THE MISSING LINK

Bridging the gap between science and conservation

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To my loving parents

CONTENTS

Cover: grey-headed kingfisher
Next page: juvenile chimpanzee

Chapter 1

INTRODUCTION

The loss of biodiversity

Biodiversity is generally defined as the variety and abundance of species, their genetic composition, and the natural communities, ecosystems, and landscapes in which they occur. Tropical countries harbour the most biodiverse ecosystems in the world, of which tropical rainforests are the richest biome of all (Olson & Dinerstein 1998; Myers 2000). However, in Africa human population density is unfortunately positively correlated to mammal, bird, snake and amphibian species richness (Balmford *et al.* 2001). This indicates the potential for conflicts between biodiversity conservation and human development in the tropics (e.g., Singapore: Brook *et al.* 2003). Conservation is of less importance in the daily lives of most people in these biodiversity hotspots than is the quest for regular food and income. Natural resources, such as timber, minerals, and wildlife are often exploited far above sustainable levels by national and international operators (Bowles *et al.* 1998; Fa *et al.* 2002, Fowler & Hobbs 2003). Human population growth and continued opening of new areas for agriculture, logging, and mining increases the pressure on rainforests and its resources throughout the tropics. Many people are dependent on these resources for their daily livelihoods (Robinson & Bennet 2000; Milner-Gulland *et al.* 2003). The rate of deforestation in Africa alone was 4 million hectares annually over the period 2000-2005 (0.3 and 1.2% per year for Central and West Africa respectively), which amounts to 55% of the global reduction in forest cover (FAO 2007). The forest loss for West and Central Africa over the period 2000-2005 was approximately 1.5 million hectares annually. This is about half the surface of the Netherlands, or eleven times the surface of the province of Utrecht that disappears each year, to put the deforestation rate in a perspective. The disappearance of these rainforests and its wildlife thus poses a threat to the survival of local people if alternative sources of protein are not available (Bennet 2002; Davies 2002; Fa *et al.* 2003; Milner-Gulland *et al.* 2003). Apart from ethic reasons to preserve these threatened ecosystems, the argument that without the services of these rainforests many people will be threatened in their survival has become a more and more important drive in biodiversity conservation throughout the tropics (Milner-Gulland *et al.* 2003).

One of the most important services provided by rainforests for local people is wildlife as an extractable resource. However, the first to suffer from high human pressure is forest wildlife (Wilkie & Carpenter 1999, Milner-Gulland *et al.* 2003, Robinson & Bennet 2004). Wildlife is a free commodity that is exploited in a ravaging way, hardly ever with any regulation to guarantee a long-term sustainable yield. Costs for materials for hunting, like cartridges and snares, are negligible. Animals caught in the forest (hereafter called bushmeat) do not only provide local households with food, but are also traded with urban centres sometimes far away. The bushmeat trade from the forest interior to large urban settlements, sometimes far away, is huge and difficult to control (Milner-Gulland & Clayton 2000; Cowlishaw *et al.* 2005; Crookes *et al.* 2005). It is estimated that approximately 5 million tons of bushmeat is harvested annually in the Congo basin alone (Fa *et al.* 2002), or on average 645 kg of bushmeat per km^2 per year (range 50-897 kg.; Wilkie & Carpenter 1999). For the Amazon basin this is far less: 0.15 million tons annually. Fa *et al.* (2002) calculated that Congo basin mammals should produce 93% of their total biomass annually to balance extraction rates, while this is only 4% for the Amazon. Harvest rates in the Congo basin surpass by far the suggested sustainable harvest rate of 20% of the production of long-lived species (production is immigration and reproduction, minus emigration and deaths, Robinson & Redford 1991, p.421).

Hunting often accompanies logging companies in the forest interior (Putz *et al.* 2001). During extractive exploitation hunters provide the labour force and their families with meat. Professional hunters follow these labourers into the forests and apart from providing the local people with meat, also transport large amounts of bushmeat to cities (Wilkie *et al.* 2000). When logging exploitation has ceased, more people turn to hunting as a source of income, increasing the pressure on wildlife. Alternatives for bushmeat are scarce and underdeveloped (Fa *et al*. 2003), or have a direct effect on the level of bushmeat exploitation (e.g., decline in fish supply increases bushmeat consumption: Brashares *et al.* 2004). Large farming or breeding projects that could provide a steady supply of protein for the human population in cities and rural areas are non-existent in most developing countries with high rainfall in Africa.

Some areas are already almost devoid of most original wildlife species, resulting in the so-called “empty forest syndrome” (Redford 1992) or are threatened with a rapid decline in species diversity (Barnes 2002; Brook *et al.* 2003; Milner-Gulland *et al.* 2003). Many species are threatened with extinction in their original habitat, which influences the stability of forest ecosystems (Redford 1992; Andresen 2000 for frugivorous seed dispersers; Redford & Feinsinger 2001 for several examples; McConkey & Chivers 2007 for gibbon).

The history of biodiversity conservation

The earliest cases of wildlife protection by setting aside protected areas (PA’s) were based either on religious grounds, with sanctuaries as PA’s, or were for the regulation of hunting by setting aside exclusive hunting reserves. Sanctuaries with a religious motivation for protection became relatively fewer over time due to the increase in other categories of PA’s. Protection within PA’s was for exclusive right of (colonial) landowners to ensure a stable stock of wildlife for private use. This method of protection was gradually taken over by governments and environmental agencies. However, the goal was not anymore the use of the protected resource, but became the prolonged protection of specific species or habitats through a set of designated PA’s. Subsequently PA’s were managed by conservation agencies, where strict protection was the doctrine, without involvement of the local people and where even evictions from PA’s took place, although the set-up of a PA network was done “in the interest of the public” (Salafsky & Wollenberg 2000, pp 1422). Nowadays there are international agreements that strive for an extensive amount of land surface to be designated as PA. At the

3rd IUCN World Parks Congress in 1992 the attendees agreed to expand the global protected area system to 10% of the earth's terrestrial surface. For the global rainforests this is currently 23.3%, but for West and Central Africa it is only 8.7% (Chape *et al.* 2003). One of the Millennium Development Goals incorporated the ratio of area protected to maintain biological diversity to total surface area as an indicator (MDG 7, target 9, www.undp.org).

In the 1990s of the last century, local populations became more involved in the management and set-up of PA’s (Salafsky & Wollenberg 2000). It was acknowledged that biodiversity could not be protected without considering the needs of the people living in or around these areas (Stearman 2000; Davies 2002; Chapin 2004). Therefore conservation organisations integrated development in their practice, which flourished with the Integrated Conservation Development Projects (ICDP). Objective of these ICDP’s was to link biodiversity conservation in PA’s with local socio-economic development. The ICDP’s aimed at providing alternative sources of livelihood to local people living in or near PA’s in an effort to reduce the pressures on PA resources (Salafsky & Wollenberg 2000).

The problem

These ICDP’s have not proven to be the solution, in spite of great attention (McShane & Wells 2004; Welsh & McShane 2004; Chapin 2004). Salafsky & Wollenberg (2000) developed a method to assess the linkage between development and conservation in a range of projects in order to rank their success, and concluded that high linkage by itself was not a guarantee of conservation success. Early warnings of possible failure of the ICDP approach, based on preliminary field experiences, were ignored whilst the Global Environmental Facility (GEF) and other multilateral agencies spent most of the available money on ICDP projects. About a decade later, when it became clear that success was not to be reached, conservation organisations changed course again (McShane 2003).

There has been an increasing call to turn attention and funding to the vast tracts of degraded land that surround relatively untouched forests (Lovejoy 1985; Daily 1995; Putz *et al.* 2001; Schwartzman *et al.* 2000; Sarukhán 2006), without reducing the establishment and proper management of protected areas. Secondary forests are the predominant landscape type in areas where rainforest used to cover the landscape. Arguments have been put forward to concentrate on these areas as they will become even more widespread, and the area covered by the original pristine rainforest will only diminish more (Putz *et al.* 2001, Robinson 2006). Managing these degraded or recovering forests requires a different approach then PA’s do, because here humans and their use of natural resources form an intricate part of the system. Management in degraded forests often resembles management that is practiced in buffer zones surrounding protected areas (Salafsky & Wollenberg 2000, Lynagh & Urich 2002). Sustainable use, or the safeguarding of livelihoods for local people, is not to be achieved because generally exploitation in a PA is not allowed.

Most large conservation organisations therefore integrate nowadays aspects of development and management of natural resources by the local populations in their policies, as they assume that without the consent of the local people conservation of biodiversity is not likely to succeed (Salafsky & Wollenberg 2000). An example is Tropenbos International, a Dutch based NGO which specifically integrates human development in its goal to achieve sustainable forest management throughout the global rainforest regions. This is expressed in their objective:

> "TBI's overall objective - improving forest management for the benefit of people, conservation and sustainable development - emphasizes the role of people. A better forest management should benefit people. People who live in or near forests and depend on forest and forest resources for their livelihood, but also those who depend on forests for the environmental services they provide and whose employment and income depend directly or indirectly on the forest industry." (www.tropenbos.org).

The development organisations on the other hand are increasingly aware that without safeguarding biodiversity, future livelihoods of the local people are at risk. For instance, the Dutch development organization SNV has incorporated this aspect in their objectives:

> "More than one billion poor people directly and indirectly depend on forest resources for income, food and job opportunities. Sound forest management and resource use are essential to raise their standard of living. (…) SNV supports the shift in emphasis from forests as a protected resource to a potentially productive asset. SNV's main clients are intermediate organizations whose input is the key for ensuring that sustainable management of forests contributes to reduction of poverty." (www.snvworld.org).

Many actors, be it development organisations, conservation organisations, governments or NGO's, collaborate more in order to safeguard biodiversity and sustained livelihoods for local populations at the same time. This requires a new approach for the use and protection of the environment and its resources. I do not deny the necessity of PA's with effective management, however, given the increasing pressure on the diminishing pristine forests, it is my opinion that it is imperative and inevitable that more attention should be given to the regulated use of resources outside PA's.

Although conservation policies and activities are developing progressively towards a more sustainable use and protection of biodiversity, we are still far from halting or at least slowing down the loss of this biodiversity. One of the main reasons is the lack of funding for the regulation of the use and the protection of biodiversity (James *et al.* 2001; Balmford & Whitten 2003; Halpern *et al.* 2006). As described earlier, there is also the failure to integrate socio-economic development with conservation activities (e.g., ICDP's). Another more pragmatic reason for this lack of success is that within conservation projects there often is a "knowledge-doing gap" or an "assessment-planning gap" (Knight *et al.* 2006a). There is disproportionally much knowledge available on biodiversity (e.g., through long term systematic surveys on species, ecosystems, etc.), and on the link between human activities and biodiversity (e.g., the link between hunting, logging and for example wildlife abundances). However, this information is not translated and used for effective conservation activities in the field (Whitten *et al.* 2001. The uneven division of attention between assessment and application has led to the so-called implementation crisis (Knight *et al.* 2006b).

The preponderance of systematic assessment studies is probably the heritage of the role classical science fulfilled within conservation biology in history (Knight *et al.* 2006a). Research used to aim at increasing the general understanding of ecosystems and associated biological processes. This kind of research required large areas of undisturbed ecosystems, hence the fruitful combination of biological science and protected areas. Although scientific attention on the effects and modelling, such as the development of harvesting models, increased over time, most research conducted in the tropics was still based on fundamental scientific methodologies (Western 2003). Surveys, descriptive studies, monitoring, etc., were still required to follow rigid western-based scientific procedures. Studies or surveys were aimed at obtaining information for purposes that did not necessarily comply with the general objective of a project within which the research was embedded. The objective of conservation projects increasingly changed from the description and exclusive protection of biodiversity towards the promotion of sustainable use practices and integrated management. The discrepancy between development of conservation science and of conservation practice as described above is worrying (Terborgh 2004).

Now that it is accepted that we are faced with an implementation crisis, we need to look for concrete changes in conservation biology that are required to increase the applicability of research to attain effective conservation (Robinson 2006). Central question of this thesis is: what is the position of current classical scientific activities within conservation biology, and what changes in these scientific activities can bridge this “knowledge-doing gap”? I address several possible solutions to this problem, based on case studies that provide practical advice for on-the-ground management. The call for the integration of local knowledge in classical scientific activities (Sheil & Lawrence 2004) is tested in two case studies (chapters 2 and 3). I discuss furthermore the combination of the results of separate research activities with the objective to provide a more comprehensive and solution-based advice for conservation planners (chapters 2, 3, and 4). And finally, the relevance of classical fundamental scientific research and its practicality for conservation is discussed based on two additional studies (chapters 5 and 6). By presenting these case studies and their combined potential I intend to stimulate the development and testing of research activities that are more based on the requirements of conservation planners and practitioners, while using existing and newly developed scientific techniques.

The case

Working in a Biodiversity Conservation Project myself, charged with providing scientific information for the set-up of a management plan I was confronted with the problem of having to comply with two criteria: providing useful data for management purposes and providing these data following a scientific sound basis. These criteria are very different and require a different approach. To provide scientific or statistically sound data in the conventional way requires much time and large amounts of money (e.g., transect and aerial surveys), which are both nowadays often unavailable within project budgets. Methodologies are mostly copied from previous experiences where survey and modelling methods were used that originate from the time when scientific research was more the goal of a conservation project than it was a means to achieve effective management. Involvement and commitment of local people was therefore limited. Currently project management needs results fast, which often does not comply with scientific standards. The dilemma prompted me to search for alternative methods for data-gathering that would still stand up to rigorous scientific scrutiny. While searching for applicable wildlife survey methods that fitted within the budget and time frame of the project I realized that many current methodologies in conservation science have not changed, while the approach to conservation practice has. The restriction in time and budget pressed me to look for other sources of information. Some of the information is available with local people as local expert knowledge (Sheil & Lawrence 2004). Although local experts are often not familiar with conservation science, their knowledge on animal ecology and biological data is generally extensive. A new approach was needed to extract and use this knowledge and translate this to useful results.

The questions on how to improve the practicality of research and how to integrate local knowledge were addressed in several studies conducted in the World Bank/Global Environment facility (WB/GEF) Campo-Ma'an Biodiversity Conservation and Management Project in Southern Cameroon as from 2000. The project had three executing agencies: the former Cameroonian Ministry of Environment and Forests (MINEF), Tropenbos International, and the Dutch development organisation SNV. The general objective of the WB/GEF project was to improve the management of protected areas in Cameroon, to protect and conserve the unique biodiversity of Cameroon, and to promote socio-economic development of the local communities based on the sustainable use of natural resources. The Campo-Ma’an project included the preparation of a zoning plan that took into account different levels of protection, sustainable timber production, multiple-use and community forestry. The formulation and implementation of the management plans for each zone was part of the regional land use plan. Main threats to the success of the plans were unsustainable forest exploitation, hunting, and slash and burn agriculture. The specific objectives of the Campo-Ma’an Project were:

- conservation of the unique tropical rainforest and biodiversity in the area, among others through zonification;
- development and implementation of a sustainable land use plan and a socio-economic development plan for the project area, as basis for biodiversity protection and community development;
- participation of concession holders in the development and implementation of management plans for timber production areas.

Surveys and research were conducted within the framework of the development of a management plan for the Campo-Ma’an project. Contemporary survey methods and scientific activities were discussed, tested, and compared to newly developed ones. Particular attention was given to the objective to integrate local knowledge in research activities in order to promote commitment and ownership of the people affected by the management plan that was to be implemented. A second objective was to plan and execute research activities in such a way that they would complement each other, thereby producing a more comprehensive data set for conservation planners.

Study Area

The integrated management area of the Campo-Ma’an Management and Biodiversity Conservation Project is located in the Southwest corner of Cameroon and is called “Unité Technique Opérationnelle” (UTO). This UTO was created in August 1999 and covers an area of 7,772 km^2, located between latitudes 2°10’-2°52’ N and longitudes 9°50’-10°54’ E (Fig.

1). Objective of the UTO was the conservation of the rich biodiversity of the area, the sustainable use of its resources, and economic development (De Kam *et al.* 2002). The project area was divided in a protection zone (which was to become a national park in 2000) and a multiple use zone. This multiple use zone consisted of industrial oil palm and rubber plantations (Socapalm and Hevecam resp.; 7% of the total UTO surface); four logging concessions (31%), some of which have been logged over twice or even three times since the 1960s; an agro-forestry zone (27%), and a coastal zone (1%). The later established national park Campo-Ma'an covered 34% of the UTO area (Fig. 1). The overall objective was to achieve development and conservation through integrated management of the different uses, with a long-term vision on conservation and socio-economic development (De Kam *et al.* 2002).

The project area is bordered by the Atlantic Ocean and is covered mainly with Guinea-Congolian rainforest. The altitude ranges from 0 to 1100 m.; a mountain chain extends from southwest to northeast. Annual rainfall varies from 1670 to 2950 mm. The rainforests of the Campo-Ma'an area are unique in its species richness: it contains 114 endemic plant species (Tchouto *et al.* 2006), more than 300 bird species of which 24 are rare or endangered. Up to half of all mammal species found in Cameroon and two-thirds of those found in dense forests are recorded in the area (De Kam *et al*. 2002). A possible explanation for this high rate of endemism and species richness is that the site is part of a series of postulated rain forest refuge areas in Central and West Africa (Rietkerk *et al.* 1995; Tchouto *et al.* 2006). During the last ice age (18,000 yrs ago) most of the tropical rainforests in West and Central Africa had disappeared, except for a few isolated areas, the rainforest refugia (Tchouto *et al.* 2006). Here survival and speciation may have contributed to the high rate of endemism and species richness.

The richness of the flora and fauna of the area was already recognised by the Cameroonian government in 1932 with the establishment of the "Réserve de faune de Campo". However, the actual level of protection in this reserve was non-existent, proven by the fact that most of the reserve was under logging concession by 1968 (de Kam *et al*. 2002). In January 2000, the Campo-Ma'an National Park (CMNP) was established within the UTO, with a surface area of 2,640 km^2. Part of the CMNP had been logged several years prior to its creation, and an abandoned road dissects the CMNP from west to east. This road was re-opened in 2002, and abandoned again in 2006, after the logging company who maintained the road closed its concession. The government of Cameroon again expressed its interest in preserving the rich biodiversity of Campo-Ma'an in 2006, by nominating it a World Heritage site (http://whc.unesco.org/en/tentativelists/4021).

The latest figure (in 2001) on population size indicated that about 61,000 people live in the UTO, spread over 167 towns and villages (Tchouto *et al.* 2006). The average population density is approximately 10 people per km^2, which is generally considered low, especially knowing that most people live in the rubber and palm plantations, in the main town Kribi, and along the coast. Most important activities in the UTO are agriculture, logging, hunting and fishing. Subsistence hunters and farmers in the UTO obtain more than 75% of their animal proteins from bushmeat (Annaud & Carriere 2000).

Many botanical surveys have been conducted in the area (see Tchouto *et al.* 2006 for a list), but little zoological research has been done. Only three peer-reviewed scientific studies have been published on wildlife, all on primates (Mitani 1991; Matthews & Mathews 2002; Matthews & Matthews 2004).

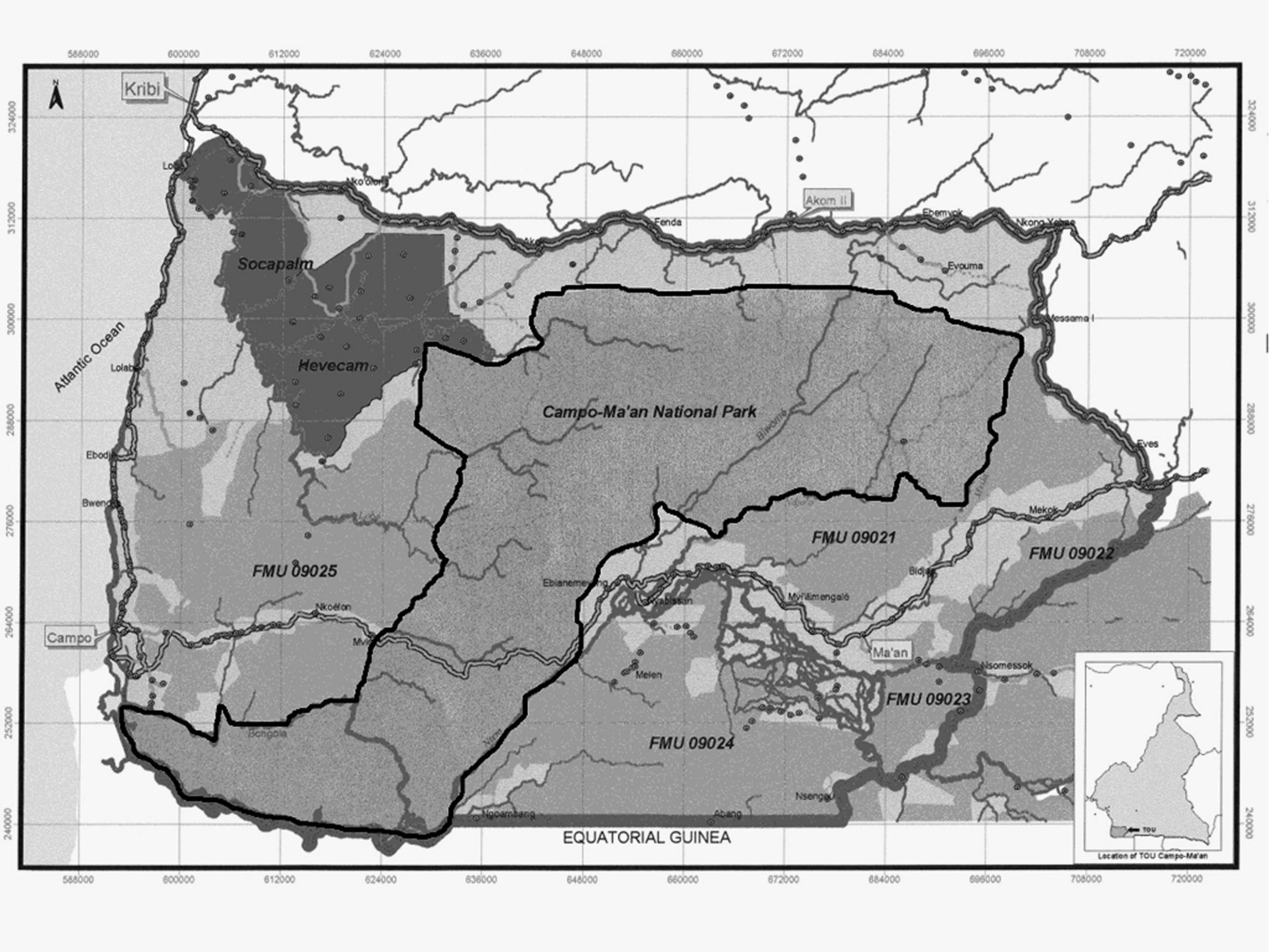

Figure 1. Map of Campo-Ma’an study area in the South of Cameroon. FMU’s are logging concessions (Forest Management Units).

Set up of thesis

The collection of baseline data, constraints of existing survey techniques, and the development of a new, cheap and fast method of estimating wildlife densities in tropical rainforests are discussed in the first part of this thesis. This new method of wildlife survey has been developed to reduce costs, time constraints, and dependence on outside scientific expertise. Ultimately this method should increase local ownership of the wildlife management plan, as the method is based on local knowledge, and it could be easily taken over in order to promote adoption of this work by local experts. Commitment of local people to sustainable use and conservation should improve with this methodology (Chapter 2).

The second part describes a vulnerability assessment method to indicate which wildlife species are vulnerable to rapid decline and local extinction. This assessment helps to prioritize what is needed to improve effectiveness of wildlife management. The vulnerability assessment is based on factors that define survival and vulnerability to decline and extinction of wildlife species in tropical rainforests. Information from literature is used in combination with local knowledge in order to supplement the known regional or global situation with the local data (Chapter 3).

When baseline data and ranking of species for conservation and management according to vulnerability is known, the development of a monitoring system logically follows. The third part discusses the importance of monitoring wildlife use, based on a practical example, a bushmeat market survey. Monitoring the bushmeat trade is hampered by logistical and methodological problems, but forms an important link between socio-economic development and conservation of threatened species, as the bushmeat trade is the interface between wildlife as a resource and as an asset (Chapter 4).

The fourth part of this thesis illustrates an example of a negative effect of an integrated project approach: a human-wildlife conflict that originated when multiple-use functions are assigned to a project area. The effects of a road through the Campo-Ma’an National Park on wildlife road-crossing behaviour of a variety of species are described and possible mitigating actions are suggested (Chapter 5).

The cascading effects of this road construction are illustrated in the last part, with a case study on the invasion of an exotic plant species along the logging roads. This invasive shrub species (*Chromolaena odorata*) threatens the gorilla in the project area, and potentially threatens this ape in the whole of its range (Chapter 6).

In the synthesis I will try to answer the questions on the position of conservation science in current conservation practice, based on literature and the case studies. The results of the first three studies (chapters 2, 3, 4) are combined and analysed whether they complement and reinforce each other. Furthermore, the use and practicality of classical long-term scientific research is discussed on the basis of two specific case studies.

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Next page: taking interview for PLEO

Chapter 2

POOLING LOCAL EXPERT OPINIONS FOR ESTIMATING MAMMAL DENSITIES IN TROPICAL RAINFORESTS

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Abstract

Methods currently used for assessing wildlife density in rainforests are time and money consuming. The precision of the most commonly used methods are disputed, but accepted because more exact methods are not available. In this study a new method of wildlife density estimation is explained. The new method is less time and money consuming, but yields comparable results with classical methods. The method was tested in the field in Cameroon and compared with transect surveys in the area and with relevant literature. The PLEO (Pooled Local Expert Opinion) method is based on the knowledge of local experts. A number of hunters were asked to estimate wildlife abundance in a specified area, after which the density per km^2 was calculated for 33 wildlife species. These estimates were pooled and extrapolated for the whole study area. Elephant (*Loxodonta africana*) density outside the National Park was estimated to be 0.06 animals per km^2, and 0.3 inside. Buffalo (*Syncerus caffer*) density for the study area was estimated at 0.2 animals per km^2 and gorilla (*Gorilla gorilla)* density at 1.05 per km^2. Transect surveys carried out at the same time for considerably more money, taking far more time, produced too few data too calculate densities. The evaluation of the PLEO-method was favourable and the method offers a substitute for conventional methods of estimating wildlife density in rain forests. The methodology is simple and it can be incorporated in many tropical biodiversity and conservation projects. It can also be used for long-term monitoring of wildlife status in a given area. In contrast with classical methods, the PLEO-method is low in cost and assures local ownership of the results.

Introduction

Monitoring of biodiversity has been defined by the World Bank as the gathering of data to enable detection of changes in the status, security and utilisation of biological diversity for the purpose of improving the effectiveness of biodiversity management (World Bank 1998). Monitoring of wildlife populations is considered essential for the management of hunting, tourism and biodiversity conservation, especially regarding the bushmeat crisis (Wilkie & Carpenter 1999; Spector & Forsyth 1998). Bushmeat is a term used for all animal species caught for consumption, and it is called a crisis because the overexploitation of this resource is regarded as a major threat to the survival of many species (Fa *et al*. 2003; Barnes 2002; Robinson & Bodmer 1999).

Assessment and subsequently monitoring of wildlife populations in tropical rainforests is tedious. Animals are elusive, leave few tracks, and visibility in the forest is limited. The most commonly used methods are transect surveys, consisting of line, recce and strip transects. Other less used methods are point sampling (e.g., monitoring of forest clearings, Magliocca *et al*. 1999) and sweep census (White & Edwards 2000; Tutin *et al*. 1995).

For regular monitoring, line and strip transects are costly and time consuming. Transect surveys should be done during a whole year in order to adjust for seasonal changes in behaviour of the animals (Fashing & Cords 2000). Furthermore, to census effectively, a team of at least two to three persons is needed in order to detect all signs, tracks and nests (White & Edwards 2000). The number of species that can be monitored is limited because only a few animal species leave clear tracks, or are readily detected by their vocalisations. Moreover, in most cases less than 0.5 % of the study area is sampled when conducting transect surveys. Most important though, is that transect data are often too scarce to calculate density estimates, as was the case in this study (see also Matthews & Matthews 2000). Results are in that case often presented as indications of abundance, preventing any reliable comparisons. All these constraints limit the effectiveness of transect surveys as a tool for the monitoring of wildlife population trends (Wilkie & Carpenter 1999). Apart from the methodological constraints, biodiversity management projects are strictly bound to budgets and fixed time frames, arguing for effective, less time and cost consuming methods (World Bank 1998).

This study describes a new method of mammal density estimation aimed at obtaining results with low costs and within a short time frame. The results were compared with literature on mammal densities in the region. The developed method is based on Pooled Local Expert Opinion (PLEO) and uses the knowledge of local hunters and wildlife specialists. The crux is that instead of estimating wildlife densities oneself, the researcher asks people who hunt or work in particular areas for longer periods to estimate wildlife densities in their particular hunting area. The opinion of several experts is pooled to give an overall estimate of wildlife densities for the study area. The PLEO method is developed in such way that local authorities with limited specific training can easily adopt it. Furthermore, to implement this method does not require extensive education and can be used by persons with a basic biological and statistical background. This ensures local ownership of the method and its results.

The reliability and practicality of the PLEO method is discussed on the basis of a case study in Cameroon. The results were used to test whether the density of certain species was higher within Campo-Ma'an National Park than outside, and to estimate the population sizes of the most threatened species in the study area. A cost and time evaluation was performed, comparing transect surveys with the PLEO method.

Methods

Study site

The study was carried out within the framework of the Campo-Ma'an biodiversity conservation and management project in southern Cameroon. This is a World Bank - Global Environmental Facility (GEF) project, with three executing agencies: the Cameroonian Ministry of Environment and Forests (MINEF), Tropenbos International and the Dutch development organisation SNV.

The integrated management area of the Campo-Ma'an Management and Biodiversity Conservation Project is located in the Southwest corner of Cameroon and is called “Unité Technique Opérationnelle” (UTO). This UTO was created in August 1999 and covers an area of 7,772 km^2. The project area is bordered by the Atlantic Ocean and is covered mainly with Guinea-Congolian rainforest. The altitude ranges from 0 to 1100 m.; a mountain chain extends from southwest to northeast. Annual rainfall varies from 1670 to 2950 mm.

In January 2000, the Campo-Ma'an National Park (CMNP) was established within the UTO, consisting of 2,640 km^2, or 34% of the area. Part of the Park had been logged several years prior to its creation. The UTO consists of four additional logging concessions (31%), which have been logged over twice or even three times, an agro-forestry zone (27%), industrial rubber and oil palm plantations (7%) and a coastal zone (1%). Subsistence hunters and farmers in the UTO obtain more than 75% of their animal proteins from bushmeat (Annaud & Carriere 2000). Preliminary inquiries indicated that wildlife abundance within the plantations and the coastal zones was negligible, and most hunting took place outside these areas. The latter two areas were therefore not included in the survey due to their disparate character.

Methodology

The origin of the Pooled Local Expert Opinion lays in the Delphi procedure (Dalkey & Helmer 1963), based on the subjective opinion of several experts, providing a group judgement for subject matters where precision is lacking. The method has proved to be a practical and efficient way to obtain a “best estimate” in uncertain contexts (Prins & Wind 1993). Originally, this method was developed to obtain opinions about future prospects in commercial trade (Dalkey & Helmer 1963). In this project it concerns mammal density as estimated by local hunters.

The information on mammal density was obtained by means of semi-structured interviews. Villages were selected so that they were spread evenly throughout the UTO in order to avoid overlap. When the villages were visited, the village chief was consulted to assist in the selection of hunters to be interviewed. Our requirement was that the hunter should still hunt, should have hunted regularly in the same area, and should be considered an expert by his peer group in the village.

In order to compare results with other studies and to extrapolate the results to the whole UTO, density should be expressed in animals per km^2. However, hunters are often not accustomed to the abstract perception of the metric system. Therefore, a clearly bound area within the region where a respondent hunted was selected (henceforth called a sample area). The respondent was made to understand that the interview was only about animals within this particular sample area. There were as many sample areas as respondents, as each respondent estimated the abundance of his particular sample area. Therefore, each sample area had only one abundance estimate from one respondent. A species list was provided in the local language, supported with drawings of each animal.

The interviewer defined the sample area by selecting an area that was bound by logging roads, streams, rivers or hills, which were named or indicated by the respondent and localised by us on a georeferenced map. Sample areas were kept small (7-17 km^2) for the repeatable recall of estimated population sizes from the hunters' memory. The surface of the sample area was calculated with the help of GIS.

The hunter was asked for an estimate, as is normal under the methodology of market forecasting, from which PLEO is derived. Therefore, he was asked, for example, whether there were more or less than 100 elephants in the sample area; if less, whether there were more or less than 50, and so on. In the end, the estimate would be "about 15", or "between 25 and 50". If ranges were given (e.g. 25 – 50), the mid-point of this range was used for further analysis. If a species lived in groups, the hunter was asked how many groups lived in the sample area. Often species have fixed territories, of which the hunter is aware, which made providing the number of groups fairly easy. When the number of groups was known, he was asked to estimate the number of individuals per group. This average group size was then multiplied by the number of groups, resulting in an overall estimate of abundance for the sample area.

Related species, like the various duiker and monkey species, offer the possibility to compare densities, thereby providing the hunter with a double check on his estimates. When considering bay duiker (*Cephalophus dorsalis*) for example, the hunter was asked whether it was more abundant than yellow-backed duiker (*Cephalophus silvicultor*). When he confirmed this, it was checked whether it complied with his estimates given in the interview. If it did not fit, he was asked to reconsider his estimates.

Interviews took place in four sessions, in the dry season (January and February), and during the rainy season (August and October). Each session took 6–14 days. The interviews included 10 respondents with hunting zones located within the CMNP (hunting was still common in the park at that time), and 24 respondents hunting outside the CMNP.

The number of interviews was set at 34, as the aim was to get a sample size of at least 5% of the total study area. With 34 samples, the sample size (that is, the surface of all sample areas combined) was 5.9% of the total study area. A 5% sample count is sought in census estimates in savannas (Norton-Griffiths 1975), and is used here by lack of suggestions for rain forests.

A selection of medium to large mammals was made, resulting in a list of 30 species (Table 1). Three reptile species that were specifically hunted and much in demand were added to this list. For Latin names of the species we follow Kingdon (1997).

Mandrill (*Mandrillus sphinx*) is treated in a special way for the following reason. When the respondent encounters a group of mandrills he would probably mention the number in the group (for example, 150 individuals) as the number that live in the sample area. Mandrills live in large (600), sometimes very large (>800) groups (Abernethy *et al*. 2002) and they occupy extensive home ranges of up to 5000 ha. (Sabater Pi 1972; Kingdon 1997). The problem is that mandrills move continuously, up to 10 km a day (Gautier-Hion *et al*. 1999), thereby only residing in the sample area part of the year. Calculating the consecutive density for the sample area probably results in too high a density, as the sample area is smaller than the (average) home range of mandrills. The results were therefore corrected for this overestimation. The density estimate of the respondent was multiplied by the ratio (sample area: home range). As home range 3000 ha was taken (Sabater Pi 1972; Kingdon 1997).

Analysis

The density for each species per sample area was calculated by dividing the number of animals by the surface of the sample area. The 34 density estimates per species were then pooled, resulting in one overall density estimate per species for all sample areas together.

Densities for inside and outside the CMNP were tested for differences using a student t-test on untransformed data. If there was a significant difference, samples were processed separately for inside and outside the CMNP. Non-significant different samples were treated as coming from one population. On the basis of *a priori* reasoning, fifteen species were expected to differ in density between the inside and the outside of the CMNP. Six species were expected to be more abundant outside than inside the park. A reason for this can be foraging behaviour that is often linked to certain habitats such as plantations (*viz*., dwarf antelope *Neotragus batesi* and talapoin *Miopithecus ogouensis*), agriculture (*viz*., marsh cane rat *Thryonomys swinderianus*, brush-tailed porcupine *Atherurus africanus* and yellow-backed duiker), roads (*viz*., civet *Civettictis civetta*) and settlements (*viz*., talapoin and cane rat). Nine others were expected to be more abundant inside the CMNP: red-capped mangabey (*Cercocebus torquatus)*, grey-cheeked mangabey (*Lophocebus albigena*), black colobus (*Colobus satanas*), golden cat (*Felis aurata*), water chevrotain (*Hyemoschus aquaticus*), leopard (*Panthera pardus*), chimpanzee (*Pan troglodytes*), giant pangolin (*Smutsia gigantea*) and elephant. These species were expected to be sensitive to disturbances and hunting, or dependent on primary forest. These *a priori* assumptions led to one-sided t-tests for the above-mentioned 15 species. The test was performed two-sided if no difference was expected.

The pooled densities were extrapolated for the entire study area. Densities are estimates, and thus indices; therefore the geometric mean was calculated for each species for all sample areas together. The data were root-transformed to correct for overestimates, resulting in a normal distribution. The final density per species for the total study area was calculated after re-transforming the mean density of each species.

The density estimate for the whole study area was used to estimate population sizes if there was no significant difference between inside and outside the CMNP. In the case of significant differences, the densities for inside and outside the CMNP were used separately for further analysis. Resulting population sizes are estimates and were therefore rounded in the following way: results from 0 - 100 were rounded to the nearest 10; 100 – 5,000 to the nearest 100; 5000 - 20,000 to the nearest 500, and > 20,000 to the nearest 1,000.

All abundance estimates used to advise project management were based on the lower confidence limits of the data, following the precautionary principle (Foster *et al*. 2000). Using these lower confidence levels allows for a certain degree of security when setting hunting quota or taking conservation measures.

Table 1: Medium to large wildlife species used in this study. Scientific and English nomenclature based on Kingdon (1997), local names obtained from interviews.

Common name	Scientific name	Mvai	Bulu	Baka	Yassa	Mabea
Ungulates						
Blue duiker	*Cephalophus monticola*	Okpwen	Opwang	Koan	Vieli	Kwan
Peters' duiker	*C. callipygus*	Mvin	Mvin	Mvindi		Mvindi
Bay duiker	*C. dorsalis*	Sô	Sô	Lékeno	Esibo	Kiana
Black-fronted duiker	*C. nigrifrons*	Angbwan	Zoum		Muchèbe	
Yellow-backed duiker	*C. silvicultor*	Zip	Zip	Zibo		Ziou
Sitatunga	*Tragelaphus spekei*	Emvou	Emvoul	Moulé		Emvoul
Water chevrotain	*Hyemoschus aquaticus*	Viong	Vion	Yén		Yong
Dwarf antelope	*Neotragus batesi*	Ojoué	Odjoé	Nojoé	Ojoi	Ojoi
Forest elephant	*Loxodonta africana cyclotis*	Zock	Zok	Diogo		Jouer
Forest buffalo	*Syncerus caffer nanus*	Nyat	Nyat	Nyadé		Nyari
Red river hog	*Potamochoerus porcus*	Ngoé Afane	Ngoé Afane	Ngo		Ngou
Primates						
Moustached monkey	*Cercopithecus cephus*	Ossôk	Ossôk		Songo	Cho’ou
Putty-nosed monkey	*C. nictitans*	Avembe	Avembe	Yembe	Ebondi	Yombo
Crowned monkey	*C. pogonias*	Essouma	Essouma	Pundé	Mpinde	Pundi
De Brazza's monkey	*C. neglectus*	Foung	Foung	Poum		Fang
Northern Talapoin	*Miopithecus ogouensis*	Ozem, Azem	Ozeme	Njingi		Njingi
Red-capped mangabey	*Cercocebus torquatus*	Eka-Foum	Ekaafum		Mosako	Nso’oe
Grey-cheeked mangabey	*Lophocebus albigena*	Kak	Kak	Nsa’ag		Boundi
Black colobus	*Colobus satanas*	Mvon	Mvon			
Mandrill	*Mandrillus sphinx*	Seuk, Zombo	Sêk			Scha’a
Chimpansee	*Pan troglodytes*	Wo’o	Waa	Wa’aga	Wo’a	Wo’a
Gorilla	*Gorilla gorilla*	Ngui	Ngui	Wangui		Ngui
Carnivores						
Leopard	*Panthera pardus*	Ze	Ze	Ndié		Njie
Golden cat	*Felis aurata*	Etoungou	Ebio	Rebué		Ebiok
Swamp otter	*Aonyx congica*	Abang	Abang	Léban		Bong
Spot-necked otter	*Lutra maculicollis*	Abang	Abang	Gya	Ilondo	Bong
African civet	*Civettictis civetta*	Zoé	Zoé	Gyobo	Zoi	
Diverse						
Giant pangolin	*Smutsia gigantea*	Avil, Zokka	Avii	Vima		
Brush-tailed porcupine	*Atherurus africanus*	Ngom	Ngom	Ngombo		Kumbo
Marsh cane rat	*Thryonomys swinderianus*	Mvep	Mvep	Mvegé		Mvep
African dwarf crocodile	*Osteolaemus tetraspis*	Nkom				Cumbi
Python	*Python sebae and P. regius*	Mvom				Mbo’ama
Monitor lizard	*Varanus niloticus*	Nka'a				Nkombi

Cost and time

The case study, in which the PLEO method was tested, took approximately six weeks of fieldwork: three interview sessions of two weeks each. Data entry and analysis took one month. Hence for an area of about 500,000 ha, the total interview time and wildlife status assessment took about 10 weeks (2.5 months). Training of staff was not included; therefore, when it is the first time for a project to apply this method, extra time should be allocated for training.

Costs were far less than what was involved in transect surveys. In terms of personnel for the PLEO method, one interviewer (also project leader), an assistant, a driver and an occasional guide or translator were needed. The highest cost was the salary of the interviewer. Apart from the wages, there were “motivational costs” for the respondents, as they were kept from conducting their daily business. Further costs included transport, accommodation and overhead. The total costs of the whole assessment amounted to approximately 10,000 € (= 7,000,000 FCFA: exchange rate 700 FCFA = 1€).

Line transect surveys involved three teams in the UTO, deployed for one year, under supervision of a wildlife expert. Each team consisted of one assistant-researcher, one guide, one guide/trail maker, a cook and a driver. The costs per month for the three teams together (including the costs of the supervision) were about 5,700 € (4,000,000 Fcfa). For full coverage in time and space, the sampling was continued for one year, including data analysis, amounting to a total cost of approximately 68,000 € (12 x 5,700). Obviously, the costs involved in assessing wildlife densities by means of PLEO were far lower than the conventional line transect method.

Results

The most abundant ungulate was blue duiker (*Cephalophus monticola*) with a density of 19.5 animals per km^2, and the least abundant was elephant, outside as well as inside the CMNP: 0.02 and 0.31 animals per km^2 (Table 2). Buffalo were also rare: 0.2 animals / km^2.
Of the primates, putty-nosed monkey (*Cercopithecus nictitans*) was most abundant (9.2 animals per km^2). The rarest primate was de Brazza’s monkey (*Cercopithecus neglectus*): 0.07 animals/km^2, followed by the black colobus (1.61 in CMNP, and 0.29 outside CMNP, Table 2).

No species that was expected to show no differences between the densities inside and outside the Park, showed significant differences. Of the fifteen species for which *a priori* differences were expected between inside and outside the CMNP, eight species showed significant differences. The species that did differ significantly, showed mostly higher densities within the CMNP than outside: water chevrotain; elephant; red-capped and grey-cheeked mangabey; black colobus; leopard and golden cat (Table 2). Only the plantation-preferring dwarf antelope was more abundant outside than inside CMNP, as was expected.

The PLEO results were tested in two ways. We first compared the densities of species found in this study against the densities found in the relevant literature. It was expected that the results of this study should not differ from densities found in other studies in the region. Densities of all studied species (if available; see Table 3) were compared with results of the same species found in the literature (no significant difference: Sign test: 2-tailed, p=1.000, n= 85; see Table 3). Second, the density found in this study was plotted against body size of the same species. A linear relation with a negative correlation was expected, where large body size should correspond with low density. The relation between body size and density is clearly indicated by the trend line in figure 1. Density decreases with increasing body size (Spearman rank correlation coefficient r=-0.36, p<0.05, n=33).

Estimated population sizes of the 33 animal species are presented in Table 4. The three most common species were: blue duiker: 138,000 animals, putty-nosed monkey: 65,000 animals, and brush-tailed porcupine: 122,000 animals. Mandrill population size was estimated at 23,000. The forest "big five" population sizes in the study area were: buffalo 1,400 animals; leopard 660 animals (inside plus outside the CMNP); gorilla 7,500 animals; chimpansee16,000 animals and elephant 880 animals (inside plus outside the CMNP).

Table 2: Wildlife densities (d in animals/km^2) inside and outside CMNP, Cameroon, as calculated through PLEO. cl = 90% confidence levels. The species marked with * differed significantly between inside and outside the CMNP, t-test $P<0.05$, $n=34$. Species marked with > were tested one-sided (see Analysis).

	Animals/km^2						**t-test in–out CMNP**
	Inside CMNP		**Outside CMNP**		**Total study area**		
Ungulates	d	cl	d	cl	d	cl	p-value
Blue duiker	18.83	10.18-30.12	19.8	13.58-27.18	19.51	14.48-25.28	0.8
Peters' duiker	3.19	0.88-6.95	2.02	1.03-3.32	2.33	1.36-3.57	0.36
Bay duiker	7.85	4.19-12.65	8.48	5.42-12.21	8.29	5.89-11.09	0.65
Black-fronted duiker	1.01	0.11-2.79	0.59	0.19-1.21	0.7	0.29-1.27	0.55
>Yellow-backed duiker	3.16	0.86-6.89	5.03	2.77-7.96	4.43	2.69-6.60	0.25
Sitatunga	5.46	2.16-10.28	4.76	3.22-6.60	4.96	3.51-6.66	0.53
>Water chevrotain*	2.09	0.26-5.67	0.65	0.34-1.07	0.99	0.48-1.69	0.02
>Dwarf antelope*	0.7	0.09-1.87	4.37	2.20-7.29	2.96	1.55-4.84	0.04
>Forest elephant*	0.31	0.09-0.67	0.02	0.00-0.05	0.07	0.02-0.14	0
Forest buffalo	0.49	0.07-1.27	0.11	0.02-0.27	0.2	0.07-0.38	0.15
Red river hog	5.9	2.88-9.99	4.29	2.56-6.47	4.74	3.22-6.55	0.73
Primates							
Moustached monkey	7.82	3.54-13.79	7.59	4.99-10.73	7.66	5.44-10.25	0.93
Putty-nosed monkey	9.12	4.70-14.99	9.17	5.86-13.23	9.16	6.49-12.28	0.84
Crowned monkey	6.67	3.92-10.15	6.91	4.33-10.10	6.84	4.85-9.18	0.65
De Brazza's monkey	0.09	0.00-0.27	0.06	0.00-0.19	0.07	0.01-0.17	0.68
>Northern talapoin	5.86	1.13-14.29	4.52	1.80-8.48	4.9	2.43-8.22	0.39
>Red-capped mangabey*	5.91	2.57-10.63	1.67	0.95-2.60	2.65	1.64-3.90	0
>Grey-cheeked mangabey*	1.87	0.93-3.15	0.56	0.25-0.99	0.87	0.51-1.32	0.02
>Black colobus*	1.61	0.59-3.14	0.29	0.09-0.62	0.57	0.27-0.98	0.01
Mandrill	4.03	1.81-7.11	2.88	2.07-3.83	3.2	2.36-4.17	0.21
>Chimpanzee	3.25	1.40-5.88	1.92	1.11-2.93	2.27	1.49-3.22	0.17
Gorilla	1.8	1.06-2.74	0.8	0.41-1.31	1.05	0.68-1.50	0.26
Carnivores							
>Leopard*	0.22	0.02-0.60	0.04	0.01-0.08	0.07	0.03-0.15	0.03
>Golden cat*	0.09	0.01-0.24	0.01	0.00-0.02	0.02	0.01-0.05	0.03
Swamp otter	1.14	0.27-2.59	0.88	0.37-1.62	0.95	0.49-1.56	0.88
Spot-necked otter	1.3	0.35-2.87	0.8	0.31-1.53	0.94	0.47-1.56	0.74
>African civet	0.23	0.06-0.49	0.38	0.15-0.74	0.33	0.16-0.57	0.18
Diverse							
>Giant pangolin	1.22	0.61-2.03	1.49	0.87-2.29	1.41	0.94-1.98	0.2
>Brush-tailed porcupine	16.32	6.43-30.72	17.61	11.23-25.42	17.23	11.85-23.61	0.46
>Marsh cane rat	0.2	0.01-0.98	0.55	0.14-1.23	0.43	0.13-0.90	0.32
African dwarf crocodile	2.45	0.91-4.75	3.24	1.70-5.28	3	1.82-4.47	0.57
Python	1.11	0.07-3.35	1.05	0.52-1.75	1.07	0.55-1.75	0.51
Monitor lizard	2.19	0.86-4.13	2.85	1.80-4.16	2.65	1.81-3.66	0.64

Table 3: Wildlife densities in present study (in animals/km^2), in other studies in the Campo-Ma'an area, in Cameroon and in the West and Central African region. *=densities for outside and inside (respectively) of the Campo-Ma'an National Park (t-test, $p<0.05$, n=34).

Species	Density in Campo-Ma'an area (N/km^2)		Density for West and Central Africa, found in literature (N/km^2)			
	This study	Other studies	Cameroon	Gabon	C.A.R	Other
Ungulates						
Blue duiker	19.51		2.78[7];	0.539	12.45/18.7[10]	24.2[13]; 10.2-20.6[14]; 30.4-53[13]
Peters' duiker	2.33		8.48[8]; 7.35[7]		1.06/1.12[10]	
Bay duiker	8.29		7.35[7];		1.51/0.26[10]	1.9-2.7[14]
Black-fronted duiker	0.7		7.357			1.3-2[14]
Yellow-backed duiker	4.43			0.219		1.6[13]; 0.5-1.6[14]
Sitatunga	4.96					
Water chevrotain	0.65 / 2.09*			0.539		1.414
Dwarf antelope	4.37 / 0.70*			0.539		
Forest elephant	0.02 / 0.31*	0.27[5]; 0.18/0.95[6]		0.679	0.8611	0.01-0.02[15];
Forest buffalo	0.2	0.045		0.519		0.27-0.67[15]
Red river hog	4.74			0.869		
Primates						
Moustached monkey	7.66	8.43[1]; 13.6[2]; 29.8[3]	5.82/13.2/16.7[8]			
Putty-nosed monkey	9.16	32.52[1]; 24.3[2]; 41.5[3]	8.94/9.59/15.14[8]			
Crowned monkey	6.84	13.54[1]; 13[2]; 13.8[3]	4.81/4.89/5.62[8]			
De Brazza's monkey	0.07					0.416
Northern talapoin	4.9					
Red-capped Mangabey	1.67 / 5.91*	13.69[1]; 9.7[2]				
Grey-cheeked mangabey	0.56 / 1.87*	2.721				
Black colobus	0.29 / 1.61*					7.5/30-33[17]
Mandrill	3.2	4 - 27[2]				
Chimpanzee	2.27	0.784		0.079	0.01-0.13[11]; 0.16[12]	0.02-0.2[15]; 0.03-0.36[13]; 2.8-4.71[18]
Gorilla	1.05	0.24		0.29	0.89-1.45[11]; 1.5/1.6[12]	

Table 3 continued.

Species	**Density in Campo-Ma’an area (N/km^2)**		**Density for West and Central Africa, found in literature (N/km^2)**			
	This study	**Other studies**	**Cameroon**	**Gabon**	**C.A.R**	**Other**
Leopard	0.04 / 0.22*			0.29		1.3[19]; 2[15]
Golden cat	0.01 / 0.09*			0.079		
Swamp otter	0.95					
Spot-necked otter	0.94					
African civet	0.33			1.09		
Diverse						
Giant pangolin	1.41					
Brush-tailed porcupine	17.23			0.69	3.081	5513
Marsh cane-rat	0.43					
African dwarf crocodile	3					
Python	1.07					
Monitor lizard	2.65					

References: 1: Mitani (1991); 2: Campo area, calculated from Matthews and Matthews (2002); 3: Ma’an area, calculated from Matthews & Matthews (2002); 4: Matthews & Matthews (unpublished data); 5: Bekhuis (unpublished data); 6: Ngandjui (unpublished data); 7: Fimbel *et al*. (2000), red duikers together: 7.35; 8: Ngandjui & Blanc (2000); 9: Prins & Reitsma (1989); 10: data from net hunts and line transects, respectively, Noss (2000); 11: Carrol (1986); 12: Blom *et al*. (2000); 13: data on West Africa were left out, Eves & Bakarr (2001); 14: for areas of no hunting, Hart (2000); 15: Ivory Coast, Caspary *et al*. (2001); 16: Thomas (1991); 17: Cameroon data left out, because too old, Fleury & Gautier-Hion (1999); 18: Hashimoto (1995); 19: Plumptre & Reynolds (1994) and 20: Furuichi *et al*. (1997).

Table 4: Total population size (N) per species in Campo-Ma'an area, Cameroon. lcl = lower confidence level (90%). Lcl is mentioned for management purposes. Species marked with an asterix (*) show significant difference in densities between inside and outside the CMNP, t-test P<0.05, n=34. Species with population sizes beneath the MVP are in italics (see Discussion).

	Inside CMNP		**Outside CMNP**		**Total study area**	
	N	lcl	N	lcl	N	lcl
Ungulates						
Blue duiker					138,000	102,000
Peters' duiker					16,500	9,500
Bay duiker					59,000	42,000
Black-fronted duiker					4,900	2,100
Yellow-backed duiker					31,000	19,000
Sitatunga					35,000	25,000
Water chevrotain*	5,500	700	2,900	1,500	8,400	
Dwarf antelope*	1,800	200	19,500	10,000	21,300	
Forest elephant *	800	200	80	10	880	
Forest buffalo					1,400	500
Red river hog					33,000	23,000
Primates						
Moustached monkey					54,000	38,000
Putty-nosed monkey					65,000	46,000
Crowned monkey					48,000	34,000
De Brazza's monkey					500	100
Northern talapoin					35,000	17,000
Red-capped mangabey*	15,500	6,500	7,500	4,000	23,000	
Grey-cheeked mangabey *	4,900	2,400	2,500	1,100	7,400	
Black colobus *	4,200	1,500	1,300	400	5,500	
Mandrill					23,000	17,000
Chimpanzee					16,000	11,000
Gorilla					7,500	4,800
Carnivores						
Leopard *	600	60	170	40	770	
Golden cat *	200	30	30	0	230	
Swamp otter					6,500	3,500
Spot-necked otter					6,500	3,300
African civet					2,400	1,100
Diverse						
Giant pangolin					10,000	6,500
Brush-tailed porcupine					122,000	84,000
Marsh cane rat					3,000	900
African dwarf crocodile					21,000	13,000
Python					7,500	3,900
Monitor lizard					18,500	13,000

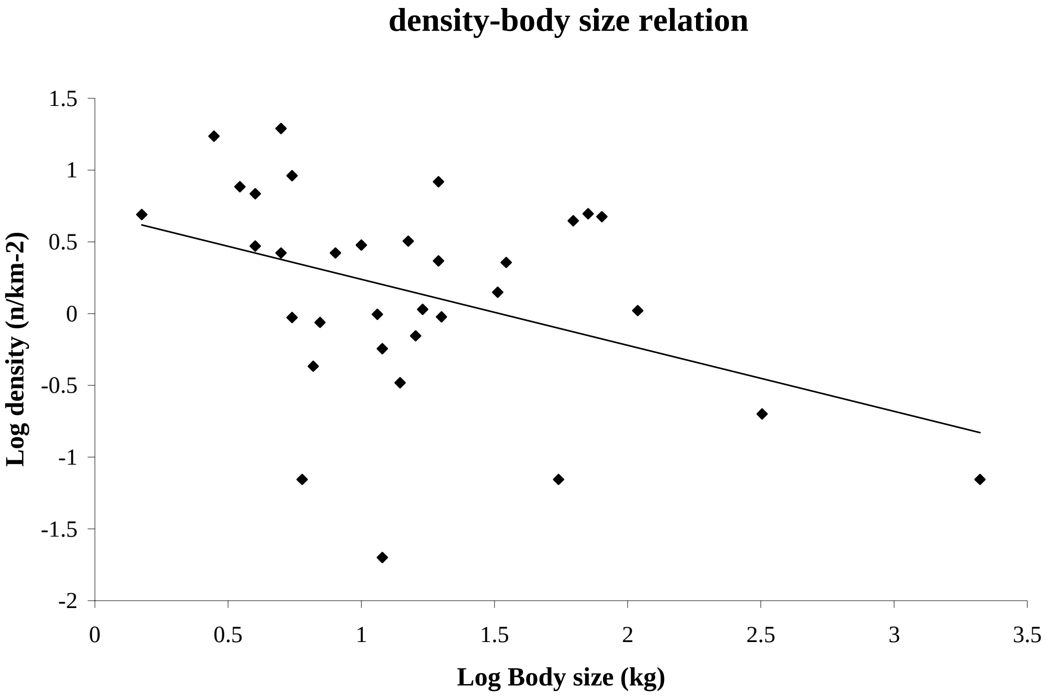

Figure 1: Relation between population densities obtained by PLEO and body size for wildlife species in the Campo-Ma'an area, Cameroon (Spearman rank correlation coefficient r= -0.36, p<0.05, n=33).

Discussion

The aim of this study was to test whether the PLEO method yielded results similar to those of classical methods in wildlife density estimates. Unfortunately, the line transect surveys that were conducted simultaneously with the PLEO produced too few data to calculate densities, except for elephant and buffalo. Indicators of abundance were given instead, which undermined comparison between the two methods in the study area. This is a major weakness in transect surveys: although it cost seven times more and took nine months more than PLEO, it did not produce more than density estimates for elephant and buffalo. As a consequence the comparisons were made with earlier studies in the area (Mitani 1991; Matthews & Matthews 2002 and some unpublished data), and with other studies in the region. Based on these comparisons, we conclude that the PLEO method is substitutable for line transect surveys. However, for ideal comparison, a calibration is needed. This could not be performed in the present study, due to the lack of reliable line transect data, but should be done in order to eliminate any doubts about the validity of the PLEO method. It is remarkable that surveying over twenty transects twelve times each did not produce enough data to estimate wildlife densities. This makes calibration with line transects as accepted wildlife estimators difficult, and stresses the need for more cost-effective field methods.

Of the 85 samples, 42 PLEO-based densities were lower, and 43 were higher than results of other studies; this showed that the PLEO results fell well within the range of estimates by conventional methods (2-tailed Sign test: p=1.000, n= 85). The density obtained by PLEO was compared consecutively with each density found in literature. As a consequence multiple comparisons per species occurred, resulting in a sample size of 85, whereas only 24 species were studied in this test. To make sure this didn't produce a bias we repeated the test with the densities found in the literature, which were averaged per species (resulting n=24). In this case no difference was found either (2-tailed Sign test: p=0.152, n=24)

The correlation between body size and density is as would be expected (Damuth 1981; Fa and Purvis 1997), and is an additional support to the acceptance of the method (Figure 1). This correlation shows that the results are consistent, although systematic over or underestimates are not uncovered.

For elephant and several other species sufficient data from other studies were found in the literature to discuss them in more detail (Table 3).

The elephant density of 0.31 animals/km^2 in the CMNP was similar to the results of transect surveys, which resulted in a density of 0.27 animals/km^2 for the southern part of the CMNP (Bekhuis unpublished data). The study of Ngandjui (unpublished data), based on line and recce transect sampling in the same area, resulted in a density of 0.95 animals/km^2. During transect cutting in 1999, Ngandjui found for the southern part of the CMNP a density of 0.18 animals/km^2. A problem is that not all transects were covered, and the surveys were done in one single period (November/December 2001). The only valid comparison is therefore with the results of Bekhuis. Other elephant densities per km^2 in Cameroon range from similar: 0.3 for Boumba-Bek (Ekobo, unpublished data) and 0.56 for Dja (Williamson and Usongo, unpublished report MINEF, project ECOFAC Cameroon 1995), to 1.75 (Wildlife Conservation Society, unpublished data) and 2.17 (Ekobo 1995) for Lobéké National Park. The Lobéké area is in the extreme east of Cameroon, were human population densities are low, which probably explains the higher densities. Dja is more to the east as well, and less populated than the Campo-Ma'an area, but is closer to large cities (e.g., Yaoundé) than Lobéké, which would explain the intermediate density. Examples of elephant densities in neighbouring countries are 0.67 for Gabon (Prins & Reitsma 1989) and 0.86 in Central African Republic (Carrol 1986). The density of human settlements and proximity to urban areas can probably explain the low densities found in the Campo-Ma'an study area compared to the other sites.

Buffalo density calculated here was lower than the three other known studies: 0.20 animals/km^2 for this study against 0.51 animals/km^2 for Gabon (Prins & Reitsma 1989), 0.42 animals/km^2 for the Lopé reserve in Gabon (White 1992 in: Blake 2002), and 0.27 – 0.67 animals/km^2 for Ivory Coast (Caspary *et al.* 2001). One of two transect surveys carried out in the CMNP showed a much lower density: 0.04 animals/km^2 (Bekhuis, unpublished data), the other transect survey produced insufficient data to calculate densities. Prins & Reitsma (1989) found the buffalo density in Gabon to be low, and ascribed this to the low amount of palatable grasses. The study by Blake (2002) in Congo showed that buffalo prefer open swamp vegetation and clearings, and abundance was associated with watercourses. His conclusion was that buffalo abundance was linked with highly patched food distribution, and much less on *terra firma* forest (Blake 2002). The absence of any clearings with palatable grasses in the Campo-Ma'an area could be the cause of the low buffalo density, but perhaps also the influence of human pressure, as is suggested for elephant.

It should be noted that it is notoriously difficult to estimate buffalo density in rainforests. This is especially the case with transect surveys, as buffalo prefer old logging roads and clearings for foraging, since they feed mainly on grass (Kingdon 1997; Blake 2002), and use elephant trails and (old) logging roads to move from one food patch to another (Blake 2002). This could result in an underestimation of buffalo density, because line transects rarely follow trails or roads, as they should be placed at random.

The density of 1.05 animals/km^2 is higher than what Matthews & Matthews (unpublished data) found in 2000, for the same area based on transect surveys (0.2 animals /km^2), but lower than studies in eastern Cameroon (1.71-2.89 animals /km^2, Wildlife Conservation Society and WWF, unpublished data). For the increasing gorilla density towards the east of the country the same explanation can be suggested as for buffalo and Elephant: the reducing human population pressure from east to west. In comparison with neighbouring countries, the gorilla density found in this study is average (Table 3).

Gorilla density is dependent on forest type, with a preference for secondary forest (Tutin & Fernandez 1984), but in this study no significant difference was found between inside the CMNP and its surroundings. This is probably due to the recent creation of the National Park, in 2000, and its history as a logging concession before that. Secondary forests are therefore still abundant within, as well as outside the CMNP.

Chimpanzee density in the Campo-Ma'an area was high (2.27 animals/km^2) compared to earlier estimates in the same area (Matthews & Matthews unpublished data): 0.78-0.86 animals/km^2. Only one study in Uganda showed a higher density, 2.8–4.7 animals/km^2 (Hashimoto 1995). Prins & Reitsma (1989) ascribed the low density of chimpanzees based on their data in Gabon (0.07 animals/km^2) to the time of year they collected the data. According to them chimpanzee lived more in the canopy during the rainy season, which was when they did their survey. This is overcome in our study, as the data are based on interviews, and not transects, and cover both wet and dry seasons.

Leopard was rare in the Campo-Ma'an area with 0.07 individuals/km^2. Prins & Reitsma (1989) found a density of 0.2 animals per km^2 in Gabon, while in Ivory Coast the density was a factor 30 higher: 2 animals/km^2 (Caspary *et al*. 2001). Matthews & Matthews (unpublished data) already remarked that leopards were rare in the UTO, but no density was mentioned. The high level of snaring in the area could be the reason for low leopard density. Hunters frequently set several hundred snares each, which they visit every few days. Leopard often get caught in snares set for other game.

Data on mandrill densities are sparse, and the study area is the only place where the mandrill is protected in Cameroon, which accounts for the special attention given to this species. However, no difference was found between the CMNP and its surroundings, resulting in a density of 3.2 animals per km^2 for the entire study area.

The study of Matthews & Matthews (2002) resulted in higher density estimate of 4 - 27 mandrills/km^2 for the southwest part of the study area, for the rest of the study area not enough data were available. Abernethy *et al*. (2002) argued, based on film records, that mandrill group sizes in other studies are probably greatly underestimated. Observers were generally able to only count 60 % of the true group size shown on film. The density of 3.2 animals per km^2 found in this study is in that case not high: 23,000 individuals (Table 4) with an average of 600 individuals per group results in a total of 35 groups for the study area of 500,000 ha.

Cane rat (0.43 animals/km^2) and civet (0.33 animals/ km^2) are relatively rare, since their habitat is predominantly near fields, villages and roads, while the study dealt with areas located further in the forest interior.

Most species that are threatened are so because they have small population sizes, or show a rapid decline in population numbers (Hilton-Taylor 2000). A reference is needed to define the level of threat. The Minimal Viable Population (MVP, Soulé 1987) was used in this study, and was set at five thousand animals for all species concerned.

Eleven species had a population size beneath the critical MVP of 5000: black-fronted duiker *Cephalophus nigrifrons* (4900), grey-cheeked mangabey (both inside, as well as outside CMNP: 4900 and 2500, respectively), black colobus (4200 and 1300), elephant (800 and 80), buffalo (1400), water chevrotain outside the CMNP (2900), de Brazza's monkey (200), leopard (600 and 60), golden cat (200 and 30), civet (2400) and cane rat (3000).

For the UTO the mentioning of species with a population size beneath this MVP is important, as the UTO is an isolated area relative to other rainforest areas in the region. Neglecting the importance of this isolation, coupled with the small population sizes can have as a consequence the extirpation of the above-mentioned species from the UTO. Although gene flow is possible, replenishment of dwindling populations by immigration is not likely to occur, and local extinction due to chance events can be envisaged with such small populations.

These results show that the Campo-Ma'an area received National Park status just in time. Even though the area has a protection status, it may be too small for some of these species. This calls for the integrated management of the CMNP and its surrounding areas.

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Next page: male mandrill

Chapter 3

THE ASSESSMENT OF VULNERABILITY OF WILDLIFE FOR SETTING CONSERVATION PRIORITIES

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Abstract

To rank wildlife species according to their vulnerability to local extinction is difficult, because the quantification of risk proneness has been developed mainly on a global scale. Too often this actual extinction risk of species is used for setting local conservation priorities. We present a new priority setting method that ranks animals in a rainforest according to their local vulnerability to extinction or to major population declines. It is a fast, efficient and cost-effective method to set priorities in conservation management. Information from literature and local knowledge from hunters was used to assess the vulnerability of 33 wildlife species. The result is a vulnerability list, where species are ranked according to their vulnerability to major declines and extinction. To produce this list we developed a system where the vulnerability of each species was determined on the basis of twelve factors. These factors were assumed to be of importance to the survival of a population, and were scored with information from interviews with local hunters, and from the literature. The method was used in a case study in the Campo-Ma'an area in south Cameroon, Africa. In this study the most vulnerable species was the mandrill, followed by the elephant, chimpanzee, and buffalo. The listing of the most threatened species in the IUCN Red list was confirmed for the Campo-Ma'an area. This study illustrates the limitation of using solely the extinction risk of species for setting local conservation priorities by illustrating the discrepancy between the results of local listing and the IUCN Red listing. This emphasizes the need for local data for an adequate assessment of vulnerability of wildlife on a regional or local scale. We argue that this method provides a useful tool for managers to allocate time and money to the species that need them most.

Introduction

The assessment and subsequent monitoring of animal populations is the foundation of effective wildlife management. Accepting that it is impossible to monitor all species present in a tropical rain forest due to cost and time constraints, we developed a new priority setting method that selects species for further monitoring in a reliable but time and cost-effective way. Adopting international accepted assessment methods may seem the obvious way to follow, such as the IUCN Red List of threatened species (IUCN 2006). However, we propose a method that focuses on the local scale, where intrinsic factors known from literature are combined with extrinsic factors that are site-specific and which are defined by local people. The arguments for this choice are that existing assessments often are tailored either for specific species (e.g., Zhou & Pan 1997), or cannot be carried out by lack of data or resources (O’Grady *et al*. 2004), or are based on global status without consideration of the regional or local situation (Breininger *et al*. 1998; Rodríguez *et al*. 2000). Furthermore, the listing by the IUCN is a process that aims at quantifying the probability of a species going extinct, which in itself is not enough to set conservation priorities, although in practice this happens often (Miller *et al*.2006).

Identification of threatened flagship species to highlight the conservation value of an area is not what local conservation practitioners need, however: what they do need is an assessment that ranks most animals in their area according to their need for direct management priorities and conservation efforts, as the threat situation of local species is often not covered in existing risk assessments (e.g., Costa *et al*. 2005; Miller *et al*. 2007). The urgency for rapid vulnerability assessments which circumvent lengthy and costly species-specific studies (O’Grady *et al*. 2004) becomes therefore all the more pertinent. As “extinction risk” is a probabilistic statement that must be made within a specified time frame, and should consider the spatial and temporal scale of threats that a species is exposed to, the IUCN Red Lists alone cannot be used for setting conservation priorities (Mace & Lande 1991; Miller *et al*. 2006, 2007). The vulnerability assessment in this study is specifically meant for priority setting in local conservation activities, and not for quantification of the actual extinction risk. The ultimate goal of this study was to develop a rapid and cost-effective method that provides a comprehensive vulnerability list of site-specific wildlife species. With this list, which ranks species according to their vulnerability to rapid decline or local extinction, selection of target species for detailed monitoring and allocation of funds for conservation efforts can be well-argued.

The susceptibility of wildlife populations to major declines and (local) extinction partly determines the conservation attention they require (Lande 1998). To determine this vulnerability is therefore a way to set priorities in conservation efforts (c.f., Breininger *et al*. 1998). There are many factors that define the vulnerability of a species to rapid decline or extinction. Most of the factors are controlled by human influence, whether direct (e.g., hunting: Bodmer *et al*. 1997) or indirect (e.g., habitat alteration, Brooks *et al*. 2002). The importance of human influence is assumed to be appropriate because most extinctions are ultimately caused by anthropogenic factors (Diamond *et al*. 1989; Forester & Machlis 1996; Lande 1998). The selection of factors that define the vulnerability of species is therefore commonly based on ecological, anthropological and socio-economic grounds (Breininger *et al*. 1998; Purvis *et al*. 2000; O’Grady *et al*. 2004). However, a division can be made in intrinsic and extrinsic factors, where the intrinsic factors describe species traits that are globally recognised, and extrinsic factors describe traits that are locally dependent.

We developed a system where the vulnerability of each species was assessed on the basis of twelve factors. These factors were assumed to be of importance to the survival of a population. Selection of these factors was done based on their proven influence, but also on their relative ease of assessment. The intrinsic factors were assessed from the literature. The extrinsic factors determining site-specific vulnerability were scored by local hunters. Additionally, international experts on wildlife conservation in tropical rainforests were asked to weight each vulnerability factor per species. Ranking of the species according to the cumulative vulnerability produced the final listing. We argue that this method of ranking species according to their vulnerability facilitates setting local priorities for conservation, thereby enabling wildlife managers to allocate time and money to the species that most need it.

Methods

Study site.

The integrated management area of the Campo-Ma'an Management and Biodiversity Conservation Project, in the south of Cameroon, is covered mainly with Guinea-Congolian rainforest. The altitude ranges from 0 to 1,100 m; annual rainfall varies from 1,670 to 2,950 mm. There are 2300 known plant species of which 29 are endemic to the area (Tchouto *et al.* 2006), 80 mammal species, and it is one of the richest reptile areas in Africa (De Kam *et al.* 2002).

The integrated management area is called the "Operational Technical Unit" (OTU). The OTU was created in August 1999 and covers an area of 7,770 km^2. In January 2000, the Campo-Ma'an National Park (CMNP) was created within the OTU, and consists of 2,640 km^2, or 34% of the area. The OTU also contains four logging concessions (31%), an agro-forestry zone (26%), rubber and oil palm plantations (7%) and a coastal strip (<1%).

There are 59,000 people living in the OTU, 24,000 of these live in the rubber- and oil palm concessions. Those who do not work in the plantations or logging companies, and who are not fishermen, are subsistence hunters or farmers. These hunters and farmers obtain more than 75% of their animal proteins from bushmeat (Annaud & Carriere 2000).

Methods

Assessing the local vulnerability of wildlife species involved several stages. First, species were selected that would be included in this study. Second, the factors determining the vulnerability were selected. Third, a decision was made as to which factors could be quantified from literature sources (intrinsic factors), and which factors by extracting local knowledge through interviews (extrinsic factors). Some site-independent factors are intrinsically linked to vulnerability (e.g., reproduction), and individual species could be scored through interpretation of the literature. Other factors, shaped by human preferences (e.g., taste), were dependent on the local situation and were scored through interviews with hunters. Fourth, a panel of international experts was individually asked to weight the factors for their importance to the survival of the different species, this information was then used to adjust the factor scores. Finally, all species were ranked according to their overall calculated vulnerability value. In the text we use the term "population" rather than species, as this study concerns "populations" in a well-defined area.

Interviews and literature study took place in 2001. Most medium to large-bodied mammals (n=30), which were known to occur in the area plus three reptile species were selected for analysis, totalling 33 wildlife species. For the species data on the intrinsic factors a literature review was conducted. Several reference works on African mammals provided most data (Estes 1991; Rowe 1996; Harvey *et al.* 1987). These reference works are compilations of peer-reviewed publications on species or species groups. We used the ISI Web of Knowledge™ and a web-based search engine (Google scholar) to perform specific searches for peer-reviewed articles on each species to verify if the data were up-to-date, and to provide data that were still missing. If no information was found in the reference works or through the literature research, we estimated the data by comparing with similar species.

Selecting and scoring of factors.

Twelve factors were selected that affect vulnerability. The six intrinsic factors were quantified with the most recent information available in the literature (Table 1) and described below. Factors were assigned a score from one (lowest vulnerability) to ten (highest vulnerability) when discrete and were categorised in classes when continuous (Table 2). Each class was then scored from one to ten.

Life span. Species are assumed to be more sensitive to disturbances when they are long-lived. Species with “slow” life histories cannot compensate for high mortality with increased fecundity (Purvis *et al.* 2000). This is illustrated by a study by Bodmer *et al.* (1997) who discovered that, under the same hunting pressure, mammals with greater generation times showed greater declines in abundance than species with shorter generation times. Species with a long life span are therefore considered more vulnerable. Maximum life span was obtained from the literature.

Reproduction. If reproduction in a declining population is slow, recovery will be slow also, and disturbances or over-exploitation can topple a population towards extinction. Species with low fecundity are frequently threatened by overharvesting (Fitzgibbon *et al.*1995; Bodmer *et al.* 1997; Lande 1998) and are considered to be more vulnerable than rapidly reproducing species (Purvis *et al.* 2000). In this study, reproduction is averaged over four components: gestation period, litter size, birth interval, and age of first reproduction.

Social organisation. To live solitary or in groups influences detectability for hunters and survival after detection (Fitzgibbon *et al.* 1995). More individuals are killed when a group living species is detected by a hunter, whereas solitary animals provide only one prey. The anti-predation strategy of animals (i.e., living in groups; van Schaik & van Noordwijk 1983) does not work when hunted by humans; for example, the hunting pressure on group living primates is higher than on solitary duikers (Bodmer *et al.* 1988 in: Fitzgibbon *et al.* 1995). Furthermore, species living in groups are more vulnerable to rapid decline or extinction than are solitary animals because persistence depends on a larger unit than the individual (Stephens *et al.* 2002; Purvis *et al.* 2000). Social organisation is defined as living solitary, in pairs, or in groups.

Home range. The size of the home range of a species influences the sensitivity of the species to over-harvesting, habitat fragmentation, decline of the total habitat area, natural disasters, etc. (Root *et al.* 1988; Purvis *et al.* 2000). Animals with small home ranges can survive in small relic populations and are therefore considered less vulnerable than are animals with large home ranges.

Table 1: Factors defining wildlife vulnerability, scored with information from the literature and interviews. Low vulnerability received low scores; high vulnerability received high scores. See text for explanation of the factors.

Factor	Source	Low risk	High risk	Cont./disc.
Life span	literature	short	long	continuous
Reproduction	literature	high	low	continuous
Social organisation	literature	solitary	group	discrete
Home range	literature	small	large	continuous
Locomotion	literature	arboreal	terrestrial	discrete
Weight	literature	low	high	continuous
Encounter rate	interviews	seldom	often	discrete
Hunting method	interviews	opportunistic	premeditated	discrete
Catchability	interviews	difficult	easy	discrete
Taste	interviews	poor	good	discrete
Market value	interviews	low	high	continuous
Traditional value	interviews	low	high	discrete

Locomotion. Locomotion influences a species' detectability for a hunter and its escape chances. Tree dwelling species are assumed to have an advantage because they are not easily snared, are often out of reach of the hunter, and can escape quicker. Ground-dwelling animals, on the contrary, are easier to track and are more susceptible to be snared. Ground dwelling animals are therefore considered more vulnerable. Three categories of locomotion are defined: terrestrial, semi-terrestrial and arboreal.

Weight. The prey's weight influences the pay-off between work and revenue for hunters. Large-bodied species are favoured prey and are disproportionally sought after (Fitzgibbon *et al*. 1995; Wilkie & Carpenter 1999; Robinson & Bennet 2000). Large body size increases risk proneness to population decline or extinction (Purvis *et al.* 2000; Cardillo & Bromham 2001). The average body weight was found in the literature (Table 3). If there was pronounced sexual dimorphy, average group size and composition was used to calculate a weighted mean weight.

For the six extrinsic factors that are either locally dependent, or for which interpretation is subjective, information was obtained by means of semi-structured interviews. Fourteen local experts (active hunters, selected by their peers, see Van der Hoeven *et al.* 2004) were asked to score each factor per species between 1 and 10. A high score meant that the respondent rated the species high in vulnerable (Table 1).

Scores for similar species were compared in order to check whether the respondent comprehends the set up. He was asked, for example, whether species A was as tasty as species B, or whether he encountered species A more often than species B. At the end of the interview the first species were double-checked for earlier misinterpretations.

Encounter rate. Encounter rate is defined as the relative frequency a hunter will encounter a species in his area. It is assumed to be a proxy for density (Hill *et al.* 2003); therefore, species that are rarely encountered are supposed to be vulnerable because of low abundances (Alvard *et al*. 1997; Robinson & Bennet 2000).

Hunting method The method of hunting is defined as opportunistic when anything that is encountered is shot, or anything that walks in a trap is taken. It is premeditated when traps are set to catch specific animals, or a hunt is directed at a particular species (Robinson & Bennet 2000). Species upon which hunting is premeditated are assumed to be more vulnerable than species which are opportunistically hunted (Damiana *et al.* 2005).

Catchability. Territorial behaviour, predictable activity patterns, and use of the home range are examples that influence catchability (Fitzgibbon *et al*. 1995; Purvis *et al*. 2000). Catchability is defined here as the ease with which an animal can be captured or killed. This factor is often used in hunting models to describe hunting efficiency (e.g., Milner-Gulland 2001). The vulnerability of a species will rise when the species is easy to catch because the hunter might intentionally go for this species.

Taste. We assume the hunter to be biased in his choice of prey before the hunt based, amongst other things, on the taste of the prey’s meat (Lahm 1993; Bennet, 2002). Highly preferred species (e.g., brush-tailed porcupine, blue duiker and bush pig; Lahm 1993) are more actively pursued and are therefore more vulnerable.

Market value. A driving factor in the prey choice by the hunter is the market value (Apaza *et al*. 2002), which is defined as the price per kilogram. If an animal is highly prized for its hide or meat it pays to spend more effort in hunting those animals (e.g., brush-tailed porcupine: Jori *et al.*1998; Cowlishaw *et al.* 2005). These species are particularly vulnerable to overharvesting. Respondents were asked to give an average price for the entire animal, for a quarter, or for a part (about 2 kg). The average weight of the animal was taken from the literature (Table 3).

Traditional value. Some species have a traditional value in local culture. They can function, for example, as traditional medicine, decoration, or spiritual messengers (Adeola 1992; De Boer & Baquete 1998; Bennet & Robinson 2000; Costa-Neto 2004). Species that have a traditional value are specifically sought after and therefore more vulnerable than species without this traditional value. Protection from exploitation of certain “taboo” or “totem” species as is found in other regions in Central and West Africa was not observed in the Campo-Ma’an area. None of the hunters interviewed named a species that was not hunted for spiritual reasons. Some species did have restrictions in use, like several species that were not allowed to be eaten by pregnant women, but this did not prevent others from hunting and consuming them (e.g., the royal antelope, pers. obs.).

Species are assumed to be less vulnerable when assigned a low score, and more vulnerable when assigned a high score. Table 2 depicts the classification of the factors and their corresponding scores.

Table 2: Classification of factors for scoring vulnerability of wildlife species. Low scores mean low vulnerability, high scores mean high vulnerability. Yr=years, mo=months, ha=hectares, 1 $ = 730 FCFA (June 2001). For sources: see Table 3. Factors that are not depicted are directly scored from 1 – 10 by the interviewee, e.g., taste. Classes were selected in such a way that data were distributed evenly.

Score	**Life span (yr)**	**Reproduction**				**Social organisation (group size)**	**Home range (ha)**	**Loco-motion**	**Weight (kg)**	**Market value (FCFA)**
		Gestation period (mo)	Litter size	Birth interval (mo)	Age of fertility (mo)					
1	0-4	0-1	>9	0-5	0-10	1	<12.5		0-3	<200
2	4-8	1-2	8-9	5-10	10-20	2	12.5-25		3-6	200-250
3	8-12	2-3	7-8	10-15	20-30	3-5	25-50	Arboreal	6-12	250-300
4	12-16	3-4	6-7	15-20	30-40	6-10	50-100		12-24	300-350
5	16-20	4-5	5-6	20-25	40-50	11-15	100-200		24-48	350-400
6	20-24	5-6	4-5	25-30	50-60	16-20	200-400	Arb/Ter	48-96	400-450
7	24-28	6-7	3-4	30-35	60-70	21-25	400-800		96-192	450-500
8	28-32	7-8	2-3	35-40	70-80	26-30	800-1600		192-384	500-550
9	32-36	8-9	1-2	40-45	80-90	31-35	1600-3200	Terrestrial	384-768	550-600
10	>36	>9	0-1	>45	>90	>35	>3200		>768	>600

Table 3: Data from the literature on factors determining vulnerability of 33 wildlife species as developed in this study.

Species	Weight (kg)	Max. Life span	Home range (ha)	Gestation period	Litter size[1]	Birth interval	Age of fertility	Group size (n)	Loco-motion[2]
Small mammals									
Cephalophus monticola	5	10*	2,5 - 4	4.5	1	12	24	2	T
C. nigrifrons	16	8	*10-20*	7	1	*10-15*	*20-30*	2	T
Hyemoschus aquaticus	11.5	11*	*10-20*	4	1	13-14	24	1	T
Neotragus batesi	4	6*	2-4	6	1	*10-15*	*20-30*	1	T
Thryonomys swinderianus	6.6	4*	*<12.5*	3	2-8	6	6	1	T
Atherurus africanus [a]	2.8	15*	2-5	3.5	1-2	4	24	8	T
Large mammals									
Cephalophus dorsalis	19.5	8	12-20	7	1	*10-15*	*20-30*	2	T
C. callipygus	19.5	8	*10-20*	7	1	*10-15*	*20-30*	2	T
C. sylvicultor	62.5	11	*15-25*	5	1	*5- 10*	12	2	T
Tragelaphus spekei [b]	60 - 105	20*	*15-25*	7	1	12	48	3	T
Potamochoerus porcus	80	20*	20-1000	5	2-4	12	24	15	T
Smutsia gigantea	32.5	*15*	*50-100*	5	1	*10-15*	*30-40*	1	T
Tree-dwelling monkeys									
Cercopithecus nictitans	5.5	25*	60	6	1	24	48-66	20	A
C. cephus	3.5	22*	35	7	1	27	48-60	20	A
C. pogonias	4	20*	50-100	6	1	24	48-66	14	A
C. neglectus	6	22*	5-10	6	1	27	45-66	6	A
Colobus satanas	12	20*	60	6	1	20	60	15	A
Myopithecus ogouensis	1.5	28*	300	6	1	12	48-114	45	A
Lophocebus albigena	7	33*	400	6	1	33	36	15	A
Ground-dwelling monkeys									
Mandrillus sphinx	12.5 – 24.5	45*	1000-5000	6	1	17	39(♀)	95	T/A
Cercocebus torquatus	8	21*	500	6	1	13	32	35	T/A

Table 3. Continued.

Species	Weight (kg)	Max. Life span	Home range (ha)	Gestation period	Litter size[1]	Birth interval	Age of fertility	Group size (n)	Loco-motion[2]
Apes									
Gorilla gorilla	90 – 185	50*	800 - 1800	9	1	48	78-132	12	T
Pan troglodytes	35	45*	500	8	1	60	135-156	26	T/A
Elephant									
Loxodonta africana cyclotis	1950 – 2350	58	>3200	22	1	48	96	8	T
Buffalo									
Syncerus caffer nanus	320	26*	*>3200*	11	1	24	66	10	T
Reptiles									
Osteolaemus tetraspis	*10*	*20*	*<12.5*	*6*	*>9*	*10-15*	*40-50*	1	T
Varanus niloticus [c]	*3.3*	8	*50 - 100*	*6*	>9	*5-10*	30-40	1	T
Python spp.	*17*	*25*	*100*	*6*	*>9*	*10-15*	*60-70*	1	T
Carnivores									
Panthera pardus	55	20*	900 - 6300	3	2-3	25	24	1	T
Felis aurata	12	*10*	*1500 - 2000*	*3*	*3-4*	*15-20*	*20-30*	1	T
Aonyx congica	20	11*	500	2	2-3	*10-15*	*20-30*	2	T
Lutra maculicollis	5.3	11*	*25 - 50*	2	2	12	24	2	T
Civettictis civetta	14	14*	*50 - 100*	3	4	12	12	1	T

* Maximum lifespan from captive animals.

[1] The average of these four components forms the variable “reproduction”

[2] T= terrestrial, A= arboreal, T/A= both.

Sources: Estes (1991); Depierre and Vivien (1992); Rowe (1996); Harvey *et al.*(1987), and Kingdon (1997).

[a] Emmons (1983) and Jori *et al.* (1998)

[b] Magliocca *et al.* (2002)

[c] De Buffrénil & Hémery 2002

The authors estimated data in italics by comparing with similar species.

Expert weighting

Each factor has a different impact on the different species. We therefore decided to weight each factor according to their importance to the vulnerability of each species. This was achieved by asking 8 (inter)national experts to weight the 12 factors. Experts were selected for their in-depth knowledge in the field of wildlife management, animal ecology, and human-wildlife relations. Ideally the weighting should be done per species, but because weighting 12 factors for 33 species is more than can be asked from experts on a voluntary base, the species were divided in groups. A total of nine groups were selected (Table 4). Grouping was based on ecological grounds and on general features.

Table 4: Grouping of species for weighting by experts. Mammal nomenclature follows Kingdon (1997).

Group	Species
Small mammals	blue duiker (*Cephalophus monticola*); black-fronted duiker (*C. dorsalis*); dwarf antelope (*Neotragus batesi*); water chevrotain (*Hyemoschus aquaticus*); marsh cane rat (*Tryonomys swinderianus*) and brush-tailed porcupine (*Atherurus africanus*)
Medium-sized mammals	bay duiker (*Cephalophus dorsalis*); yellow-backed duiker (*C. sylvicultor*); peters' duiker (*C. callipygus*); sitatunga (*Tragelaphus spekei*); red river hog (*Potamochoerus porcus*) and giant pangolin (*Smutsia gigantea*).
Tree-dwelling monkeys	putty-nosed monkey (*Cercopithecus nictitans*); moustached monkey (*C. cephus)*; crowned monkey (*C. pogonias*); de Brazza monkey (*C. neglectus*); black colobus (*Colobus satanas*); northern talapoin (*Myopithecus ogouensis*) and grey-cheeked mangabey (*Lophocebus albigena*).
Ground-dwelling monkeys	mandrill (*Mandrillus sphinx*) and red-capped mangabey (*Cercocebus torquatus*)
Apes	Chimpanzee (*Pan paniscus*) and lowland gorilla (*Gorilla gorilla)*
Elephant	forest elephant (*Loxodonta africana cyclotis*)
Buffalo	forest buffalo (*Syncerus caffer nanus*)
Reptiles	african dwarf crocodile (*Osteolaemus tetraspis*); python (*Python spp.*) and monitor lizard (*Varanus niloticus*).
Carnivores	leopard (*Panthera pardus*); golden cat (*Felis aurata*); swamp otter (*Aonyx congica*); spot-necked otter (*Lutra macullicollis*) and african civet (*Civettictis civetta*).

First, experts were asked to name the most important factor per species group, then the least important, then the most important but one, and the least important but one, etc. The most important factors received a high weight (1-5); factors with less importance received a low weight (6-12).

The weighting of all experts was averaged per species group and inversed (maximum score - actual score).

Transformation and ranking

An extra analysis was done since the values obtained by the interviews are a measure of observed risk and are not nominal values. These values were root-transformed because they are proxy parameters of risk and then added which results in an estimate of the relative risk.

Finally, all values of the twelve factors obtained after the weighting by experts and after transformation were added, labelling each species with one final “vulnerability value”. This resulted in the vulnerability list.

To reduce the chance of over-dimensionality of the model and to prevent the chance of double weighting of factors, we explored whether factors could be removed from the analysis. Removing of factors enhances robustness of the method. To decide which factors could be removed a Principal Component Analysis (PCA) was performed. The PCA was used to explore the correlation between the twelve factors. With the resulting biplot pairs of correlated factors could be selected and one factor of each pair could be removed from the analysis.

Results

The factor scores show which factors most influence vulnerability (see Table 5 for average scores per species for each factor). Weight, for example, was very important for elephant (*Loxodonta africana cyclotis*, score 10), but not for porcupine (*Artherurus africanus*, score 1), while catchability was more important for porcupine (score 7.9 against 2.1 for elephant). The results of the factor scores illustrated that elephant, mandrill (*Mandrillus sphinx*), and the apes (Chimpanzee *Pan troglodytes* and Gorilla *Gorilla gorilla*) usually received high scores and were considered therefore most vulnerable. Examples of species that received low scores were otters (spot-necked otter *Lutra maculicollis* and swamp otter *Aonyx congica*), dwarf antelope (*Neotragus batesi*), and yellow-backed duiker (*Cephalophus silvicultor*).

The opinion of international experts on the importance of the selected factors for the different species groups is presented in Table 6. They considered hunting method important for the vulnerability of most species (7.63 – 9.75 on a scale of 0 – 12) and locomotion much less important (2.63 – 4.5 on a scale of 0 – 12).

The elephant generally scored high in the weighting process, except for the factors catchability and taste. In other words, experts believed that taste is not a motive for hunters to hunt elephant and that elephants are hard to catch for local hunters. These considerations diminish the influence of this factor on the overall vulnerability of the species. Least important of all was social organisation of reptiles, it was thought to have hardly any influence (0.38 on a scale of 0 –12).

Taste and locomotion were positively correlated, as were traditional value and market value. All these four factors were negatively correlated with the factors encounter rate and catchability. This is evident by the alignment and direction of the arrows in the PCA biplot (Fig. 1a).

The factors locomotion and traditional value were removed from the analyses, after which the new ranking was compared with the original 12-factor ranking by performing a Pearson's correlation test. This test confirmed that the removal of two factors had no effect on the ranking of the species (2-tailed; r= 0.938; p< 0.01, n=33).

The final vulnerability list is given in Table 7, with the most vulnerable species at the top.

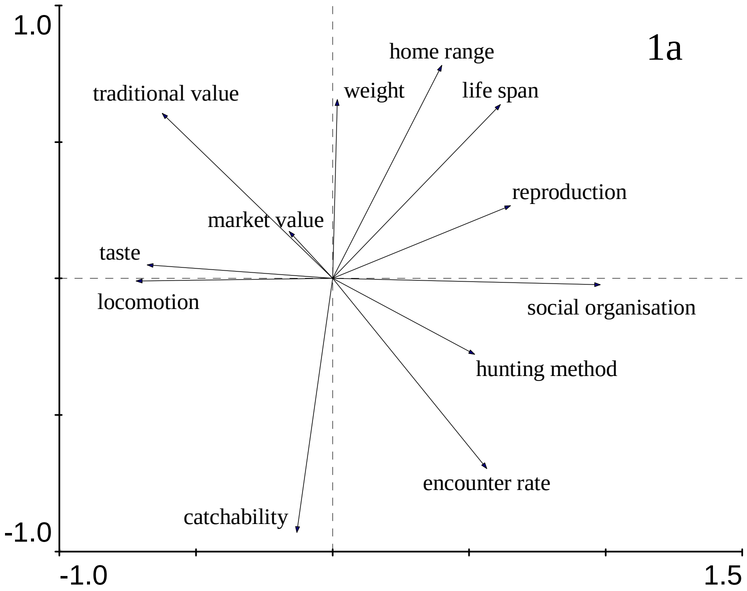
Table 5: Averaged scores per species for each factor that determines vulnerability, obtained through interviews and literature study. High scores indicate high vulnerability.

Species	life span	reproduction	Soc. organisation	home range	locomotion	weight	encounter rate	hunting method	catchability	taste	market value	traditional value
Small mammals												
Cephalophus monticola	3	5.1	2	1	9	2	7.5	5.6	8.2	5.2	5	1.3
C. nigrifrons	3	5.8	2	2	9	4	2.8	2.5	5.3	5.1	5	2.4
Hyemoschus aquaticus	3	5	1	2	9	3	2.6	3.1	5.4	7	7	1.5
Neotragus batesi	2	5.5	1	1	9	2	5.3	2.3	7.8	7.1	3	1.7
Thryonomys swinderianus	1	2.8	1	1	9	3	7.3	6.1	6.5	6.9	5	1.2
Atherurus africanus	4	4.1	4	1	9	1	5.9	6.9	7.9	7.4	10	1.6
Medium-sized mammals												
Cephalophus dorsalis	3	5.8	2	2	9	4	4.9	3.8	5.3	4.5	4	2.3
C. callipygus	3	5.8	2	2	9	4	3.1	3.1	5.1	5.4	4	1.9
C. sylvicultor	4	4.8	2	2	9	6	3.3	2.7	2.8	3.6	1	2.2
Tragelaphus spekei	5	6.3	3	2	9	6	5.1	6.3	4.7	7.4	2	2.3
Potamochoerus porcus	5	4.8	5	7	9	6	3.2	4.9	4.1	8	1	1.4
Smutsia gigantea	2	5.5	1	4	9	5	1.7	2.4	4.3	9.2	8	1.3
Tree-dwelling monkeys												
Cercopithecus nictitans	6	6.8	6	4	3	2	7.4	5.1	5.4	5.7	6	1
C. cephus	5	7.3	6	3	3	2	7.9	5.3	6	6.6	9	1.6
C. pogonias	5	6.8	5	4	3	2	7	4.7	5.5	6.5	8	1.1
C. neglecticus	5	7	4	1	3	2	2.7	3.3	4.6	6.1	5	1.1
Colobus satanas	5	6.5	5	4	3	3	4.5	3.8	3.1	5.1	5	1.1
Myopithecus ogouensis	7	7	10	6	3	1	8.4	3.3	6.4	5.4	7	1.1
Lophocebus albigena	8	6.8	5	6	3	3	5	3.6	4.2	4.2	7	1.2
Ground-dwelling monkeys												
Mandrillus sphinx	10	6	10	9	6	4	6.9	7.2	6.5	6.7	8	1.1
Cercocebus torquatus	5	5.8	9	7	6	3	6	5	5.8	5.5	7	1
Apes												
Gorilla gorilla	10	9.8	5	8	9	7	4.6	5.4	3.2	6.1	2	2.9
Pan troglodytes	10	9.5	8	7	6	5	5.7	5	3.8	6.4	6	4.1
Elephant												
Loxodonta africana	10	10	4	10	9	10	5.1	7.3	2.1	7.7	6	6.3
Buffalo												
Syncerus caffer	6	8	4	10	9	8	5	7.5	2.3	7.3	3	2.1
Reptiles												
Osteolaemus tetraspis	7	3.8	1	1	9	3	5.6	7	6.9	8.7	10	4.4
Varanus niloticus	2	3.5	1	4	9	2	3.6	2.5	4.3	8.2	8	2.4
Python sebae	9	4.3	1	2	9	4	2	1.7	3.9	7.4	10	7.1
Carnivores												
Panthera pardus	5	4.8	1	10	9	6	1.3	1.6	1.6	6.7	10	6.5
Felis aurata	4	3.5	1	9	9	3	1.5	2.8	3.3	6.3	10	1.8
Aonyx congica	3	4	2	7	9	4	3.6	1.7	2	3.7	2	3.3
Lutra maculicollis	3	4.3	2	3	9	2	3.9	2.2	2.9	4.2	4	3.5
Civettictis civetta	4	3.8	1	4	9	4	3.9	3.3	4.1	7.2	6	4.4

Table 6: Weight attached by international experts to factors that determine vulnerability per species group. A factor is important to the vulnerability of a species group when assigned a high value. See text for arrangement of species groups.

	Small mammals	**Medium-sized mammals**	**Tree dwelling monkeys**	**Ground dwelling monkeys**	**Apes**	**Elephant**	**Buffalo**	**Reptiles**	**Carnivores**	**average**
Life span	4.50	3.75	4.25	4.38	7.00	7.00	5.63	4.13	6.00	**5.2**
Reproduction	6.88	6.00	5.88	5.75	8.00	8.88	6.50	4.63	4.88	**6.4**
Social organisation	1.38	0.75	8.13	8.25	6.00	4.50	4.38	0.38	0.63	**3.8**
Home range	4.50	3.13	5.00	4.38	3.88	6.63	4.75	3.00	5.88	**4.6**
Locomotion	4.50	3.13	2.63	3.00	3.38	2.88	2.63	2.75	3.38	**3.1**
Weight	4.00	7.13	5.50	5.13	6.13	7.25	9.00	5.50	4.50	**6.0**
Encounter rate	10.13	8.88	9.25	9.63	6.38	5.75	5.88	7.88	6.63	**7.8**
Hunting method	9.50	9.75	9.38	9.38	8.13	7.63	8.63	8.00	8.38	**8.8**
Catchability	9.75	8.75	7.00	7.50	5.00	1.75	5.00	9.13	6.25	**6.7**
Taste	3.25	4.00	2.63	1.75	2.25	2.50	3.63	6.50	4.00	**3.4**
Market value	5.75	7.50	5.75	5.88	6.75	8.63	8.13	8.63	8.13	**7.2**
Traditional value	1.88	3.25	0.63	1.38	3.13	2.63	1.88	5.50	7.00	**3.0**

Figure 1: PCA of factors determining vulnerability of wildlife before 1(a), and after 1(a) removal of two factors, namely “locomotion” and “traditional value” (R^2=68.9 and R^2=68.1 respectively). Direction and length of arrows indicate nature and strength of correlation between factors.



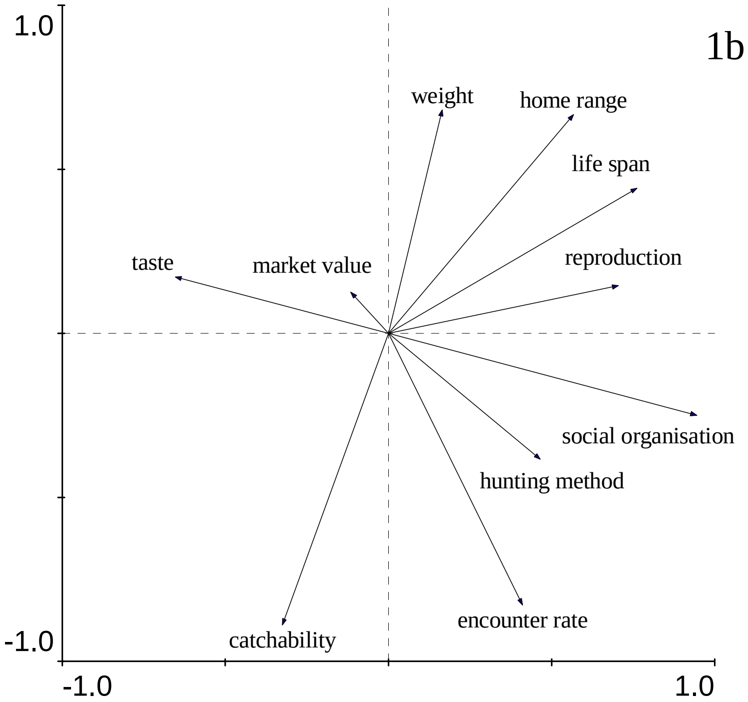


Table 7: Vulnerability list: ranking of species according to their vulnerability, determined by valuation of ten factors through literature research and interviews. The symbol * =status according to the IUCN Red List of threatened species (IUCN 2006): Vu=vulnerable; EN=endangered; LR/cd=Lower Risk, conservation dependent; LR/nt=Lower Risk, near threatened; LC= Least Concern; DD=Data Deficient. Mammal nomenclature follows Kingdon (1997)

Rank	Species		Score	IUCN
1	Mandrill	*Mandrillus sphinx*	66.00	Vu
2	Elephant	*Loxodonta africana*	65.08	Vu
3	Chimpansee	*Pan troglodytes*	61.67	EN
4	Buffalo	*Syncerus caffer*	59.96	LR/cd
5	Talapoin	*Myopithecus ogouensis*	59.90	
6	Red-capped mangabey	*Cercocebus torquatus*	59.51	LR/nt
7	Moustached monkey	*Cercopithecus cephus*	59.19	LC
8	Gorilla	*Gorilla gorilla*	58.22	EN
9	Putty-nosed monkey	*Cercopithecus nictitans*	57.60	LC
10	Crowned monkey	*Cercopithecus pogonias*	56.89	LC
11	Grey-cheeked mangabey	*Lophocebus albigena*	56.15	LC
12	African dwarf crocodile	*Osteolaemus tetraspis*	53.96	Vu
13	Brush-tailed porcupine	*Atherurus africanus*	53.15	LC
14	Black colobus	*Colobus satanas*	51.84	Vu
15	Sitatunga	*Tragelaphus spekei*	51.30	LR/nt
16	Blue duiker	*Cephalophus monticola*	50.69	LC
17	Red river hog	*Potamochoerus porcus*	49.52	LC
18	Giant pangolin	*Smutsia gigantea*	48.63	LC
19	De Brazza's monkey	*Cercopithecus neglectus*	48.32	LC
20	Leopard	*Panthera pardus*	47.81	LC
21	Marsh cane rat	*Thryonomys swinderianus*	47.48	LC
22	Python	*Python sebae*	47.44	
23	Bay duiker	*Cephalophus dorsalis*	46.89	LR/nt
24	Civet	*Civettictis civetta*	46.81	LC
25	Golden cat	*Felis aurata*	46.60	Vu
26	Monitor lizard	*Varanus niloticus*	46.59	
27	Water chevrotain	*Hyemoschus aquaticus*	46.10	DD
28	Peters' duiker	*Cephalophus callipygus*	45.25	LR/nt
29	Dwarf antelope	*Neotragus batesi*	44.91	LR/nt
30	Black-fronted duiker	*Cephalophus nigrifrons*	44.70	LR/nt
31	Yellow-backed duiker	*Cephalophus sylvicultor*	41.00	LR/nt
32	Spot-necked otter	*Lutra maculicollis*	40.54	LC
33	Swamp otter	*Aonyx congica*	40.52	DD

Discussion

This research attempts to develop a method for locally setting priorities for conservation of rainforest wildlife. Ranking species according to their vulnerability to decline and extinction was assumed to identify those species that need local conservation attention most. The order of the species listed is of more importance than the value attached to it, as this value is only a means for ranking.

In order to enhance the robustness of the procedure and to reduce the weight of factors that were strongly related, two factors were removed. Although each of the six most correlated factors could have been removed (taste and locomotion; traditional and market value; encounter rate and catchability), the factors locomotion and traditional value were removed. This decision was based more on the nature of the factors than on strongest correlation. Locomotion was removed because the division in only three classes was assumed to be restrictive, and the experts weighted this factor low (last but one). Furthermore, there are indications that being arboreal could be a disadvantage instead of an advantage, as wildlife is more conspicuous moving trough the canopy (Isaac & Cowlishaw 2004). Traditional value was removed for similar reasons; no reference was found and it was ranked as least important by the experts. Moreover, in our study area only few species have a known traditional value (pers. obs.). However, this could be very different for other regions, for example, South America or Asia (Costa-Neto 2004) where traditional use of wildlife for medicinal purposes forms a serious threat to local wildlife. The results of the tests support the choice because none of the eliminated factors influenced the ranking of the species on the vulnerability list. A subsequent PCA on ten remaining factors confirmed that correlation between factors decreased (Fig. 1b). The final vulnerability list was thus based on the ten remaining factors.

Most other factors used in this study are also used in other risk or vulnerability assessments (e.g., Purvis *et al*. 2000). The first and most obvious factor that we did not use is the abundance or population size of a species, and the change hereof (O'Grady *et al*. 2004). We argue that this information frequently lacks and was therefore not incorporated in our vulnerability assessment so as to make our proposed method more generally applicable. Yet we have information on the densities of the study species in the study area (Van der Hoeven *et al.* 2004) and the results were used in combination with the vulnerability list by the wildlife managers in the study area. A separate study on abundances validates the vulnerability assessment and vice versa. Yet, in case of budget or time constraints (as is often the case), a vulnerability assessment in itself should provide essential indicative information for conservation planning.

The vulnerability list is based not only on international expertise, but also on local expertise, specifically from indigenous hunters, thereby combining both the international and local assessments. Apart from pointing out the obvious species as being threatened (e.g., elephant and apes) this method also analysed the status of lesser-known species at the local level (e.g., moustached monkey *Cercopithecus cephus*). These lesser-known species can be of more importance to resource conservation than flagship species because they form an important resource for local people. Assessment of only a few flagship or focal species is often not adequate (Lindenmayer *et al*. 2002). Regulation of hunting of common species should facilitate protection of rare and threatened species since this could reduce the hunting pressure on endangered species.

The listing according to vulnerability is not rigid; on the contrary, it should stimulate discussion to generate attention for species that need to be monitored more closely than others. There was, for example, no clear a priori reason for the moustached monkey to be ranked as high as the fifth place because it is a common monkey species that is frequently encountered in primary and secondary forest. No factor proved to be specifically high, the moustached monkey ranked generally higher for several factors. Previous research in the area, however, described a steady decline in density for this species in the past 15 years (Mitani 1991; Matthews & Matthews 2002; Van der Hoeven *et al.* 2004). This trend has also been observed in the Central African Republic (Philippe Auzel, personal communication 2002). This decline marks the species as vulnerable which supports its position on our list.

When the vulnerability list is compared to the IUCN Red List of threatened species (IUCN 2006) the listing of the most threatened species in the IUCN Red list was confirmed for the Campo-Ma'an area (Table 7). Of the seven species that are listed as being vulnerable or endangered (the second and third highest IUCN threat categories) in the study area, six are found amongst the top fourteen, including the first three. However, there are some clear differences between the local vulnerability list and the IUCN Red list. Some species listed as threatened by the IUCN are ranked low on the local list, and some species not listed as threatened by the IUCN are ranked high on the local list. It should be noted that not all species listed by the IUCN are threatened; species that are listed as LC (Least Concern) are assumed not to be threatened. The clearest example concerns the duikers. All duikers that occur in the Campo-Ma'an area are listed by the IUCN as threatened (LR/nt), except for the blue duiker. In our assessment we found the exact opposite: all duikers listed as threatened by IUCN are at the bottom of the local list (thus not considered as threatened) while the blue duiker is listed considerably higher. This discrepancy between both listings points to the limitations when using only the IUCN Red list at a specific locality. When such a discrepancy emerges, specific attention needs to be given to these species. It turns out, based on density assessments conducted in the study area at the same time (Van der Hoeven *et al.* 2004), that the duikers are generally more abundant in our study area than elsewhere in the region. Combining local data and a local vulnerability assessment provides specific information that has different implications for management. The disparity between global and local ranking is a known problem (i.e., Breininger *et al*. 1998) and emphasizes the importance of local based assessments as is pointed out by Rodríguez *et al*. (2000, 2004). Although efforts are made to adjust extinction risk assessments to regional criteria (i.e., Gärdefors *et al*. 2001; Miller *et al.* 2007), the local assessment in the present study demonstrates that setting conservation priorities should not be solely based on the basis of extinction risk, but should be combined with local data. The vulnerability assessment described in this study bridges the gap between higher level risk assessments, such as the IUCN Red lists, and the actual priority setting that is needed in the field.

The density assessment of the same species in the same area (Van der Hoeven *et al*. 2004) provides additional information on the wildlife status. When the density assessment and the vulnerability list are combined, they can assist in the formulation and improvement of wildlife management activities, as illustrated with the example on duikers. The vulnerability list itself does, however, provide a robust indication of which species need special attention. With only local interviews and literature research to conduct, the required time for producing the vulnerability list is minimal. This allows for a rapid assessment of the general vulnerability of a wildlife population in a given area, after which more thorough and species-specific management measures can be taken.

Acknowledgements

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Next page: blue duiker caught with a snare

Chapter 4

THE PRICE, STATE AND DIVERSITY OF A BUSHMEAT MARKET SUPPLY IN CAMEROON, CENTRAL AFRICA; THE SEARCH FOR CLEAR INDICATORS.

Christiaan A. van der Hoeven, Willem F. de Boer and Herbert H. T. Prins

Abstract

The bushmeat crisis is now accepted to be a major threat to biodiversity as well as to the sustained livelihoods of local people. Clear indicators to asses whether bushmeat trade is sustainable are required. Socio-economical and biological data on bushmeat markets can hold the key to a relative cost-effective way of monitoring bushmeat trade. However, data analyses are hardly ever conclusive on the sustainability of the off-take. Theoretical models that include various variables perform quite well in simulations, but long-term, multi-variable datasets provide no unequivocal definition on the sustainability of the bushmeat trade. This study selected several factors that are thought to indicate the state of the trade and exploitation pressure, and analyzed the relations between dependent factors and several independent factors that are hypothesized to influence this state. The factors assumed to be indicative of the bushmeat trade are: the price, the state (the percentage smoked meat), and the diversity of the bushmeat for sale (in terms of number of species for sale). These are thought to be related to several independent factors that influence hunting pressure, which are: human population level size, as a proxy for the demand for bushmeat; distance between markets, distance to a National Park; and the wildlife density in the area surrounding the market. We conclude that price and state of the supply prove to be useful indicators, based on their relation to factors indicating bushmeat exploitation levels. Diversity of supply is useful only in combination with the state of the product or with the percentage of rodents at the supply.

Introduction

The bushmeat trade is one of the major threats to Central and West African rainforest wildlife (Robinson *et al.* 1999; Barnes 2002, Robinson & Bennet 2000, Milner-Gulland *et al.* 2003, Walsh *et al.* 2003). It does not only threaten wildlife, but could also decrease the protein supply for the local human population (Bennet 2002; Fa *et al.* 2003; Brashares *et al.* 2004). This problem, commonly referred to as the “bushmeat crisis”, is extensively discussed in literature (e.g., special issues: Oryx, vol. 36 (4), 2002; and Conservation Biology, vol. 16 (3), 2002). Most studies on the bushmeat crisis can be divided in two categories: a biological and a socio-economical approach. The biological approach involves the study of population dynamics of wildlife, sustainable harvesting models, and relations between human population pressure and wildlife densities (e.g., Robinson & Redford 1991; Milner-Gulland & Akçakaya 2001; Stephens *et al.* 2002; Rowcliffe *et al.* 2003). The socio-economical category of studies involves, amongst others, the analysis of the commodity chain (Wilkie & Godoy 2001; Cowlishaw *et al.* 2005b), the possible alternatives for this limited resource, income related influences, and ethnic considerations (Demmer *et al.* 2002; de Merode *et al.* 2004; Fa *et al.* 2004; Basset 2005; Damania *et al.* 2005; Wilkie *et al.* 2005; Sirén *et al.* 2006). The integration of these two approaches and translation to conservation policies have so far not come up with satisfactory results to solve the crisis (Davies 2000; Williams *et al.* 2003). What is clear from these discussions, however, is that still much information is needed, at the biological as well as at the socio-economic level. The bushmeat market is the interface where both socio-economic as well as biological data are readily available. The supply side, considered here as the bushmeat available at the market, is a reflection of the bushmeat exploitation (Juste *et al.* 1995, Fa *et al.* 2000).

It is virtually impossible to assess whether a given bushmeat market is based on sustainable off-take, even when large datasets are available (see for example Crookes *et al.* (2006), but as exception to this: Cowlishaw *et al.* 2005a). Long term market studies provide useful data mostly on only one or a few species, which results in conclusions on these specific species and not on the general market dynamics or the overall exploitation (e.g., for wild pigs in Indonesia, Milner-Gulland & Clayton 2002). No studies have come up so far with a set of variables that indicate general exploitation level. In this study we propose a selection of factors that can measure the state of the exploitation. We make the distinction between state and level, as level suggests a quantifiable figure which has proven to be very difficult to obtain. The general question is whether a state of bushmeat exploitation can be found, based on only a few selected variables. We focused on the supply side, as market supply data are straightforward and relatively easily obtainable.

Three market features are hypothesized to represent the state of bushmeat exploitation: the diversity of the supply, the price, and the state of the supply (smoked meat versus fresh). These three dependent factors are influenced by four independent factors: the human population size (as a proxy for the demand for bushmeat), the distance between local and city market, the distance to a National Park, considered as a major bushmeat source, and finally the wildlife density in the surrounding area. In this study we explore the relations between these dependent and independent factors to verify whether our presumptions on the mechanism of the bushmeat supply hold, and whether these three factors can be used for a rapid assessment of a bushmeat market.

Species diversity of the bushmeat supply

Hawkes *et al.* (1982) explored the applicability of the Optimal Foraging Theory (OFT) on the description of the resource exploitation by hunter-gatherers. Their main conclusion was that high-ranking resources, in terms of a cost-benefit analysis, were almost always large to medium sized mammals. Although plant material, smaller animals, and animal products were collected too, the hunter-gatherers always preferred to pursue a medium or large prey when the possibility presented itself. The fact that most vulnerable and threatened wildlife species in the world are in the large to medium body size category supports this theory (IUCN 2000; Fa *et al*. 2000). Hawkes *et al.* (1982) found that foraging results were independent of abundances, and that high ranking prey was always taken (see also chapter 3, this thesis). Hence we assumed that the proportion of these species within the supply at the market provides a good reflection of the actual abundances of these species in the source area. In other words: when an area is not overexploited, and wildlife species diversity and densities are high, the OFT predicts that only the most appreciated and lucrative species are taken. Lower ranking species are left untouched. Therefore, in an unexploited area this should translate to a lower diversity of species at the offtake and market, as most small mammal species are not taken. When on the other hand, the resource area is regularly exploited and medium to large sized mammals have become rare, then hunters will switch to a wider range of smaller species in order to satisfy their protein demand. This will result in a higher diversity of the offtake, as smaller mammals (e.g., rodents) are added to the offtake, which results then in a higher diversity on the market, where it forms the supply. The last situation is when an area is overexploited, and all medium to large mammals have already disappeared and smaller categories of mammals are in decline, then the diversity will decrease again. All that is left to hunt are small species such as rodents and small primates (Fa *et al*. 2000).

For human population size, as a proxy for hunting pressure, it is hypothesized that when human population sizes increase, species diversity at the market will first increase (B; Fig. 1), until hunting pressure is so high that the supply dries up, and diversity decreases again (A; Fig. 1). The same reasoning applies for wildlife density around a market site. When wildlife density is high, diversity will be low (B; Fig. 1; when wildlife density decreases, diversity at the market supply will increase, until densities are so low that diversity at the market will decrease again (A; Fig. 1). When market sites are compared and the dependent variables are known, then a site can be described as overexploited (situation A) or relatively healthy (situation B).

We furthermore hypothesize that the species diversity at the bushmeat market is positively correlated with distance to the National Park; and negatively correlated to distance to the city (Table 1).

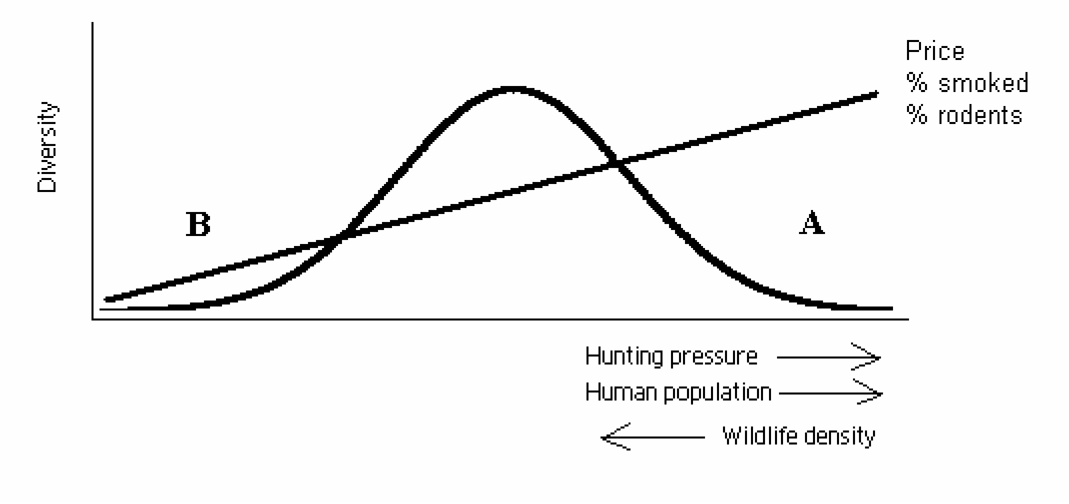


Figure 1: Change of the dependent factors: the species diversity (bell-curve); the price of bushmeat; the percentage smoked meat, and the percentage rodents found at a bushmeat market in relation to independent factors.

Table 1: Expected correlations between dependent and independent factors.

	Population size	Distance to City	Distance to NP	Wildlife density
Diversity	+/–	–	+	+/–
Price	+	–	+	–
State (% smoked)	+	–	+	–

Bushmeat prices

General market rules dictate that when a product becomes scarce, prices will rise (Brown & Field 1978). A more direct and appealing indicator is not available: one would expect that when wildlife becomes scarce due to overexploitation, prices of bushmeat will rise. To our knowledge, no study has addressed this direct relationship. Several studies analyzed the relations between socio-economic factors and bushmeat prices, such as income, taste, cultural preferences, etc. (Wilkie & Godoy 2001; Wilkie *et al*. 2005), but none examined the direct relation between abundance of wildlife in a catchment area and the prices of bushmeat at the market. Statements on this issue have been made, claiming that rare species are similar in price to common species (e.g., Wilkie & Godoy 2001), but as far as we now this is a more anecdotal than structural claim. For that reason, the first factor in this study that should be linked to price is the wildlife density in the surrounding area. The other independent factors are distance from source to the market, and human population size. The larger the urban settlement, the larger the catchment area should be that is needed to provide for the bushmeat demand. Large urban settlements act thus as a sink for its surroundings. A protected area, such as a National Park, can be considered as the main source. The distance from the market to the source and the sink should be reflected in the price. We therefore hypothesize that prices are negatively correlated with the wildlife density in the source area, and with distance to cities, and are positively correlated with human population size at the surveyed market and with distance to the National Park (Table 1).

State of the bushmeat supply

The bushmeat that is sold on the market comes in three states: fresh, smoked and cooked. The latter is not considered here, as cooked meat could have been either smoked or fresh before it is turned into a dish. The division of the supply in two states is assumed to indicate the distance of the source of the bushmeat to the market. Meat deteriorates quickly in the tropics and needs either to be quickly delivered to the market, or should be preserved. The most common way of preserving meat in tropical rainforests is by smoking (Cowlishaw *et al*. 2005b). When human population pressure increases, wildlife resources in the surrounding areas are expected to decline (Harcourt *et al.* 2001). So hunters must travel further (Milner-Gulland & Clayton 2002), and need to preserve their merchandise from deteriorating before reaching the market. The percentage of smoked meat at a market can therefore be an indicator for the distance between market and the source, and consequently the exploitation state around the market site. We hypothesize that the percentage of smoked meat is positively correlated with the human population size and with the distance from the National Park, and negatively correlated with the distance to cities and with wildlife densities in the area surrounding the market (Table 1).

Species composition of the bushmeat supply

Apart from the average species diversity, the specific composition of the bushmeat supply may indicate the state of exploitation (Fa *et al*. 2000, Rowcliffe *et al*. 2003). When hunting pressure increases, hunters switch to prey species that are easier to obtain, which is reflected in the supply at the market. When most preferred species are gone, some typical species or species groups are left to exploit. In general small, high-reproduction species replace large, low reproduction species. Fa *et al*.(2000) surveyed markets while surveying the populations in the source area at the same time, and found that when medium to large species were hunted to near extinction, hunters changed to rodents and small monkeys (“hunting down size-classes”). This was reflected in the market supply: the percentage of rodents increased while wildlife biomass in the surrounding areas decreased. In this study we therefore compared the occurrence of rodents at the different market sites in relation to the independent factors described above. Rodent occurrence was expected to be positively correlated to human population sizes, to distance from national park, and negatively correlated to distance from city and wildlife density.

Methods

The study was conducted from August 2001 to November 2002 in the integrated management area of the Campo-Ma’an Management and Biodiversity Conservation Project, located in the Southwest corner of Cameroon, Central Africa (2°21’29 N, 10°09’05 E). The integrated management area was established in January 2000, and is covered mainly with Guinea-Congolian rainforest (De Kam *et al.* 2002, Tchouto *et al.* 2006). The altitude ranges from 0-1100 m; a mountain chain extends from Southwest to Northeast. Annual rainfall varies from

1670-2950 mm. Within the project area are logging concessions, rubber and oil palm plantations, agriculture zones, and a National Park. Approximately 60,000 people live within the area, in 167 villages. The port of Kribi borders the project area and is the main hub through which diverse products are transported (timber, rubber, bushmeat, etc.).

For this study five markets were selected where the independent factors were known to be different (hereafter called sites, Fig. 2). The first was in the rubber plantation "Hevecam", where villages are solely constructed for plantation and factory labourers; the second in Ipono, where a large sawmill is situated; the third in Campo where a small port is located; the fourth in Akom II, which is mainly a cross roads, serving transporters on their way to the main international port of Douala; and the last is the remote village of Ma'an, isolated in the south-east of the project area. This village has no particular function except as a political centre.

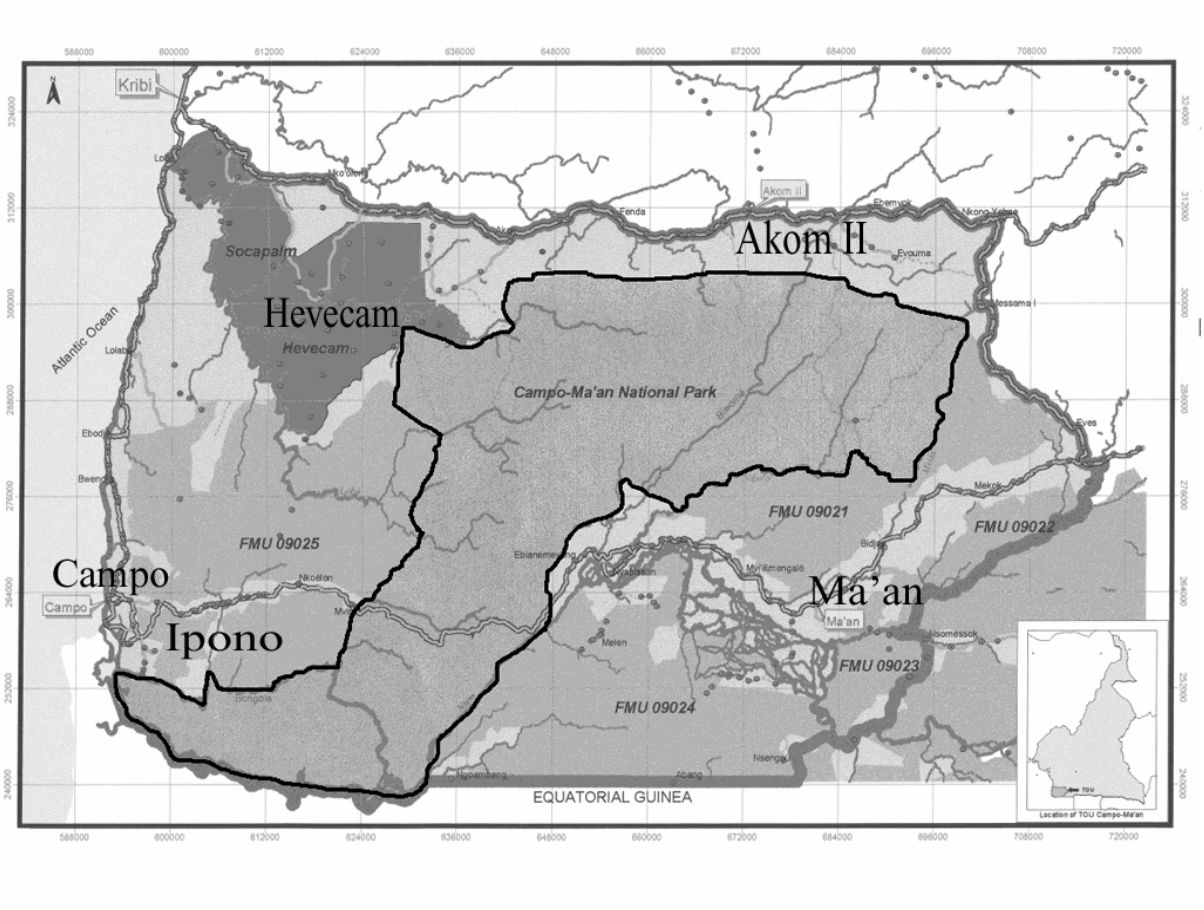


Figure 2: Map of the study area with the sites of the bushmeat survey.

In each of these five sites, one surveyor was selected who knew the place and its inhabitants intimately. Their established position within the community was essential for data collection. Extracting data on bushmeat prices and composition is complicated; bushmeat traders are reluctant to share information because much of their produce is acquired illegally. Most people are aware of legislation which prescribes that only authorized persons can hunt and trade wildlife species, and that permits are required for both hunting and selling bushmeat. As hardly anyone has a license, theoretically all trade in bushmeat is illegal. Information on their bushmeat supply and their origin is only shared with people who are trusted not to pass this information on to the authorities.

Each surveyor selected four to six persons that traded in bushmeat at their site. The surveyor visited these traders once a month, during the weekly market or around pay-day, when supply is at its top. With a simple questionnaire the surveyor noted what the saleswomen (most traders were women) had for sale, recording: species; parts for sale; number; state, and price of the bushmeat. Bushmeat is sold in three portions: the entire animal, a quarter animal ("gigot") or in chunks ("morceau"). Chunks were generally the same size, as they were mostly cooked and sold per piece for the same price. The number was recorded as the quantity per category. When a saleswoman knew how many chunks came from one animal, this was also noted, and used to estimate the total number of entire animals. State of the bushmeat could be fresh, smoked or cooked. Price was noted per unit; thus per entire animal, quarter or chunk.

Independent factors

The independent factors were known up front (Table 2). The distance between sites and city (Distance to city) and the NP (Distance to NP) was recorded using a GPS, human population size at the site was taken from the latest survey (de Kam *et al.* 2002). Wildlife density was indexed. Densities of surveyed species (Wildlife density) were compiled from samples within a radius of 25 km around each site, from a wildlife study conducted in the same period in the project area (van der Hoeven *et al.* 2004). These densities were averaged per species for all sites; after which the deviation from this average (index=100) was used to illustrate differences in densities between sites.

Table 2: Data for independent factors used to analyse bushmeat market in Campo-Ma'an.

Site	Population size*	Distance to city (km)	Distance to NP (km)	Wildlife density (index)**
Hevecam	10,000	50	14	59
Ipono	1,408	84	4	105
Campo	1,627	75	14	106
Akom II	1,720	86	8	83
Ma'an	1,293	209	17	113

* De Kam *et al.* 2002; ** Van der Hoeven *et al.* 2004.

Dependent factors

All dependent factors were calculated per site per month. The diversity of the bushmeat market supply was calculated as the number of species recorded per saleswoman per month. We used an index to standardize the bushmeat price. First the average price per entire animal, quarter, and chunk of each species per month for all five sites together was calculated. This average was indexed as 100, after which the site-specific prices and their deviation from this index were calculated, first per entire animal, quarter and chunk, then these three indices were averaged to result in one index per species, per site. Prices are lower than the monthly average when below 100, and higher when above 100. The proportion of the meat that was smoked, defined as % smoked, was calculated per site, per month. Here percentages were also first calculated per entire animal, quarter and chunk, after which they were averaged to give one percentage per species. Percentage smoked and % fresh adds up to 100 %.

Analysis

First, a Canonical Correspondence Analysis (CCA) was performed to explore relations between factors, sites, and species. The resulting triplot illustrates the strength and direction of associations between factors, sites, and species. Furthermore, a general linear model (GLM) was built for each dependent factor incorporating all independent factors in the model, and following a backward elimination of non-significant factors. The GLM was corrected for time by introducing month as fixed factor in the model. The Levene’s test was used to test for equality of variances, and the Kolmogorov-Smirnov test was used to confirm the normality of the residuals.

Species were labelled and named in the CCA triplot; correlations were analyzed between rodents and the independent factors, using the triplot, and a GLM.

Results

Blue duiker and African brush-tailed porcupine were the two most encountered bushmeat species, covering respectively 20% and 19% of the supply (Table 3).

Data on dependent factors are given in Table 4.

Table 3: Total numbers observed, and percentages of species found at bushmeat markets in the Campo-Ma'an area, Cameroon, divided per site (in %) and per category. The numbers correspond to the points in the CCA triplot. Names according to Kingdon (1997). Wildlife identified at group level but not at species level was lumped (as "rodents", "primates", "carnivores", etc.).

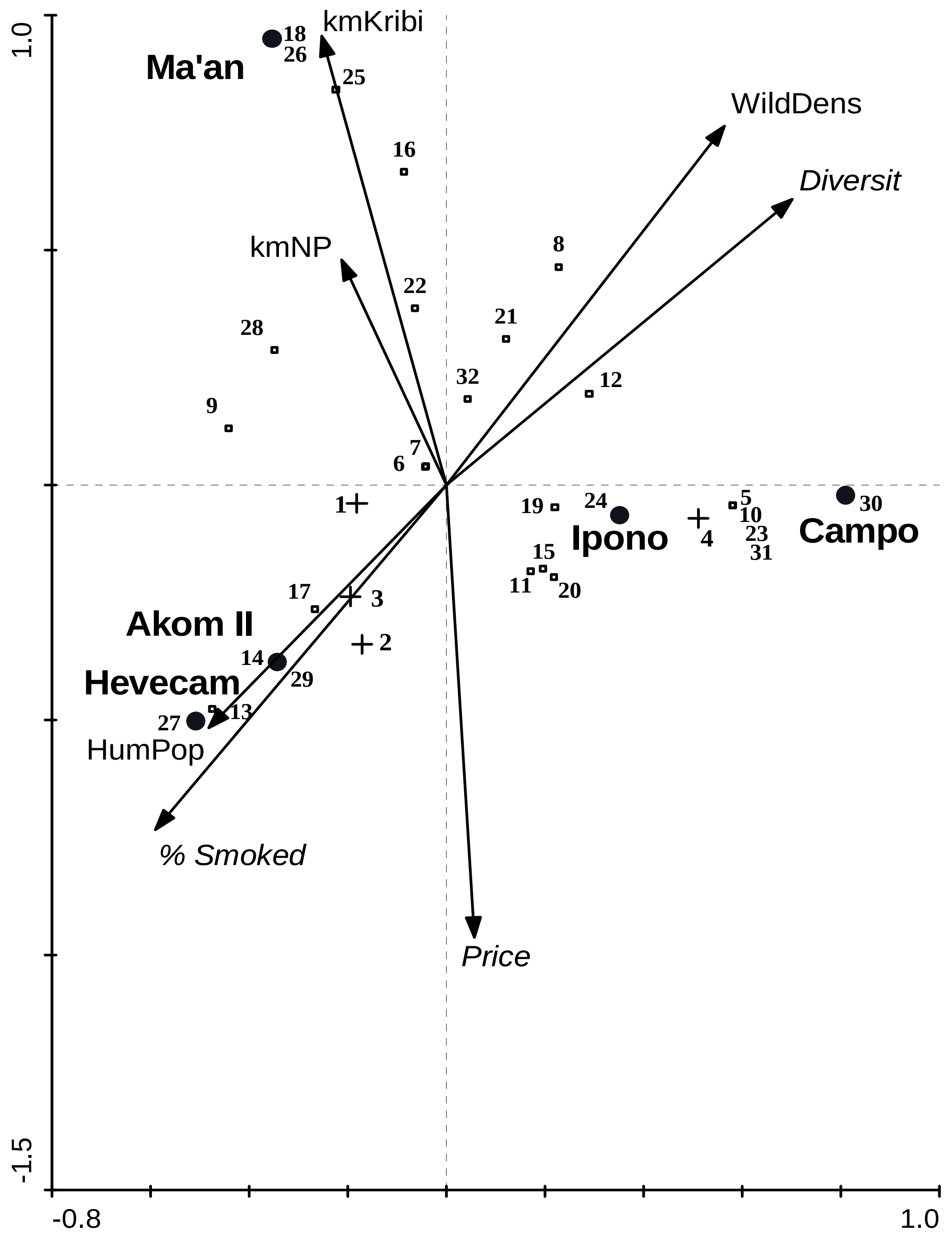
	Species		**Total n**	**Total %**	**Hevecam**	**Ipono**	**Campo**	**Akom II**	**Ma'an**
1	Brush-tailed porcupine	*Atherurus africanus*	155	19	31	17	9	24	23
2	Giant pouched rat	*Cricetomys emini*	75	9	16	5	6	18	0
3	Marsh cane-rat	*Thryonomys swinderianus*	18	2	4	0	2	4	1
4	Rodent		93	12	4	11	31	3	1
	Rodents		**341**	**43**	**55**	**34**	**48**	**48**	**25**
5	Peters' duiker	*Cephalophus callipygus*	2	0	0	1	1	0	0
6	Bay duiker	*Cephalophus dorsalis*	48	6	7	6	5	4	7
7	Blue duiker	*Cephalophus monticola*	161	20	13	27	14	29	25
8	Yellow-backed duiker	*Cephalophus silvicultor*	2	0	0	0	1	0	1
9	Duiker		10	1	4	0	0	0	3
10	Water chevrotain	*Hyemoschus aquaticus*	3	0	0	1	1	0	0
11	Red river hog	*Potamochoerus porcus*	14	2	2	4	1	0	0
12	Sitatunga	*Tragelaphus spekei*	25	3	1	4	6	0	4
	Artiodactyls		**265**	**33**	**27**	**43**	**27**	**33**	**40**
13	Red-capped mangabey	*Cercocebus torquatus*	9	1	4	0	0	1	0
14	Moustached monkey	*Cercopithecus cephus*	1	0	0	0	0	1	0
15	Putty-nosed monkey	*Cercopithecus nictitans*	22	3	0	5	4	6	0
16	Crowned monkey	*Cercopithecus pogonias*	8	1	0	1	1	0	5
17	Mandrill	*Mandrillus sphinx*	18	2	6	1	1	2	1
18	Northern talapoin	*Myopithecus ogouensis*	3	0	0	0	0	0	3
19	Monkey		35	4	3	7	6	2	2
	Primates		**96**	**12**	**13**	**13**	**11**	**11**	**10**
20	Gaboon viper	*Bitis gabonica*	4	1	1	1	1	0	0
21	Dwarf crocodile	*Osteolaemus tetraspis*	16	2	1	2	3	0	4
22	Nile monitor lizard	*Varanus niloticus*	19	2	1	1	3	2	7
23	Python		11	1	0	3	3	0	0
24	Snake		1	0	0	1	0	0	0
25	Tortoise		12	2	0	0	1	0	8
	Reptiles		**63**	**8**	**4**	**6**	**10**	**2**	**19**
26	Swamp otter	*Aonyx congica*	1	0	0	0	0	0	1
27	African civet	*Civettictis civetta*	2	0	1	0	0	0	0
28	Servaline genet	*Genetta tigrina*	2	0	0	0	0	1	1
29	African palm civet	*Nandinia binotata*	1	0	0	0	0	1	0
30	Tree pangolin	*Phataginus tricuspis*	1	0	0	0	1	0	0
31	Giant pangolin	*Smutsia gigantea*	3	0	0	1	1	0	0
32	Pangolin	*Pangolin sp.*	21	3	0	4	3	4	5
	Carnivores		**31**	**4**	**1**	**4**	**4**	**6**	**6**

Table 4: Data for dependent factors used to analyse bushmeat market relations. H=Hevecam; I=Ipono; C=Campo; A=Akom II; M=Ma'an. Diversity is average number of species found at the site for each month, prices are indexed.

Month	Diversity					Price					% Smoked				
	H	I	C	A	M	H	I	C	A	M	H	I	C	A	M
Aug-01		2.5	1.8	3.3			105	100	100			11	12	86	
Sep-01		3	2.7	3.3			100	144	101			14	0	88	
Oct-01	2.7	3			3.5	104	103			83	67	58			24
Nov-01	1.8	3.5	3.3	2.5	4	106	105	109	102	81	34	21	37	100	20
Dec-01	2.3	4.8		2.5	4.5	106	103		93	89	56	24		100	8
Dec-01	2		4	2.3	2.3	101		124	90	95	79		3		17
Jan-02	2		3.3	1.3		102		130	92		83		31		
Feb-02	1.7		4	1.5	3	105		132	97	93	56		19	100	22
Mar-02	1.4	2.5	4.7	1.8		102	101	119	97		66	43	12		
Apr-02	1.6		4	1.3		103		111	92		33		3		
May-02	1.8	4.3	3.4	1	2.3	95	96	106	103	95	59	14	5		11
Jun-02		5	2.5				100	120				28	10		
Jul-02		4	3				99	121				22	16		
Aug-02		2.7	4				103	147				19	0		

The CCA triplot (Fig. 3) clearly indicated that Hevecam, the most populated site, was associated with a high percentage of smoked meat, whereas this was lower in Campo and Ipono, and least with Ma'an. Price was negatively associated with Ma'an, the most remote site. The diversity of bushmeat supply was positively associated with Campo and Ipono, and negatively associated with Hevecam, Akom II, and Ma'an.

Figure 3: CCA triplot illustrating associations between market sites (large), dependent (italics) and independent factors and bushmeat species found at the market sites. The first two axes explain 76.4 % of the variance. % smoked = state of supply; humpop = human population; kmNP = distance to NP; kmKribi = distance to city; wilddens = wildlife density; Diversit = diversity of bushmeat supply. Rodents (numbers 1-4) are marked with +. Numbers refer to species (see Table 3 for list of names).



In the GLM, seven of the twelve expected correlations were found to be significant (Table 5). However, only three correlations had the same sign (+/-) as expected. These significant correlations supported the two hypotheses that price will rise when the market is near a city, and will fall when the market is near a National Park. The hypothesis that with decreasing wildlife density, the percentage smoked meat at the market would increase was also accepted.

A positive correlation was found between diversity at the market and wildlife density. The expected positive correlation between human population size and the % smoked meat was not found: with increasing human population size the proportion of smoked meat decreased. The percentage smoked meat did not increase either when nearing a city; on the contrary, a positive correlation was found between the distance of the city and the percentage smoked meat. Wildlife density showed a positive correlation with price, where a negative correlation was expected.

Of the four independent factors, wildlife density was significantly correlated with all dependent factors, followed by distance to the city with two significant correlations. Population size showed only one significant unexpected correlation. Distance to the Park showed only one significant expected correlation.

Table 5: Expected and found correlations between dependent and independent factors at the bushmeat market. GLM-diversity: $F_{1,\ 45}=28.77$, $p<0.001$, adj. $R^2=0.39$; GLM-price: $F_{3,\ 43}=28.45$, $p<0.001$, adj.$R^2=0.67$; GLM-state: $F_{16,\ 25}=20.24$, $p<0.001$, adj.$R^2=0.88$.

	Population size		Distance to city		Distance to NP		Wildlife density	
	expected	found	expected	found	expected	found	expected	found
Diversity	+		–		+		–	+
Price	+		–	–	+	–	–	+
State (% smoked)	+	–	–	+	+		–	–

The CCA triplot illustrated relations between sites, factors and species (Fig. 3). The occurrence of rodents at the bushmeat market was mostly explained by the factors on the second axis. The occurrence of rodents at the market sites is positively correlated to human population size, price of the bushmeat, and percentage smoked meat, and was negatively correlated to wildlife density and distance to the city. The latter was confirmed by a GLM which illustrated a negative correlation between distance to the city and occurrence of rodents ($F_{1,\ 45}=11.74$, $p<0.005$, adjusted $R^2=0.19$), hence the closer to settlements the more rodents appear at the market.

Discussion

Based on the relations that were found, we confirm that the state and diversity of the supply at a bushmeat market are two useful factors to indicate the state of exploitation. The percentage of bushmeat that is smoked is a good indicator of the wildlife density in the area surrounding the market. High wildlife densities are associated with low percentages of smoked meat. The state of the supply can therefore be used as an indicator of the abundance of wildlife in the area. When comparing markets, the percentages of smoked meat indicate which markets are situated in areas where wildlife is still abundant or has become scarce.

The diversity of species for sale at a bushmeat market in itself is not a clear indicator. As this diversity changes with exploitation level, a momentary survey of diversity alone is not enough to indicate the exploitation state. When, however, diversity is combined with other factors indicating hunting pressure, such as % rodents and % smoked meat, then diversity can be indicative of the state of exploitation. High percentages of rodents for sale at the market are strongly associated with an increasing human population pressure (Fig. 3) and with percentage of smoked meat (Fig. 3, and GLM). The latter two indicate a high level of exploitation. Hence, when diversity of supply is low, and percentage of smoked meat is high, one can assume that the level of exploitation is very high, indicating that all that is left of wildlife are small mammals, which is expressed in a high percentage of rodents at the supply (situation A in fig 1). If the diversity is low, and the percentage of smoked meat is also low, than one can assume that the bushmeat market is in the situation B (fig 1); hence, the hunting pressure is low and wildlife density is high. The percentage rodents for sale should be low in this situation. Therefore, diversity alone is not enough, but in combination with other factors gives a good indication of the state of exploitation.

The percentage of rodents is clearly indicative of the state of wildlife exploitation, as shown by the significant linear model, as the distance from a city increases, the percentage of rodents at the supply declines, indicating a decrease in hunting pressure. This is in line with what is found in Equatorial Guinea (Fa *et al.* 2000).

The relation between price and wildlife density was contrary to what was expected. The hypothesis that prices are negatively correlated to the abundance of wildlife in the catchment area is therefore rejected. One might then consider that to consumers “all meat is equal”, whether from rare mandrills or common rats: most likely there is full substitution. Price then is not a good indicator. Apparently there are other factors that influence the price of bushmeat more, for example taste preferences and transport costs (Cowlishaw *et al*. 2005b). The only relations that were as expected had an indirect character, namely the link between price and the source and sink. In this study the source was the National Park; here prices were low, whereas at the sink, the nearest city (Kribi), prices were higher. One indirect link is that when wildlife is eradicated near a city, bushmeat needs to be transported from further away, which as a consequence raises the price of the products (Milner-Gulland & Clayton 2002, Cowlishaw *et al*. 2005b).

This results in a set of four variables that could be used to assess the state of exploitation of a bushmeat market: the price, the percentage of smoked meat, the percentage of rodents, and in combination with one or both of the latter two factors, the diversity of the supply. Each of these variables is easy to obtain in a bushmeat market survey. It is required that the factors need to be collected at least twice, either at multiple sites within the study area, or twice at the same site, with sufficient time between the surveys. This is necessary to validate the position of each site. In this study, the two sites Akom II and Hevecam are in the situation of overexploitation (situation A in Fig. 1), as % rodents and % smoked are high at these sites and diversity is low (Fig. 3, Table 4), and Ma'an is in the (relative) safe situation B (Fig 1.), as % smoked, % rodents and prices are the lowest of all sites.

Although the evidence presented here validates our selected independent factors as useful indicators for quantifying the state of exploitation, there are many more factors. Several studies analyzed the relative importance of a number of variables that described a bushmeat market. Several examples are: ethnicity of consumers and producers (Fa *et al.* 2002); income of consumers and providers (East *et al.* 2005); economic activity (Basset 2005); hunting technology (Damania *et al.* 2005); or transport costs (Cowlishaw *et al.* 2005b). Although these variables (similar to the independent factors used here), have been proven to be linked to exploitation levels, they are not suitable as indicators of changes in the state of exploitation. For example, a raise in income of consumers will increase bushmeat consumption (East *et al.* 2005), but when income is known at a given site, not much can be said about the state of the exploitation, except when compared at different times or sites. What is shown in our study, however, is that it is not necessary to collect data on all these (independent) variables (e.g., income, taste preference, hunting method, etc.) in order to get an indication of the state of the exploitation. There are only a few dependent variables needed to obtain this indication. We therefore argue that building a large model with as many factors as possible will not enhance the reliability of such a study. Selecting a few easily obtainable variables (in this case price, state, diversity of the supply and percentage of rodents), should be more cost-efficient than to set up a full blown monitoring system, covering all factors involved in the market chain. This is supported by the fact that large scale monitoring studies do not seem to offer a satisfactory tool (e.g., Crookes *et al.* 2006; Milner-Gulland & Clayton 2002). We do not deny the importance of long-term monitoring studies, but when the aim is to get a quick impression of the state of a given bushmeat market and its resources, it pays to select a few factors that are known to indicate the state of the bushmeat exploitation.

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Next page: spoor plot along the road

Chapter 5

EFFECTS OF ROADS ON RAINFOREST WILDLIFE, TO CROSS OR NOT TO CROSS?

Christiaan A. van der Hoeven, Willem F. de Boer and Herbert H. T. Prins

Abstract

The negative effects of roads on wildlife in tropical rainforests in Africa are poorly understood. Road construction has high priority in Africa, with as effect that negative impacts of roads on wildlife often are neglected. This study aims at providing information on the effects of roads on crossing behaviour of rainforest wildlife. Crossing probability of forest wildlife was analyzed for association with ten different factors that were linked to road presence or road construction. Factors were divided into three classes: vegetation cover, topography and human influence. A spoor plot survey was done in southern Cameroon, Africa. Spoor plots were laid along a 32 km unpaved logging road that intersects Campo-Ma'an National Park. Tracks of several species were found frequently (e.g., genets and porcupines), while others were found only sporadically (e.g., forest duikers and apes). Differences in crossing behaviour between plots along the road and in the forest interior supported the hypothesis that the presence of a road acts as a barrier for most species. The actual physical obstacles found along the road (e.g., logs, banks, etc.) proved to be highly negatively correlated with crossing probabilities. The assumption that invasive species found in road side vegetation reduces wildlife crossings was rejected as high vegetation cover, regardless of species, was positively correlated to crossing probability. This study proves that roads have a large impact on wildlife, and indicates which factors could be altered during road construction and maintenance in order to mitigate these impacts, such as to maintain a high vegetation cover at shrub level up to the road, and to prevent the roadside from being blocked during construction.

Introduction

The promises a road brings are dualistic, on the one hand a road provides prospects of development and better access to services such as markets, schools and primary health care for the local population, but on the other hand they pose a serious threat to biodiversity. On a national level, roads are seen as a means to exploit rich natural resources with large economic profits shining at the horizon, such as timber and minerals. It can be argued that roads are necessary for the development of a country, but the threat that they pose to biodiversity should not be underestimated and be addressed adequately (Chomitz & Gray 1996; Wilkie *et al.* 2000). There are many ecological effects of roads on biodiversity, which for Europe, North America and Australia are well described in literature (for example, Forman & Alexander 1998; Spellenberg 1998; Trombulak & Frissel 2000). In Central Africa, a road can be seen as an alien object in a further continuous forest zone (Picture 7, p. 100). Most species that live in rainforests are adapted to an environment associated with low light levels and temperature, high humidity, amongst others. Roads form a completely different environment with a different microclimate (Trombulak & Frissel 2000). The faunal community in the forest interior is expected to differ from the one at a road. In other words, there are species that are expected to avoid roads, as well as species that are not disturbed by roads, or even attracted to it (Malcolm & Ray 2000). Mitigating actions are known (Burnett 1992; Alexander & Waters 2000; Malcolm & Ray 2000), implemented, and tested (Little *et al.* 2002) and some western countries have specific protocols concerning road planning and construction (e.g., Netherlands: Cuperus 2004).

In tropical rainforest areas in the developing world, however, much less studies have been done on the ecological effect of roads on biodiversity. In Latin America several studies on tropical birds and small mammals have been conducted (e.g., Burkey 1993; Laurance *et al.* 2004; Pinowski 2005), few in Asia (e.g., Kerley *et al.* 2002) and Africa (e.g., Malcolm & Ray 2000). Considering the attention for tropical rainforests as biodiversity hotspots (Myers *et al.*, 2000) and the threats to these ecosystems posed by an ever increasing road network (Chomitz & Gray 1996), the lack of attention on the ecological impacts of roads in tropical rainforests is alarming. These roads form barriers that for wildlife are often uncrossable, splitting the original population in several sub-populations. This barrier effect tends to form metapopulations: a collection of isolated subpopulations that are individually more sensitive to decline and extinction than the original population (Forman & Alexander 1998). Although genetic isolation is likely not a serious threat (Burnett 1992), local extinction of cut-off populations is.

This study concentrates on the effects of roads on wildlife crossing in Central African rainforests. The predominant infrastructures in these rainforests are unpaved (public) roads with low traffic intensity, often constructed and maintained by logging companies. There are measurable physical factors which influence the road-crossing behaviour of wildlife, which we divided in three classes: vegetation cover, topography (the arrangement of the natural and artificial physical features of an area) and human presence. The first, vegetation cover, is an important variable that characterizes the forest bordering the road. Several studies indicated clear correlations between vegetation cover and composition at different heights and distances from the road, and crossing behaviour of small mammals and birds (Goosem 2000; Malcolm & Ray 2000; Laurance *et al.* 2004). High density of understorey vegetation, low tree diversity, and low canopy cover are indications of disturbance and are related to road-avoidance behaviour, except for certain rodent species that show increased abundance in disturbed vegetation (Malcolm & Ray 2000). Road construction can also facilitate settlement of invasive species (Spellenberg 1998; Pauchard & Alaback 2004). The invasive shrub *Chromolaena odorata* (see chapter 6 of this thesis) could have a double negative effect: it not only indicates disturbance, but also displaces native vegetation that is a resource for local fauna. It was expected that areas with high infestation rates of this invasive shrub would have lower wildlife crossing probability, compared to road sides with native vegetation.

Topographical factors are, first, the across road distance between mature trees on each side of the road, which we use as an indicator of the real width of the road, as effects extend past the road surface. The larger this across road distance between mature rainforest trees, the larger the barrier is assumed to be. A second inhibiting topographic factor for animals can be the occurrence of obstacles as a result from road construction, such as pushed over logs, banks and trenches (called "slash piles" by Malcolm & Ray 2000). These barriers literary block animals in their movement. It is a regular feature along unpaved logging roads: during construction, but also during maintenance, bulldozers push debris and cut logs to the side. Sometimes deep trenches are dug next to the road for drainage of runoff water.

Apart from the road presence itself as factor of human presence, we assume that the distance to the nearest human settlement influences crossing behaviour too. As crossing is risky, we expect less crossings near human settlements, because human presence ads to the threat.

We hypothesize furthermore that the invasion of exotics in roadside vegetation enhances the barrier effect of a road on wildlife. In this study the most important invasive species is the shrub *C. odorata* which is a very aggressive colonizer that grows in sunlit places like road sides, clearings, and old pasturelands (Goodall & Erasmus 1996). African gingers (Zingiberaceae) form the main part of the native roadside vegetation and are an important food resource for much wildlife (Tutin *et al.* 1997; Doran *et al.* 2002). Zingiberaceae are currently displaced by *C. odorata* at an alarming rate (view chapter 6).

Methods

The study was conducted in Campo-Ma'an National Park (CMNP), located in the Southwest corner of Cameroon (2°21'29 N, 10°09'05 E) and covers 2,640 km^2. The CMNP was established in January 2000 and is covered mainly with Guinea-Congolian rainforest (Tchouto *et al.* 2006). The altitude ranges from 0 to 1100 m; a mountain chain extends from Southwest to Northeast. Annual rainfall varies from 1670 to 2950 mm. Part of the CMNP was logged several years prior to its gazetment. There are four logging concessions that border the park at the East and West side. An old logging road that traverses the CMNP from west to east was reopened in 2000 to allow logging transport from the eastern concessions to the port and sawmill at the coast in the west. This road is the subject of our study. The use of this unpaved road was strictly regulated; passage occurred in convoys accompanied by forestry guards only in the morning and afternoon. The road was closed and guarded from 17.00 hrs to 08.00 hrs, local time. The road is 32 km long, and divides the CMNP in two: a northern part (approximately 75%) and a southern part (approximately 25%).

Road crossing frequency by forest wildlife was studied in March 2004 with a spoor plot survey along the road and in the adjacent forest interior. A total of twenty spoor plots were laid out along the road. A spoor plot was 10 m long, 1 m wide (10 m^2), with a 3 cm sand layer. The sand was specifically selected for its consistency in dry and wet conditions and was found outside the CMNP. It was loamy sand with a small clay fraction, which leaves clear prints even from small animals. Persistence of tracks was tested before data collection.

Road spoor plots were laid out on the road surface, as close as possible to the edge to avoid disturbance by traffic, but runoff ditches were avoided because of their inclination. Plots were constructed horizontally to prevent plot erosion.

The ground surface for the plot was first cleared from grass, small herbs and loose detritus to prevent weeds form penetrating the sand layer, and to provide a solid base. Single stems of woody vegetation and large herbs (e.g., gingers) as well as large loose objects (e.g., logs and rocks) were left in place, as long as it did not prevent animals from crossing. Most road plots were on already relative clear ground, which reduced disturbance and dissimilarity form the surroundings (Picture 8, p. 101).

Six control plots were laid out in the forest interior, at regular distances from the road; three on the north side, three on the south side. Plot locations were selected that were relatively level, in not too dense vegetation. Here, also detritus and herbs (if present) were removed. Stems and large logs were left in place (Picture 1, p. 98).

Data collection was performed daily over 17 days. Data collection started the day after a plot was finished. As a limited number of plots could be constructed per day, several plots have a different number of observation days. The first constructed plot was observed for 17 days, the last plot for 11 days. After 17 days, plot surfaces became too much invaded by seedlings, and the study had to be terminated. The order of sampling changed each day, one day sampling would start at plot 1, up to plot 26, the next day in reverse order, form plot 26 to plot 1. This was done to decrease bias due to the hour of sampling. Sampling of the first plot would be at around 8.30 am, the last plot around 12 am local time. For consistency of track data, the same tracker accompanied the researchers when plots were sampled. Plot number, time, and if tracks were found, species and maximum number of individuals was noted. After data collection the plot was cleared of tracks by raking and incidental damage was repaired (e.g., when an elephant or rains had damaged the plot).

The physical surroundings of each plot were described by ten factors divided in three classes: vegetation, topography, and human pressure (Table 1). Vegetation cover was estimated at 7 m (road side) and 15 m (forest edge) from the road. Cover was estimated for shrub and tree layer together, indicating general forest structure, and for *C. odorata* and African gingers (Zingiberaceae) indicating the effect of infestation by invasive species. Secondly, topography was described by the presence or absence of a physical barrier within 5 m of the road side behind the plot, and the gap size between mature trees on either side of the road. A barrier could be a bank made by bulldozers during road construction; it could be logs that lie parallel to the road, deep trenches, or anything that could prevent animals from crossing (Pictures 5 & 6, p.100). A barrier was quantified as covering more or less than 40 % of the approach to the road behind the plot. The distance between mature rainforest trees across the road was estimated and expressed as gap size in meters. The presence of mature, large trees is an indication of the state of the forest (disturbed or not); therefore, pioneer species (e.g., Umbrella tree *Musanga sp.*) were not considered as trees. Thirdly, the distance to the nearest Park entrance was recorded.

Table 1: Classification of independent factors defining wildlife crossing probability.

Class	Factor	Abbreviation	Unit
Vegetation Cover	*C. odorata* at 7 m	PC7	%
	C. odorata at 15 m	PC15	%
	Zingiberaceae at 7 m	PZ7	%
	Zingiberaceae at 15 m	PZ15	%
	Trees & Shrubs at 7 m	TS7	%
	Trees & Shrubs at 15 m	TS15	%
Topography	Barrier	barrier	Yes or no
	Gap size between mature trees	gap	m
Human pressure	Distance to nearest park entrance	dist	km
	Presence or absence of the road	road	Yes or no

The number of track data per species was expected to vary widely, so we categorized the tracks in 14 species groups (Table 2). Categorization was based similarity in behaviour, ecology or physical characteristics (based on Kingdon 1997) and on our expectation that species or species groups would show different crossing behaviour in relation to the above-mentioned factors.

We expected species that are group-living, nocturnal, terrestrial, and/or that are large-bodied to cross roads more easily, while solitary, diurnal, arboreal, and/or small animals were expected to cross a road less. Group-living animals or animals with a larger body weight are expected to be less vulnerable to predation when crossing open spaces (Burnett 1992). Arboreal animals, even within their own habitat, are often reluctant to descend towards the ground, let alone cross open spaces that are alien to them, while terrestrials are expected to have less problems travelling on the ground. Under the cover of the night, crossing open spaces is expected to be less dangerous than in full daylight.

Number	Category
1	<3kg
2	>3kg
3	<5kg
4	>5kg
5	terrestrial
6	arboreal
7	nocturnal
8	diurnal
9	group
10	solitary
11	monkey
12	ungulate
13	rodent
14	carnivore

Table 2: Categories of species groups based on similarity in behaviour, ecology or physical characteristics.

Statistical analysis

Data were first transformed to a binary presence-absence table with crossing observations for each of the species categories per plot. The proportion of days with and without tracks was used in the analysis to correct for the difference in sample size, and to avoid pseudo replication. A stepwise generalized linear model, with a logit-link to accommodate for the binary character of the data, was used to analyze the data. Both forwards and backwards regressions were performed, selecting the best model based upon the log-likelihood and the Pearson chi square. Wald statistics ($p<0.05$) were used to test for significant correlation between the number of tracks found and each of the ten factors.

There were three possible outcomes for the regression analysis on the factors and crossing behaviour for the different species groups: either a significant correlation between factors and crossing behaviour was found in correspondence to our formulated predictions, or a significant unexpected correlation was found in contradiction with the formulated hypothesis, and the last possibility was that no significant correlation was found at all. For five out of ten factors a significant negative correlation with crossing probability was expected. If, for example, cover of *C. odorata* increased, then crossing probability was expected to decrease. The other five factors were expected to show a significant positive correlation coefficient with crossing frequencies. An increase in the distance to the park entrance, for example, was expected to increase crossing probability (compare Table 4).

A sign test was done to test for the effect of each factor. We tested for direction of difference (+ or -) for each factor over all 14 species groups.

Results

Tracks of 28 different wildlife species were recorded on 26 plots in 17 days. Tracks were found in 46% of the observation days. The average number of tracks was 1.57 per plot per day, with a total of 287. A few species were recorded very frequently (e.g., genet and porcupine, Fig. 1), while most were only seen a few times (gorilla and buffalo, Table 3).

Table 3: Species and the number of tracks (n) recorded in the spoor plots and their classification in different species groups. 1=yes, 0=no; terr/arbor = terrestrial/arboreal; noc/diur = nocturnal/diurnal; group/solo = group-living/solitary

Species		**n**	**body size**	**<3 kg**	**<5 kg**	**terr/arbor**	**noc/diur**	**group/solo**	**monkeys**	**ungulates**	**rodents**	**carnivores**
Genet		65	<5	1	1	T	N	S				X
Brush-tailed porcupine	*Atherurus africanus*	22	2.8	1	1	T	N	G			X	
Blue duiker	*Cephalophus monticola*	20	5	0	1	T	D	S		X		
Mouse		19	<5	1	1	T	N	S			X	
Palm civet	*Nandinia binotata*	17	2.6	1	1	A	N	S				X
Giant pouched rat	*Cricetomys emini*	15	1.4	1	1	T	N	S			X	
Squirrel		15	<5	1	1	A	D	S			X	
African civet	*Civettictis civetta*	13	14	0	0	T	N	S				X
Pangolin		13	2.5	1	1	A	N	S				X
Red-capped mangabey	*Cercocebus torquatus*	10	8	0	0	A	D	G	X			
Mandrill	*Mandrillus sphinx*	10	15	0	0	T	D	G	X			
Snake		10	<6	0	1	T	D	S				X
Sitatunga	*Tragelaphus spekei*	10	71	0	0	T	N	G		X		
Monitor lizard	*Varanus niloticus*	9	5	0	1	T	D	S				X
Mongoose		8	<5	1	1	T	N	S				X
Elephant	*Loxodonta Africana*	5	2100	0	0	T	D	G		X		
Small mammal		5	<5	1	1	T	N	S				
Chimpanzee	*Pan troglodytes*	4	35	0	0	T	D	G	X			
African buffalo	*Syncerus caffer nanus*	4	320	0	0	T	D	G		X		
Monkey		3	<6	0	1	A	D	G	X			
Bay duiker	*Cephalophus dorsalis*	2	19.5	0	0	T	N	S		X		
Duiker		2	>5	0	1	T	D	S		X		
Peter’s duiker	*Cephalophus callipygus*	1	19.5	0	0	T	D	S		X		
Yellow-backed duiker	*Cephalophus sylvicultor*	1	62.5	0	0	T	N	S		X		
White-nosed monkey	*Cercopithecus nictitans*	1	5.5	0	1	A	D	G	X			
Gorilla	*Gorilla gorilla*	1	109	0	0	T	D	G	X			
Dwarf antelope	*Neotragus batesi*	1	4	0	1	T	D	S		X		
Potto	*Peridicticus potto*	1	1.2	1	1	A	N	S	X			

Three factors clearly showed the strongest significant correlation with crossing behaviour: “Percentage *C. odorata* cover at 15 m” (PC15); the presence of a barrier, and the presence of the road (Table 4). The first factor showed, against expectations, a positive correlation with the crossing probability for five categories, and only one negative correlation, with the category < 3 kg. Barrier and road presence showed a clear negative correlation, as was expected, for 6 and 8 out of 14 categories respectively.

One factor, “Percentage Tree & Shrub cover at 7 m” (TS7), significantly increased the crossing probability for two species categories; > 3 kg and group-living animals (Table 4).

Two factors, “Percentage Tree & Shrub cover at 15 m” (TS15) and “Percentage Zingiberaceae cover at 15 m” (PZ15), increased significantly the road crossing chances for respectively monkeys and group-living animals.

One factor, distance to nearest Park entrance (dist), showed an unexpected significant negative correlation for two categories: diurnal species and monkeys.

Two factors showed no significant correlation in the first model, but the sign test resulted in a positive correlation for: “Percentage *C. odorata* cover at 7 m” (PC7), and a negative correlation for “Percentage Zingiberaceae cover at 7 m” (PZ7). Gap size was not correlated to any category.

In the sign test, three factors were significantly positive correlated with crossing probability (PC7, PZ15 and TS15), and four factors were significantly negative correlated with crossing probability (barrier, road, PZ7 and distance). Three factors were in contradiction to our hypothesis: “Percentage *C. odorata* cover at 7 m” (PC7) was positive correlated with crossing probability, while “Percentage Zingiberaceae cover at 7 m” (PZ7) and distance were negatively correlated with crossing probability. Barrier and road showed as expected a significant negative correlation with crossing probability, while “Percentage Zingiberaceae cover at 15 m” (PZ15) and “Percentage Tree & Shrub cover at 15 m” (TS15) showed the expected positive correlation with crossing probability.

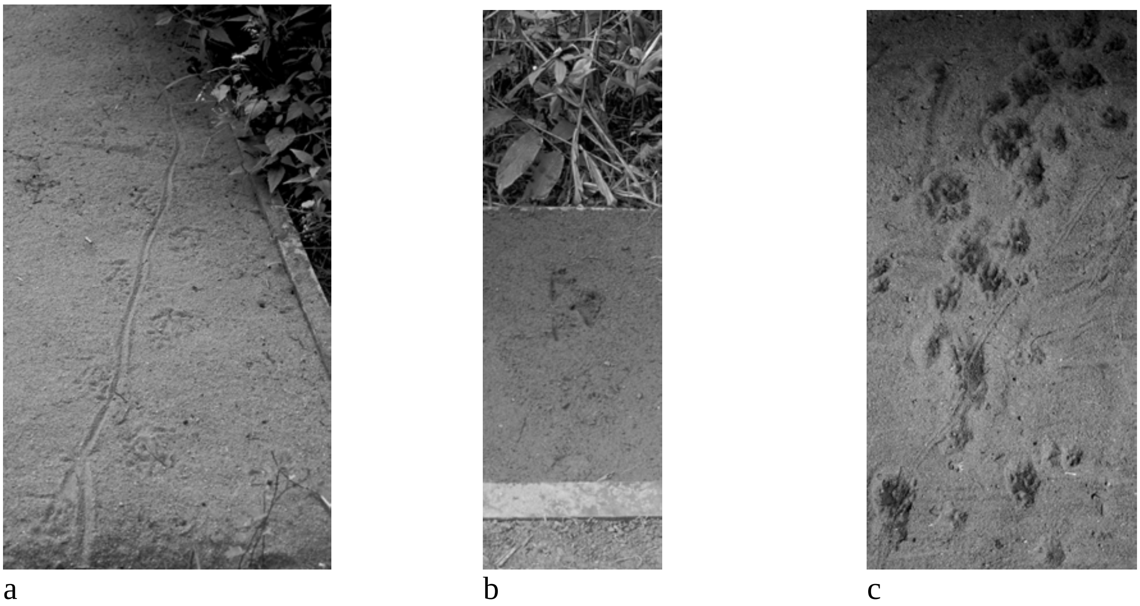

a b c

Figure 1: examples of tracks recorded at the spoor plots: monitor lizard (a), red-capped mangabey (b), and genet (c).

Discussion

The road presence seems to be the strongest indicative factor for crossing behaviour: for 8 out of 14 species categories the presence of the road significantly decreased the crossing probability. This confirms our hypothesis that the presence of a road forms a barrier for many wildlife species.

Road side associated barriers were, as expected, negatively correlated with crossing probability: when the barrier covered more than 40 % of the area behind the plot, crossing probability decreased significantly for 6 out of 14 species categories. It should be realised that these barriers often consist of piles of broken trees and branches, uprooted stumps, earth mounds and rocks, and are overgrown with herbs (Pictures 5 & 6, p.100). They can form serious impediments for wildlife, and often cannot be scaled or climbed over. With this result we conclude that a road (or its side-effects) forms a physical barrier, although we do not deny the importance of psychological and sociological effects on crossing behaviour (Burnett 1992). It shows here that specification of the predicate “physical” is necessary, preferably in measurable units that quantify the barrier impact. For two categories, “arboreal” and “monkeys”, that show no significant impact of the presence of barriers on crossing behaviour, it could be that a barrier on the ground is irrelevant as these species rarely descend to the forest floor. There was no significant relation found between small animals (<3 kg and <5 kg) and barrier, which may be because small animals can pass easier underneath barriers, or climb over them. This could also explain why “rodents” are apparently not inhibited by barriers either.

The correlation between “Percentage *C. odorata* cover at 15 m” (PC15) and crossing probability was significantly positive for 5 out of 14 species categories, implicating that with a higher cover of *C. odorata* wildlife tended to cross more than when *C. odorata* cover was lower. This is supported by the general tendency of “Percentage *C. odorata* cover at 7 m” (PC7) to be positively correlated with crossing probability (sign test, $p<0.05$, Table 4). The hypothesis that the presence of the invasive species reduces the road crossing frequency was therefore rejected. A possible explanation for this outcome could be that the increased cover of foliage (irrespective of the species) overrules the negative impact of the invasive species on the native vegetation. *C. odorata* forms a dense woven canopy which allows only 7 % of direct sunlight to pass to its understorey (Honu & Dang 2002). Only for small mammals (<3 kg) “Percentage *C. odorata* cover at 15 m” (PC15) seemed to be an inhibiting factor. High PC15 indicates that disturbance is higher, and effective road width is larger than when PC15 is low or absent. Maybe small mammals are more sensitive to a larger strip of *C. odorata*.

The human pressure factor “distance to nearest Park entrance” yielded only unexpected negative relations with crossing probability: the closer to the Park border, the more the “diurnal” and “monkey” species categories crossed. The general tendency for the other categories confirmed this, as most correlations between distance to the nearest entrance and crossing probability were negative (sign test, $p=0.001$). One would expect that wildlife would avoid human settlements (e.g., monkeys: Lahm *et al.* 1998, and elephants: Barnes *et al.* 1991; Blom *et al.* 2004) for reasons of safety against hunters or habitat degradation due to land clearance or other disturbances. One could speculate that poaching takes place further in the Park, where chances on detection are smaller, because guards are stationed near the Park entrances. It may also be that a few species that forage near human settlements (e.g., genet and African civet, Ray & Sunquist 2001) tend to use roads more often, distorting the data, because their tracks are found disproportional more near settlements than tracks of forest species (e.g., forest duikers).

Table 4: Correlations between factors and species categories[a] from a Stepwise Generalized Linear Model[b] and a Sign test[c]. PC7= Percentage *C. odorata* cover at 7 m; PC15= Percentage *C. odorata* cover at 15 m; PZ7= Percentage Zingiberaceae cover at 7 m; PZ15= Percentage Zingiberaceae cover at 15 m; TS7= Percentage Tree & Shrub cover at 7 m; TS15= Percentage Tree & Shrub cover at 15 m; dist=distance to human settlement.

	Negative correlation expected					Positive correlation expected				
Category / factor	PC7	PC15	barrier	gap	road	PZ7	PZ15	TS7	TS15	dist
<3 kg	+	-*	-	-	-*	-	+	-	+	-
>3 kg	+	+*	-*	-	-*	-	+	+*	+	-
<5 kg	+	-	-	-	-*	-	+	-	+	-
>5 kg	+	+*	-*	-	-	-	+	+	+	-
Terrestrial	+	-	-*	-	-*	-	+	-	+	-
Arboreal	-	-	-	-	-	-	+	-	+	-
Nocturnal	+	-	-	-	-*	-	-	-	+	-
Diurnal	+	+	-*	+	+	-	+	+	+	-*
Solitary	+	-	-*	+	+	-	+	-	+	-
Group-living	+	+*	-*	-	-*	-	+*	+*	-	-
Monkeys	+	+*	-	-	-	+	-	+	+*	-*
Ungulates	-	+*	-*	+	-	-	+	+	+	+
Rodents	-	-	-	-	-*	+	-	-	-	-
Carnivores	+	-	-	+	-	-	+	-	+	-
Sign test p[c]	0.029*	0.395	0.000***	0.09	0.006**	0.006**	0.029*	0.183	0.006**	0.001***

[a] += positive correlation, **-** = negative correlation.

[b] Asterix indicates significant correlation, Wald p<0.05.

[c] *:$p<0.05$, **: $p<0.01$ ***: $p=0.001$

Two factors showed no significant correlation in the linear model, but showed opposing correlations in the sign test. “Percentage *C. odorata* cover at 7 m” (PC7) was positively correlated to crossing probability (p<0.05), while “Percentage Zingiberaceae cover at 7 m” (PZ7) was negatively correlated (p<0.01). It would be expected, based on the other vegetation results, that cover would facilitate crossing and PZ7 would also be positively correlated to crossing probability. The negative effect of PZ7 is probably caused by an increase in PC7, which is linked with higher crossing probability. This would explain the increase in crossing probability when PZ7 decreases. This is probably due to the effect hat higher PC7 increases crossing probability,

One factor showed no significant relation with any species category whatsoever: gap size between trees (“gap”). No satisfying explanation can be found for this. Road width has been suggested as an important factor (Forman & Alexander 1998, Laurance *et al.* 2004), but this is not found in this study.

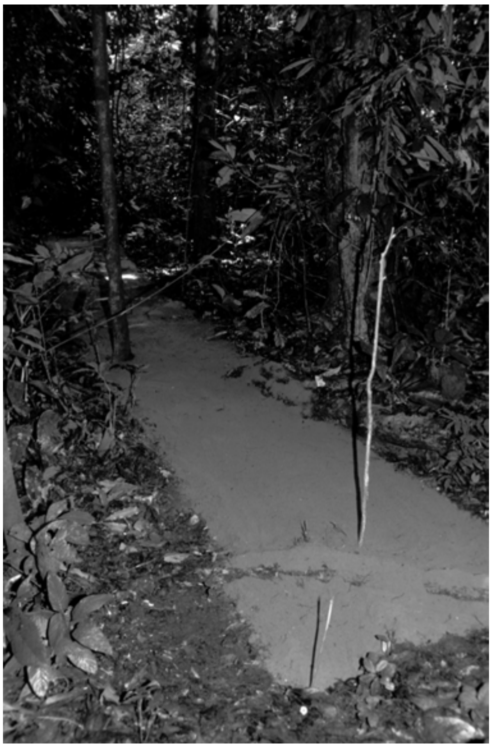
Two species categories showed no significant correlation with any of the factors studied. These were arboreal and carnivorous species. Although it was expected that arboreal species would avoid roads, the spoor plot method probably is not the right way of testing this. In general, arboreal species are expected not to descend to the ground, except, for example, when they need to cross a road. The negative effect of roads on arboreal species is therefore probably masked by the positive effect of these arboreal species descending to the forest floor. This indicates that studies on forest wildlife should approach arboreal species differently than terrestrial species, like for example by setting traps at different heights in the forest (Malcolm & Ray 2000). As for carnivores and scavengers, there are species known to use roads as ways of conduct and as hunting ground (e.g., African civet, Ray & Sunquist 2001), which obliterates the distinction between road and forest, but not enough data were available for each carnivore species to analyze this further. Another explanation is the fact that small animals (mostly rodents, but probably all animals with fixed home ranges) align their home ranges along linear environmental discontinuities (Burnett 1992), which would probably result in an underestimate of crossing frequencies, as these species would remain within their home range.

Although vegetation succession stages are a well-used indicator of disturbances on animal behaviour (Laurance *et al.* 2004; Malcolm & Ray 2000), we found that the factors with the strongest influence are the physical factors: the presence of the road and the barrier alongside it. As no study has focused on the specific physical barrier effects of a forest road, this study forms a clear stimulus for road construction to take into account the secondary effects of road construction. Based on the strong effect of the presence of the road itself, it is doubtful whether negative effects can be fully mitigated. Conscious construction could however, decrease the impact. By avoiding parallel obstacles, such as shoving logs perpendicular to the road, or stacking them away from the road, or by avoiding the construction of deep trenches or steep banks, crossing could be made easier.

There was an indication that tree and shrub cover is important for animals to facilitate road crossing (Figure 2a & b), even if this cover is provided by invasive species. Arguably, wildlife is probably more preoccupied with the potential dangers of lingering near or at a road side than with feeding, which nullifies the negative effect of displacement of the native vegetation by exotics. The general indication that dense vegetation right up to the road is a strong stimulant factor, enhancing wildlife crossings, has clear implications for road side management (Pictures 2 & 7, p. 98, 101).

Figure 2a: A logging road with cleared verges about 5 m. in width. Crossing for wildlife is very risky due to high exposure. Illustration: Hendrik Rypkema.

Figure 2b: A logging road with verges where vegetation is allowed to grow. Crossing the road is far less risky for wildlife in this situation. Illustration: Hendrik Rypkema



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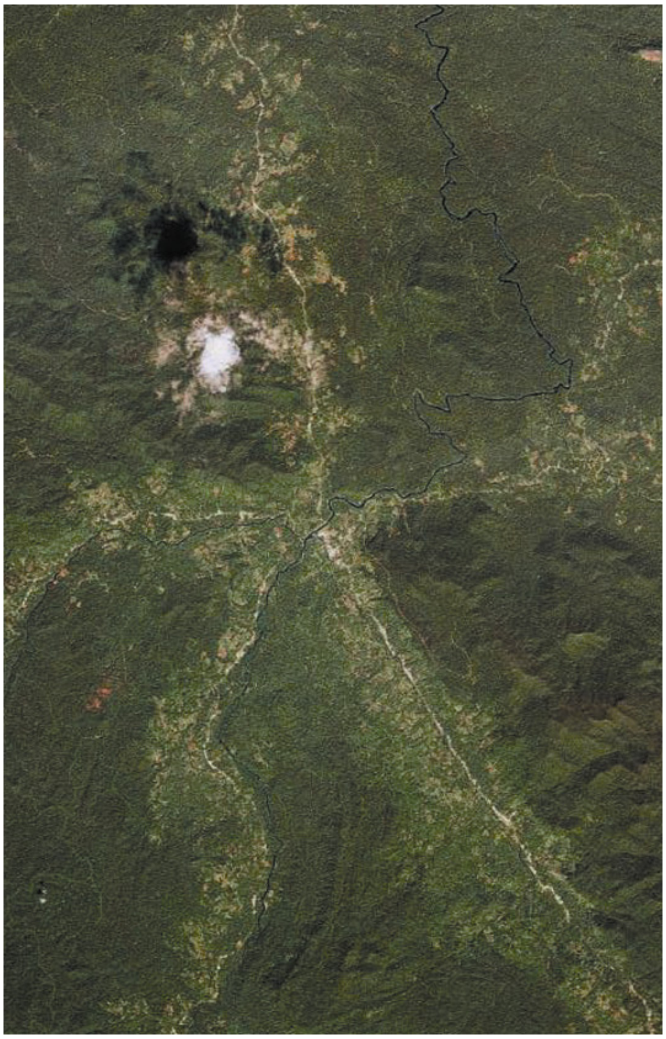
Spoor plot in the forest interior (Picture 1). In collaboration with a logging company vegetation at regular intervals along the road through the Park was allowed to grow up to the road side, forming a “corridor” (Picture 2).

3

Parts of the African civet are used for traditonal medical practices (Picture 3).

5



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Examples of a physical barrier formed by logs and debris (Picture 4) and steep road banks (Picture 5). Deforestation along roads in study area (Picture 6, source: Google Earth)

7

8

9

Logging truck passing through a “corridor” (Picture 7). Laying a spoor plot (Picture 8) by the local team (Picture 9).

10

11

The rare swamp otter (Picture10), and the common, but much hunted giant pouched rat (Picture 11).

Next page: logging trucks passing trhough the Park


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Chapter 6

INVASIVE PLANT SPECIES THREATENS GORILLA IN EQUATORIAL AFRICA

Christiaan A. van der Hoeven and Herbert H.T. Prins

Abstract

The invasive plant species *Chromolaena odorata* threatens the gorilla *Gorilla gorilla* in Central African forests by outcompeting plant species of the African ginger family (Zingiberaceae). These African gingers are the most important food species and therefore form a key resource for this endangered ape. Zingiberaceae occupy sunlit places, such as abandoned fields and road sides. The invasive *C. odorata* occupies the same niche, but is much more aggressive and has outcompeted most Zingiberaceae. Zingiberaceae do not occur in the forest interior, but survive mostly in the forest edge where shade from trees and shrubs prevent *C. odorata* from growing. Abundance of these gingers has been largely reduced in only two years time along a road network in Campo-Ma'an National Park in Cameroon. It is estimated that cover of Zingiberaceae in this area has been reduced by 95%. Main reason for this *C. odorata* dominance is the standing road maintenance policy. Clearing of road sides halts natural succession, which provides the opportunity for *C. odorata* to take over. The invasion of this shrub and subsequent disappearance of the original vegetation threatens the gorilla in their already precarious existence.

Introduction

The western lowland gorilla (*Gorilla gorilla gorilla*) is a species threatened by habitat destruction, hunting and virulent diseases (Huijbregts *et al.* 2002; Walsh *et al.* 2003). The species is marked as endangered on the IUCN red list of threatened species, but experts argue it should be lifted to the critical endangered level because of its rapid decline in the last decade (Butynski *et al.* 2000; Walsh *et al.* 2003). Here we report that an invasive plant species from South America poses a new threat for this charismatic primate. The invasive species is the perennial shrub *Chromolaena odorata* L., and is replacing native ginger species (Zingiberaceae), which are the main food source for gorilla.

At present, *C. odorata* is considered a pest even though it was introduced as a cover crop in coffee and oil palm plantations in West Africa in the 1950s (De Rouw 1991). This species is listed as one of the 100 "World's Worst" invaders by the Invasive Species Specialist Group of the World Conservation Union (www.issg.org). It has invaded vast areas of the tropics (McFayden & Skarratt 1996). *Chromolaena odorata* is a heliophytic species, with highest reproduction and germination success in places with much sunlight, such as road sides and fallow fields (Epp 1987; De Rouw 1991; Witkowski & Wilson 2001). It does not produce seed below a light intensity of 6.5 mJ m^{-2} day^{-1} (Joshi *et al.* 2006). It does, however, occur in old secondary forests and even in shaded forests, where sterile plants and a seedbank form the dormitory source for future (re-) colonisation when a gap in the canopy appears (Witkowski & Wilson 2001; Joshi *et al.* 2006). The species does not occur above 1000 m altitude or in areas with extreme drought (Kushwaha *et al.* 1981; McFayden & Skarratt 1996). *C. odorata* is characterised by a high reproductive potential, excellent dispersal, fast germination and rapid initial growth (Joshi *et al.* 2006). Therefore, when conditions are favourable *C. odorata* will form a dense canopy, which suppresses the development of seedlings underneath its foliage (Honu & Dang 2002). Native species are outcompeted and the original vegetation changes into a mono-specific *C. odorata* stand (Fig. 1). This has already led to problems for Nile crocodile reproduction (Leslie & Spotila 2001), but also for grazing and browsing by large herbivores in Africa (Witkowski & Wilson 2001).

C. odorata is currently invading the forests in the rainforest area of southern Cameroon. These invasions take place in typical slash-and-burn situations, but also in the old secondary forest (pers. obs.). The persistence and propagation of *C. odorata* is stimulated by standing policy of road maintenance in rainforests, which constitutes regular mowing of the road sides. This regular mowing maintains the road side vegetation in a permanent state of arrested succession, which favours *C. odorata* dominance (Kushwaha *et al.* 1981; De Rouw 1991). A similar effect is found in Nepal where grazing replaces mowing (Joshi *et al.* 2006). *Chromolaena odorata* invasion resulted in the near-disappearance of wild gingers because this species occupies the same niche as Zingiberaceae do (Fig. 2). In the African rainforest Zingiberaceae typically occur in sun-lit places like road verges, old logging tracks and abandoned cultivation (White *et al.* 1995). Zingiberaceae are outcompeted in this heliophytic habitat by the more rapidly growing and faster reproducing *C. odorata*. Wild gingers do not occur in closed forest either, and are hardly found in natural tree-falls. Here Marantaceae (e.g., *Haumania sp.*, *Megaphrinium sp.*) dominate the herb/shrub layer and form an important gorilla food source (White *et al.* 1995; Doran *et al.* 2002).

The stem, pith and fruit of most Zingiberaceae provide major food for gorilla (Doran *et al.* 2002; White *et al.* 1995) and other large wildlife (Tutin *et al.* 1997). Gorilla depends on Zingiberaceae, mostly *Aframomum sp.*, as building material (Tutin *et al.* 1995; Mehlman & Doran 2002) and as food source, especially at times when fruit is scarce (Doran & McNeilage 1998). These Zingiberaceae species are therefore considered key species or important fallback species, which sustain the apes in periods of food scarcity (White *et al.* 1995). Ape diet is generally quantified by the use of relative occurrences of food species in dung samples or in feeding trails. Food species are considered important when they rank high in counts or samples. Doran *et al.* (2002) found *Aframomum* in dung samples on more than 50% of the days sampled and in more than 30% of the dung samples, ranking this species highest. In 63% of the feeding trail samples, one or two species of *Aframomum* were found. Only *Haumania danckelmaniana* (Marantaceae) was encountered more (79%, Doran *et al.* 2002). Rogers *et al.* (1990) ranked Zingiberaceae even as number one food species, with 87% of the feeding trail surveyed containing *Aframomum spp*. Based on a study in Gabon Tutin *et al.* (1997) called *Aframomum spp.* keystone food species. Overall indication is that Zingiberaceae species provide a permanent bulk resource on which gorilla rely, especially during fruit-scarce times.

Because Zingiberaceae provide a key resource for gorillas, their disappearance threatens the subsistence of the ape. We compared cover of these native gingers, Marantaceae and the invasive *C. odorata* to verify this threat to the ape in the rainforest zone of Cameroon.

Methods

The study was conducted in March 2004 in Campo Ma'an National Park in southern Cameroon (2°19'N, 10°12'E). The area represents a Guinea-Congolian lowland rainforest (Tchouto *et al.* 2006). Selective logging took place until 1999, a year prior to the establishment of the park. The density of gorilla, chimpanzee (*Pan troglodytes*), mandrill (*Mandrillus sphinx*) and other vertebrates was established between 1998 and 2002 (Van der Hoeven *et al.* 2004). Besides *Chromolaena odorata* and Zingiberaceae, cover of Marantaceae species was also estimated as this family provides together with Zingiberaceae the main food resource for gorilla. Preliminary observations suggested a strong light dependency of both the indigenous Zingiberaceae and the invasive *C. odorata*. Three separate areas were therefore sampled: road verge, forest edge and forest interior. These areas are characterized by an increase of vegetation cover and therefore a decrease of light level at the forest floor. At each sample location, we made cover estimates of *C. odorata* and of the different functional groups using a modified Londo-scale (which uses classes of 10 percent points). The functional groups we discerned were ferns, grasses, Zingiberaceae, Marantaceae, shrubs (woody species shorter than 5 m) and trees (woody species taller than 5 m). The cover of *C. odorata*, the Zingiberaceae and the Marantaceae was plotted as dependent variables against the sum of the cover of the light intercepting trees and shrubs, and tested using a Kruskall-Wallis test, as data were not normally distributed. Percentages can exceed 100% because of the occurrence of multiple layers of foliage. We sampled vegetation cover at 97 sites in the road verge, at 78 sites in the forest edge and at 48 sites in the forest interior (n=223). Sampling was done in the second week of March 2004, which is at the start of the rainy season.

Figure 1. Road side colonised and dominated by the invasive shrub species *Chromolaena odorata*, Campo-Ma'an National Park, Cameroon, 2004.

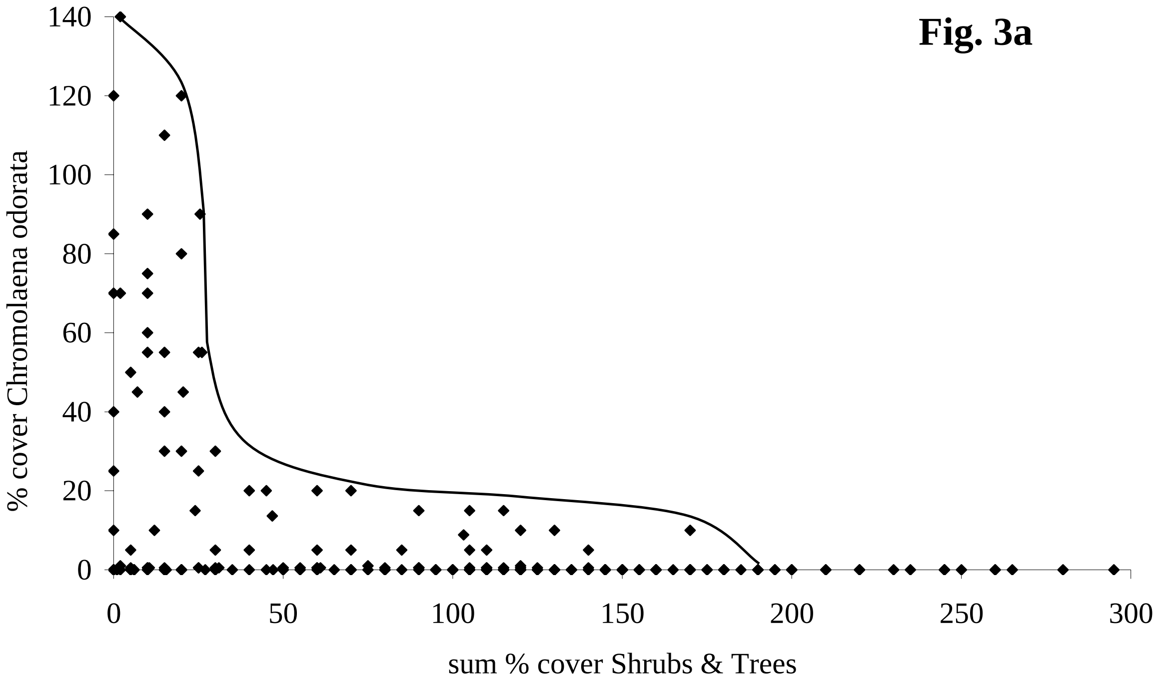
Figure 2. Road side in Campo-Ma'an National Park, Cameroon in 2004. The original vegetation dominated by African gingers (Zingiberaceae) is not yet invaded by the invasive shrub species *Chromolaena odorata.*

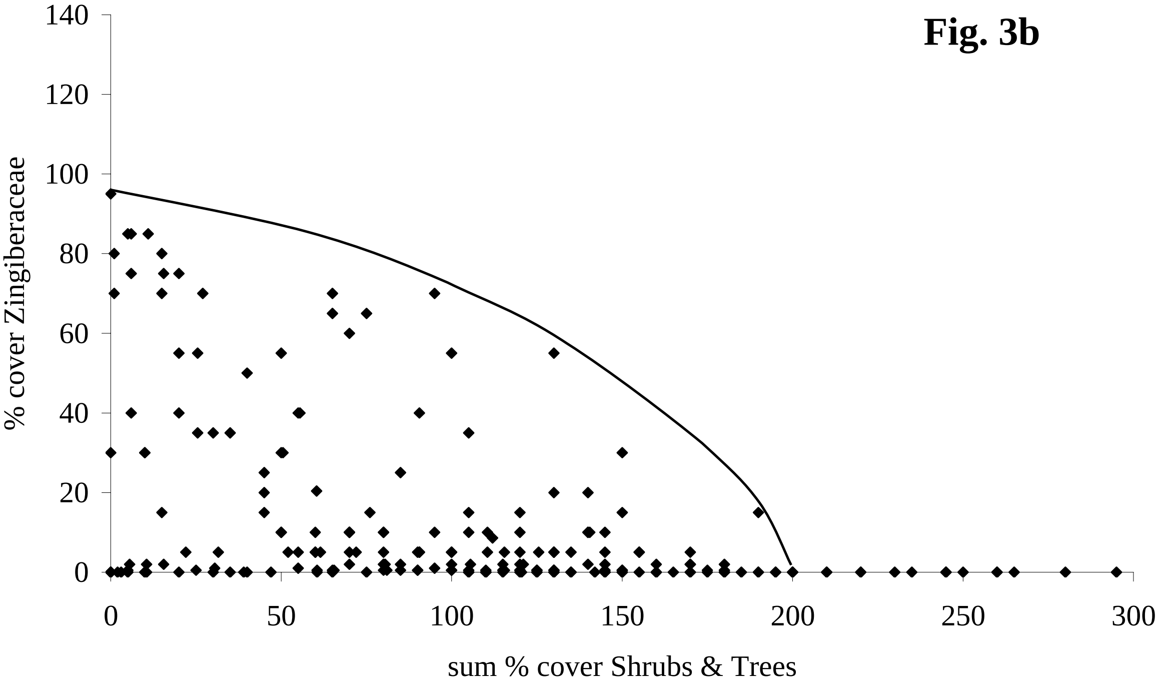
Results

Chromolaena odorata invaded road verges all along a 32 km long road through Campo-Ma'an National Park in only two years time (2002-2004) and penetrated the forest along a network of old logging tracks for more than 15 km. Invasion varied from negligible to total invasion, resulting in a patchy distribution of mono-specific *C. odorata* stands, and areas where both Zingiberaceae as well as *C. odorata* occurred. Individual *C. odorata* plants were found all along the road however, indicating that the colonisation process was still underway.

Zingiberaceae and *C. odorata* cover decreased significantly from the road verge to the forest interior (Figure 3a and 3b, Kruskall-Wallis test $p<0.001$). Peak *C. odorata* cover was where tree and shrub cover was less than app. 30%, which is predominantly in the road verge, where trees and shrubs are suppressed by regular mowing. *C. odorata* is virtually absent where tree and shrub cover is higher than 150%. Zingiberaceae cover was also highest at low tree and shrub cover, and was absent above 175% tree and shrub cover. Marantaceae cover was highest in the forest interior, between 50 and 250% tree and shrub cover. Marantaceae cover decreased drastically beneath 50% tree and shrub cover (Figure 3c, Kruskall-Wallis test, $p<0.001$). The level of co-occurrence of *C. odorata* with Zingiberaceae and with Marantaceae is illustrated in Figure 4.

Figure 3. Niche take-over by invasive plant species (*Chromolaena odorata*). The cover of *Chromolaena odorata* (3a), Zingiberaceae (3b) and Marantaceae (3c) were plotted as dependent variable against light-intercepting cover of shrubs and trees. Shrub and Tree cover include *C. odorata*, except when plotted against *C. odorata* itself. Percentages can exceed 100% because of the occurrence of multiple layers of foliage.





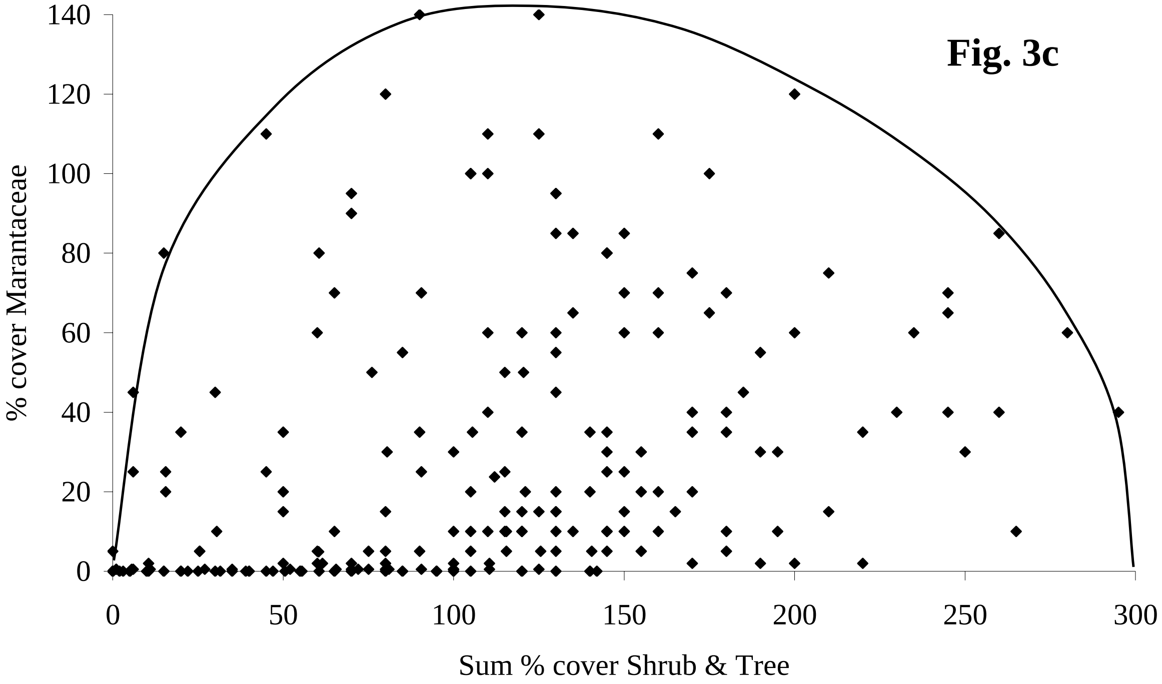


Fig. 3c
140
120
100
80
60
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% cover Marantaceae
0
50
100
150
200
250
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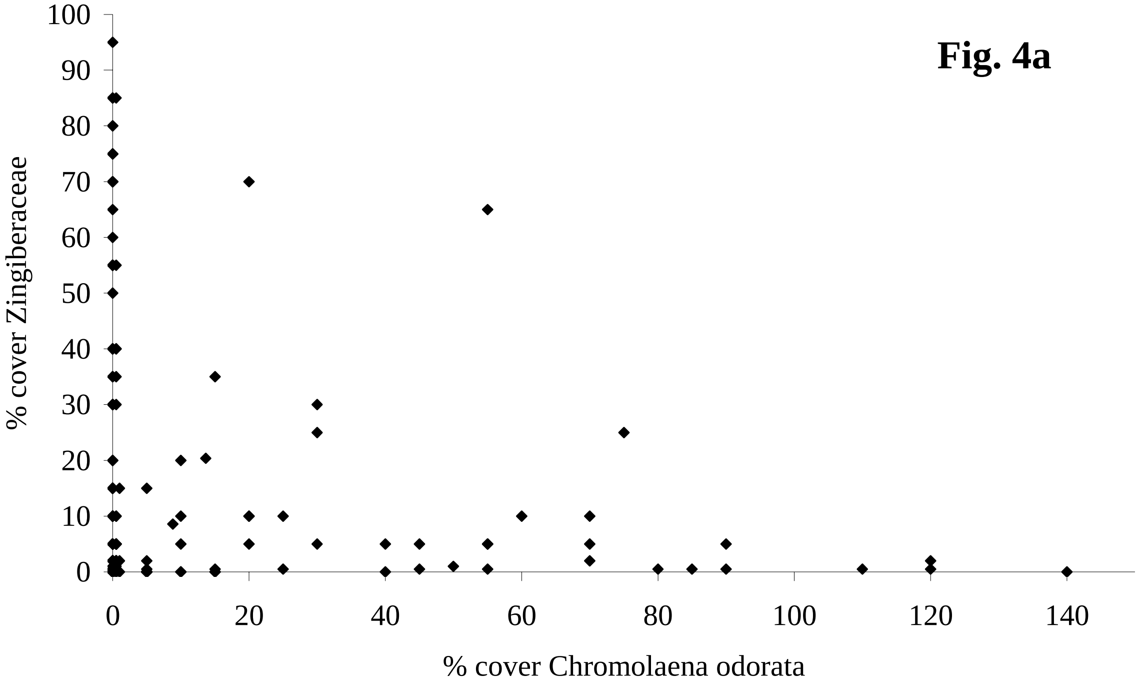
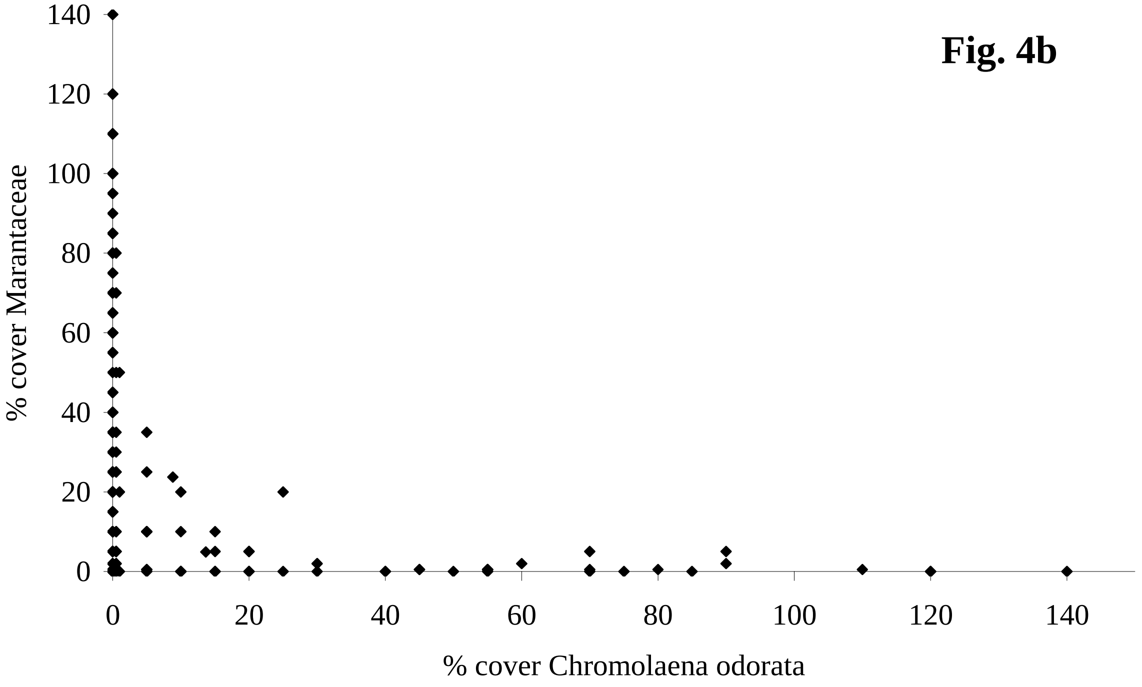
Sum % cover Shrub & Tree

Figure 4. Competition between invasive plant species (*Chromolaena odorata*) and native Zingiberaceae and Marantaceae. Zingiberaceae share a large part of their niche with *C. odorata* (4a), whereas Marantaceae hardly co-occur with *C. odorata* (4b). Percentages can exceed 100% because of the occurrence of multiple layers of foliage.





Discussion

Zingiberaceae and *C. odorata* are both light dependent; where the latter thrives at places with 0 - 30% tree and shrub cover, and Zingiberaceae occur mostly when tree and shrub cover is 0 - 130% (Fig. 3). When both species are plotted against each other it becomes clear that Zingiberaceae and *C. odorata* share the same niche (fig. 4a). Zingiberaceae will remain dominant only at places with more than 150% tree and shrub cover, where *C. odorata* is absent, as can be seen from Fig. 3. This means a reduction of cover of Zingiberaceae of 95% when *C. odorata* has invaded all areas with tree and shrub cover <130%. Zingiberaceae were not found in treefalls, which rules out the possibility of treefalls as a refuge for wild gingers. Marantaceae are the most dominant species found in treefalls. Given that Zingiberaceae need much light to germinate, just as *C. odorata* does, and given that at exactly those sunlit places *C. odorata* replaces Zingiberaceae, we predict that these native gingers will disappear from the most suitable habitat.

Chromolaena odorata and Marantaceae hardly share the same niche (fig. 4b), as Marantaceae inhabit the forest interior, where light levels are too low for both Zingiberaceae and *C. odorata.* Marantaceae are therefore not threatened by *C. odorata*. However, given the large quantities and high relative abundance of Zingiberaceae in gorilla diet, it is doubtful whether Marantaceae alone can provide enough bulk food for the gorilla when the Zingiberaceae have disappeared.

If the vegetation in disturbed forest would be allowed to regrow after clearing, *C. odorata* incidence would decrease, and after about five to ten years only a few sterile plants would remain, waiting for a chance to recolonise when the vegetation is cleared again (Kushwaha *et al.* 1981; De Rouw 1991; Joshi *et al.* 2006). With the necessity of regular clearing of roadsides for road maintenance, this natural succession and consequent suppression of encroachment of *C. odorata* will not happen. Logging companies claim that there is no alternative for clearing, as the laterite road bed needs direct sunlight to dry in order to remain hard, otherwise intensive use by logging trucks would ruin the road surface. Clearing is furthermore done for safety reasons, such as to create a better view, to prevent wildlife from crossing, and to prevent hazardous treefalls. Few large tracts of forests remain in Central Africa that are not yet penetrated by logging or exploration roads (Wilkie *et al.* 2000; Minnemeyer 2002). As roads are the main lines of conduct for spreading of *C. odorata* over long distances (De Rouw 1991), this calls for a change of road maintenance policy, or the acceptance of the total invasion of Central African rainforests by *C. odorata*.

Given its light dependency we do not believe that *C. odorata* will invade the rainforest itself, like it does in, for example, the monsoon forests of Bandipur NP, India (pers. obs.) and Nepal (Joshi *et al.* 2006). We predict, however, a widespread invasion of abandoned cultivation, logging tracks and road verges in all secondary lowland rainforests of Central Africa, exactly in the last strongholds of the embattled populations of the lowland gorilla. When the whole region where gorillas live is invaded, which is well under way, most of the Zingiberaceae will have disappeared. Already stressed by human encroachment, diseases, poaching and other disturbances, the effect of this reduction in food availability thus threatens the existence of the gorilla.

Acknowledgements

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Next page: Memvele falls

Chapter 7

SYNTHESIS & CONCLUSIONS

The implementation crisis

The pressure on biodiversity around the world is increasing every day. Some even speak of the sixth mass extinction already taking place (Pimm 2000; Brooks *et al.* 2002). As is discussed in the first chapter of this thesis, the loss of biodiversity poses a real threat to humankind. The protection and sustainable use of our natural resources has become an international priority because of this threat. The international community recognised this threat and designed the Convention on Biological Diversity in 1992 to improve conservation of biodiversity. Currently 190 countries have signed the treaty (www.biodiv.org). This increased awareness has furthermore led in the early 1980s to the development of conservation biology as a new scientific discipline which is dedicated specifically to phenomena that affect the maintenance, loss, and restoration of biological diversity (Soulé 1985). The Society for Conservation Biology (SCB) was founded in 1985 to stimulate this discipline and has currently over 10,000 members, ranging from scientists to conservation managers. Their platform is the peer-reviewed scientific journal Conservation Biology. Dedication and funding among many conservation organisations and inter-governmental institutes has increased substantially in the past few decades, although there is still a large gap between available funds and what is needed for conservation activities (James *et al.* 2001; Balmford & Whitten 2003; Halpern *et al.* 2006).

Has all this attention increased conservation success? Although much has been achieved, pressure on biodiversity is ever increasing, particularly in the tropics (Meffe *et al.* 2006). There is currently much debate whether conservation biology is effective in reaching its main goal: to improve conservation of biodiversity. On the 20th anniversary of the SCB in 2006 the editors of the journal Conservation Biology took stock of the current state of conservation biology (Meffe *et al.* 2006). While advancement has been made, the overall picture is not a positive one, and more calls are heard for a structural change in conservation biology (McShane 2003; Terborgh 2004; Robinson 2006). There is, for example, a need for new initiatives to address the interface between science and policy, and a necessity to actively promote the translation of conservation biology to conservation on the ground (Meffe *et al.* 2006). An often heard statement found in scientific forums is that conservation biology does not provide adequate answers for the questions posed by management on the ground, which hampers effective conservation. Long-term studies in conservation biology are often not based on specific management requests, but more on fundamental scientific interests, resulting in a “knowing-doing gap” or an “assessment-planning gap” (Knight *et al.* 2006a).

There is too much focus on assessments only (mainly surveys), without suggestions for planning or practical solutions, or on-the-ground management (Whitten *et al.* 2001). Examples of how to translate survey results into practical advice and then use them for applying to conservation are rare (Knight *et al.* 2006a).

This discrepancy between research attention and the information requirements of practitioners has led to the so-called “implementation crisis” (Knight *et al.* 2006b). In order to actively seek solutions for the problems at hand conservation biology “should focus research on the causes rather than the symptoms of biodiversity and ecosystem services losses” and “must focus on tangible, rather than intrinsic values” (Sarukhán 2006: p. 675; and Western 2003: p. 16, respectively). Whitten *et al.* (2001) formulated it even stronger by questioning whether conservation biology is a “displacement behaviour for academia”, given the missing link between scientific theory and on-the-ground management.

In this synthesis I consider the causes of the implementation crisis found in literature, and discuss the case studies form the field that address this problem. I specifically combine and compare classical conservation science with newly developed methodologies. These new methods are based on the requirements of conservationists in the field, and link different research activities in order to deliver a more comprehensive set of information for conservation planning and implementation. I try to answer to the observed call for a new direction of conservation biology, which asks for scientific activities to be more geared towards implementation than towards knowledge gathering *per sé*. The main conclusion of this thesis is that integration of local knowledge in classical wildlife surveys in combination with a vulnerability assessment for priority setting in conservation delivers a tool that facilitates priority setting for conservation. Several other scientific activities that are regularly conducted in conservation projects are discussed based on case studies, with recommendations on how to generate more useful information for effective conservation.

In any conservation project, whether it concerns a protected area or a multiple-use zone, the general approach to the set-up and management of such an area follows a number of clear steps that each requires specific scientific information. In general there are two phases: the descriptive and the implementation phase. Although there is no clear protocol for the development and management of a given area, these steps are almost always required. There is, as is discussed in this synthesis, an uneven division of attention between the descriptive and implementation phase. The flow chart (Fig. 1) illustrates the stages in the set-up of a conservation project, with main activities and examples. Individual steps normally involve studies that are conducted separately, but in this thesis several studies are intended to reinforce one another. In the flow chart I have indicated which steps are addressed by the different chapters of this thesis.

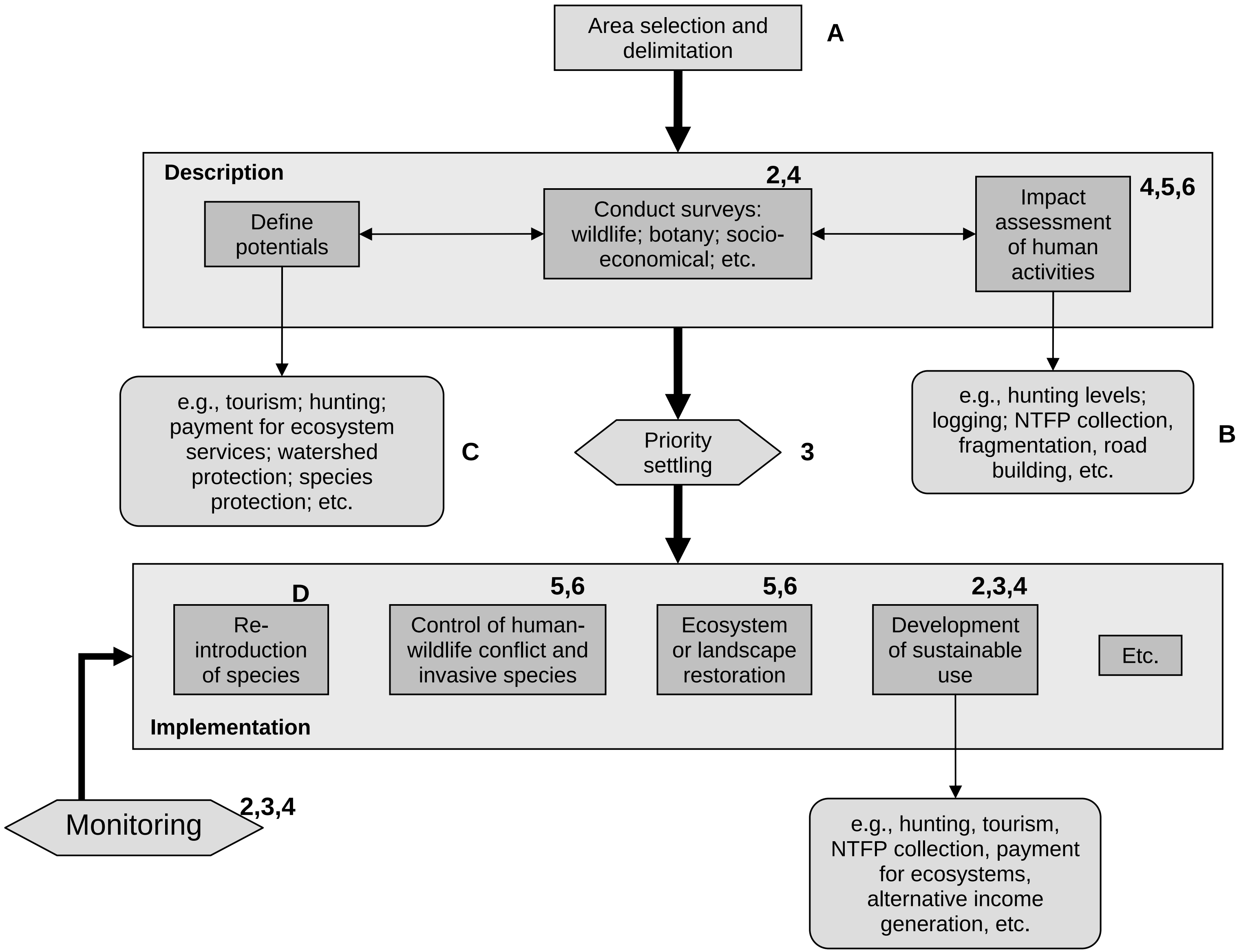


Figure 1. Flow chart of steps needed for development and management of conservation projects. Numbers refer to chapters of this thesis that address specific steps. A: this was done by the Project Campo-Ma'an and the government of Cameroon, B: this was the expected output of the project at large, C: this lies in the domain of the Governement of Cameroon, D: this was not considered as an issue in this project.

Underlying causes for the implementation crisis

Reasons according to literature for the failure to bridge the gap between knowledge gathering and implementation are numerous. One is the opinion that environmental problems are too complex to lend themselves to standard scientific approach of problem definition, data-gathering, analysis and decision making (Ludwig *et al.* 1993, Ludwig 2001). There is too much uncertainty in the data, there is a lack of controls and replicates, targets keep shifting, and there are no clear objectives, which makes addressing the problem difficult (Ludwig *et al.* 1993; Berkes 2004).

Another reason is that to solve the problems faced in biodiversity conservation, the involvement of different classical academic disciplines is needed (Drew & Henne 2006), and other sources of knowledge should be used, such as traditional ecological knowledge (Huntington 2000; Davis & Wagner 2003; Mauro & Hardison 2000). Integration or at least collaboration between scientific disciplines (e.g., socio-economics, sociology, biology, etc.) and the use of other sources of knowledge has hardly taken place. But, however difficult, integration or merging of knowledge from different sources is increasingly suggested as the means to a more successful link between science and management, because this will take away existing barriers between scientific disciplines, which hamper effective conservation activities (Huntington 2000; Ludwig 2001; Berkes 2004; Sheil & Lawrence 2004; Sheil *et al.* 2005; Drew & Henne 2006). Collaboration will become only more pertinent because most extinctions take place in human-dominated landscapes, thereby challenging conservation biology to broaden its scope (Sarukhán 2006; Robinson 2006).

It does not help when conservation biology has a fixation on its own field, with insufficient consideration of its practical implication (Whitten *et al.* 2001). Inadequate practicality of conservation biology is also caused by the struggle for funds, and the pressure for scientists to publish results in peer-reviewed journals. When different organisations compete for funding from the same source, sharing of information, experience and methodology is hampered because one might miss out on funding opportunities (McShane 2003). Fierce competition in the race for scientific acknowledgement might withhold scientists form publishing relevant research that might be better for management, but less so for scientific advancement. This can result in a more conservative and thus “safer” approach that may guarantee academic survival, but which precludes practical studies from being published (Sheil & Lawrence 2004). It is speculated that this explains the relative low number of publications by local conservation scientists in international renowned journals, because they are busy with actual conservation instead of securing scientific acknowledgement through publication of research (Whitten *et al.* 2001; Western 2003). These divergent goals stand in the way of collaboration between conservationists, let alone between different scientific disciplines (McShane 2003).

The result of this “knowledge-doing gap” is that conservation science is put above the practical needs and loses touch with the field of conservation practice (Fazey *et al.* 2006). This is illustrated by the predominance of scientific studies conducted in the description phase, and the lack of studies in the implementation phase (Fig. 1). Conservation practitioners require specific information for planning and management, preferably as fast and as accessible as possible, while conservation scientists hold on to the classical ways of producing data with as high as possible levels of confidence (Danielsen *et al.* 2005). As a consequence information acquired is often not tailored to the management needs (e.g., a disproportionate number of assessment studies), or information is not available because research takes too long (e.g., theoretical modelling, or long term analyses and surveys; Sheil & Lawrence 2004). Information is furthermore unavailable because agreement between scientists is lacking, and scientists refrain from drawing conclusions on data because confidence is lacking (Ludwig *et al.* 1993; Robinson 2006).

The potential of local knowledge

From the multitude of essays that discuss the "knowledge-doing gap" the overall opinion emerges that more successful conservation can be achieved if the local population will be more integrated in the conservation process: from problem definition to knowledge sharing, data gathering, policy setting, and actual implementation and monitoring (see box 1). This does, however, depend on the goal of the project. If strict protection is the objective, than the integration of local people will be limited, because generally exploitation within a National Park is not allowed. I concentrate on the areas where humans are an integral part of the system, and integration of their knowledge and opinion is therefore more important than when dealing with management of strictly protected areas. The advantages of integrating local people in conservation biology are that collaboration helps scientists to shift from theory to practice (Sheil & Lawrence 2004; Huntington 2000; Basset *et al.* 2004; Bawa *et al.* 2004). The fulfilment of this integration is twofold. First, there is the use of the local knowledge in research and consequently conservation planning. Second, there is the actual involvement of local people in conservation practice, from the policy setting level through the research, implementation and finally the "use" level. Throughout these levels local people are asked for their opinion and ideas, and are actively involved in planning and assessment activities. The use of local knowledge is based on the notion that classical conservation science can not do it alone anymore (Moller *et al.* 2004; Sheil & Lawrence 2004; Fazey *et al.* 2006). By incorporating this knowledge in conservation biology the "knowing-doing gap" can be bridged (Knight *et al.* 2006a). I use the term local knowledge which covers the many different definitions used in literature, for example: local ecological knowledge (LEK, Davis & Wagner 2003), traditional ecological knowledge (TEK, Huntington 2000), and indigenous knowledge (IK, Mauro & Hardison 2000). The Convention on Biological Diversity even has a significant part of its policy dedicated to local knowledge and the sharing of knowledge and techniques. Articles that specifically address these issues as being important to improve conservation are: 8j, traditional knowledge, innovations and practices, (with its own Working Group); 17, exchange of information; and 18, technical and scientific cooperation (CBD 2007, see www.biodiv.org).

The actual involvement of local people in the conservation process serves a different purpose, namely to promote involvement in and commitment to conservation. I make this distinction because although both processes (use of local knowledge and integration of local people in conservation activities) have taken place some time, the motivation behind each has been and may still be different. On the one hand, actual involvement of local people in conservation used to be mainly for assistance and guiding in the field. Currently, however, local people and their needs are accepted as intricately associated with any activities that are planned in their area, and should therefore be involved in the whole process affecting biodiversity management (Berkes 2004; Western 2003; Sheil & Lawrence 2004; Sheil *et al.*

2005). The use of local knowledge on the other hand, used to be a means to serve conservation biologists in reaching their academic goals, irrespectively of the opinions of the local people regarding conservation planning.

The potential and the possibilities of the use of local knowledge in conservation biology have been extensively discussed in literature. Quoted here are just a few examples where it is clear that much is discussed in future tense (Huntington 2000), instead of discussing practical applications or case studies (emphasis added by the author):

- "A satisfactory resolution *may well* hinge on special sorts of local knowledge and institutions that will only become available if local people are welcomed as active and influential participants" (Ludwig 2001, p. 763).
- "Combination of formal science training and traditional skills and the insights offered by paraconservationists *are worth nurturing*" (Western 2003, p. 17).
- "Tropical biology research *must become* more participatory, involving local communities and according respect to alternative knowledge systems in addition to modern scientific knowledge. Partnership with local communities may be an important key to the success of conservation and restoration efforts" (Bawa *et al.* 2004, p. 443).
- "Collaborative research projects that involve local people from the outset *generate possibilities* for complementary use of scientific and traditional knowledge" (Berkes 2004, p. 627).
- "Making meaningful progress *in the future* will entail a willingness on the part of conservation scientists and practitioners to work with indigenous, mobile and local communities in new ways, in which the tools of western science are offered in support of local conservation practices" (Brosius 2004, p. 611)
- "Collaboration with local communities shows how biological evaluations are more efficient with local guidance, and *reveals potential* for collaborations between local communities and those concerned with conservation. Conservationists who engage with local views *can* benefit from an expanded constituency, and from new opportunities for pursuing effective conservation" (Sheil *et al.* 2005, p. 17)
- "… academics and practitioners will also need to be open to the perspective that it *is a worthwhile endeavour to find* appropriate ways to elicit, communicate and share experiential knowledge" (Fazey *et al.* 2006, p. 8).

The technical issues of how to translate or quantify local knowledge to allow scientific scrutiny by peers has rarely been discussed in literature (Huntington 1998 and 2000; Davis & Wagner 2003; Danielsen *et al.* 2005). The case studies described in chapters 2 and 3 of this thesis illustrate how the use of local knowledge can be: 1) structured in a transparent and reproducible way, 2) merged with scientific information and methodology, and 3) validated by comparisons with similar studies.

Based on the results presented in chapters 2 and 3, I argue that the use of local knowledge can serve both goals: it can help conservation biology become more effective by merging local knowledge with classical science, as well as stimulate participation and integration of local people in conservation activities. The use of local knowledge can induce the democratic legitimization of conservation activities within multiple use zones. If people affected by conservation activities have a say in the planning process, then motivation to comply with conservation could increase because they have influence on the planning, as it is also based on their knowledge. When conservation plans and measures are based on the preconceived ideas of western conservationists without consideration or involvement of local people, commitment or motivation to cooperate in the conservation process will be low (Sheil & Lawrence 2004). In order to achieve improved conservation the expertise and knowledge of classical science should be employed to use local knowledge to the fullest extent.

The integration of local knowledge in resource management, and consequently in conservation, has successfully been tested and applied in the domain of fisheries. Marine scientists have accepted that the knowledge of local fishermen could supplement lacking knowledge in classical science. They proceeded in testing and integrating this local knowledge in modelling and fisheries management (Huntington 2000; MacKinson 2001; Davis & Wagner 2003; Fraser *et al.* 2006). This has led to promising results in a more effective management of these resources (Pomeroy 1996; Berkes *et al.* 2000; Moller *et al.* 2004). Remarkably, this has not happened yet in wildlife science, despite the significant number of scientists and scientific journals that deal with wildlife conservation and management (see the first paragraph). Although the advantages of integrating local knowledge in conservation biology -particularly in wildlife management- have extensively been discussed in literature, the practical implementation has not. Most articles are about the possibilities and potential (box 1) and less on actual examples on how to apply this in the field (Davis & Wagner 2003, but see for an exception: Steinmetz *et al.* 2006). Reasons why this integration does not happen are:

- the inertia and stubbornness of scientists to change their behaviour (Huntington 2000; Knight *et al.* 2006a),
- the lack of interest in actual developments in the conservation field (Terborgh 2004),
- the lack of case studies for comparison (Knight *et al.* 2006a),
- and the aversion of people, including biologists, to delegate control and status (Sheil & Lawrence 2004).

Given the developments and experience in other disciplines (social sciences, fisheries, and ethnobotany), and the many theoretical essays on the potential of integration of local knowledge in classical science, this disregard by conservation biology seems out of date. The integration of local knowledge in classical conservation science should be fairly straightforward: collect data (information from local people) in a transparent and justifiable way and analyse them using traceable and reproducible methods. The knowledge should preferably be quantifiable in order to make comparisons and to enable replication. Good practice dictates that any information should be adequately replicated, and cross-validated whenever possible (Sheil & Lawrence 2004), but in practice clear research protocols when it comes to using local knowledge for conservation biology are lacking.

Using local knowledge in practice

Chapter 2 of this thesis is a case study of how local knowledge is used to assess wildlife densities in a biodiversity conservation project area. The method used is the Pooled Local Expert Opinion (PLEO), in which the estimates of the local experts were pooled to give an overall density estimate for wildlife. Information on wildlife abundances was extracted from local experts (hunters) through interviews in such a way that after a relative simple analysis average wildlife densities were obtained for most medium to large sized mammals in the study area. A representative sample of interviewees was selected in order to cover the whole project area, while selecting enough to achieve an adequate sample size. Local experts were selected by their peers in their village, as this should produce the best results (Davies & Wagner 2003). Hunters were asked to estimate wildlife numbers in a known area, which allowed for calculation of wildlife density. Averaging densities from all interviews resulted in a density for each species for the whole project area. These results were compared to classical (transect) surveys conducted at the same time in the same area for validation of the method. The PLEO method was favourable in comparison to the classical transect survey in the same area, as more results were obtained in less time and for less money. The resulting densities from the PLEO were similar to results from other surveys in comparable areas in the region.

The second case study of the use of local knowledge is described in chapter 3. In this chapter a vulnerability list was produced which ranked the species according to their vulnerability to rapid decline or local extinction. A combination of local knowledge with information obtained from classical biological science was used to produce results that could help project management to set conservation priorities. Local experts were asked to rank species for a set of factors that were presumed to be of influence on their vulnerability. These factors define the attractiveness of the species for hunters to pursue them specifically, and make them therefore vulnerable for overexploitation. These factors could not be quantified by information from literature, as they are locally dependent. Examples of these factors are taste, commercial and traditional values, and catchability. This information was merged with biological information obtained from scientific literature. Factors from literature included, for example, home range size, body size and life history traits. Factors were ranked per species similarly to the factors defined by local knowledge. Both rankings were combined, producing a ranking list based on both classical science as well as on local knowledge.

These two case studies illustrate three advantages in which the issues of integration of local knowledge in conservation biology are addressed. Firstly, the use of local knowledge is formalised in a structured way, because data collection and analysis are repeatable and transparent. Knowledge was extracted in a manner that allowed comparison with other results, even when these were obtained with classical survey methods. Methodology was developed in a way that the study can be repeated requiring minimal technical training. Secondly, the required information is obtained in a way that directly serves conservation management with useful data (density estimates and a vulnerability list ranking all major wildlife species). The assessments were furthermore cheaper and faster than classical survey methods. Classical transect surveys at the same time took one year to be completed, compared to the two months it took to complete the PLEO survey. The budget needed for the transect survey was seven times higher than that of the PLEO. In spite of the large budget and the extended period needed for the classical surveys, these transect surveys did not produce enough data to provide useful density estimates for any species, primarily due to the large area that needed to be surveyed. A recognised feature of assessments based on local knowledge is that it can provide large amounts of data in a relative short period with limited effort (Moller *et al.* 2004; Danielsen *et al.* 2005). The risk of over- or underestimates because data are derived from subjective sources was anticipated by having a large enough sample size, in this case enough interviewees. Comparison with classical surveys indicated that there was no significant deviation (neither consistently higher, nor lower), and confirmed the PLEO method as a useful method for density estimation in tropical rainforests.

The third advantage is that the commitment to policies based on the data should increase. The local experts deliver information which will be used for conservation planning. The knowledge that management plans are also based on their input can increase the commitment to the implementation of these plans. Unfortunately, the Campo-Ma'an project was ended prematurely due to a change of policy by the Netherlands Ministry of Development Cooperation. Due to the lack of funding resulting from this change of policy the management plan that was developed was actually never implemented. This made testing the assumption that commitment to conservation action increased with integration of local knowledge unfeasible.

Using classical surveys

Although the advantages of the use and integration of local knowledge for monitoring in wildlife management are recognised, the ability to detect wildlife population trends still proves to be difficult. While the involvement and participation of local people in the management and monitoring process provide several clear advantages, it does not provide an adequate solution so far, because short-time population trends can still not be detected (Noss *et al.* 2005; Basset *et al.* 2004). For the moment, until the PLEO method is tested for detecting population trends, it serves primarily as a baseline survey, or starting point for long term population monitoring.

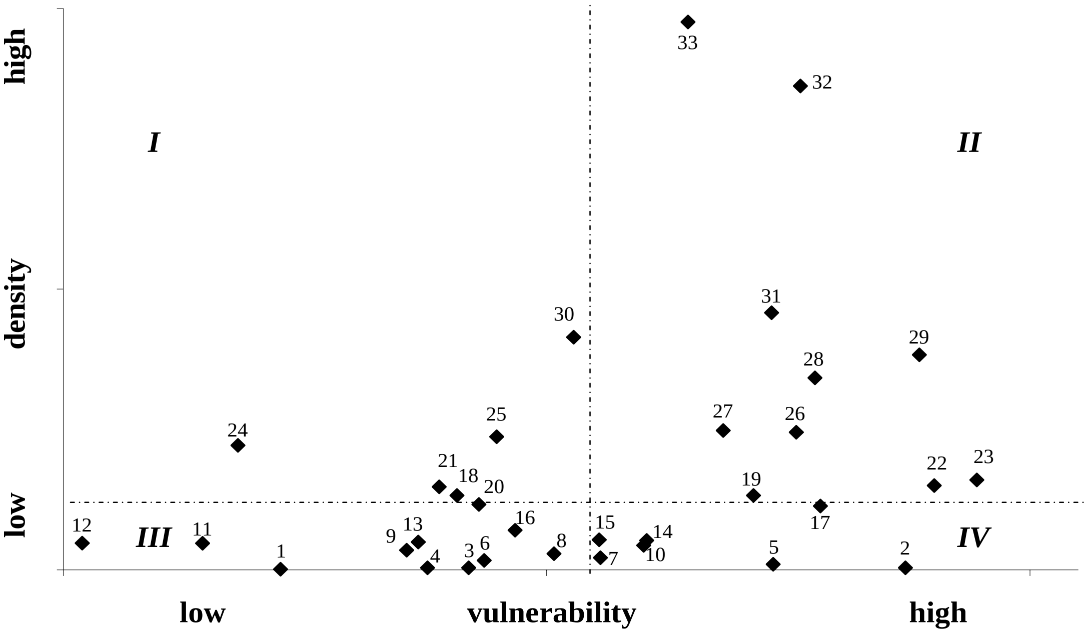
Other methods for indicating changes in wildlife populations have been used to assess effects of management interventions on wildlife populations. One example is studying the harvest rates of hunters in tropical rainforests. It is suggested that the catch-per-unit-effort can indicate whether wildlife populations are in decline or not (Noss *et al.* 2005). Similar to this is the study of the bushmeat trade, where it is assumed that the supply of wildlife for sale on a market can indicate the state of the source population, and consequently the sustainability of the offtake (Fa *et al.* 2000). At the time of the study, bushmeat market surveys were still regarded as a potentially adequate way to assess the state of wildlife in a given area. A market survey was therefore launched, aimed at identifying a limited set of indicators that could be easily monitored, and which should indicate the state of the resource population (chapter 4). This survey was considered as an additional tool for management, supplementary to the actual wildlife density assessment (chapter 2) and the priority-setting vulnerability assessment (chapter 3). Although several factors were found to be indicative of the wildlife population that was being harvested, nothing, however, could be said about the sustainability of the trade. Other studies found in literature that were completed after the one described in chapter 4, which were based on much longer and more comprehensive research, did not provide an answer to the question of sustainability either (e.g., Crookes *et al.* 2006). The bushmeat trade is obviously very complex and does not provide a single approach that gives satisfying answers about the sustainability of the hunting. Market surveys neither result in detecting accurate enough trends in wildlife population sizes. However, this market survey illustrates that with only a few indicators instead of a full factor model an indication can be given about the general state of the wildlife. In combination with the other surveys (see below), the usefulness of a few indicating factors proves its' worth. An exception to the above-mentioned disadvantages of market surveys is when a long-term study is directed at a single species, which allows money and time to be fully dedicated to one species (e.g., babirusa in Indonesia, Milner-Gulland & Clayton 2002; helmeted guineafowl in Cameroon, Njiforti 1997).

When the different information sources are combined (chapters 2, 3, and 4) an overall picture emerges that provides conservation planners and practitioners with adequate information to prioritise conservation activities, more than when only one of the three studies is used. The density assessment provides the base line, which is reinforced by the vulnerability assessment and vice versa. Species that rank high on the vulnerability list would be expected to be low in density in the study area. If a species is not low in density, but ranks high on the vulnerability list, further research is needed to identify why this species is perceived as vulnerable, and what this might implicate for management. The combination of both assessments becomes clear when the local densities and vulnerability are depicted together (Fig. 2). The species are divided in four categories, which are: high local density, low local vulnerability (I); High local density, high local vulnerability (II), Low local density, low local vulnerability (III), and low local density, high local vulnerability (IV). The latter category, where species have small population sizes and are ranked high in vulnerability indicates the species that at first sight are in the highest threat category. This is confirmed considering the species that fall in this category, like for example chimpanzee (nr. 17), elephant (nr. 2), gorilla (nr. 14), and buffalo (nr. 5). These particular species are internationally considered to be highly threatened (IUCN 2007), which is confirmed by the present study. Although outliers like porcupine (nr. 32) and blue duiker (nr. 33) seem to be safe as they occur in high densities, they do score high on vulnerability. This can be explained by the data from the market survey, where these two species are found to be the species that are most frequently found for sale. Hunting pressure on these abundant species is high, which explains the high vulnerability. Care should be taken when considering regulating the use of these species, even if they seem still abundant. Several other detailed examples are elaborated in the relevant chapters, with species-specific information that was collected after these species were found to be more vulnerable than assumed based on the density assessment alone. Although the division in four categories might seem rather arbitrary, it draws attention to species that could use some extra scrutiny before for example any plans for regulating hunting can be implemented.

The bushmeat market survey indicates the pressure on wildlife on a local scale and what factors influence this pressure. The state and diversity of the supply at different market sites in the project area can be related to the distribution of wildlife and their vulnerability. When the dynamics of the bushmeat trade and the distribution of wildlife are described, a picture emerges which can direct conservation activities to target different areas, species, and processes in a given area. Two of the five bushmeat market sites in the project area clearly showed signs of overexploitation because percentages rodents and smoked meat were high (both indicators of high hunting pressure) in comparison to the other sites. This was confirmed by the PLEO, which indicated that wildlife densities in a radius of 25 km around these sites were much lower than around the three other sites. Combining the results of the

PLEO and the market survey indicates that management efforts, such as regulation of hunting, should be adjusted to the local situation. The different case studies provide detailed enough information for management interventions to be adjusted to specific sites within the project area. The effectiveness of conservation should thus improve when activities are tailored to the local situation.

Figure 2: Density and vulnerability of 33 wildlife species in the Campo-Ma'an area, Cameroon. *I*= high density, low vulnerability; *II*= high density, high vulnerability; *III*= low density, low vulnerability; *IV*= low density, high vulnerability. Numbers correspond to species. Species in bold are highly vulnerable and low in density: they need most local attention.



1	Golden cat	**10**	**Grey-cheeked mangabey**	19	Red-capped mangabey	28	Crowned monkey
2	**Forest elephant**	11	Spot-necked otter	20	Monitor lizard	29	Moustached monkey
3	De Brazza's monkey	12	Swamp otter	21	Dwarf antelope	30	Bay duiker
4	Leopard	13	Water chevrotain	22	African dwarf crocodile	31	Putty-nosed monkey
5	**forest buffalo**	**14**	**Gorilla**	23	Mandrill	32	Brush-tailed porcupine
6	African Civet	**15**	**Python**	24	Yellow-backed duiker	33	Blue duiker
7	**Marsh cane rat**	16	Giant pangolin	25	Red river hog		
8	Black colobus	**17**	**Chimpanzee**	26	Northern talapoin		
9	Black-fronted duiker	18	Peters' duiker	27	Sitatunga		

The role of classical science

As argued in the preceding section, conservation biology in the 21st century should shift attention towards integration of local knowledge in classical scientific activities. However, I do not deny the need for fundamental long-term biological research and its use for conservation biology. Relevance of classical science concerning long term studies on animal ecology and processes at ecosystem level are needed to put changes in, for example, extinction levels in perspective (Terborgh 2004). The value of long-term classical scientific research for direct conservation planning is illustrated in a study on the effect of human intervention in a protected area, in this case the effect of the construction of a road on wildlife behaviour in a National Park in Cameroon (chapter 5). Project management had decided that a road through a National Park would benefit local populations by connecting them to certain basic social services, such as education and primary health care. It was furthermore seen as a useful control measure because all logging transport had to use this road. By regulating and controlling transport, illegal trade in wildlife and timber was foreseen to decrease. There were, however, serious concerns about the effect of the road on the wildlife populations in the park, as the road could form a barrier, dividing the park in two. To assess the effect of the road on the survival of a wildlife population, an experimental study was conducted that studied the effect of the road on wildlife behaviour. The study resulted in several clear suggestions for project management that could mitigate the negative impact of the road on wildlife. These suggestions concerned mainly advice on road construction and maintenance that favour wildlife crossing behaviour.

While studying the effects of the road on local wildlife an indirect but nonetheless serious side effect of road construction was discovered (chapter 6). An invasive shrub was found to have colonised most of the sides of the road through the National Park. This invasive species, *Chromolaena odorata*, originating from South America, had replaced most original vegetation found on the road sides. Normally road sides in rainforest areas are dominated by Zingiberaceae, the African ginger family. From existing studies it was known that Zingiberaceae form a key resource for several large mammals in Africa like gorilla, chimpanzee, mandrill and elephant. The main habitat of the Zingiberaceae is disturbed areas with ample sunlight, as is the case for *C. odorata*. However, *C. odorata* is an aggressive, rapid growing and fast reproducing species that quickly replaces native vegetation in sunlit places. Invasion is still going on and most ginger habitat will probably be colonised in the near future, displacing native Zingiberaceae. The invasion of this shrub and subsequent disappearance of the original ginger vegetation could threaten the survival of the gorilla.

These two examples illustrate the translation of scientific results to practical advice for mitigating negative effects of human interventions. In these two cases knowledge obtained through long-term classical research was needed. Ethological data on gorilla feeding behaviour and ecological data on niche occupation by invasive species and native flora was needed to assess the effects of road construction in the National Park.

Conclusions

Integrate local knowledge in research activities.
The integration of local knowledge in wildlife research activities is possible in a scientific sound way. It does not only produce concrete results, it is also much cheaper and faster than classical wildlife surveys. The case studies confirm that integration is possible if experience from “classical science” is used to validate data obtained from “local knowledge”, and if repeatability and transparency is secured. Merging the best of both worlds is possible when dogmas of classical conservation biology are released. By shifting research towards practical solution-based goals, and by acting before there is scientific consensus, the much needed information becomes faster and more readily available for conservation planners. This thesis provides a start for further consolidation of the use of local knowledge within conservation biology.

Complementary or corroborative research activities produce results tailored to practical needs.
Research activities for biodiversity conservation should be based on the questions posed by conservation planners and practitioners. Although this seems obvious, in conservation projects this often is not the case. I have described in this synthesis that attention is unevenly distributed between research activities within conservation biology, resulting in a “knowing-doing gap”. The case studies presented in this thesis are all based on specific questions originating from management on the ground. The best available methods of finding solutions for these questions were sought, achievable within available funds and time. When adequate methods were not available they were specifically developed. Individual studies produced results that were combined with results from other studies conducted at the same time. This conveys an overall picture with as particular objective to provide conservation planners with practical advice or useful data. A wildlife density survey was combined with a bushmeat market survey and a vulnerability ranking of the wildlife in the project area. This combination of three studies identified several wildlife species and several specific areas that required conservation attention. Although each separate study could stand alone, it is the combination of the different studies that provides the added value.

Long-term classical scientific studies in conservation biology: only if they produce practical advice for management.
The need for classical research activities within a conservation project, such as experimental or long-term complex biological studies, proves its worth if called upon with specific questions. Conducting more fundamental scientific research only for the sake of discovering new facts should not be labelled ‘conservation biology’. When planning research activities one should consider that funding or conducting long-term fundamental research can lead to conflicts in budgeting and time allocation within a conservation project, and may subsequently hamper the practicality of the research. Research activities should be based on specific questions, for example, what will be the consequences of planned activities (e.g., road building, exploitation, etc.). Answers should provide practical advice on what can be done to mitigate negative effects on biodiversity. Proving that there is a link between a certain activity and biodiversity is not enough: research should rather aim for directing management. Long-term, more fundamental scientific research should have its own niche, with separate funding,

and different criteria for assessment of success and relevance for conservation. This assures that funds and time that are designated for conservation are used specifically for practical research that solves the implementation crisis.

Change of attitude of conservation biologists.
Conservation planners (the project managers) should clearly state what information is needed to improve conservation. Conservation biologists (the scientists) should aim research at solving problems expressed by the planners, instead of following their own preconceived research trajectory. In this thesis several case studies are presented that provide a starting point for further development of practical research. More case studies are needed to further test new methods, and alternative methods need to be sought for research activities that do not provide adequate information for conservation planners. This will require a change in attitude from conservation biologists. Based on the quite loud call from literature to do so, it now lies with the leading conservation biologists to put their money (and expertise) where their mouth is.

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Summary

Conservation biology is faced with an implementation crisis. This crisis is the result of a “knowledge-doing” or “assessment-planning” gap. One reason for this is that there is a discrepancy between systematic classical scientific assessments or surveys, and actual implementation in the field. This thesis explores the state of conservation biology by discussing the practicality of several research activities that are needed in most biodiversity conservation projects. Classical conservation science is compared and combined with newly developed methodologies. The objective was to produce a more comprehensive information package for conservation planning and implementation. Research activities were analysed on complementarity, cost and time constraints, and on the possibilities of integrating local knowledge. This study was conducted in GEF-Campo-Ma’an project in the tropical rainforest area of south Cameroon.

In **chapter 2** a new method of wildlife density estimation is explained, which is less time and money consuming, but yields comparable results with classical methods. Methods currently used for assessing wildlife density in rainforests are time and money consuming. The precision of the most commonly used methods is disputed, but accepted because more exact methods are not available. The method was tested in the field and compared with transect surveys in the area and with relevant literature. The PLEO (Pooled Local Expert Opinion) method is based on the knowledge of local experts. A number of hunters was asked to estimate wildlife abundance in a specified area, after which the density per km^2 was calculated for 33 wildlife species. These estimates were pooled and extrapolated for the whole study area. Elephant (*Loxodonta africana*) density outside the National Park was estimated to be 0.06 animals per km^2, and 0.3 inside. Buffalo (*Syncerus caffer*) density for the study area was estimated at 0.2 animals per km^2 and gorilla (*Gorilla gorilla)* density at 1.05 per km^2. Transect surveys carried out at the same time for considerably more money, taking far more time, produced too few data to calculate densities. The evaluation of the PLEO-method was favourable and the method offers a substitute for conventional methods of estimating wildlife density in rain forests. The methodology is simple and it can be incorporated in many tropical biodiversity and conservation projects. It can also be used for long-term monitoring of wildlife status in a given area. In contrast with classical methods, the PLEO-method is low in cost and assures local ownership of the results.

In **chapter 3** a new method is presented that ranks medium to large-sized mammals in a rainforest according to their vulnerability to extinction or to major population declines. It is a fast, efficient and cost-effective method to set priorities in conservation management. Information from the literature and local knowledge from hunters and forestry people were combined to assess the status of 33 wildlife species. The result is a vulnerability list, where species are ranked according to their vulnerability to major declines and extinction. To produce this list we developed a system where the risk-proneness of each species was determined on the basis of thirteen factors. These factors were assumed to be of importance to the survival of a population, and were scored with information from interviews with local hunters, and from the literature. The method was tested in the Campo-Ma’an area in south Cameroon. In this study the most vulnerable species was the mandrill, followed by the elephant, chimpanzee, and buffalo. Five of the ten most vulnerable species are on the IUCN red list of threatened species, which justifies the use of the new method to set local conservation priorities. We argue that for on-the-ground management this method provides a useful tool to allocate time and money to the species that need them most.

Because detecting trends in wildlife populations remains difficult, other ways of monitoring are needed. Gathering socio-economical and biological data on bushmeat markets could be a relative easy way of monitoring the bushmeat trade. Although collecting data is fairly straightforward, analyses of these data are hardly ever conclusive on the sustainability of the off-take. Theoretical models that include as much variables as possible do perform quite well in simulations, but long-term, multi-variable datasets provide no clear-cut answer yet. **Chapter 4** presents a survey that selected only a few factors that are thought to indicate the state of the trade, and analyzed the relations between these dependent factors and several fixed (independent) factors that are hypothesized to influence this state. The factors assumed to be indicative of the bushmeat trade are: the price, the state (the percentage smoked meat), and the diversity of the bushmeat for sale (in terms of number of species for sale). These are thought to be related to several factors that influence hunting pressure, which are: human population size (as a proxy for the demand for bushmeat); distance between local market and city markets; and distance to the National Park; and the wildlife density in the area surrounding the market. Two clear relations were found after analysis. These are a negative correlation between the price of the bushmeat and the distance to the city; and a negative correlation between the percentage of smoked meat at the market and the wildlife density in the area surrounding the surveyed market.

The negative effects of roads on wildlife in tropical rainforests in Africa are poorly understood. Road construction has high priority in Africa, with as effect that negative impacts of roads on wildlife often are neglected. **Chapter 5** provides information on the effects of roads on crossing behaviour of rainforest wildlife. Crossing probability of forest wildlife was analyzed for association with ten different factors that were linked to road presence or road construction. Factors were divided into three classes: vegetation cover, topography and human influence. Spoor plots were laid along a 32 km unpaved logging road that divides Campo-Ma’an National Park. Tracks of several species were found frequently (e.g., genets and porcupines); while others were found only sporadically (e.g., forest duikers and apes). Differences in crossing behaviour between plots along the road and in the forest interior supported the hypothesis that the presence of a road acts as a barrier for most species. The actual physical obstacles found along the road (e.g., logs, banks, etc.) proved to be highly negatively correlated with crossing probabilities. High vegetation cover was positively correlated to crossing probability. This study proves that roads have a large impact on wildlife, and indicates which factors could be altered during road construction and maintenance in order to mitigate these impacts, such as to maintain a high vegetation cover at shrub level up to the road, and to prevent the roadside from being blocked during construction.

During the study described in chapter 5, the colonisation of roadsides by an invasive plant species was discovered. We surveyed the level of invasion of the roadsides by the invasive shrub *Chromolaena odorata* and found that native plant species of the African ginger family (Zingiberaceae) were outcompeted (**chapter 6**). Zingiberaceae form a key resource for the lowland gorilla. Abundance of these gingers has been largely reduced through displacement by *Chromolaena odorata* in two years time in Campo-Ma’an National Park in

Cameroon. The invasion of this shrub in the whole rainforest region of central Africa and subsequent disappearance of the original vegetation threatens the gorilla in its already precarious existence.

Merging local knowledge in classical scientific activities is possible in a scientific sound way. The advantages in terms of cost and time benefits, plus the potential of increased commitment of the local population to conservation argues for this approach to be adopted on a larger scale. When the first three case studies (chapters 2, 3, and 4) are combined, species and site specific information can be generated that provides on-the-ground management with the means to set conservation priorities. By combining the vulnerability assessment, the base-line density assessment and the level of exploitation in the market survey, a comprehensive analysis of the state of the wildlife emerges. The risk of not noticing threatened species is in this way reduced.

Although more practical and solution-based research within conservation biology is imperative for more effective conservation, fundamental and long-term classical research remains necessary. But only, given the time and money constraints, if they are based on specific questions posed by conservation planners and practitioners. The case studies (chapters 5 and 6) are examples where there is a clear link between classical scientific research and the requirements from conservation planners.

Reconsideration of the role of conservation science is necessary, and testing of new methods proves that it is worthwhile to leave the beaten track. This does require a change of attitude of the conservation biologists. Only by trying new approaches and by testing new methods can one advance conservation. This thesis provides a start by presenting case studies which should stimulate further progress in conservation biology.

Samenvatting

Er heerst een crisis in de biologische tak van wetenschap die zich bezig houdt met natuurbescherming (engels: conservation biology). Deze crisis is het gevolg van de kloof die er is tussen “kennisvergaring” voor, en het daadwerkelijk uitvoeren van de natuurbescherming. Deze “kennisvergaring-uitvoering” kloof is onder meer een gevolg van de discrepantie tussen tijd, budget en aandacht die besteed wordt aan systematisch en fundamenteel wetenschappelijk biologisch onderzoek en het meer praktisch en probleemoplossend gericht onderzoek. In dit proefschrift wordt de toepasbaarheid van de huidige natuurbeschermingsbiologie geanalyseerd aan de hand van voorbeelden uit de praktijk. De klassieke natuurbeschermingbiologie wordt vergeleken en gecombineerd met nieuwe onderzoeksmethodes. Doel is het leveren van op een wetenschappelijke manier verkregen informatie aan uitvoerders in het veld, met als inzet dat deze informatie snel en makkelijk bruikbaar moet zijn. Onderzoeksactiviteiten worden geanalyseerd op kosten en baten en op complementariteit met andere onderzoeksgebieden. De studie is uitgevoerd in het GEF Campo-Ma’an project in het zuiden van Kameroen.

In **hoofdstuk 2** wordt een nieuwe methode voor het schatten van wildstanden gepresenteerd. Deze schattingsmethode levert sneller gegevens op en is goedkoper, terwijl de resultaten vergelijkbaar zijn met data verkregen via klassieke methodes. De huidige methodes voor het schatten van wildstanden in regenwoud kosten veel tijd en geld. Oorzaak is dat er weinig zicht is in het regenwoud en omdat ondanks de hoge soortenrijkdom de dichtheden laag zijn (veel diersoorten, maar weinig individuen per soort). De meest gebruikte schattingsmethodes zijn niet erg precies, maar bij gebrek aan beter worden deze methodes nog geaccepteerd als best toepasbaar.

De in dit hoofdstuk beschreven methode is gebaseerd op de kennis van de lokale bevolking, welke samengevoegd wordt om te komen tot een beeld van het hele gebied. De methode is getest in het veld en vergeleken met conventioneel transectonderzoek dat gelijktijdig uitgevoerd is. De resultaten worden vergeleken met andere onderzoeken in vergelijkbare gebieden op basis van de literatuur. Een aantal jagers is gevraagd om aantallen te schatten van 33 zoogdiersoorten, in een gebied dat van te voren afgebakend was op een kaart. Deze absolute aantallen werden omgerekend naar dichtheden (per km^2), waarna de dichtheden van de verschillende jagers per soort gemiddeld werden en vervolgens geëxtrapoleerd naar het hele studiegebied. De dichtheid van bijvoorbeeld olifanten buiten het nationale park werd geschat op 0.06 dieren per km^2; veel lager dan de 0.3 binnen het park. Het aantal buffels werd geschat op 0.2 dieren per km^2 en het aantal gorilla’s op 1.05 per km^2.

Ondanks dat de traditionele transectstudies meer tijd vergde (vier keer zo lang) en meer kostte (zeven keer zoveel), leverde de conventionele methode te weinig gegevens op om dichtheden te kunnen berekenen. De in hoofdstuk 2 beschreven methode is relatief eenvoudig en goedkoop en daarom goed te integreren in natuurbeheersprojecten in het tropische regenwoud. Tevens is betrokkenheid van de lokale bevolking beter aangezien de gegevens, en dus beleid gestoeld op deze gegevens, hen toebehoren. De hier beschreven methode vormt daarom een goed alternatief voor conventionele methodes van wildstandschattingen in tropisch regenwoud.

In **hoofdstuk 3** wordt een nieuwe methode gepresenteerd die wildsoorten in het regenwoud rangschikt naar risico op uitsterven of snelle achteruitgang van populaties. Het is een snelle, efficiënte en goedkope methode om prioriteiten te stellen in natuurbeheersprojecten. Informatie uit de literatuur en lokale kennis van jagers en boswachters werd gecombineerd om de risicostatus van 33 wildsoorten in het studiegebied te bepalen. Het resultaat is een lijst waar de soorten gerangschikt zijn naar hun risico op snelle achteruitgang en mogelijk (lokaal) uitsterven.

Om deze lijst op te stellen is een systeem ontwikkeld waar de risico-gevoeligheid van elke soort werd bepaald aan de hand van 13 factoren. Deze factoren zijn geselecteerd op hun belang voor het voortbestaan van een wildpopulatie. De factoren werden gewaardeerd door de jagers (bijvoorbeeld: smaak en marktwaarde) en met gegevens uit de literatuur (bijvoorbeeld gewicht en levensduur). De methode werd getest in het Campo-Ma’an gebied. De soort die volgens deze methode het meest risicogevoelig was is de mandril, gevolgd door de olifant, chimpansee, en buffel. Vijf van de tien meest risicogevoelige soorten staan op de rode lijst van bedreigde diersoorten van de IUCN (World Conservation Union). Dit rechtvaardigt het gebruik van deze nieuwe methode voor het stellen van prioriteiten in lokaal natuurbeheer. In dit proefschrift wordt gesteld dat deze methode een bruikbaar instrument is voor mensen in het veld, waardoor middelen besteed kunnen worden daar waar dit het hardst nodig is.

Omdat het detecteren van een trend in wildstanden volgens de geëigende methodes moeilijk is, zijn andere methodes van monitoren nodig. Het verzamelen van socio-economische en biologische gegevens op de markten waar wild voor consumptie wordt verkocht vormt een potentieel makkelijk alternatief. Maar ook al lijken deze marktsurveys eenvoudig, het blijkt dat na analyse deze gegevens niet resulteren in een eenduidige conclusie over de duurzaamheid van de exploitatie van wild. Theoretische modellen met zoveel mogelijk factoren doen het goed in simulaties, maar multivariabele datasets gebaseerd op langlopende veldonderzoeken blijken nog steeds geen duidelijk beeld op te leveren.

Hoofdstuk 4 beschrijft een studie waarin enkele factoren zijn geselecteerd waarvan verwacht wordt dat ze een indicatie geven over de status van de handel in wild. De ‘bushmeat’ markt vormt op zijn beurt een indicatie van de jachtdruk in het regenwoud. Dit zijn afhankelijke factoren, te weten: de marktprijs van het wild, de staat van het wild (gerookt of vers) en de diversiteit van het aanbod (aantal soorten dat te koop wordt aangeboden). Daarnaast zijn er onafhankelijke factoren die samen met de afhankelijke factoren een duidelijk beeld zouden moeten geven van de jachtdruk en daarom van de effectiviteit van het wildbeheer. De onafhankelijke factoren zijn: het aantal inwoners (een maat voor de vraag); de afstand tussen de lokale markt en grote stad; de afstand van de markt tot het nationale park; en de wilddichtheid in de directe omgeving van de markt.

De analyses leverden twee duidelijke relaties op: een negatieve correlatie tussen de prijs en de afstand tot de stad, en een negatieve correlatie tussen het percentage gerookt vlees op de markt en de wilddichtheid in de directe omgeving van de markt. Met andere woorden: hoe dichter bij de stad, hoe duurder het wild werd en hoe hoger de wilddichtheid in de omgeving, hoe minder gerookt vlees er werd aangeboden.

De effecten van wegen door tropisch regenwoud op het gedrag van wilde dieren zijn tot dusver nauwelijks onderzocht. In sub-Sahara Afrika heeft wegenbouw vaak prioriteit, met negatieve gevolgen van deze weg op het wild als gevolg. In **hoofdstuk 5** worden de effecten van een weg op het oversteekgedrag van wild beschreven. De relatie is geanalyseerd tussen het oversteekgedrag van wild en tien factoren waarvan wordt aangenomen dat ze geassocieerd zijn met de aanwezigheid van een weg of de bouw daarvan. De factoren zijn ingedeeld in drie klassen: vegetatie kenmerken (bedekking), fysieke factoren (topografie) en menselijke invloed. Sporenplots langs een 32 kilometer lange weg die het nationale park Campo-Ma'an in tweeën deelt en een aantal plots dieper in het bos leverde gegevens over welke diersoorten de weg overstaken en welke niet. Sommige soorten werden vaak waargenomen langs de weg (bijv. genetkat en stekelvarken), terwijl van anderen slechts incidenteel sporen werden gevonden (bijv. bosduiker en mensaap). De verschillen in frequentie van sporen die gevonden werden langs de weg en in het bos bevestigen de hypothese dat de weg voor veel soorten een barrière vormt in hun natuurlijke beweging. De aanwezigheid van fysieke obstakels langs de weg (bijvoorbeeld omgezaagde bomen, hoge wegkanten of wallen en diepe greppels) was sterk negatief gecorreleerd met oversteekfrequentie van wild. Een hoge bedekkingsgraad van de vegetatie was echter sterk positief gecorreleerd met oversteekfrequenties. Deze studie bewijst dat wegen een groot effect hebben op het wild, en geeft aan wat veranderd kan worden om de negatieve gevolgen van een weg op natuurlijk gedrag van wild tegen te gaan. Belangrijkste factoren zijn dat tijdens de aanleg en het onderhoud van een weg ervoor gezorgd moet worden dat fysieke barrières tot een minimum beperkt worden. Waar mogelijk moet de vegetatie in de wegbermen zoveel mogelijk intact gelaten worden om het wild schuilmogelijkheden te bieden.

Tijdens het onderzoek naar oversteekgedrag van wild in hoofdstuk 5 werd ontdekt dat een invasieve soort struik uit Zuid-Amerika een groot gedeelte van de wegbermen had gekoloniseerd. De mate van kolonisatie werd onderzocht, waarbij duidelijk bleek dat deze struik (*Chromolaena odorata*) de van nature in open plekken en wegbermen voorkomende Afrikaanse gemberplant (*Aframomum spp.*, Zingiberaceae) verdrongen had (**hoofdstuk 6**). Deze Zingiberaceae vormen de belangrijkste voedselbron voor de laaglandgorilla. De Zingiberaceae waren in het Campo-Ma'an Park in twee jaar tijd voor het grootste gedeelte verdrongen door *Chromolaena odorata*. Omdat deze soort dezelfde niche bezet als de favoriete voedselplant van de gorilla (en andere wildsoorten), en gezien het feit dat de spreiding van deze plant dermate snel en agressief plaatsvindt in heel Centraal Afrika, wordt de gorilla in zijn voortbestaan bedreigd.

Het integreren van lokale kennis in traditionele wetenschappelijke studies is mogelijk op een wetenschappelijk verantwoorde manier. De voordelen in termen van kosten, tijd en betrokkenheid van de lokale bevolking pleiten ervoor dat deze benadering op grotere schaal ingezet wordt. Als de drie praktijkvoorbeelden (Hoofdstukken 2, 3 en 4) worden gecombineerd wordt er soorts- en lokatie-specifieke informatie gegenereerd die natuurbeheersprojecten de mogelijkheid bieden om snel en effectief prioriteiten te stellen in geplande of bestaande activiteiten. Door de risicogevoeligheid van soorten, de wildstandschatting en de mate van jachtdruk te combineren ontstaat een duidelijk beeld van de status van het wild. Omdat informatie gegeven wordt die alle middelgrote tot grote zoodieren beschrijft, wordt het risico dat er soorten ongemerkt achteruitgaan beperkt, met een effectiever wildbeheer als gevolg.

In dit proefschrift wordt betoogd dat het ontwikkelen en testen van meer praktisch en probleemoplossend onderzoek essentieel is voor het op een hoger plan tillen van de natuurbeschermingsbiologie. Dit wil echter niet zeggen dat fundamenteel wetenschappelijk en lange-termijn onderzoek achterhaald is. Dit soort onderzoek moet gedaan blijven worden, echter, gegeven de beperkt beschikbare fondsen en tijdsdruk die natuurbeheersprojecten vrijwel altijd ondervinden, zal dit gebaseerd moeten worden op de vraag vanuit de natuurbeheerders en planners in het veld. Uit de praktijkvoorbeelden (hoofdstuk 5 en 6) blijkt dat traditioneel wetenschappelijk onderzoek duidelijk gekoppeld kan zijn aan een concrete vraag vanuit het veld.

Het heroverwegen van de rol van natuurbeschermingsbiologie is noodzakelijk en het testen van nieuwe methodes zoals beschreven in dit proefschrift toont aan dat het de moeite waard is om af te wijken van de gebaande paden. Dit vereist echter een verandering van houding van de wetenschappers die claimen onderzoek te doen voor natuurbescherming. Natuurbescherming kan zich slechts ontwikkelen als nieuwe methodes wordt gezocht en vooral in het veld worden getest. Dit proefschrift geeft met praktijkvoorbeelden een aanzet hiertoe, in de hoop dat het de discussie over en verdere ontwikkeling van natuurbescherming stimuleert.

Résumé

La biologie de la conservation fait face à une crise de mise en oeuvre. Cette crise résulte d'un déficit de savoir-faire ou d'évaluation-planification. Une raison pour cet état de chose est la divergence entre les enquêtes ou évaluation systématiques scientifiques classiques et l'exécution des projets sur le terrain. Cette thèse explore l'état des lieux de la biologie de la conservation en débattant l'aspect pratique de plusieurs activités de recherche qui sont nécessaires dans la plupart de projets de conservation de la biodiversité. La science de la conservation classique est comparée et associée à des méthodologies nouvelles développées. L'objectif était de produire un paquet exhaustif d'information pour la planification et la mise en oeuvre de la conservation. Les activités de recherche étaient analysées en fonction de la complémentarité, les contraintes de coûts et de temps, et selon les possibilités d'intégration des connaissances locales. Cette étude a été menée dans le cadre du projet GEF-Campo-Ma'an dans la région de forêt tropicale humide du Sud Cameroun.

Dans le **chapitre 2** une nouvelle méthode d'estimation de la densité de la faune est expliquée, qui prend peu de temps et d'argent, mais produit des résultats comparables aux méthodes classiques. Les méthodes actuelles utilisées pour l'estimation de la densité de la faune dans la forêt tropicale humide sont onéreuses et prennent trop de temps. La précision de la plupart des méthodes les plus utilisées est discutable, mais acceptée parce que des méthodes plus exactes ne sont pas disponibles. Cette méthode a été testée sur le terrain et comparée aux échantillonnages par transects dans la région et de la littérature appropriée. La méthode de PLEO (Pooled Local Expert Opinion) est basée sur les connaissances des experts locaux. Un nombre de chasseurs était appelé à estimer l'abondance de la faune dans une région spécifique, après quoi la densité au km^2 a été calculée pour 33 espèces fauniques. Ces estimations ont été regroupées et extrapolées pour toute la région d'étude. La densité d'éléphant (*Loxodonta africana*) hors du Parc National a été estimée à 0.06 animaux au km^2, et 0.3 à l'intérieur. Celle du Buffle (*Syncerus caffer*) dans la région d'étude fut évaluée à 0.2 animaux par km^2 et le gorille (*Gorilla gorilla*) à 1.05 au km^2. Les échantillonnages par transects effectués pendant la même période par contre, ont coûté considérablement beaucoup plus d'argent et de temps; et ont produit beaucoup moins de données pour pourvoir calculer les densités. L'évaluation de la méthode-PLEO était favorable et la méthode offre un substitut pour les méthodes conventionnelles d'estimation des densités de la faune dans les forêts humides. Cette méthodologie est simple et peut être incorporée dans plusieurs projets de biodiversité et conservation dans les régions tropicales. Elle peut aussi être utilisée pour le suivi-évaluation sur le long terme de l'état de la faune dans une région donnée. Contrastant avec les méthodes classiques, la méthode-PLEO est moins chère et assure l'appropriation locale des résultats.

Dans le **chapitre 3** une nouvelle méthode est présentée qui classe les mammifères de moyenne et grande taille dans la forêt humide, en fonction de leur vulnérabilité à l' extinction ou à une réduction majeure de population. C'est une rapide, efficiente et moins chère méthode pour fixer les priorités en matière de gestion de la conservation. Les informations de la littérature et les connaissances locales des chasseurs et population de forêt ont été regroupées en vue d’évaluer l’état de 33 espèces fauniques. Le résultat est une liste de vulnérabilité, où les espèces sont classées selon leur vulnérabilité aux disparitions et réductions majeures. Pour produire cette liste nous avons développé un système où la prédisposition au risque de chaque espèce a été déterminée sur la base de treize facteurs. Ces facteurs sont présumés être d’importance pour la survie d’une population, auxquelles étaient reportées des données des interviews auprès des chasseurs locaux, et de la littérature. La méthode a été testée dans la région de Campo-Ma’an du Sud Cameroun. Dans cette étude l’espèce la plus vulnérable est le mandrill, suivie de l’éléphant, le chimpanzé et le buffle. Cinq des espèces les plus vulnérables figurent sur la liste rouge des espèces menacées de l’ UICN, ce qui justifie l’utilisation de la nouvelle méthode pour établir les priorités locales de conservation. Nous argumentons que la gestion sur le terrain de cette méthode fournit un outil utile pour l’allocation des ressources en argent et en temps aux espèces qui en ont le plus besoin.

Parce que la détection des tendances au sein des populations fauniques reste difficile, d’autres voies pour le suivi sont nécessaires. Regrouper les données socio-économiques et biologiques sur les marchés de viande de brousse pourrait être une méthode relativement facile pour faire le suivi du commerce de la viande brousse. Bien que la collecte des données soit assez simple, les analyses de ces données sont rarement concluantes en matière de durabilité des prélèvements. Les modèles théoriques qui incluent autant de variables que possible présentent de bonnes performances lors des simulations, sur le long terme cependant, les ensembles de données à variables multiples ne fournissent pas encore des résultats clairs et tranchés. Le **Chapitre 4** fait la présentation d’une enquête où seuls quelques facteurs supposés indiquer l’état du commerce de la viande brousse ont été sélectionnés. L’analyse des relations entre ces facteurs dépendants et plusieurs facteurs fixes (indépendants) qui par hypothèse influencent cet état a été effectuée. Ces facteurs supposés être indicateurs de l’état du commerce de la viande brousse sont: le prix, l’état (le pourcentage de la viande séchée), et la diversité de la viande de brousse en vente (en terme de nombre des espèces en vente). Ceux-ci sont supposés liés à plusieurs facteurs qui influencent la pression de la chasse qui sont: la taille de la population humaine (comme un indicateur de la demande de la viande brousse); la distance entre les marchés locaux et les marchés des villes; la distance au Parc National; et la densité de la faune dans la zone avoisinant le marché. Seules deux relations claires ont été trouvées après l’analyse. Celles-ci sont une corrélation négative entre le prix de la viande de brousse et la distance entre la marches locales et la ville, et une corrélation négative entre le pourcentage de la viande séchée sur le marché et la densité de la faune dans la zone avoisinant le marché où l’enquête est menée.

Les effets négatifs des routes sur la faune dans les forêts tropicales humides en Afrique sont peu connus. La construction de la route est une grande priorité en Afrique, avec comme effet que les impacts négatifs de la construction des routes sur la faune sont généralement négligés. Le **Chapitre 5** fournit les informations sur les effets des routes sur le comportement de traversée de la faune de la forêt humide. La probabilité de traversée de la faune de forêt humide a été analysée pour associer à dix facteurs différents qui étaient liés à la présence ou à la construction de la route. Les facteurs étaient divisés en trois classes: le couvert végétal, la topographie ou l’influence humaine. Les parcelles à empreintes ont été disposées sur 32 km le long d’ une route forestière non bitumée qui divise en deux le Parc National de Campo-Ma’an. Les traces de plusieurs espèces ont été fréquemment retrouvées (ex: genettes, et porcs-épics); pendant que d'autre étaient retrouvées que de manière sporadique (ex: céphalophe de forêt et des singes). Les différences dans les comportements de traversée entre les parcelles le long de la route à l'intérieur de la forêt a confirmé pour la plupart l'hypothèse selon laquelle la présence d'une route agit comme une barrière. Le véritable obstacle physique le long de la route (ex: les billes de bois, les rives, etc.) s'avère être grandement en corrélation avec la probabilité de traversée. Cette étude prouve que les routes ont un grand impact sur la faune, et indique les facteurs qui pourraient être éviter lors des travaux de construction et de maintenance pour diminuer ces impacts tels que: le maintien d'un grand couvert végétal au niveau des buissons jusqu' au niveau de la route, et l'obstruer les bordures de la route lors de travaux de construction.

Pendant l'étude décrite au chapitre 5, la colonisation des abords des routes par une espèce de plante invasive a été découverte. En effet nous avons enquêté le niveau d'invasion des abords de la route par d'espèce d'arbuste Chromolaena odorata et découverts que les espèces de plante locale de la famille du gingembre africain (Zingiberaceae) était les plus détruites par ces plantes invasives (**Chapitre 6**). L'abondance de ces gingembre a été largement réduite du fait de l'invasion à Chromolaena odorata en deux ans dans le Parc National de Campo-Ma'an. L'invasion par cet arbuste dans toute la région de forêt tropicale humide de l' Afrique Centrale et la disparition subséquente de la végétation originale menace l'existence déjà précaire du gorille.

Il est possible d'associer de manière scientifiquement sensée les connaissances locales dans les activités scientifiques classiques. Les avantages en terme de bénéfices en coûts et temps, et le potentiel d'un engagement croissant de la population locale à la conservation militent pour une adoption à large échelle de cette approche.

Quand les premiers trois études de cas (chapitres 2, 3, et 4) sont conjugués, les informations spécifiques au site et sur les espèces peuvent être générées qui fournissent a la gestion sur le terrain les moyens de déterminer les priorités de la conservation. En associant l'évaluation de la vulnérabilité, les données de base sur les densités et le niveau d'exploitation sur les enquêtes de marche, une analyse exhaustive de l'état de la faune émerge. Le risque de ne pas noter les espèces menacées est de cette manière réduit.

Bien qu'une recherche plus pratique basée-sur-solution au sein de la biologie de la conservation soit impérative pour une conservation plus effective, une recherche classique fondamentale reste nécessaire. Mais seulement, lorsque les contraintes financières et de temps sont remplies, et si elle se base sur des questions spécifiques posées par les planificateurs et les praticiens de la conservation. Les études de cas (chapitre 5 et 6) sont des exemples où le lien entre la recherche scientifique classique et les conditions des planificateurs de la conservation est clair.

La reconsidération du rôle de la science de la conservation est nécessaire, et tester de nouvelles méthodes prouve que quitter les sentiers battus en vaut le coup. Ceci requiert un changement d'attitude des biologistes de la conservation. Seulement en essayant de nouvelles approches et en testant de nouvelles méthodes peut on faire avancer la conservation. Cette thèse en fournit un début en présentant des études de cas qui pourraient stimuler davantage de progrès dans le domaine de la biologie de la conservation.

Acknowledgements

This thesis is the outcome of a process in which I wrestled to combine science with practical field work for the project I worked for. As the starting point was a regular job with little scientific pretences, but rather the production of biological information fast and easily accessible for project management, transforming and upgrading the data and research to a scientific level worthy of a PhD title was quite difficult, to put it mildly. However, I find it very important that my experiences, not only the results but also the methodological process, need to be made known to the people who undoubtedly have or will encounter the same problems as I did when dealing with biodiversity conservation. As expressed in this thesis, there is a gap between conservation practitioners and conservation scientists, part as a result of the lack exchange of information within and between the two. I have tried to fill this gap to a certain extent. Hopefully it will help colleagues in the field, but also in the developed world where policies are made and funding is distributed. Many people helped me to achieve this.

It was my promotor Herbert Prins, who was my scientific adviser for my regular work at the Campo-Ma'an project, who offered me the possibility to write a PhD thesis at the Resource Ecology Group at the Wageningen University. This gave me the opportunity to reach my goal as described above in the best way possible. For this I am very grateful! That is the official thanks. The real support, however, is not by providing this opportunity, but rather the faith in the outcome (thus my person), and the endless process of stimulating and subtle leading my quest in a manner where along the way I learned the value of science in general and the worth of doing a PhD in particular. If I had known what writing a PhD would entail at the time when Herbert asked me (while we floated in the Atlantic Ocean at the beach in Kribi!) I do not know whether I would have started at all! But now I have and for your support, Herbert, I am really grateful, thanks!

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Living in a remote place in Cameroon intensifies the relation with your direct colleagues. Beautiful as Kribi can be (white beaches, nice nightlife, phenomenal natural beauty, rich cultural life); you try to keep some of your home-country feelings alive. Patricia Bekhuis, Jacqueline van de Pol, Nic Hendriks, Jacintha de Huu, Johan Verhoeff, Vranck de Wildt, Tinus de Kam, Monique van Meegeren and Jean Pierre Fines helped to give me the great time

I had in Kribi and the Campo-Ma’an area! I hope we keep in contact as we have done until now!

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Far away from the comfort and pleasure of Kribi were the people that did the hard work, without whom I could not have written this thesis. Loin de la confort et plaisir de Kribi se trouve les gens qui ont fait le vrai travail, sans lequel la réalisation de cette thèse n’était pas possible. Pour les contacts locaux, et la collection des données sur la marché de viande de brousse (chapitre 4) je remercie Mbotto Albert à Campo, Ondoa Zo’onna Jean Claude à Hevecam, Ndjetoh Micheline et Ndemengana Verlaine à Ma’an, Oyene Eleye Léonie à Ipono et la defunt Zamo Christine Liliane à Akom II.
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Back in the Netherlands my co-promotor Fred de Boer of the Resource Ecology group of Wageningen University did a fantastic job in helping me analyse the data (over and over again), revising manuscripts and suggesting new approaches. All this with continious optimism, although I can imagine there surely were moments that you must have sighed and maybe even doubted whether there would ever be an ending… Your practical advice and ability to relativize formed a nice combination with the more philosophical and “bird view” approach of Herbert. All students are thanked for their companionship and help: Jasja, Nicol, Thomas, Geerten, Euridice, Jasper, Anna, Jort, Bjorn, Roy, Juul, Gabriela, and all former and new students I did not mention. The REG staff is equally thanked, for work and pleasure:

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This thesis is about biodiversity conservation in tropical rainforests, and could have been done anywhere in the developing world where rainforests are found. However, it is my parents and their choice to work and live in Zambia in the 1970s that are responsible for my love for Africa. How difficult it must have been for you to see that your son yearned and actually planned to go back, while you had to leave Africa with the great sorrow of losing Jesse to this continent. When I first went back (to Ivory Coast, in 1994) I did not fully realise what this meant to you. Later I did, and I am deeply thankful for your visits during my various stays abroad. For your love, your belief in me and your always constructive advice and thoughts, and for your devotion to always help less fortunate people, I dedicate this book to you!

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Curriculum vitae

Christiaan Aat van der Hoeven was born in Deventer, the Netherlands, on 22 March 1970. After graduating from the Atheneum at the Rijnlands Lyceum in Oegstgeest in 1989, he studied biology at the Utrecht University, where he obtained his Masters Degree in 1997. He specialised in animal behaviour, with two field studies in Africa. The first was an ethological study on the cryptic olive colobus (*Colobus verus*) at the Taï Monkey Project in the Taï National Park in Ivory Coast in 1994 and 1995. This study was conducted in the framework of the Ethology department of Utrecht University (supervision: Dr. E.H.M. Sterck and Prof. dr. J.A.R.A.M. van Hooff) and the Max Planck Institüt für Verhaltensphysiology, Seewiesen, Germany (supervision: Dr. R. Noë). The second was a behavioural ecology study on elephants in Waza National Park, Cameroon in 1996. This study was conducted in the framework of the Institute of Environmental Sciences in Leiden at the field site of the Centre d'Etude de l'Environnement et du Développement in the extreme North of Cameroon (supervision: dr. Hans de Iongh and dr. Hans Bauer).

He worked as a research coordinator in 1998 and 1999 at the Ray Research Centre in the Benoué region in North Cameroon. He supervised Msc. students in research on ecology and behaviour of the Kob antelope (*Kobus kob*).

After conducting a short desk study in 1999 on the program “Biodiversity in Disturbed Ecosystems”, which is a collaboration between the Netherlands Organisation of Scientific Research (NWO) and Tropenbos International, he returned in 2000 to Cameroon, this time in the humid South, at the GEF Campo-Ma’an project in Kribi. This was a World Bank/GEF project with three executing partners: Tropenbos International, SNV (a Dutch development organisation), and the Cameroonian Ministry of Environment and Forestry (the former MINEF). He was contracted to develop a wildlife and National Park management plan. It was during this contract that in consultation with his scientific supervisor (Prof. H.H.T. Prins) it was decided to write a PhD thesis on his work in the Campo-Ma’an project. When he was back in the Netherlands he proceeded to write the current thesis at the Resource Ecology Group of the Wageningen University.

Because funding for writing a thesis is hard to come by, he worked part-time from 2002 until now, first as a secretary for the Working group on Ecology and Development, and now still as programme officer for the Ecosystem Grants Programme of the IUCN Netherlands Committee.

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PE&RC PhD Education Certificate

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Review of Literature (5.6 credits)

- Which criteria are currently used in defining biodiversity hotspots in African rainforests? (2003)

Writing of Project Proposal (5.6 credits)

- A new method for assessing mammal status in tropical rainforests based on local knowledge (2003)

Post-Graduate Courses (2.8 credits)

- Biodiversity assessment; Biodiversity school RUL (1998)
- Advanced statistics; PE&RC (2003)

Deficiency, Refresh, Brush-up and General courses (0.3 credits)

- Course on communication; PE&RC (2003)

Competence Strengthening / Skills Courses (4.2 credits)

- Basic statistics; PE&RC (2003)
- Scientific writing; PE&RC (2003)
- Project management and time planning; PE&RC (2003)

Discussion Groups / Local Seminars and Other Scientific Meetings (4 credits)

- Forest & conservation ecology (2003 and 2004)
- Current themes in ecology: global ecology, ecological implications of adaptive behaviour, biological invasions (2003 and 2004)
- KNAW symposium: responsibilities of environmental research: anthropogenic influences on the earth as ecosystem (2004)

PE&RC Annual Meetings, Seminars and the PE&RC Weekend (0.3 credits)

- Fontis workshop on large herbivores (2003)
- PE&RC symposium ecosystems response to climate change (2004)

International Symposia, Workshops and Conferences (10.3 credits)

- Links between biodiversity conservation, livelihoods and food security: the bushmeat problem (IUCN/Traffic/FAO), Cameroon (2000)
- Cameroon bio-monitoring network (2001)
- Congress: globalisation & localisation: UvA, CEDLA, VU, IAC, Tropenbos, etc. (2003)
- Gaining ground: in pursuit of ecological sustainability; IFAW Forum, Ireland (2004)

Courses in Which the PhD Candidate Has Worked as a Teacher

- Theoretical course Park & Wildlife management; CML. Leiden University; 6 days (2004 and 2005)