Is There a Future for the Amphibians of the Ankaratra Massif Reserve? Understanding the Role of Landscape Change

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MSc

Todas as correções determinadas pelo júri, e só essas, foram efetuadas.

O Presidente do Júri,

Porto, ______/______/_________

“Madagascar, like Africa, is not a place for the faint-hearted.”

David A. Burney

In the forword of “Extinct Madagascar”

“When you realize the value of all life, you dwell less on what is past and concentrate more on the preservation of the future.”

Dian Fossey

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Ericeira, 23 de Julho de 2017

Resumo

Madagáscar é uma das regiões do mundo onde se pode encontrar uma riqueza inigualável em fauna e flora endémicas. Esta alta percentagem de endemismo é resultado da sua longa evolução enquanto ilha, sem ligações continentais desde que se separou da Índia, há aproximadamente 66 a 90 milhões de anos. Esta biodiversidade única é também fruto de uma geologia estável e de uma grande variabilidade climática ao longo da ilha. Infelizmente, Madagáscar está entre as regiões do mundo que mais estão a sofrer pela degradação dos seus habitats.

Contando com mais de 500 espécies existentes em Madagáscar, os anfíbios são um dos grupos faunísticos mais distintos aqui, sendo de uma diversidade extrema e tendo-se adaptado a todos os biomas desta ilha. Apesar disto, estes anfíbios estão em declínio no número de espécies e de populações. A degradação e perda de habitats causados pela desflorestação continuam a ser uma das principais razões para este declínio.

A Reserva do Ankaratra Massif situa-se nas planícies centrais do Madagáscar e tem sofrido longamente por degradação dos seus recursos naturais, devido à exploração ilegal de carvão, métodos de agricultura de corte e queima e pela exploração económica de uma plantação de pinheiro. Esta Reserva é muito recente (oficialmente reconhecida em 2015) e foi criada com o intuito de proteger três espécies microendémicas que apenas se encontram neste lugar – um réptil e dois anfíbios. O Ankaratra Masssif sustenta pradarias, savanas e floresta tropical de montanha. Para além disto, existem também terras de cultivo e plantações alóctones de pinheiro, como já foi mencionado.

Estando esta Reserva incorporada na Alliance of Zero Extinction, é agora necessário identificar os padrões de desflorestação e a interação entre a comunidade de anfíbios aqui presente e o seu habitat. Este estudo pretende aumentar o conhecimento desta região através (1) da caracterização da comunidade de anfíbios de ribeiros de grandes altitudes, (2) da avaliação das dinâmicas dos padrões de desflorestação e das interações entre diferentes tipos vegetativos, (3) da avaliação de mudanças nas funções do ecossistema entre 1985 e 2016, e através (4) da análise de como mudanças estruturais no ecossistema poderão afetar a diversidade de anfíbios presentes no Ankaratra Massif.

Um recenseamento de anfíbios ao longo de um período de dois anos foi realizado ao longo de troços de água de altas altitudes (acima dos 2000m acima do nível médio da água do mar). Para detetar a ocorrência de desflorestação e investigar alterações funcionais no ecossistema entre 1985 e 2016, foi utilizado o sistema de deteção remota através de satélites. Para obter os mapas de vegetação desde 1985, foi aplicada um método de classificação de imagens onde são definidas *à priori* as categorias vegetativas desejadas (supervised image classification). Dois índices de vegetação (NDVI e Albedo) foram usados para estimar alterações na produtividade primária e na temperatura da superfície terrestre. Uma abordagem de modelos múltiplos (multi-model inference) foi desempenhada para estimar a resposta desta comunidade de anfíbios às alterações passadas e presentes que ocorrem no seu habitat.

Este estudo revela uma paisagem intensamente dinâmica, que parece sofrer bastante aquando do aumento da instabilidade política nacional. A comunidade de anfíbios do Ankaratra Massif parece ser composta por espécies extremamente adaptáveis a diferentes habitats, com exceção das duas rãs microendémicas (*Boophis williamsi* e *Mantidactylus pauliani*) que estão consideravelmente adaptadas ao seu habitat de topo de montanha. Os valores de riqueza específica, abundância e capacidade de renovação da sua população existentes hoje em dia no Ankaratra Massif são fruto de alterações passadas e presentes nos processos funcionais e estruturais desta ecorregião ameaçada de extinção. Assim, para assegurar a sobrevivência das espécies únicas que aqui habitam, é de extrema importância que seja criado um plano eficiente de gestão da Reserva, que alie o rigor científico de estudos existentes sobre o assunto com o apoio da população local, pois são estas pessoas que têm o poder de salvaguardar o futuro da Reserva do Ankaratra Massif no seu dia-a-dia.

Palavras-chave

Madagáscar, Planalto Central, Floresta Tropical de Montanha, Anfíbios, Endemismo, Desflorestação, NDVI, Albedo, Deteção Remota

Abstract

Madagascar’s long isolation from continental land, its geological stability and climatic disparity has resulted in it being one of the most important regions in the world in percentage of endemic fauna and flora. Unfortunately, it is also among the regions experiencing exceptional habitat degradation. One of the most unique groups of animals present here are the amphibians, a largely diverse group of vertebrates that has adapted to all of the island’s biomes. With more than 500 known amphibian species and despite this large diversity, Malagasy amphibians are following the global trend of broad decline both in species and in population numbers. Habitat degradation or loss caused by deforestation still remains one of the main reasons for this decline.

The Ankaratra Massif Reserve is situated in Madagascar’s Central Highlands and has been suffering degradation to a large extent due to illicit charcoal exploitation, uncontrolled slash and burn agriculture and due to the economic exploitation of pinewood. The Reserve is very recent (officially recognized only in 2015) and was created mostly to protect the three critically endangered microendemic species it is home to – one reptile and two amphibians. The Massif supports Tropical Montane Cloud Forest, savannah and grasslands as well as a non-native pine plantation that is economically exploited and agricultural lands.

Incorporated in the Alliance of Zero Extinction, it is now critical to identify the patterns of deforestation and the interactions between the amphibian community living there and the habitat they live in. This study helps to decrease the existing knowledge gap by (1) characterizing the high-altitude amphibian community associated with montane streams, (2) estimating land cover dynamics and deforestation rates from 1985 to 2016, (3) assessing changes in the ecosystem functioning variables from 1985 to 2016 and by (4) analyzing how structural changes in the ecosystem might have affected the amphibian diversity of the Ankaratra Massif.

A 2-year survey on amphibian occurrence and abundances was conducted along high-elevation streams (above 2000m above sea level) and Satellite Remote Sensing was used to detect deforestation and functional changes in the study area from 1985 to 2016. Supervised image classification was used to attain land cover/land use maps since 1985. Two satellite-derived vegetation indices (NDVI and Albedo) were used to assess changes in vegetation productivity and in land surface temperature and a multi-model inference approach was performed to estimate how the surveyed amphibian community is responding to these changes in the environment.

This study reveals the highly dynamic landscape of the Ankaratra Massif, which seems to suffer intensified pressure whenever the political instability of the country increases. Its amphibian community seems to be composed of highly adaptable species except for the two microendemic species (*Boophis williamsi* and *Mantidactylus pauliani*) which seem to be much more specialized to this mountain-top habitat. Present-day values of amphibian richness, abundances and turnover capacity are to some extent a result of past and present changes in structural and functional processes of this threatened ecoregion. To ensure the survival of these unique species, an efficient management plan needs to be set up, taking into account rigorous scientific knowledge combined with the support of the local population, who are the ones in the best geographical position to safeguard the future of the Ankaratra Massif Reserve.

Keywords

Madagascar, Central Highlands, Tropical Montane Cloud Forest, Amphibians, Endemism, Deforestation, NDVI, Albedo, Remote Sensing

Index

List of Figures

List of Tables

List of Abbreviations

AIC – Akaike Information Criterion

AICc – Corrected Akaike Information Criterion

ASA – Amphibian Survival Alliance

ASG – Amphibian Specialist Group

a.s.l. – above sea level

Bd – *Batrachochytrium dendrobatidis*

CR – Critically Endangered

DN – Digital Numbers

DOPA – Digital Observatory for Protected Areas

EDGE – Evolutionary Distinct & Globally Endangered

EFV – Ecosystem Functioning Variable

GAA – Global Amphibian Assessment

GLM – Generalized Linear Model

GPL – Global Panzootic Lineage

IUCN – International Union for Conservation of Nature

LC – Least Concern

LCC – Land Cover Class

ME – Microendemism Rate

MMI – Multi-Model Inference

Mya – Millions of years ago

N – Specimen abundance

NDVI – Normalized Difference Vegetation Index

NE – Not Evaluated

NIR – Near Infrared

NT – Near Threatened

OLI – Operational Land Imager

OOB – “out-of-bag”

PIFs – Pseudo Invariant Features

RF – Random Forests

SAVI – Soil Adjusted Vegetation Index

SR – Species Richness

SRS – Satellite Remote Sensing

SSC – Species Survival Commission

STC – Species Turnover Capacity

SWIR – Shortwave Infrared

TIRS – Thermal Infrared Sensor

TM – Thematic Mapper

TMCF – Tropical Montane Cloud Forest

ToA – Top of Atmosphere

UFA – Union Forestière d’Ambatolampy

VIF – Association Vondrona Ivon’ny Fampandrosoanavif

VIFs – Variation Inflation Factors

USGS – United States Geological Survey

WWF – World Wildlife Fund

1. Introduction

1.1 Madagascar, Biodiversity's Paradise

Madagascar is the fourth largest island in the world, after Greenland, New Guinea and Borneo, being approximately 590,000km^2 in size. Originally part of the supercontinent Gondwana, it split from Africa around 160 Mya and from India approximately 66-90 Mya, having had no mainland connections ever since (Kusky et al. 2007; Ali and Aitchison, 2008). Nowadays, Madagascar is climatically very diverse and, consequently and contrary to most large tropical islands, has extreme major biomes which exhibit well-defined borders between them (Vences et al., 2009). These biomes go from tropical humid forests in the north and east to dry deciduous forests in the west, to subarid spiny forests and shrubland in the south.

This isolation, geological stability and climate disparity has resulted in Madagascar being the number one region in percentage of endemic fauna and flora and being among the 3 richest hotspots (area displaying extraordinary concentrations of endemic species and suffering exceptional degradation of natural habitat) on Earth (Myers et al., 2000; Wilmé et al., 2006). Madagascar is characterized by being home to a great diversity of species, as for example, within the amphibians and reptiles, and on the other hand by the complete absence of other groups otherwise globally distributed, such as the inexistence of groups such as canids, felids, cervids, bovids and anthropoid primates (see Dewar and Richard, 2007). This demonstration of Madagascar's unique biodiversity extends to all fauna and flora and its evolution is still a subject that generates a lot of questions, although there is great evidence that its biota largely evolved in isolation, with a restricted number of colonisations by overseas dispersal (Vences et al., 2003; Samonds et al., 2012), the most recent ones coming from Africa (Crottini et al., 2012). Thus, it has been suggested that the extant species richness has mainly resulted from within-island speciation processes rather than from immigration (Crottini et al., 2012). One of the most unique groups of animals present here are the amphibians, being 99.9% endemic (although the only non-endemic species – *Ptychadena mascariensis* – is already known to be a complex of three candidate new species, Zimkus et al., 2017) and having radiated to all the island's biomes, spreading throughout 313 named species (as of February 2017; Scherz, 2017), with new ones being discovered every year and others still waiting to be named and described (Vieites et al., 2009; Rosa et al., 2012; Perl et al., 2014).

For centuries, humans have admired frogs for their beauty and for their utility to mankind in the most varied fields, from the use for food resources to tribal hunting and rituals, from pharmaceutical uses and discoveries to medical teaching and research (Tyler et al., 2007). Despite this great fascination and usefulness, amphibians worldwide are in broad decline, being the most threatened group in the IUCN Red List (Hoffmann et al., 2010) with 32% known to be threatened or extinct and 25% had insufficient data to be assessed at the time the last Global Amphibian Assessment report was presented (GAA, updated in 2008). Since then, numbers are expected to have risen, as 42% of assessed species were suffering from population decline at the time of the GAA report.

In addition to these alarming numbers, in recent years dozens of amphibian species worldwide have been decimated and driven to extinction by the global spread of a deadly fungus species (Skerratt et al., 2007), *Batrachochytrium dendrobatidis* (*Bd*). There exist various lineages of this fungus, one of which (the Global Panzootic Lineage, GPL) is the disease-causing lineage. The disease causes death through cardiac arrest motivated by compromised osmotic regulation due to *Bd*-infected thickened skin (Voyles et al., 2009). In the past few years, a new fungal pathogen of the same genera was described, *Batrachochytrium salamandrivorans,* causing the same symptoms as Bd in salamanders (Martel et al., 2013).

Recent studies (e.g. Bletz et al., 2015a; Kolby and Skerratt, 2015) reported on the widespread presence of the chytridiomycosis causing fungus in Madagascar. DNA screening shows that the lineage present in Madagascar is very similar to the GPL (Bletz et al., 2015a).

More than 4% of the total worldwide described amphibian diversity can be found in Madagascar (AmphibiaWeb, as of September 19th, 2017), at least one fourth of which is threatened with extinction (Andreone and Randriamahazo, 2008). All amphibians found here are anurans from the superfamily Ranoidea (Glaw and Vences, 2003), except for the invasive black-spined toad (*Duttaphrynus melanostictus*) that has been recently and accidentally introduced from southeast Asia into Madagascar (Vences et al., 2017).

Malagasy frogs are highly diversified, displaying all sorts of mimetisms and having evolved numerous unique reproductive strategies (Glaw and Vences, 2007; Andreone et al., 2010). Regional endemism, or microendemism (species known from five or less locations; Glaw and Vences, 2003), is very high, especially in high-elevation habitats. Although the high degree of microendemism found throughout Madagascar still remains poorly understood (Vences et al., 2009), there is some evidence that it occurs through specialization to specific environments, restricting dispersal ability in heterogeneous environments (Wilmé et al., 2006). Nowadays, these environments are becoming increasingly fragmented and this astonishing natural legacy has been continuously under anthropogenic pressure right from the beginning of human settlement (Goodman and Jungers, 2014). A large number of Malagasy amphibians are forest-specialists, which makes them very vulnerable to deforestation, being likely that at least some unknown frog species have already become extinct (Andreone and Luiselli, 2003).

Human settlement in Madagascar is thought to have happened at least 2000 years ago (Randrianja and Ellis, 2009; Goodman and Jungers, 2014) and nowadays only an estimated 10% of its natural habitats remain (Goodman and Benstead, 2005). It has been estimated that the island was originally covered with approximately 11 million hectares of primary rainforest, having been reduced to about 3.8 million hectares by 1985 (see Glaw and Vences, 2007). The background for this extreme rate of forest clearance has been remained the same throughout the years, led by the socio-economic needs of the population (Goodman and Jungers, 2014). This has worsened in recent years, as the Malagasy population is growing exponentially (Myers, 1993) and where 80% still depend solely on subsistence farming and on the use of charcoal for cooking fuel (Ecosia, 2017). Despite restricted extent of occurrence caused by deforestation and habitat loss being the main threat to Malagasy amphibians (Andreone et al., 2005; Andreone et al., 2007), other menaces (which are not kept back by the borders of protected-areas) exist, such as environmental contamination, disease, the invasion by exotic species and climate change, and, to a smaller extent, illegal pet trade (Glaw and Vences, 2003; Moore and Church, 2008). Moreover, amphibians are extremely sensitive to environmental changes in their habitat, especially due to their characteristic cutaneous respiration and to their complete exposure to the abiotic conditions in their niche during ontogenesis. For all these reasons, Madagascar is currently among the world’s most critical conservation priorities (Goodman and Benstead, 2005).

Protecting amphibian habitat will improve water sources (Hocking and Babbitt, 2014) helping, as a natural consequence, the local populations who depend on them and will preserve complex food-webs, as amphibians generally play an important role both as prey and as predators across their whole life cycle (Glaw and Vences, 2003). Furthermore, the higher the amount of frogs present in an ecosystem, the better the insect populations can be controlled (Leonard, 1993) – insects that cause disease and are a menace to agriculture. Additionally, the finding of new amphibian species can lead to new and potentially important discoveries (e.g. of new toxins important for pharmacological studies; see Tyler et al., 2007).

In Madagascar, coordinated amphibian conservation planning was launched as a result of a workshop which took place in Antananarivo, September 2006. This meeting was attended by national and international experts and institutions and resulted in the publication of the first Sahonagasy Action Plan (Andreone and Randriamahazo, 2008) and in the appointment of an official position for the Amphibian Executive Secretariat, to coordinate and prioritize amphibian conservation and research plans in Madagascar. This action plan acknowledges that special attention to monitoring should be given to sites along elevational transects, as montane species are particularly vulnerable to extinction from upslope displacement due to global warming.

1.1.1 The Central Highlands

Madagascar's Central Highlands (or the High Plateau), as the name illustrates, consists of a mountain system with a north-south orientation in the centre of the island. Once inhabited by now extinct iconic species such as elephant birds (Aepyornithidae), giant lemurs, Madagascan Dwarf hippopotamus and a giant species of tortoise (Goodman and Jungers, 2014), people have largely taken over this part of the island, as it is now one of the most densely populated regions of the country (Rainforest Trust, 2016). Humans began to settle here at least one millennia ago (Dewar and Wright, 1993), starting to change it into the greatly anthropogenic landscape seen today. Therefore, the wide belief that its natural habitats have been completely ruined has led to a huge lack of research and conservation plans in this area (Raxworthy and Nussbaum, 1996).

Montane forest and woody formations once covered much of this region (Ganzhorn et al., 2001) but present-day vegetation in Madagascar's highlands is typically patchy, often characterized by low density and largely exotic patches of landscape, comprised of introduced eucalyptus, pines, acacias and fruit trees, that are increasing significantly in area (McConnell et al., 2015). Montane forest in the Central Highlands flourished during the climatic oscillations of the Pleistocene (Burney, 1996), having decreased to a forest-woodland-grassland mosaic during the early and middle Holocene (Burney, 1987). Forest-adapted taxa then underwent a drastic range decrease approximately 4000yrbp in a pre-human environment (Burney, 1999). Nowadays the montane forest is restricted to the highest mountain regions and during the interglacial periods it might have act as refugia for cold-adapted fauna and flora species (Burney, 1996). Both montane forest and montane heathland are classified by the World Wildlife Fund (WWF) as critically endangered ecoregions (Crowley, 2000a; Crowley, 2000b).

The island's three highest peaks (Maromokotro, on the Tsaratanana Massif, Boby Peak, on the Andringitra Massif, and Tsiafajavona, on the Ankaratra Massif) lie along the central plateau and harbour the last fragmented remains of montane humid forest. White (1983) describes four types of primary montane vegetation: moist montane forest in less exposed areas, with the presence of mosses; sclerophyllous montane forest with the presence of lichen; montane heathland on exposed ridges above 2000m and rupicolous shrubland restricted to rocky scenarios. Due to the frequent burning of these mountains across the last century, a great part of this vegetation is no longer primary (Raxworthy and Nussbaum, 1996).

Although Madagascar has been at the centre of a considerable number of studies on biodiversity quantification and preservation, only a small fraction of these studies has focused on the Highlands and an even smaller fraction has focused on the high montane domain (Gardner, 2009). As mentioned above, their current patchy land cover, largely dominated by exotic species, has been the cause of mostly being ignored by conservationists due to their lesser biological value (McConnell et al., 2015). However, even though primary vegetation has nearly disappeared, this type of land cover is relevant to soil and water conservation, carbon budgets and other aspects of environmental management. For a proper management of wildlife resources, it is therefore essential to characterize the transition in vegetation that is shaping the landscape of these highlands.

1.1.2 The Ankaratra Massif

The Ankaratra Massif is situated in the central high plains of Madagascar, 72km southwest of the capital, and it includes the third highest peak of the island at 2643m above sea level (a.s.l.). It is an extinct volcanic range, extending over an area of approximately 2500 km^2 and originally covered by a great amount of montane forest [also known as Tropical Montane Cloud Forest (TMCF) when talking about the tropics, due to the recurrent occurrence of low cloud cover] but it is now suffering high degradation rates. Part of the forest is replaced by grassland areas, intertwined with plots of montane heathland dominated by shrubs, especially in areas above 2000m a.s.l. (Vences et al., 2002; also see Burney, 1996) This transition from montane sclerophyllous forest to montane ericoid thicket is also visible on Madagascar's other three Massifs (Crowley, 2000a): Tsaratanana in the north (2876m a.s.l.), Andringitra in the south (2658m a.s.l.) and Marojejy in the northeast (2133m a.s.l.). All these massifs have been Nature Reserves for much longer than Ankaratra, and thus have been studied much more intensively.

The climate in the Massif follows the general seasonal pattern for Madagascar, having a dry and cooler season from May to October and a warm and wet season from November to April.

Deforestation is a big problem in this region, mainly due to illicit charcoal exploitation, uncontrolled slash and burn agricultural methods (Fig. 1.1) and economic exploitation of the area for pinewood plantation. This said, the native forest that has survived in Ankaratra has done so thanks to the existence of the Manjakatompo Forest Station (Gade, 1996). However, Manjakatompo Forest Station covers only 6.5km^2 and has no management plan (Crowley, 2000a).

Figure 1.1 – Satellite image of fire in our study area (1995). The darker area in the centre of the image, under the smoke, is the higher elevation area of the Ankaratra Massif.

Following the pattern described by Jenkins (1987) for Madagascar's massifs, Ankaratra bares poor species richness but significant endemism that is solely confined to this Massif. Here, the borders of a new Nature Reserve were identified in 2010 and officially recognized and established in August 2015 (Amphibian Survival Alliance, 2015). As one of the few Nature Reserves in Madagascar that do not host any lemur species, this Reserve was specifically established to protect three Critically Endangered species: two amphibians (*Boophis williamsi* and *Mantidactylus pauliani*), that are probably the most threatened species of amphibians in Madagascar, and one gecko species (*Lygodactylus mirabilis*).

The Ankaratra Massif is home to 15 species of amphibians (Table 1.1), two of which are microendemic to this mountain range, existing nowhere else in the world: the Williams' Bright-eyed frog, *Boophis williamsi,* (Guibé 1974) and the Madagascar frog *Mantidactylus pauliani,* Guibé 1974 (Vences et al., 2002). They are both classified as critically endangered (CR), following IUCN Red Listing Criteria (IUCN, 2016e, 2016h), and *M. pauliani* is listed as an EDGE priority species. These two species are thought to have originally inhabited a large portion of the montane forest of the central highlands of Madagascar (Glaw and Vences, 2003) but nowadays are only found along a few streams in high elevation grassland (2000-2400m a.s.l.) with relict forest patches (Andreone et al. 2005; IUCN, 2016a, 2016b) and in open savannah areas (Rabemananjara, 2011). Hardly anything is known about the biology and the ecology of these two species. *Boophis williamsi* and *Mantidactylus pauliani* spent thirty years since their discovery without being reported (Vences et al., 2002) and only recently some aspects of these species' life history have been unveiled, such as their longevity and sexual maturity (Andreone et al., 2014). Other issues related to their basic biology and ecology such as their feeding behaviour, reproductive habits or development remain unknown.

During the past decade, the habitat of these species has undergone severe habitat loss and contamination, particularly after great part of the allochthonous pinewood forest was burned in 2010 (subsequent to political instability and conflicts between villagers, ACSAM, 2011), which together with the other forms of deforestation mentioned above, lead to water pollution in the streams where these CR species inhabit and breed (Rabemananjara, 2011). Moreover, similarly to what is happening in other tropical montane habitats, they may be affected by upslope displacement due to global warming (Raxworthy et al., 2008; Hirschfeld et al., 2016).

Table 1.1 – Amphibian species found on the Ankaratra Massif and respective conservation status.

Species	IUCN Criteria
Blommersia domerguei (Guibé, 1974)	LC
Blommersia kely (Glaw & Vences, 1994)	LC
Boophis ankaratra Andreone, 1993	LC
Boophis goudotii Tschudi, 1838	LC
Boophis microtympanum (Boettger, 1881)	LC
Boophis williamsi (Guibé, 1974)	CR
Heterixalus betsileo (Grandidier, 1872)	LC
Heterixalus rutenbergi (Boettger, 1881)	LC
Mantidactylus alutus (Peracca, 1893)	LC
Mantidactylus brevipalmatus Ahl, 1929	LC
Mantidactylus pauliani Guibé, 1974	CR
Mantidactylus sp. aff *curtus Ca19*	NE
Plethodontohyla tuberata (Peters, 1883)	NT
Ptychadena mascareniensis (Duméril & Bibron, 1841)	LC
Scaphiophryne madagascariensis (Boulenger, 1882)	NT

Regular monitoring by several research teams since 2001 have shown that *B. williamsi* is very rare, with less than 10 adult individuals found per 400m to 1000m of stream transect, while *M. pauliani* has shown higher densities. It is not clear whether the reason for this difference in abundances is due to their different ecology. *B. williamsi* is supposed to be an arboricole frog, inhabiting areas of rudimentary gallery forest and bushy vegetation along fast-flowing streams where the species reproduces during breeding season, whilst *M. pauliani* is an aquatic frog with some degree of site fidelity, generally found on rocks along fast-flowing streams (Vences et al., 2002). Preserving both high savannah and the adjacent natural forest is critical for ensuring the survival of these two species.

From the beginning, Ankaratra has been one of the eight focus sites of the National Monitoring Program for the early detection of *Bd* (Weldon et al., 2013) as it is a high-altitude site (where *Bd* is more likely to occur and where it could be more virulent) and as it hosts the two microendemic and CR frog species mentioned above. Also, being close to the capital it is suggested to have high potential for becoming a perfect study site where disease dynamics and conservation measurements can be put in place. Since 2012, *Bd* has been repeatedly reported from this site (Bletz et al., 2015a; Kolby and Skerratt, 2015) and, to date, two expeditions have been organized to this location to isolate the *Bd* strain occurring in Madagascar in order to perform infection trials, test for its virulence and assess its risks on the native amphibian fauna of Madagascar. Additionally, *Ranavirus* has already been detected in Ankaratra (Kolby et al., 2015).

1.2 Satellite Remote Sensing, Land Cover and Ecosystem Functioning

The threats facing biodiversity, the challenges for conservation and the need to report and understand the processes that shape biodiversity at a regional to a global scale have led to the enhancement of the use of earth observation technology, such as Satellite Remote Sensing, in the analysis and understanding of these issues.

Species are sensitive to changes in multiple aspects of landscapes (e.g. forest structure, productivity or thermal variation). Habitat degradation and destruction occurs not only via the loss of optimal area, but also via changes in its functional attributes, which likely affect population dynamics, by alteration of reproduction and migration behaviours due to new environmental factors.

The flux of energy, nutrients and organic matter through an ecosystem (denominated as ecosystem functioning, Valentini et al. 1999), shows a shorter response to environmental changes than vegetation structure (Wiegand et al., 2004). This flux of energy can be directly and indirectly measured through various processes, such as plant primary production, nutrient recycling, ecosystem gas exchange, evapotranspiration and decomposition cycles. The measurement of these processes can then provide us with indirect information on the health and complexity of the entire ecosystem under study.

Ecosystem functioning can be measured through defined biophysical characteristics, named Ecosystem Functional Attributes (EFAs, Alcaraz-Segura et al. 2009). These, in turn, can be indirectly measured from space, with the help of a satellite network. This means that there are a set of proxies which can be measured to help assess different EFAs, such as the estimation of plant primary productivity via the way plants absorb and reflect visible and infrared light waves. The most common measurement of functional vegetation condition is called Normalized Difference Vegetation Index (NDVI, Rouse 1974), which is linked to vegetation cover, biomass and net primary productivity (Reed et al., 1994; Rocchini et al., 2016), being associated with the highest number of remote sensed ecological applications (Pettorelli, 2013). This index makes use of the large amount of sunlight in the red wave length range (0.4 to 0.7µm) that is absorbed by the chlorophyll for photosynthesis, and of the near-infrared light (0.7 to 1.1µm) that is reflected back into the atmosphere by the cell structure on leaves (Jensen 2007). Based on this, it is possible to determine the amount of vegetation growth in a determined study area with the use of reflectance data collected by satellites. As well as being the mostly used remote sensing index in ecology, NDVI was also the first index to be used to produce global maps of the Earth's biophysical cover (DeFries and Townshend, 1994). Since then, NDVI has been proved to efficiently characterize vegetation functioning at a regional scale and across different time ranges, such as seasonal and annual dynamics (Alcaraz et al., 2006). It has also been shown that this index can provide essential information on vegetation dynamics, allowing research to be made on the relationships between animal populations and environmental variability (Pettorelli et al., 2011).

The Soil Adjusted Vegetation Index (SAVI; Huete 1988) complements the NDVI by correcting for the influence of soil brightness where vegetative cover is low. This index is also based on reflectance properties captured by satellite sensors. In places where vegetative cover is low, leaving the soil surface exposed, the reflectance of light caught by the satellite may influence vegetation index values. Thus, Huete (1988) developed a modification to the NDVI formula, introducing the soil brightness factor (L), allowing for comparisons to be made between studies across different soil types. This way, different soil types, with different properties that reflect more or less light back to the satellites, will not influence NDVI values across studies. SAVI is found to be mostly useful in intermediately vegetated areas.

NDVI and SAVI are both satellite-based vegetation indices, where the obtained signal depends on the condition of the vegetation at the time it was acquired (Wegmann and Leutner, 2016). The thermal imprint of a landscape is another important functional component for biodiversity. Different metrics can be estimated to illustrate the thermal characteristics of an ecosystem, varying according to method and to satellite platform considered. Land surface Albedo is a proxy for thermal variation, measured by the amount of radiation reflected from the Earth’s surface back into the atmosphere. This varies with land surface materials: the darker the material, the less it reflects back into the atmosphere.

The ability of satellite derived vegetation and thermal indices to out-perform widely used environmental predictors, such as precipitation patterns in predicting species distributions, is being increasingly recognized (Rasmussen et al., 2006) especially as climate patterns are altering at an extensive rate. Therefore, underlying ecological regulators of species distributions will have to be taken into account in future studies. Our ability to conserve biodiversity will depend on our capacity to understand the role of these ecological regulators and anticipate their effect on the composition of species communities (Cabello et al., 2012). The use of Satellite Remote Sensing (SRS) is greatly expanding this capacity. Its use in biodiversity and resource management and monitoring is strongly expanding, while communication and interdisciplinary between conservationists, wildlife managers and the remote sensing community is also increasing (Pettorelli et al., 2014a). Satellite imagery is extremely useful for analysing and classifying environmental conditions and for detecting changes in land cover and land use, improving monitoring and warning systems (areas of potential biodiversity change) of protected areas (Duro et al., 2007). Satellite imagery is increasingly freely available, having a high temporal and spatial resolution and giving researchers access to reliable global information on spatio-temporal changes in ecosystems. This allows for a better understanding of anthropogenic pressures and gives evidence on the effectiveness of various conservation programmes and management plans (Pettorelli et al., 2014a). SRS is also very useful when field data is limited and difficult to collect, due to, for example, economic reasons, political instability or the large range of the study area.

When using SRS-based information, two important terms need to be introduced – Land Cover and Land Use. The first, Land Cover, refers to the types of vegetation or materials that cover the Earth’s surface. Secondly, we have Land Use which specifies the functional roles that land plays in economic activities (Campbell, 1983).

In this study, satellite data from the Landsat satellite network platform was used to gather images and functional information, as it provides one of the most extensive and continuous imagery archives, dating back to 1972.

1.3 Objectives

Being a region of high priority for conservation, incorporated in the Alliance of Zero Extinction programme (Langaha 2010), the Ankatrata Nature Reserve has been developed by Malagasy authorities with the purpose of protecting its particular native wildlife. Even though it has recently become a protected area, little work has been done on the Ankaratra Massif to increase the knowledge on its ecological patterns. Despite being a place that has suffered a huge change in vegetation cover in the recent past and has been undergoing intensive anthropogenic exploitation, land use dynamics and biological interactions between different land cover classes have barely been studied. The conservation organization “Association Vondrona Ivon’ny Fampandrosoanavif (VIF)” is now supposed to manage the reserve through forest guard units assembled in collaboration with local communities. These units regularly patrol the reserve to stop illegal logging and have effectively decreased logging rates. However, it is now critically important to associate these efforts with a scientifically rigorous characterization of forest dynamics so that the current efforts in protecting these last remaining patches of montane forest can be better directed.

To safeguard the CR species from extinction we need to understand spatial and temporal land cover and land use dynamics in order to improve land management policies, minimizing conflicts between local people and endemic biodiversity.

Therefore, five main objectives were established:

(1) Analyze the diversity of the high-altitude amphibian community of the Ankaratra Massif Reserve, based on a two-year survey (2011-2013);
(2) Characterize landscape dynamics from 1985 to 2016, using multi-temporal analyses on land cover change and on landscape functioning, based on satellite images;
(3) Map deforestation and assess deforestation rates throughout the years under analysis;

(4) Analyse the responses of the amphibian community to changes in landscape conditions considering the four chosen biodiversity metrics (species richness, amphibian abundance, microendemism rate and species turnover capacity) and three main processes (deforestation, primary productivity and thermal variation);

(5) Discuss the implications of the findings for the future management of the Ankaratra Massif Reserve.

2. Material and Methods

2.1 Study Area

The study area spreads over 158,42km^2 and 77% of the Ankaratra Massif Reserve, Madagascar (Fig. 2.1). The Ankaratra Massif has been identified for its important resources such as water supplies and unique biodiversity since 1960, being a region of high priority for conservation (Rahantaliosa et al., 2011). Elevation ranges from 1514m above sea level (a.s.l) to 2643m a.s.l, the third highest peak in Madagascar.

The climate on the Massif follows the general seasonal pattern for Madagascar, having a dry and cooler season from May to October and a warm and wet season from November to April. The colder and dryer months are from June to August whereas the hotter and rainier months are December to February. In Ankaratra, winter temperatures can drop to below zero at night (Rakotozafy, 2017).

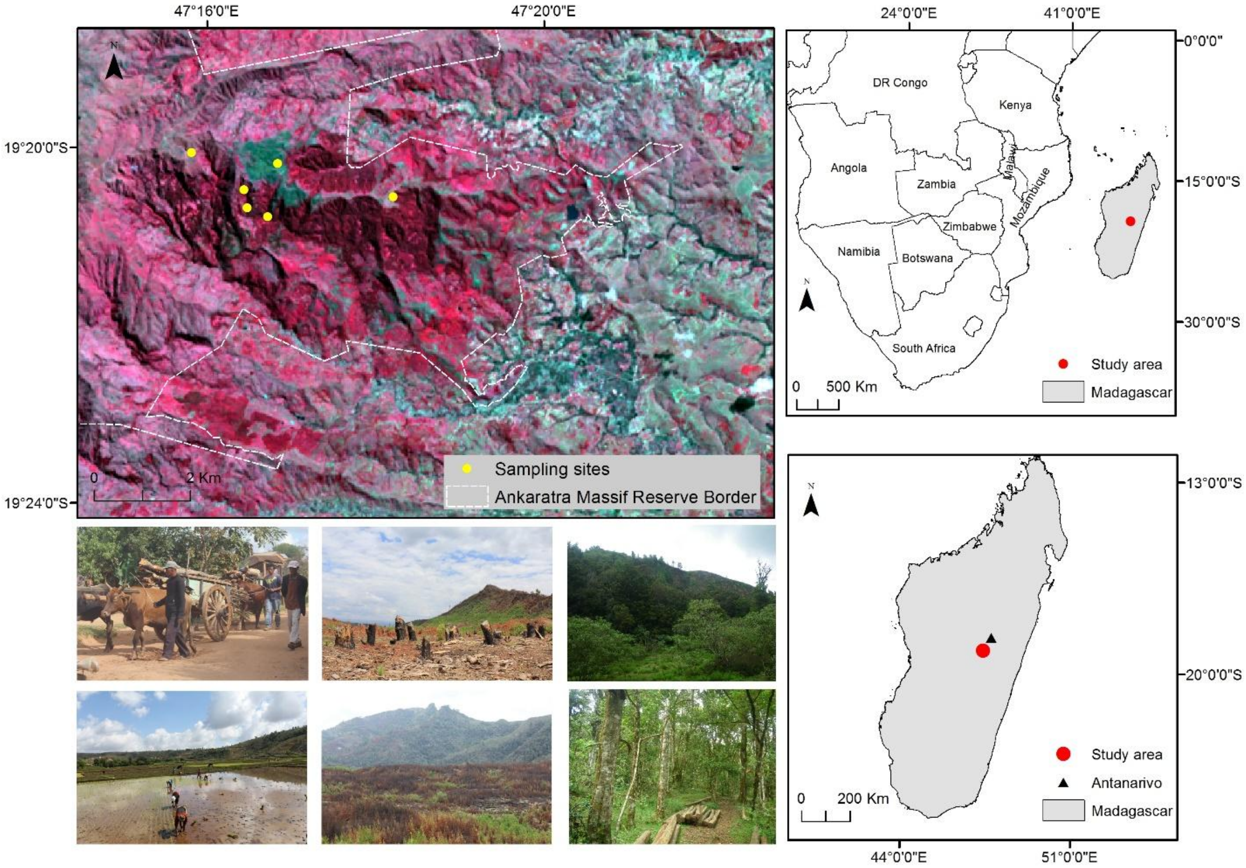

Figure 2.1 – Study area. Clockwise figure description: Study area showing sampling points for the 2011 – 2013 survey; Caption of southern sub-Saharan Africa; Madagascar, with study area and capital city depicted; Six photos from the Ankaratra Massif. Clockwise photo description: Farmers with cattle; Forest clearance; Forest; Rice paddies; Burnt heathland; Illegal logging in remaining native forest.

2.2 Dataset Collection and Assembling

2.2.1 Amphibian Community Data Collection

Amphibian surveys were conducted in six sampling sites distributed along high-altitude streams (>2000m a.s.l.; Table 2.1) and differing in habitat characteristics (native forest, savannah, exotic forest) and state of degradation. Surveys were carried out from 2011 to 2013, both in wet and in dry season, and were directed mainly at assessing occurrence and abundances of *Boophis williamsi* and *Mantidactylus pauliani*.

At each site, one transect of approximately 100 metres long, with varied widths according to the stream structure, was established. The selected streams are not directly connected to each other, to guarantee that gene flow among them is resultant from movements of adult dispersal. Each transect was surveyed twice during each survey trip, with both diurnal and nocturnal visits. During these surveys, the team, supervised by the association Langaha (a local herpetological organization based in Antananarivo), conducted an exhaustive search to capture all adult and larval frogs present along the monitored transects. Access Appendix I for detailed characteristics on the topography and temporal sampling of each site.

In order to verify the efficiency and level of completeness of the surveys, species accumulation curve were carried out (Fig. 3.2). Accumulation curves estimate the rate at which species not previously sampled during the survey can be found within it. The R software package BiodiversityR was used for this purpose.

Table 2.1 – Sampling Sites

Sampling Sites	Coordinates	Sampled Years	Altitude (above sea level)
Ambohimirandrana	S 19°20'45'' E 47°16'46''	2011 - 2013	2250 – 2314m
Tavolotara	S 19°20'45'' E 47°16'45''	2011 - 2013	2000 – 2020m
Tsimiaramianadahy	S 19°20'02'' E 47°15'83''	2011 - 2013	2364 – 2410m
Analafohy	S 19°20'39.0'' E 47°16'30.3''	2012 - 2013	2082m
Ambitsika	S 19°20'09.0'' E 47°16'51.5''	2012 - 2013	2208m
Anosiarivo	S 19°20'30.4'' E 47°18'14.0''	2013	2062m

2.2.2 Satellite Imagery and Pre-Processing

A set of seven multi-spectral satellite images retrieved from the Landsat satellite platform were treated to map and analyze land cover, estimate deforestation rates and to derive landscape functioning values (Table 2.2). The dataset covered the study area between 1985 and 2016. Landsat 5 Thematic Mapper (TM) and Landsat 8 Operational Land Imager and Thermal Infrared Sensor (OLI-TIRS) images, with a spatial resolution of 30 metres, were retrieved from USGS Earth Explorer (https://earthexplorer.usgs.gov) in Geotiff format. Landsat 5 and 8 images are composed by seven and eleven spectral bands, respectively. For the land cover mapping analysis, the following bands were considered: blue, green, red, near infrared (NIR), shortwave infrared (SWIR) 1 and shortwave infrared (SWIR) 2. Data in these images is stored in digital numbers (DN). In order to allow further analysis, it was converted into Top of Atmosphere (ToA) Reflectance using the radiometric rescaling coefficients provided in the product's metadata.

In order to allow multi-temporal comparison between images from both types of satellite sensors, radiometric normalization was applied to Landsat 5 TM images, using pseudo-invariant features (PIFs) – features whose reflectance properties hardly change throughout time (in this case stable forest patches) – and applying the following formula (Schott et al., 1988):

$$DN'_{2i} = \frac{\sigma_{1i}}{\sigma_{2i}} DN_{2i} + \overline{DN_{1\iota}} - \frac{\sigma_{1i}}{\sigma_{2i}} \overline{DN_{2\iota}} \tag{1}$$

where DN'_{2i} is the normalised image,

σ is the standard deviation of each image,

DN_{2i} is the image that will be normalised to the master image,

$\overline{DN_{1\iota}}$ and $\overline{DN_{2\iota}}$ are the means of the pseudo-invariant pixels of master image and of the image to be normalised, respectively.

Simplifying we can have:

$$y = ax + c \tag{2}$$

where a - $\frac{\sigma_{1i}}{\sigma_{2i}}$,

x - image to be normalised,

c - $\overline{DN_{1\iota}} - \frac{\sigma_{1i}}{\sigma_{2i}} \overline{DN_{2\iota}}$.

Table 2.2 - Landsat images used for analyses.

Acquisition Date	Source	Sensor Type	File Name	Spatial Resolution
08/03/1985	USGS EarthExplorer	Landsat 5 TM	LT5159073 1985067	30m
20/03/1995	USGS EarthExplorer	Landsat 5 TM	LT5159073 1995079	30m
18/05/2005	USGS EarthExplorer	Landsat 5 TM	LT5159073 2005138	30m
10/05/2008	USGS EarthExplorer	Landsat 5 TM	LT5159073 2008131	30m
22/07/2011	USGS EarthExplorer	Landsat 5 TM	LT5159073 2011203	30m
22/04/2013	USGS EarthExplorer	Landsat 8 OLI/TIRS	LC8159073 2013112	30m
30/04/2016	USGS EarthExplorer	Landsat 8 OLI/TIRS	LC8159073 2016121	30m

2.3. Multi-Temporal Land Cover Mapping and Assessment of Forest Cover Change and Deforestation Rates

A supervised image classification approach with Random Forests (RF; Breiman, 2001), a machine learning algorithm, was used to map land cover in the study area, from 1985 to 2016. This algorithm builds a set (forest) of independent classification trees and combines the prediction from all the trees in the final model (Cutler et al., 2007). For each date, the input dataset for the classification was composed by the chosen Landsat spectral bands and by the three composite satellite-derived indices (NDVI, SAVI, Albedo; see section 2.4). Vegetation indices have already been successfully used to discriminate different land cover types in previous studies across Africa (Achard and Blasco, 1990).

Five land cover classes (forest, shrubland, grassland, crop and barren land) were defined based on previous descriptions of the study area and from the observation of high resolution Google Earth images. There exist two different types of forest in Ankaratra which were not differentiated when analysing and classifying the satellite images: the native montane forest and the non-native plantations of *Pinus* sp. (Rahantaliosa et al., 2011). It was decided to map all forest as a single land cover class due to the in-field knowledge needed to accurately attempt a discrimination between the two forest categories and due to the implications that misclassification can bring to Madagascar’s forest policies. Shrublands are characterized by the presence of shrubs or short trees and composed of two subclasses, which, here too, are not differentiated when analysing and classifying the satellite images: 1) savannah, where shrubland is fairly open so grasses and other short plants grow between the shrubs, 2) forest regrowth, where there have not been ongoing destructive anthropogenic activities and shrubland is composed of continuous areas of growing trees. Grasslands are composed of open areas of plants from the Gramineae family. Crops are constituted by cultivated areas, mainly potato plantations and rice fields (paddies), and pastures for the grazing of livestock. Barren land is any eroded land caused by deforestation, recent fires or infertile soil patches.

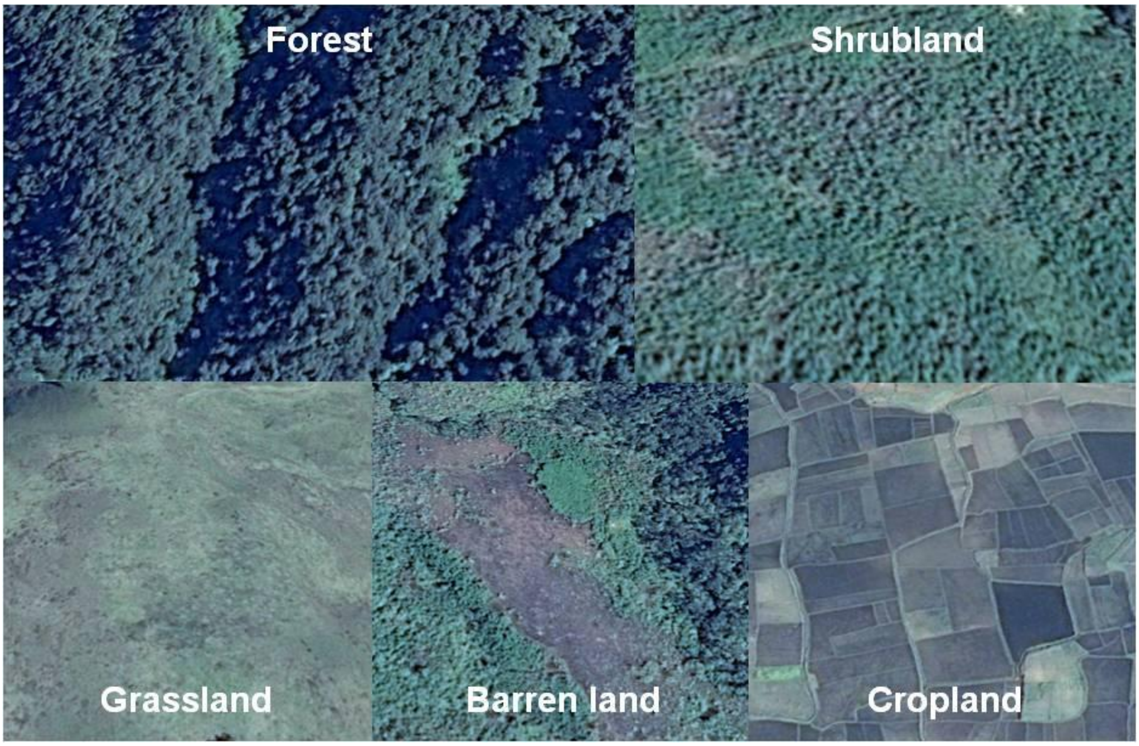

Figure 2.2 – The five defined Land Cover Classes as seen on Google Earth satellite images.

Training polygons, representative of each land cover class, were created (Fig. 2.3) in ENVI software v.4.7 (EXELIS, 2009). The training areas consist of groups of pixels with similar reflectance, known to be part of a specific land cover class. The Random Forest algorithm is used to create a classification model which, having learnt the patterns in the training data, is then used to estimate the classification map of the whole image.

Due to misclassification of crops at high altitude and on steep slopes, a post-classification ruleset was applied to eliminate crops at higher altitudes than 2000m a.s.l. and at slopes steeper than 8 degrees, using masking technique in ENVI software. This rule was defined based on previous knowledge of the study area.

To validate and assess the accuracy of each classification, a set of fifty randomly chosen sites were created (Fig. 2.4). The location of the validation polygons was constant throughout all classified images, unless they were in mixed vegetation areas and thus had to be moved to a uniformed vegetation area. Very high resolution images from Google Earth were used as a further supporting tool to verify the accuracy of the classified images. Classification accuracy was calculated through “out-of-bag” error (OOB). As RF algorithm consists of a set of trees where each tree only trains on a subset of the full training dataset, OOB is the part of the training dataset that was left out of the training of each tree and is used to estimate the error rate.

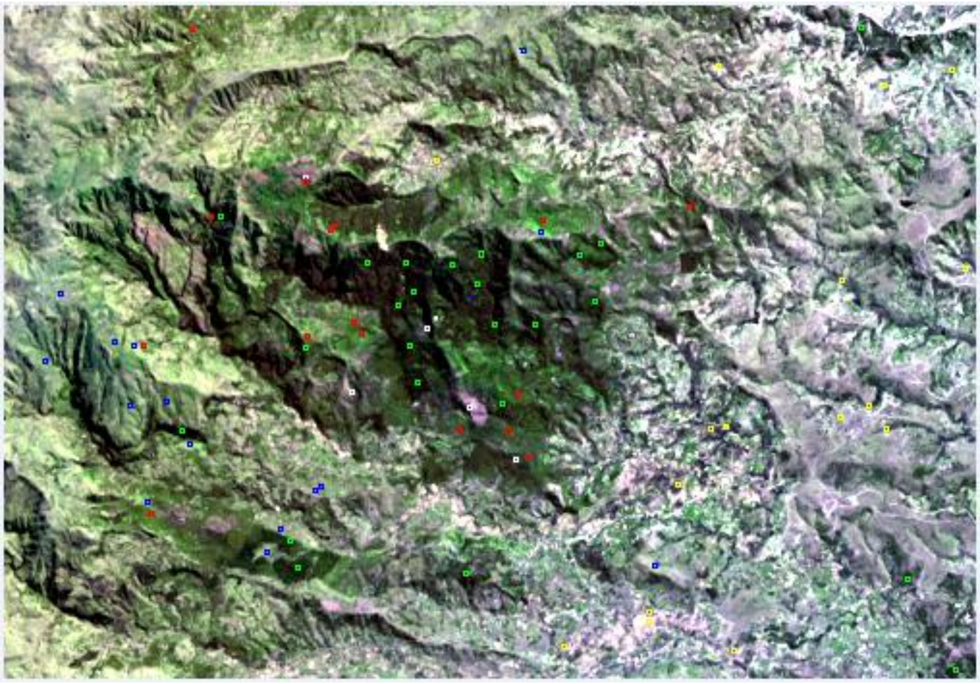

Figure 2.3 – Training polygons used for each land cover class. Shrubland – red polygons; Grassland – blue polygons; Forest – green polygons; Barren land – white polygons; Crop – yellow polygons. Image date: 30/04/2016.

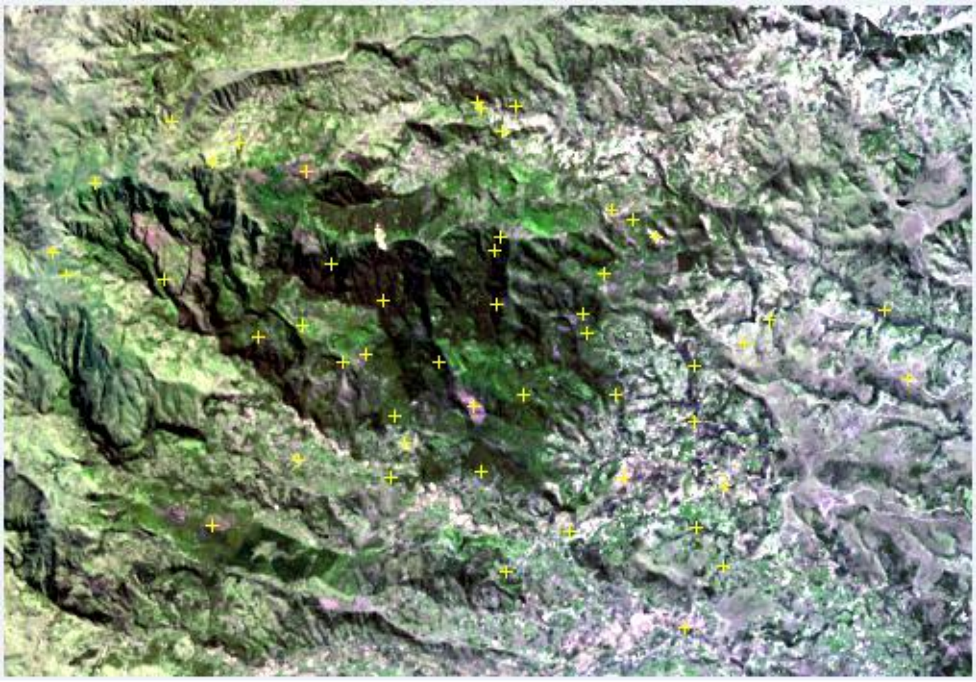

Figure 2.4 – Validation polygons (yellow crosses). Image date: 30/04/2016.

All classification operations were performed in ENVI software v.4.7 and in Python v.2.7 programming language (available at https://www.python.org) using the following packages: future (Schofield, 2016), Matplotlib (Hunter, 2007), NumPy (van der Walt et al., 2011), Pandas (McKinney, 2010) and skicit-learn (Pedregosa et al., 2011).

Annual deforestation rates were assessed using the forest cover class at each date and the equation (3) described by Puyravaud (2003):

$$r = \frac{1}{t_1 - t_2} ln \frac{A_2}{A_1} \tag{3}$$

where r is the annual rate of change of forest cover,
t_1 is the earliest year analysed and t_2 is the most recent,
A_1 and A_2 are the forest cover at time t_1 and t_2 respectively.
The lower the value of r the higher the annual deforestation rate.

2.4. Characterization of Landscape condition through Ecosystem Functioning Variables (EFVs)

The use of remote sensing derived functional metrics is increasingly used to monitor environmental change and its impacts on biodiversity. Three satellite-derived indices – the Normalized Difference Vegetation Index (NDVI), the Soil Adjusted Vegetation Index (SAVI) and Albedo – were chosen to characterize three aspects of the landscape functioning of Ankaratra’s ecosystem throughout the last thirty years. These three metrics were estimated for 1985, 1995, 2005, 2011, 2012, 2013 and 2016.
SAVI was only used for image classification purposes while NDVI and Albedo were also used in the multi-model inference approach performed later on in the analyses.

NDVI is given by the differential reflection captured by the red (R) and near-infrared (NIR) bands, allowing the monitoring of the density and intensity of green vegetation growth. It is calculated through the following equation (4):

$$NDVI = \frac{NIR - R}{NIR + R} \tag{4}$$

NDVI values range from -1 to 1, where negative values are given by water, clouds and snow, values close to zero represent rock or bare soil and within the positive values, the higher the value, the denser the green vegetation cover.

SAVI adjusts for soil brightness in areas with low vegetation cover, where soil brightness can interfere in the radiation received by the satellite sensor. It is estimated in the same way as the NDVI, but it adds the soil brightness factor (L). The value of L varies with the amount of green vegetation: L=0 in areas of dense green vegetation and L=1 in areas with no green vegetation. When L=0, SAVI=NDVI. Here, as SAVI was estimated for the whole study area, we used the default value of L=0.50, as we have a mixture of forest, shrubland and grassland. The following equation (5), described by Huete (1988), was used:

$$SAVI = \frac{NIR - R}{NIR + R + L} \times 1 + L \tag{5}$$

SAVI values also range from -1 to 1. Similarly to NDVI, the lower the value, the lower the vegetation cover and the higher the amount of exposed, bare soil.

Land surface Albedo measures how much radiation is reflected off the Earth’s surface, which influences land surface temperatures. It was estimated using the following equation (6):

$$Albedo = \frac{0.356b_1 + 0.130b_3 + 0.373b_4 + 0.085b_5 + 0.072b_7 - 0.0018}{0.356 + 0.130 + 0.373 + 0.085 + 0.072} \tag{6}$$

where b_1 - blue band
b_3 - red band
b_4 - NIR band
b_5 - SWIR1 band
b_7 - SWIR2 band

This equation is a normalized formula to calculate Albedo from Landsat images with DNs, developed by Smith (2010) based on the work of Liang (2000).

Albedo values range from 0 to 1. High Albedo values indicate high reflectance substances like snow and low values refer to low reflectance surfaces (e.g. concrete), which absorb more solar radiation and thus become hotter surfaces than high reflectance substances.

2.5 Modelling the Response of the Amphibian Community to Landscape Change

2.5.1 Multi-Model Inference Framework: Hypothesis and Competing Models

A multi-model inference (MMI) approach with model ranking based on the Akaike Information Criterion (AIC; Akaike, 1974) was used to understand the role of landscape and its dynamism on the high-altitude amphibian community of the Ankaratra Massif. This method requires the *a priori* definition of a set of competing hypotheses to be tested and ranked by order of importance – defining which hypothesis better explains the observed response variables and measuring how close to the “truth” each one stands. Based on literature, three main hypotheses described by a set of predictors were defined (see Table 2.3): deforestation (H1); primary productivity of the system (H2) and thermal variation (H3). H1 expresses the influence that forest loss may have on the surveyed amphibian species of Ankaratra; H2 and H3 hypothesize the influence that ecosystem energy processes, namely primary productivity and temperature, may exert on them. Each hypothesis was a competing model and it was expected that deforestation played a major role in explaining the current patterns of the high-elevation amphibian community of the Ankaratra Massif.

As all species present in this study are non-migrants (IUCN Red List) and amphibian movement is restricted due to physiological constraints (such as specific thermal environment and water availability; Feder and Burggren, 1992), two spatial units were considered: a 15 metre and a 30 metre radius from sampling point [from here on these will be referred to as 30m buffer (diameter) and 60m buffer (diameter)]. Having a circular buffer surrounding each sampling point ensures that the predictor variables analysed are specific to each sampling point and not a consequence of change throughout the entire study area. Having two different spatial extents for these buffers captures the variation of ecological processes across a broader scale. Likewise, two temporal units were studied: long-term changes within buffer (T1) and recent disturbances within buffer (T2). T1 incorporates changes in predictor variables from 1985 to 2016 and T2 involves changes in the ecosystem during sampling years (2011 to 2013). Using these buffers, the value for each predictor was extracted.

Table 2.3 – Competing hypotheses under analysis and associated rationales.

Hypothesis	Predictor	Description	General Rationale
Deforestation (H1)	Deforestation Rate	Annual rate at which deforestation occurs	Deforestation negatively affects forest specific species (Schneider-Maunoury et al., 2016)
Primary productivity of the system (H2)	NDVI	Normalized Difference Vegetation Index is linked to vegetation cover, biomass and net primary productivity	Amphibian richness and abundance tend to decrease in a gradient with decreasing vegetation cover (Trimble and van Aarde, 2014)
Thermal variation (H3)	Albedo	Albedo is a proxy for land surface temperature, measuring the amount of radiation reflected from the Earth's surface back into the atmosphere	Variations in temperature have a great impact on the survival of amphibians, as they hugely rely on atmospheric temperature to maintain their ideal body temperature (Carey and Alexander, 2003)

2.5.2 Response Variables

In order to measure the response of the amphibian community to the set of hypothesis established (H1 – 3), four response variables were defined based on the dataset introduced in section 2.2.1.: species richness (SR), amphibian abundance (N), microendemism rate (ME) and species turnover capacity (STC). These metrics were estimated considering the average values obtained in each sampling site (except for Anosiarivo, which was only sampled in 2013 and thus was not included in this set of analyses) from the 2-year survey. The five sampling sites that were surveyed from 2011 to 2013 were sampled twice a year, allowing for a good time representation of the high-altitude amphibian community present on the Ankaratra Massif. More specifically, SR is the mean number of species found during the two-year field survey. N is the mean number of adult and juveniles specimens found. ME was calculated as the number of *B. williamsi* and *M. pauliani* specimens found divided by the total number of adults, juveniles and tadpoles found. STC was calculated by dividing all tadpoles found by the total number of adults, juveniles and tadpoles found.

Although modelled with the same predictor variables, each response variable was modelled independently.

2.5.3 Predictor Variables

The three predictors associated with the three competing hypotheses (H1 – 3) were used to model species richness (SR), amphibian abundance (N), microendemism rate (ME) and species turnover capacity (STC). As already mentioned, each predictor was individually estimated at two different temporal (T1 and T2) and spatial (30 and 60m) scales for each sampling site considered in this set of analyses. These predictors are:

deforestation rate, which refers to the per year percentage at which forest is lost (estimated with equation 3); EFV's NDVI, proxy for primary productivity of the ecosystem; and Albedo, proxy for land surface temperature. EFV's SAVI was not included in these analyses so as not to exceed the limit number of variables allowed for a robust modelling technique since the number of competing hypotheses should ideally be smaller than the sample size (Burnham et al., 2011). Also, it was correlated with NDVI.

2.5.4 Statistical Analyses

At first, all predictors were checked for multicollinearity through examining Spearman correlation. When predictors had a correlation higher than 0.70, one of them was excluded from the analysis. After this model calibration, variance inflation factors (VIFs) were additionally calculated and hypotheses were only tested if VIF < 3, which indicates that there is no collinearity between them (Zuur et al., 2010). To relate the response variables to the predictors, generalized linear models (GLMs; McCullagh and Nelder, 1989) were used. After testing which distribution of errors best fitted each response variable, testing the distribution of the raw data and using the level of significance of residual deviance, species richness was modelled with a Poisson distribution of errors; amphibian abundance and microendemism rate were fitted with a Negative Binomial; and species turnover capacity was modelled with a Gaussian distribution of errors. To rank and identify the most parsimonious hypothesis explaining each response variable, the corrected AIC (AICc) values were used (Akaike, 1974). AIC is a measure of information loss where the lower the AIC value, the better the model explains the response variable. AIC scores are generally displayed as ΔAIC, which is the difference between the best model presented (smallest AIC value) and each one of the consecutive best following models. Thus, the best model has a ΔAIC of zero. AICc was used to correct AIC to the number of observations made. Where a Poisson distribution of errors was used, a dispersion test was also performed to further test the fitting capacity of the model (Cameron and Trivedi, 1990).

To quantify how much each model contributes to the distribution of the response variables, Deviance Explained was also calculated:

$$Deviance\ Explained = 1 - \frac{residual\ deviance}{null\ deviance} \tag{7}$$

where the null deviance is associated to the null hypothesis.

These analyses were carried out in R software v.3.4.0 (R Development Core Team, 2017).

3. Results

3.1 Amphibian Community Patterns of Ankaratra Massif Reserve

Due to the presence of three species endemic to Ankaratra (2 amphibians and 1 gecko, *Lygodactylus mirabilis*), its biodiversity has been relatively well studied (Vences et al. 2002; Andreone et al., 2007, 2014). To assess the abundance of the two critically endangered species of amphibians inhabiting Ankaratra (*B. Williamsi* and *M. pauliani*), a two-year survey was performed in this area (Rahantaliosa et al., 2011). Despite the more specific nature of this survey (targeting high-altitude aquatic amphibians), it covers 67% of the amphibian species known for this area (Vences et al., 2002).

Of the 15 species of amphibians known to occur on the Ankaratra Massif, 10 species were repeatedly surveyed and presence and abundances data were included in this study (Table 3.1). The five species not encountered are species that can only be found at lower altitudes, with the exception of *Plethodontohyla tuberata* that can be found up to 2400m a.s.l. but, being a strictly terrestrial species, was not encountered during this survey, which was targeted to aquatic species.

All species found during the survey are non-migrants and classified as least concern (LC) by the IUCN Red Listing Criteria (IUCN, 2016a, 2016b, 2016c, 2016d, 2016f, 2016g, 2016i, 2017a, 2017b), except for *B. williamsi* and *M. pauliani,* which are CR, as already mentioned. Generally, these species are classified as LC due to their local abundance, adaptation to diversified habitats and due to some degree of tolerance to the degrading of these habitats. Despite this, some species are caught for human consumption (the large *Boophis goudotii* is caught and sold in restaurants under the name of “cuisses des nymphes”) and their numbers are suspected do be declining.

The genus *Boophis* is mainly composed of tree frog species (Glaw and Vences, 2007). In Ankaratra the species of this genus vary greatly in the extent of their toleration to habitat degradation. Some inhabit degraded sites and rice fields as well as rainforests, breed in slow moving or permanent waters (e.g. *B. goudotii*), others tolerate degraded areas and open habitat, as long as trees and fast flowing unpolluted streams are available (*B. ankaratra* and *B. mycrotympanum*). On the contrary, *B. williamsi* is a highly localized mountain-top species and relies on montane forest for surviving and mountain brooks for breeding.

Table 3.1 - Amphibian species of the Ankaratra Massif (data from Andreone et al., 2007, 2014; Vences et al., 2002; Rahantaliosa et al., 2011).

	Blommersia domerguei (-, 1) (males 16 mm)	*Blommersia kely* (+, 1) (males 14-16mm)	*Boophis ankaratra* (+, 1) (males 24-28mm)	*Boophis goudotii* (+, 1, 2) (males 50-100mm)	*Boophis microtympanum* (+, 1) (males 27-42mm)	*Boophis williamsi* (+) (males 40-43mm)	*Heterixalus betsileo* (-, 1) (males 18-28mm)	*Heterixalus rutenbergi* (-,1) (males 25mm)
1992 - 2001								
2011 - 2013								

	Mantidactylus alutus (+, 1, 2) (males 24-26mm)	*Mantidactylus brevipalmatus* (+, 1) (males 28-45mm)	*Mantidactylus* sp. aff. *curtus* Ca19 (+, 1, 2) (males 37-43mm)	*Mantidactylus pauliani* (+) (males 25-34mm)	*Plethodontohyla tuberata* (+, 1, 2) (males 35-45mm)	*Ptychadena mascareniensis* (-, 1, 2) (males 30-50mm)	*Scaphiophryne madagascariensis* (-, 1, 2) (males 41-56 mm)
1992 - 2001							
2011 - 2013							

+ species found at elevations of > 2000m; - species found at elevations of < 2000m; ^{1} tolerates open/degraded habitat; ^{2} tolerates agricultural land.

Likewise, among the species belonging to the genus *Mantidactylus,* a great degree of toleration to a variety of habitats is observed. For instance, *M. alutus* and *M.* sp. aff. *curtus* Ca19 inhabit both rice fields (or other agricultural areas) and montane forest, *M. brevipalmatus* can be found at high-altitude savannahs and heathlands but apparently they cannot tolerate agricultural lands. *M. pauliani* and *B. williamsi* inhabit fast flowing montane streams, while *Blommersia kely* and *Ptychadena mascareniensis* can be found in many different habitats.

In order to analyze amphibian community patterns, four biodiversity metrics were chosen (Fig. 3.1): species richness (SR), amphibian abundance (N), microendemism rate (ME) and species turnover capacity (STC). As surveys were carried out twice a year, it was possible to analyse community trends in the surveyed populations in the wet and in the dry season. This survey was performed opportunistically and thus, it was decided to profit from the available data even though it was not collected under a strict standardized sampling protocol (in relation to month of sampling across the year and not in relation to sampling methodology). Although the importance of analyzing robust raw data is acknowledged, it is important to keep in mind that Ankaratra is a very isolated area, where sampling is conditioned by bad weather conditions and road unavailability. In the future, it will be important to collect survey data across standardized sampling months. Despite this, inter-seasonal analyses were performed to achieve a primary understanding of possible seasonal behaviour. More precise analyses will have to be made in the future, once established sampling months are defined.

Differences can be seen between wet and dry seasons and between sampling sites. Species richness varies a lot between sampling sites (from 3 to 8), the mean SR being higher in the dry season. Similarly, it can be observed that there is a much larger range of amphibian abundance during the dry season, its mean surpassing the number of specimens found during the wet season. This might be due to the low availability of water bodies in these drier months, causing species to aggregate in smaller areas, therefore increasing detection rates. The highest N (outlier) is always found in Tavolotara (a mean number of 334 individuals encountered in the wet season and 341 in the dry season). The mean proportion of microendemic individuals found was much higher during the dry season than the wet season, whereas the maximum proportion (a mean proportion of 85% of microendemic species found in Tavolotara) was the same across seasons. No microendemic species were found in Anosiarivo in either of the seasons. Species Turnover Capacity (number of tadpoles / number of specimens found throughout the sampling years) has a higher range of occurrences in the wet season, with the exception of an outlier situated in Anosiarivo, where almost 100% of the individuals found there during the dry season were tadpoles.

Overall, it seems clear that the dry season seems much more favourable for amphibian detection.

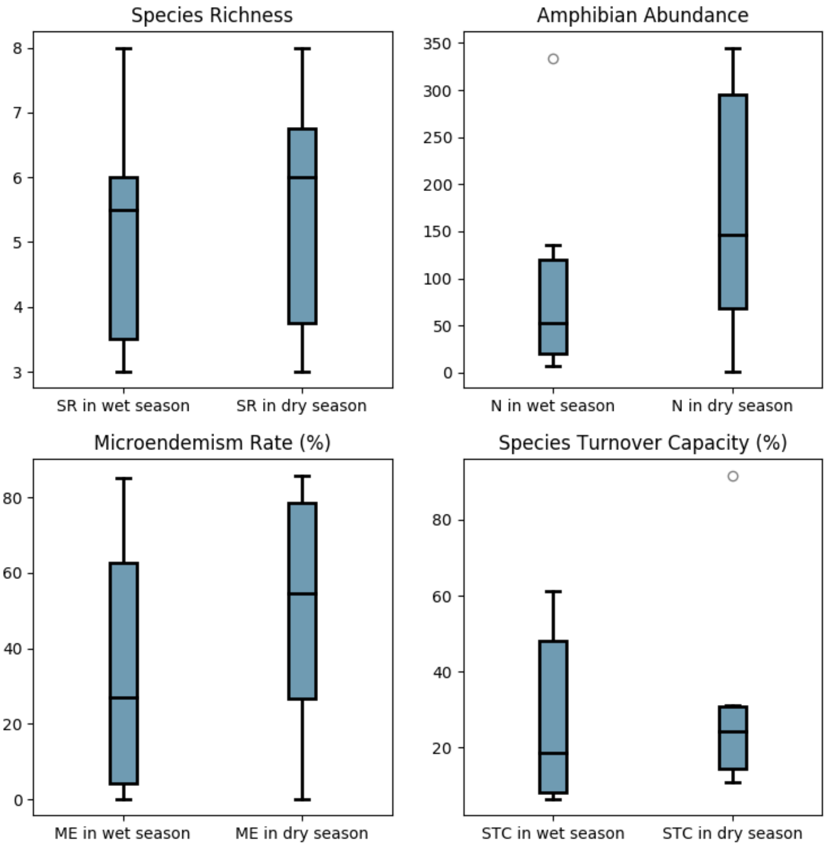

Figure 3.1 - Amphibian community patterns in Ankaratra Massif Reserve from 2011 – 2013. SR – Species Richness (mean value of 2 sampled years); N – Amphibian Abundance (mean value of 2 sampled years); ME – Microendemism Rate (number of adults and juveniles of *Boophis williamsi* and *Mantidactylus pauliani* found throughout the 2 sampling years/ number of specimens found throughout the sampling years*100); STC – Species Turnover Capacity (number of tadpoles found throughout the 2 sampling years/ number of specimens found throughout the sampling years*100). SR and N are presented in absolute values, ME and STC are presented in percentage.

The species accumulation curves show that the more the sampling sites, the higher the probability of sampling all amphibian species present in the study area (Fig. 3.2 a) but, as the curve did not reach a plateau, it would be relevant to increase the number of sampling sites in further surveys. Regarding the number of microendemic species found, the curve reached the plateau state at site 3 (Fig. 3.2 b), meaning that surveying as few as three sampling sites between the 6 analysed in this study is enough if the sole objective of the survey is to encounter *B. williamsi* and *M. pauliani.*

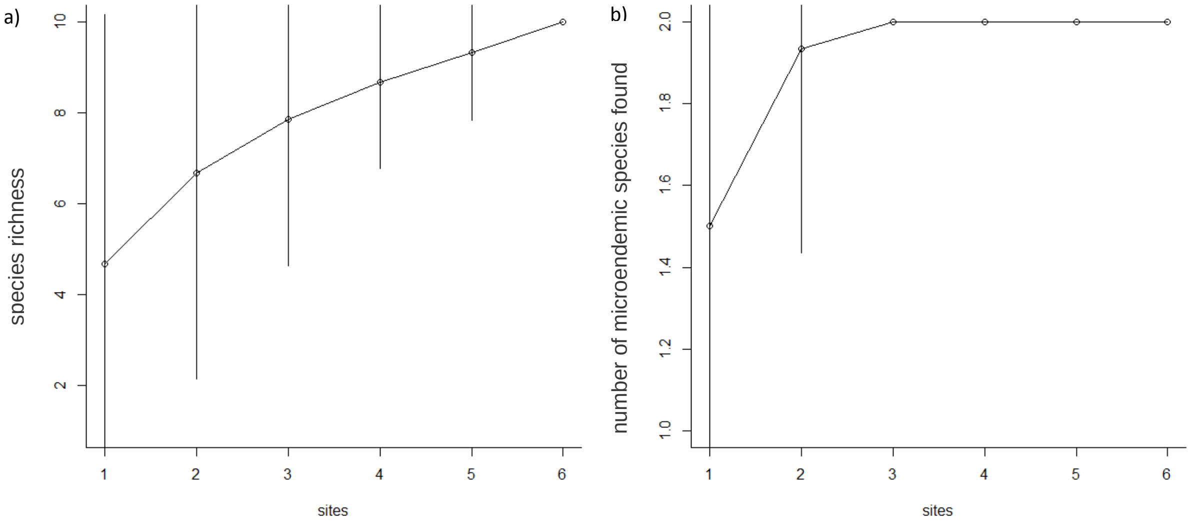

Figure 3.2 – Species accumulation curves. a) species richness accumulation curve; b) microendemic species accumulation curve.

3.2 Landscape Changes in the Ankaratra Massif Reserve

3.2.1 Land Cover and Deforestation

The multi-temporal land cover analysis accomplished using a supervised classification of Landsat satellite images with Random Forest algorithm between 1985 and 2016 reveals that the Ankaratra Massif is a very dynamic landscape with a notable role in forest conversion into other types of land cover (Fig. 3.3; Table 3.2; Table 3.4).The most recent analysed landscape was mainly composed by grasslands (48.9%), followed by woody vegetation [shrubland (21.7%) and forest (9.9%)] and crop areas (15.5%). This land cover pattern prevailed during the entire period of analysis, even throughout the periods of expansion and contraction that all land cover classes suffered (Table 3.2). Despite this extremely dynamic landscape, a steady trend of forest clearing can be observed since 2005 (annual deforestation rate of approximately 6% between 2005 and 2016, Fig. 3.4). In 2016 only approximately 10% of the whole study area was covered by forest (Table 3.2), although a large amount of regrowth is present (Fig. 3.3). Additionally, a relatively large increase in barren land is observable in the 2016 land cover analysis (3.87% in 2016 compared to 1.83% in 1985, Table 3.4).

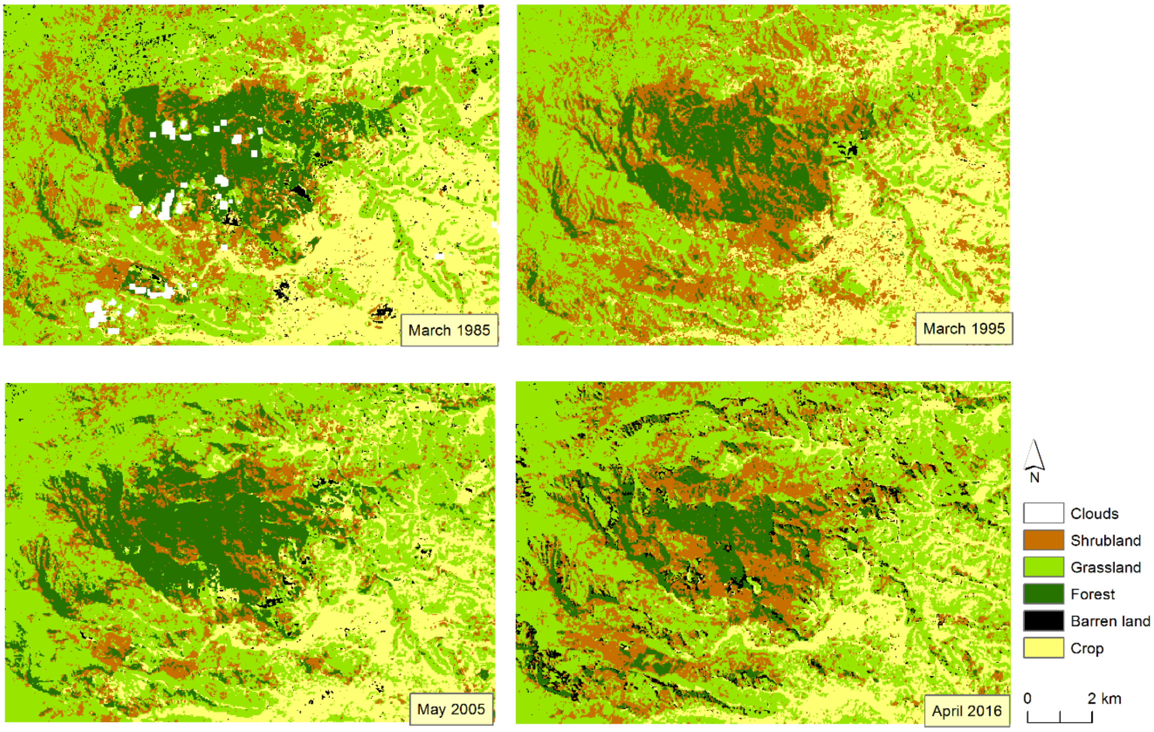

Figure 3.3 – Land cover maps obtained from supervised classification of Landsat imagery for 1985, 1995, 2005 and 2016.

Table 3.2 - Percentage of each Land Cover Class (LCC) throughout analysed years.

% LCC	1985	1995	2005	2016
Shrubland	14.97	27.19	13.27	21.79
Grassland	42.33	40.13	49.33	48.92
Forest	12.40	10.19	17.32	9.88
Barren Land	1.83	0.14	0.68	3.87
Crop	26.98	22.35	19.40	15.54
Cloud Cover	1.48	0.00	0.00	0.00

Accuracy assessments on land cover classification based on “out-of-bag” error (OOB) indicated that the maps for the Ankaratra Massif Reserve were reliable. The classification tree algorithm had occasional difficulty in distinguishing shrubland from forest due to the fact that regrowth could be in an advanced phase of progression. In these situations the two categories could only be distinguishable by the naked eye analysing Google Earth images with high spatial resolution, as pixels could be very similar in both classes at that stage. There was also some confusion in classification between grassland and crop, specifically in wetter years and with rice paddies. Despite this, classification accuracy is consistently very high (classification values above 90%; Table 3.3). When validating the classification data, accuracy assessments indicated that all classifications of the images registered an overall accuracy of more than 80%

with the exception of 1995 which had an overall accuracy of 75% (Table 3.3). The best class accuracy was achieved for the forest land cover class (ranging from 82% to 100% accuracy). On the other hand, crop and barren land were the classes with the highest variability in accuracy values (crop: 57% to 100%; barren land: 0% to 100%).

Table 3.3 – Accuracy of supervised classification performed on Landsat images and posterior validation. Validation accuracy of each land cover class.

Land Class		Year			
		Accuracy (%)			
		1985	1995	2005	2016
Total image	Classification	94.1	97.0	90.8	94.8
	Validation	92.0	75.0	91.0	80.0
Shrubland	Validation	83.0	55.0	100.0	100.0
Grassland	Validation	100.0	68.0	89.0	64.0
Forest	Validation	93.0	100.0	82.0	100.0
Barren Land	Validation	100.0	33.0	100.0	0.0
Crop	Validation	57.0	83.0	100.0	85.0

Comparatively to all analyzed time intervals, the period between 1985 and 1995 was relatively mild in terms of forest clearing activities, with an annual deforestation rate of 1.96% (Table 3.4). From 1995 to 2005 the trend is of 5.30% annual regrowth, which increased the forested area in 40%. Deforestation rate is drastically inverted from 2005 to 2016, the rate of forest clearing rising to 5.10% annually, determining the decrease of forested area to a mere 15.66km^2 in 2016. A severe increase of deforestation in 2010 is observed, year where a huge fire occurred across a large portion of the pine plantations of Ankaratra. Interestingly, in recent years it seems that deforestation has been stabilizing (Table 3.4).

Table 3.4 - Forested area and respective deforestation rate for both long-term and short-term analyses.

Year	Forested Area (km2)	Time scale	Deforestation Rate (%/year)	Year	Forested Area (km2)	Time scale	Deforestation Rate (%/year)
1985	19.64	1985 - 1995	-1.96	2011	21.01	2011 - 2013	-12.20
1995	16.15	1995 - 2005	5.30	2013	16.46	2013 - 2016	-1.66
2005	27.44	2005 - 2016	-5.10	2016	15.66		
2016	15.66						

When deforestation rate value is positive, this indicates forest regrowth. The lower the deforestation rate value, the higher the annual amount of cleared forest.

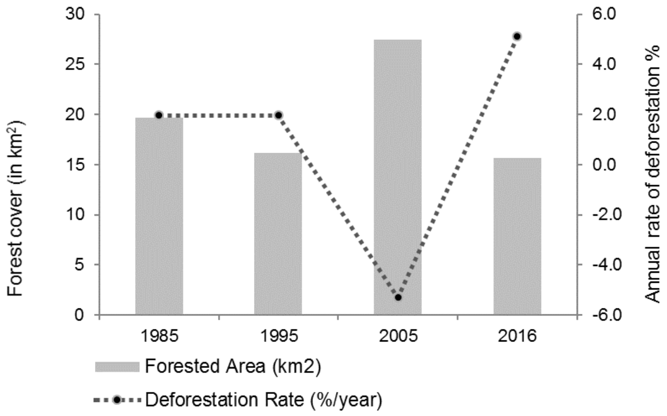

Figure 3.4 – Graphic analysis of long-term deforestation rates. To facilitate visual interpretation please note that, the higher the annual rate of deforestation value, the higher the values of forest clearance. True deforestation rate for 1985 is not available because the forest cover previous to this year was not assessed in this study.

Figure 3.5 shows which portion of the forest has remained stable since 1985. Merely 42% of the forest has remained stable throughout these thirty years of analyses. Only three of the six sampling points are within or in the close proximity of the forest areas that have been maintained stable.

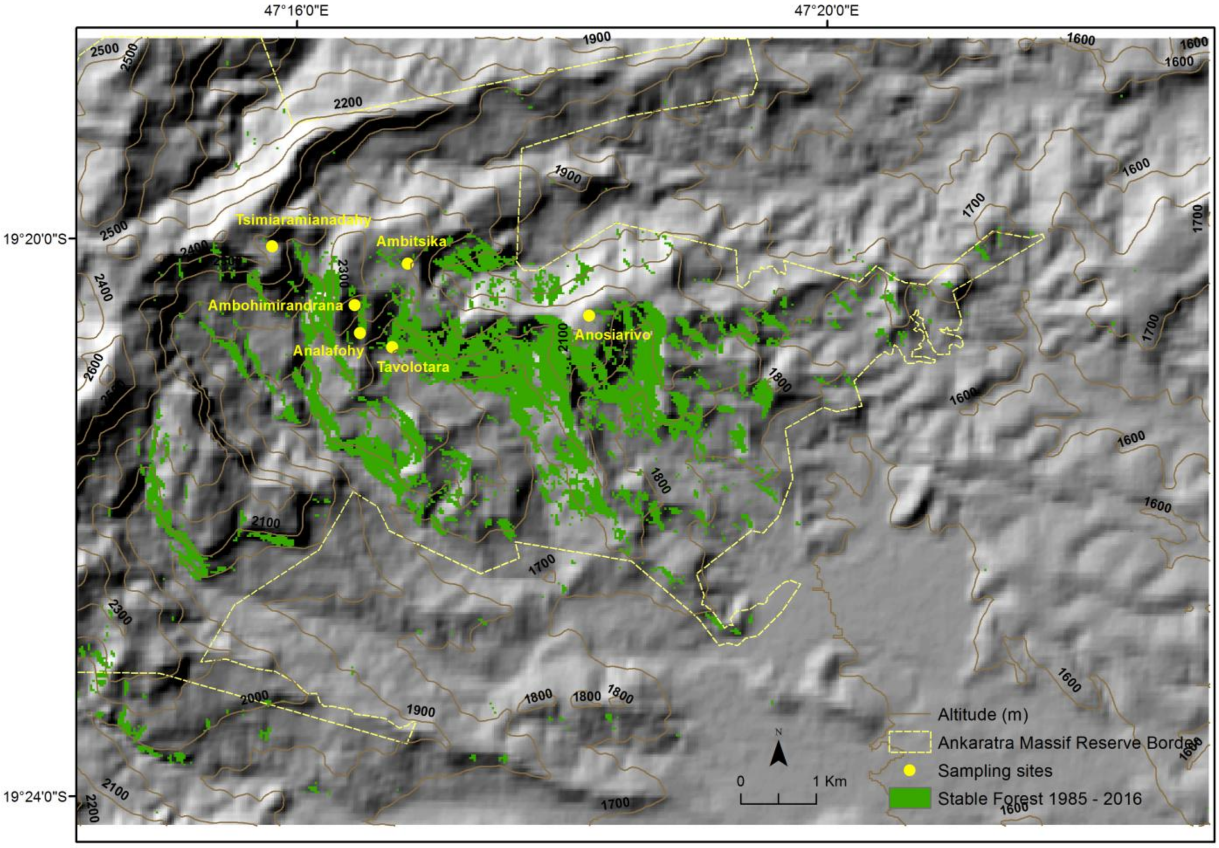

Figure 3.5 – Forest that has been maintained stable throughout the 30 years under analysis.

3.2.2 Changes in Landscape Functional Attributes

Three indices (NDVI, SAVI and Albedo) were used to characterize and measure alterations in the functional attributes of landscape between 1985 and 2016 (Fig. 3.6).

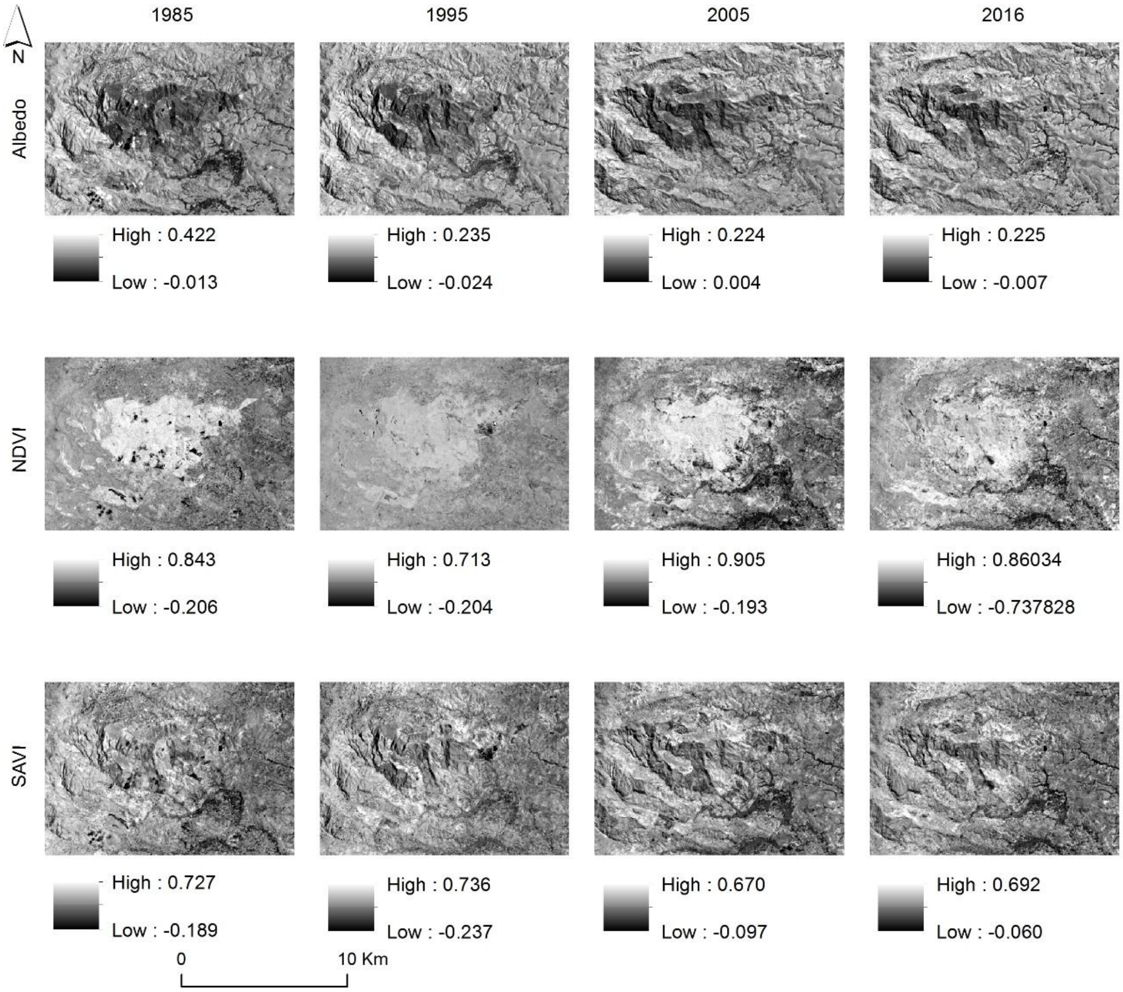

Figure 3.6 – Changes in Ecosystem Functional Attributes throughout the years under analysis.

A baseline value of NDVI, SAVI and Albedo for the study area was calculated throughout the analysed years so to have a reference value that can be referred to when considering the equilibrium of the ecosystem (Table 3.5). This way it is possible to have a proxy of how the system has changed in relation to this equilibrium and how it will tend to evolve in the future (Fig.3.7).

Table 3.5 – Summary statistics for the vegetation indices for the entire study area. The baseline value is the median values across the whole study area since 1985.

Year	NDVI		SAVI		Albedo	
	Median	Standard Deviation	Median	Standard Deviation	Median	Standard Deviation
1985	0.684	0.118	0.390	0.085	0.120	0.036
1995	0.673	0.496	0.376	0.093	0.117	0.035
2005	0.665	0.099	0.372	0.078	0.116	0.032
2016	0.677	0.067	0.379	0.087	0.114	0.030
Baseline	0.675	0.109	0.377	0.086	0.116	0.033

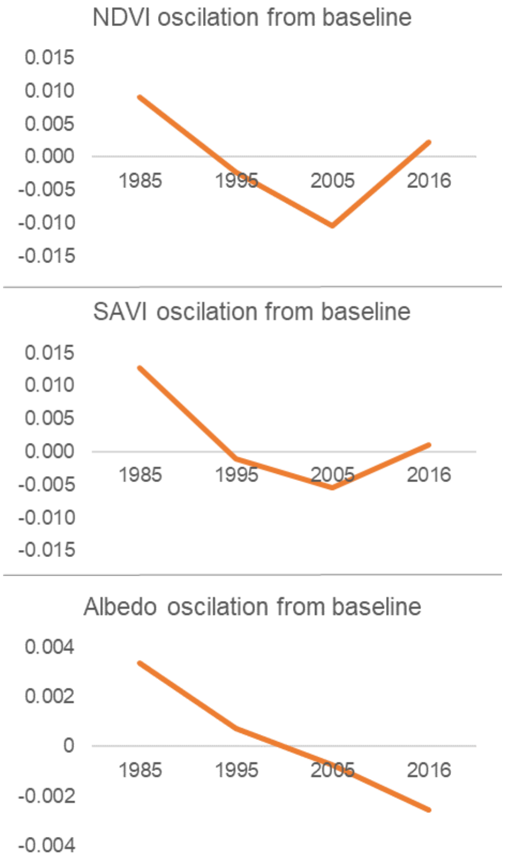

Figure 3.7 – Ecosystem Functional Attributes (EFAs) oscillation in relation to the baseline conditions of the whole study area.

Results show that the median values of all three indices have decreased since 1985 for the whole study area, although 2016 shows a slight increase relative to the baseline for NDVI and SAVI values (Table 3.5). More specifically, NDVI and SAVI values were the highest in 1985, following a decreasing trend until 2005. From 2005 to 2016, values started to recover, reaching the baseline (Fig. 3.7). NDVI and SAVI are extremely related to one another, as NDVI is a proxy to greenness abundance and SAVI corrects for areas with lower greenness, being a proxy to the presence of bare land. Therefore, greenness decreased from 1985 to 2005 with a consequent expansion of bare soil. Then, from 2005 to 2016 the increase of these values suggests the system to be recovering once more. Albedo, a proxy for thermal conditions or land surface temperature, is decreasing throughout the study area, meaning land surface temperature is increasing, as land surface is absorbing more radiation, however little it may be. As can be seen in Figure 3.6, Albedo's trend has been to decrease sharply.

3.3 Landscape Change and Amphibian Patterns in Sampling Sites

All sampling sites have suffered recent forest clearing with the exception of Tavolotara, which seems to be the most stable site in terms of ecosystem structure, having maintained its forest cover since 2005 (Table 3.6). Tsiamiaramianadahy, appears to be a stable combination between open savannah and heathland and the most stable sampling site from 1985 up until 2013 is Anosiarivo, which has had constant forest cover throughout time, although it has recently suffered from forest clearance. Likewise, Ambitsika had a stable forest cover until 2005, suffering intensive forest clearing after that but it now seems to be recovering at a rapid pace. Analafohy seems to be a stable balance between gallery forest and shrubland, although recent clearing can be observed. Ambohimirandrana has also suffered from recent forest degradation after a period of regrowth.

Table 3.6 – Land cover class at each sampling point in each analysed year. Analysis within 60m buffer.

Sampling Site	1985	1995	2005	2011	2013	2016	In-field description	Description based on long-term image analysis
Ambohimirandrana	Shrub (100%)	Shrub (99%) Grass (1%)	Forest (79%) Shrub (21%)	Forest (19%) Barren (19%) Shrub (62%)	Forest (75%) Shrub (25%)	Forest (16%) Shrub (52%) Grass (12%) Barren (20%)	Narrow brook in an open savannah area. Existing slope of 20 to 45 degrees.	Old regrowth with recent clearing
Tavolotara	Forest (83%) Shrub (17%)	Forest (46%) Shrub (54%)	Forest (100%)	Forest (100%)	Forest (100%)	Forest (99%) Shrub (1%)	Brook on average slope in a forest environment	Old forest clearing
Tsimiaramianadahy	Shrub (17%) Grass (83%)	Shrub (98%) Grass (2%)	Forest (2%) Grass (98%)	Forest (1%) Grass (53%) Shrub (46%)	Grass (100%)	Shrub (5%) Grass (92%) Barren (3%)	Moderate flowing brook in the open savannah area	Open savannah/heathland
Analafohy	Forest (60%) Shrub (40%)	Forest (54%) Shrub (46%)	Forest (100%)	Forest (100%)	Forest (99%) Shrub (1%)	Forest (40%) Shrub (36%) Grass (1%) Barren (23%)	Moderate flowing brook in a gallery forest	Long-term stable primary or secondary forest with very recent clearing
Ambitsika	Forest (100%)	Forest (100%)	Forest (100%)	Forest (20%) Barren (80%)	Barren (100%)	Forest (48%) Shrub (52%)	Moderate flowing brook near a crest with degraded exotic forest	Long-term stable primary or secondary forest with very recent clearing
Anosiarivo	Forest (100%)	Forest (93%) Shrub (7%)	Forest (100%)	Forest (100%)	Forest (100%)	Forest (49%) Shrub (51%)	Moderated flowing brook in a degraded natural forest	Long-term stable primary or secondary forest with very recent clearing

At an ecosystem functional level, sampling sites show important changes, varying according to time and space scale considered (Fig. 3.8). NDVI values (proxy for primary productivity) were always high (Table 3.5), indicating that greenness was maintained in all plots, across the different time scales. Despite this, mean NDVI values have decreased approximately 8% in recent years, from 0.81 to 0.75 (Fig. 3.8). The lowest NDVI value in the recent years was detected for Ambitsika (at 30m buffer = 0.61; at 60m buffer = 0.58), a site in degraded exotic forest, although the lowest NDVI long-term values have been detected for Tsiamiaramianadahy (at 30m buffer = 0.70; at 60m buffer = 0.69), which was to be expected being a site in a mostly open area. SAVI values have notably varied from site to site since 1985, greatly homogenizing around 0.35 in recent years, putting into evidence two outliers: a very low SAVI value at Ambitsika (SAVI=0.18), suggesting this to be the site with the higher extent of bare soil, and a comparatively high value at Tsiamiaramianadahy (SAVI=0.41) where, however, we can see a slight decrease of the mean SAVI values in recent years, suggesting an increase in barren land across sites. Contrary to the general diminishing trend of the study area, sampling locations do not seem to have changed much when looking at Albedo values, with the exception of the coolest site – Tsiamiaramianadahy – which has had a decrease of Albedo=0.14 to Albedo=0.13, meaning it has become slightly hotter. As a final remark, we note that Tsiamiaramianadahy is characterized by slightly different ecosystem functional variables than the other sampling sites. This is probably due to the nature of this sampling site, being a brook flowing through open savannah and heathland.

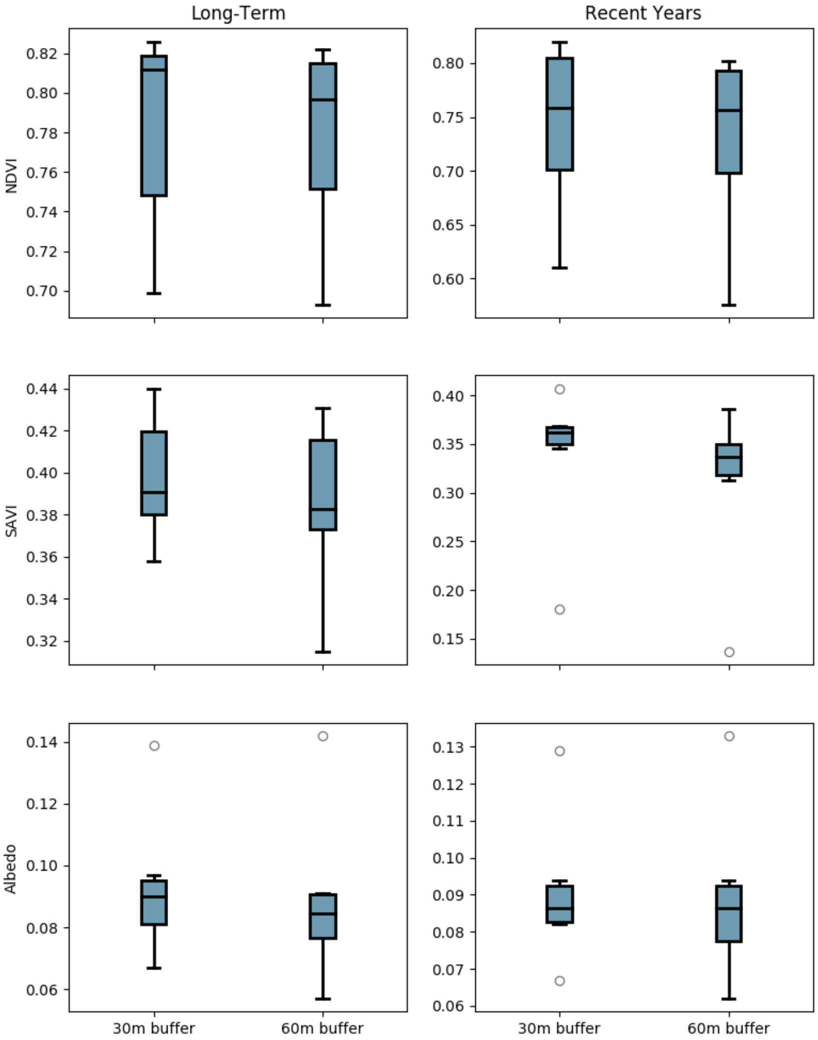

Figure 3.8 – Boxplots representing the predictors associated to each spatial and temporal analysis for all sampling sites. Long-term values were calculated by the median of 1985, 1995, 2005 and 2016. Recent year values were calculated by the median of the years during which sampling occurred – 2011, 2012 and 2013.

Regarding the amphibian community present at each sampling site, the higher percentage of microendemisms can be found at Tavolotara and Analafohy, both around 85% (Table 3.7). Despite this, Tavolotara has a very low STC. Anosiarivo has the highest STC but the lowest SR and no microendemism were found here (it is important to take into account that this site was only surveyed during 2013). Nearly 50% of the amphibians found in Ambitsika were microendemic but restricted to only one of the two species – *M. pauliani.*

Table 3.7 – Summary statistics for biodiversity metrics calculated for each sampling site.

Sampling Sites	SR	N	ME	STC
Ambohimirandrana	6	372	46.33	21.59
Tavolotara	6	751	85.49	9.94
Tsimiaramianadahy	7	281	17.77	25.12
Analafohy	5	410	85.10	11.60
Ambitsika	4	66	49.62	24.43
Anosiarivo	3	46	0.00	65.22

SR – Species Richness (mean value of the 2 sampled years); N – Amphibian Abundance (mean value of the 2 sampled years); ME – Microendemism Rate (number of adults and juveniles of *Boophis williamsi* and *Mantidactylus pauliani* found throughout the 2 sampling years/ number of specimens found throughout the sampling years*100); STC – Species Turnover Capacity (number of tadpoles found throughout the 2 sampling years/ number of specimens found throughout the sampling years*100). SR and N are presented in absolute values, ME and STC are presented in percentage.

3.4 Amphibian Community Responses to Landscape Change: Deforestation and Ecosystem Functional Attributes

Three hypotheses – deforestation (H1), primary productivity (H2) and thermal variation (H3) – were defined to assess the role of landscape changes over amphibian community patterns represented by four dimensions (species richness, amphibian abundance, microendemism rate and species turnover capacity). For this set of analyses, data from only five out of the six sampling sites was used, to avoid the creation of bias in the models, since Anosiarivo was sampled only in 2013. Also, SAVI measures were not included, as this index was found to be very correlated with NDVI.

Results indicated that the most parsimonious hypothesis varied according to the biodiversity community metric selected and also according to time and spatial scale. Species richness and microendemism rate are both mostly affected by long-term changes in the ecosystem variables (H2 and H3, respectively), whereas amphibian abundance and species turnover rate were more influenced by recent changes in ecosystem variables (H2), at a 60m buffer and at a 30m buffer respectively. Although not pointed out as the best model to explain the current biodiversity patterns on the Ankaratra Massif, deforestation rate cannot be discarded as a predictor affecting the response variables, especially when analysing species richness. Having pointed this out, the best model in all three other biodiversity metrics explains nearly 100% of the measured values (Deviance Explained; Kindt and Coe, 2005). Full models (models containing all predictor variables) always had the highest percentage of explained deviance, but were never the best explaining model, as, in this model ranking method, a simple explanation is always preferable than a more complex one. Here, full models constantly showed ΔAICc > 2, thus being rejected for not being good competitors with the best model (Anderson and Burnham, 2002). Temporal scale is much more important for the response variables analysed than spatial scale, as there is no big difference between the same models within the same temporal scale, although spatial scales might be too similar to get very different responses.

Accordingly, long-term NDVI (proxy for primary productivity) at a 60m resolution is the model that best describes present day SR – higher levels of NDVI relating to lower levels of SR (Table 3.8; H2: AICc = 27.74, ΔAICc = 0.00, w_i = 0.56, Deviance Explained = 94.80%, Dispersion Test = 0.010, SpCorr = -0.97). However, the Akaike weight value (w_i = 0.56), suggests that deforestation rates may also play an important role, since this model is only 1.27 times better supported than the best model (evidence ratio = 0.56/0.44). Higher levels of deforestation rate negatively affect SR (Table 3.12; H1: AICc = 28.21, ΔAICc = 0.47, w_i = 0.44, Deviance Explained = 45.54%, Dispersion Test = 0.105, SpCorr = 0.53). Recent changes Albedo (proxy for thermal variation, H3) within both buffers also explains a great deal of the species richness present on the Massif (30m: AICc = 27.79, ΔAICc = 0.00, w_i = 0.55, Deviance Explained = 89.45%, Dispersion Test = 0.021, SpCorr = 0.97; 60m: AICc = 27.84, ΔAICc = 0.00, w_i = 0.60, Deviance Explained = 84.22%, Dispersion Test = 0.031, SpCorr = 0.97) as well as deforestation rate within a 30m buffer (AICc = 28.16, ΔAICc = 0.37, w_i = 0.45, Deviance Explained = 50.97%, Dispersion Test = 0.096, SpCorr = 0.73). Albedo is positively related to SR (Table 3.12), which means the higher the Albedo values (cooler land surface temperatures), the more species exist.

Table 3.8 – Results of Multi-Model Inference approach explaining observed species richness in Ankaratra Massif Reserve.

	Species Richness									
	Spatial Extent	Hypotheses	Loglik	df	AICc	ΔAICc	w_i	Deviance Explained (%)	Dispersion Test	SpCor
Long-term ecosystem variables	30 m	H2	-8.90	2	27.80	0.00	0.56	88.70	0.021	-0.97
		H1	-9.13	2	28.27	0.47	0.44	39.80	0.115	0.37
		H1*H2	-8.89	3	47.79	19.99	0.00	89.80	0.019	-
	60 m	H2	-8.87	2	27.74	0.00	0.56	94.80	0.010	-0.97
		H1	-9.11	2	28.21	0.47	0.44	45.54	0.105	0.53
		H1*H2	-8.86	3	47.73	19.98	0.00	96.44	0.007	-
Recent ecosystem variables (2011-2013)	30 m	H3	-8.90	2	27.79	0.00	0.55	89.45	0.021	0.97
		H1	-9.08	2	28.16	0.37	0.45	50.97	0.096	0.73
		H1*H3	-8.88	3	47.75	19.96	0.00	93.90	0.012	-
	60 m	H3	-8.92	2	27.84	0.00	0.60	84.22	0.031	0.97
		H1	-9.32	2	28.65	0.81	0.40	0.07	0.191	0.16
		H1*H3	-8.87	3	47.74	19.90	0.00	94.94	0.009	-

The competing hypotheses are listed in descending order from the best to the least fit hypothesis determined by AICc values. H1 – Deforestation rate; H2 – NDVI; H3 – Albedo. H2 and H3 are proxies for ecosystem functioning.

Loglik log-likelihood; *df* degrees of freedom; *AICc* Corrected Akaike Information Criterion value; *ΔAICc* measures the difference between the best model (*ΔAICc* = 0) and the remaining models; w_i represents the Akaike weights and measures how much each model weighs when explaining the response variable; *Deviance explained* measures how much of the response variable is explained by each competing model; *Dispersion test* measures how much dispersion exists within the model.-*SpCor* is the Spearman correlation between the response variable and each predictor.

Recent changes in NDVI seem to affect present day amphibian abundances the most (Table 3.9; AICc = 71.10, ΔAICc = 0.00, w_i = 0.99, Deviance Explained = 95.06%, SpCorr = 1.00). As there is such a high correlation between NDVI and amphibian abundance, no other model was taken into account when analysing this biodiversity metric. This relation follows a positive trend (Table 3.12), the higher the NDVI values, the more amphibians were found.

Table 3.9 – Results of Multi-Model Inference approach explaining observed amphibian abundance in Ankaratra Massif Reserve.

	Amphibian Abundance								
	Spatial Extent	Hypotheses	Loglik	df	AICc	ΔAICc	w_i	Deviance Explained (%)	SpCor
Long-term ecosystem variables	30 m	H3	-27.75	3	85.50	0.00	0.53	11.90	-0.20
		H1	-27.85	3	85.71	0.21	0.47	8.40	-0.05
		H1*H3	-27.53	4	Inf	Inf	0.00	19.00	-
	60 m	H3	-27.49	3	84.97	0.00	0.63	20.30	-0.20
		H1	-28.01	3	86.03	1.05	0.37	2.70	0.36
		H1*H3	-27.37	4	Inf	Inf	0.00	23.87	-
Recent ecosystem variables (2011-2013)	30 m	H2	-20.77	3	71.54	0.00	0.99	94.61	1.00
		H1	-26.01	3	82.02	10.49	0.01	54.93	0.71
		H1*H2	-20.75	4	Inf	Inf	0.00	94.66	-
	60 m	H2	-20.55	3	71.10	0.00	0.99	95.06	1.00
		H1	-25.46	3	80.93	9.83	0.01	63.65	0.56
		H1*H2	-20.54	4	Inf	Inf	0.00	95.07	-

The competing hypotheses are listed in descending order from the best to the least fit hypothesis determined by AICc values. H1 – Deforestation rate; H2 – NDVI; H3 – Albedo. H2 and H3 are proxies for ecosystem functioning.

Loglik log-likelihood; *df* degrees of freedom; *AICc* Corrected Akaike Information Criterion value; *ΔAICc* measures the difference between the best model (*ΔAICc* = 0) and the remaining models; w_i represents the Akaike weights and measures how much each model weighs when explaining the response variable; Deviance explained how much of the response variable is explained by each competing model. *SpCor* is the Spearman correlation between the response variable and each predictor.

Microendemism rates were best explained by long-term changes in thermal conditions (Table 3.10; AICc = 68.16, ΔAICc = 0.00, w_i = 0.98, Deviance Explained = 80.23%, SpCorr = -0.50). This relationship does not favour the presence of high levels of microendemic species found, as the two variables are negatively correlated – the higher the Albedo values, the less microendemic species will be found. For this variable, no other predictors seem to be of importance to explain observed variables.

Recent changes in NDVI were fully supported (w_i = 1) as the best model in explaining species turnover capacity (Table 3.11; AICc = 42.12, ΔAICc = 0.00, Deviance Explained = 97.22%, SpCorr = -1.00). According to Spearman's correlation, NDVI is linearly correlated with amphibian abundances and with species turnover capacity.

Table 3.10 – Results of Multi-Model Inference approach explaining observed microendemism rate in Ankaratra Massif Reserve.

	Microendemism Rate								
	Spatial Extent	Hypotheses	Loglik	df	AICc	ΔAICc	w_i	Deviance Explained (%)	SpCor
Long-term ecosystem variables	30 m	H3	-20.23	3	70.46	0.00	0.94	69.10	-0.50
		H1	-23.02	3	76.03	5.57	0.06	10.00	-0.21
		H1*H3	-19.84	4	Inf	Inf	0.00	73.50	-
	60 m	H3	-19.08	3	68.16	0.00	0.98	80.23	-0.50
		H1	-23.26	3	76.52	8.36	0.02	0.04	0.10
		H1*H3	-19.08	4	Inf	Inf	0.00	80.28	-
Recent ecosystem variables (2011-2013)	30 m	H2	-22.19	3	74.38	0.00	0.73	33.92	0.90
		H1	-23.19	3	76.38	2.01	0.27	2.62	0.35
		H1*H2	-20.03	4	Inf	Inf	0.00	71.63	-
	60 m	H1	-21.51	3	73.02	0.00	0.71	49.23	0.56
		H2	-22.40	3	74.81	1.79	0.29	28.17	0.90
		H1*H2	-21.44	4	Inf	Inf	0.00	50.44	-

The competing hypotheses are listed in descending order from the best to the least fit hypothesis determined by AICc values. H1 – Deforestation rate; H2 – NDVI; H3 – Albedo. H2 and H3 are proxies for ecosystem functioning.

Loglik log-likelihood; *df* degrees of freedom; *AICc* Corrected Akaike Information Criterion value; *ΔAICc* measures the difference between the best model (*ΔAICc* = 0) and the remaining models; w_i represents the Akaike weights and measures how much each model weighs when explaining the response variable; Deviance explained how much of the response variable is explained by each competing model. *SpCor* is the Spearman correlation between the response variable and each predictor.

Table 3.11 – Results of Multi-Model Inference approach explaining observed species turnover rate in Ankaratra Massif Reserve.

	Species Turnover Rate								
	Spatial Extent	Hypotheses	Loglik	df	AICc	ΔAICc	w_i	Deviance Explained (%)	SpCor
Long-term ecosystem variables	30 m	H2	-14.89	3	59.77	0.00	0.53	4.95	0.10
		H1	-15.01	3	60.02	0.25	0.47	0.08	0.05
		H1*H2	-14.85	4	Inf	Inf	0.00	6.42	-
	60 m	H1	-14.44	3	58.88	0.00	0.52	20.43	0.20
		H3	-14.50	3	59.01	0.12	0.48	18.47	-0.36
		H1*H3	-13.42	4	Inf	Inf	0.00	47.07	-
Recent ecosystem variables (2011-2013)	30 m	H2	-6.06	3	42.12	0.00	1.00	97.22	-1.00
		H1	-13.07	3	56.13	14.04	0.00	54.09	-0.71
		H1*H2	-5.40	4	Inf	Inf	0.00	97.86	-
	60 m	H2	-7.72	3	45.44	0.00	0.99	94.59	-1.00
		H1	-12.23	3	54.46	9.02	0.01	67.12	-0.56
		H1*H2	-7.52	4	Inf	Inf	0.00	95.01	-

The competing hypotheses are listed in descending order from the best to the least fit hypothesis determined by AICc values. H1 – Deforestation rate; H2 – NDVI; H3 – Albedo. H2 and H3 are proxies for ecosystem functioning.

Loglik log-likelihood; *df* degrees of freedom; *AICc* Corrected Akaike Information Criterion value; *ΔAICc* measures the difference between the best model (*ΔAICc* = 0) and the remaining models; w_i represents the Akaike weights and measures how much each model weighs when explaining the response variable; Deviance explained how much of the response variable is explained by each competing model. *SpCor* is the Spearman correlation between the response variable and each predictor.

Table 3.12 – Model-averaged coefficients (β) and unconditional standard errors (STE) for the AICc-based models.

	Species Richness			
	Spatial Extent	Hypothesis	β	STE
Long-term ecosystem variables	30 m	H2	-0.0952	0.1614
		H1	0.0527	0.1432
	60 m	H2	-0.0979	0.1615
		H1	0.0565	0.1453
Recent ecosystem variables (2011-2013)	30 m	H3	0.0916	0.1558
		H1	0.0646	0.1598
	60 m	H3	0.0968	0.1576
		H1	-0.0019	0.1194

	Microendemism Rate			
	Spatial Extent	Hypothesis	β	STE
Long-term ecosystem variables	30 m	H3	-0.0176	0.0068
		H1	-0.0003	0.0024
	60 m	H3	-1.956E-02	5.14E-03
		H1	-6.239E-06	1.07E-03
Recent ecosystem variables (2011-2013)	30 m	H2	0.0080	0.0077
		H1	0.0009	0.0047
	60 m	H1	0.0104	0.0085
		H2	0.0029	0.0060

	Amphibian Abundance			
Long-term ecosystem variables	30 m	H3	-0.0018	0.0032
		H1	-0.0011	0.0028
	60 m	H3	-0.0027	0.0035
		H1	0.0006	0.0038
Recent ecosystem variables (2011-2013)	30 m	H2	9.245E-03	1.28E-03
		H1	4.098E-05	5.99E-04
	60 m	H2	9.738E-03	1.38E-03
		H1	5.826E-05	7.11E-04

	Species Turnover Capacity			
Long-term ecosystem variables	30 m	H2	-0.1182	0.4250
		H1	0.0124	0.3954
	60 m	H1	-0.2328	0.4332
		H3	0.2083	0.4218
Recent ecosystem variables (2011-2013)	30 m	H2	-0.9851	0.1007
		H1	-0.0007	0.0250
	60 m	H2	-0.9620	0.1674
		H1	-0.0089	0.0917

The competing hypotheses are listed in descending order from the best to the least fit hypothesis determined by AICc values. H1 – Deforestation rate; H2 – NDVI; H3 – Albedo.

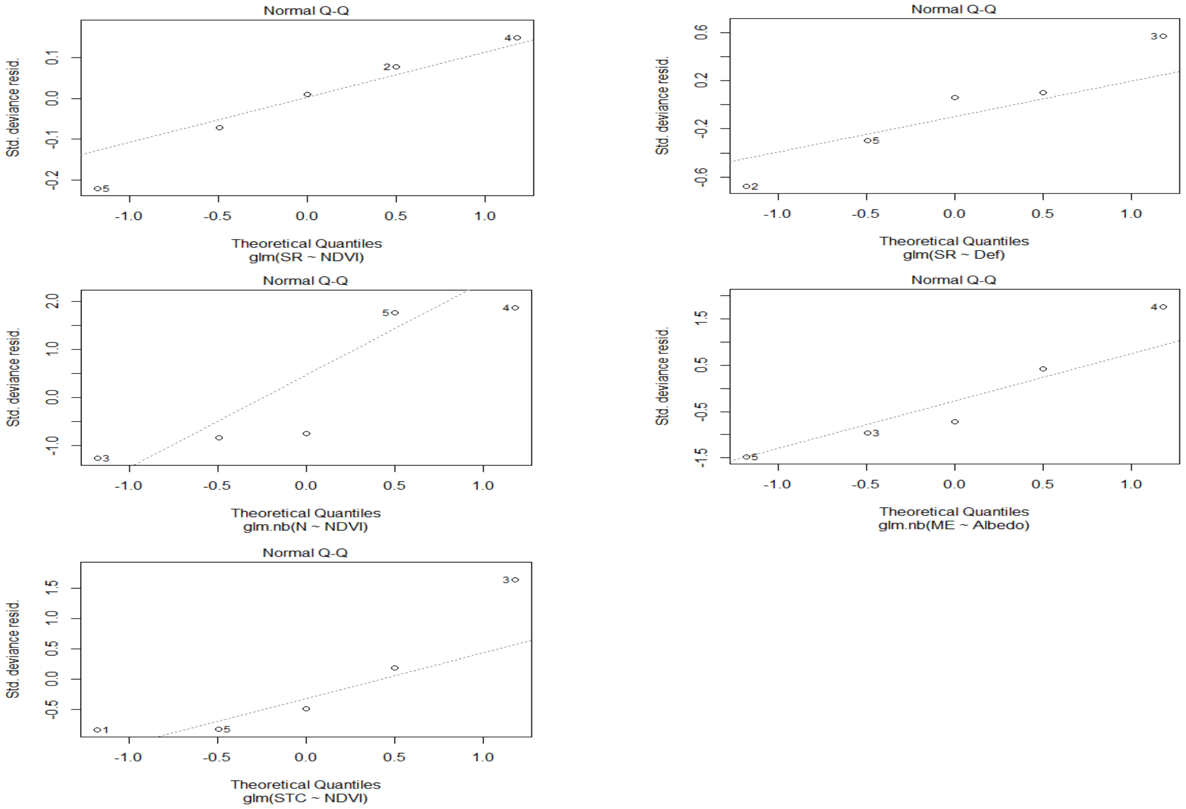

Figure 3.9 – Plot of the residuals of the best fitting model for each response variable.

4. Discussion

4.1 Amphibian Community Patterns and Environmental Change in the Ankaratra Massif Reserve

The amphibian community of the Ankaratra Massif Reserve analysed in this study is composed mostly of species that are fairly able to adapt to different habitats, with the exception of the two microendemic species – *B. williamsi* and *M. pauliani* – which are highly specialized species that do not penetrate into the agricultural matrix that surrounds the Reserve. Moreover, in addition to the already mentioned anthropogenic factors that threaten these species, climate change can also represent a threat (Pearson et al., 2014), especially for microendemic high-altitude species, such as *B. williamsi* and *M. pauliani.* The presence of the chytridiomycosis causing fungus, which has been recently detected in Madagascar (Kolby, 2014; Kolby et al., 2015; Bletz et al., 2015a, 2015b) might represent another possible threat. In Ankaratra, different species (*M. pauliani* included) have already been tested positive for this fungus albeit so far no verified symptoms of chytridiomycosis have been detected.

When analysing species abundances (N) and percentage of tadpoles (STC), this amphibian community shows some degree of seasonality in some of the sampled sites (Fig. 3.1), but it cannot be said that this seasonality is shared throughout the whole Massif. In fact, there is a large range of variation in seasonality between sites and they seem not to share a common pattern. Seasonality is most likely to affect sites with open areas of grassland and shrubs, as levels of greenness and, consequently, humidity are more variable here (Williams-Linera et al., 1998; Fig. 3.7). Microclimates within dense forested canopies are more stable throughout the year (Hardwick et al., 2015). The unclear observed pattern could be due to the sampling method that was temporally inconsistent – it occurred once every season throughout the two sampling years but it did not occur during a defined month within each season – or it could be due to the low seasonality that generalist, highly adaptable species seem to have, contrary to the more sensitive specialist species (Andreone, 1994). Tadpoles were encountered throughout the entire year and this is probably due to being in high altitude streams. Although frogs will probably breed during the wet season (Glaw and Vences, 2007), tadpoles are prone to prolong their larval stage for up to six months when conditions are not optimal for their metamorphosis. However, further studies of the ecology of these species, particularly of *B. williamsi* and *M. pauliani* are required to answer this question. Although no evidence of seasonality was detected across the surveyed period for the two microendemic species, lines of arrested growth on bones are present in both species (Andreone et al., 2014) and *B. williamsi* males can display nuptial pads (Vences et al, 2002; Andreone et al., 2014) - both these features being signs of seasonality. In tropical and subtropical amphibian species, lines of arrested growth are thought to be the result of seasonal fluctuation of food availability that is a response to seasonal changes in temperature (Guarino et al., 1998). One possible explanation for the apparent absence seasonality observed throughout the survey is that since surveys were specifically focused to monitor the population trends of these two critically endangered species, more effort was invested in finding individuals of these species. Alternatively, it is worth noting that surveys were focused along the streams, where species might concentrate during the breeding season for reproduction and where they might aggregate during the dry season due to the reduced availability of other humid places.

The higher number of amphibians was found in Tavolotara, where the forest has been fully recovered after an old forest clearing (Table 3.5), followed by Analafohy (Table 3.6). ME was also the highest there. Both these sites have high percentages of forest cover (Table 3.5). Overall, fewer species were found in Ambitsika and Anosiarivo (Table 3.6). These two sites have suffered recent forest clearing and displayed an estimated forest coverage of less than 50% in 2016 (Table 3.5).

4.1.1 The Story of an Unnamed Species

During the survey, whose results are here analysed, the team that carried out the sampling identified some individuals as *Mantidactylus mocquardi* (or *M. sp. aff. mocquardi*)*, a* species that has never been reported for the Ankaratra Massif. There are two possible explanations for this occurrence: 1) the individuals identified as *M. mocquardi* are, in reality, remarkably large female of *M.* sp. aff. *curtus* Ca19, that have prominent femoral glands (generally present only in males but present in a vestigial form also in females of the genus *Mantidactylus*); 2) the specimens truly belong to a further new species that has not yet been reported for the area. A study conducted by Vieites et al. (2009) genetically examined 2,850 specimens from all over Madagascar, discovering a large percentage of undescribed diversity. For Ankaratra, 40 individuals of *Mantidactylus* were analysed and all of them were assigned to the candidate new species *Mantidactylus* sp. 19 (Vieites et al., 2009, Supplementary Material; here named as *M.* sp. aff. *curtus* Ca19). Recently collected material (September and October 2010 and November 2014) confirms the wide occurrence of *Mantidactylus* sp. aff. *curtus* Ca19, but also reveals the occurrence of a further new species for the area (*M.* sp. aff. *curtus* Ca18; A. Crottini pers. Comm.) which was already known from Ambohitantely (another locality in the Central Highlands of Madagascar). This discovery increases the biodiversity value of the Ankaratra Massif Reserve, as *M.* sp. aff. *curtus* Ca19 is currently only known from this site, with a potential four microendemic species solely inhabiting this Massif. This confirms Ankaratra as an important centre of microendemism, where repeated effort should be invested in order to preserve this unique biodiversity.

4.1.2 A Changing Landscape

The analysis on land cover and functional attributes of the Ankaratra Massif Reserve with Landsat satellite data from the last thirty years (1985 – 2016) highlighted that the amphibian community present here is settled in a very dynamic landscape (Fig. 3.3), marked by an intense pressure on forest ecosystem. Grasslands have been the most stable land cover class throughout the time interval analyzed, covering nearly 50% of the study area (Table 3.2). They mostly dominate the higher altitude plains, while forest is generally present on the more irregular terrain. The higher altitude grasslands are likely to occur here naturally and not be the consequence of anthropogenic activities, unlike the grasslands on the lower Plateau level (Vorontsova et al., 2016). To test the degree of disturbance of these grasslands, a species inventory should be carried out to identify levels of endemism and degree of heterogeneity.

The area occupied by crops seems to have decreased to almost half since 1985. This could be due to the growing exploitation of the lowlands for rice and potato production, leaving the higher agricultural lands to be converted into pastures for the grazing of livestock. Pastures can be easily confused with natural grasslands when performing the supervised image classification. Other possible explanations for this apparent decrease of agricultural lands may be the decline in soil fertility (Vågen et al., 2006) or may be due to the conversion of rice paddies into grassland as a result of a reduced amount of water available (Vallan, 2002), both conditions caused by deforestation. Also, there has been a clear increase in barren land since 1995.

Although a global growing trend of conversion of Tropical Montane Cloud Forest into agricultural and animal husbandry field areas has been observed over the past years (Muñoz-Villers and López-Blanco, 2008), in Ankaratra an increase in the area covered by shrubs and small trees suggests that forest regrowth is occurring, even though the area of remaining forest is the lowest for the last thirty years. Approximately 18% of forest cover was lost between 1985 and 1995, a period during which the political situation in Madagascar was highly unstable, weakening the already fragile environmental policies and resulting in illegal forest clearance in Madagascar's highlands (Vågen, 2006). From 1995 to 2005, there seems to have been a rapid recovery of forest coverage, with an increase of the forested area to nearly 40% of its 1985 values. This tendency contrasts with that reported for sub-tropical regions by Keenan et al. (2015) and is most likely the consequence of forest management by a federation of villagers' association, the Union Forestière d'Ambatolampy (UFA), which signed a 20-year contract to manage Ankaratra in 1998 (ASA & ASG, 2010). After this large restoration, a new period of deforestation was observed in the 2005 to 2016 period, with a 5.10% of forest clearance every year. This deforestation process sharply increased in 2007, which coincides with the suspension of the UFA's contract by the Regional Directorate (ASA & ASG, 2010). We must additionally take into account that this large fluctuation in forested areas may be in part influenced by the economic exploitation of pine trees in the surveyed area, however keeping in mind that a pine tree will not be harvested before it is 20 years old (Townsley et al., 2001). It was not possible to differentiate between forested areas used for this purpose and native forest using the remote sensing techniques applied in this work because discriminating dominant tree species between forests (in this case differentiating forests in which pine trees were dominant from forests with a more heterogeneous assemblage of trees) could not be done as both types of forests produced similar NDVI values and similar NDVI temporal trends. This is a very common case among assemblages of different plant species (Pettorelli et al., 2005) and it is an issue that should definitely be taken into consideration in the future development of this work.

During the time of the amphibian survey, an exceptionally high deforestation rate was observed, with 12.20% of forest clearance every year, from 2011 to 2013. This considerable rate of forest clearance is most likely resultant from, yet again, turbulent political events that started in 2009, a time during which the illegal exploitation of wood for charcoal and bush fires intensified (Rahantaliosa et al., 2011). When analyzing the fraction of forest that has remained stable since 1985, the result is a small, fragmented area, where only 42% of the 1985 forest remained (Fig. 3.4).

Forest fragmentation acts in synergy with forest clearance, leaving an ever-decreasing area of optimal habitat for forest interior dwelling species. This is due to changes in the abiotic characteristics of forest edges, such as reduced humidity and increased maximum daily temperatures relative to the forest interior (Didham and Lawton, 1999; Hardwick et al., 2015). The abiotic changes that occur after forest clearance, lead to altered species abundances, distributions and composition (Ewers and Didham, 2006). Additionally, species that avoid the surrounding matrix tend to decline or disappear in fragments (Gascon et al., 1999) and Lehtinen et al. (2003) show that Malagasy herpetofauna strongly respond to altered microclimates near cleared forest edges.

Schneider-Maunoury et al. (2016) found that the edge sensitivity effects on herpetofauna can be identified as far as approximately 250m into the forest. As the forest cover of the Massif has been decreasing since 2005 (Fig. 3.3), edge sensitivity must be taken into account when designing a management plan for the conservation of the two microendemic amphibians, as this seems to be correlated with extinction vulnerability (Lehtinen et al., 2003) and specialized species are more vulnerable to fragmentation, by not being able to cross unsuitable matrices between forest patches (Henle et al., 2004). *B. williamsi* is suspected to be highly vulnerable to edge effects, due to predicted adult dispersal limitations through unsuitable habitats. These limitations seem to affect *M. pauliani* to a smaller extent because being a stream frog it can possibly better tolerate the unfavourable microclimatic conditions created by deforestation (Vallan, 2000).

Another dimension of the landscape strongly affected by landscape/land cover change is the functional dimension. This component is especially relevant for the amphibian community. Of all climatic components that make up an ecosystem, temperature and moisture are the two that cause greater impact on amphibians (Carey and Alexander, 2003). A frog maintains its body temperature by heat exchanges with the environment (through air, water, soil and solar radiation) and preserves the humidity of its skin according to the humidity within its surroundings (Hutchison & Dupré, 1992). Humidity is both connected to air temperature and to the density of the canopy/greenness coverage of the land (Hardwick et al., 2015).

Landscape functional change effects were assessed indirectly through satellite derived information. Thermal variation was accounted for through changes in Albedo values, which is a proxy for land surface temperature. The amount of greenness related to vegetation productivity was measured through NDVI. While a decrease in vegetation productivity values (NDVI) from 1985 to 2005 was observed, these appear to be increasing again in recent years (Table 3.7; Fig. 3.6). Contrarily, Albedo values do not suggest to be returning to their baseline values, but seem to be in a continuing downward trend, meaning land surface temperature is slowly but steadily increasing (Table 3.7; Fig. 3.6). According to Hardwick et al. (2015), average maximum temperatures can rise up to 2.5°C in logged forests relative to undisturbed forests, having a high impact on biodiversity, especially on ectotherms from the tropics that are already living close to their optimum temperature (Deutsch et al., 2008). This increase of land surface temperature is particularly critical for amphibians, whose physiological and biochemical processes have an increase in rate by two to threefold for every 10°C rise in body temperature (Rome et al., 1992).

Deforestation of this high altitude montane cloudy forest will further impact on the region's microclimate, as the presence of the almost constant fog will most likely disappear along with the decreasing forest. This fog is of extreme importance for sustaining high atmospheric-humidity levels, providing extra hydrological and nutritional input ("occult" precipitation) to the forest and reducing solar irradiance (see Bruijnzeel et al., 1998).

4.2 The Role of Temporal and Spatial Scale of Ecosystem Functional Variables on the Amphibian Community of the Ankaratra Massif Reserve

Along the highly dynamic landscape of the Ankaratra Massif, hypotheses associated to the ecosystem functioning variables resulting from the alterations in land cover were the most supported when explaining the four biological components of the amphibian community (species richness, amphibian abundance, microendemism rate and species turnover rate). Time and spatial extent affect each of the four biological aspects here analyzed differently, however the temporal scale seems to play a more important role (see section 3.4 in Results). The fact that the spatial scale does not playing such an important role as the temporal scale might be due to the spatial scales defined being very similar to each other. Thus, more disparate spatial scales should be tested.

Ecosystem functioning variables have a more immediate effect on biodiversity than the deforestation rates *per se,* whose influence is only detected on a long-term time scale and on the number of species that can now be found on the Ankaratra Massif. Species richness is mostly affected by long-term changes in the amount of greenness coverage (vegetation productivity), but, as already mentioned, long-term deforestation rate also plays an important role, by negatively affecting it (Tables 3.8 and 3.10). Species richness around the globe has been found to be constrained by multiple factors working together (Buckley and Jetz, 2007), so it is of no surprise to find that recent changes in land surface temperature also play an important role in shaping this biodiversity metric. This relationship is found to work in the direction that the cooler the land surface temperature, the higher the number of species will be present in a given place. Despite all of these variables being meaningful in explaining species richness, Buckley and Jetz (2007) observed that if only one single predictor were to be mentioned, then the energy supply of an ecosystem would be the strongest to predict amphibian species richness. This is consistent with the multi-model inference of this study as vegetation productivity (H2) is the hypothesis that explains greater deviance and the factor that has the smallest dispersion values. Here, the relation between H2 and species richness works in the way that higher NDVI values will present places with fewer species. Grasslands and recently grown shrubs have higher NDVI values than forest due to their higher photosynthetic activity. In our study area, grasslands below 2000m are most likely to be signs of habitat disturbance, being pastures or cleared forest. Likewise, recently grown shrubs throughout the study area are also sign of the system trying to recover back into forest or into old growth savannah. Thus, having intermediate values of NDVI, it can be inferred that the forest environment has the highest species richness.

Long-term changes in the thermal condition of the landscape, proxy by Albedo effect on land surface temperature was the variable that better explained the current numbers of *B. williamsi* and *M. pauliani*, suggesting that the microendemic amphibian species that inhabit Ankaratra are very sensitive to alterations in the thermal conditions associated to land cover change. This brings implications regarding global warming as decreasing humidity and increasing temperatures can be limiting factors affecting amphibians adapted to the stable climatic conditions within primary habitats (see Vallan, 2002). This issue is particularly critical concerning mountain-top species, such as *B. williamsi* and *M. pauliani*, as the area available for suitable occupancy decreases (Elsen and Tingley, 2015).

Recent changes (analyzed period between 2011 and 2013) in the amount of greenness present in the ecosystem play the largest role when analyzing amphibian abundance and species turnover rate. When looking at the values used to analyze this temporal scale, it must be kept in mind that these three years suffered from intense deforestation rates (12.20% every year), which immediately affects the observed values of NDVI. As vegetation dynamics are intrinsically linked to local climate (Pettorelli et al., 2005), this implies that the number of individuals [both of adults and juveniles (N) and of tadpoles (STC)] found throughout the survey could be the result of the deforestation that has been occurring in recent years since this, as seen above, alters the microclimate of the region. Although both amphibian abundance and species turnover rate appear to be extremely correlated to the recent changes in greenness values (primary productivity), the first response variable is positively correlated to greenness and the second response variable is negatively correlated with it (Tables 3.9 and 3.11). This could be due to the tadpoles being found in the streams. Water has much lower NDVI values than other surfaces (Pettorelli et al., 2005). Nonetheless, these results precisely coincide with the work done by Semlitsch et al. (2009), who demonstrate that forest clearance negatively affects the juvenile and adult stages but positively affects oviposition and larval stages, which are more water dependent phases of an amphibian's life cycle. Similarly, Skelly et al. (2005) found that larval development in herbivorous tadpoles is positively related to light levels in breeding ponds, because the increase in water temperature enhances periphyton activity.

4.3 Insights for Conservation in the Ankaratra Massif Reserve

Being the first extensive study on the landscape of the Ankaratra Massif, this work has shown that this area is under an increasing pressure on its available land resources, although there has been a recent reduction in the rate at which forest is being cleared (from 2013 to 2016, relative to the previous analyzed periods). Nonetheless, the maintenance of the Central Highland's grasslands and savannah/heathland alongside the remnant montane forest patches is of extreme importance for maintaining the irreplaceable biodiversity of the Ankaratra Massif Reserve.

Previous studies have found a negative relation between deforestation and stream flow, with reported reductions of the amount of water in streams following forest clearance (Moraes et al., 1998; Vågen, 2006). Tropical Montane Cloud Forest is essential to maintain the quality of upland water resources (Bruijnzeel and Proctor, 1995). Thus, it is essential to protect the remaining forest, not only for the safeguarding of the unique species present here, but also to preserve the water resources it offers, being the only source of drinking water and irrigation for Ankaratra's population (Rakotozafy, 2017).

Preservation of even the smallest forest fragment can be of significant value for the survival of a considerable amount of fauna and flora (Andreone 2007). Furthermore, as old-growth primary forests are continuously being degraded and lost, the conservation value of secondary tropical forests becomes of upmost importance and, with time, can restore the functional attributes of old-growth forests (DeWalt et al., 2003).

Madagascar's montane herpetofauna is relatively resistant to fire (Raxworthy and Nussbaum 1996). Theses authors propose that if post-fire secondary heathland is protected from cattle grazing and from further burning, it could act as dispersal corridors between montane forest blocks, offering new opportunities for conservation.

All species are not equal in terms of conservation importance. When managing protected areas, this has to be taken into account so as to direct conservation efforts efficiently, as direct actions towards all components of an ecosystem is logistically and economically unrealistic. In this Reserve, it is vital to direct efforts to the conservation of the microendemic species that are found here; not only focusing on both the amphibian species on which this work was focused, but also turning conservationists' attention to the Marvellous gecko (*Lygodactylus mirabilis)*. Not only are these species found nowhere else in Madagascar nor in the world, but being forest dwelling species, these two amphibians are more prone to extinction with the ever decreasing forest. Moreover, all of these three microendemics are strictly mountain-top species and thus very susceptible to warming temperatures and upslope displacement.

Abundance measurements are very important indices when analysing ecosystem health, providing different information and, for this issue, being more useful than SR which fails to capture fragmentation impacts on population viability (Banks-Leite et al., 2012). Declining populations, caused by degraded and altered habitat, eventually lead to extinction, even if only on a local scale (Schneider-Maunoury et al., 2016). Species richness is important to assess how much biodiversity an ecosystem holds but it does not indicate health levels, as it quantifies small populations as being equal to large populations. Thus, species richness *per se* is a poor indicator of conservation progress in a protected area and of conservation value of an ecosystem (Barlow et al., 2007; Gardner et al., 2007). For an effective assessment of the progress and evolution of the Ankaratra Massif as a Nature Reserve, seasonal abundance surveys of its three microendemic species (*B. williamsi, M. pauliani* and *L. mirabilis*) must occur as well as regular assessment of habitat health.

It is recommended that controlled reforesting is taken into consideration, together with the preservation of the open areas.

Finally, the coupling of in-field observation with remote sensing data can contribute to the long-term monitoring of the Ankaratra Massif Reserve, by allowing the estimation of landscape change, related change in functional attributes of its ecosystem and by supporting the modelling of its biodiversity.

If the measures described above are taken into action, with the addition of controlling stream pollution, there might still be a future for the amphibians of the Ankaratra Massif Reserve, especially for the microendemic species that survive there and nowhere else in the world. If the efficient protection of this montane habitat does not occur and degradation of TMCF continues at the rates observed from 2005 to 2016 (especially in the period between 2011 and 2013), then surely we will witness the extinction of yet two more irreplaceable amphibian species and with them the local Malagasy communities will lose a part of their unique ecosystem on which they rely on for keeping their water sources clean and their mountain from eroding.

5. Conclusions

This study makes a first preliminary assessment of the high-altitude amphibian community present in the newly established Ankaratra Massif Reserve and how it relates to changes in landscape conditions.

Due to the small number of sampled sites and to the imprecise temporal sampling, further intensive studies should be performed in order to increase the study's robustness. To do this, it is proposed that future surveys be performed under a strict seasonal turnaround, sampling both dry and wet seasons in the same month across all years, so that data will not be biased by the sampling period (surveys taking place at the beginning of the season one year and at the end of the season in another year), and the length of the transect should be standardized.

Also, if at all possible, it will be interesting to extend the number of sampling sites, in order to maximize the efficiency of detection and have better abundances data for all the surveyed species of the Ankaratra Massif Reserve, including the sites at lower altitude. Another downside of this study is the time inconsistency across sampled sites, as some sites have data available throughout the two years of the survey period, whereas others have only one year of data available. These points are surely true weak points of this study, however we must always keep in mind that remote areas are extremely difficult to be surveyed and, although far from perfect, the data here analysed is the only data so far available on these Critically Endangered frog species endemic to the Ankaratra Massif. Therefore, this data deserves to be considered of unique value.

Regarding land use/land cover transitions and ecosystem interactions throughout time, SRS-based techniques were of the utmost importance to uncover the fluctuations that have been happening in the past thirty years. Increasingly SRS-based information is being used in ecology studies and to monitor diverse aspects of biodiversity levels around the world (Pettorelli et al., 2014b). The future of both SRS and of ecology and conservation is for them to evolve together as new initiatives such as the Digital Observatory for Protected Areas (DOPA; http://dopa.jrc.ec.europa.eu/en) start to appear, seeing that conservation becomes an ever more global concern. This study provides one more successful example of the alliance between these disciplines in the broadening of the knowledge of a recently developed nature reserve of which little is still known.

Having acknowledged the occurrence of extensive habitat degradation and knowing of the presence of Bd in Ankaratra, it is vital that a greater effort is made towards the conservation of its microendemic species by including them in the captive breeding programmes that are being carried out in Madagascar (Andreone et al., 2008; Dawson et al., 2014). Additionally, starting to gather information on the ecology and genetics of *B. williamsi* and *M. pauliani* is crucial for a more efficient conservation planning in order to prevent their extinction. Additionally, it would be interesting to see whether the diversification time between these two microendemic species and their respective sister species coincides with the decrease of montane forest after the Late Glacial Maximum (see Burney et al., 2004) in order to assess if these are natural endemisms or induced by intense anthropogenic activity.

Although the deforestation rate decreased between 2013 and 2016, efficient reforestation should take place in critical areas inhabited by *B. williamsi* and *M. pauliani* to ensure the stabilization of abundance in numbers by increasing core habitat areas. A two-year survey of *M. pauliani* populations is currently taking place under the EDGE 2016 program “Distribution, abundance and gene flow of *Mantidactylus pauliani* in Ankaratra Special Reserve”, by Lovasoa Manuelle Sylviane Rakotozafy in Ankaratra Special Reserve. Seeing that *B. williamsi* abundance numbers are exceptionally low (IUCN, 2016e), it is of critical importance that a similar programme is set up concerning this species. The possibility to establish a similar work is currently under discussion (A. Crottini, pers. comm.)

I*n situ* recognition of the areas used for the economic exploitation of pine wood is essential in order to be able to map them and acknowledge them for a more efficient and sustainable planning and maintenance of the Reserve.

As a final remark, most of the recently established protected areas in Madagascar are incorporated in the Durban Vision (see Virah-Sawmy et al., 2014), which aims to maximize conservation but also rural development (IUCN, 2003). The Ankaratra Massif Reserve is integrated in the SAPM (Système d’Aires Protégées de Madagascar), which, in consistency with the Durban Vision, adopts three main objectives (see Gardner, 2009): 1) to conserve all Madagascar’s biodiversity; 2) to conserve Madagascar’s cultural heritage and 3) to encourage sustainable use for development and poverty alleviation. Thus, with this in mind, it is suggested that the portion of forest that has been stable throughout the last 30 years in Ankaratra becomes strictly prohibited for resource extraction, leaving the remaining area of the Reserve to be managed through sustainable agriculture and resource extraction.

Overall, this study brings to light past landscape dynamics and brings together information that will hopefully help improve management plans within the Ankaratra Massif Reserve. It also shows how land management by local associations appears to have had very positive results on forest management and should therefore be enforced. Similarly, for a positive and long-term outcome, it is preferable that sustainable exploitation measures are planned by conservationists together with the local population so that this endangered ecoregion is ensured for the future.

6. References

Achard, F., Blasco, F., 1990. Analysis of vegetation seasonal evolution and mapping of forest cover in West Africa with the use of NOAA AVHRR HRPT data. Photogramm. Eng. Remote Sens. 56, 1359–1365.

ACSAM, 2011. The Ankaratra Massif and its threatened amphibians. http://www.sahonagasy.org/conservation/ankaratra. [accessed 1st August 2017]

Akaike, H., 1974. A New Look at the Statistical Model Identification. IEEE Trans. Automat. Contr. 19, 716–723.

Alcaraz-Segura, D., Cabello, J., Paruelo, J., 2009. Baseline characterization of major Iberian vegetation types based on the NDVI dynamics. Plant Ecol. 202, 13–29.

Alcaraz, D., Paruelo, J., Cabello, J., 2006. Identification of current ecosystem functional types in the Iberian Peninsula. Glob. Ecol. Biogeogr. 15, 200–212.

Ali, J.R., Aitchison, J.C., 2008. Gondwana to Asia: Plate tectonics, paleogeography and the biological connectivity of the Indian sub-continent from the Middle Jurassic through latest Eocene (166-35 Ma). Earth-Science Rev. 88, 145–166.

Amphibian Survival Alliance, 2015. Ankaratra Massif Reserve. http://www.amphibians.org/explore/projects/ankaratra-massif/. [accessed 1st August 2017]

Anderson, D.R., Burnham, K.P., 2002. Avoiding Pitfalls when using Information-Theoretic Metohds. J. Wildl. Manage. 66, 912–918.

Andreone, F., 1994. The amphibians of Ranomafana rain forest, Madagascar - preliminary community analysis and conservation considerations. Oryx 28, 207–214.

Andreone, F., Bungard, M., Freeman, K., 2007. Threatened frogs of Madagascar. Cascine Vica-Rivoli (Torino).

Andreone, F., Cadle, J.E., Cox, N., Glaw, F., Nussbaum, R.A., Raxworthy, C.J., Stuart, S.N., Vallan, D., Vences, M., 2005. Species review of amphibian extinction risks in Madagascar: Conclusions from the global amphibian assessment. Conserv. Biol. 19, 1790–1802.

Andreone, F., Carpenter, A.I., Cox, N., Du Preez, L., Freeman, K., Furrer, S., Garcia, G., Glaw, F., Glos, J., Knox, D., Köhler, J., Mendelson, J.R., Mercurio, V., Mittermeier, R.A., Moore, R.D., Rabibisoa, N.H.C., Randriamahazo, H., Randrianasolo, H., Raminosoa, N.R., Ramilijaona, O.R., Raxworthy, C.J., Vallan, D., Vences, M., Vieites, D.R., Weldon, C., 2008. The challenge of conserving amphibian megadiversity in Madagascar. PLoS Biol. 6, 0943–0946.

Andreone, F., Crottini, A., Rabemananjara, F.C.E., Randrianirina, J.E., Razafindrabe, T., Tessa, G., 2014. Age structure, population estimate and Bd-status of two Critically Endangered frogs from the Ankaratra Massif (Madagascar), *Boophis williamsi* and *Mantidactylus pauliani* (Amphibia: Mantellidae). Scr. Herpetol. 17–29.

Andreone, F., Luiselli, L.M., 2003. Conservation priorities and potential threats influencing the hyper-diverse amphibians of Madagascar. Ital. J. Zool. 70, 53–63.

Andreone, F., Randriamahazo, H., 2008. Sahonagasy Action Plan. Conservation programs for the amphibians of Madagascar. Bogotá, Museo Regionale di Scienze Naturali, Conservation International and IUCN/SSC Amphibian Specialist Group, 96pp.

Andreone, F., Rosa, G.M., Noël, J., Vences, M., Crottini, A., Raxworthy, C.J., 2010. Living within fallen palm leaves: The discovery of an unknown *Blommersia* (Mantellidae: Anura) reveals a new reproductive strategy in the amphibians of Madagascar. Naturwissenschaften 97, 525–543.

Andreone, F., Vences, M., Glaw, F., Randrianirina, J.E., 2007. Remarkable records of amphibians and reptiles on Madagascar's central high plateau. Trop. Zool. 20, 19–39.

ASA & ASG, 2010. Ankaratra Massif: Home to Two Critically Endangered Amphibian Species and Numerous Other Threatened Species. http://www.amphibians.org/news/ankaratra-massif-home-two-critically-endangered-amphibian-species-numerous-other-threatened-species/. [accessed 4th September 2017]

Banks-Leite, C., Ewers, R.M., Metzger, J.P., 2012. Unraveling the drivers of community dissimilarity and species extinction in fragmented landscapes. Ecology 93, 2560–2569.

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., Hoogmoed, M.S., Leite, R.N., Lo-Man-Hung, N.F., Malcolm, J.R., Martins, M.B., Mestre, L.A.M., Miranda-Santos, R., Nunes-Gutjahr, A.L., Overal, W.L., Parry, L., Peters, S.L., Ribeiro-Junior, M.A., da Silva, M.N.F., da Silva Motta, C., Peres, C.A., 2007. Quantifying the biodiversity value of tropical primary, secondary, and plantation forests. Proc. Natl. Acad. Sci. U. S. A. 104, 18555–60.

Bletz, M.C., Rosa, G.M., Andreone, F., Courtois, E.A., Schmeller, D.S., Rabibisoa, N.H.C., Rabemananjara, F.C.E., Raharivololoniaina, L., Vences, M., Weldon, C., Edmonds, D., Raxworthy, C.J., Harris, R.N., Fisher, M.C., Crottini, A., 2015a. Widespread presence of the pathogenic fungus *Batrachochytrium dendrobatidis* in wild amphibian communities in Madagascar. Sci. Rep. 5, 1–10.

Bletz, M.C., Rosa, G.M., Andreone, F., Courtois, E.A., Schmeller, D.S., Rabibisoa, N.H.C., Rabemananjara, F.C.E., Raharivololoniaina, L., Vences, M., Weldon, C., Edmonds, D., Raxworthy, C.J., Harris, R.N., Fisher, M.C., Crottini, A., 2015b. Consistency of Published Results on the Pathogen *Batrachochytrium dendrobatidis* in Madagascar: Formal Coment on Kolby et al. Rapid Response to Evaluate the Presence of Amphibia Chytrid Fungus (*Batrachochytrium dendobatidis*) and Ranavirus in Wild Amphibian. PLoS One 10, 1–10.

Breiman, L., 2001. Random Forests. Mach. Learn. 45, 5–32.

Bruijnzeel, L.A., Proctor, J., 1995. Hydrology and Biogeochemistry of Tropical Montane Cloud Forests: What Do We Really Know?, in: Hamilton, L.S., Juvik, J.O., Skatena, F.N. (Eds.), Tropical Montane Cloud Forests. Ecological Studies (Analysis and Synthesis). Springer, New York, pp. 38–78. doi:10.1007/978-1-4612-2500-3

Bruijnzeel, L.A., Veneklaas, E.J., Ecology, S., Jan, N., 1998. Climatic Conditions and Tropical Montane Forest Productivity: The Fog Has Not Lifted Yet. Ecology 79, 3–9.

Buckley, L.B., Jetz, W., 2007. Environmental and historical constraints on global patterns of amphibian richness. Proc. R. Soc. Biol. Sci. 274, 1167–1173.

Burney, D.A., 1987. Late Holocene vegetational change in central Madagascar. Quat. Res. 28, 130–143.

Burney, D.A., 1996. Climate change and fire ecology as factors in the quaternary biogeography of Madagascar. Biogéographie de Madagascar 49–58.

Burney, D.A. 1997. Theories and facts regarding Holocene environmental change before and after human colonization. Natural change and human impact in Madagascar: 75-89.

Burney, D.A., Burney, L.P., Godfrey, L.R., Jungers, W.L., Goodman, S.M., Wright, H.T., Jull, A.J.T., 2004. A chronology for late prehistoric Madagascar. J. Hum. Evol. 47, 25–63.

Burnham, K.P., Anderson, D.R., Huyvaert, K.P., 2011. AIC model selection and multimodel inference in behavioral ecology: Some background, observations, and comparisons. Behav. Ecol. Sociobiol. 65, 23–35.

Cabello, J., Fernández, N., Alcaraz-Segura, D., Oyonarte, C., Piñeiro, G., Altesor, A., Delibes, M., Paruelo, J.M., 2012. The ecosystem functioning dimension in conservation: Insights from remote sensing. Biodivers. Conserv. 21, 3287–3305.

Cameron, A.C., Trivedi, P.K., 1990. Regression-based tests for overdispersion in the Poisson model. J. Econom. 46, 347–364.

Campbell, J.B., 1983. Mapping the Land: Aerial Imagery for Land Use Information. Resource Publications in Geography. Association of American Geographers, Washington DC.

Carey, C., Alexander, M.A, 2003. Climate change and amphibian declines: is there a link ? Divers. Distrib. 9, 111–121.

Crottini, A., Madsen, O., Poux, C., Strauss, A., Vieites, D.R., Vences, M., 2012. Vertebrate time-tree elucidates the biogeographic pattern of a major biotic change around the K-T boundary in Madagascar. Proc. Natl. Acad. Sci. 109, 5358–5363.

Crowley, H., 2000a. Montane Grasslands and Shrublands: Madagascar. https://www.worldwildlife.org/ecoregions/at1011 [accessed 26th July 2017]

Crowley, H., 2000b. Tropical and Subtropical Moist Broadleaf Forests. Southern Africa: Central Madagascar. https://www.worldwildlife.org/ecoregions/at0118 [accessed 26th July 2017]

Cutler, D.R., Edwards, T.C., Beard, K.H., Cutler, A., Hess, K.T., Gibson, J., Lawler, J.J., 2007. Random Forests for Classification in Ecology. Ecology 88, 2783–2792.

Dawson, J., Edmonds, D., Freeman, K., Iambana, B., Moore, M., Rakotoarisoa, J.C., 2014. Working Together to Develop Madagascar’s Amphibian Captive Breeding Capacity. FrogLog 22, 61–62.

DeFries, R.S., Townshend, J.R.G., 1994. NDVI-derived land cover classifications at a global scale. Int. J. Remote Sens. 15, 3567–3586.

Deutsch, C.A., Tewksbury, J.J., Huey, R.B., Sheldon, K.S., Ghalambor, C.K., Haak, D.C., Martin, P.R., 2008. Impacts of climate warming on terrestrial ectotherms across latitude Thermal Safety margin. Proc. Natl. Acad. Sci. 6, 6668–6672.

DeWalt, S.J., Maliakal, S.K., Denslow, J.S., 2003. Changes in vegetation structure and composition along a tropical forest chronosequence: Implications for wildlife. For. Ecol. Manage. 182, 139–151.

Dewar, R.E., Richard, A.F., 2007. Evolution in the hypervariable environment of Madagascar. Proc. Natl. Acad. Sci. 104, 13723–13727.

Dewar, R.E., Wright, H.T., 1993. The culture history of Madagascar. J. World Prehistory 7, 417–466.

Didham, R.K., Lawton, J.H., 1999. Edge Structure Determines the Magnitude of Changes in Microclimate and Vegetation Structure in Tropical Forest Fragments. Biotropica 31, 17–30.

Duro, D.C., Coops, N.C., Wulder, M.A., Han, T., 2007. Development of a large area biodiversity monitoring system driven by remote sensing. Prog. Phys. Geogr. 31, 235–260.

Ecosia Blog, 2017. Madagascar: Reforesting the world’s oldest island. http://blog.ecosia.org/post/156090848455/madagascar-holds-a-vast-and-rich-treasure-of. [accessed 1st June 2017]

Elsen, P.R., Tingley, M.W., 2015. Global mountain topography and the fate of montane species under climate change. Nat. Clim. Chang. 5, 772–776.

EXELIS, 2009. ENVI- IDL 4.7, USA

Ewers, R.M., Didham, R.K., 2006. Confounding factors in the detection of species responses to habitat fragmentation. Biol. Rev. Camb. Philos. Soc. 81, 117–142.

Feder, M.E., Burggren, W.W., 1992. Environmental Physiology of the Amphibians. University of Chicago Press.

Gade, D.W., 1996. Deforestation and Its Effects in Highland Madagascar. Mt. Res. Dev. 16, 101–116.

Ganzhorn, J.U., Lowry, P.P., Schatz, G.E., Sommer, S., 2001. The biodiversity of

Madagascar: One of the world's hottest hotspots on its way out. Oryx 35, 346–348.

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 Nat. 2, 2–29.

Gardner, T.A., Barlow, J., Peres, C.A., 2007. Paradox, presumption and pitfalls in conservation biology: The importance of habitat change for amphibians and reptiles. Biol. Conserv. 138, 166–179.

Gascon, C., Lovejoy, T.E., Bierregaard, R.O., Malcolm, J.R., Stouffer, P.C., Vasconcelos, H.L., Laurance, W.F., Zimmerman, B., Tocher, M., Borges, S., 1999. Matrix habitat and species richness in tropical forest remnants. Biol. Conserv. 91, 223–229.

Glaw, F., Vences, M., 2003. Introduction to Amphibians, in: Goodman, S.M., Benstead, J.P. (Eds.), The Natural History of Madagascar. The University of Chicago Press, pp. 883–898.

Glaw, F., Vences, M., 2007. A Fieldguide to the Amphibians and Reptiles of Madagascar. 3rd Edition, Cologne, 496 pp

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

Goodman, S.M., Jungers, W.L., 2014. Extinct Madagascar. The University of Chicago Press.

Guarino, F.M., Andreone, F., Angelini, F., 1998. Growth and Longevity by Skeletochronological Analysis in *Mantidactylus microtympanun*, a Rain-Forest Anuran from Southern Madagascar. Copeia 1, 194–198.

Hardwick, S.R., Toumi, R., Pfeifer, M., Turner, E.C., Nilus, R., Ewers, R.M., 2015. The relationship between leaf area index and microclimate in tropical forest and oil palm plantation: Forest disturbance drives changes in microclimate. Agric. For. Meteorol. 201, 187–195.

Henle, K., Davies, K.F., Kleyer, M., Margules, C., Settele, J., 2004. Predictors of species sensitivity to fragmentation. Biodivers. Conserv. 13, 207–251.

Hirschfeld, M., Blackburn, D.C., Doherty-Bone, T.M., Gonwouo, L.N., Ghose, S., Rödel, M.-O., 2016. Dramatic Declines of Montane Frogs in a Central African Biodiversity

Hotspot. PLoS One 11, e0155129.

Hocking, A., Babbitt, K., 2014. Amphibian Contributions to Ecosystem Services. Herpetol. Conserv. Biol. 9, 1–17.

Hoffmann, M., Hilton-Taylor, C., Angulo, A., Böhm, M., Brooks, T.M., Butchart, S.H.M., Carpenter, K.E., Chanson, J., Collen, B., Cox, N.A., Darwall, W.R.T., Dulvy, N.K., Harrison, L.R., Katariya, V., Pollock, C.M., Quader, S., Richman, N.I., Rodrigues, A.S.L., Tognelli, M.F., Vié, J.-C., Aguiar, J.M., Allen, D.J., Allen, G.R., Amori, G., Ananjeva, N.B., Andreone, F., Andrew, P., Ortiz, A.L.A., Baillie, J.E.M., Baldi, R., Bell, B.D., Biju, S.D., Bird, J.P., Black-Decima, P., Blanc, J.J., Bolaños, F., Bolivar-G, W., Burfield, I.J., Burton, J.A., Capper, D.R., Castro, F., Catullo, G., Cavanagh, R.D., Channing, A., Chao, N.L., Chenery, A.M., Chiozza, F., Clausnitzer, V., Collar, N.J., Collett, L.C., Collette, B.B., Fernandez, C.F.C., Craig, M.T., Crosby, M.J., Cumberlidge, N., Cuttelod, A., Derocher, A.E., Diesmos, A.C., Donaldson, J.S., Duckworth, J.W., Dutson, G., Dutta, S.K., Emslie, R.H., Farjon, A., Fowler, S., Freyhof, J., Garshelis, D.L., Gerlach, J., Gower, D.J., Grant, T.D., Hammerson, G.A., Harris, R.B., Heaney, L.R., Hedges, S.B., Hero, J.-M., Hughes, B., Hussain, S.A., Icochea M, J., Inger, R.F., Ishii, N., Iskandar, D.T., Jenkins, R.K.B., Kaneko, Y., Kottelat, M., Kovacs, K.M., Kuzmin, S.L., La Marca, E., Lamoreux, J.F., Lau, M.W.N., Lavilla, E.O., Leus, K., Lewison, R.L., Lichtenstein, G., Livingstone, S.R., Lukoschek, V., Mallon, D.P., McGowan, P.J.K., McIvor, A., Moehlman, P.D., Molur, S., Alonso, A.M., Musick, J.A., Nowell, K., Nussbaum, R.A., Olech, W., Orlov, N.L., Papenfuss, T.J., Parra-Olea, G., Perrin, W.F., Polidoro, B.A., Pourkazemi, M., Racey, P.A., Ragle, J.S., Ram, M., Rathbun, G., Reynolds, R.P., Rhodin, A.G.J., Richards, S.J., Rodríguez, L.O., Ron, S.R., Rondinini, C., Rylands, A.B., Mitcheson, Y.S., Sanciangco, J.C., Sanders, K.L., Santos-Barrera, G., Schipper, J., Self-Sullivan, C., Shi, Y., Shoemaker, A., Short, F.T., Sillero-Zubiri, C., Silvano, D.L., Smith, K.G., Smith, A.T., Snoeks, J., Stattersfield, A.J., Symes, A.J., Taber, A.B., Talukdar, B.K., Temple, H.J., Timmins, R., Tobias, J.A., Tsytsulina, K., Tweddle, D., Ubeda, C., Valenti, S. V, van Dijk, P.P., Veiga, L.M., Veloso, A., Wege, D.C., Wilkinson, M., Williamson, E.A., Xie, F., Young, B.E., Akçakaya, H.R., Bennun, L., Blackburn, T.M., Boitani, L., Dublin, H.T., Fonseca, G.A.B., Gascon, C., Lacher Jr., T.E., Mace, G.M., Mainka, S.A., McNeely, J.A., Mittermeier, R.A., Reid, G.M., Rodríguez, J.P., Rosenberg, A.A., Samways, M.J., Smart, J., Stein, B.A., Stuart, S.N., 2010. The impact of conservation on the status of the world’s vertebrates. Science (80-.). 330, 1503–1509.

Huete, A.R., 1988. A soil-adjusted vegetation index (SAVI). Remote Sens. Environ. 25, 295–309.

Hutchison, Victor H., Dupré, R. K., 1992. Thermoregulation, in: Feder, M.E., Burggren, W.W. (Eds.), Environmental Physiology of the Amphibians. The University of Chicago Press, pp. 206-249.

IUCN 2003. The Durban Action Plan. One of the outputs to the 2003, 5th IUCN World Parks Congress held in Durban, South Africa.

IUCN SSC Amphibian Specialist Group. 2016a. *Blommersia kely*. The IUCN Red List of Threatened Species 2016: e.T57491A84172784. Downloaded on 08 June 2017.

IUCN SSC Amphibian Specialist Group. 2016b. *Boophis ankaratra*. The IUCN Red List of Threatened Species 2016: e.T57391A84161674. Downloaded on 08 June 2017.

IUCN SSC Amphibian Specialist Group. 2016c. *Boophis goudotii*. The IUCN Red List of Threatened Species 2016: e.T49542645A49542679. Downloaded on 08 June 2017.

IUCN SSC Amphibian Specialist Group. 2016d. *Boophis microtympanum*. The IUCN Red List of Threatened Species 2016: e.T57415A84163802. Downloaded on 08 June 2017.

IUCN SSC Amphibian Specialist Group. 2016e. *Boophis williamsi*. The IUCN Red List of Threatened Species 2016: e.T57435A84165560. Downloaded on 14 July 2016.

IUCN SSC Amphibian Specialist Group. 2017a. Mantidactylus alutus. The IUCN Red List of Threatened Species 2017: e.T57456A84168725. Downloaded on 08 June 2017.

IUCN SSC Amphibian Specialist Group. 2016f. *Mantidactylus brevipalmatus*. The IUCN Red List of Threatened Species 2016: e.T57469A84170158. Downloaded on 08 June 2017.

IUCN SSC Amphibian Specialist Group. 2017b. Mantidactylus curtus. The IUCN Red List of Threatened Species 2017: e.T57474A84170807. Downloaded on 08 June 2017.

IUCN SSC Amphibian Specialist Group. 2016g. *Mantidactylus mocquardi.* The IUCN Red List of Threatened Species 2016: e.T57506A84174666. Downloaded on 08

June 2017.

IUCN SSC Amphibian Specialist Group. 2016h. *Mantidactylus pauliani.* The IUCN Red List of Threatened Species 2016: e.T57509A84175039. Downloaded on 14 July 2016.

IUCN SSC Amphibian Specialist Group. 2016i. *Ptychadena mascareniensis*. The IUCN Red List of Threatened Species 2016: e.T76317565A79825430. Downloaded on 08 June 2017.

Jenkins MD (1987) Madagascar, An Environmental Profile. International Union for Conservation of Nature and Natural Resources, Gland.

Jensen JR (2007) Remote sensing of the environment: an earth resource perspective, 2nd edn. Prentice Hall, Upper Saddle River, NJ

Keenan, R.J., Reams, G.A., Achard, F., de Freitas, J. V., Grainger, A., Lindquist, E., 2015. Dynamics of global forest area: Results from the FAO Global Forest Resources Assessment 2015. For. Ecol. Manage. 352, 9–20.

Kindt, R., Coe, R., 2005. Tree diversity analysis. A manual and software for common statistical methods for ecological and biodiversity studies. World Agroforestry Centre (ICRAF), Nairobi, Kenya.

Kolby, J.E., 2014. Presence of the amphibian chytrid fungus *Batrachochytrium dendrobatidis* in native amphibians exported from Madagascar. PLoS One 9.

Kolby, J.E., Skerratt, L.F., 2015. Amphibian chytrid fungus in Madagascar neither shows widespread presence nor signs of certain establishment. PLoS One 10, 1–6.

Kolby, J.E., Smith, K.M., Ramirez, S.D., Rabemananjara, F., Pessier, A.P., Brunner, J.L., Goldberg, C.S., Berger, L., Skerratt, L.F., 2015. Rapid response to evaluate the presence of amphibian chytrid fungus (*Batrachochytrium dendrobatidis*) and ranavirus in wild amphibian populations in Madagascar. PLoS One 10, 1–21.

Kusky, T.M., Toraman, E., Raharimahefa, T., 2007. The Great Rift Valley of Madagascar: an extension of the Africa–Somali diffusive plate boundary? Gondwana Research 11.4, 577-579.

Langaha, 2010. Evaluation rapide des états des populations des deux espèces en danger critique d'extinction d'Ankaratra: *Boophis williamsi* et *Mantidactylus pauliani*. Conservation International. 25 pp

Lehtinen, R.M., Ramanamanjato, J., Raveloarison, J.G., 2003. Edge effects and extinction proneness in a herpetofauna from Madagascar. Biodivers. Conserv. 12, 1357–1370.

Leonard, M.D., 1993. Notes on the Giant Toad, *Bufo marinus* (L.), in Puerto Rico. J. Econ. Entomol. 26, 67–72.

Liang, S., 2000. Narrowband to broadband conversions of land surface albedo I Algorithms. Remote Sens. Environ. 76, 213–238.

Martel, A., Spitzen-van der Sluijs, A., Blooi, M., Bert, W., Ducatelle, R., Fisher, M.C., Woeltjes, A., Bosman, W., Chiers, K., Bossuyt, F., Pasmans, F., 2013. *Batrachochytrium salamandrivorans* sp. nov. causes lethal chytridiomycosis in amphibians. Proc. Natl. Acad. Sci. U. S. A. 110, 15325–9.

McConnell, W., Viña, A., Kull, C., Batko, C., 2015. Forest Transition in Madagascar's Highlands: Initial Evidence and Implications. Land 4, 1155–1181.

McCullagh, P., Nelder, J.A., 1989. Generalized Linear Models, 2nd Edition. ed. Chapman and Hall.

Moore, R.D., Church, D.R., 2008. Implementing the Amphibian Conservation Action Plan. Int. Zoo Yearb. 42, 15–23.

Moraes, J.M., Pellegrino, G.Q., Ballester, M. V., Martinelli, L.A., Victoria, R.L., Krusche, A. V., 1998. Trends in hydrological parameters of a southern Brazilian watershed and its relation to human induced changes. Water Resour. Manag. 12, 295–311.

Muñoz-Villers, L.E., López-Blanco, J., 2008. Land use/cover changes using Landsat TM/ETM images in a tropical and biodiverse mountainous area of central-eastern Mexico. Int. J. Remote Sens. 29, 71–93.

Myers, N., 1993. Tropical Forests: The Main Deforestation Fronts. Environ. Conserv. 20, 9.

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

Pearson, R.G., Stanton, J.C., Shoemaker, K.T., Aiello-Lammens, M.E., Ersts, P.J., Horning, N., Fordham, D.A., Raxworthy, C.J., Ryu, H.Y., McNees, J., Akçakaya, H.R., 2014. Life history and spatial traits predict extinction risk due to climate change. Nat. Clim. Chang. 4, 217–221.

Perl, R.G.B., Nagy, Z.T., Sonet, G., Glaw, F., Wollenberg, K.C., Vences, M., 2014. DNA barcoding Madagascar's amphibian fauna. Amphibia-Reptilia 35, 197–206.

Pettorelli, N., 2013. The normalized difference vegetation index. Oxford University Press.

Pettorelli, N., Laurance, W.F., O'Brien, T.G., Wegmann, M., Nagendra, H., Turner, W., 2014a. Satellite remote sensing for applied ecologists: Opportunities and challenges. J. Appl. Ecol. 51, 839–848.

Pettorelli, N., Ryan, S., Mueller, T., Bunnefeld, N., Jedrzejewska, B., Lima, M., Kausrud, K., 2011. The Normalized Difference Vegetation Index (NDVI): Unforeseen successes in animal ecology. Clim. Res. 46, 15–27.

Pettorelli, N., Safi, K., Turner, W., 2014b. Satellite remote sensing, biodiversity research and conservation of the future. Philos. Trans. R. Soc. B Biol. Sci. 369, 20130190–20130190.

Pettorelli, N., Vik, J.O., Mysterud, A., Gaillard, J.M., Tucker, C.J., Stenseth, N.C., 2005. Using the satellite-derived NDVI to assess ecological responses to environmental change. Trends Ecol. Evol. 20, 503–510.

Puyravaud, J.P., 2003. Standardizing the calculation of the annual rate of deforestation. For. Ecol. Manage. 177, 593–596.

R Core Team, 2017. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL https://www.R-project.org/.

Rabemananjara, F. C. E., Randrianirina, J., Randriambahiniarime, M., Randrianasolo, T., Razafindrabe, J., Andreone, F., and Rabibisoa, N. 2011. Brief note on the most threatened Amphibian species from Madagascar: *Boophis williamsi* (CR) and *Mantidactylus pauliani* (CR) in Ankaratra massif, Madagascar. FrogLog 97: 38.

Rahantaliosa, H., Rakotomalala, M., Rabemananjara, F.C.E., Rahantamalala, J., Rabibisoa, N.H., 2011. Ankaratra massif: Highland focal amphibian site for conservation. FrogLog 97, 36–37.

Rainforest Trust, 2016. A Sanctuary for Ankaratra's Amphibians. https://www.rainforesttrust.org/project/creating-a-sanctuary-for-ankaratras-amphibians/. [accessed 17th October 2016]

Rakotozafy, 2017. Conducting Research in The Misty Mountain Cold Forest of

Madagascar! http://www.edgeofexistence.org/edgeblog/?p=9329. [accessed 28th August 2017]

Randrianja, S., Ellis, S., 2009. Madagascar: a short history. HURST Publishers, London.

Rasmussen, H.B., Wittemyer, G., Douglas-Hamilton, I., 2006. Predicting time-specific changes in demographic processes using remote-sensing data. J. Appl. Ecol. 43, 366–376.

Raxworthy, C.J., Nussbaum, R.A., 1996. Montane Amphibian and Reptile Communities in Madagascar. Conserv. Biol. 10, 750–756.

Raxworthy, C.J., Pearson, R.G., Rabibisoa, N., Rakotondrazafy, A.M., Ramanamanjato, J.B., Raselimanana, A.P., Wu, S., Nussbaum, R.A., Stone, D.A., 2008. Extinction vulnerability of tropical montane endemism from warming and upslope displacement: A preliminary appraisal for the highest massif in Madagascar. Glob. Chang. Biol. 14, 1703–1720.

Reed, B.C., Brown, J.F., VanderZee, D., Loveland, T.R., Merchant, J.W., Ohlen, D.O., 1994. Measuring phenological variability from satellite imagery. J. Veg. Sci. 5, 703–714.

Rocchini, D., Leutner, B., Wegmann, M., 2016. From Spectral to Ecological Information, in: Wegmann, M., Leutner, B., Dech, S. (Eds.), Remote Sensing and GIS for Ecologists: Using Open Source Software. Pelagic Publishing, UK, Exeter, pp. 150–165.

Rome, L.C., Don Stevens, E., John-Alder, H.B., 1992. The influence of temperature and thermal acclimation on physiological function, in: Feder, M.E., Burggren, W.W. (Eds.), Environmental Physiology of the Amphibians. The University of Chicago Press, pp. 183-205.

Rosa, G.M., Andreone, F., Crottini, A., Hauswaldt, J.S., Noël, J., Rabibisoa, N.H., Randriambahiniarime, M.O., Rebelo, R., Raxworthy, C.J., 2012. The amphibians of the relict Betampona low-elevation rainforest, eastern Madagascar: An application of the integrative taxonomy approach to biodiversity assessments. Biodivers. Conserv. 21, 1531–1559.

Rouse J.W., Haas R.H., Schell J.A., Deering D.W. 1974. Monitoring vegetation systems in the Great Plains with ERTS. Proc Third Earth Resources Technology Satellite-1 Symp, December 10–15 1974, Greenbelt, MD, 3:301–317, NASA, Washington, D.C.

Samonds, K.E., Godfrey, L.R., Ali, J.R., Goodman, S.M., Vences, M., Sutherland, M.R., Irwin, M.T., Krause, D.W., 2012. Spatial and temporal arrival patterns of Madagascar’s vertebrate fauna explained by distance, ocean currents, and ancestor type. Proc. Natl. Acad. Sci. U. S. A. 109, 5352–7.

Scherz, M. D., 2017. Spotlight on Madagascar. Froglog 25, 11.

Schneider-Maunoury, L., Lefebvre, V., Ewers, R.M., Medina-Rangel, G.F., Peres, C.A., Somarriba, E., Urbina-Cardona, N., Pfeifer, M., 2016. Abundance signals of amphibians and reptiles indicate strong edge effects in Neotropical fragmented forest landscapes. Biol. Conserv. 200, 207–215.

Schott, J.R., Salvaggio, C., Volchok, W.J., 1988. Radiometric scene normalization using pseudoinvariant features. Remote Sens. Environ. 26, 1–16.

Semlitsch, R.D., Todd, B.D., Blomquist, S.M., Calhoun, A.J.K., Gibbons, J.W., Gibbs, J.P., Graeter, G.J., Harper, E.B., Hocking, D.J., Hunter, M.L., Patrick, D. a., Rittenhouse, T. a. G., Rothermel, B.B., 2009. Effects of Timber Harvest on Amphibian Populations: Understanding Mechanisms from Forest Experiments. Bioscience 59, 853–862.

Skelly, D.K., Halverson, M.A., Freidenburg, L.K., Urban, M.C., 2005. Canopy closure and amphibian diversity in forested wetlands. Wetl. Ecol. Manag. 13, 261–268.

Skerratt, L.F., Berger, L., Speare, R., Cashins, S., McDonald, K.R., Phillott, A.D., Hines, H.B., Kenyon, N., 2007. Spread of chytridiomycosis has caused the rapid global decline and extinction of frogs. Ecohealth 4, 125–134.

Smith, R.B., 2010. The heat budget of the earth’s surface deduced from space. Yale University Center for Earth Observation: New Haven, CT, USA

Townsley, J., Gaulke, P., Ingram, D., 2001. USDA Forest Service Mission, Fianarantsoa, Madagascar In Support of the Landscape Development Interventions Project. Section 3 : Economic Analysis Of Pine Plantation Management in Haute Matsiatra.

Trimble, M.J., van Aarde, R.J., 2014. Amphibian and reptile communities and functional groups over a land-use gradient in a coastal tropical forest landscape of high richness and endemicity. Anim. Conserv. 17, 441–453.

Tyler, M.J., Wassersug, R., Smith, B., 2007. How frogs and humans interact:

Influences beyond habitat destruction, epidemics and global warming. Appl. Herpetol. 4, 1–18.

Vågen, T.-G., 2006. Remote sensing of complex land use change trajectories-a case study from the highlands of Madagascar. Agric. Ecosyst. Environ. 115, 219–228.

Vågen, T.-G., Shepherd, K.D., Walsh, M.G., 2006. Sensing landscape level change in soil fertility following deforestation and conversion in the highlands of Madagascar using Vis-NIR spectroscopy. Geoderma 133, 281–294.

Vallan, D., 2002. Effects of anthropogenic environmental changes on amphibian diversity in the rain forests of eastern Madagascar. J. Trop. Ecol. 18, 725–742.

Vallan, D., 2000. Influence of forest fragmentation on amphibian diversity in the nature reserve of Ambohitantely, highland Madagascar. Biol. Conserv. 96, 31–43.

Valentini R., Baldocchi D.D., Tenhunen J.D., 1999. Ecological controls on land-surface atmospheric interactions. In: Tenhunen JD, Kabat P (eds) Integrating hydrology, ecosystem dynamics and biogeochemistry in complex landscapes. Wiley, Berlin, pp 105–11

Vences, M., Andreone, F., Glaw, F., Raminosoa, N., Randrianirina, J.E., Vieites, D.R., 2002. Amphibians and reptiles of the Ankaratra Massif: Reproductive diversity, biogeography and conservation of a montane fauna in Madagascar. Ital. J. Zool. 69, 263–284.

Vences, M., Brown, J.L., Lathrop, A., Rosa, G.M., Cameron, A., Crottini, A., Dolch, R., Edmonds, D., Freeman, K.L.M., Glaw, F., Grismer, L.L., Litvinchuk, S., Milne, M.G., Moore, M., Solofo, J.F., Noël, J., Nguyen, T.Q., Ohler, A., Randrianantoandro, C., Raselimanana, A.P., van Leeuwen, P., Wogan, G.O.U., Ziegler, T., Andreone, F., Murphy, R.W., 2017. Tracing a toad invasion: lack of mitochondrial DNA variation, haplotype origins, and potential distribution of introduced Duttaphrynus melanostictus in Madagascar. Amphibia-Reptilia 38, 197–207.

Vences, M., Vieites, D.R., Glaw, F., Brinkmann, H., Kosuch, J., Veith, M., Meyer, A., 2003. Multiple overseas dispersal in amphibians. Proc. R. Soc. London B 270, 2435–2442.

Vences, M., Wollenberg, K.C., Vieites, D.R., Lees, D.C., 2009. Madagascar as a model region of species diversification. Trends Ecol. Evol. 24, 456–465.

Vieites, D.R., Wollenberg, K.C., Andreone, F., Köhler, J., Glaw, F., Vences, M., 2009. Vast underestimation of Madagascar's biodiversity evidenced by an integrative amphibian inventory. Proc. Natl. Acad. Sci. U. S. A. 106, 8267–8272.

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

Vorontsova, M.S., Besnard, G., Malakasi, P., Moat, J., Clayton, W.D., Ficinski, P., Savva, G.M., Nanjarisoa, O.P., Razanatsoa, J., Randriatsara, F.O., Kimeu, J.M., Luke, W.R.Q., Kayombo, C., Linder, H.P., 2016. Madagascar's grasses and grasslands: anthropogenic or natural? Procedings R. Soc. B 283, 2015–2262.

Voyles, J., Young, S., Berger, L., Campbell, C., Voyles, W.F., Dinudom, A., Cook, D., Webb, R., Alford, R.A., Skerratt, L.F., Speare, R., 2009. Pathogenesis of Chytridiomycosis, a Cause of Catastrophic Amphibian Declines. Science (80-.). 326, 582–586.

Wegmann, M., Leutner, B., 2016. Introduction to Remote Sensing and GIS, in: Wegmann, M., Leutner, B., Dech, S. (Eds.), Remote Sensing and GIS for Ecologists: Using Open Source Software. Pelagic Publishing, UK, Exeter, pp. 22–39.

Weldon, C., Crottini, A., Bollen, A., Rabemananjara, F.C.E., Copsey, J., Garcia, G., Andreone, F., 2013. Pre-emptive national monitoring plan for detecting the amphibian chytrid fungus in Madagascar. Ecohealth 10, 234–240.

White, F., 1983. The vegetation of Africa. United Nations Educational, Social and Cultural Organization. Paris

Wiegand, T., Snyman, H.A., Kellner, K., Paruelo, J.M., 2004. Do grasslands have a memory: Modeling phytomass productionof a semiarid South African grassland. Ecosystems 7, 243–258.

Williams-Linera, G., Domínguez-Gastelú, V., García-Zurita, M.E., 1998. Microenvironment and Floristics of Different Edges in a Fragmented Tropical Rainforest. Conserv. Biol. 12, 1091–1102.

Wilmé, L., Goodman, S.M., Ganzhorn, J.U., 2006. Biogeographic Evolution of Madagascar's Microendemic Biota. Science (80-.). 312, 1063–1065.

Zimkus, B.M., Lawson, L.P., Barej, M.F., Barratt, C.D., Channing, A., Dash, K.M., Dehling, J.M., Du Preez, L., Gehring, P.S., Greenbaum, E., Gvoždík, V., Harvey, J., Kielgast, J., Kusamba, C., Nagy, Z.T., Pabijan, M., Penner, J., Rödel, M.O., Vences, M., Lötters, S., 2017. Leapfrogging into new territory: How Mascarene ridged frogs diversified across Africa and Madagascar to maintain their ecological niche. Mol. Phylogenet. Evol. 106, 254–269.

Zuur, A.F., Ieno, E.N., Elphick, C.S., 2010. A protocol for data exploration to avoid common statistical problems. Methods Ecol. Evol. 1, 3–14.

6.2 Python Packages Used

future: Schofield, Ed., 2016. Python Charmers Pty Ltd, Australia, and Python Charmers Pte Ltd, Singapore. http://python-future.org/

Matplotlib: John D. Hunter, 2007. Matplotlib: A 2D Graphics Environment, Computing in Science & Engineering, 9, 90-95, DOI:10.1109/MCSE.2007.55 (publisher link)

NumPy: Stéfan van der Walt, S. Chris Colbert and Gaël Varoquaux., 2011. The NumPy Array: A Structure for Efficient Numerical Computation, Computing in Science & Engineering, 13, 22-30, DOI:10.1109/MCSE.2011.37 (publisher link)

Pandas: Wes McKinney, 2010. Data Structures for Statistical Computing in Python, Proceedings of the 9th Python in Science Conference, 51-56

skicit-learn: Fabian Pedregosa, Gaël Varoquaux, Alexandre Gramfort, Vincent Michel, Bertrand Thirion, Olivier Grisel, Mathieu Blondel, Peter Prettenhofer, Ron Weiss, Vincent Dubourg, Jake Vanderplas, Alexandre Passos, David Cournapeau, Matthieu Brucher, Matthieu Perrot, Édouard Duchesnay, 2011. Scikit-learn: Machine Learning in Python, Journal of Machine Learning Research, 12, 2825-2830

6.3 R Packages Used

AER: Christian Kleiber and Achim Zeileis (2008). Applied Econometrics with R. New York: Springer-Verlag. ISBN 978-0-387-77316-2. https://CRAN.R-project.org/package=AER

BiodiversityR: Kindt, R. & Coe, R. (2005) Tree diversity analysis. A manual and software for common statistical methods for ecological and biodiversity studies. World Agroforestry Centre (ICRAF), Nairobi. ISBN 92-9059-179-X.

car: John Fox and Sanford Weisberg (2011). An {R} Companion to Applied Regression, Second Edition. Thousand Oaks CA: Sage. http://socserv.socsci.mcmaster.ca/jfox/Books/Companion

Hmisc: Frank E Harrell Jr, with contributions from Charles Dupont and many others. (2017). Hmisc: Harrell Miscellaneous. R package version 4.0-3. https://CRAN.R-project.org/package=Hmisc

lme4: Douglas Bates, Martin Maechler, Ben Bolker, Steve Walker (2015). Fitting Linear Mixed-Effects Models Using lme4. Journal of Statistical Software, 67(1), 1-48. doi:10.18637/jss.v067.i01.

MASS: Venables, W. N. & Ripley, B. D. (2002) Modern Applied Statistics with S. Fourth Edition. Springer, New York. ISBN 0-387-95457-0

modEvA: Barbosa A.M., Brown J.A., Jimenez-Valverde A. and Real R. (2016). modEvA: Model Evaluation and Analysis. R package version 1.3.2.

https://CRAN.R-project.org/package=modEvA

MuMIn: Kamil Barton (2016). MuMIn: Multi-Model Inference. R package version 1.15.6. https://CRAN.R-project.org/package=MuMIn

rcompanion: Salvatore Mangiafico (2017). rcompanion: Functions to Support Extension Education Program Evaluation. R package version 1.5.7. https://CRAN.R-project.org/package=rcompanion

vegan: Jari Oksanen, F. Guillaume Blanchet, Michael Friendly, Roeland Kindt, Pierre Legendre, Dan McGlinn, Peter R. Minchin, R. B. O'Hara, Gavin L. Simpson, Peter Solymos, M. Henry H. Stevens, Eduard Szoecs and Helene Wagner (2017). vegan: Community Ecology Package. R package version 2.4-3. https://CRAN.R-project.org/package=vegan

6.4 Google Earth Images

“Ankaratra”. 19°20’58.23’’S and 47°17’47.93’’E. **Google Earth.** October 23, 2003.

“Ankaratra”. 19°20’58.23’’S and 47°17’47.93’’E. **Google Earth.** May 5, 2013.

7. Appendix

I – Field Data

Table 7.1 – The transects of the survey of the critically endangered species *Boophis williamsi* and *Mantidactylus pauliani* and all the associated amphibians between June 2011 and June 2013.

Transects	Coordinates	Altitudes	Transect Length	Habitat type	Observations
Ambohimirandrana	S 19°20.450' E 47°16.461' S 19°20.387' E 47°16.413'	2250 - 2314m	170m	Narrow brook in an open and sloppy of 20 to 45 degrees savannah area.	Degraded zone from the fire burning in October 2010. The area has been recovered again for the last 6 months by herbaceous plants.
Tavolotara	S 19°20'45''.5 E 47°16'45''.2	2000 - 2020m	100m	Brook on average slope in a forest environment.	Intact environment but sometimes polluted by the waste of the damages upstream.
Tsimiaramianadahy	S 19°20.029' E 47°15.832'	2364 - 2410m	100m	Moderated flowing brook in the opened savannah area.	High mountain permanent transect for *Boophis williamsi* and *Mantidactylus pauliani.*
Analafohy	S 19°20'39.0'' E 47°16'30.3''	2082m		Moderated flowing brook in a gallery forest.	New survey site for the altitude 2080m. This transect is chosen to facilitating the survey path to save time in the future studies. We will replace the Maharavana transect into this one for all future surveys.
Ambitsika	S 19°20'09.0'' E 47°16'51.5''	2208m		Moderated flowing brook near a crest with degraded exotic forest.	This transect was chosen for the survey of the impact of the degradation of the exotic forest on the Amphibian species, especially the critically endangered *Mantidactylus pauliani* and *Boophis williamsi.*
Anosiarivo	S 19°20'30.4'' E 47°18'14.0''	2062m		Moderated flowing brook in a degraded natural forest.	This transect has been surveyed before 2009 with presence of *Mantidacylus pauliani*. On 2010, no individual has been observed. For this period of habitat restoration made by VIF Association, we chose the transect to evaluate the capacity of the population to spread again into this area.

Table 7.2 – Survey results.

Year	Season	Sampling Site	Nº of species found	Nº of specimens found	N (adults + juveniles)	Total nº tadpoles found	*B. williamsi* (adults + juveniles)	*M. pauliani* (adults + juveniles)
2011	Dry (June, August)	Ambohimirandrana	6	377	225	152	5	167
2011	Humid (November, December)	Ambohimirandrana	6	262	255	7	8	71
2012	Dry Begining (May)	Ambohimirandrana	6	117	117	1	3	68
2012	Dry End (September)	Ambohimirandrana	6	64	61	1	2	35
2013	Humid (January, February)	Ambohimirandrana	5	57	36	21	1	11
2013	Dry (May, June)	Ambohimirandrana	5	243	184	59	18	128
2011	Dry (June, August)	Tavolotara	7	627	522	105	10	503
2011	Humid (November, December)	Tavolotara	8	877	851	26	4	792
2012	Dry Begining (May)	Tavolotara	4	82	79	3	0	75
2012	Dry End (September)	Tavolotara	2	269	266	1	2	266
2013	Humid (January, February)	Tavolotara	8	111	71	40	1	41
2013	Dry (May, June)	Tavolotara	6	288	236	49	4	229
2011	Dry (June, August)	Tsimiaramianadahy	7	342	206	136	15	44
2011	Humid (November, December)	Tsimiaramianadahy	7	178	169	9	6	21
2012	Dry Begining (May)	Tsimiaramianadahy	4	38	36	2	0	10
2012	Dry End (September)	Tsimiaramianadahy	8	112	105	7	0	6
2013	Humid (January, February)	Tsimiaramianadahy	6	28	8	20	0	3
2013	Dry (May, June)	Tsimiaramianadahy	8	146	100	38	5	40
2011	Dry (June, August)	Analafohy	-	-		-	-	-
2011	Humid (November, December)	Analafohy	-	-		-	-	-
2012	Dry Begining (May)	Analafohy	-	-		-	-	-
2012	Dry End (September)	Analafohy	8	418	389	29	0	372
2013	Humid (January, February)	Analafohy	3	45	34	11	0	32
2013	Dry (May, June)	Analafohy	5	356	301	55	11	282
2011	Dry (June, August)	Ambitsika	-	-		-	-	-
2011	Humid (November, December)	Ambitsika	-	-		-	-	-
2012	Dry Begining (May)	Ambitsika	-	-		-	-	-
2012	Dry End (September)	Ambitsika	3	33	33	0	0	11
2013	Humid (January, February)	Ambitsika	5	18	7	11	0	0
2013	Dry (May, June)	Ambitsika	3	80	59	21	0	54
2011	Dry (June, August)	Anosiarivo	-	-		-	-	-
2011	Humid (November, December)	Anosiarivo	-	-		-	-	-
2012	Dry Begining (May)	Anosiarivo	-	-		-	-	-
2012	Dry End (September)	Anosiarivo	-	-		-	-	-
2013	Humid (January, February)	Anosiarivo	3	34	15	19	0	0
2013	Dry (May, June)	Anosiarivo	3	12	1	11	0	0

II – Example of a Script for Supervised Classification of Images using Random Forest Algorithm (Python)

```
# Import Python 3's print function and division
from __future__ import print_function, division

# Import GDAL, NumPy, and matplotlib
from osgeo import gdal, gdal_array
import numpy as np
import matplotlib.pyplot as plt
from sklearn.ensemble import RandomForestClassifier
import pandas as pd

# Tell GDAL to throw Python exceptions, and register all drivers
gdal.UseExceptions()
gdal.AllRegister()

# Read in our image and ROI image
img_ds = gdal.Open(‘image_file_name.tif', gdal.GA_ReadOnly)
roi_ds = gdal.Open('raster_file_name.tif', gdal.GA_ReadOnly)

img = np.zeros((img_ds.RasterYSize, img_ds.RasterXSize, img_ds.RasterCount),
               gdal_array.GDALTypeCodeToNumericTypeCode(img_ds.GetRasterBand(1).DataType))
for b in range(img.shape[2]):
    img[:, :, b] = img_ds.GetRasterBand(b + 1).ReadAsArray()

roi = roi_ds.GetRasterBand(1).ReadAsArray().astype(np.uint8)

# Display them
plt.subplot(121)
plt.imshow(img[:, :, 6], cmap=plt.cm.Greys_r)
plt.title('SWIR1')

plt.subplot(122)
plt.imshow(roi, cmap=plt.cm.Spectral)
plt.title('ROI Training Data')
plt.show()
```

```
#Pairing Y with X


# Find how many non-zero entries we have -- i.e. how many training data samples?
n_samples = (roi < 255).sum()
print('We have {n} samples'.format(n=n_samples))


# What are our classification labels?
labels = np.unique(roi[roi < 255])
print('The training data include {n} classes: {classes}'.format(n=labels.size, classes=labels))


# We will need a "X" matrix containing our features, and a "y" array containing our labels
#     These will have n_samples rows
#     In other languages we would need to allocate these and them loop to fill them, but NumPy can be faster


X = img[roi < 255, :]  # include 9th band, which is Fmask, for now
y = roi[roi < 255]


print('Our X matrix is sized: {sz}'.format(sz=X.shape))
print('Our y array is sized: {sz}'.format(sz=y.shape))


# Mask out clouds, cloud shadows, and snow using Fmask
#clear = X[:, 8] <= 1


X = X #[clear, :8]  # we can ditch the Fmask band now
y = y #[clear]


print('After masking, our X matrix is sized: {sz}'.format(sz=X.shape))
print('After masking, our y array is sized: {sz}'.format(sz=y.shape))


# Initialize our model with 500 trees
rf = RandomForestClassifier(n_estimators=500, oob_score=True)


# Fit our model to training data
rf = rf.fit(X, y)


print('Our OOB prediction of accuracy is: {oob}%'.format(oob=rf.oob_score_ * 100))
#feature importance scores
```

```
bands = [1, 2, 3, 4, 5, 6, 7]

for b, imp in zip(bands, rf.feature_importances_):
    print('Band {b} importance: {imp}'.format(b=b, imp=imp))

# crosstabulation to see the class confusion
# Setup a dataframe -- just like R
df = pd.DataFrame()
df['truth'] = y
df['predict'] = rf.predict(X)

# Cross-tabulate predictions
print(pd.crosstab(df['truth'], df['predict'], margins=True))

# predict the entire image

# Take our full image and save as image (
new_shape = (img.shape[0] * img.shape[1], img.shape[2])

img_as_array = img[:, :,:7].reshape(new_shape)

# Now predict for each pixel
class_prediction = rf.predict(img_as_array)

# Reshape our classification map
class_prediction = class_prediction.reshape(img[:, :, 0].shape)

geo = img_ds.GetGeoTransform()
proj = img_ds.GetProjection()

shape = img.shape
driver = gdal.GetDriverByName("GTiff")

dst_ds = driver.Create( "class_prediction.tif", shape[1], shape[0], 1, gdal.GDT_Float32)
dst_ds.SetGeoTransform( geo )
dst_ds.SetProjection( proj )
dst_ds.GetRasterBand(1).WriteArray(class_prediction)
dst_ds = None
```

```
# Display classification
# Visualize

# First setup a 5-4-3 composite
def color_stretch(image, index, minmax=(0, 10000)):
    colors = image[:, :, index].astype(np.float64)

    max_val = minmax[1]
    min_val = minmax[0]

    # Enforce maximum and minimum values
    colors[colors[:, :, :] > max_val] = max_val
    colors[colors[:, :, :] < min_val] = min_val

    for b in range(colors.shape[2]):
        colors[:, :, b] = colors[:, :, b] * 1 / (max_val - min_val)

    return colors

img321 = color_stretch(img, [3, 2, 1], (0, 8000))

# See https://github.com/matplotlib/matplotlib/issues/844/
n = class_prediction.max()
# Next setup a colormap for our map
colors = dict((
    (0, (0, 0, 0, 255)),  # Nodata
    (1, (0, 150, 0, 255)),  # Forest
    (2, (0, 0, 255, 255)),  # Water
    (3, (0, 255, 0, 255)),  # Herbaceous
    (4, (160, 82, 45, 255)),  # Barren
    (5, (255, 0, 0, 255))  # Urban
))
# Put 0 - 255 as float 0 - 1
for k in colors:
    v = colors[k]
    _v = [_v / 255.0 for _v in v]
    colors[k] = _v
```

```
index_colors = [colors[key] if key in colors else
        (1, 1, 1, 0) for key in range(1, n + 1)]
cmap = plt.matplotlib.colors.ListedColormap(index_colors, 'Classification', n)

# Now show the classmap next to the image
plt.subplot(121)
plt.imshow(img321)

plt.subplot(122)
plt.imshow(class_prediction, cmap=cmap, interpolation='none')

plt.show()
```

III – Example of a Script for Validation of the Supervised Classification using Random Forest Algorithm (Python)

```
# Import Python 3's print function and division
from __future__ import print_function, division


# Import GDAL, NumPy, and matplotlib
from osgeo import gdal, gdal_array
import numpy as np
import matplotlib.pyplot as plt
from sklearn.ensemble import RandomForestClassifier
import pandas as pd
from sklearn.metrics import roc_curve, auc, roc_auc_score, confusion_matrix, r2_score, classification_report
from itertools import cycle


# Tell GDAL to throw Python exceptions, and register all drivers
gdal.UseExceptions()
gdal.AllRegister()
# Read in our image and ROI image
img_ds = gdal.Open('image_file_name.tif', gdal.GA_ReadOnly)
val_ds = gdal.Open('raster_validation_file_name.tif', gdal.GA_ReadOnly)


img = np.zeros((img_ds.RasterYSize, img_ds.RasterXSize, img_ds.RasterCount),
               gdal_array.GDALTypeCodeToNumericTypeCode(img_ds.GetRasterBand(1).DataType))
for b in range(img.shape[2]):
    img[:, :, b] = img_ds.GetRasterBand(b + 1).ReadAsArray()


val = val_ds.GetRasterBand(1).ReadAsArray().astype(np.uint8)


# Display them
```

```
plt.subplot(121)
plt.imshow(img[:, :, 0], cmap=plt.cm.Greys_r)
plt.title('classification')

plt.subplot(122)
plt.imshow(val, cmap=plt.cm.Spectral)
plt.title('Validation Data')

plt.show()

#Pairing Y with X

# Find how many non-zero entries we have -- i.e. how many training data samples?
n_samples = (val > 0).sum()
print('We have {n} samples'.format(n=n_samples))

# What are our classification labels?
labels = np.unique(val[val > 0])
print('The training data include {n} classes: {classes}'.format(n=labels.size, classes=labels))

# We will need a "X" matrix containing our features, and a "y" array containing our labels
#     These will have n_samples rows
#     In other languages we would need to allocate these and them loop to fill them, but NumPy can be faster

X = img[val > 0, :]  # include 8th band, which is Fmask, for now
y = val[val > 0]

print('Our X matrix is sized: {sz}'.format(sz=X.shape))
print('Our y array is sized: {sz}'.format(sz=y.shape))
```

```
# crosstabulation to see the class confusion
# Setup a dataframe -- just like R
df = pd.DataFrame()
df['truth'] = y # validation areas
df['classified'] = X #classified areas

# Cross-tabulate predictions
print(pd.crosstab(df['truth'], df['classified'], margins=True))

confusion_matrix (y, X)
print(classification_report(y,X))
```

IV – Script for Calculation of Normalized Difference Vegetation Index (Python)

```
# Import the Python 3 print function
from __future__ import print_function

# Import the "gdal" and "gdal_array" submodules from within the "osgeo" module
from osgeo import gdal
from osgeo import gdal_array

# Import the NumPy module
import numpy as np
from numpy import *

#import the Matplotlib
import matplotlib.pyplot as plt

# Open a GDAL dataset
dataset = gdal.Open('image.tif', gdal.GA_ReadOnly)

# Allocate our array using the first band's datatype
image_datatype = dataset.GetRasterBand(1).DataType

image = np.zeros((dataset.RasterYSize, dataset.RasterXSize, dataset.RasterCount),
                 dtype=gdal_array.GDALTypeCodeToNumericTypeCode(image_datatype))

# Loop over all bands in dataset
for b in range(dataset.RasterCount):
    # Remember, GDAL index is on 1, but Python is on 0 -- so we add 1 for our GDAL calls
    band = dataset.GetRasterBand(b + 1)

    # Read in the band's data into the third dimension of our array
    image[:, :, b] = band.ReadAsArray()

#print('Red band mean: {r}'.format(r=image[:, :, 2].mean()))
#print('NIR band mean: {nir}'.format(nir=image[:, :, 3].mean()))
```

```
b_red = 1 #please indicate position of the band
b_nir = 2

ndvi = (image[:, :, b_nir] - image[:, :, b_red]) / (image[:, :, b_red] + image[:, :, b_nir])

#gettting geographical information of the tiff
geo = dataset.GetGeoTransform()
proj = dataset.GetProjection()

shape = image.shape
driver = gdal.GetDriverByName("GTiff")

dst_ds = driver.Create( "ndvi.tif", shape[1], shape[0], 1, gdal.GDT_Float32)
dst_ds.SetGeoTransform( geo )
dst_ds.SetProjection( proj )
dst_ds.GetRasterBand(1).WriteArray(ndvi)

dst_ds = None  # save, close
print(ndvi)
print(ndvi.max())

#we are setting up the image display
plt.figure(figsize=(8,14))
plt.subplot(122)
plt.imshow(ndvi, cmap=plt.cm.Greys_r)
plt.title('NDVI')
plt.show()
#plt.colorbar()
```

V – Script for Calculation of Soil Adjusted Vegetation Index (Python)

```
# Import the Python 3 print function
from __future__ import print_function

# Import the "gdal" and "gdal_array" submodules from within the "osgeo" module
from osgeo import gdal
from osgeo import gdal_array

# Import the NumPy module
import numpy as np
from numpy import *

#import the Matplotlib
import matplotlib.pyplot as plt

# Open a GDAL dataset
dataset = gdal.Open('image.tif', gdal.GA_ReadOnly)

# Allocate our array using the first band's datatype
image_datatype = dataset.GetRasterBand(1).DataType

image = np.zeros((dataset.RasterYSize, dataset.RasterXSize, dataset.RasterCount),
                 dtype=gdal_array.GDALTypeCodeToNumericTypeCode(image_datatype))

# Loop over all bands in dataset
for b in range(dataset.RasterCount):
    # Remember, GDAL index is on 1, but Python is on 0 -- so we add 1 for our GDAL calls
    band = dataset.GetRasterBand(b + 1)

    # Read in the band's data into the third dimension of our array
    image[:, :, b] = band.ReadAsArray()

#print('Red band mean: {r}'.format(r=image[:, :, 2].mean()))
#print('NIR band mean: {nir}'.format(nir=image[:, :, 3].mean()))
```

```
b_red = 1 #please indicate position of the band
b_nir = 2
L = 0.5

savi = (1.0 + L) * (image[:, :, b_nir] - image[:, :, b_red]) / (image[:, :, b_nir] + image[:, :, b_red] + L)

#gettting geographical information of the tiff from the original image
geo = dataset.GetGeoTransform()
proj = dataset.GetProjection()

shape = image.shape
driver = gdal.GetDriverByName("GTiff")

dst_ds = driver.Create( "savi.tif", shape[1], shape[0], 1, gdal.GDT_Float32)
dst_ds.SetGeoTransform( geo )
dst_ds.SetProjection( proj )
dst_ds.GetRasterBand(1).WriteArray(savi)

dst_ds = None  # save, close
print(savi)
print(savi.max())

#we are setting up the image display
plt.figure(figsize=(8,14))
plt.subplot(122)
plt.imshow(savi, cmap=plt.cm.Greys_r)
plt.title('SAVI')
plt.show()
#plt.colorbar()
```

VI - Example of a Script for Multi-Model Inference using Generalized Linear Models (R software)

```
##Upload packages##
library(lme4)
library(MuMIn)
library(vegan)
library(car)
library(Hmisc)
library(rcompanion)
library(modEvA)
library(AER)

#load(data)
#check for correlation between variables using Spearman’s correlation
rcorr(as.matrix(file[ ,2:5]), type="spearman")

#check whether the data is normally distributed
SR<-density(SR_30_samp$SR)
plot(SR)
def<-density(SR_30_samp$Def)
plot(def)
NDVI<-density(SR_30_samp$NDVI)
plot(NDVI)
alb<-density(SR_30_samp$Albedo)
plot(alb)

#Remove predictors that are correlated to each other
indices<-subset(file, select=c("response_variable","predictor1", "predictor2"))
#Full model
Glm_1<-glm(response_variable~predicotr1+predictor2,data=indices,family=poisson)
#calculate multicollinearity between predictors
vif(Glm_1)
summary(Glm_1)

#Test for deforestation
Glm_2<- glm(response_variabe~predictor1,data=indices,family=poisson)
summary(Glm_2)
```

```
#test for ecosystem functioning
Glm_3<- glm(response_variable~predictor2,data=indices,family=poisson)
summary(Glm_3)

#Model ranking
Va.mods <- list(Glm_1, Glm_2, Glm_3)
aictab <- model.sel(Va.mods)
aictab
print.data.frame(aictab,digits=2)
x <-model.avg(Va.mods, beta = TRUE, revised.var = TRUE)
summary(x, digits = 3)

#To test whether Poisson distribution fit the data (p>0.05)
1 - pchisq(summary(Glm_1)$deviance, summary(Glm_1)$df.residual)
1 - pchisq(summary(Glm_2)$deviance, summary(Glm_2)$df.residual)
1 - pchisq(summary(Glm_3)$deviance, summary(Glm_3)$df.residual)

#Test for overdispersion for best model
dispersiontest(Glm_3)

#plot residuals of best model
plot(Glm_3)
```