifaune de la forêt de Mikea. In *Inventaire Floristique et Faunistique de la Forêt de Mikea: Paysage Ecologique et Diversité Biologique d'une Préoccupation Majeure pour la Conservation*, A.P. Raselimanana and S.M. Goodman (Eds.), pp. 53–68. *Recherches pour le Développement, Série Sciences Biologiques* 21.

Ramanamanjato, J.-B. and Rabibisoa, N. 2002. Evaluation rapide de la diversité biologique des reptiles et amphibiens de la Réserve Naturelle Intégrale d'Ankarafantsika. In *Une Evaluation Biologique de la Réserve Naturelle Intégrale d'Ankarafantsika, Madagascar*, L.E. Alonso, T.S. Schulenberg, S. Radilofe and O. Missa (Eds.), pp. 98–193. *RAP Bulletin of Biological Assessment* 23.

Raselimanana, A.P. 2004. l'Herpétofaune de la forêt de Mikea. In *Inventaire Floristique et Faunistique de la Forêt des Mikea: Paysage Ecologique et Diversité Biologique d'une Préoccupation Majeure pour la Conservation*, A.P. Raselimanana and S.M. Goodman (Eds.), pp. 37–52. *Recherches pour le Développement, Série Sciences Biologiques* 21.

Raselimanana, A.P. 2008. Herpétofaune des forêts sèches malgaches. In *Les Forêts Sèches de Madagascar*, S.M. Goodman and L. Wilmé (Eds.), pp. 46–75. *Malagasy Nature* 1.

Raxworthy, C.J. 1995. *Field survey of amphibians, reptiles and small mammals from the Toliara region, Madagascar, Oct. 5–30 1995*. Unpublished Report, University of Michigan.

Raxworthy, C.J., Ramanamanjato, J.-B. and Raselimanana, A. 1994. Les reptiles et les amphibiens. In *Inventaire Biologique de la Forêt de Zombitse*, S.M. Goodman and O. Langrand (Eds.), pp. 41–57. *Recherches pour le Développement, Série Sciences Biologiques* N° spécial.

Schulenberg, T.S. and Randrianasolo, H. 2002. Evaluation ornithologique rapide de la Réserve Naturelle Intégrale d'Ankarafantsika, Madagascar. In *Une évaluation biologique de la Réserve Naturelle Intégrale d'Ankarafantsika, Madagascar*, L.E. Alonso, T.S. Schulenberg, S. Radilofe and O. Missa (Eds.), pp. 88–92. *RAP Bulletin of Biological Assessment* 23.

Thomas, H., Kidney, D., Rubio, P. and Fanning, E. 2006. *The Southern Mikea: a Biodiversity Survey*. Frontier, London.

Appendix 3

Supplementary materials for Chapter 8

File A3.1 English translation of questionnaire administered to 208 charcoal producers along the Route Nationale 9, south-west Madagascar, between December 2010 and June 2011.

Research Questionnaire: Charcoal production and livelihood changes

Interviewer: Interview No.

Interview location: Date:

How did you find this interviewer (find on sarety, in house etc):

Excuse me, I am carrying out a survey about charcoal production in this area, and especially about the factors that lead people to become charcoal producers. To do this I am asking questions from many charcoal producers throughout this region, from Belalanda to Tsianisiha. Would you mind if I ask you a few questions? This will take about twenty minutes, and you will be offered 200 Ar for your time.

Before we start, I would like to explain a little about my study. I will not ask you your name, so the results will be anonymous, and I ask you to answer the questions as honestly as possible. The answers you give will be used by a student for his personal research, and he may use the answers he collects to make recommendations to WWF about how to improve the charcoal industry and alternative development activities for the area.

1 - Basic information

Firstly, I would like to ask just a few general questions about your life.

A – how old are you? ____________________

B – how many children do you have? ____________________

C – Have you ever been to school? (Yes/No) ____________________

D – *[If yes]* Between what ages did you go to school? ____________________

E – What ethnicity are you? ____________________

F – Where do you live? (village, commune) ____________________

G – How long have you been living in that place? ____________________

[The following questions are asked only if the respondent has not been in the same location all his life]

H – Where did you live before you lived at your present location (village, district, region) ____________________

I – For what reasons did you decide to leave your area of origin? (Prompt)

____
____

J – For what reasons did you chose your current home as a place to settle?

____
____

2 – Livelihood activities

A - The next group of questions are about the activities that you carry out day to day to get food and money for yourself and your family. I am going to give a list of activities, and for each one I will ask you to tell me how important it is for you and your family as a source of income. I will ask you to give a score of 1-3 for each activity, depending on how important it is to you.

1 – I never carry out this activity.
2 – I sometimes carry out this activity, but not very often (for example once a week, or two months per year). It brings some revenue to my household but it is small compared to other activities.
3 – I carry out this activity quite a lot (for example three days a week, or for six months a year). It is an important source of revenue for my household.

Activity	**1**	**2**	**3**
Farming (growing crops in sedentary fields)			
Hatsake (slash-and-burn agriculture)			
Keeping livestock (cattle, goats, sheep, pigs, chickens etc)			
Fishing and collecting marine products			
Hunting of wilds animals (Tandraka, tambotriky, lambo etc)			
Cutting wood in the forest and selling it			
Producing charcoal			
Collecting medicinal plants, yams, honey and other wild products			
Selling things (in market, owning a shop)			
Waged labour – i.e. a job			
Supported by other people (e.g. family members)			
Any other activities that are not on the list *[please specify]*:			
____			
____			
____			

B – Now I would like to ask the same question again, but about how important each activity was five years ago. This is because I would like to know if your life has changed in the last five years, or if is mainly the same. We will use the same scoring system as before, from 1-3.

1 – I never carry out this activity.
2 – I sometimes carry out this activity, but not very often (for example once a week, or two months per year). It brings some revenue to my household but it is small compared to other activities.
3 – I carry out this activity quite a lot (for example three days a week, or for six months a year). It is an important source of revenue for my household.

Activity	**1**	**2**	**3**
Farming (growing crops in sedentary fields)			
Hatsake (slash-and-burn agriculture)			
Keeping livestock (cattle, goats, sheep, pigs, chickens etc)			
Fishing and collecting marine products			
Hunting of wilds animals (Tandraka, tambotriky, lambo etc)			
Cutting wood in the forest and selling it			
Producing charcoal			
Collecting medicinal plants, yams, honey and other wild products			
Selling things (in market, owning a shop)			
Waged labour – i.e. a job			
Supported by other people (e.g. family members)			
Any other activities that are not on the list *[please specify]*:			
______________________________			
______________________________			
______________________________			

C – In summary, what do you think have been the major changes in your livelihood activities in the last five years?

____

3 – Factors influencing change

[Ask these questions only to respondents who have said that their livelihoods have changed].

You have mentioned that the mix of activities that you carry out today has changed since five years ago. Now I would like to know more about why your life has changed. In other words, I would like to know what has changed in your life, or in the world around you, that made you decide to change your activities.

A - What are the reasons for the change in activities? ______________________________

____

[If the respondent cannot give a clear answer, then you can try suggesting reasons. Factors might include:]

- Lack of rain and increasing dryness
- Life is harder/more expensive
- Loss of agricultural land or infrastructure
- Problems in life or in family needing a lot of money
- Not enough fish to catch
- Problems of insecurity/cattle theft

4 – Charcoal production

The next group of questions are specifically about charcoal production

A – Do you produce charcoal all year, or only some of the year? Which months?
____
____

B – If you produce charcoal in some months, why is that? What do you do in the rest of the year, and why?
____
____

C – Where do you produce charcoal? How far is this from your home?
____
____

D – What difficulties do you face in your life as a charcoal producer?
____
____

6 – And finally

A – Do you think that your life is harder or easier now than it was five years ago? i.e. is it harder or easier to find the money and food that you need?

Easier Harder About the same

Thank you very much for participating in this interview, goodbye.

Appendix 4

Publications associated with the thesis

The following papers, book and monograph chapters, editorials and natural history notes have been published during my registration as a doctoral candidate at the University of Kent and relate to Madagascar's biodiversity or its conservation.

Gardner, C.J. and Jasper, L.D. 2014. A record of carnivory by the crested drongo (*Dicrurus forficatus*). *Malagasy Nature* 8: 105–106.

Gardner, C.J. 2014. Publishing for conservation and development? *Madagascar Conservation & Development* 9: 3–4.

Gardner, C.J. and Jasper, L. 2014. Northwards range extension of the littoral rock thrush *Monticola imerinus* in south-west Madagascar. *Bulletin of the African Bird Club* 21: 89–90.

Virah-Sawmy, M., **Gardner, C.J.** and Ratsifandrihamanana, A.N. 2014. The Durban Vision in practice: experiences in participatory governance of Madagascar's new protected areas. In *Conservation and Environmental Management in Madagascar*, I. Scales (Ed.), pp. 216–252. Routledge, London.

Ferguson, B., **Gardner, C.J.,** Andriamarovololona, M.M., Healy, T., Muttenzer, F., Smith, S., Hockley, N. and Gingembre, M. 2014. Governing ancestral land in Madagascar: have policy reforms contributed to social justice? In *Governance for Justice and Environmental Sustainability: Lessons across Natural Resource Sectors in Sub-Saharan Africa*, M. Sowman and R. Wynberg (Eds.) pp. 63–93. Routledge, London.

Gardner, C.J. and Jasper, L. 2014. *Accipiter henstii* (Schlegel, 1873) (Falconiformes: Accipitridae): new distribution record from southwest Madagascar. *Check List* 10: 164–165.

Taylor, B. and **Gardner, C**. 2014. Nectar feeding by the day gecko *Phelsuma mutabilis* (Squamata: Gekkonidae) on the mangrove tree *Sonneratia alba* (Lythraceae) in southwest Madagascar. *Herpetology Notes* 7: 85–87.

Gardner, C.J., Germain, N. and de Foucault, C. 2013. Heaviest recorded specimen of Madagascar ground boa *Acrantophis madagascariensis* (Serpentes: Boidae). *Herpetology Notes* 6: 553–554.

Walker, R.C.J., Castellano, C.M., Ogle, M., Rafeliarisoa, T.H., Rakotondrainy, R., Ramahaleo, T.A., Randriamahazo, H. and **Gardner, C.J**. 2013. Proposed action plan for the conservation of the Madagascar spider tortoise, *Pyxis arachnoides*. In *Turtles on the Brink in Madagascar: Proceedings of Two Workshops on the Status, Conservation, and Biology of Malagasy Tortoises and Freshwater Turtles,* C.M. Castellano, A.G.J. Rhodin, M. Ogle, R.A. Mittermeier, H. Randriamahazo, R. Hudson and R.E. Lewis (Eds.), pp. 153 –161. *Chelonian Research Monographs* 6.

Walker, R.C.J., **Gardner, C.J**., Rafeliarisoa, T.H., Smith, I., Razafimanatsoa, R. and Louis Jr., E.E. 2013. Conservation of the Madagascar spider tortoise (*Pyxis arachnoides*) amid changing land use policy: assessing the spatial coincidence of relict populations with protected areas and mining concessions. In *Turtles on the Brink in Madagascar: Proceedings of Two Workshops on the Status, Conservation, and Biology of Malagasy Tortoises and Freshwater Turtles*, C.M. Castellano, A.G.J. Rhodin, M. Ogle, R.A. Mittermeier, H. Randriamahazo, R. Hudson and R.E. Lewis (Eds.), pp. 135–145. *Chelonian Research Monographs* 6.

Gardner, C. and Jasper, L. 2013. New locality records for *Boophis xerophilus* and *B. doulioti* (Anura: Mantellidae) from Bezà Mahafaly Special Reserve, southwest Madagascar. *Herpetology Notes* 6: 477–478.

Andriamalala, G., Peabody, S., **Gardner, C.J.** and Westerman, K. 2013. Using social marketing to foster sustainable behaviour in traditional fishing communities of southwest Madagascar. *Conservation Evidence* 10: 37–41.

Westerman, K. and **Gardner, C.J**. 2013. Adoption of socio-cultural norms to increase community compliance in permanent marine reserves in southwest Madagascar. *Conservation Evidence* 10: 4–9.

Gardner, C. 2013. Observation of reed bed use by *Furcifer verrucosus* (Squamata: Chamaeleonidae) in southwest Madagascar. *Herpetology Notes* 6: 21–22

Raselimanana, A.P., Raherilalao, M.J., Soarimalala, V., **Gardner, C.J.**, Jasper, L.D., Schoeman, M.C. and Goodman, S.M. 2012. Un premier aperçu de la faune de vertébrés du bush épineux de Salary-Bekodoy, à l'ouest du Parc National de Mikea, Madagascar. *Malagasy Nature* 6: 1–23.

Gardner, C. and Jasper, L. 2012. *Paroedura picta* in southern Madagascar: diet and predation by the introduced *Hemidactylus frenatus*. *Herpetology Notes* 5: 457–458.

Gardner, C.J. 2012. Social learning and the researcher-practitioner divide. *Oryx* 46: 313 –314.

Gardner, C. and Jasper, L. 2012. Cannibalism in *Chalarodon madagascariensis* (Squamata: Iguanidae) from southwest Madagascar. *Herpetology Notes* 5: 127–128.

Gardner, C.J., De Ridder, C., De Ridder, B. and Jasper, L. 2012. Birds of Ambondrolava mangrove complex, southwest Madagascar. *Check List* 8: 1–7.

Gardner, C., Jasper, L. and Vincke, X. 2011. Sakalava weaver nesting association with raptors: an alternative hypothesis. *Malagasy Nature* 5: 129–131.

Gardner, C.J. and Jasper, L. 2011. Status of barn swallow *Hirundo rustica* in south-west Madagascar. *Bulletin of the African Bird Club* 18: 207–210.

Gardner, C., Jasper, L. and Razafinarivo, N. 2011. A new, isolated population of *Oplurus* (Iguanidae) from Tsingy de Bemaraha National Park, western Madagascar. *Herpetology Notes* 4: 253–254.

Ferguson, B. and **Gardner, C.J.** 2010. Looking back and thinking ahead – where next for conservation in Madagascar? *Madagascar Conservation & Development* 5: 75–76.

Andriamalala, G. and **Gardner, C.J.** 2010. L'utilisation du *dina* comme outil de gouvernance des ressources naturelles: leçons tirés de Velondriake, sud-ouest de Madagascar. *Tropical Conservation Science* 3: 447–472.

Irwin, M.T., Wright, P.C., Birkinshaw, C., Fisher, B., **Gardner, C.J.**, Glos, J., Goodman, S.M., Loiselle, P., Rabeson, P., Raharison, J.-L., Raherilalao, M.J., Rakotondravony, D., Raselimanana, A., Ratsimbazafy, J., Sparks, J., Wilmé, L. and Ganzhorn, J.U. 2010. Patterns of species change in anthropogenically disturbed habitats of Madagascar. *Biological Conservation* 142: 2351–2362.

Gardner, C. and Jasper, L. 2010. New locality records for *Phelsuma breviceps* and *Cryptoblepharus boutonii* ssp. from Andavadoaka, southwest Madagascar. *Herpetology Notes* 3: 233–235.

Gardner, C.J. and Jasper, L. 2009. Possible new *Mirza* taxon from the Fiherenana River valley, Atsimo-Andrefana Region. *Lemur News* 14: 46–49.

Gardner, C. and Jasper, L. 2009. The urban herpetofauna of Toliara, southwest Madagascar. *Herpetology Notes* 2: 239–242.