

Cooperation or Competition
Dilemma for Resource Managers
in
Sustainable Wildlife Utilisation

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Cooperation or Competition Dilemma for Resource Managers in Sustainable Wildlife Utilisation

Emmanuel Mwakiwa

Thesis

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To
my wife: Joyce;
my daughters: Josephine Lorraine, Joylynn Tarnia & Jacqueline Rose;
my mother: Rachel Makiwa;
&
in memory of my father: Jackson Makiwa

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Abstract

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Wildlife as part of biodiversity is a global natural resource. However, landowners have some control over the future of wildlife on their land. Wildlife could be managed by the state or private landowners. The survival of the wildlife and their habitats is determined by how these landowners decide to use the land and the renewable resources on it. Some complication come into place given that wildlife usually roam on land held over by more than one owner providing more challenge to its management. In addition, wildlife as a natural resource has multiple uses that generate revenues for the betterment of the landowners. The uses could be consumptive or non-consumptive. Each landowner has multiple objectives which might be conflicting which poses even a greater challenge to the sustainable wildlife management.

To meet their objectives wildlife managers use management tools. Some of the tools used include constructing or closing of artificial waterpoints, fire management, fencing, and population manipulation through culling/hunting or translocation of animals. However, use of these management tools can lead to unintended or opposite effects if they are not well understood. There are direct and indirect effects of the tools on biodiversity. Landowners could be tempted to excessively use some of the tools in order to achieve their objectives. In addition, most studies have concentrated on either the ecological or economic effects of the wildlife management tools. For the landowner, it is essential that he comprehends both the ecological and economic effects of the wildlife management tools for the sustainable management of wildlife, a contribution of this study.

The main objective of this study is to assess the ecological and economic implications of some wildlife management tools on the landowners' welfare. I use simple ecological economic analytical models based on the Pontryagin's maximum principle to perform the analyses. The Savanna ecosystem model which is a spatially explicit, process-oriented model is also used to further explore the effects of one of the wildlife management tools on landowner's multiple objectives.

One of the tools that is analyzed in this thesis is the improvement of land productivity through increasing of vegetation quality. Given that, it is usually not easy to increase the land size in response to increased incentives, some landowners might consider increasing the land productivity. The results show that utilization of wildlife can contribute to wildlife conservation and enhancement of welfare as a result of investment by landowners into habitat quality improvement. However, the use of a wildlife management tool has direct and indirect effects as demonstrated by another framework presented in this thesis on waterpoints. Waterpoints are used by wildlife managers to supplement natural water supplies which in turn support herbivore populations, like elephants. A private oriented landowner may be interested only in maximization of profits or personal benefits either from elephant offtake and/or tourism revenue, thus might ignore the negative effects that could be brought about by elephants to biodiversity. In such case, the game reserve management as the authority entrusted with sustainable

management of the game reserve should use economic instruments such as subsidies or payments for the compliant landowners and/or taxes or charges for the non-compliant landowners to encourage compliance with sustainable wildlife management practices.

The Savanna ecosystem model is used to explore the effects of waterpoints on elephant density (representing an economic objective) and biodiversity (representing an ecological objective). The model is used to analyze the differential impact of waterpoints on the Kruger National Park's regions under 26 waterpoints manipulation scenarios. The model is also used to analyze elephant impact on vegetation biomass diversity in four regions of Kruger National Park. The results showed that constructing (or closing) extra waterpoints in one region does not necessarily translate into higher (or lower) elephant densities in that region, but the effect depends on the vegetation and other conditions of the region in comparison to neighbouring regions. In one of the regions, the model showed that there is a trade-off between elephant density and vegetation biomass diversity. In another region, elephants' effect on vegetation biomass diversity follows the intermediate disturbance hypothesis, whilst in other regions the relationship is positive. The intermediate disturbance hypothesis postulates that there would be a higher diversity of vegetation structure at intermediate elephant densities whilst at extreme levels of both low and high disturbance the diversity would be reduced. The model thus suggests that different strategies should be adopted for different regions, e.g., an adaptive management strategy could be used for one of the regions where waterpoints are switched on and off depending on the elephant density.

Another wildlife management tool that is analyzed is the use of physical barriers like fences. Physical barriers could be utilised by landowners to separate different wildlife uses which might be conflicting. Landowners or game reserve management are often faced with the decision whether to undertake consumptive (hunting) and/or non-consumptive (tourism) use on their properties. A theoretical model is constructed to examine these cases. The results show that that the two uses can be undertaken in the same contiguous area if the consumptive use is not dominating.

In conclusion, what emerges from this work is that given that the landowner's welfare is not only affected by his own actions but also his neighbours' *modi operandi*, then the landowner should consider all levels of cooperation with his neighbours in order to fully maximize his welfare. This includes cooperation in terms of which management tool(s) he and/or his neighbour should use. The frameworks presented in this thesis could be used by landowners (both state and private) to analyze the effects of their management actions on their welfare.

Keywords: analytical modelling; Associated Private Nature Reserves; consumptive use; elephants; Kruger National Park; land productivity; non-consumptive use; waterpoints; Savanna ecosystem model; South Africa

General introduction

Emmanuel Mwakiwa

1.1 Wildlife as a natural resource

Wildlife as part of biodiversity is a global natural resource. Conserving wildlife does not only benefit the custodians of wildlife at landowner and country levels, but also has positive effects at global level. However, as mentioned by Child (2000), landowners have some control over the future of wildlife on their land. The survival of the wildlife and their habitats is determined by how these landowners decide to use the land and the renewable resources on it. They would stimulate wildlife conservation if their welfare requirements are met (Brandon and Wells, 1992; Child, 1988). However, some complication come into place given that wildlife usually roam on land held over by more than one owner providing more challenge to its management.

Wildlife as a natural resource has multiple uses that generate revenues for the betterment of the landowners. The uses could be consumptive or non-consumptive. Non-consumptive uses include: tourism, photographic safaris or wildlife viewing. On the other hand consumptive uses include: hunting for trophies, harvesting for meat and/or hides. Furthermore, there are also some external benefits which accrue to others besides the landowners directly managing the wildlife. Some of these benefits include revenues which accrue to safari operators, the state, and local communities and airlines, charter companies, taxidermy companies, hotels, restaurants, firearms and ammunitions dealers, camp equipment manufacturers, amongst others.

Wildlife population, including elephants, in southern Africa especially South Africa has been steadily increasing in contrast to the downward trend observed in other parts of the continent. For instance in Kruger National Park alone, the annual elephant count has increased from fewer than 7000 in 1995 to close to 13000 animals in 2005 (Marshall, 2004). Elephants and other larger herbivores are an important component of the ecosystems that they live in.

Elephants are regarded as a charismatic megafauna species because of their size, intelligence, and social structure (Owen-Smith, 1992). Due to their size, they engineer the ecosystem through direct or indirect impact on the availability of resources to other organisms and creation of habitat heterogeneity and possibly homogeneity under certain conditions (Owen-Smith, 1992; Kerley et al., 2008; Valeix et al., 2008). Elephants can cause noticeable effects on vegetation composition and influence the balance between trees, shrubs and herbaceous layers in plant communities (Holdo, 2007; Fornara and du Toit, 2008; Kerley et al., 2008; Valeix et al., 2008).

1.2 Wildlife management

Wildlife could be managed by the state or private landowners. Both entities play a crucial role in wildlife management in southern African countries such as South Africa (Child, 2000; Muir-Leresche and Nelson, 2000). The study area for this thesis is South Africa. I infer to both public and private landowners involvement in wildlife management.

1.2.1 Wildlife management by the public sector

On behalf of the state, the South African National Parks (SANParks) is the authority responsible for managing a system of 21 national parks throughout South Africa and other conservation areas, covering a total area of more than four million hectares of land, or about 5.8% of the total area of South Africa (Hearne et al., 2000; Krug, 2001; ABSA, 2003; Department of Environmental Affairs and Tourism, 2005).

As an authority vested with wildlife management, SANParks has three key objectives, viz, conservation, people-centred conservation and eco-tourism (SANParks, 2006). Biodiversity conservation is mentioned to be SANParks major goal. However, recently, revenue collection through tourism also gained a prominent role in the operation of the wildlife protected areas to support in the attainment of the major goal of biodiversity conservation. In September 1998, South African government's Department of Environmental Affairs and Tourism articulated the need for SANParks to prepare for a lesser dependence on state funding, which would increasingly be aimed at funding the essential conservation requirements (Fearnhead, 1999; Mabunda, 2004; SANParks, 2007).

The Kruger National Park (KNP), as the biggest park managed by SANParks is increasingly under pressure to generate its own funding as park management funding shifts from government grants to income from tourism. The KNP is the one of the oldest national park in Africa, established in 1898, and one of the 10 largest game reserves in the world. The park is

South Africa's flagship conservation area and is regarded as one the best managed national parks in Africa. Stretching 350 km from north to south, at most 60km in width and covers an area of 1,948,528 hectares. The KNP is located to the northeast of South Africa and is bordered to the east by Mozambique and to the north by Zimbabwe. The park is known for its diverse range of species: 336 trees, 49 fish, 34 amphibians, 114 reptiles, 507 birds and 147 mammals (SANParks, 2008). Among the mammal species present in the park are the big five – lion, elephant, leopard, buffalo and rhino. The KNP was ranked 7 among the top 10 attractions visited by international tourists in South Africa, accounting for 15% and 13% of all foreign arrivals in 1999 and 2000 respectively (World Tourism Council, 2002).

KNP is divided into four regions, south, central, north and far north, with the far north of the park as the most difficult area to access (SANParks, 2009). There are a total of six perennial rivers that cross the park from west to east. At least 300 boreholes were drilled between the 1930s and 1980s to provide reliable surface water throughout the year (Gaylard et al., 2003). However, the KNP revised the water provision policy in 1997, which resulted in at least half of the waterpoints being closed down (Pienaar et al., 1997; Smit et al., 2007b).

1.2.2 Wildlife management by the private sector

In South Africa, privately protected areas are more than public protected areas both in terms of numbers and total area covered (Watkins et al., 1996). There are over 5000 game ranches and about 4000 mixed game and livestock ranches, which together constitute about 13% of the country's total land area, compared with 5.8% covered by public conservation areas (Hearne et al., 2000; Krug, 2001; ABSA, 2003).

In South Africa, there are three main types of private landownership involved in wildlife management namely: game ranching, private conservancy, and private nature reserves. Game ranching involves the raising of wildlife animals for a variety of products, including meat, hides, feathers and antlers. Game ranching involves intensive, small-pasture production of wild animals and it involves extensive, wide-ranging production. Game ranching is often combined with domestic livestock ranching. On the other hand, private conservancy consists of a group of wildlife ranches where the landowners pooled natural and financial resources together to conserve wildlife and they operate through a committee. Members sometimes practise normal farming activities and operations in combination with wildlife conservation. Each conservancy has its own constitution containing a set of legally binding wildlife management and conservation objectives (Krug, 2001).

Private nature reserve is the third type of land ownership and would be the main focus of this thesis. I will focus on the case of Associated Private Nature Reserves (APNR). Management objectives of private nature reserves vary from preservation to the sustainable use of wildlife. Some nature reserves have formed associations whereby adjoining landowners pooled their resources to create larger units but still retaining individual ownership. One of the oldest private reserves in the north-eastern lowveld of South Africa, is the Timbavati Private Nature Reserve which was formed in the 1950s (Krug, 2001). In addition, in some cases private nature reserves have entered into agreements with state-owned national parks to enable the removal of fences between them. An example of such an arrangement is the agreement between Kruger National Park and the Associated Private Nature Reserves (comprising of Timbavati, Klaserie, Umbabat and Balule Private Nature Reserves) to the west of Kruger which led to the removal of fences between Kruger National Park and these neighbouring private reserves in 1994 (Krug, 2002; APNR, 2005).

Little is known about the private sector's potential role in managing and supporting long term biodiversity (Scriven and Eloff, 2003). Some of the landowners in the game reserves do not entirely depend on their reserves for revenue generation as they have other sources of income (APNR, 2005). Private landowners maintain largely natural conditions on their land thereby contributing to conservation.

1.3 Wildlife management tools

To meet their objectives wildlife managers use management tools. Some of the tools used include constructing or closing of artificial waterpoints, fire management, fencing, and population manipulation through culling/hunting or translocation of animals (Perrings and Walker, 1997; Slotow et al., 2005; de Boer et al., 2007; Grant et al., 2002). However, the use of these management tools can lead to unintended or opposite effects if they are not well understood (Pienaar et al., 1997). There are direct and indirect effects of the tools on biodiversity. Landowners could be tempted to excessively use some of the tools in order to achieve their objectives (Witkowski, 1983; Parker and Witkowski, 1999).

So there is need for better understanding of the effects of wildlife management tools before the landowners use them to manipulate wildlife. Getting a better understanding of the economic and ecological effects of the management tools is important for the sustainable management of wildlife. Managers have been concerned by the unwanted increases of some species at the expense of others in protected areas (Prins and van der Jeugd, 1993; Pienaar et al., 1997; Grant et al., 2002; Chamaillé-Jammes et al., 2007; Holdo et al., 2009). The

development of predictive models and ecological economic analytical frameworks of biodiversity dynamics could therefore assist.

1.4 The TEMBO programme

This research is part of the Temporal Elephant Movement and Bio-economic Optimization (TEMBO) programme. The TEMBO integrated programme looked at an important tri-trophic system: vegetation - herbivores - humans in and around the Kruger National Park (KNP), South Africa. The integrated programme comprised six subprojects that together aimed at analysing and predicting the spatio-temporal distribution of elephants in conservation areas in relation to the nutrient status of the vegetation, in order to carry out a spatially explicit cost-benefit analysis for the optimisation of resource management for commercial purposes and conservation. This subproject will contribute to the overall programme by using the analytical models and a spatio-temporal model for elephant distribution to analyse management actions. It further incorporates the management objectives of the different land users (Kruger National Park and surrounding private and corporate conservation areas).

1.5 Research objectives

The main objective of this study is to assess the ecological and economic implications of some wildlife management tools on the landowners' welfare. This objective was guided by the following specific research questions:

- (a) What is the effect of improving wildlife habitat quality on landowner's welfare?
- (b) What is the impact of waterpoint manipulation on both ecological and economic objectives?
- (c) What are the consequences of closing and opening waterpoints at regional level in a big national park?
- (d) Under what conditions would a landowner choose between wildlife consumptive and/or non-consumptive use?

I use simple ecological economic analytical models to answer these questions. Although I made several simplifications to the models, the findings will be useful to wildlife management since scientific theory is often developed from testing simple models in order to better understand complex systems (Rosenzweig, 1973; Noy-Meir, 1975; Starfield, 1997). I also use the Savanna ecosystem model (Coughenour, 1993; Kiker, 1998; Boone et al., 2002) which is a spatially explicit, process-oriented model to explore the effects of waterpoints on biodiversity.

1.6 Outline of the thesis

Due to increasing human population, high land prices and inelastic supply of land, it is usually not easy to increase the land size in response to increased incentives. Under such conditions, chapter 2 analyzes the effects of improving land productivity on landowner's welfare.

Landowners operate in a multiple objective environment, and sometimes these objectives are in conflict. Representing multiple objectives where the landowner has two objectives – one ecological and another economic, chapter 3 examines the direct and indirect effects of establishing waterpoints on wildlife diversity. Still looking at waterpoints and multiple conflicting objectives, chapter 4 analyzes the effects of waterpoints establishment or closure using the Savanna ecosystem model.

Given that there are some conflicting but also complementary effects of having non-consumptive use and consumptive use in the same contiguous area, chapter 5 explores the effects on landowner's welfare who is contemplating on whether to completely separate, partially separate, or mix these two uses.

Chapter 6 looks at the main findings and review them in context of the other chapters. In addition, a number of recommendations drawn from the study are set forth with respect to tools and strategies used in wildlife management.

Analyzing the effects of wildlife habitat quality enhancement on landowner's welfare using bio-economic modelling

Emmanuel Mwakiwa; Johannes D. Stigter; Willem F. de Boer; John W. Hearne; Rob Slotow; Cornelis van der Waal; Yolanda Pretorius; Andrew K. Skidmore; Mike Peel; Ignas M.A. Heitkönig; Cornelia C. Grant; Frank van Langevelde; Henrik J. de Knegt; Herbert H.T. Prins

Abstract

It is usually not easy to increase the land size in response to increased incentives. In that case the landowner could increase land productivity instead. We introduce the concept of investing in habitat quality improvement by developing a mathematical model for wildlife utilization. The model has two state variables, vegetation biomass and herbivore biomass and two control variables, the investment in habitat quality improvement and the herbivore offtake rate. We use empirical results from a large scale nutrient supplement experiment that we carried out in the Associated Private Nature Reserves, South Africa, to calibrate the model and demonstrate the effects of an increase in land productivity on elephant biomass. We show that utilization of wildlife can contribute to wildlife conservation and enhancement of welfare as a result of investment by landowners into habitat quality improvement.

Keywords: resource utilization; herbivore offtake; fixed land size; optimization; elephants

2.1 Introduction

There are different strategies that landowners can use to increase their welfare in wildlife management. Traditional management instruments include harvesting and investment in habitat. Regarding the latter they could adjust the size of their landholding so as to accommodate more or less wildlife (Swanson, 1994; Alexander and Shields, 2003). However, it is usually not easy for landowners to increase land size given the increasing human population, exorbitant land prices and inelastic supply of land (Brown, 2008; Kuhnen, 1998; Foldvary, 2006). Instead, the landowner can improve wildlife habitat by increasing land productivity. In nutrient-poor systems, nutrient additions through fertilization provide a viable way to increase land productivity. Land quality or productivity could also be improved or maintained through stocking density/rate management, fire management, and waterpoints management, among others (Dublin et al., 1990; Arrow et al., 1995; Perrings and Walker,

1997; Meyer and Ausubel, 1999; Prins et al, 2000; Liedloff et al., 2001; Bond and Archibald, 2003; Bond et al., 2003; Brown, 2008).

Private landowners in southern Africa play a crucial role in wildlife conservation. For instance, there are over 5000 game ranches and about 4000 mixed game and livestock ranches in South Africa. Together they cover about 13 % of the country's total land area, compared with 5.8 % for all officially declared conservation areas (Hearne et al., 2000; ABSA, 2003). The landowners of private game ranches and some private nature reserves aim to obtain profits from wildlife through both consumptive and non-consumptive uses (Huffaker, 1993; Hearne et al., 1996; Hearne and McKenzie, 2000; Hearne et al., 2000; ABSA, 2003; de Boer et al., 2007).

The objective of this paper is to develop a generalized framework for exploring the implications of habitat quality enhancement for the welfare of the landowner and conservation of wildlife. Although we have made several simplifications to the model we are convinced that the findings will be useful to wildlife management since scientific theory is often developed from testing simple models in order to better understand complex systems (Rosenzweig, 1973; Noy-Meir, 1975; Starfield, 1997).

We focus on analyzing the direct and indirect effects of having two state and two control variables for decision making of the landowner. We also explore the dependence of vegetation and herbivore biomasses, their own interest rate (Clark and Munro, 1975). We also carry out a simple sensitivity analysis to check the effect of small change in parameter values on the variables.

This paper introduces the concept of investing in habitat quality enhancement to the wildlife bio-economic literature, and analytically considers its implications. We use empirical results from a large scale nutrient supplement experiment carried out in the Associated Private Nature Reserves (APNR), South Africa (van der Waal, unpublished data) to demonstrate the economic consequences of investing in habitat improvement. The soils of the APNR are mostly nutrient poor, limiting plant production and quality, and hence animal production. A viable way to increase the productivity of the system is to supply the nutrients that limit plants, e.g. nitrogen and phosphorous, through fertilization. We focus on African elephants, *Loxodonta africana*, as the representative of the herbivore assemblage. We chose elephants as the major species of the model because of their role as charismatic megafauna species and their implications in terms of engineering biodiversity, through directly or indirectly

controlling the availability of resources to other organisms and creation of habitat heterogeneity (Owen-Smith 1992; Jones *et al.* 1997). Elephants are also highly debated species in many parks in southern Africa and are given prominence in the Convention on International Trade and Endangered Species (CITES) (CITES, 2007).

This paper has five sections. In section 2, we review Swanson's model. In section 3, we introduce the model which endogenizes habitat quality. In section 4, we outline and solve the model numerically. Finally in section 5, we discuss the results implications.

2.2 Model of economic species utilization

Bio-economic modelling for analyzing the economic utilization of species was developed by Clark who analyzed optimization of profit from fisheries (Clark, 1973; Swanson, 1994; Alexander, 2000). Clark (1973) developed a model to analyze the decision-making of a sole owner seeking to maximize the present value of his harvests. Clark showed that there are three factors that may lead to species' extinction as an optimal utilization strategy. These are: high price:cost ratio of harvesting; low natural growth rate; and high opportunity cost of capital (e.g. Clark, 1973; Clark and Munro, 1975; Skonhøft, 1999). However, there are differences between the utilization of marine and terrestrial resources. Importantly, there are generally numerous competing uses of the land by its inhabitants (Skonhøft, 1999). Swanson (1994) adapted Clark's model to terrestrial species by endogenizing habitat quantity.

2.2.1 Swanson's Model

In Swanson's (1994) model, land is regarded as a control variable. For notational convenience, we suppress time notation, but time should be understood to be implicit in all variables. Swanson stipulates the problem of the landowner managing wildlife as follows:

$$\max \int_0^{\infty} [py - c(x)y - r\rho R]e^{-rt} dt \quad (1)$$

Subject to

$$\frac{dx}{dt} = F(x; R) - y \quad (2)$$

Where x is the stock level of species, y is the harvest of the species, p is the price of herbivore offtake, R is the land base and ρ is the unit price of the land resource, $c(x)$ is the

unit cost of harvest as a function of stock, $F(x)$ is the resource growth function, and r is the economic discount rate. The price of herbivore offtake is fixed.

After applying Pontryagin's maximum principle and deriving the necessary conditions associated with optimal stock level, we get the following expression (where λ is the costate variable associated with equation (2)):

$$r = \frac{\lambda F_R(x^*, R)}{\rho} \quad (3)$$

This condition requires that land be allocated to species if they generate a competitive return as compared to another resource. This means that even when Clark's conditions are not met for extinction, a species could still become extinct if it does not provide a competitive return to land (Swanson, 1994; Alexander, 2000).

Another first order condition that results from Swanson's model for the steady state where

$\frac{dx}{dt} = \frac{d\lambda}{dt} = 0$ is the following:

$$r = F_x(x^*; R) - \frac{c'(x^*)F(x^*; R)}{p - c(x^*)} \quad (4)$$

The right hand side of equation (7) can be regarded as the resource's own interest rate (Clark and Munro, 1975). The own interest rate has habitat quantity, land, as a factor which should be considered when making decision to invest in a renewable resource. In Clark's model such resource was assumed to be free and as such did not require investment.

Swanson's model, just like most analytical bio-economic models, treats the carrying capacity per unit of land as an exogenous variable, and land quantity as a control variable. This implies that landowners can increase or decrease the amount of land they have (see also Skonhøft and Armstrong, 2003). The carrying capacity in this case is defined as the maximum biomass of species that can be supported by a given unit of land. Instead, we develop an analytical model where we assume that the land available for wildlife management is fixed so that the only means to increase production for the landowner is to increase the productivity of the land. We

thus relax the assumption inherent in most wildlife bio-economic models that productivity per unit of land for the species is fixed.

2.3 Model

2.3.1 Modified Species Utilization Model

We assume vegetation is homogenous and habitat quality manipulation is represented by increasing land productivity. We also assume that there is one grazing species, and that changes in vegetation biomass are a result of changes in vegetation growth minus herbivore consumption.

The equation describing vegetation dynamics is:

$$\frac{dV}{dt} = g(V, I) - \beta X \quad (5)$$

Where V denotes the vegetation biomass; I stands for the investment in increasing land productivity; X denotes the herbivore biomass; $g(V, I)$ is the growth function of vegetation biomass, which is a function of vegetation biomass, V , and investment, I ; β represents the rate of vegetation depletion due to grazing. For simplicity, we assume that rainfall is constant. We are also assuming a non-interactive grazing system where the herbivore has no influence on vegetation growth other than removal of vegetation biomass through grazing (Caughley, 1976). Due to computational limitations, we have assumed that per animal vegetation-consumption is constant. We also assume that $g_V(V, I) > 0$, $g_I(V, I) > 0$, $g_{VV}(V, I) < 0$ and $g_{II}(V, I) < 0$, conditions that would ensure concavity of the Hamiltonian (see Bulte and van Kooten, 2001). The herbivore dynamics are given by the following population dynamics equation:

$$\frac{dX}{dt} = h(X, V) - qX \quad (6)$$

Where $h(X, V)$ is the herbivore growth function which depends on herbivore and vegetation biomasses; q denotes the rate of herbivore offtake. We assume that $h_X(X, V) > 0$, $h_V(X, V) > 0$, $h_{XX}(X, V) < 0$ and $h_{VV}(V, I) < 0$. We also assume the landowner maximize net benefit given by:

$$Max_{q,I} W = \int_0^{\infty} [pqX + T(X) - c(X)qX - ZI] e^{-rt} dt \quad (7)$$

Where $T(X)$ is the revenue realized from non-consumptive use value of wildlife, e.g., tourism; $c(X)$ is the unit cost function for harvesting herbivores which is a function of the stock level; Z is the unit cost of habitat enhancement; r is the discount rate. We assume that $T'(X) > 0$, $c'(X) < 0$ and $c''(X) > 0$.

The landowner can manipulate the rate of herbivore offtake, q , and investment, I , to maximize net benefit. We assume that an interior solution exists, thus no need to invoke Kuhn-Tucker conditions that are required when there is a corner solution (Conrad and Clark, 1987; Chiang, 1992; Clark, 2005). Maximization takes place subject to equations (5), and (6) so that the current value Hamiltonian for this problem is as follows:

$$H = pqX + T(X) - c(X)qX - ZI + \lambda[g(V, I) - \beta X] + \mu[h(X, V) - qX] \quad (8)$$

Where λ and μ are the current value costate variables for vegetation and herbivore stocks, respectively.

After using Pontryagin's maximum principle (Conrad and Clark, 1987; Clark, 2005) (Appendix 2), we get the following equations for the optimal steady state (assuming that it exists):

$$r = g_V(V^*, I^*) + \frac{[p - c(X^*)]h_V(X^*, V^*)g_I(V^*, I^*)}{Z} \quad (9)$$

$$r = h_X(X^*, V^*) - \frac{c'(X)h(X^*, V^*)}{p - c(X^*)} + \frac{T'(X^*)}{p - c(X^*)} - \frac{Zg(V^*, I^*)}{[p - c(X^*)]g_I(V^*, I^*)} \quad (10)$$

The right hand side of equations (9) and (10) represent the vegetation and herbivore own interest rate respectively. We now analyze the implications of the model.

2.3.2 Implications of the model on wildlife utilization

Swanson's model has one state and two control variables which makes his model easier to control than our model which has two states and two control variables. In Swanson's model, all the controls have direct effects on the state. In our model, there is a choice of coupling the controls, and also the controls have direct and indirect effects on the states (Fig. 2.1). For example, fertilizer (control 1) has direct effect on vegetation (state 1) but indirect effect on herbivores (state 2) through vegetation's effect on herbivores (Fig. 2.1). Herbivore offtake has direct effect on herbivore but indirect effect on vegetation through herbivore's effect (Fig. 2.1).

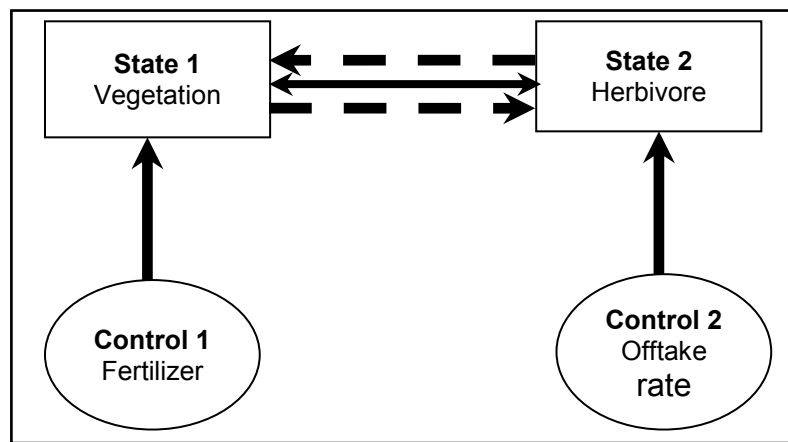


Fig. 2.1: The direct and indirect effects of controls on state variables. Direct effect of control on respective variables is represented by continuous arrows. The indirect effects of controls on state represent by broken arrow. Effects of states on each other represented by double pointed arrow.

The landowner could use both controls at the same time or could choose to use only one. However, we would like to acknowledge that such systems are complex, and there are likely to be multiple steady states or equilibria, each characterised by a different composition of species, and each associated with different properties at equilibrium (Westoby et al., 1989; Perrings and Dalmazzone, 1997; Perrings and Walker, 1997). In such systems, threshold values for different species exist which define limits of the local stability of any equilibrium. If any of these variables are driven beyond such threshold values by the management regime, the system itself will switch from one equilibrium to another (Perrings and Walker, 1997). Resilience in ecology is a measure of the amount of disturbance that an ecosystem can absorb before one equilibrium state is dislodged into that of another equilibrium state (Folke et al., 1996; Perrings and Walker, 1997). It is the capacity of the system to buffer disturbance.

An important aspect of the equilibrium point is its stability properties. If the equilibrium is stable, then the effect of a shock that lowers the steady state resource marginally will not be too serious, because the perturbation of the system will fall within the equilibrium's zone of attraction (Krutilla and Reuveny, 2006). Under these circumstances, the system will evolve back to equilibrium. But if equilibrium is unstable, a shock to the system would cause it to move to another equilibrium or would result in crashing of the system.

In our case we assume that the system is very resilient, such that the local equilibrium cannot be dislodged quickly but can withstand considerable perturbation (Folke et al., 1996; Perrings and Walker, 1997). We also assume that the steady state is stable and the shock introduced is marginal such that it would not dislodge the equilibrium point.

Since our model is linear in q but non-linear in I , the landowner could use controls in this way:

$$q = \begin{cases} 0 & \text{if } X < X^* \\ q^* & \text{if } X = X^* \\ q_{\max} & \text{if } X > X^* \end{cases}$$

$$I = I^{*1}$$

As argued by Clark (2005), this could be an acceptable practical approach because “if you know where you want to be and if many feasible approach paths are available, do not be concerned if the ideal path is not apparent” (Clark, 2005 pp 330).

The vegetation's own interest rate (right hand side of equation (9)) comprises of two terms, namely (i) marginal productivity and (ii) the herbivore net return and marginal productivity on herbivore effect.

The herbivore's own interest rate (right hand side of equation 10) comprises four terms namely: (i) marginal productivity, (ii) marginal stock effect, (iii) marginal revenue from non-consumptive use effect, and (iv) investment cost effect. The herbivore's own interest rate has habitat quality as a factor in all the effects except in the marginal revenue from non-consumptive use effect. The positive terms have an additive effect in the own interest rate whilst the negative terms have the opposite effect except for the marginal stock effect. Since

¹ Maybe $I = 0$ if there would be fixed costs associated with adding fertilizer. But this is not in the current model.

we have assumed that $c'(X) < 0$, this implies that although the marginal stock effect has a negative term – it has an additive effect in the herbivore own interest rate.

2.4 Habitat enhancement through vegetation manipulation: An experiment

Data on how wildlife habitat quality responds to the management tools, such as fire management and waterpoint manipulation is not readily available (e.g. Owen-Smith, 1992; Perrings and Walker, 1997). This renders it difficult to analyze the effects of such tools. However, we use empirical results from a recent large-scale nutrient supplementation experiment we carried out in the Associated Private Nature Reserves, South Africa, to consider the impact of habitat enhancement through vegetation manipulation on elephant biomass. Currently no landowner in the South African wildlife management areas is fertilizing his land. However, this could change in future. Moreover, we use this data because it is a proxy for other management tools available to the landowners for habitat enhancement.

2.4.1 Fertilization Experiment

The vegetation fertilization study was conducted in a mopane (*Colophospermum mopane*) woodland area in the Timbavati Private Nature Reserve in South Africa. The soil in the study area, derived from granite, is nutrient poor (Venter et al., 2003). The long term mean rainfall (1983-2004) is about 450 mm (Ingwalala rainfall station 6 km N). The mean daily temperature ranges between 9.4 °C (month June) to 33.7 °C (month January; Venter *et al.* 2003).

A field experiment was conducted where 10 m x 10 m patches were fertilized at increasing rates. Twelve, 10 x 10 m plots were marked in a 0.5 km by 1 km area and one of four fertilizer treatments were randomly allocated per plot. A commercial NPK fertilizer (3 N: 2 P: 1 K) was used and fertilizer was spread by hand in January 2006 and again in December 2007. The treatments consisted of the following fertilizer rates and are expressed as Kg N/ha: 0 (control), 12, 60 and 300 Kg N/ha, respectively. All plots were selected in grassy, open area between trees. The mean of six readings inside and six readings outside were converted to dry biomass using the calibration curve of Zambatis et al. (2006) developed for the Kruger National Park.

A regression analysis was performed on the results and the vegetation response to fertilizer was obtained (Fig. 2.1). For simplicity, we selected a quadratic model (see also France and Thornley, 1984; Belanger et al., 2000). We then used these results to model the response of

the vegetation to fertilization. The vegetation response fertilizer model that we obtained (Fig. 2.2) is similar to other model results conducted by other studies which show that vegetation biomass increases with fertilizer application upto a certain level, afterwhich the biomass begins to decrease with excess fertilization (e.g. France and Thornley, 1984; Belanger et al., 2000; Donaldson et al, 1984). The optimum range for fertilizer application is comparable to the range obtained by these studies. Although it could be far-fetched, the effects of fertilization could be comparable to effects of fire another tool used by wildlife managers. Some grasses and grasslands are said to thrive on fire and suffer if left long unburned. However, there is an optimum fire frequency range, below and beyond which biodiversity could be negatively affected (Liedloff et al, 2001; Zelder, 2007).

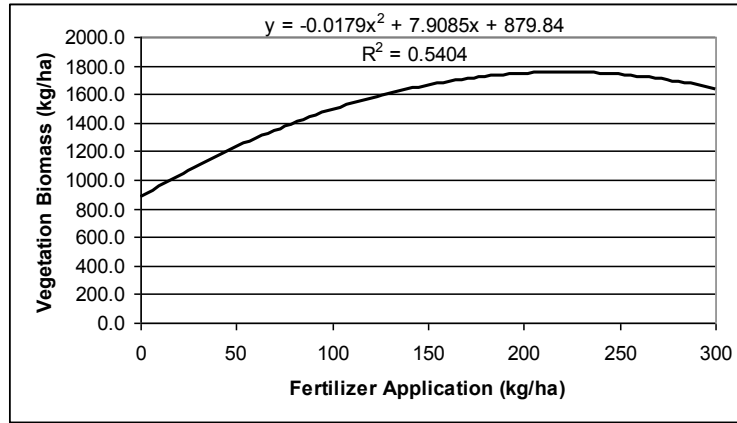


Fig. 2.2: Regression of fertilizer and vegetation response data from the vegetation response to fertilizer experiment

2.4.2 Vegetation response to fertilization model

Because of the complexity of the problem, we ignored non-consumptive use of herbivores and cost related harvest costs. We thus specify some of the components for equations (8) to

(10) as: $c(X) = T(X) = 0$; $g(V, I) = \eta V \left(1 - \frac{V}{K + I(a - bI)} \right)$; and $h(X, V) = \theta X \left(1 - \frac{\beta X}{V} \right)$.

By having $c(X) = T(X) = 0$, the effect would be: shifting up of the vegetation own interest rate (equation (9)) plane; decrease in the investment cost effect component of the herbivore own interest rate, disappearance of the two components of herbivore own interest rates namely marginal stock effect and marginal revenue from non-consumptive use effect; shifting down of the herbivore own interest rate (equation (10)) plane if non-consumptive use is dominant.

As evident from the specification we are assuming that both the vegetation and herbivore growth rate are logistic for simplicity reasons as used by many economists (e.g. Skonhøft, 1995; Perrings and Walker, 1997; Bulte and van Kooten, 2001; Fleming and Alexander, 2003; Clark, 2005). The economic and ecological analyses of technology investment on vegetation improvement are based on the assumption that there is a functional relationship between the vegetation yield and level of investment. This relationship is not known precisely and theories of yield response to technology are not well developed especially in the wildlife research field. Generally, the relationship between yield and technology obeys the law of diminishing returns, and can be described by many functions (Colwell, 1983; Ronald, 1970). For example, we expect that when you add fertilizer, the plant response at first will be positive, after which an optimum will be probably be reached, but adding more fertilizer after a certain point will result in negative vegetation responses. Vegetation generally response to fertilization conforms to the principle of diminishing marginal product (French, 1956; Overman et al, 1993; Overman and Scholtz, 2004). We assume that the land is responsive to vegetation improvement and all the land in the model responds the same to the intervention. The technology will affect both the vegetation growth rate and the maximum vegetation biomass that can be supported per unit area. We are assuming that the technology has no influence on the vegetation intrinsic growth rate but only on the carrying capacity.

The effects of fertilizer technology on vegetation improvement is captured in the term, $K + I(a - bI)$. When the investment rate is zero, the carrying capacity equals the normal K . When the landowner starts investing in a technology to improve the vegetation, it result in some increment to the total carrying capacity as long as investment rate is less than the investment rate that gives the maximum additional potential vegetation response. However, when $I > a$, the technology will result in the reduction of the carrying capacity. At this point the technology would be excessive, and instead of resulting in improvement of vegetation, the destruction of vegetation will occur. We use data from the fertilizer experiment described in the earlier subsection to parameterize the effects of the fertilizer technology. To ensure concavity of the Hamiltonian function, we consider the condition that $g_V(V, I) > 0$, $g_I(V, I) > 0$, $g_{VV}(V, I) < 0$, and $g_{II}(V, I) < 0$ which implies considering stock levels on the concave segment of the growth curve (Bulte and van Kooten, 2001). We also assume that the logistic growth function of herbivores is limited by the growth the availability of vegetation (e.g. Perrings and Walker, 1997).

Due to the complexity of the problem, we solve the model numerically. In addition, because of computational limitations, we set the economic discount rate that should be equalized with the biomass' own interest rate to 0. However, we deduce from the model, different optimum rate of returns from different combinations of optimum vegetation and herbivore biomass. The parameters and their definitions are shown in Table 2.1.

The Jacobian matrix eigenvalues for the system at steady state are: 0.221, -0.171, 0.111, and -0.061. Since two of the eigenvalues are negative, an equilibrium could be achieved, i.e. for any initial state, $[V_0, X_0]$ sufficiently close to the steady state, it is possible to determine the initial values of the co-state variables $[\lambda_0, \mu_0]$, so that the control path leads to the equilibrium point (Tino et al., 2001; Antoci et al., 2005). This is an attractive fixed point for the natural dynamics since the Jacobian matrix of system has two eigenvalues with strictly negative real parts (Antoci et al., 2005) which implies that the fixed point can be reached along the optimum control trajectory. However, as pointed out in the analytical part, for such complex systems following the bang-bang approach could be an acceptable practical approach. (see Section 2.3.2).

The optimum values for the control variables of the model, that is, investment, i and herbivore offtake, q are shown in Fig. 2.3. The peak of the plot gives the optimal profit from which the level of investment and herbivore offtake could then be deduced. We use these optimum values to express the vegetation's and herbivore's own interest rates in terms of vegetation and herbivore biomass. With this information we plot the vegetation's and herbivore own interest rate plane (Fig. 2.4).

As shown in Fig. 2.4: the vegetation own interest rate plane is steeper than the herbivore own interest rate plane due to the vegetation higher growth rate. The vegetation own interest rate is negatively related to vegetation biomass but positively related to herbivore biomass. This means that marginally increasing the vegetation biomass from the optimum value would result in a decrease in vegetation's rate of return whilst marginally increasing herbivore biomass would result in an increase in vegetation's rate of return. On the other hand, the herbivore's own interest rate is negatively related to herbivore biomass but positively related to vegetation biomass. This shows that there is a negative effect of own interest rate on its own biomass. This also shows the effect of higher interest on reducing resource stocks (Clark, 1973; Alexander, 2000).

Table 2.1: Parameter used, their meaning, values and their source

| <i>Parameter</i> | <i>Meaning</i> | <i>Value</i> | <i>Source</i> |
|------------------|---|--------------------|---|
| K | Y intercept | 879.84kg/ha | Tembo fertilizer experiments |
| a | Fertilizer response parameter 1 | 7.9085 | Tembo fertilizer experiments |
| b | Fertilizer response parameter 2 | 0.0179 ha/kg | Tembo fertilizer experiments |
| η | Vegetation intrinsic growth rate | 0.8 | Huffaker , 1993 |
| β | Rate of vegetation depletion due to grazing | 1.2 (Elephant) | Huffaker, 1993; Prins, 1993; Bothma, 2002 |
| θ | Herbivore growth rate | 0.07 (Elephant) | Calef, (1988); Bothma et al., 2002; Jolles, 2007; ABSA, 2003 |
| p | Price of herbivore offtake per unit biomass | USD8/kg (Elephant) | Thormahlen and Cochran Safaris, 2008; Bothma, 2002 |
| Z | Cost of fertilizer | USD0.2/kg | http://www.fssa.org.za/pebble.asp?reli_d=64 |
| r | Discount rate | 0 | |

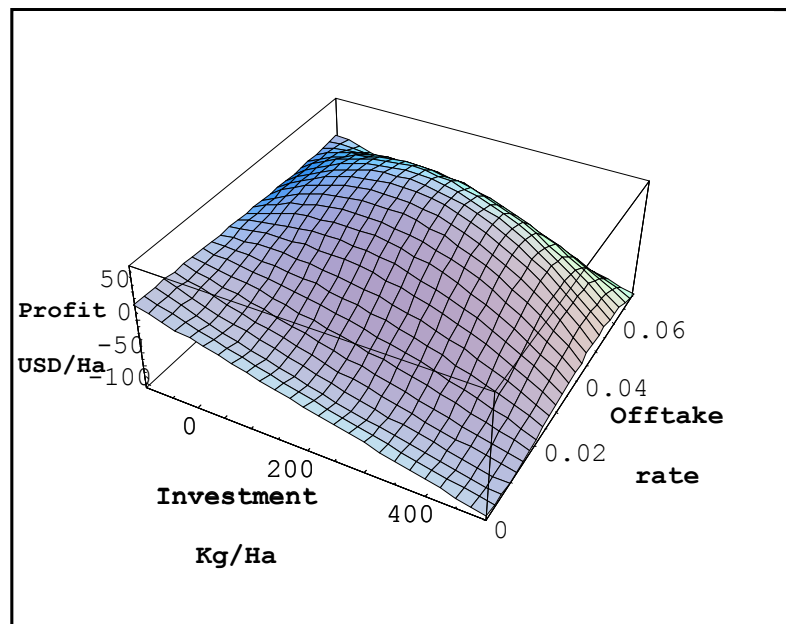


Fig. 2.3: Optimum investment and herbivore offtake rate that give the landowner optimum profit. The optimum profit lies on the peak of the plot.

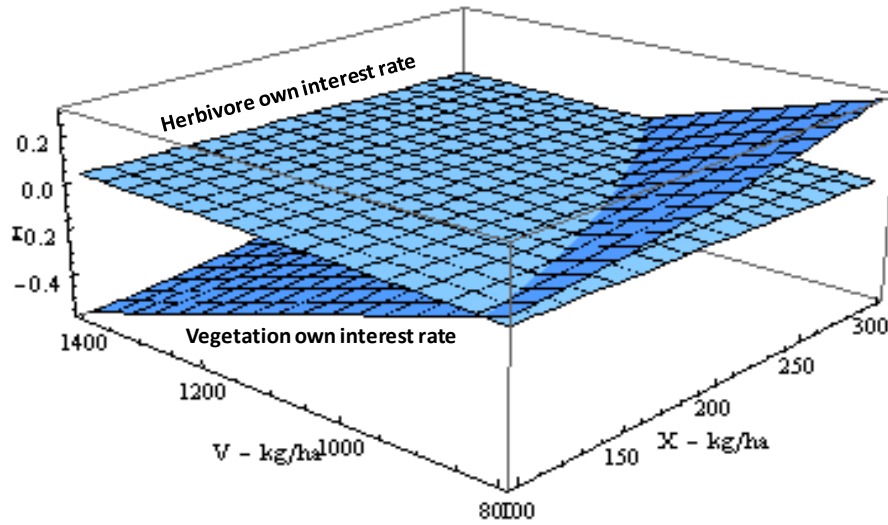


Fig. 2.4: Intersection of vegetation and herbivore own interest rate. The various rates of return from optimum combinations of vegetation and herbivore biomass assuming fixed investment and herbivore offtake is defined by the movement along the line of intersection.

The various rates of return from optimum combination vegetation and herbivore biomass assuming fixed investment and herbivore offtake is defined by the movement along the line of intersection. It is along this intersection line when the rate of return is equal zero that biomass in the model is maximized since it is also where the economic discount rate is zero. By subtracting the herbivore interest rate plane equation from vegetation interest rate plane – we get a plane of different combinations of optimum vegetation and herbivore biomass and the resultant rate of return when investment and offtake rate are fixed (Fig. 2.5). We shall call the resultant plane, Optimum Vegetation and Herbivore Biomass Rate of Return (OVHBROR) plane. The line of intersection for the vegetation and herbivore own interest rate in Fig. 2.5 lies along this plane.

The OVHBROR plane has a negative relationship with vegetation biomass but has a positive relationship with herbivore biomass. This means that marginally increasing optimum vegetation biomass would result in a drop in the rate of return whilst marginal increase in herbivore biomass would have an opposite effect. The following parameters would result in shifting up of the OVHBROR plane when they are increased: price of herbivore offtake, y intercept, fertilizer response parameter 1, and herbivore growth rate; whilst an increase in fertilizer response parameter 2 and cost of fertilizer would result in shifting down of the plane.

However, increase in the vegetation intrinsic growth rate results in increase in the slope of the OVHBROR plane on the vegetation biomass side. On the other hand, a decrease in the vegetation depletion due to grazing parameter results in decrease in the slope of the OVHBROR plane on the herbivore biomass – it becomes less positive.

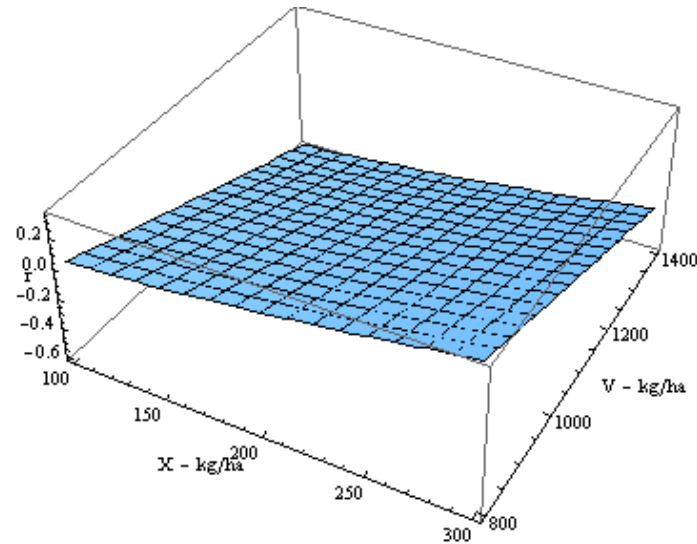


Fig. 2.5: Plane of different combinations of optimum vegetation and herbivore biomass and the resultant rate of return when investment and offtake rate are fixed

We now look at the optimum values for the vegetation and herbivore biomass including the optimum control variables, investment to enhance land productivity and herbivore offtake rate. Where there is investment in fertilization, all the variables are higher than in scenarios without investment (Table 2.2). For the elephants scenario, the optimum vegetation biomass increased by 82% due to fertilization (Table 2.2), which is similar to the herbivore increase. The optimum fertilizer application rate is 130kg/ha (Table 2.2). This happens to be within the range of optimal fertilizer rates recorded in some agricultural studies (Belanger, et al., 2000). The profit of the landowner increases by 33% (Table 2.2).

Table 2.2: Effects of improvement of vegetation through fertilization on elephant

| <i>Variable</i> | <i>Without fertilizer</i> | <i>With fertilizer (increase)</i> |
|-----------------------------|---------------------------|-----------------------------------|
| Vegetation Biomass Kg/ha | 556 | 1013 (82%) |
| Herbivore Biomass Kg/ha | 136 | 248 (82%) |
| Investment fertilizer Kg/ha | 0 | 130 |
| Offtake rate | 0.05 | 0.05 |
| Profit USD/ha | 54 | 72 (33%) |

In our model, herbivore density translates in the scenario without fertilizer to about 2.3 elephants per km², whereas with fertilizer it rises to 4.1 elephants per km² assuming elephants body mass of 6000 kg (Macdonald, 2001). These might seem to be extremely high densities given that in most national parks in South Africa the density has been around or below 0.4 elephants per km² (van Aarde, 1999; Slotow et al., 2005; Gough and Kerley, 2006). However, these results must be seen in the context of the simplistic assumptions of only one herbivore species and one vegetation type. It must also be noted that there are some areas with comparatively high elephant density. The elephants at Addo Elephant National Park, South Africa have been stocked at much higher levels, upto 4 elephants per km² during the park's history but stand currently at 2.4 elephants per km² (Gough and Kerley, 2006). In Chobe National Park, Botswana, density was 2.9 elephants per km² in 2001 (Allen, 2006). In addition, a model used by Ben-Shahar (1996) explored the number of elephants which could be sustained in mopane woodlands and indicated that the woodlands could support 3-9 elephants per km². Other studies and experiences elsewhere in Africa also show densities of more than 4 elephants per km² (Prins and Douglas-Hamilton, 1990; Kerley et al., 2008). Overall, therefore the results are not unrealistic and support the idea that investing in land quality via fertilization may pay off. Nevertheless the value of the numbers lies more in the comparative statics than in the point estimates. In other words, the policy implications of our model results should be taken more from a qualitative viewpoint than a quantitative one.

2.4.3 Sensitivity Analysis

The relationships between the model parameters and variables are shown in Fig. 2.6. The herbivore offtake rate is constant for most of the changes in parameters except for vegetation and herbivore growth rates. Vegetation biomass, herbivore biomass and investment fertilizer have a positive relationship with the parameters: fertilizer response parameter 1, vegetation intrinsic growth rate, y-intercept and price of herbivore offtake; whilst they have a negative relationship with herbivore rate of vegetation depletion, fertilizer response parameter 2 and cost of fertilizer (Fig. 2.6). Specifically, the relationship between elephants' price of offtake and its biomass is positive and it saturates around 250 kg/ha of biomass (Fig. 2.6a). It is positive in the numerical model because of the increase in marginal effect of investment on vegetation is dominating as suggested in the analytical model. The relationship between elephants' price of offtake and optimum fertilizer application is also positive (Fig. 2.6). On the other hand the relationship between fertilizer's cost and its application rate is negative (Fig. 2.6b). The relationship between fertilizer's cost and its herbivore biomass is also negative (Fig. 2.6b).

To check sensitivity of the model, values of parameters are changed positively and negatively by 50% and their effects on variables are analyzed (Table 2.3). It is observed that vegetation biomass is more sensitive to change in the fertilizer response parameter 1, herbivore biomass is more sensitive to drop in herbivore growth rate, profit is more sensitive to drop in the rate of vegetation depletion due to grazing, whilst investment is sensitive to a drop in fertilizer response parameter 2 (Table 2.3). On the other hand, vegetation biomass and herbivore biomass are less sensitive to increase in herbivore growth rate and increase in price of herbivore offtake investment is less sensitive to the change in y-intercept, whilst profit is less sensitive to fertilizer response parameter 2.

Table 2.3: The effects of 50% increase and 50% decrease in parameter values on variables

| <i>Parameter</i> | Percentage change in variable due to change in parameter by | | | | | | | |
|------------------|---|-----------|------------|--------|--------------|-----------|------------|--------|
| | 50% increase | | | | 50% decrease | | | |
| | Vegetation | Herbivore | Investment | Profit | Vegetation | Herbivore | Investment | Profit |
| K | 27 | 27 | 0 | 37 | -27 | -27 | 0 | -37 |
| a | 68 | 68 | 85 | 62 | -41 | -41 | -85 | -25 |
| η | 14 | 40 | 13 | 32 | -25 | -55 | -43 | -49 |
| β | -12 | -41 | -35 | -43 | 7 | 114 | 35 | 142 |
| θ | 5 | 5 | 23 | 70 | -28 | -28 | -70 | -62 |
| b | -15 | -15 | -33 | -8 | 45 | 45 | 100 | 25 |
| p | 5 | 5 | 23 | 70 | -28 | -28 | -70 | -62 |
| z | -12 | -12 | -35 | -15 | 7 | 7 | 35 | 21 |

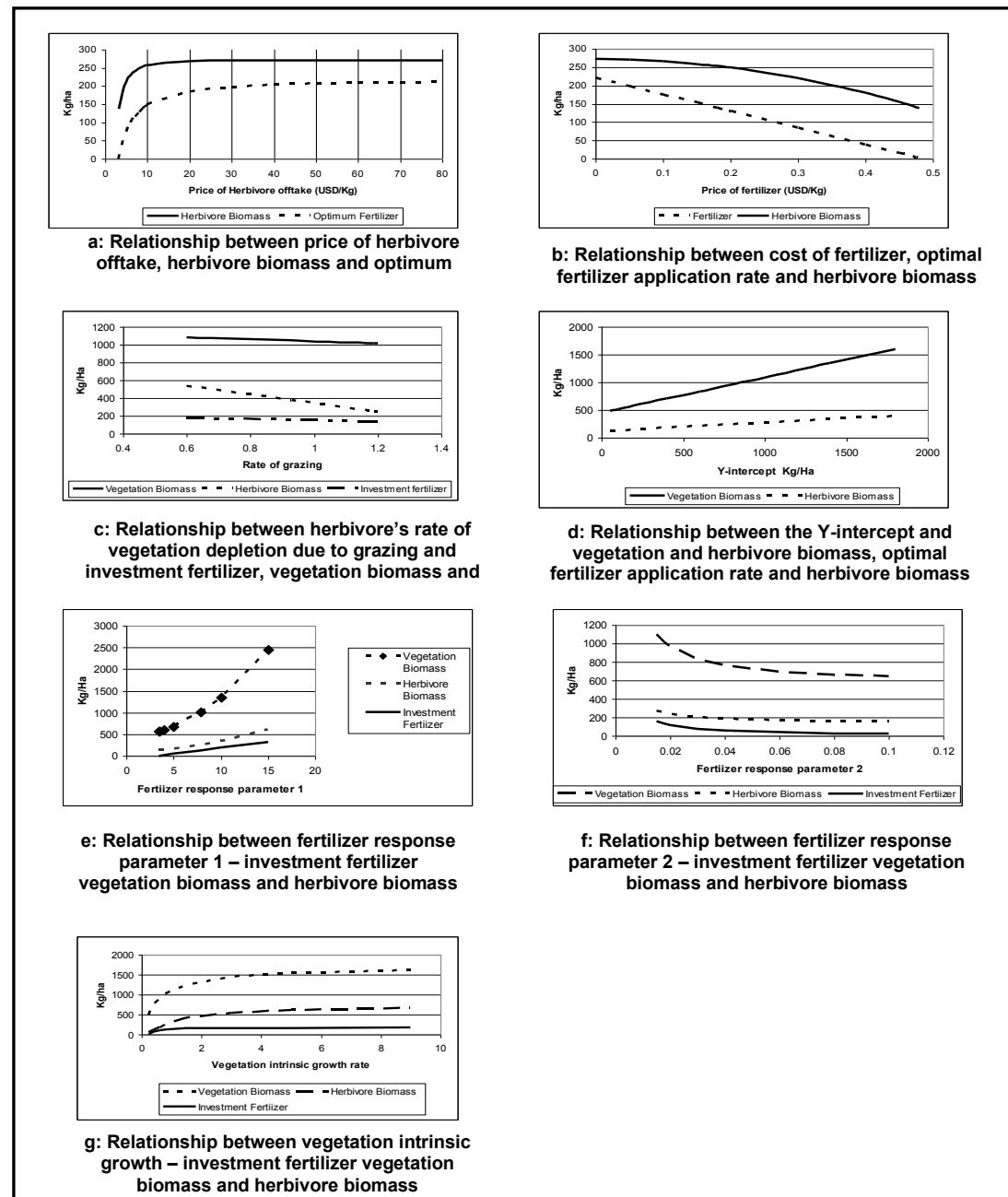


Fig. 2.6: The effect of changing model parameters on the variables

2.5 Discussion

We develop a mathematical model for wildlife utilization which introduces habitat quality improvement. From our theoretical model framework, there are some parallels between our “endogenous habitat quality model” and that of Swanson (1994). If the landowner finds wildlife attractive he acquires more land to accommodate more wildlife in Swanson's case. In our case he will enhance land productivity to accommodate more wildlife on the land. Our case is more applicable to cases where available land is fixed, reflecting the difficulty of increasing one's land size due to increasing human population, a shift from one stock variable

to two stock variables (more difficult to analyze). By having two interacting stocks depending on each other for survival, the decision rules for the landowner are also modified. What happens to one state variable affects the other. Choices on controlling the state variables are increased. There are some direct and indirect effects of the control variables on the state variables. This brings some level of realism to analytical modelling, since in real life a decision maker has to deal with several state variables given numerous control variables that affect the state directly or indirectly.

We have assumed that the economic discount rate that is equalized with the biomass' own interest rate to be zero. In the real world the discount is usually positive. In that case it would not be economic to invest in wildlife habitat investment, let alone wildlife management if the landowner is obtaining returns that are much lower than the economic discount rate. Nonetheless, our model shows that investing in wildlife eases off the trend to local extinction of large herbivores such as elephants (Swanson, 1994; Alexander, 2000). In the analytical model, we discussed about the marginal revenue from non-consumptive use. So as long this is very high it would have a significant additive value to the herbivore own interest rate thereby contributing to increasing the attractiveness of investment on wildlife management by landowners (Alexander, 2000).

We developed the optimum vegetation and herbivore biomass rate of return (OVHBROR) plane by subtracting the herbivore own interest from vegetation own interest rate equation. This plane is useful in analysing the optimum combinations of vegetation and herbivore biomasses and their respective rate of return assuming fixed controls. The OVBHROR plane can give an indication on the maximum rate of return that could be achieved from the given state variables given their controls which are fixed in the plane. This is useful given the limitation that is poised by the current computational power in solving optimal control problems. In addition there are some parallels that could be drawn from the results obtained from sensitivity analysis of both the OVBHROR and full model results. For instance, all the parameters that have been associated with increasing – that is shifting up the OVHBROR plane or increasing the slope are also associated with positive relationships with the variables. This also applies for the opposite situation.

A decrease in price of herbivore offtake resulted in a decrease in levels of investment by landowners which in turn lead to a decrease in vegetation and herbivore biomass. However, an increase in price of herbivore offtake has the opposite effect. So in this case, utilization of wildlife could generate income that could be ploughed back into wildlife conservation

through habitat enhancement. Outright banning of wildlife utilization could result in wildlife being sidelined by landowners in the long run. However, we acknowledge that this will only work in an environment where the property rights are well defined (Skonhofs and Solstad, 1998). An example is in South Africa where private landowners are gainfully engaged in wildlife utilization from both wildlife's consumptive and non-consumptive uses. ABSA estimated that tourism and hunting could together generate more than 3 million Rand per annum for an average sized ranch in South Africa (ABSA, 2003).

Given some of the simplifications that we have made in the model, the results should not be taken from a quantitative viewpoint but from a qualitative one. Although the model has shown that it is profitable to fertilize wildlife management areas, there are logistical challenges when applying fertilizer to a natural ecosystem mainly due to unknown spatial application costs in terms of labour costs and other associated costs. We tried to factor this into the model by doubling the price of fertilizer per kilogram. However, there are some spatial challenges that may confront the application process. Future work could analyze spatial issues in application of fertilizer in more detail.

Acknowledgments:

We are grateful to the Associate Private Nature Reserve (APNR) for hosting the fertilizer experiment. This study is part of an integrated programme, The Elephant Movements and Bio-economic Optimality programme (TEMBO), financially supported by the Netherlands Foundation for the Advancement of Tropical Research (Wotro), the Dr Marie Luttig trust, and Shell South Africa. We thank Erwin Bulte for his critical comments and fruitful discussions in development of this paper. We also thank Geerten Hengeveld and Enoch M. Kikulwe for productive discussion on the model.

Appendix 2: Derivations of Equation (9) and Equation (10)

The landowner's objective

$$Max_{q,I} W = \int_0^{\infty} [pqX + T(X) - c(X)Xq - ZI] e^{-rt} dt \quad (A1)$$

Subject to

$$\frac{dV}{dt} = g(V, I) - \beta X \quad (A2)$$

$$\frac{dX}{dt} = h(X, V) - qX \quad (A3)$$

The current value Hamiltonian:

$$H = pqX + T(X) - c(X)qX - ZI + \lambda[g(V, I) - \beta X] + \mu[h(X, V) - qX] \quad (A4)$$

The Pontryagin's necessary conditions for a maximum are:

$$\frac{dH}{dI} = -Z + \lambda g_I(V, I) = 0 \quad (A5)$$

$$\frac{dH}{dq} = pX - \mu X - Xc(X) = 0 \quad (A6)$$

$$\frac{dH}{dV} = \mu h_V(X, V) + \lambda g_V(V, I) \quad (A7)$$

$$-\frac{dH}{dV} = \dot{\lambda} - r\lambda \Rightarrow \dot{\lambda} = r\lambda - \frac{dH}{dV} \quad (A8)$$

$$\frac{dH}{dX} = pq - \beta\lambda - qc(X) - qXc'(X) + T'(X) + \mu[-q + h_X(X, V)] \quad (A9)$$

$$-\frac{dH}{dX} = \dot{\mu} - r\mu \Rightarrow \dot{\mu} = r\mu - \frac{dH}{dX} \quad (A10)$$

$$\frac{dH}{d\lambda} = -X\beta + g(V, I) \Rightarrow \beta = \frac{g(V, I)}{X} \quad (A11)$$

$$\frac{dH}{d\mu} = -qX + h(X, V) \Rightarrow q = \frac{h(X, V)}{X} \quad (A12)$$

Solve equations (A5) and (A6) for λ and μ , respectively

$$\lambda = \frac{Z}{g_I(V, I)} \quad (A13)$$

$$\mu = p - c(X^*) \quad (A14)$$

Take d/dt of equations (A13) and (A14)

$$\dot{\lambda} = \frac{d}{dt} \left[\frac{Z}{g_I(V^*, I^*)} \right] \quad (A15)$$

$$\dot{\mu} = \frac{d}{dt} [p - c(X^*)] \quad (A16)$$

Substituting equations (A13) and (A14) into equations (A8) and (A10)

$$\dot{\lambda} = \frac{rZ}{g_I} - \left\{ [p - c(X)]h_V(X, V) + \frac{Zg_V(V, I)}{g_I} \right\} \quad (A17)$$

$$\dot{\mu} = r[p - c(X)] - \left\{ pq - \frac{\beta Z}{g_I(V, I)} - qc(X) - qXc'(X) + T'(X) + [p - c(X)][-q + h_X(X, V)] \right\} \quad (A18)$$

We assume that an equilibrium exists so that all conditions are simultaneously met. At equilibrium $\dot{\lambda}$ and $\dot{\mu}$ are equal to zero. This makes equations (A15) to (A18) equal to zero. Given these conditions and substituting for q as shown in equation (A12) and β as in equation (11) into equation (A18), we solve for r for both equations (A17) and (A18).

$$r = g_V(V^*, I^*) + \frac{[p - c(X^*)]h_V(X^*, V^*)g_I(V^*, I^*)}{Z} \quad (A19)$$

$$r = h_X(X^*, V^*) - \frac{c'(X)h(X^*, V^*)}{p - c(X^*)} + \frac{T'(X^*)}{p - c(X^*)} - \frac{Zg(V^*, I^*)}{[p - c(X^*)]g_I(V^*, I^*)} \quad (A20)$$

Optimization of wildlife management through waterpoints manipulation: a bio-economic analysis

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Abstract

Surface water is one of the constraining resources for herbivore populations in semi-arid regions. Artificial waterpoints are constructed by wildlife managers to supplement natural water supplies, to support herbivore populations. The aim of this paper is to analyze how a landowner may realize his ecological and economic goals by manipulating waterpoints. We develop a theoretical bio-economic framework to analyze the optimization of wildlife management objectives (revenue generation from both consumptive and non-consumptive use and biodiversity conservation), using waterpoint construction as a control variable. The model provides an ecological economic framework for analyzing optimization problems where a control has direct effect on one herbivore but indirect effects on the other. A landowner may be interested only in maximization of profits or personal benefits either from elephant offtake and/or tourism revenue, thus might ignore the negative effects that could be brought about by elephants to biodiversity. If the landowner is not taking into consideration the indirect effects of tools such as waterpoints, then game reserve management as an authority entrusted with sustainable management of the game reserve on behalf of all the landowners might use economic instruments such as subsidies or payments to the landowners who are complying with sound waterpoints management and/or taxes or charges to those who are not complying so as to enforce sound waterpoints management.

Keywords: landowners; southern Africa; analytical modelling; elephant; water-dependent; water-independent; biodiversity; externalities

3.1 Introduction

In South Africa the private landowners play a crucial role in wildlife conservation. These landowners often aim to conserve wildlife through sustainable utilization of both wildlife

consumptive and non-consumptive uses (Jordie and Peddie, 1988; Pilanesburg, 2000; Mafikeng, 2001; APNR, 2005). Some private landowners have formed game reserves and employ management teams to run the reserve and ensure that sustainable wildlife management practices are practiced by the landowners (APNR, 2005). Surface water provision, fire management, fencing and animal population manipulation by culling or translocation of animals, are some of the most common intervention tools used by landowners in wildlife protected areas to achieve their objectives (Perrings and Walker, 1997; Slotow et al., 2005; de Boer et al., 2007; Grant et al., 2008). Surface water is one of the main constraining resources for herbivore populations in semi-arid regions (Western, 1975; Redfern et al., 2003). Artificial waterpoints are therefore constructed by game managers to supplement natural water supplies, to support the existing populations and to distribute evenly the impact of herbivores on vegetation (Owen-Smith, 1996; Grant et al., 2008).

It has been shown that properties with the big five, consisting of elephant, buffalo, rhino, lion and leopard, generally attract a high number of tourists (Lindsey *et al.*, 2007; Okello et al., 2008). Investment in the establishment of waterpoints is expected to increase if their creation would increase the number and visibility of such animals as elephant and buffalo which belong to the big five (Joubert, 1986; Mabunda et al., 2003; Chamaillé-Jammes et al., 2007; Smit et al., 2007a). This led to the belief that a more extensive network of waterpoints would increase the revenue generated through tourism (Joubert, 1986; Parker and Witkowski, 1999; Mabunda et al., 2003). For example waterpoints in the Klaserie Private Nature Reserve, adjoining Kruger National Park (KNP) increased from only six waterpoints in 1965, to 144 by 1980 (Witkowski, 1983; Parker and Witkowski, 1999). Although waterpoints could be beneficial to tourists and wildlife viewing, they may compromise other conservation objectives, such as biodiversity conservation (Harrington et al., 1999). Additional waterpoints could result in increased number of herbivores which could lead to increase in revenues in the short term, but increased number of herbivores could later adversely affect the vegetation (structure, species and biomass), and thereby change the animal species diversity in the long term (Walker, et al., 1987; Thrash et al., 1991; Thrash, 1998a; Thrash and Derry, 1999; Baxter, 2003), which could lead to decreased revenues from both animal offtake and tourism.

The potential increase in animal impact with increasing numbers of artificial waterpoints is an issue of concern. A certain density of waterpoints could be beneficial to the animals in a conservation area. Too few waterpoints could result in severe water shortages for populations, hence increased animal mortality – this may closer approximate the densities that can be maintained in a resource limited semi-arid environment, that is, a stabilisation of animal

densities at lower levels. On the other hand, too many waterpoints might result in increased environmental costs, such as disastrous effects on biodiversity. Too many waterpoints could also result in widespread large impacts on vegetation, an increase in predation and/or an increase in the population of water-dependent species at the expense of water-independent species – and overutilisation of the veld resulting in homogenisation of vegetation composition and structure (Smuts, 1978; Owen Smith, 1996; Harrington et al, 1999; Thrash, 1998b). It is argued that biodiversity is higher in a heterogeneous landscape than a homogenous one (Walker et al., 1987; Thrash et al., 1991; Thrash et al., 1995; Thrash, 1998a; Thrash, 1998b; Baxter, 2003). Elephant, zebra, buffalo, wildebeest and waterbuck are herbivore species considered to be water-dependent and thereby respond favourably to an increase in waterpoints (Collinson, 1983; Redfern et al, 2003; Redfern et al, 2005). On the other hand, some species such as tsessebe, roan, impala, kudu, giraffe, and warthog can tolerate limited water consumption and may survive for long periods without access to surface water during the dry season (Martin, 1983; Smithers, 1983; Estes, 1991).

Roan, sable and other antelope species are sensitive to habitat changes and have critical habitat requirements, as they depend on tall grasses (Martin, 1983). So the physiognomic changes to vegetation structure brought about by bulk feeders could result in a decrease of these water-independent species (Martin, 1983). For instance, in Kruger National Park a severe drop in the roan population was observed between 1986 and 1993 from about 450 to about 45 animals (Harrington et al, 1999; Grant et al, 2002). Some studies have claimed the cause of this decline to be the provision of numerous artificial waterpoints in the roan range, which attracted the heavy grazers such as zebra and wildebeest, particularly during drought conditions (Harrington et al, 1999; Grant et al, 2002).

There are no economic studies addressing the issue of waterpoints from an ecological economic point of view. The intention of this paper is therefore to address these issues. We aim to analyze how wildlife managers may achieve their objectives of generating returns from wildlife by manipulating waterpoint densities, whilst at the same time contributing to sustainable wildlife conservation. Although we have made several simplifications to the model, the findings are useful to wildlife management since scientific theory is often developed from testing simple models in order to better understand complex systems (Rosenzweig, 1973; Noy-Meir, 1975; Starfield, 1997). We develop a theoretical bio-economic framework to analyze the optimization of wildlife management objectives using waterpoint manipulation and herbivore offtake as control variables. The underlying assumption is that surface water availability can be manipulated through provision of artificial waterpoints at

relevant scale to influence wildlife populations (Redfern, 1995; Redfern et al, 2005). For model transparency, we consider a single species model with and without environmental costs, and a two species model – with elephants representing water-dependent species and roan representing water-independent species. We regard the occurrence of the second species as a proxy for biodiversity, which can be justified from the Harrington et al. (1999) study.

3.2 The model

We firstly consider a case where an increase in elephant numbers does not have any negative impact on other herbivore species. However, in the second case, we will consider a situation where the increase in the number of elephants has a negative impact on biodiversity.

3.2.1 The single species model

In the single species model, we assume two economic activities *viz.* tourism and hunting. Although tourism and hunting are considered mutually exclusive especially at a very small scale, we are assuming that the scale is large enough to allow the two activities to occur at different places or time. We use elephants as the representative herbivore species for the reasons that we will articulate later. For notational convenience, we suppress the time notation, but time should be understood to be implicit in all variables. The population dynamics of the elephant is given by the following equation:

$$\frac{dX}{dt} = h(X, W) - qX \quad (1)$$

Where: $h(X, W)$ is the growth function of elephant which depends on its stock (X) and number of waterpoints, W (Chamaillé-Jammes et al., 2007), q is the rate of elephant offtake. We assume that the distribution of perennial water sources within the ecosystem is constant over time and that surface water availability can be manipulated at scales relevant to elephant management (Western, 1975; Redfern et al, 2005). We adopt the notation: the superscript denotes derivative with respect to the variable shown in the superscript. We also assume that $h^X(X, W) > \text{or} < 0$, $h^{XX}(X, W) < 0$ and $h^W(X, W) > 0$. In this case we assume that the index of welfare (Π) is a measure of the landowner's net benefit. The landowner can manipulate the rate of elephant offtake through hunting, q , and manipulating the number of waterpoints W , influencing his net benefit. The profit function for the landowner is given by:

$$\Pi = pqX + TX - ZW \quad (2)$$

Where pqX is the revenue generated from selling hunting licences. We assume that p , the price of a hunting licence, is fixed. TX is the revenue realized from non-consumptive use of elephants. We have assumed a linear relationship between revenues from non-consumptive use of elephants and population sizes of the species to simplify the problem. Z is the cost of maintaining a waterpoint.

The above specification of the model would suffice in the case where elephants do not have any effect on biodiversity. In this case, the landowner would be interested only in maximizing revenue through tourism and hunting. There are several proxies used for measuring biodiversity, in the one species model we adopt species richness – the total number of species present (Jarvinen and Vaisanen, 1978; Magurran, 1988) but in the two species model we let roan, the water-independent species represent biodiversity.

Elephants are regarded as a charismatic megafauna species because of their size, apparent intelligence, social structure and they are ecosystem engineers or architects because of their size through direct or indirect impact on the availability of resources to other organisms and creation of habitat heterogeneity and possibly homogeneity under certain conditions (Owen-Smith 1992; Jones et al. 1997; Kerley et al., 2008; Valeix et al., 2008). Elephants can have noticeable effects on vegetation composition and influence the balance between trees, shrubs and herbaceous layers in plant communities (Holdo, 2003; Holdo, 2006; Roux, 2006; Fornara and du Toit, 2007; Holdo, 2007; Fornara and du Toit, 2008; Kerley et al., 2008; Levick and Rogers, 2008; Valeix et al., 2008). We assume that there is a trade-off between elephants and biodiversity. Elephants at intermediate density levels might promote biodiversity (Cochrane, 2003), as suggested by the correlation between increase in the elephant population, correlated to an increase in some populations for which medium-term facilitation has been suggested, as elephants may generate more browse resources through coppicing (Skarpe et al., 2000; Baxter, 2003; Makhabu et al, 2006; Fornara and du Toit, 2007 ; de Knecht, 2008). However, extremely high density of elephants may also have negative effects on biodiversity (Tchamba, 1995, Trollope et al. 1998; Baxter, 2003; Kerley et al 2008), as suggested by a number of studies where a negative correlation between the elephant biomass and the browser biomass existed across ecosystems (Fritz et al., 2002; Parker, 1983; Owen-Smith, 1989). This suggests that there could be an optimum level above which elephant have a negative impact on biodiversity (Whyte et al., 2003).

By considering Fig. 3.1, we add an additional objective, namely biodiversity conservation to the model. The increase in number of the elephants can result in environmental costs such as loss of biodiversity (Fig. 3.1). This is important in the case where the landowner has multiple objectives – profit maximization and biodiversity conservation. These environmental costs can be denoted by the following:

$$M = M(X) \quad (3)$$

Where $M(X)$ are the environmental costs dependent on the elephant stock.

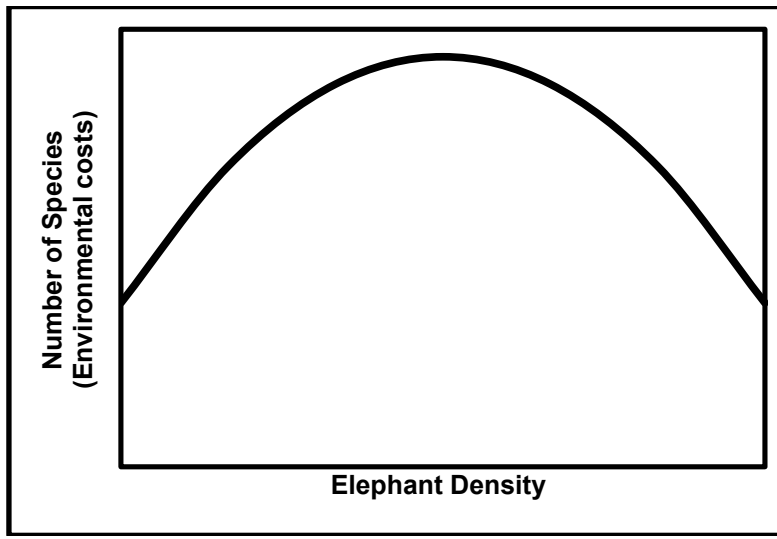


Fig. 3.1: The hypothetical relationship between elephant density and biodiversity as defined by the number of species.

3.2.1.1 Optimal management in the single species model

We start by analyzing the case where elephant does not impose costs on the environment. In this case, the question for the landowner is how the two production activities, tourism and hunting, could be balanced against each other so as to maximize the current value of profit using the two control variables, elephant offtake and waterpoints. The objective of the landowner is maximization of present value (PV) of welfare (Π), given as follows:

$$\text{Max}_{q,W} PV\Pi = \int_0^{\infty} [pqX + TX - ZW]e^{-rt} \quad (4)$$

The current-value Hamiltonian of the problem is given by the following:

$$H = pqX + TX - ZW + \lambda[h(X, W) - qX] \quad (5)$$

In this current-value Hamiltonian equation, X is the state variable, W and q are the control variables, and λ is the current value costate variable for herbivore stock.

After using the Pontryagin's maximum principle (Clark, 2005; Conrad and Clark, 1987), (Appendix 3) we get the following equations:

$$h^w(X, W) = \frac{Z}{p} \quad (6)$$

$$r = h^x(X, W) + \frac{T}{p} \quad (7)$$

The equations (6) and (7) are the reduced form of the necessary condition for a maximum when an interior solution is assumed to be present, that is, where stock size, harvesting rate, and waterpoints at the steady state are positive in the long term.

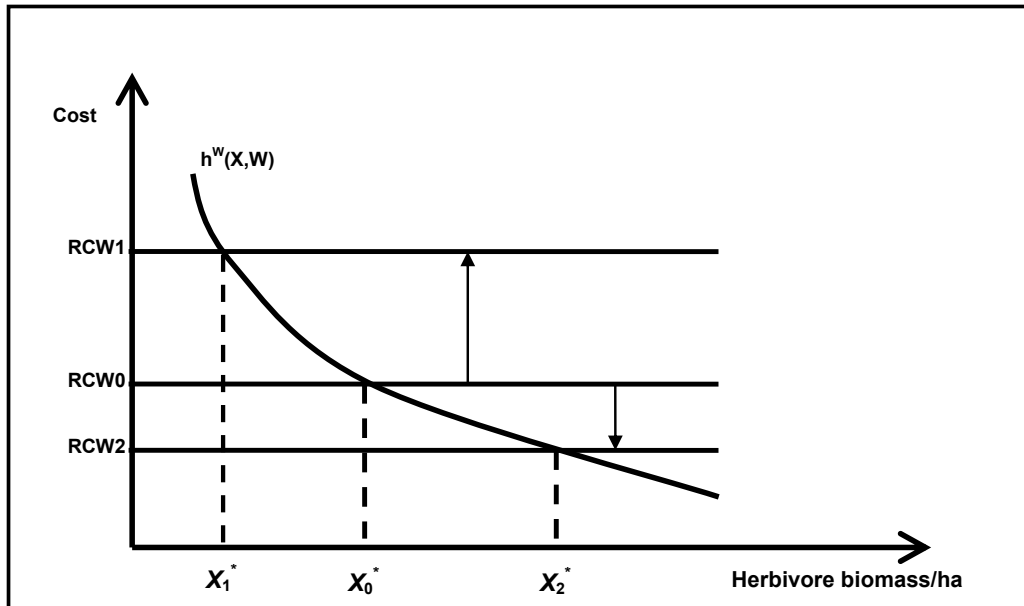


Fig. 3.2: Effects of change in cost of waterpoints maintenance (Z) and price of herbivore offtake (p) on the relative cost of waterpoint maintenance (RCW).

Equation (6) reflects the impact of the waterpoint control term in the objective function. The left hand side of equations (6) is the marginal effect of waterpoints on elephants representing the demand curve for waterpoints. The right hand side is the ratio of cost of waterpoint maintenance and price of herbivore offtake which represents relative cost of waterpoint

maintenance (RCW). When cost of waterpoint maintenance goes up or price of herbivore offtake goes down, the relative cost of waterpoint maintenance shift from RCW0 to RCW1 because of less demand of waterpoints (Fig. 3.2). This lead to the optimum herbivore biomass to decrease from X_0^* to X_1^* . On the other hand, when the cost of waterpoint maintenance decreases or price of herbivore offtake goes up, the relative cost of waterpoint maintenance shift from RCW0 to RCW1. This results in increase demand for waterpoints and lead the optimum herbivore biomass to increases from X_0^* to X_2^* (Fig. 3.2). The left hand side of equation (7) is r , the economic discount rate determined by the market forces. The right hand side of equation (7) can be regarded as the elephant's own interest rate (Clark and Munro, 1975) which consists of two components: (i) the marginal productivity of the elephant; and (ii) the marginal revenue from non-consumptive use term.

Overall, it is apparent from all the components of the elephant's own interest rate that if the landowner increases the number of waterpoints, it would result in an increase in the elephant marginal productivity. In our model, the price of elephant offtake has a direct effect on the elephant's own interest rate, whilst the waterpoints maintenance cost has an indirect effect. The effect of a permanent increase in the price of elephant offtake on the elephant's own interest rate is ambiguous (Table 3.1). It has a positive or negative effect on the marginal productivity whilst a negative on the marginal benefits from non-consumptive use term. If the increase in elephant's offtake results in the rate of increase in the marginal productivity surpassing the rate of decrease in the marginal benefits from non-consumptive use term, then that would result in a positive effect on the elephant's own interest rate. The effect of the cost of waterpoint maintenance is negative (Table 3.1), indicating that if the cost of waterpoint maintenance increase, this may prompt the landowner to close some of the waterpoints, leading to a drop in elephant population. The effect of an increase in the marginal benefits from non-consumptive use is positive (Table 3.1). In this model, possible negative effects of elephants on biodiversity are not taken into consideration.

Table 3.1: Comparative statics – effect of change in the price of herbivore offtake (p) and the cost of waterpoints maintenance (Z) marginal benefits from non-consumptive use on the elephant's own interest (one species model)

| | $h^X(X, W)$ | $\frac{T}{p}$ | Overall |
|-----|-------------|---------------|---------|
| p | -/+ | - | -/+ |
| Z | - | 0 | - |
| T | + | + | + |

3.2.1.2 Optimality with environmental costs

When the environmental costs caused by elephants are taken into consideration, the picture changes. In this case, the landowner would be maximizing the present value (PV) of welfare (Π) as follows:

$$\text{Max}_{q,W} PV\Pi = \int_0^{\infty} [pqX + TX - ZW - M(X)]e^{-rt} \quad (8)$$

The new current-value Hamiltonian is given by the following:

$$H = pqX + TX - ZW - M(X) + \lambda[h(X, W) - qX] \quad (9)$$

After using the Pontryagin's maximum principle (Conrad and Clark, 1987; Clark, 2005), we get the same equation (6), however, equation (7) is modified as follows:

$$r = h^X(X, W) + \frac{T}{p} - \frac{M'(X)}{p} \quad (10)$$

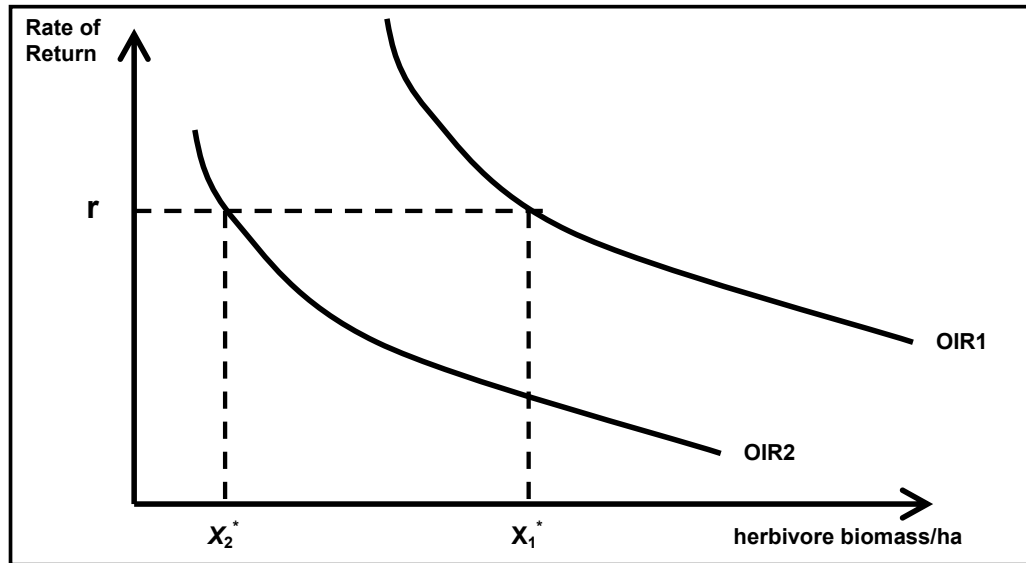


Fig. 3.3: The effect of considering environmental costs by elephants in a game reserve area.

Under the scenario where environmental costs are taken into consideration by the landowner, the elephant population and waterpoints density will be lower than in the case where environmental costs are not taken into consideration (Fig. 3.3; equation (7); equation (10)). We get the opposite picture if we assume that elephant density has not yet reached its optimum density – then there would be environmental benefits. In a case where there are environmental costs, the strategy of the game reserve management would be to close waterpoints until the optimum elephant density is attained. On the other hand, if there are

environmental benefits, then constructing of more waterpoints until the optimum density of elephants is attained would be the best strategy. However, this model until now does not explicitly model the effects of elephants on biodiversity. We address this in the two species model.

3.2.2 The two species model

We now adopt the notation: the subscript denotes respective species whilst superscript denotes derivative with respect to the variable shown in the superscript. Now we consider two species where one species, X_1 , is water-dependent, represented by elephant and the other species, X_2 is water-independent, represented by roan which symbolizes biodiversity.

The dynamics of the elephant is given by:

$$\frac{dX_1}{dt} = h_1(X_1, X_2, W) - q_1 X_1 \quad (11)$$

On the other hand the dynamics of roan, is given by:

$$\frac{dX_2}{dt} = h_2(X_1, X_2) - q_2 X_2 \quad (12)$$

Where: i denotes the species and in this case $i = 1, 2$; $h_1(X_1, X_2, W)$ is the growth function of elephants, which depends on its own stock, roan stock, and additional water provision through waterpoints, W ; $h_2(X_1, X_2)$ is the growth function of roan, which depends on its own stock, and elephant; q_i is the rate of herbivore offtake for species i . We assume that the following: $h_1^1(X_1, X_2, W) > 0$; $h_2^2(X_1, X_2) > 0$; $h_1^W(X_1, X_2, W) > 0$, implying that additional waterpoints would result in an increase in elephant population size; $h_2^1(X_1, X_2) < 0$ and $h_1^2(X_1, X_2, W) < 0$ implying that there is competition for forage between the two species. For model simplicity we assume that there is only one forage type available for which both species compete. Waterpoints have an indirect effect on roan through the growth of elephant population.

For this case we also assume that the index of welfare (Π) is a measure of the landowner's net benefit. The landowner can manipulate the rate of herbivore offtake of species i , q_i , and the number of waterpoints W , that would maximize his net benefit. The profit function for the landowner is given as follows:

$$\Pi = p_1 q_1 X_1 + p_2 q_2 X_2 + T_1 X_1 + T_2 X_2 - ZW \quad (13)$$

Where: $p_i q_i X_i$ is the revenue generated from selling the hunting licences of herbivore species, i . We assume that p_i , which is the price of hunting licence of herbivore species i , is fixed, $T_i X_i$ is the tourism revenues realized from species i .

The objective of the landowner is maximization of the present value (PV) of welfare (Π), given as:

$$\text{Max}_{q_i, W} PV\Pi = \int_0^{\infty} [p_1 q_1 X_1 + p_2 q_2 X_2 + T_1 X_1 + T_2 X_2 - ZW] \quad (14)$$

The current-value Hamiltonian of the problem is now:

$$\begin{aligned} H = & p_1 q_1 X_1 + p_2 q_2 X_2 + T_1 X_1 + T_2 X_2 - ZW \\ & + \lambda_1 [h_1(X_1, X_2, W) - q_1 X_1] + \lambda_2 [h_2(X_1, X_2) - q_2 X_2] \end{aligned} \quad (15)$$

In this current-value Hamiltonian equation, X_1 and X_2 are the state variables, W , q_1 and q_2 are the control variables, λ_1 and λ_2 are the current value costate variables for the herbivore stock species X_1 and X_2 respectively.

3.2.2.1 Optimality in the two species model

After using the Pontryagin's maximum principle, we get:

$$h_1^W(X_1, X_2, W) = \frac{Z}{p_1} \quad (16)$$

$$r = h_1^1(X_1, X_2, W) + \frac{T_1}{p_1} + \frac{p_2 h_2^1(X_1, X_2)}{p_1} \quad (17)$$

$$r = h_2^2(X_1, X_2) + \frac{T_2}{p_2} + \frac{p_1 h_1^2(X_1, X_2, W)}{p_2} \quad (18)$$

Equations (16), (17) and (18) are the reduced form of necessary conditions for a maximum when an interior solution is assumed to be present. Equation (16) is similar to equation (6) in the single species model. Unlike in the single species model, the two species model has two

herbivore's own interest rates, for elephant and roan, as represented by the right hand sides of equations (17) and (18) respectively. Each herbivores' own rate comprise the following components: (i) the herbivores marginal productivity, (ii) the herbivore's marginal benefits from non-consumptive use term and (iii) the marginal effect of the competing herbivore on the herbivore term.

The landowner would establish higher density of waterpoints which would result in thicker stocks of elephants and thinner stock of roan with possibility of local extinction of roan under these conditions – if ecological factors: the marginal effect of elephant on roan is very low (elephant are a weak competitor) and the marginal effect of roan on elephant is very high (roan outcompete elephants) and the economic factors, the price of elephant offtake is high and the price of roan offtake is very low. The landowner would close some of the waterpoints resulting in lower density of waterpoints leading to thinner elephant stock and thicker stocks of roan under the opposite conditions.

The effect of change in the elephant offtake price on the elephant's own interest rate can be deduced as follows: if the price of elephant's offtake increases, the landowner would establish more waterpoints this would result in: ambiguous effect on the elephant's marginal productivity; negative effect on the marginal benefits from elephant's non-consumptive use term; ambiguous effect on the marginal effect of elephant on roan term (would have a positive effect if the increase in the price of herbivore's offtake – denominator, is more than the increase in the marginal effect of elephant on roan – numerator). Overall, the effect of an increase in the elephant offtake price on the elephant's own interest rate is ambiguous (Table 3.2). An increase in the price of elephant's offtake would have positive effects on the elephant's own interest rate if total positive effects from the elephant's marginal productivity and the marginal effect of elephant on roan term outnumber the negative effects from the marginal benefits from elephant's non-consumptive use term. The effects of an increase in price of elephant offtake (p_1), price of roan offtake (p_2), waterpoint maintenance cost (Z), elephant's marginal benefits from non-consumptive use (T_1) and roan's marginal benefits from non-consumptive use (T_2) on the elephant's own interest rate are summarised in Table 3.2 and on the roan's interest rate in Table 3.3.

Table 3.2: Comparative statics for the elephant's own interest rate – the effects of an increase in price of elephant offtake (p_1), price of roan offtake (p_2), waterpoint maintenance cost (Z), elephant's marginal benefits from non-consumptive use (T_1) and roan's marginal benefits from non-consumptive use (T_2) on the elephant's own interest rate

| | $h_1^1(X_1, X_2, W)$ | $\frac{T_1}{p_1}$ | $\frac{p_2 h_2^1(X_1, X_2)}{p_1}$ | Overall |
|-------|----------------------|-------------------|-----------------------------------|---------|
| p_1 | -/+ | - | -/+ | -/+ |
| p_2 | - | 0 | - | - |
| Z | - | 0 | - | - |
| T_1 | + | + | - | -/+ |
| T_2 | - | 0 | + | -/+ |

Table 3.3: Comparative statics for the roan's own interest rate – the effects of an increase in price of elephant (p_1), price of roan offtake (p_2), waterpoint maintenance cost (Z), elephant's marginal benefits from non-consumptive use (T_1) and roan's marginal benefits from non-consumptive use (T_2) on the roan's own interest rate

| | $h_2^2(X_1, X_2)$ | $\frac{T_2}{p_2}$ | $\frac{p_1 h_1^2(X_1, X_2, W)}{p_2}$ | Overall |
|-------|-------------------|-------------------|--------------------------------------|---------|
| p_1 | - | 0 | - | - |
| p_2 | -/+ | - | -/+ | -/+ |
| Z | + | 0 | + | + |
| T_1 | - | 0 | + | -/+ |
| T_2 | + | + | - | -/+ |

3.3 Implications of the model

In the one species model without environmental costs/benefits, controls have direct effects on state variable and nothing else which is far from reality. In the one species model with environmental costs/benefits, the controls have direct effects on the state variable and indirect effects on the environmental costs/benefits. However, the environmental costs/benefits have no control which directly affects them.

In the two species model, the controls, elephant offtake and waterpoints have direct effects on elephant stock, whereas they have indirect effects on roan stock through the influence of elephant stock on roan stock (Fig. 3.4). So the landowner, could use waterpoints and elephants to influence directly the elephant stock and indirectly the roan stock. On the other hand, he could influence the roan stock directly by using roan offtake, and this would influence indirectly the elephant stock. Although, the two species model renders the modelling exercise complex, it brings some realism into analytical modelling.

Just as for the case of the single species model, in the two species model, the effect of an increase in the elephant offtake price on the elephant's own interest rate is ambiguous. An increase in the price of elephant's offtake would have negative effects on the elephant's own interest rate if the positive effects from the elephant's marginal productivity and the marginal

effect of elephant on roan term are less than the total negative effects from the marginal benefits from elephant's non-consumptive use term (equation (17)). In such case, if the elephant offtake price increases, it would reduce the elephant population, and it would result in an increase in the roan population, representing biodiversity, as a result of a reduction in competition from elephant. So, if there are extremely high prices of elephant offtake it might result in the local extinction of elephants. In other words to preserve elephants, the price of its offtake must be kept at relatively low prices. This would be in agreement with Clark's model (1973) and policies that have been used by CITES where a moratorium on ivory trade is expected to lead to a reduction in the price of ivory (Clark and Munro, 1975; Skonhoft, 1995; Barnes, 1996; Bulte and van Kooten, 1999a; Bulte and van Kooten, 1999b; Alexander, 2000).

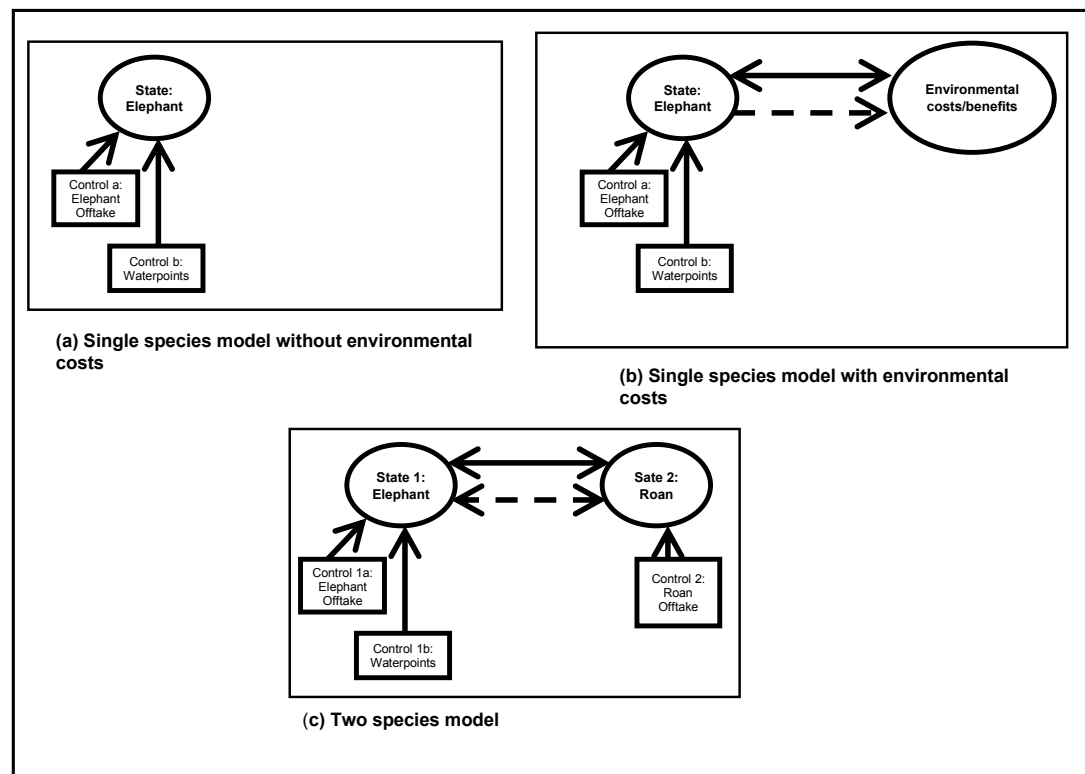


Fig. 3.4: Demonstrating the direct and indirect effects of controls on the models. Continuous arrows depict direct effects whilst broken arrow depicts indirect effects of controls states or environmental costs/benefits.

In the one species model without environmental costs considered, waterpoints are assumed to affect only the herbivore that uses them, and it ignores the indirect effects of waterpoints on other species (Fig. 3.4). In the model where environmental costs/benefits are introduced, the indirect effects of waterpoints are captured in the elephant's own interest (equation (10)) as externalities caused by increasing number of elephants. However, this formulation does not capture the ecological and economic interaction of elephants and other species in the

ecosystem. In the two species model, the direct and indirect effects of waterpoints are captured through the two herbivore's own interest rates which show that there is an ecological and economic interaction of the species (equations (17); equation (18)).

If the elephant's non-consumptive use value is dominant, then an increase in the elephant's marginal benefits from non-consumptive use would be an incentive for the landowner to establish more waterpoints. This would result in a positive effect on the elephant's own interest rate, because it will increase the marginal revenues from elephant's non-consumptive use, overriding the loss in other effects of the elephant's own interest rate (equation (17)). An increase in elephant non-consumptive use could be as a result of a growth in international demand for wildlife-viewing tourism (Barnes, 1996). In such a case an increase in elephant's marginal benefits from non-consumptive use will have a positive effect on the elephant's own interest rate plane, shifting it upwards, whilst the roan's own interest rate plane will shift downwards as a result of more competition for forage as elephant stock increases (Fig. 3.5). On the other hand, if the roan's (biodiversity's) non-consumptive use value is dominant, then an increase in roan's marginal benefit from non-consumptive use would make the landowner to close up some waterpoints. This would result in an overall positive effect on the roan's own interest rate, because the gain in marginal revenues from roan's non-consumptive use effects will surpass the loss in other effects of the roan's own interest rate. An increase in marginal benefits from biodiversity could be from society's increase in perceived value of biodiversity (Gowdy, 1997; Fuller et al., 2007; Alho, 2008).

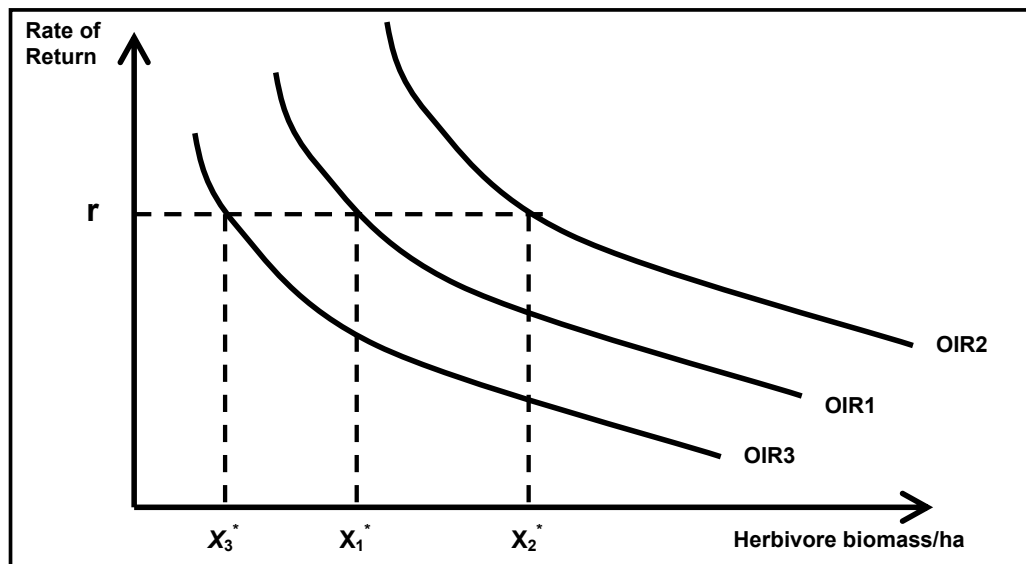


Fig. 3.5: The positive and negative effects of the herbivore's own interest rate components on the own interest rate (OIR) plane projected on the resource stock, rate of return plane (for simplicity the plane represented by curve in discount rate-herbivore biomass axis).

In extreme cases where the elephants are regarded as having extremely high non-consumptive use value relative to biodiversity, then it would be optimal for landowner to keep them at higher biomass levels at the expense of biodiversity (Tisdell, 2005). However, it could be that the elephant density has already reached its optimum density such that a subsequent increase in elephant density leads to decrease in biodiversity. In the one species model, this is not accommodated in the model. However, in the one species model with environmental costs, it occurs when elephant density has reached its optimum density and subsequent increase in elephant density leads to a decrease in biodiversity (Fig. 3.1). In the two species model it is assumed that there is competition between the two species for forage. In such case, if the landowner responds to the increase in value of elephants by increasing waterpoints, the water-independent species is adversely affected since its population would decrease due to increased competition from increased population of the water-independent species. The opposite case applies where biodiversity is regarded as having extremely high value, the landowner would close some of the waterpoints which would result in thinner stocks of elephants. However, in cases, where both species have relatively moderate values, the landowner would sustain both species (Fleming and Alexander, 2003).

The condition for obtaining an interior solution would be satisfied if the ecological competition between the two species is not too intense. In addition, the conditions for an interior solution would hold if the marginal benefits from non-consumptive use between the two species do not sharply differ. If T_1 is very large compared to T_2 , it would become optimal for the landowner to completely deplete the roan population. This places a very large value on the marginal benefits from non-consumptive use of elephants, T_1 . Usually there is a strong campaign among preservationist organizations and animal rights movements for the preservation of elephants at all costs (Hutton, 1995; Barnes, 1996; Hollander, 1998). This places a very large value on the marginal benefits from non-consumptive use of elephants, T_1 . In addition, since the preservationist organizations and animal rights movements advocate for little or no elephant hunting (Hollander, 1998), they drive the price of elephants to very low levels thereby increasing elephant's own interest rate (right hand side of equation 17) whilst reducing the biodiversity's own interest rate being represented by roan here (right hand side of equation 18).

3.4 Discussion

If the landowner only considers the species that is going to be affected directly by the control, then the indirect effects of such a control could have large consequences for his net returns

and for biodiversity. In such a case, species favoured by the control would be boosted at the expense of the other species indirectly affected. In the one species model, when environmental costs are internalized, the optimum stock level of the herbivore is reduced, assuming the herbivore causes negative effects on the environmental. Due to taking into consideration of the externalities (negative effects) to biodiversity that could be brought about by increasing the density of the herbivore species, the optimum herbivore biomass would be lower than in the case where the negative effects are externalized. So it is prudent to include the externalities into the landowner's maximization problem. In practice externalities are difficult to quantify (Jordan, 1995), but this does not mean they should be left out (Bulte et al., 2003; Clark, 2005). This is because ignoring them may have severe long term consequences. The long term consequence of the waterpoints would result in biodiversity loss. This in turn would result in huge economic loss that was not evident in the short term in addition to the ecological loss. Thus the landowner needs to consider both direct and indirect ecological economic effects of a control tool at his disposal. Ecological information of the direct and indirect effects of waterpoints on different herbivores should be provided to the landowner so that he has advanced understanding of the ecological and economic effects of waterpoints thus allowing for sound decision making. Ecological studies should provide information to establish the estimated optimum waterpoints density expected for a given area. It would be good if the landowner on his own takes into consideration the negative effects that could be brought about by such management tools as waterpoints.

However, a landowner may be interested only in maximization of commercial profits from elephant offtake and tourism revenue, and might ignore the negative effects that could be brought about by increasing elephant population to biodiversity (Ntiamoa-Baidu, 1997; Tisdell, 2005). So the landowner would manipulate waterpoints to maximize his objective. This is represented by the one species model that we have discussed in this paper. In this case the landowner would establish excessive waterpoints, resulting in more than optimum elephant biomass without regarding the environment costs. The environmental costs would be considered to be externalities to the landowner since they would not be taken into account (Tisdell, 2005). This would lead to negative effects in the long term. So the game reserve management as an authority entrusted with sustainable management of the game reserve on behalf of all the landowners, should ensure that the landowner takes these externalities into consideration and incorporates them into his production. The game reserve management may do this by providing information which would enable the landowner to take environmental costs into consideration in his maximization problem.

If the landowner is not taking into consideration the negative effects of tools such as waterpoints, then game reserve management may use economic instruments such as subsidies or payments to the landowners who are complying with sound waterpoints management; and/or taxes or charges to landowners exceeding a certain density of waterpoints (Wallart, 1999; Bulte et al., 2003). This is a command and control type method (The Royal Society, 2002). Theoretically, the density of waterpoints that is ecologically and economically optimal, could be determined for each landowner, so that a charge could be applied for additional waterpoints established by the management and/or payment be paid for waterpoints closed below the optimum number/density. The game reserve management should however be ready for possible resistance in implementation of taxes from the landowners who might form a lobbying group which could make implementation difficult (Bulte et al., 2003; Brown, 2000, Wallart, 1999).

By considering two species differentially affected by waterpoints, ecological economics effects could be determined and corrective action could be taken to avoid adverse effects of waterpoints on the species that is negatively or indirectly affected. However, in an environment where there are various species differentially affected by waterpoints, the problem is even more complex. Nevertheless, our model gives a good framework to start to analyze such cases. The species could be classified into groups according to how they compete and with regard to their environmental requirements (e.g., water dependency versus water-independency). Our framework could then be used to analyze the ecological economic effects of these different species groups. So information would be required on how a control such as waterpoints construction affects the different species' groups directly and indirectly.

Acknowledgments

This study is part of an integrated programme, The Elephant Movements and Bio-economic Optimality programme (TEMBO), financially supported by the Netherlands Foundation for the Advancement of Tropical Research (WOTRO), the Dr Marie Luttig trust, and Shell South Africa. We would like to thank the Associated Private Nature Reserves (APNR) for providing assistance in the study which resulted in this paper. We thank Erwin Bulte for his critical comments and fruitful discussions in development of this paper.

Appendix 3: Derivations of Equation 6 and 7

The landowner's objective is given by the following:

$$Max_{q,W} PV\Pi = \int_0^{\infty} [pqX + TX - ZW]e^{-rt} \quad (A1)$$

$$\text{Subject to } \frac{dX}{dt} = h(X, W) - qX \quad (A2)$$

The current value Hamiltonian:

$$H = pqX + TX - ZW + \lambda[h(X, W) - qX] \quad (A3)$$

The Pontryagin's necessary conditions for a maximum are:

$$\frac{dH}{dq} = pX - \lambda X = 0 \quad (A4)$$

$$\frac{dH}{dW} = -Z + \lambda h^W(X, W) = 0 \quad (A5)$$

$$\frac{dH}{dX} = pq + T + \lambda[-q + h^X(X, W)] \quad (A6)$$

$$-\frac{dH}{dX} = \dot{\lambda} - r\lambda \Rightarrow \dot{\lambda} = r\lambda - \frac{dH}{dX} \quad (A7)$$

$$\frac{dH}{d\lambda} = -qX + h(X, W) = 0 \Rightarrow q = \frac{h(X, W)}{X} \quad (A8)$$

Solve equations (A4) for λ

$$\lambda = p \quad (A9)$$

Take d/dt of equation (4)

$$\dot{\lambda} = \frac{d}{dt}[p] \quad (A10)$$

Substituting equation (A9) into equations (A5) and (A7)

$$-Z + ph^W(X, W) = 0 \quad (A11)$$

$$\dot{\lambda} = rp - \{pq + T + p[-q + h^X(X, W)]\} \quad (A12)$$

We assume that an equilibrium exists so that all conditions are simultaneously met. At equilibrium $\dot{\lambda}$ is equal to zero. This makes equations (A10) to (A12) equal to zero. Given these conditions and substituting for q as shown in equation (A8) into equation (A11), solve equations (A11) and (A12).

$$h^W(X, W) = \frac{Z}{p} \quad (A13)$$

$$r = h^x(X, W) + \frac{T}{p} \quad (\text{A14})$$

Elephant and biodiversity: manipulating waterpoints to meet management objectives

Emmanuel Mwakiwa; Mike Coughenour; John W. Hearne; Henrik J. de Knegt; Willem F. de Boer; Edward Kohi; Johannes D. Stigter; Herbert H.T. Prins

Abstract

Game reserve managers use tools to meet their objectives in wildlife management. One such tool is the construction or closing of artificial waterpoints. Elephant distribution and density can be steered through artificial waterpoints. Elephants are known to be important drivers of vegetation dynamics in African savannas. However, the direction of the effects of waterpoints on the dynamics of both vegetation and herbivores is highly debated. Given a huge ecosystem like the Kruger National Park (KNP), which has a decentralized regional management system, it would be important to analyse these effects at regional level. We used the Savanna ecosystem model to analyse the impact of waterpoints on the KNP's regions under 26 waterpoints manipulation scenarios. We used the model to analyse elephant impact on vegetation biomass diversity in four regions of KNP. The results showed that a changing waterpoint density has different effects on both elephant density and vegetation biomass diversity in the different regions. Constructing or closing extra waterpoints in one region does not necessarily translate into higher or lower elephant densities respectively, but the effect depends on the vegetation and other conditions of the region in comparison to neighbouring regions. The model showed that there is a trade-off between elephant density, representing an economic objective and vegetation biomass diversity, representing an ecological objective. Increase in elephant density resulted in decrease or increase in vegetation biomass diversity depending on the elephant densities. The model suggests that different strategies should be adopted for different regions, e.g., an adaptive management strategy could be used for certain regions where waterpoints are switched on and off depending on the elephant density.

Keywords: Savanna ecosystem model, elephant, vegetation biomass diversity, intermediate disturbance hypothesis, region, trade-off

4.1 Introduction

Wildlife management problems are often characterized by multiple conflicting objectives that should be achieved simultaneously (Chikumbo et al., 2001). An objective can be biological, social or economic. National park managers often face a challenge to develop wildlife management strategies that guarantee long term wildlife sustainability. In this paper, we consider a case where the management of a national park is pursuing two potentially conflicting objectives, an ecological objective to maintain species diversity, and an economic one aimed at increasing tourism revenues through maintaining high elephant densities.

To meet their objectives, national park managers use management tools, such as surface water provisioning through constructing or closing of artificial waterpoints, fire management, fencing, and population manipulation through culling or translocation of animals (Perrings and Walker, 1997; Slotow et al., 2005; de Boer et al., 2007; Grant et al., 2002). Artificial waterpoints are used by game managers to supplement natural water supplies in semi-arid regions to support the existing populations, and to alter the density and distribution of wildlife and its impact on the vegetation (Chamaillé-Jammes et al., 2007; de Beer and van Aarde, 2008; Grant et al., 2002).

Pursuing these different conflicting objectives requires the managers to make trade-offs, with regard to wildlife impact on the plant and herbivores community or with regard to the tourist revenues. The effects of waterpoints on the dynamics of both vegetation and herbivores are highly debated (Pienaar, 1985; Pienaar et al., 1997; Thrash and Derry, 1999; Baxter, 2003; Mabunda et al., 2003; Redfern et al., 2003; Chamaillé-Jammes et al., 2007; Smit et al., 2007b). For example, Kruger National Park (KNP) has a problem with regard to some species dwindling in numbers, which according to some studies is due to the high prevalence of artificial waterpoints. So, KNP management has closed some waterpoints in order to address these problems (Pienaar et al., 1997; Grant et al., 2002). Hence, getting a better understanding of the interactions and feedbacks among herbivores and waterpoints could be important for the management of biodiversity in many savanna ecosystems. Managers have long been concerned by the unwanted increases of some species at the expense of others in protected areas (Western, 1975; Prins and van der Jeugd, 1993; Pienaar et al., 1997; Grant et al., 2002; Western 2006; Chamaillé-Jammes et al., 2007; Luske et al, 2009; Holdo et al., 2009). The development of predictive models of biodiversity dynamics could therefore assist. It is thus important that a better quantitative understanding of how multiple factors that influence savannas interact with each other is obtained, both to understand the functioning of these

systems and to understand the implications of certain management interventions (Holdo et al., 2009).

It is difficult to predict the impact of elephants on vegetation diversity. Some studies have shown a negative effect of increasing elephant density on vegetation cover and heterogeneity (Parker, 1983; Fritz et al., 2002) whilst others have shown a positive effect (Skarpe et al., 2000; Baxter, 2003; Makhabu et al., 2006; Fornara and du Toit, 2007; de Knecht et al., 2008; Kalwij et al., 2010). On the other hand, some have suggested that there would be a higher diversity of vegetation structure at intermediate elephant densities whilst at extreme levels of both low and high disturbance the diversity would be reduced, following the intermediate disturbance hypothesis (Tchamba, 1995; Baxter, 2003; Whyte et al., 1999; Kerley et al., 2008; Fig. 4.1). These differences in vegetation diversities are expected to influence the diversity of the associated fauna. Various conservation areas therefore stimulate the increase of vegetation heterogeneity, e.g., through mosaic burning, in an effort to create multiple ecological niches and thereby increase the faunal species diversity, as maximizing species diversity is often a management objective (Kalwij et al., 2010).

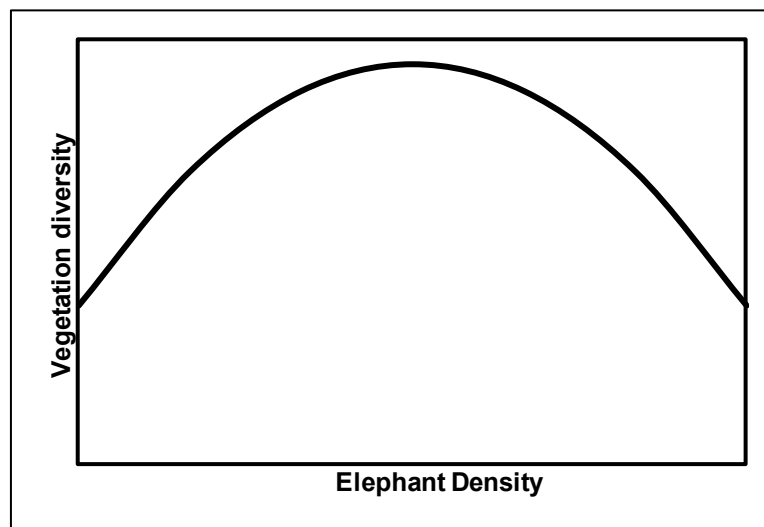


Fig. 4.1: Predicted relationship between elephant density and vegetation diversity as stipulated by the intermediate disturbance hypothesis.

An ecosystem can pass through various stages of disturbance in different places and at different times (Whyte et al., 1999). The species diversity in a large national park could differ per region, because different regions have a different resilience, which differentially affects the species interactions (Whyte et al., 1999). For instance, when waterpoints are used to manipulate elephant densities in different regions that are not physically separated, it would be interesting to understand the impact of these different densities on the diversity of the

vegetation. It is not clear whether those regions with higher numbers of waterpoints would also have higher elephant densities. This is because elephant density is not only affected by water but also by other resources such as vegetation biomass, slope, forage quality and temperature (de Knegt et al., 2010).

Elephants, apart from affecting the diversity of the vegetation, also have an economic value as shown by people who are willing to pay substantial amounts of money to view them (Okello, et al., 2008). For countries as South Africa, ecotourism is partly dependent on the charismatic megafauna such as elephants (Heal, 2005). Barret et al. (2006) found that the growth in tourism (tourists per protected area) was positively associated with elephant population size, although the causal relationship was unclear. Hence, national park managers invest in the establishment of waterpoints, anticipating an increase in the number and visibility of species like elephant, ultimately leading to an increase in income through increased tourism (Mabunda et al., 2003; Chamaillé-Jammes et al., 2007; Smit et al., 2007b).

We used the Savanna ecosystem model, for short Savanna (Coughenour, 1993) to analyse the different impact of waterpoints on the vegetation heterogeneity in the different regions of KNP under several waterpoints manipulation scenarios. We also analysed elephant impact on vegetation biomass diversity in these regions. We tested whether the impact of elephant on vegetation biomass follows the intermediate disturbance hypothesis or whether it is linearly increasing or decreasing. We treated elephant density as a proxy for the economic objective and vegetation biomass diversity to represent the ecological objective.

We would like to contribute to the discussion of sustainable utilization of wildlife resources under multiple and conflicting objectives, where national park managers use waterpoints in different regions of a national park as a tool to influence the elephant distribution. Evaluating how ecosystem components change in response to management actions requires a model that is complex enough to represent those management actions fairly (Hilborn, 1995). Savanna, with a spatially explicit landscape, enables a regional-scale analysis. Our goal is to provide resource managers with a scientific basis for the environmental impacts of different waterpoints management strategies, so that informed decision can be taken with regard to biodiversity conservation and income generation through tourism.

The present policy for management of elephant in the KNP (Whyte et al. 1999) focuses on the extent and intensity of elephant impacts on biodiversity, and is based on three assumptions:

- a. That fluctuations of conditions and population responses are an inherent attribute of ecosystems and contribute to biodiversity;
- b. That elephants are important agents of habitat modification and thus contribute to biodiversity;
- c. That it is possible to manipulate elephant populations in different regions of the national park through differential use of management tools, such as waterpoint density.

Given these principles, KNP manages elephants on a zone basis. The policy proposes that the KNP be divided into six zones: two botanical reserves, two high-elephant-impact zones and two low-elephant-impact zones (Whyte et al., 1999; Owen-Smith et al., 2006).

4.2 Methods

4.2.1 Study area

Biodiversity conservation is KNP's bedrock for the organisation's activities against which other core functions, such as tourism, are anchored (SANParks, 2009). Another objective of KNP is to raise income through tourism that will cover the costs of biodiversity conservation (SANParks, 2009).

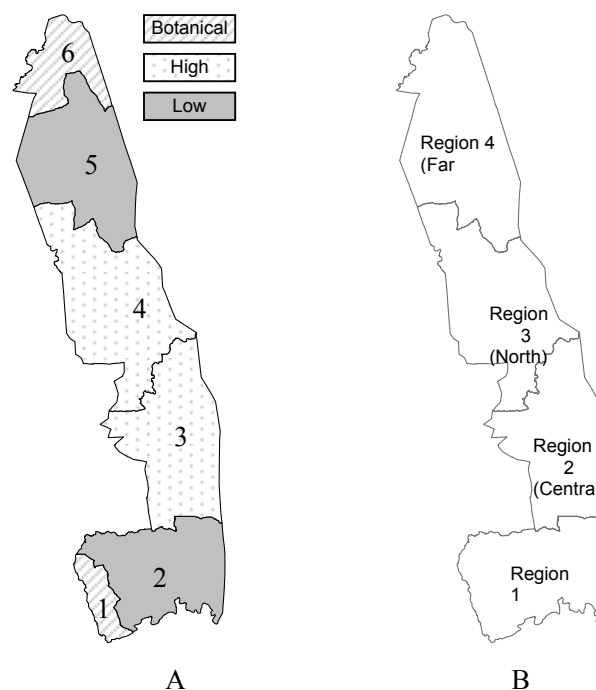


Fig. 4.2: A – Elephant impact zones in Kruger National Park. B – Regions as used in this paper.

There is a total of six perennial rivers that cross the park from west to east. At least 300 boreholes were drilled between the 1930s and 1980s to provide reliable surface water throughout the year (Gaylard et al., 2003). However, the KNP revised the water provision policy in 1997, which resulted in at least half of the waterpoints being closed down and is even considering closing more waterpoints (Pienaar et al., 1997; Smit et al., 2007b). KNP was divided into 6 elephant impact zones (Fig. 4.2A), grouped for simplicity into 4 larger zones in this analysis (Fig. 4.2B).

Model Input Data Processing

Data on the spatial variation in vegetation, elephant population, topography, and climate, together with data for distances to rivers, pans, waterpoints, and boundaries was provided by the Scientific Services, KNP. Statistics for the elevation (m above sea level), slope (degrees) were derived from SRTM 90m Digital Elevation Data (see <http://srtm.csi.cgiar.org/index.asp>). Herbaceous biomass data (g/m^2) was derived from Smit et al. (2007). Woody vegetation cover, the areal percentage that is covered with trees or shrubs, was derived from the MODIS Vegetation Continuous Fields dataset, which estimates per grid cell (500x500 m resolution) the percentage of bare soil, herbaceous vegetation and woody vegetation. The method is described by Hansen et al. (2002b) and validated for African woodland in Hansen et al. (2002a).

Land type classification and water holding capacity and depth data were derived from Kiker (1998) and Venter (1990). The Venter land type map, with 58 types, was used as a basis for assigning plant functional group compositions (Venter, 1990). Tree and shrub cover maps were derived from the woody cover map. Coordinates of functioning artificial waterpoints (boreholes) provided by KNP was used to make GIS coverage, from which a map was created illustrating the distances to artificial water. Coordinates of rivers, natural waterholes, and seasonal surface water (pans) were also provided, and corresponding distance to water maps were created. Surface water was assumed to be available only during the wet season (December to March).

Real rainfall data for the KNP was used. This was based on daily precipitation data for 20 stations in the KNP which was used to make a single monthly weather file covering a 30 year period from 1977 to 2006. Mean annual rainfall increases from north to south (Table 4.1). The south region (region 1) has the highest density of perennial rivers followed by the north region, far north region and central region, respectively (Table 4.1). The south region also has the highest herbaceous biomass (Table 4.1).

Table 4.1: Summary of ecological characteristics of Kruger National Park regions (source: Kruger National Park)

| | <i>Region 1 (South)</i> | <i>Region 2 (Central)</i> | <i>Region 3 (North)</i> | <i>Region 4 (Far North)</i> |
|--|-----------------------------|-------------------------------|-----------------------------|---------------------------------|
| Area (km ²) | 41967 | 4199 | 4944 | 5646 |
| Mean annual rainfall (mm)* | 675 | 592 | 509 | 492 |
| Perennial river density (m/km ²) | 54.3 | 8.7 | 40.9 | 24.9 |
| Waterpoint density (n/km ²) | 0.005 | 0.011 | 0.005 | 0.006 |
| Pan density (n/km ²) | 0.004 | 0.008 | 0.005 | 0.014 |
| Mean elevation (m a s l) | 339 | 306 | 336 | 373 |
| Average herbaceous biomass (g/m ²) | 3419 | 3280 | 2079 | 2909 |
| Average tree cover (%) | 16.4 | 13.0 | 13.0 | 13.9 |

* Source: WorldClim database (<http://www.worldclim.org/>)

4.2.2 The Savanna model

Simulation modelling is a valuable tool for exploring situations that would be difficult and costly to examine with field experiments. Savanna is a spatially explicit, process-oriented model of grassland, shrubland, savanna, and forested ecosystems at both local and regional scales over annual to decadal time scales (Coughenour, 1993; Coughenour and Singer, 1996; Kiker, 1998; Boone et al., 2002; Christensen et al., 2003; Weisberg and Coughenour, 2003). Savanna is composed of different submodels comprising the hydrology, plant biomass production, plant population dynamics, ungulate herbivory, ungulate spatial distribution, ungulate energy balance, and ungulate population dynamics. The development of the model began more than 15 years ago, to support research in the Turkana District of Kenya (Coughenour, 1993). Adaptations of the model were done for the Rocky Mountain National Park, Colorado (Weisberg and Coughenour, 2003); Yellowstone National Park, Wyoming (Coughenour and Singer, 1996); Kruger National Park, South Africa (Kiker, 1998); inner Mongolia, China (Christensen et al., 2003); and other areas.

We used Savanna to predict the effects of the increased and decreased water availability on elephant distribution. Savanna quantifies the impacts of herbivores on the vegetation. We adapted the Savanna model for Kruger National Park. Elephant numbers were held constant at the 2006 level of 11454. The distribution of elephants is expected to vary in the regions in response to differences in distances to water, manifested by simulated additional waterpoints under different waterpoints scenarios (see below). We fixed the elephant population at a constant level, but regional differences are created by attracting or dispersing elephants through opening or closing of waterpoints respectively.

Runs were performed using 5km square grid cells which gives a feasible simulation time for the entire extent of the study area, 19000 km². The Savanna model was configured with 10 plant functional groups, including three herbaceous groups, shrubs, palatable and unpalatable fine-leaf and broad-leaf trees, and mopane and marula trees. Data collected in the KNP Veld Condition Assessment (VCA) surveys was provided by KNP, which were utilized by Savanna as initial values for tree (either fine or broad leaved) and shrub species compositions and plant sizes in the different vegetation types. Canopy sizes were converted to canopy cover. Separately, species palatability and maximum heights were determined from existing sources (Kiker, 1998). Total canopy cover was calculated over the following functional groups: shrub, fine-leaved palatable tree, fine-leaved unpalatable tree, broad-leaved palatable tree, broad-leaved unpalatable tree, mopane trees, and marula trees.

Model parameters for plants and animals were largely based upon previous parameterizations for Kruger by Kiker (1998), and for the Serengeti-Ngorongoro area by Boone et al. (2002).

Elephant Spatial Distribution Submodel

The elephant spatial distribution submodel dynamically simulates elephant distributions over the simulated landscape. It is a dynamic habitat selection model. The habitat suitability index (HSI) of each grid-cell is computed for each week of the year. This index is a dimensionless scalar where 0 indicates no use, and 1.0 indicates maximum preference. The HSI's are summed up across all grid cells and standardized so they sum to 1.0. Then all elephants are distributed according to the standardized HSI given by the formula below.

$$\text{HSI} = F1(\text{preferred forage}) \times F2(\text{green herb}) \times F3(\text{energy intake rate}) \times \min[F4(\text{slope}), F5(\text{tree cover})] \times F6(\text{range location}) \times F7(\text{humans}) \times F8(\text{random variable})$$

The habitat suitability index of a grid-cell is affected by the continuous 0-1 scalar functions of preference-weighted forage biomass, green herbaceous biomass, slope, tree cover, range location, and humans. Importantly, what is being modelled is the suitability for foraging, since forage intake rate is multiplied by elephant density to determine offtake.

The forage function takes the value of the sum of the products of dietary preference weights times forage biomass over all dietary items in the grid-cell. Elephants can select areas with more abundant green herbaceous biomass. They may also select areas where digestible energy intake rates are highest, which is affected by forage abundance and quality. The minimum of the slope and tree cover effects is taken, hypothesizing that either of these effects can override

the other. A uniform random variate (0.8-1.0) is introduced that prevents the modelled animals from attaining an ideal free distribution due to imperfect knowledge of resource distributions, physical inaccessibility, and time lags for travel. The range function is a binary 0 or 1 that represents whether the grid-cell is in or out of the population's range.

Model implementation

A series of model simulations were run to simulate the water scenarios (see below). In these simulations, elephants were distributed using the Savanna spatial redistribution model (see above). Elephants were assumed to prefer relative level topography, areas with a minimal amount of tree cover, areas with more green herbaceous biomass, and areas with higher predicted digestible forage intake rates (de Knecht et al., 2010). The effect of distance to water was such that the effect on HSI declined linearly from 1 at 0 km from water to 0 at 12 km from water.

The key output response variable was elephant impact on vegetation biomass diversity. Normally, vegetation diversity is calculated based upon relative species abundances. However, here diversity is calculated based upon relative abundances of cover or biomass in different plant functional groups and size classes. The model was configured to simulate plant functional groups rather than species, due to the difficulty of having sufficient information at the species level and thus the focus here is on function rather than species composition. The distribution among plant sizes was of interest due to the possible consequences for vegetation structural heterogeneity and thus, habitats for species with different requirements. This program calculates diversity based upon above ground biomass of woody plants and herbaceous plants over the different functional groups. Diversity is calculated for each of the four KNP regions.

Diversity measure

Diversity was assessed using woody plant functional groups and size cohorts, along with open cover. The vegetation structural diversity was quantified by using the Shannon-Wiener diversity index on the relative abundance of plant functional groups, using the biomass in each functional group over several plant size classes as an index of abundance (Gillison, 2002). We used 10 plant functional groups: herb increasers type 1, herb increasers type 2, herb decreasers, shrubs, fine leaf palatable trees, fine leaf unpalatable trees, broad leaf palatable trees, broad leaf unpalatable trees, mopane trees, and marula trees. As for the plant size classes: for trees we summed across 6 size classes and for herbaceous plants only one size class was used. The Shannon-Wiener index (H') is given by the following formula:

$$H' = - \sum_{i=1, S; j=1, N} P_{ij} * \ln P_{ij}$$

where: P_i is the fraction of the entire population made up of the plant functional group i ; P_j is the fraction of the entire population made up of plant size class j ; S is the total numbers of plant functional groups encountered; N is the total numbers of plant size classes encountered.

4.2.3 Scenarios

A series of artificial water scenarios were created in GIS, and were randomly selected to suit requirements of each scenario. 26 scenarios were selected to address potential management questions regarding waterpoints management in the KNP. We considered the effects of increasing, or decreasing waterpoint density and the current waterpoints status. A series of simulations was run and the output compared to the control situation, the current waterpoints scenario (scenario P). The scenarios involved: no artificial waterpoints, quartering, halving, doubling and quadrupling the number of artificial waterpoints in each of the four zones (Table 4.2).

For each scenario, the Savanna model was run for 30 years, using the rainfall data from 1977 to 2006. For each scenario, we analysed for each region what happened each year to the vegetation biomass diversity (Shannon-Wiener index) of the plant functional groups and size classes, and the elephant density. We then compared the elephant densities and vegetation diversity over the different scenarios.

We tested whether the data followed a normal distribution. We then used one way ANOVA test and Tukey post-hoc test to check for significant effects between the scenarios in each region. We also tested for the linear and quadratic relationships between elephant density and vegetation biomass diversity in each region for the following scenarios: scenarios A (no artificial waterpoints), scenario F (quartering waterpoints), scenario K (halving waterpoints), scenario P (current situation), scenario Q (doubling waterpoints), and scenario V (quadrupling waterpoints). We only report and discuss the scenarios that were significant.

Table 4.2: Different waterpoints scenarios used in the Savanna model with number of artificial waterpoints in the four regions.

| Scenario | Number of waterpoints | | | |
|-------------|-----------------------|----------|----------|----------|
| | Region 1 | Region 2 | Region 3 | Region 4 |
| A | 0 | 0 | 0 | 0 |
| B | 0 | 46 | 24 | 32 |
| C | 20 | 0 | 24 | 32 |
| D | 20 | 46 | 0 | 32 |
| E | 20 | 46 | 24 | 0 |
| F | 20*0.25 | 46*0.25 | 24*0.25 | 32*0.25 |
| G | 20*0.25 | 46 | 24 | 32 |
| H | 20 | 46*0.25 | 24 | 32 |
| I | 20 | 46 | 24*0.25 | 32 |
| J | 20 | 46 | 24 | 32*0.25 |
| K | 20*0.5 | 46*0.5 | 24*0.5 | 32*0.5 |
| L | 20*0.5 | 46 | 24 | 32 |
| M | 20 | 46*0.5 | 24 | 32 |
| N | 20 | 46 | 24*0.5 | 32 |
| O | 20 | 46 | 24 | 32*0.5 |
| P (current) | 20 | 46 | 24 | 32 |
| Q | 20*2 | 46*2 | 24*2 | 32*2 |
| R | 20*2 | 46 | 24 | 32 |
| S | 20 | 46*2 | 24 | 32 |
| T | 20 | 46 | 24*2 | 32 |
| U | 20 | 46 | 24 | 32*2 |
| V | 20*4 | 46*4 | 24*4 | 32*4 |
| W | 20*4 | 46 | 24 | 32 |
| X | 20 | 46*4 | 24 | 32 |
| Y | 20 | 46 | 24*4 | 32 |
| Z | 20 | 46 | 24 | 32*4 |

Note: The numbers 20 (Region 1), 46 (Region 2), 24 (Region 3), and 32 (Region 4) represent the current number of waterpoints in each respective region. The multiple numbers i.e., “*0.25” indicate quartering, “*0.5” halving, “*2” doubling and “*4” quadrupling of the current scenario.

4.3 Results

4.3.1 Effects of waterpoints manipulation within a region

Region 1

We first investigated what happens to the elephant density in region 1 when waterpoints are manipulated in this region whilst the number of waterpoints is not changed in the other

regions. Removing all waterpoints and having a quarter of the waterpoints in region 1 (scenarios B and G; Fig. 4.3A) significantly decreased the elephant density compared to the current waterpoints situation, with a further reduction in elephant density when the waterpoints were quadrupled (scenario W; Fig. 4.3A). Halving and doubling of waterpoints in region 1 did not significantly change the elephant density (scenarios L and R; Fig. 4.3A). It is apparent from Fig. 4.3B, that in scenarios where waterpoints were reduced in one region or in all the regions (scenarios A to O) decreased the vegetation biomass diversity compared to the current scenario and scenarios where waterpoints were increased (scenarios P to Z). Another observation is that vegetation biomass diversity decreased when waterpoints were doubled (scenario R), but when waterpoints were quadrupled (scenario W) the diversity was not significantly different from the current situation (scenario P; Fig. 4.3B). Quadrupling the number of waterpoints in region 2 resulted in a higher elephant density in region 1 showing the indirect effects of having waterpoints in another region. However, waterpoints manipulation in the regions that are further from region 1, that is, region 3 and 4, resulted in no significant effect on either elephant density nor vegetation diversity in region 1 (Fig. 4.3A; Fig. 4.3B).

Region 2

Quartering the waterpoints in region 2, when waterpoints in the other regions are not manipulated (scenario H), increased the elephant density (Fig. 4.3C), whereas other scenarios of directly manipulating waterpoints in Region 2 (scenarios C, M, S and V) do not significantly affect elephant densities. On the other hand, all the waterpoints reduction scenarios (scenarios A to O) result in a significantly lower vegetation biomass diversity than the current and all the waterpoints increasing scenarios (scenarios P to Z; Fig. 4.3D).

There are indirect effects of reducing the waterpoints in other regions on region 2's elephant density. Closing all the waterpoints in region 3 or 4 (scenarios D and E respectively), or quartering of waterpoints in all regions or in region 3 or 4 (scenarios F, I and J respectively), or halving the waterpoints in region 1, 3 or 4 (scenarios L, N, and O respectively) resulted in an increased elephant density in region 2 compared to region 2's elephant density under the current waterpoints scenario (scenario P; Fig. 4.3C). However, increasing the number of waterpoints in other regions did not have a significant effect on either elephant density or vegetation biomass diversity in region 2 as compared to the current scenario (Fig. 4.3C; Fig. 4.3D).

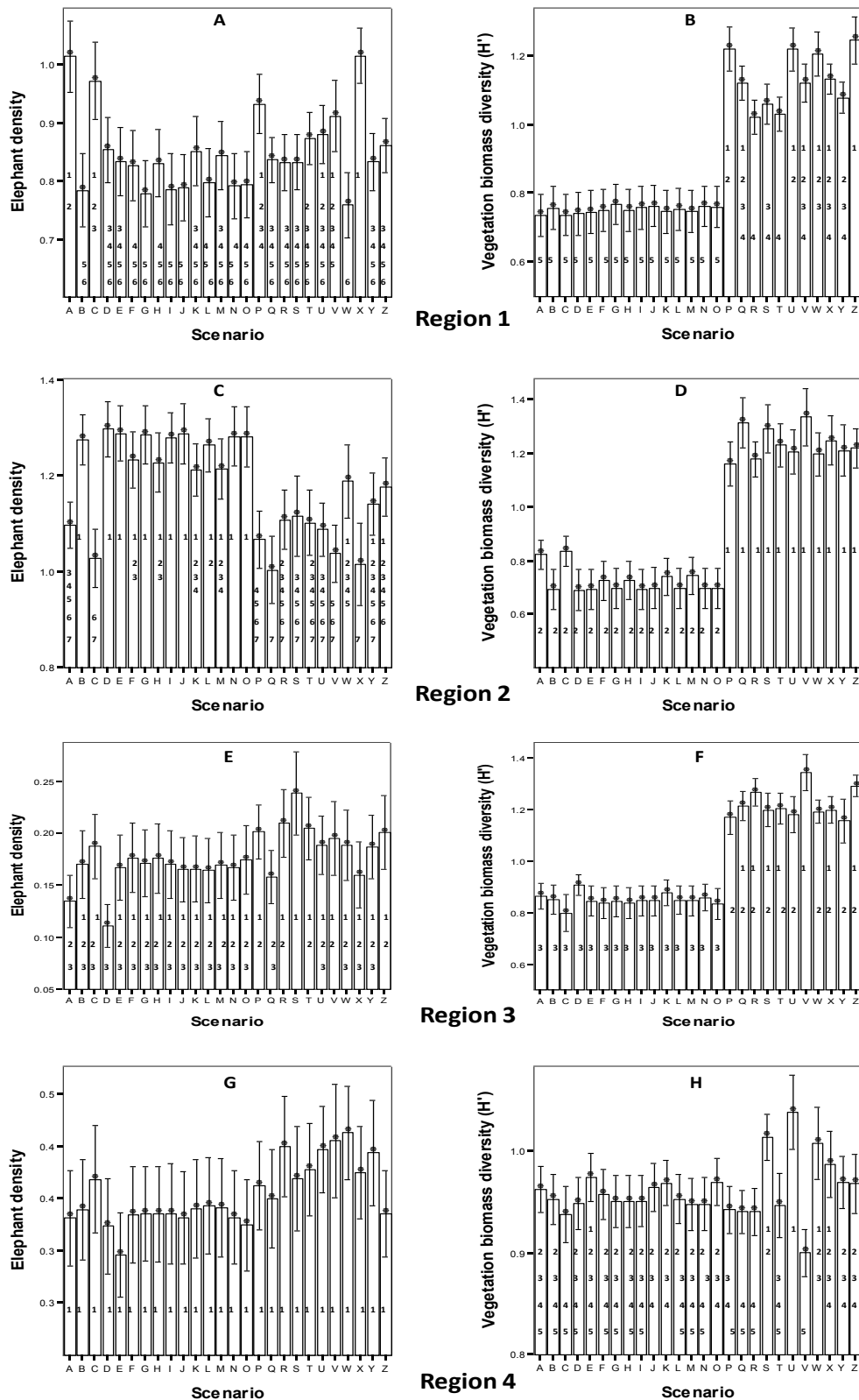


Fig. 4.3: Mean ($\pm 95\%$ CL) elephant density (number/km²) and vegetation biomass diversity (Shannon Wiener Index - H') in regions 1 to 4 under different artificial waterpoints scenarios. For scenarios A-Z, see Table 4.2. Numbers in bars are subgroups based on multiple comparison tests.

Region 3

Closing of all the waterpoints in region 3 (scenario D) resulted in a reduced elephant density than the current waterpoints scenario (scenario P; Fig. 4.3E). However, all the other waterpoints scenarios did not have a significant effect on elephant density (Fig. 4.3E). Just like in regions 1 and 2, all the waterpoints reduction scenarios resulted in decreased vegetation biomass diversity compared to the current and the waterpoints increasing scenarios (Fig. 4.3F). However, the scenario where the waterpoints were quadrupled in all regions (scenario V) resulted in a higher vegetation biomass diversity than under the current waterpoints condition (scenario P; Fig. 4.3F).

Region 4

In region 4, elephant density was not significantly affected by a change in number of waterpoints (Fig. 4.3G). Doubling the waterpoints in region 4 (scenario U) resulted in increased vegetation biomass diversity. In addition, when waterpoints were doubled in region 2 (scenario S), the vegetation diversity increased (Fig. 4.3H). Quadrupling the number of waterpoints in all regions (scenario V) significantly reduced the vegetation diversity in region 4 (Fig. 4.3H).

4.3.2 Elephant impact on vegetation biomass diversity

When we tested for the relationship of elephant density on vegetation biomass in region 1 for all the scenarios, there was a significant negative linear relationship under the absence of waterpoints ($R^2_{adj}=0.11$; $n=30$; $B=-0.375$, $p<0.05$; $t=-2.129$; Fig. 4.4A).

Region 1

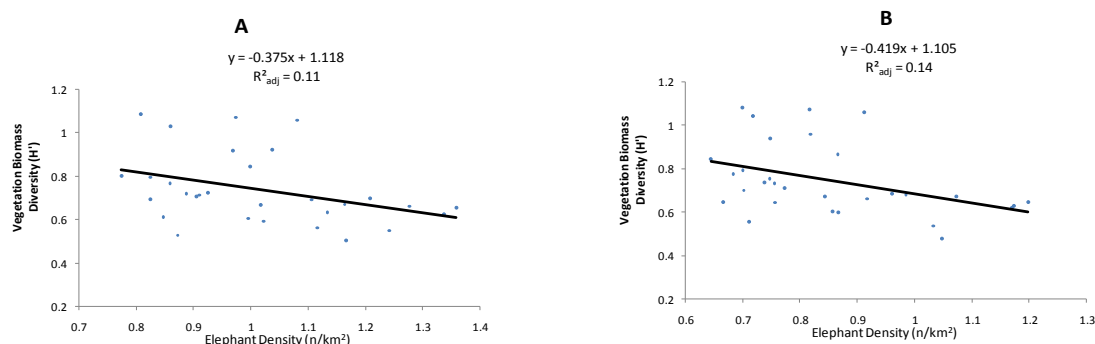


Fig. 4.4: Relationship between elephant density (number/km²) and vegetation biomass diversity (Shannon Wiener - H') in Region 1 under the absence of artificial waterpoints (A) and halving the number of existing waterpoints (B) scenarios.

Also when the number of waterpoints were halved the vegetation biomass diversity decreased ($R^2_{adj}=0.14$; $n=30$; $B=-0.418$, $p<0.05$; $t=-2.354$; Fig. 4.4B). The other scenarios did not yield statistical significant relationships.

Region 2

For region 2, the relationship between elephant density and vegetation biomass diversity was significantly linear and negative under the following scenarios: no artificial waterpoints ($R^2_{adj}=0.44$; $n=30$; $B=-0.79$, $p<0.01$; $t=-4.882$; Fig. 4.5A), quartering waterpoints ($R^2_{adj}=0.65$; $n=30$; $B=-0.987$, $p<0.01$; $t=-7.402$; Fig. 4.5B), and halving waterpoints ($R^2_{adj}=0.58$; $n=30$; $B=-0.994$, $p<0.01$; $t=-6.436$; Fig. 4.5C).

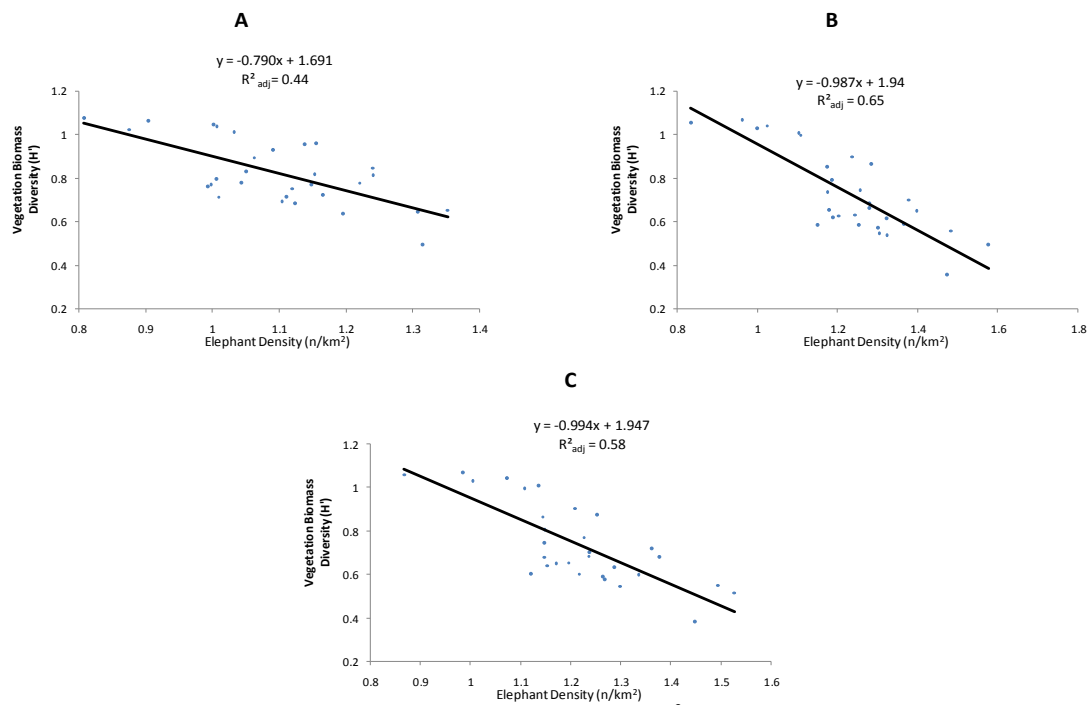


Fig. 4.5: Relationship between elephant density (n/km^2) and vegetation biomass diversity (Shannon Wiener - H') in Region 2 under the no artificial waterpoints (A), quartering (B) and halving waterpoints (C) scenarios.

Region 3

For region 3, the significant relationship between elephant density and vegetation biomass was quadratic for the scenarios without artificial waterpoints ($R^2_{adj}=0.17$; $n=30$; quadratic term: $t=-12.154$, $p<0.05$; linear term: $t=3.327$, $p<0.01$; Fig. 4.6A), under doubling of waterpoints ($R^2_{adj}=0.17$; $n=30$; quadratic term: $t=-2.433$, $p<0.05$; linear term: $t=2.077$, $p<0.05$; Fig. 4.6E), and under quadrupling the number of waterpoints ($R^2_{adj}=0.15$; $n=30$; quadratic term: $t=-2.302$, $p<0.05$; linear term: $t=1.935$, $p<0.05$; Fig. 4.6F). However, the significant relationship between elephant density and vegetation biomass was linear and positive for

quartering the waterpoints ($R^2_{adj}=0.27$; $n=30$; $B=0.98$, $p<0.01$; $t=3.426$; Fig. 4.6B), halving the waterpoints ($R^2_{adj}=0.28$; $n=30$; $B=0.872$, $p<0.01$; $t=3.504$; Fig. 4.6C), and for the current waterpoints conditions ($R^2_{adj}=0.1$; $n=30$; $B=0.892$, $p=0.05$; $t=2.017$; Fig. 4.6D).

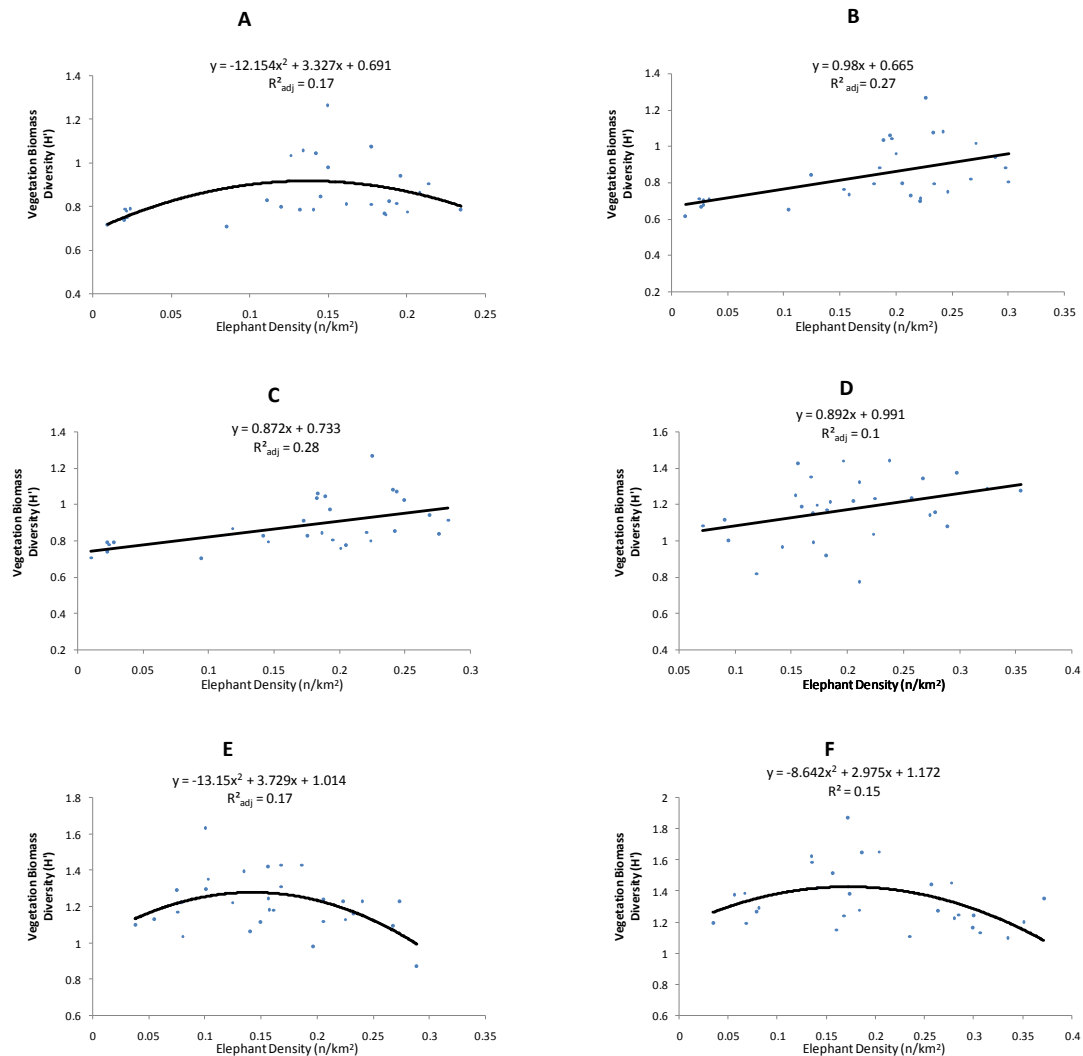


Fig. 4.6: Relationship between elephant density (n/km²) and vegetation biomass diversity (Shannon Wiener - H') in Region 3 under the no artificial waterpoints (A), quartering (B), halving (C), current conditions (D), doubling (E) and quadrupling (F) of waterpoints scenarios.

Region 4

When we tested for the relationship between elephant density and vegetation biomass diversity for region 4, the linear relationship was significant and positive for the scenarios: no artificial waterpoints ($R^2_{adj}=0.1$; $n=30$; $B=0.181$; $t=2.08$; $p<0.05$; Fig. 4.7A), quartering waterpoints ($R^2_{adj}=0.17$; $n=30$; $B=0.24$; $t=2.637$; $p<0.05$; 7B), and halving waterpoints ($R^2_{adj}=0.1$; $n=30$; $B=0.17$; $t=2.027$; $p=0.05$; Fig. 4.7C). However, the relationship was linear

for the scenarios: current waterpoints ($R^2_{adj}=0.26$; $n=30$; $B=-0.294$; $t=-3.341$; $p<0.01$; Fig. 4.7D) and quadrupling waterpoints ($R^2_{adj}=0.13$; $n=30$; $B=-0.17$; $t=-2.324$; $p<0.05$; Fig. 4.7E).

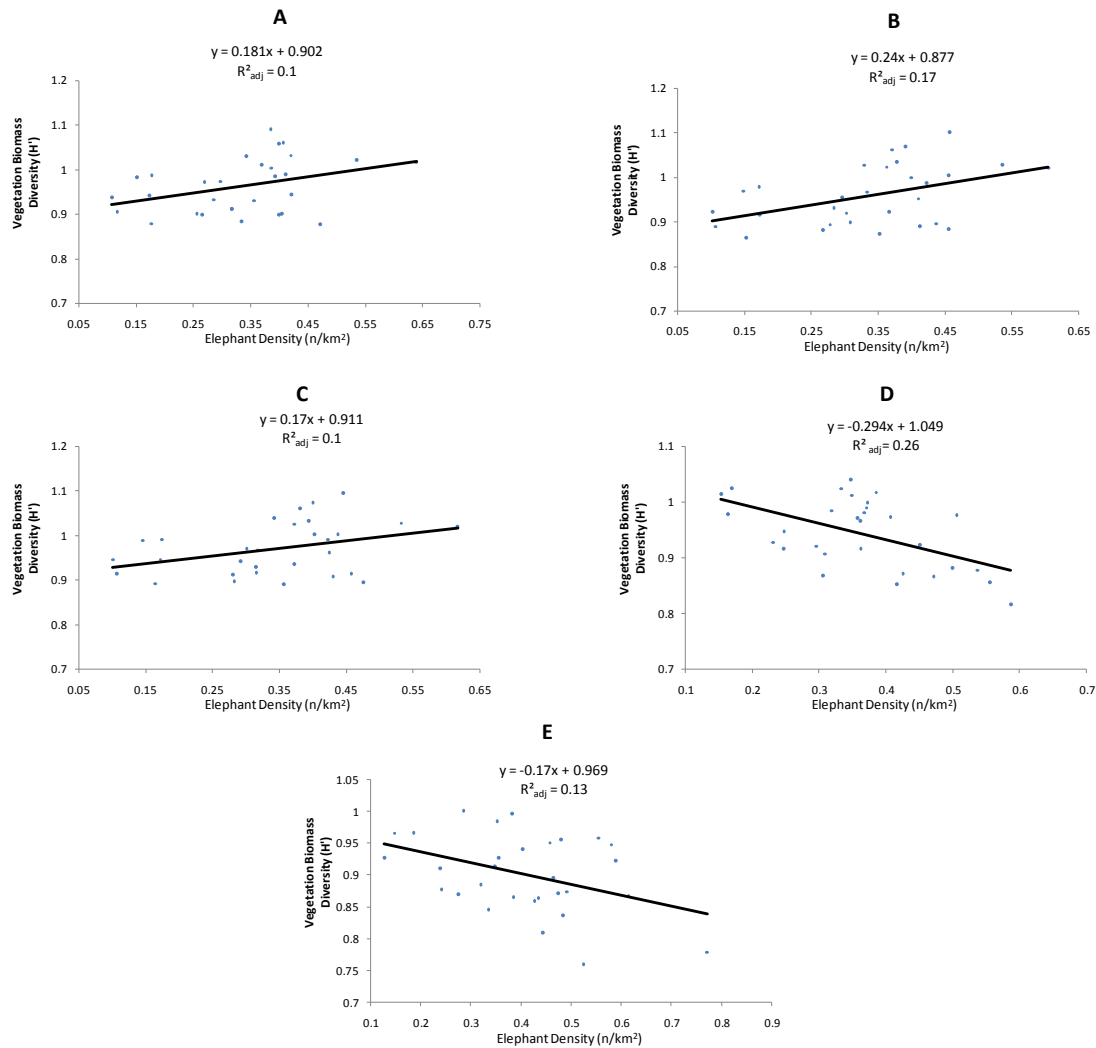


Fig. 4.7: Relationship between elephant density (n/km^2) and vegetation biomass diversity (Shannon Wiener - H') in Region 4 under the no artificial waterpoints (A), quartering (B), halving (C), current conditions (D) and quadrupling (E) of waterpoints scenarios

4.4 Discussion

If we only consider the scenarios that had a significant linear relationship between elephant density and vegetation biomass diversity: regions 1 and 2 had a negative relationship whilst region 3 had a positive relationship (Fig. 4.8). However, for region 4, there was a positive relationship for the waterpoints reduction scenarios, whilst for the current and increasing waterpoints scenario, the relationship between elephant density and vegetation biomass diversity was negative (Fig. 4.8). For the cases where the relationship between elephant density and vegetation biomass diversity was negative, in regions 1 and 2, this suggests a

trade-off between the two objectives, increasing elephant density decreased the vegetation biomass diversity and vice versa (Parker, 1983; Fritz et al., 2002). Elephant density is representing an economic objective whilst vegetation biomass diversity is representing an ecological objective.

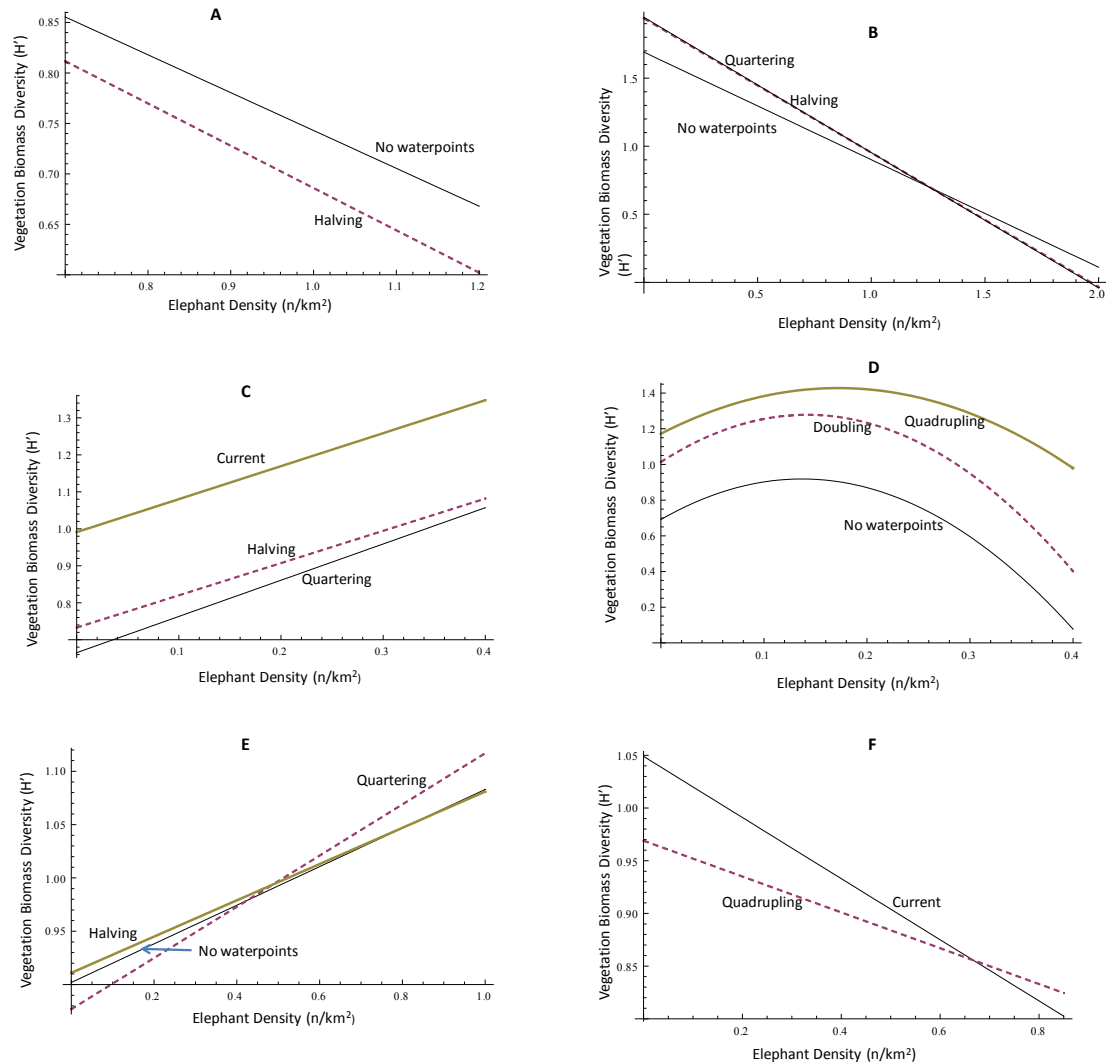


Fig. 4.8: Comparison of relationships between elephant density and vegetation biomass diversity that were significant in: region 1 (A); region 2 (B); linear relationship in region 3 (C); quadratic relationship in region 3 (D); positive relationship in region 4 (E); negative relationship in region 4 (F).

The results in region 1 suggest that, adding waterpoints reduces the vegetation biomass diversity, since the graph of the scenario without waterpoints is above that of halving the current waterpoints density (Fig. 4.8A). So the best strategy in this region, from a vegetation diversity point of view, would be to close all waterpoints. This could be justified given that region 1 has the highest density of perennial rivers and receives the highest amount of rainfall in comparison to the other 3 regions (Table 4.1). This would support the policy of closing

waterpoints by the KNP management to promote diversity (Pienaar, 1985; Pienaar et al. 1997).

The model suggests that adding waterpoints in Region 2 at lower elephant densities (lower than 1.25 elephants/km²) would increase the vegetation biomass diversity whereas at higher densities (higher than 1.25 elephants/km²) would decrease the vegetation biomass diversity (Fig. 4.8B). However, quartering and halving of the waterpoint density in this region have almost similar effects (Fig. 4.8B). So for regions 2, adding waterpoints would be a good strategy when elephant densities are very low. However, for higher elephant densities, addition of waterpoints could result in a drop of the vegetation biomass diversity. Elephants at low densities cause very low disturbances to the vegetation and hence results in low vegetation biomass diversity (Kerley et al 2008; Pringle 2008). So when wildlife managers add waterpoints at such low elephant densities in the region, elephants would be attracted to the region and through their disturbances result in higher vegetation biomass diversity. However, in a case where the elephant density in the region is already high, adding more waterpoints could nonetheless attract additional elephants which would cause excessive disturbance to the ecosystem of the region but now at the expense of the vegetation biomass diversity. Adaptive management could be prescribed in this region, that is, waterpoints are not closed completely but they could be switched on and off depending on the elephant density.

As for region 4, when the waterpoints were reduced, the relationship between elephant density and vegetation biomass diversity became positive (Fig. 4.8E), whereas it shifted to negative, under the addition of waterpoints (current and quadrupling scenarios; Fig. 4.8F). This shows that although waterpoints could be used as a strategy to increase the vegetation biomass diversity and the elephant density in region 4, excessive use of this tool would result in a trade-off between the two objectives. When waterpoints are added to region 4, elephant are attracted to this region, causing an increase in vegetation biomass diversity (Kerley et al 2008; Pringle 2008). However, adding even more waterpoints would attract more elephants, which could lead to excessive damage to the vegetation, and thereby result in a loss of the vegetation biomass diversity.

The elephant density in region 3 had a hump-shaped effect on the vegetation biomass diversity under the no artificial waterpoints, and under the doubling and quadrupling waterpoints scenarios, following our predictions (Fig. 4.8). Our model suggests that the elephant density that results in a maximum vegetation biomass diversity (H') of 0.919 in region 3 under the no artificial waterpoints is 0.136 elephants/km² (Fig. 4.8). This elephant

density is within the range of the current elephant density in the KNP (van Aarde et al., 1999; Slotow et al., 2005; Gough and Kerley, 2006). However, under the doubling scenario, the elephant density that resulted in the maximum vegetation biomass diversity (H') of 1.278, was 0.142 elephants/km² (Fig. 4.8). Under the quadrupling waterpoints scenario, the vegetation biomass diversity increased to 1.42, under an elephant density of 0.172 elephants/km². When both the quadratic and the linear relationships between the elephant density and the vegetation biomass diversity in Region 3 are considered, the model suggests that adding more waterpoints in region 3 results in higher vegetation biomass diversity.

The model supports the policy taken by KNP management to shut down some of the waterpoints, a decision taken in 1997 (Pienaar, 1985; Pienaar et al. 1997) but for region 3, the model suggests that additional waterpoints are necessary. Thus different management options seem to be required in different zones. It should be noted that the elephant population size in this model was fixed and that results might differ if the population dynamics of elephant were included, but the qualitative argument that we have raised in this paper would remain the same.

Only the simulation results from region 3 supported the intermediate disturbance hypothesis. This is in agreement with findings of Holdo et al. (2009), who found that the spatial heterogeneity in tree cover varied unimodally with elephant population density in a modelling study of the Serengeti. Results from the other regions do not support the intermediate hypothesis, however, these regions could be at different phases of the unimodal relationship (Wiens 1997; Whyte et al., 1999). Elephants are important agents of disturbance and as such are able to change species richness (Kerley et al 2008; Pringle 2008). Elephants, can modify the ecosystem through direct or indirect impacts on the availability of resources to other organisms and affect habitat heterogeneity, and thereby trigger cascading effect in the system (Jones et al., 1997; Kerley et al., 2008; Valeix et al., 2008). This implies that relatively low levels of diversity are observed at extreme levels of both low and high elephant densities. However, at intermediate densities, the vegetation biomass diversity is maximal. In this case elephants are the agents of the disturbance. So as to preserve vegetation diversity, conservation efforts must explicitly consider the disturbance process (Hobbs and Huenneke, 1992) brought about by differences in elephant density, to meet both ecological and economic objectives.

The results showed that waterpoints have different effects on both elephant density and vegetation biomass diversity in the different regions. Closing or construction of waterpoints in

one region does not necessarily translate into higher or lower elephant densities respectively, as this depends on the habitat suitability index of the region in comparison to the other regions. So, although waterpoints management could be a tool that could be used to manipulate the spatiotemporal distribution of water-dependent species such as elephants (Chamaillé-Jammes et al., 2007, Kalwij et al., 2010), the impact should be analysed on the largest possible spatial scale (e.g., the park in this case). However, if the analysis is carried out at a regional scale, and regions are not physical separated, then using waterpoints to influence elephant distribution in a particular region might actually give opposite and unintended results, as we have demonstrated in this study. This is particularly essential to larger national parks like the KNP, where the emphasis is on managing elephants on a regional scale (Whyte et al., 1999). It also applies to individual landowners who share elephants with other landowners in a large game reserve where physical barriers between the multiple land properties have been removed (Krug, 2001; APNR, 2005; Druce et al. 2009). So before the managers decide to open or close waterpoints in one region to attract or repel elephants respectively, they need to understand how the dynamics between elephant and vegetation in one region would be affected by the other regions and vice versa. However, we acknowledge that this is a simplified system that shows that under these constructs elephants seem to be able to influence vegetation diversity. Hence, further work is required, ideally combined with experimental studies to test these predictions in the field.

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Optimization of net returns from wildlife consumptive and non-consumptive uses by game reserve management

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Abstract

Hunting is a wildlife consumptive use whilst tourism is non-consumptive. Landowners or game reserve management are often faced with the decision whether to undertake consumptive and/or non-consumptive use on their properties. We use a theoretical model to examine cases where the game reserve management decides on how much land to devote to hunting and how much to tourism or a combination of both. We use three scenarios to analyze the effects of these decisions: (1) the hunting area is separated from the tourism area but wildlife is shared; (2) hunting and tourism co-exist in the same area; and (3) hunting and tourism area separated by a fence. The key aspect to emerge from our work is that consumptive and non-consumptive uses are not mutually exclusive, but that careful planning is needed to ensure that multiple management objectives can be met. Further, our results indicate that the two uses can be undertaken in the same contiguous area. Whether they are spatially or temporally separated depends on the magnitude of the consumptive use. Where the consumptive use is not dominating, the two are compatible in the same shared area.

Keywords: hunting; tourism; land management; land-use; South Africa; wildlife economics

5.1 Introduction

In countries like South Africa, Zimbabwe, and Namibia some control has been given to private landowners over the use of wildlife (Child, 2000; Muir-Leresche and Nelson, 2000). For example, in South Africa, privately protected areas are more numerous and more extensive in area than public nature parks (Watkins et al., 1996). There are over 5000 game ranches and about 4000 mixed game and livestock ranches in South Africa, and together they

cover about 13% of the country's total land area, compared with 5.8% for all officially declared conservation areas (Hearne et al., 2000; Krug, 2001; ABSA, 2003). Fox and du Plessis (2000) found that 80% of nature conservation in South Africa takes place on private land, while Bothma (2002) showed that there was an increase of 2.5% between 1998 and 1999 in the area of land utilized for game farming.

Wildlife has multiple uses that generate revenues for the betterment of the landowners. For example, wildlife is used for tourism, for photographic safaris or wildlife viewing (non-consumptive uses), or for hunting for trophies or for selling meat and/or hide (both consumptive uses). Landowners can either concentrate on consumptive or become involved in non-consumptive use or a mixture of both (Aylward, 2003; de Boer et al 2007).

Game viewing produces a number of revenue sources for the game reserve management, such as entrance fees, accommodation charges, sales from restaurants and shops, trails, rides and tour income, and other revenues, such as permits and fines (Aylward, 2003). Game viewing is the major reason why tourists visit game reserves (e.g., Vial 1996; Langholz and Kerley, 2006). In addition, it is also mentioned that the higher the density or probability of seeing the big five (elephant, buffalo, rhino, lion and leopard), the higher the number of tourists (Lindsey et al., 2007; Okello, et al., 2008). Lindberg et al. (2003) found a significant positive relationship between the number of wildlife species and property size in studies carried out on a group of privately protected areas in KwaZulu-Natal province, South Africa.

Hunting plays a major role in generating income for landowners or game reserve managers involved in consumptive use (Barnett and Patterson, 2006). In South Africa how much these landowners can hunt is based on quotas set by themselves under the auspices of provincial nature conservation authorities generally determined by the availability of species in terms of densities (APNR, 2005; Barnett and Patterson, 2006). Like other forms of wildlife control, hunting is under increasing scrutiny by various interest groups (Wittman et al., 1998; Campbell et al., 2009). There has been lobbying against hunting practices by several animal-welfare and rights groups, especially for species that hold special appeal to humans in terms of their charisma and size, like elephant (Peterson, 2004; Burke et al., 2008; Lötter et al., 2008; Kettles and Slotow, 2009). Such management interventions that are perceived negatively also have the potential to reduce public appeal, and hence tourism (Burke et al., 2008; Lötter et al., 2008). Moreover, hunting does not only affect the stock in terms of a number of animals removed, but also negatively affects the visibility or level of habituation of the remaining animals (Leader-Williams and Hutton, 2005; Reimer, et al., 2009). This

represents an important concern where hunting and tourism occurs on shared land (Reimer, et al., 2009).

Several analytical models have been built to analyze the interaction between ecological and economic factors in wildlife management. The models have mainly dealt with land competition between wildlife and other land-uses, such as livestock production, in order to meet specific objectives (e.g., Schulz and Skonhøft, 1996; Kinyua et al., 2000; Fleming and Alexander, 2003). Usually these studies came up with different strategies on how to optimize land allocation amongst these competing land-uses, which are usually wildlife and livestock. There are few analytical models that investigate competition between hunting and tourism.

There are three main types of private landownership involved in wildlife management: game ranching, private conservancy, and private nature reserves also known as game reserves. Game ranching involves the raising of wildlife animals for a variety of products, including meat, hides, feathers and antlers. Private conservancy consists of a group of wildlife ranches where the landowners pool natural and financial resources to conserve wildlife, and they usually operate through a joint management committee (Krug, 2001). Management objectives of game reserves vary from preservation to sustainable use of wildlife. Wildlife use involves obtaining profits from both consumptive and non-consumptive uses (Huffaker, 1993; Hearne and McKenzie, 2000; ABSA, 2003).

In South Africa, some game reserves have formed associations whereby adjoining landowners pooled their resources to create larger units but still retained individual ownership of the land. One of the oldest private reserves in the north-eastern lowveld of South Africa, is the Timbavati Private Nature Reserve which was formed in the 1950s (Krug, 2001). This reserve is part of the Associated Private Nature Reserves (APNR). The APNR consists of Timbavati, Klaserie, Umbabat and Balule Private Nature Reserves, each of which is made up of smaller, individually owned, land parcels. The reserves have land totalling approximately 1850 km². Each reserve comprises land belonging to a number of farms and owners. In addition, each reserve has its own management structure in place. Management runs the reserve on behalf of the landowners. Landowners in each reserve contractually devote their land to wildlife conservation, so they can not convert to other land-use practices other than wildlife management.

The landowners in a game reserve can be classified into two groups with different objectives: commercial and non-commercial. Commercial landowners are involved in tourism and have

lodges on their properties. These landowners provide accommodations and associated tourism services (Eagles, 2001). On the other hand, non-commercial landowners have their land for the owners' recreation, aesthetic appeal, place of retirement, land investment or to contribute to wildlife conservation. The landowners with non-commercial objectives usually have other sources of income. Hunting by the game reserve management is usually practised throughout the reserve. Table 5.1 shows the classification and number of land properties for the APNR and associated percentage of landsize in each category.

Table 5.1: Classification of land properties in the Associated Private Nature Reserves according to commercial and non-commercial objectives.

| <i>Game Reserve Name</i> | <i>Commercial (% area)</i> | <i>Non-commercial (% area)</i> |
|--------------------------|----------------------------|--------------------------------|
| Timbavati | 9 (31) | 39 (69) |
| Klaserie | 5 (11) | 115 (89) |
| Umbabat | 1 (6) | 18 (94) |
| Balule | 15 (33) | 168 (67) |
| Total | 30 (22) | 340 (78) |

Economists' models of renewable resource exploitation have utilized various abstractions for both analytical tractability and to focus attention on variables of particular interest in the policy process (Sanchirico and Wilen, 1999). We investigate how the game reserve management could decide on how much land to devote to hunting and how much to tourism or a combination of both. We use these three scenarios to analyze the effects of these decisions: (1) the hunting area is separated from the tourism area but wildlife is shared; (2) hunting and tourism co-exist in the same area; and (3) the hunting area and tourism area are separated by a fence.

5.2 Models

We modelled the management for a game reserve, which would like to decide how much land to be used for wildlife non-consumptive and/or consumptive uses. The objective of the management is to maximize its net revenue from a combination of both wildlife consumptive and non-consumptive uses. We analyze three scenarios which are presented in Fig. 5.1. To substantiate some of the findings in the theoretical parts, we carry out a brief numerical analyses. Although we have made several simplifications to the model, the findings are useful to wildlife management since the application of simple model-based strategies is a good starting point for complex management decisions (Rosenzweig, 1973; Starfield, 1997). We adopt the notation where the subscript denotes the derivative with respect to that variable.

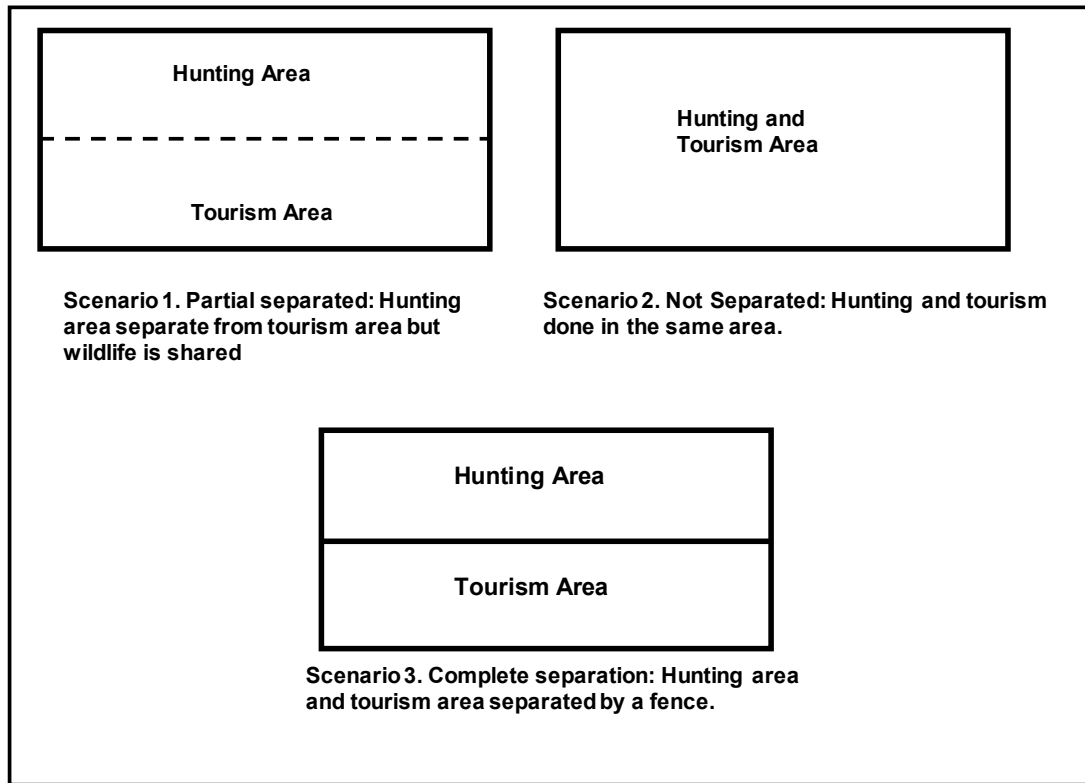


Fig. 5.1: Schematic presentation of wildlife management cooperation scenarios between different land-use activities.

5.2.1 Scenario 1: Hunting and tourism area partially separated

In Scenario 1 we analyze the case where the game reserve management has separated the hunting area from the tourism area, but there is no barrier between the two areas, so wildlife roams freely between the two areas. The proportion of land under hunting is denoted by A and that under tourism is denoted by L . Land base is fixed so the relationship between hunting and tourism land-uses is as follows:

$$L + A = 1 \quad (1)$$

We assume that there is one wildlife species. The dynamics for this species is given by:

$$\frac{dX}{dt} = h(X) - q \quad (2)$$

where: X denotes the wildlife stock at time t (for notational convenience, we suppress the time notation, but time should be understood to be implicit in all variables); $h(X)$ is the

growth function of wildlife which is dependent on wildlife stock ($h(X)$) is a logistic function with the formulation $h(X) = \tau X[1 - X/K]$, where τ is the wildlife's intrinsic growth rate and K is the carrying capacity). From this formulation: $h > 0$, $h_X > 0$ for $0 < X < \bar{X}$ and $h_X < 0$ for $X > \bar{X}$ (\bar{X} is the value of X when $h_X = 0$); $h_{XX} < 0$. We postulate that q is the wildlife offtake. For simplicity we ignore age or sex structure since we believe that this will not affect the general findings of the model.

The game reserve management derives its income from selling hunting licences and tourism. Its profit function (ε) is given by:

$$\varepsilon = b(X, A, p)q + Tf(X, L) \quad (3)$$

Where: $b(X, A, p)$ is the unit profit function of hunting wildlife which depends on the wildlife stock, the area under hunting activity; p is the given market price of wildlife offtake. To understand the various signs of the partial derivatives given below, we will break down and specify the unit profit function. The unit profit function contains both unit income and unit costs. The price per offtake is fixed which would give the income formulation as pq . On the other hand, the assumption is that costs decrease as either or both wildlife increases or land increases – due to economies of scale with the formulation for the unit costs as $z + c1 * e^{-\sigma X} + c2 * e^{-\phi A}$ (where z , σ and ϕ are constants) which means costs associated with size of wildlife ($c1 * e^{-\sigma X}$) and those associated to size of land ($c2 * e^{-\phi A}$) will both decrease to some asymptote. This would then give the complete formulation for the unit profit function of hunting as $b(X, A, p) = pq - [z + c1 * e^{-\sigma X} + c2 * e^{-\phi A}]$. From this formulation: $b_A(X, A, p) > 0$, and $b_X(X, A, p) > 0$ implying that there are economies of scale by having larger hunting area and a larger wildlife stock respectively (Benson, 2003). From the formulation, $b_{Ap}(X, A, p) = 0$ and $b_{Xp}(X, A, p) = 0$, because the price of wildlife offtake does not affect the hunting area and the wildlife stock directly. Other assumptions made on the unit profit function: in Scenarios 1 and 2 scenarios, $b_{XA}(X, A, p) = 0$ because in those two scenarios, wildlife is shared, so neither hunting nor tourism is influencing the wildlife stock directly. Further assumptions which conforms to the formulations above are: $b_{AA}(X, A, p) < 0$ and $b_{XX}(X, A, p) < 0$, which signifies that the increase in profit due to a rise in land or wildlife stock sizes will do so at a decreasing rate.

$f(X, L)$ is the net tourism profit which depends on the wildlife stock and amount of land under tourism; T is a tourism parameter. To understand the various signs of the partial derivatives given below and to validate our assumptions, we specify the net tourism profit function as $(aX/(b+X))(cL/(d+L))$; where a , b , c , and d are parameters. From this specification, $f_X(X, L) > 0$, implying that with increasing numbers of wildlife, the profit from tourism would rise as more tourists would be attracted (Lindsey et al., 2007; Okello, et al., 2008). We also assume $f_L(X, L) > 0$, meaning that larger areas under tourism would attract more tourists, since it would give them a larger area for wildlife viewing (Hearne and McKenzie, 2000). We base these assumptions on the observation that the larger the land property, the higher the diversity of wildlife species, and the higher the landscape diversity for tourists (Aylward, 2003; H. Killian, personal communication). We also assume that $f_{XL}(X, L) > 0$, implying multiplicative effects of land and wildlife stock on each other. We also assume that $f_{XX}(X, L) < 0$ and $f_{LL}(X, L) < 0$, implying a rise in profit due to an increase in wildlife or land stock, respectively, would do so at a decreasing rate. For model simplicity, in this scenario we assume that there is uniform mixing of wildlife between hunting and tourism area (Sanchirico and Wilen, 1999).

We assume that the objective of the game reserve management is to maximize present value of net income from hunting and tourism, as given by:

$$Max_{q, A} PV = \int_0^{\infty} [b(X, A, p)q + Tf(X, L)]e^{-rt} dt \quad (4)$$

where r is the economic discount rate. We assume that $r > 0$.

Maximization takes place subject to equations (1) and (2), so that the current value Hamiltonian (Clark, 2005) for this problem is as follows:

$$H = b(X, A, p)q + Tf(X, L) + \mu[h(X) - q] \quad (5)$$

with X as the state variable; A and q the control variables; and μ the costate variable for the wildlife stock.

It can be shown (Clark and Munro, 1975; Conrad and Clark, 1987; Clark, 2005; Appendix 5) that the optimal harvest strategy will be the one that brings wildlife biomass (X), to the steady state level in this scenario, X_1^* , given by the modified Golden rules (equations (6) and (7)).

$$r = h_x(X) + \frac{b_x(X, A, p)h(X)}{b(X, A, p)} + \frac{Tf_x(X, L)}{b(X, A, p)} \quad (6)$$

$$Tf_L(X, L) = h(X)b_A(X, A, p) \quad (7)$$

The right hand side of equation (6) is the wildlife's own interest rate and comprises the following terms: (i) marginal productivity, (ii) marginal hunting profitability effect : unit profit ratio, and (iii) marginal effect of wildlife on tourism net revenue : unit profit ratio.

Under this scenario the optimum profit made by the game reserve, ε_1^* , is given by:

$$\varepsilon_1^* = b(X_1^*, A_1^*, p)q_1^* + Tf(X_1^*, L_1^*) \quad (8)$$

Implications of Scenario 1

Equation (7) gives the equilibrium condition between demand and supply for land (Fig. 5.2). The curve $Tf_L(X, L)$ emanating from LHS of equation (7) could be termed the demand schedule of land for tourism. On the other hand, the curve, $h(X)b_A(X, A, p)$ in Fig. 5.2, emanating from RHS of equation (7) constitutes the demand function of land for hunting. The intersection of these two curves is the amount of land that should be devoted to hunting, A , (hence, the remainder would be for tourism, L), given the optimum opportunity cost of land, OP on the vertical axis. So the right hand side of equation (7) gives the demand schedule of land for hunting, whilst the left hand side constitutes the demand schedule of land for tourism (Schulz and Skonhofs, 1996).

It is apparent that the curve representing the marginal benefit of land due to tourism, will shift outwards if tourism parameter, T , thereby increase land under tourism at the expense of land under hunting.

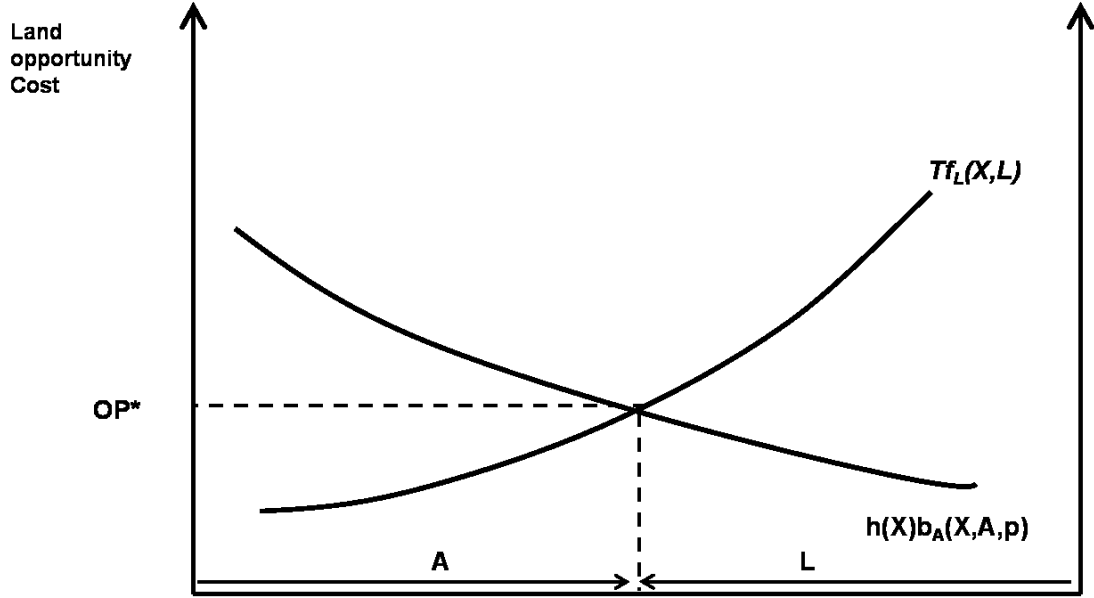


Fig. 5.2: Theoretical equilibrium condition for land. The curve $Tf_L(X,L)$ is the demand of land for tourism. The curve, $h(X)b_A(X,A,p)$, is the demand of land for hunting. The intersection of these two curves is the amount of land that should be devoted to hunting, A, and to tourism, L, and the optimum opportunity cost of land, OP^* on the vertical axis. The relationship $L+A=1$ holds. Compare with Fig. 5.6

We let η be the hunting opportunity cost of land, therefore:

$$\eta = h(X)b(X, A, p) \quad (9)$$

We then calculate the total derivate of η , $d\eta$.

$$d\eta = [h(X)b_{AP}(X, A, p)]dp + [h(X)b_{AA}(X, A, p)]dA + [h_X(X)b_A(X, A, p) + h(X)b_{XA}(X, A, p)]dX \quad (10)$$

As indicated by equation (10), an increase in wildlife optimal stock has two effects, namely change in unit net profit from hunting, $h_X(X)b_A(X, A, p)$ and the effect of wildlife stock growth changes from land, $h(X)b_{XA}(X, A, p)$. From the assumptions made, $h_X(X)$ could either be negative or positive and $b_{XA}(X, A, p) = 0$. This implies that an increased optimal stock could result in either a negative or positive shift in the demand for hunting land (Schulz and Skonhoft, 1996).

The marginal effect of wildlife on tourism's net revenue contributes to keeping the stocks larger than would have occurred if the stocks were only kept for hunting (Equation (6)). Rearranging equation (6) would give:

$$[r - h_X(X)]b(X, A, p) = b_X h(X) + Tf_X(X, L) \quad (11)$$

Since both terms to the right hand side of equation (11) are positive, this means that X^* , the equilibrium wildlife stock, would be at the point where $r - h_X(X) > 0$ holds, or in other words, the point where the wildlife's marginal productivity is less than the discount rate.

Comparative statics

The process for calculating the comparative statics are given in the Appendix 5. We first explore the effect of permanent changes in the price of wildlife offtake. The effect is given by the following (for convenience, we now use short hand by skipping the functions' dependent variables):

$$\frac{dX^*}{dp} = \frac{[(r - h_X)b_p]b_{AA}h - [-Tf_{XL} + b_A(h_X - r)]b_{Ap}h}{D} \quad (12)$$

The first term of the numerator portrays the asset valuation of the wildlife stock, whilst the second reflects the increased competition for land due to an increased price of wildlife offtake. In the second term, the marginal net revenue from tourism is reflected in our model, showing that, since wildlife is shared between hunting and tourism, whatever happens to the wildlife stock due to an increase in price of wildlife offtake, will affect the revenue streams from tourism (Equation (12)). $dX^*/dp < 0$, so the effects of a price increase on the wildlife stock is unambiguously negative.

$$\frac{dA^*}{dp} = \frac{-[b_X(2h_X - r) + bh_{XX} + b_{XX}h + Tf_{XX}]b_{Ap}h - [(r - h_X)b_p][-Tf_{XL} + b_Ah_X]}{D} \quad (13)$$

Equation (13) shows the effect of a permanent change in price of wildlife offtake on the hunting area. The first term of the numerator represents the effect of substituting land with wildlife biomass, and due to the second order conditions for an optimum (Schulz and Skonhøft, 1996), the effect of the first term of the numerator is positive. The second term in

the numerator reflects the demand for land, due to changes in wildlife stock. The effect of a change in price of wildlife offtake on this second term is ambiguous, it will depend on whether Tf_{XL} is greater or less than $b_A h_X$. If $Tf_{XL} > b_A h_X$, implying that tourism is more competitive than hunting, then $dA^*/dp > 0$. Thus when the price of the wildlife offtake increases, $b_A h$ would increase relative to Tf_{XL} , so more land would be allocated to hunting since it would have marginally become more competitive. However, even if $Tf_{XL} < b_A h_X$ it is not sufficient to result in $dA^*/dp < 0$, the second term must be negative and dominating the first term of the numerator. This could be due to hunting being more competitive than tourism. This would be such that the wildlife offtake price effect, which is positive, would dominate the land effect, which in this case is negative.

$$\frac{dX^*}{dr} = \frac{b_{AA}hb}{D} < 0 \quad (14)$$

The relationship between the optimal wildlife stock and the discount rate is negative (Equation (14)), indicating that if the discount rate increases, the wildlife stock becomes less attractive, and this results in a smaller wildlife stock, since it would now be economically justified to consume and invest somewhere else, where the returns are higher (Clark, 2005).

$$\frac{dA^*}{dr} = \frac{-b[-Tf_{XL} + b_A h_X]}{D} \quad (15)$$

The numerator in equation (15) reflects the demand of land for hunting minus the demand of land for tourism. The effect of the discount rate on hunting area is ambiguous (Equation (15)). It would depend on whether Tf_{XL} is smaller or greater than $b_A h_X$, determining which of the two activities is competitive. If $Tf_{XL} > b_A h_X$ then $\frac{dA^*}{dr} > 0$, which implies that the area allocated to hunting would increase when the interest rate increases. On the other hand if $Tf_{XL} < b_A h_X$, then $\frac{dA^*}{dr} < 0$.

5.2.2 Scenario 2: Hunting and tourism on the same area

In this scenario, we explore the case where the game reserve management is undertaking hunting and tourism on the same area. The Hamiltonian function in this case would be:

$$H = b(X, \Pi, p)q + Tf(X, \Pi) - qS + \mu[h(X) - q] \quad (16)$$

Where S is the unit damage cost caused by hunting on tourism. We assume hunting is perceived negatively by tourists and therefore reduces tourism due to its negative effects on the number of game and its visibility (Hearne and McKenzie, 2000; Bothma, 2002; Burke et al., 2008; Lötter et al., 2008; Reimer, et al., 2009). In addition, Π is the total area of the game reserve.

The optimal harvest strategy will be the one that brings wildlife biomass (X), to the steady state level in this scenario, X_2^* , given by the modified Golden rules (equations (17) and (18)).

$$r = h_x(X) + \frac{b_x(X, \Pi, p)h(X, \Pi)}{b(X, \Pi, p) - S} + \frac{Tf_x(X, \Pi)}{b(X, \Pi, p) - S} \quad (17)$$

$$0 = h(X)b_\Pi(X, \Pi, p) + Tf_\Pi(X, \Pi) \quad (18)$$

If we compare equations (6) in Scenario 1, reflecting the wildlife's own interest rate where the hunting area is separated from the tourism area but wildlife is shared (OIR1 in Fig. 5.3), and equation (17), describing the wildlife's own interest rate in Scenario 2, where hunting and tourism occur in the same area (OIR2 in Fig. 5.3), an interesting observation can be made. The steady state of wildlife biomass in Scenario 1 (X_1^*) is smaller than the optimum biomass in Scenario 2 (X_2^*) (Fig. 5.3). The undertaking of hunting and tourism in the same area comes with a cost. This cost S , reduces the denominator of both the marginal hunting profitability effect and the marginal effect of wildlife on the tourism net revenues, the second and third terms to the right of equation (17) respectively, thus increasing the wildlife's own interest rate. To show the effects of S , on the wildlife's own interest rate, we use a graph. We let OIR($S=0$) and $X_{S=0}^*$, to denote the wildlife's own interest rate and optimum biomass when $S = 0$ respectively, and OIR($S>0$) and $X_{S>0}^*$ to denote the wildlife's own interest rate and optimum biomass when $S > 0$ respectively. The effect of S is that it causes the wildlife's own interest rate to shift outwards from OIR($S=0$) to OIR($S>0$), and that the wildlife biomass increases from $X_{S=0}^*$ to $X_{S>0}^*$ (Fig. 5.4). Hence, S causes an increase in optimal wildlife offtake.

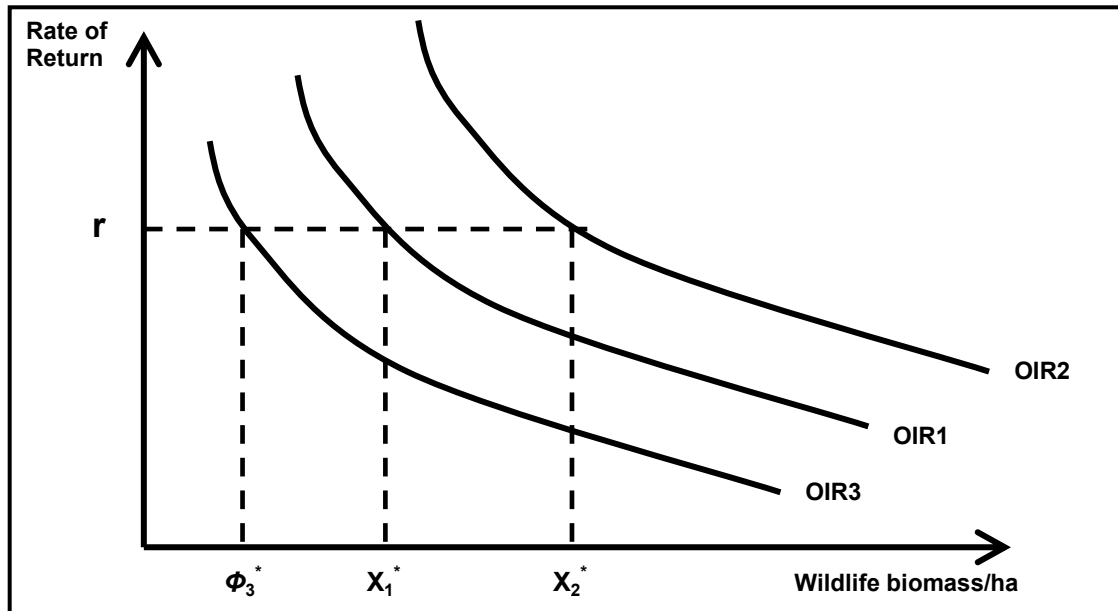


Fig. 5.3: Comparison of optimum wildlife biomass for scenario where hunting and tourism areas are partially separated (X_1^*) not separated (X_2^*) completely separated (Φ_3^*) and their respective own interest rate (OIR) curves. In the scenario where there is complete separation, the wildlife biomass, (Φ_3^*), is only for the hunting section

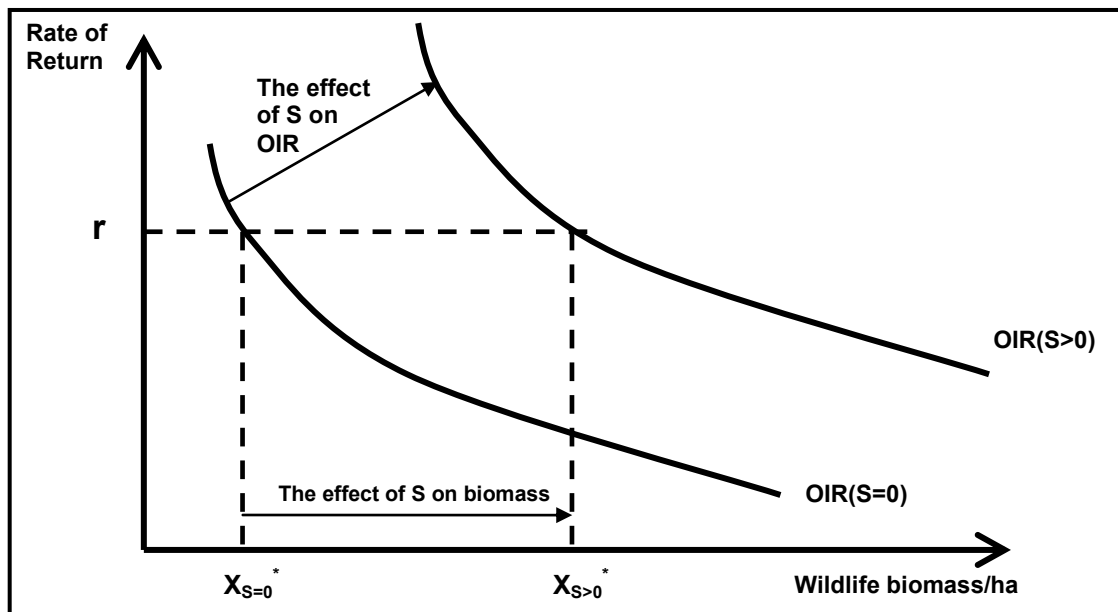


Fig. 5.4: The effect of S , the unit damage cost caused by hunting on tourism, on optimum wildlife biomass. Increasing S causes the wildlife's own interest rate to shift outwards from $OIR(S=0)$ to $OIR(S>0)$. The wildlife biomass shifts from $X_{S=0}^*$ to $X_{S>0}^*$.

Let ω be the hunting and tourism opportunity cost of land.

$$\omega = h(X)b_{\Pi}(X, \Pi, p) + Tf_{\Pi}(X, \Pi) \quad (19)$$

The demand for tourism and hunting land (equation (19)) would be higher than the demand for hunting land alone (equation (9)), because of the increased size of land under both hunting and tourism.

After calculating the total derivative of ω , we deduce that the effects of an increase in wildlife optimal stock has the following effects: an increase in the profit from hunting, $h_X(X, \Pi)b_{\Pi}(X, \Pi, p)$ and the effect of wildlife stock growth changes from land, $h(X, \Pi)b_{\Pi}(X, \Pi, p)$, which are the same as in the case where the hunting area is separated from the tourism area but where wildlife is shared. However, these effects are larger, as hunting is now occurring on a larger area. There is one additional effect due to hunting and tourism occurring in the same area, namely the increase in tourism net profit from the effect of wildlife stock growth from land, $Tf_{X\Pi}(X, \Pi)$.

Under this scenario, the optimum profit made by the game reserve, ε_2^* , is given by:

$$\varepsilon_2^* = b(X_2^*, \Pi^*, p)q_2^* + Tf(X_2^*, \Pi^*) \quad (20)$$

5.2.3 Scenario 3: Hunting and tourism areas separated by a fence

In this scenario, the hunting area is completely separated from the tourism area by a fence, so there are two separate wildlife populations which are not mixing.

The profit function from tourism is now given by $Tf(\Omega, L)$, where Ω is the wildlife biomass in the game reserve section where tourism is being undertaken. On the other hand, the profit function from hunting is given by $b(\Phi, A, p)q$ where Φ is the wildlife biomass in the game reserve section where hunting is practised. The Hamiltonian in this case is given by:

$$H = b(\Phi, A, p) + \mu[h(\Phi, A) - q] + Tf(\Omega, L) \quad (21)$$

In this scenario, the optimal harvest strategy that will result in the wildlife biomass steady state level, Φ_3^* , in the hunting section, is given by the modified Golden rules (equations (22))

and (23)). On the other hand, the optimum wildlife biomass in the tourism section is given by Ω_3^* which we do not model explicitly here.

$$r = h_\Phi(\Phi, A) + \frac{b_\Phi(\Phi, A, p)h(\Phi, A)}{b(\Phi, A, p)} \quad (22)$$

$$Tf_L(\Omega, L) = h(\Phi, A)b_A(\Phi, A, p) + h_A(\Phi, A)b(\Phi, A) \quad (23)$$

Comparison of the wildlife's own interest rate, where there is a fence separating the hunting area from the tourism area (equation (22) depicted by OIR3 curve in Fig. 5.3) to that of the case where hunting area is separated from tourism area but wildlife is shared (equation (6) depicted by OIR1 in Fig. 5.3), shows that the herbivore's own interest rate would shift downwards, implying that the wildlife stocks per unit area in the hunting section of the game reserve would drop (Fig. 5.3) if this section is separated from tourism section. However, in the tourism section, the wildlife per unit area increases.

The right hand of equation (23) shows the demand function for land for hunting. If a comparison of the demand for land for hunting is made between the scenario where hunting and tourism land is separated but wildlife is shared (i.e., Scenario 1) (RHS of equation (7)) and this case where wildlife is not shared (RHS of equation (23)), then the following can be deduced: the demand for land for hunting in the former scenario is higher than in the latter scenario as long as hunting dominates tourism in benefiting from increased wildlife biomass as we will now illustrate.

Fig. 5.5 illustrates the effect of scenarios 1 and 3 on the demand schedules for land for hunting versus tourism. To clearly illustrate the point, we assume that demand schedules of land for tourism and hunting are straight lines: for Scenario 1 represented by TD1 and HD1, respectively, and for Scenario 2, TD3 and HD3, respectively. So we reconstruct Fig. 5.2 and analyze the effects when the demand schedule for hunting is relatively steeper and flatter on opportunity cost of land and the size of allocated land to hunting (conversely to tourism), as we move from Scenario 3 (full separation of hunting and tourism areas) to Scenario 1 (partial separation where only wildlife is shared).

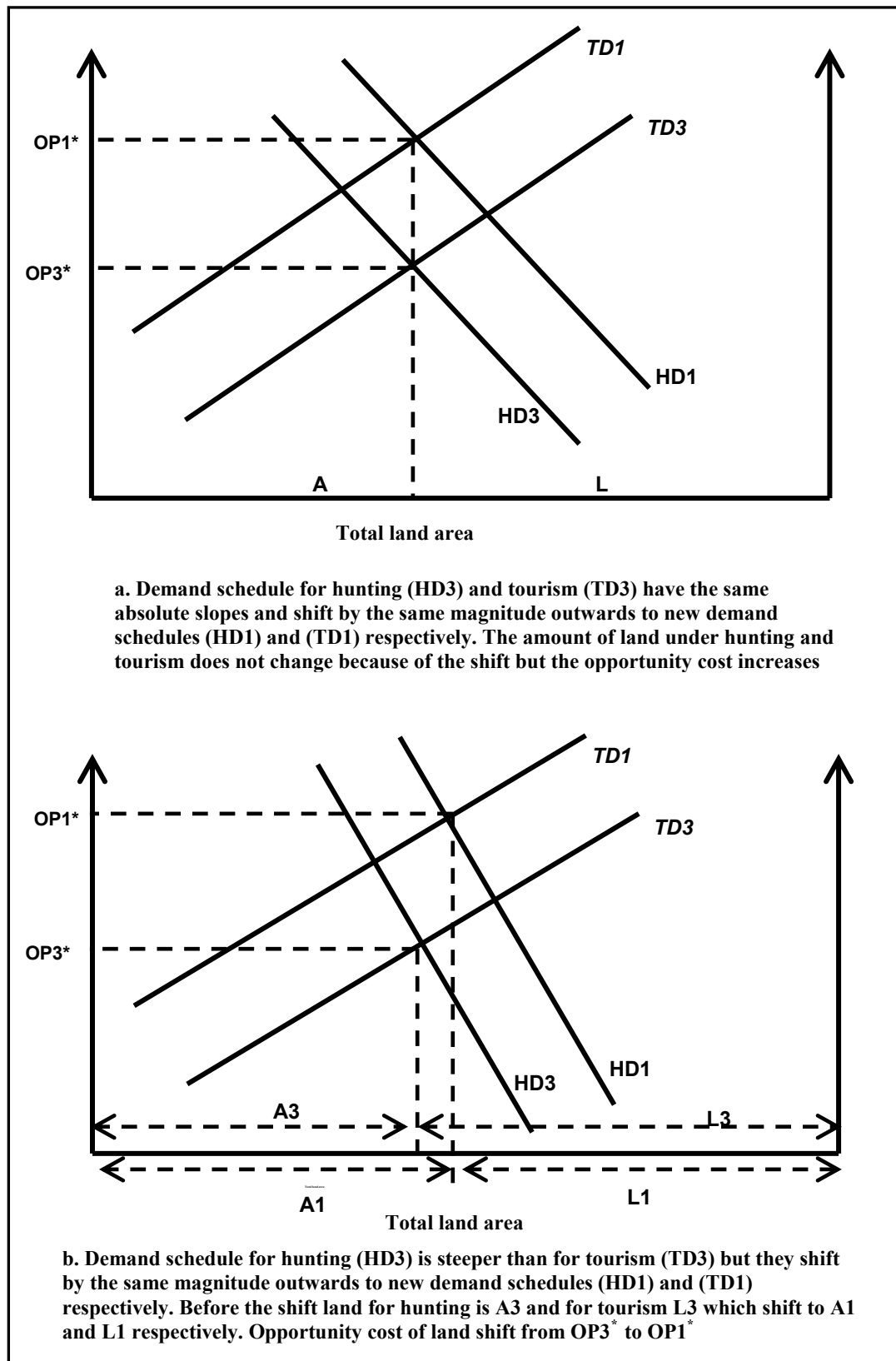


Fig. 5.5: Comparison of the effect of shift upwards of demand schedules for tourism and hunting when the demand schedule for hunting is flatter (a) and steeper (b).

As shown in Fig. 5.5, when the demand schedule for hunting land is steeper (implying that hunting is more dominating), then a shift in both of the demand schedules would favour allocation of more land to hunting at the expense of tourism. (Fig 5.5b). However, if the demand schedules have the same absolute slope then if there is a shift in the demand schedules by the same magnitude as in the earlier case, there would be no change in allocation of land between hunting and tourism, but the opportunity cost of land would shift upwards (Fig. 5.5a). So as we shift from Scenario 3 (complete separation of hunting from tourism), to Scenario 1 (hunting and tourism on separate area, but wildlife is shared), the opportunity cost of land goes up, but whether tourism or hunting is allocated more land depends on which activity is dominating.

We now shift focus to the hunting opportunity cost of land. Let ψ be the hunting opportunity cost of land in this case where the hunting area and tourism area are separated by a fence.

$$\psi = h(X, A\Phi)b_A(\Phi, A, p) + h_A(\Phi, A)b(\Phi, A) \quad (24)$$

We calculate the total derivative of ψ , then determine the effects of an increase in wildlife optimal stock as follows: increase in the unit profit from hunting, $h_\Phi(\Phi, A)b_A(\Phi, A, p)$ and the effect of wildlife stock growth changes from land, $h(\Phi, A)b_{\Phi A}(\Phi, A, p)$, which are similar to the case where the hunting area is separated from the tourism area but wildlife is shared (Scenario 1). However, these effects are smaller given that hunting is now occurring on a smaller area. There are two additional effects due to the hunting area being fenced off from the tourism area. These are: the effect of a change in unit profit from the change in wildlife stock $h_A(\Phi, A)b_\Phi(\Phi, A, p)$ and the effect of a change in unit hunting profit from land $b(\Phi, A, p)h_{\Phi A}(\Phi, A)$. Under this scenario the optimum profit made by the game reserve, ε_3^* , is given by equation (25).

$$\varepsilon_3^* = b(\Phi_3^*, A_3^*, p)q_3^* + Tf(\Omega_3^*, L_3^*) \quad (25)$$

5.4 Numerical example on demand for tourism and hunting land

In sub-section 5.2.1 specifically referring to Fig. 5.4 we illustrated the interaction between the demand of tourism land versus that of hunting land theoretically. To substantiate the results of some of the theoretical findings in the previous sub-sections, we now undertake a simple and

short numerical exercise. We use the equations formulations that were described earlier in the paper. Our concern now is not to model the precise tourism and hunting demand functions but to explore the general implications of interactions between these two land uses when selected parameter values change. Hence the parameters of the model have not been estimated. This is the subject of further work. Table 5.2 gives values of parameter for the functions used.

Table 5.2: Parameters and their values

| <i>Type</i> | <i>Symbol</i> | <i>Parameter description</i> | <i>Value</i> |
|------------------------------------|---------------|---|--------------|
| Herbivore ecological parameters | τ | Herbivore intrinsic growth rate | 0.2 |
| | K | Carrying capacity | 500 |
| Demand for hunting land parameters | p | Herbivore offtake price | 5 |
| | c1 | Herbivore hunting cost parameter | 1 |
| | c2 | Land hunting cost parameter | 1 |
| | z | General hunting cost parameter | 50 |
| | σ | Herbivore exponential parameter | 10 |
| | ϕ | Land exponential parameter | 10 |
| Demand for tourism land parameters | T | Tourism parameter | 100 |
| | a | Herbivore tourism rate | 1 |
| | b | Herbivore tourism half saturation coefficient | 5 |
| | c | Land tourism rate | 1 |
| | d | Land tourism half saturation coefficient | 1 |

Fig. 5.6 is similar to Fig. 5.2; the only difference is that the former is plotted using numerical data. We now explore numerically the effects of change in key parameters on the respective land-uses A and L . Table 5.3 shows the effect of a small change in key parameters on the size of the land for hunting, A , which implies opposite relationship for land for tourism, L , since $A=1-L$.

The demand for hunting land model is mainly sensitive to the change of two parameters: price of herbivore offtake (positive effect on hunting land) and land exponential parameter (negative effect on hunting land) (Table 5.3). A positive change in both of the herbivore ecological parameters, result in increase in the land under hunting at the expense of land under tourism (Table 5.3). The parameters that are numerators in the demand for tourism model have a negative effect on land for hunting when increased whilst the denominators have positive effects (Table 5.3).

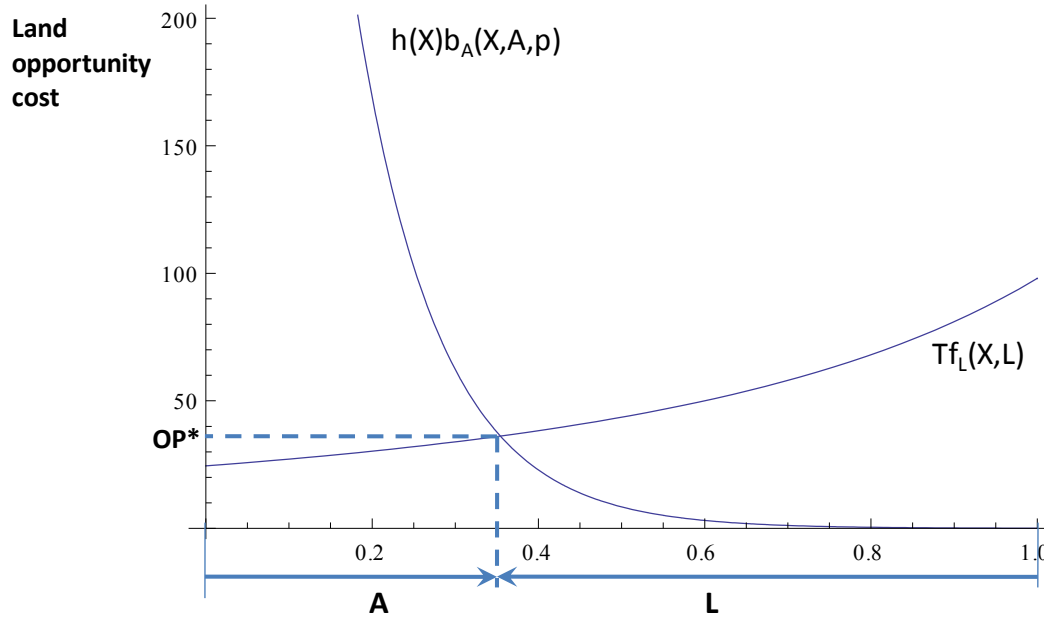


Fig. 5.6 Equilibrium condition for land plotted using numerical data. The curve $Tf_L(X,L)$ is the demand of land for tourism. The curve, $h(X)b_A(X,A,p)$, is the demand of land for hunting. The intersection of these two curves is the amount of land that should be devoted to hunting, A , and to tourism, L , and the optimum opportunity cost of land, OP^* on the vertical axis. The relationship $L+A=1$ holds. Compare with Fig. 5.2

Table 5.3: Change effect of key parameters on the size of land for hunting, A

| Parameter type | Key parameter | Change effect on A |
|------------------------------------|---|----------------------|
| Herbivore ecological parameters | Carrying capacity (K) | + |
| | Herbivore intrinsic growth rate (τ) | + |
| Demand for hunting land parameters | Price of herbivore offtake (p) | + |
| | Land exponential parameter (ϕ) | - |
| | Tourism parameter 1 (T) | - |
| Demand for tourism land parameters | Herbivore tourism rate (a) | - |
| | Herbivore tourism half saturation coefficient (b) | + |
| | Land tourism rate (c) | - |
| | Land tourism half saturation coefficient (d) | + |

5.4 Discussion

We examined from a theoretical point of view, under what conditions a landowner could allocate land between consumptive use and/or non-consumptive land-use which are potentially conflicting objectives. Our work uses a similar framework as Schulz and Skonhoft (1996), as we both assume that the amount of land is fixed. However, Schulz and Skonhoft analyzed two competing land-uses, wildlife and livestock production, where wildlife in the parkland generates externalities outside the park where livestock is produced. In our case we

analyzed two activities within the wildlife land-use area which are potentially conflicting. In addition, our approach offers richness in the sensitivity analysis, and we also substantiate some of the theoretical analytical findings using a simple and short numerical analysis.

The figure for demand for tourism and hunting land in the analytical part (Fig. 5.2) is similar to the one plotted using numerical data (Fig. 5.6). In the theoretical part of the analysis, the effect of change in price of wildlife offtake on land for hunting (equation (13)) was indicated to be ambiguous depending on whether Tf_{XL} is greater or less than $b_A h_X$. If $Tf_{XL} > b_A h_X$, then $dA^*/dp > 0$. From the numerical part of the analysis, the effect of change in price of wildlife on hunting land is positive. Numerically, $Tf_{XL} = 0.00059$ and $b_A h_X = -0.05801$. So the numerical work substantiates the theoretical part of the analysis. Under this finding, it means that when price wildlife offtake increases, the managers would allocate more land to hunting so that they capture the increased benefit (Schulz and Skonhofs, 1996). Also substantiating our findings is that in the theoretical part, the increase in tourism parameter, T , is said to shift the demand curve for tourism land outwards thereby resulting in increase in L , land for tourism, at the expense of A . This is also what the numerical part shows, since increase in T has a negative effect on A hence a positive effect on L (Table 5.3).

When the land-use is separate, but the wildlife resource shared, the negative effect of an increase in the price of wildlife offtake on the wildlife stock means that an increase in the price of wildlife offtake would lead to a decrease in wildlife stock. So, high prices of wildlife offtake would lead to a decrease in wildlife stock. In other words to keep higher stock of wildlife the price of its offtake must be kept at relatively low prices. This is in agreement with Clark's model and policies that have been used by CITES, where a moratorium on the trade of ivory which is expected to lead to higher stocks of elephants since the market for ivory would be depressed (Clark and Munro, 1975; Barnes, 1996; Bulte and van Kooten, 1999). So for the sake of wildlife conservation, the best strategy the management would be to introduce a moratorium or hunting quotas to ensure that wildlife stock is not locally depleted due to an increase in price of the hunted animal (e.g., Barnett and Patterson, 2006). If the management is not able to do that themselves, the authority under whose auspices quota are set, should step in. A point of debate that ought to be discussed would be at which level of grain or scale the quota should be set or maintained, and whether local overharvesting is allowed but global protection is conducted.

We will use the assumption that there is more wildlife biomass per unit area if there is only non-consumptive tourism than when there is only hunting (Munro, 1990; Schulz and Skonhøft, 1996). Given this assumption it would mean that by combining tourism and hunting there is a potential loss of wildlife biomass per unit area than when tourism alone is practiced. With this in mind we present the summary of model scenarios in Table 5.4.

Table 5.4: Comparisons of optimum biomass and profit for the three scenarios: from a wildlife point, Scenario 1 with non-domineering hunting appears to be the best, which also appears to be the optimum for income.

| <i>Hunting and Tourism areas</i> | <i>Wildlife biomass values {Comments between brackets}</i> | <i>Profit value {Comments between brackets}</i> |
|-----------------------------------|--|--|
| Partially separated (Scenario 1) | X_1^* {Biomass lower than in Scenario 3 as long as hunting is not dominating; wildlife offtake higher than in Scenario 3 as long as tourism is not dominating} | $b(X_1^*, A_1^*, p)h_1^* + T(X_1^*, L_1^*)$ {Profit higher than in Scenario 3 as long as hunting is not dominating; Profit is lower/higher than in Scenario 2 if land/biomass effect is dominating} |
| Not separated (Scenario 2) | X_2^* {Biomass lower than in Scenario 1; wildlife offtake higher than in Scenario 1 as long as the negative effect of hunting is not dominating} | $b(X_2^*, \Pi^*, p)h_2^* + T(X_2^*, \Pi^*)$ {Profit is higher/lower than in Scenario 1 if land/biomass effect is dominating} |
| Completely separated (Scenario 3) | $\Phi^*(A_3^*) + \Omega^*(L_3^*)$ {Biomass higher than in Scenario 1 as long as hunting land use is not completely dominating; wildlife offtake lower than in Scenario 1} | $b(\Phi^*, A_3^*, p)h_3^* + T(\Omega^*, L_3^*)$ {Profit lower than in Scenario 1 as long as tourism is not dominating} |

As we move from scenarios where hunting and tourism areas are completely separated (Scenario 3) through where they are partially separated (Scenario 1) to where they are not separated (Scenario 2), the wildlife biomass per hectare decreases (this is what we term the biomass effect) but the size of land increases (the land effect), but wildlife offtake increases (the wildlife offtake effect) (Table 5.4; Fig. 5.3). So if the management is interested in maintenance of wildlife biomass, they would prefer completely separating hunting and tourism sections of the game reserve, but this is as long as hunting is not completely dominating over tourism. If the management is interested in the maximization of profit then they would choose Scenario 2 and would combine hunting and tourism areas, as long as tourism is not completely dominating over hunting and the land effect is dominating over the biomass effect. If the biomass effect is dominating the land effect then the landowners would be better off by choosing Scenario 1. Further, if tourism is dominating then Scenario 1 would be the best option where wildlife biomass is highest. If the management are interested in satisfying their hunting clientele, they would choose Scenario 2 as long as the negative effect of tourism is not very high.

In South Africa there are numerous game reserves and ranches where hunting and tourism is occurring together or wildlife is shared between the wildlife consumptive and non-consumptive uses, as evidenced by the removal of fences separating private properties and even fences that are separating private and public reserves (Lindberg et al., 2003; APNR, 2005; Patterson and Khosa, 2006). However, some lobbyists and animal rights group advocate that hunting and tourism should not co-exist (Hearne and McKenzie, 2000; ABSA, 2003), which would be analogous to attaching an extremely high value on the negative effect of hunting on tourism. If their assessment is correct, then the best strategy would be a complete separation of the two land-use activities, a fence would be erected separating the hunting from the tourism areas. Although this is not part of our model, another way to look at this would be to hold the land-use activities at different times, that is hunting during the hunting season whilst engaging in tourism the remainder of the time (Hearne and McKenzie, 2000; ABSA, 2003). This ensures that tourism and hunting benefit from each other through a larger optimal wildlife stock than if only hunting was practised. In this case, then the model would need to accommodate a time variable, since tourism and hunting land-uses would be sharing wildlife and area but a time aspect would be separating the two activities. Critical to the applying of the model to facilitate choices, is thus the knowledge of the size of the depressing effect of hunting on tourism.

The model brings some insight into wildlife management decisions by a game reserve management or a landowner with a very large property, who needs to consider how to combine wildlife consumptive and non-consumptive uses. Our framework might be useful for analyzing wildlife management cases where both land size and land-uses are fixed and land-use activities are consumptive and non-consumptive. Areas that could be further explored include models for more than one species. However, we maintain that our framework could assist in analyzing a significant number of real world wildlife management cases. The key aspect to emerge from our work is that consumptive and non-consumptive use are not mutually exclusive, but that careful planning is needed to ensure that multiple reserve objectives can be met. Further, our results indicate that the two uses can be undertaken in the same contiguous area. Whether they are spatially (or temporally) separated depends on the magnitude of the consumptive use. Where the consumptive use is not dominating, the two could be compatible in the same shared area.

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Appendix 5

The game reserve Management's objective is given by the following:

$$Max_{q,A} PV = \int_0^{\infty} [b(X, A, p)q + Tf(X, L)]e^{-rt} dt \quad (A1)$$

Subject to

$$\frac{dX}{dt} = h(X) - q \quad (A2)$$

The current value Hamiltonian:

$$H = b(X, A, p)q + Tf(X, L)X + \mu[h(X) - q] \quad (A3)$$

The Pontryagin's necessary conditions for a maximum are:

$$\frac{\partial H}{\partial q} = b(X, A, p) - \mu = 0 \quad (A4)$$

$$\frac{\partial H}{\partial A} = qb_A(X, A, p) - Tf_L(X, L) = 0 \quad (A5)$$

$$\frac{\partial H}{\partial X} = Tf_X(X, L) + \mu h_X(X) + qb_X(X, A, p) \quad (A6)$$

$$-\frac{\partial H}{\partial X} = \dot{\mu} - r\mu \Rightarrow \dot{\mu} = r\mu - \frac{\partial H}{\partial X} \quad (A7)$$

$$\frac{\partial H}{\partial \mu} = -q + h(X) = 0 \Rightarrow q = h(X) \quad (A8)$$

Solve equations (A4) for μ

$$\mu = b(X, A, p) \quad (\text{A9})$$

Substituting equation (A9) into equation (A7)

$$\dot{\mu} = rb(X, A, p) - [Tf_X(X, L) + b(X, A, p)h_X(X) + qb_X(X, A, p)] \quad (\text{A10})$$

We assume that an equilibrium exists so that all conditions are simultaneously met. At equilibrium $\dot{\mu}$ is equal to zero. This makes equations (A10) equal to zero. Given these conditions and substituting for q as shown in equation (A8) into equations (A5) and (A11), solve equations (A5) and (A10) and re-arranging the equations we get the following:

$$r = h_X(X) + \frac{b_X(X, A, p)h(X)}{b(X, A, p)} + \frac{Tf_X(X, L)}{b(X, A, p)} \quad (\text{A11})$$

$$Tf_L(X, L) = h(X)b_A(X, A, p) \quad (\text{A12})$$

Equation (A11) can be rearranged as:

$$rb(X, A, p) - [Tf_X(X, L) + b(X, A, p)h_X(X) + b_X(X, A, p)h(X)] = 0 \quad (\text{A13})$$

Taking the total derivative of equation (A13) and expressing it as a matrix gives (for convenience we now use short hand):

$$\begin{bmatrix} [b_X(2h_X - r) + bh_{XX} + b_{XX}h + Tf_{XX}] & [-Tf_{XL} + b_A(h_X - r)] \\ [-Tf_{XL} + b_Ah_X] & [b_{AA}h] \end{bmatrix} \begin{bmatrix} dX^* \\ dA^* \end{bmatrix} = \begin{bmatrix} (r - h_X)b_p & b & -f_X \\ -b_{Ap}h & 0 & f_L \end{bmatrix} \begin{bmatrix} dp \\ dr \\ dT \end{bmatrix} \quad (\text{M1})$$

The left hand side of the matrix is for variables of the models whilst the right hand side is for parameters of the model. The determinant of the matrix on the left hand side of matrix (M1), D is given as follows:

$$D = [b_X(2h_X - r) + bh_{XX} + b_{XX}h + Tf_{XX}][b_{AA}h] - [-Tf_{XL} + b_A(h_{XX} - r) + b_{XA}h][-Tf_{XL} + b_Ah_X + b_{XA}h] \quad (A14)$$

D will strictly be greater than zero due to the second order conditions for a maximum (Schulz and Skonhøft, 1996). For calculating the comparative static in the main text, calculating $\frac{dX^*}{dp}$, you replace the column for the variable X^* (in the left hand side of matrix (M1)) with the column of p (in the right hand side of matrix (M1)) and calculate the determinant, D_p . Then $\frac{dX^*}{dp}$ would be given by $\frac{D_p}{D}$.

Synthesis

Emmanuel Mwakiwa

6.1 Introduction

The aim of this thesis was to provide wildlife management tool frameworks for both public and private landowners. For the foreseeable future in Africa, the consumptive and non-consumptive uses of wildlife resources will all be essential in contributing to sustainable wildlife management (Kelso, 1993; Lewis and Carter, 1993; Western and Wright, 1994; Pearce, 1995; Prins et al., 2000). The challenge is therefore to use appropriate wildlife management tools in order to maximize community benefits without jeopardizing Africa's wildlife heritage.

In southern African countries such as South Africa, both the state and private sector play important roles in wildlife management (Child, 2000; Muir-Leresche and Nelson, 2000). In this thesis, the study areas were Kruger National Park (KNP) and Associated Private Nature Reserves (APNR) located to the west of KNP. The APNR consists of several private nature reserves namely, Timbavati, Klaserie, Umbabat and Balule. Each nature reserve has a management board which has been entrusted to manage the reserve on behalf of the owners (Krug, 2001). Despite the management board, each landowner retains ownership of his/her property in the nature reserve. One level of cooperation that these landowners have established is the removal of fences not just amongst themselves in one nature reserve or amongst the nature reserves in the APNR but also between APNR and the Kruger National Park (APNR, 2005). Latter in the chapter, I will analyze another level of cooperation, besides sharing wildlife, which could be established between two landowners living adjacent to each other using an example of a landowner involved in non-consumptive use and the other in consumptive use. This is also known as landowners' client traversing rights agreement.

Both the state and private landowners have multiple objectives which vary amongst themselves. They use different tools and strategies to meet their management objectives in

wildlife management. Some of the most common intervention tools used by landowners in wildlife protected areas include surface water provision, fire management, fencing and animal population manipulation by culling/hunting or translocation (Perrings and Walker, 1997; Slotow et al., 2005; de Boer et al., 2007; Grant et al., 2008). In this thesis, I used bio-economic modelling approach to analyze the effects of a few wildlife management tools on the objectives of the landowners.

The management tools that I looked in this thesis are: increasing habitat quality (chapter 2), surface water manipulation (chapters 3), waterpoints opening or closure (chapter 4), full, partial use or non-use of physical barriers to incorporate non-consumptive and consumptive use of wildlife by one landowner (chapter 5). I developed several analytical frameworks using the Pontryagin's maximum principle (Clark, 1973; Swanson, 1994) to analyze these management tools and formulated strategies that could be used by wildlife managers to meet their wildlife management objectives (Chapter 2, 3, and 5). I also used the Savanna ecosystem model (Coughenour, 1993; Kiker, 1998; Boone et al., 2002) which is a spatially explicit, process-oriented model to further explore the effects of one of the management tools, viz. waterpoints establishment or closure (Chapter 4). There is scope for future work in further analyzing the tools and strategies that the landowners use in sustainable wildlife management.

In this closing chapter, the main findings are reviewed in the context of the other chapters. In addition, a number of recommendations drawn from the study are made. I also present the relevance and contribution of this project to the overall Temporal Elephant Movement and Bio-economic Optimization (TEMBO) integrated programme objectives. The integrated programme comprised six subprojects that together aimed at analysing and predicting the spatio-temporal distribution of elephants in conservation areas in relation to the nutrient status of the vegetation, in order to carry out a spatially explicit cost-benefit analysis for the optimisation of resource management for commercial purposes and conservation. This subproject contributed to the overall programme by using the analytical models and a spatio-temporal model for elephant distribution to analyse management actions in order to meet management objectives of the different land users (KNP, and surrounding private and corporate conservation areas).

6.2 Wildlife management tools and strategies

Chapter 2 assumes that it is usually not easy for landowners to increase land size given the increasing human population, high land prices and inelastic supply of land (Brown, 2009; Kuhnen, 1998; Foldvary, 2006). In such case, the landowner can improve wildlife habitat by

increasing land productivity. At the theoretical level, the model in chapter 2 enriches the analytical framework of the traditional harvest model of renewable resources by developing a bio-economic model with two state and two control variables. Compared to Clark's (Clark, 1973) and Swanson's (Swanson, 1994) models which only considered one species, I look at two species where one species depends on the other. This model also benefits from a simple numerical application on the effects of fertilizer on both the vegetation and herbivore biomass. It is shown that investment in vegetation improvement reduces the trend to herbivore extinction (Swanson, 1994). The analysis in this chapter also shows that utilization of wildlife rather than non-use can contribute to wildlife conservation and enhances landowner's welfare as it encourages investment by landowners in habitat quality improvement (Swanson, 1994; Alexander, 2000).

In chapter 3, I provide an ecological economic framework for analyzing optimization problems where a control has direct effect on one herbivore but indirect effect on the other. A landowner may be interested only in maximizing profits or personal benefits either through elephant offtake and/or tourism revenue, and thus might not realize that there are negative ecological effects that could be caused by elephants to biodiversity. When managers employ a certain tool, they might only take into consideration the direct effects of that tool but might not take into account the indirect impact of that tool. In chapter 3, I consider the waterpoints. Waterpoints as a management tool, affects the relationship between the abundance of water-dependent species and water-independent species. If the landowner is not taking into consideration the indirect effects of tools such as waterpoints, then the authorities responsible for sustainable management of the wildlife should use economic instruments such as subsidies or payments to the landowners who are complying with sound waterpoints management and/or taxes or charges to those who are not.

In chapter 4, I continue working with waterpoints but I use the spatially explicit Savanna ecosystem model to analyze the effects of manipulating waterpoints on both ecological and economic objectives of the KNP. The model supports the idea of treating the park as zones since different elephant densities have different impact on these zones given the resources within the region or zone (Whyte et al., 1999) as given by the habitat suitability index in the Savanna model. Given that different regions of the same ecosystem have different resilience due to differences in their biological make-up (Wiens, 1997; Whyte et al., 1999), so waterpoints have differential effects on both elephant density and vegetation biomass diversity in the different regions. Establishing waterpoints (or closing) in one region does not necessarily translate into higher (lower) elephant densities but depends on the dynamics of

habitat suitability index of the region in comparison to other regions. Although some studies found that waterpoint provision is a tool that can be used to manipulate the spatio-temporal distribution of water-dependent species (Chamaille'-Jammes et al., 2007, Kalwij et al., 2010), this may only happen when looking at a very large scale. However, on a regional scale, waterpoint manipulation to influence elephant distribution in a particular region may actually yield opposing and unintended results. This is particularly critical in larger national parks such as the KNP where elephant management focuses at a regional scale (Whyte et al., 1999). So managers of large national parks should carefully consider where they place the waterpoints since the effect of waterpoints on wildlife distribution is not as straight forward as one might be tempted to think *prima facie*.

Chapter 5 focuses on whether a landowner should pursue consumptive or non-consumptive wildlife utilization or both. The key aspect to emerge from this chapter is that consumptive and non-consumptive uses are not usually mutually exclusive, but that careful planning is necessary to ensure that multiple reserve objectives can be met. The analysis however, indicates that the two uses can be undertaken in the same contiguous area spatially or temporally. Whether they are spatially or temporally separated depends on the magnitude of the consumptive use. Where the consumptive use is not dominating, the two are compatible in the same shared area.

6.3 Sustainable utilization of wildlife management tools and strategies

Chapter 3 illustrates that by using a single tool to manage wildlife, some wildlife species might be promoted at the expense of the others. The findings of chapter 3 should be taken into context of other management tools such as the manipulation of vegetation quality (chapter 2). This means that improving the vegetation quality might result in promotion of certain species of vegetation which might offset the balance amongst different species in an ecosystem. In such circumstances, wildlife managers need to take into consideration direct and indirect effects of a management tool on different species of wildlife so that corrective action could be taken to avoid adverse effects of that tool on a group of species that are negatively or indirectly affected.

When dealing with a large national park which is managed at regional levels, increasing (decreasing) number of waterpoints does not necessarily result in increased (decreased) elephant density or vegetation diversity in a region (chapter 3). This insight is useful for landowner whose land is adjacent to other landowners and there are no physical barriers between the properties or when the barriers are not effective in restricting wildlife movement

between the properties. So using a certain management tool to repel or attract wildlife on one's land might actual result in unintended results. In such cases, landowners living adjacent to each other might need to further cooperate on top of just sharing wildlife roaming on their properties.

The entire thesis is thus relevant for the KNP and private landowners surrounding KNP. It is also relevant for other landowners involved in wildlife management particularly in the African context. For instance, chapter 4 is of interest to private landowners surrounding KNP who share elephants with other landowners in a large nature reserve where physical barriers between the multiple land properties have been removed. Establishing waterpoints (or closing) on one's property by individual landowners does not necessarily translate to higher (or lower) elephant densities on that property. Therefore, before managers decide to use waterpoints on one property to attract (or repel) elephants, they need to understand elephant-vegetation dynamics.

Furthermore, the results provide new insights into how landowners could manage waterpoints for National Parks and private protected areas alike. KNP feels that the waterpoints established by surrounding private landowners are excessive, which is why it has been closing its own waterpoints (Pienaar et al., 1997). But some of the private landowners have established waterpoints to attract wildlife which in turn they believe would attract more tourists (Witkowski, 1983; Parker and Witkowski, 1999). The results illustrate how major role-players in wildlife management could further cooperate despite being involved in potentially conflicting operations and still achieve their objectives.

6.4 Bargaining for clients' traversing rights

As already indicated, landowners in natures reserves in the APNR are already cooperating at a certain level given that they have removed the fence amongst their landholdings. They share the wildlife which roams on their properties. The landowners neighbouring each other might be involved in different and sometimes potentially conflicting wildlife land-uses, for example, consumptive and non-consumptive land-uses. There is need for them to consider another level of cooperation, that is, negotiation of traversing rights for their respective clients, like is the norm in some other private reserves (as in Welgevonden Private Game Reserve close to Vaalwater, South Africa). These landowners could continue practising their activities on their respective land without traversing rights agreement. The case is similar to the one considered in chapter 5 under the scenario where the hunting area is separated from the tourism area but wildlife is shared (scenario 1). On the other hand, they could give each other full traversing

rights, a case I will analyze now. Whether they agree to give each other traversing rights or not, it depends on a bargaining process. In Welgevonden, traversing rights had been established at the inception of the Reserve, and owners could only buy in under the condition that they accepted these traversing rights. Owners in the APNR base their rights on landownership rules that were in existence before the establishment of the Reserve, and new rules cannot be imposed from the top down. An investigation into bargaining rules is thus insightful because this can provide an avenue for further optimization for the individual landowners, who now are clearly constrained by the neighbours' *modi operandi*.

The framework for analysis is exactly as found in chapter 5 under the scenario where hunting and tourism are done in the same area (scenario 2). The difference is that instead of both revenues from hunting and tourism accruing to one decision maker, it now accrues to the respective landowner and there is now bargaining between the landowner involved in non-consumptive use (tourism) and the other involved in consumptive use (hunting). I ignore the effect of the unit damage cost caused by hunting on tourism which is better dealt with in chapter 5. To avoid lengthy and repetitive model description, I will just move to the salient part of the analysis.

There are two players involved. One landowner is involved in hunting (Agent A) and the other is involved in tourism (Agent B). I then define a "threat point" which is made up of the payoffs each agent would receive if there were no cooperation that is, no clients' traversing rights agreement (Munro, 1999). The assumption is that each player would accept a payoff from cooperation through traversing rights agreement, if it is greater than the threat point payoff. The less the player is likely to gain from noncooperation (that is no traversing rights agreement), the stronger one's bargaining power.

Adapting from chapter 5, the relationship between the two landowners' properties is given by the following:

$$L + A = \Pi \tag{1}$$

Where: the property under hunting is denoted by A and that under tourism is denoted by L .

When there is no cooperation, that is no traversing rights, the scenario would be analogous to the scenario where hunting area is separate from tourism area but wildlife is shared (scenario 1) in chapter 5. So the hunting landowner's profit function (ε) is given by:

$$\varepsilon = b(X, A, p)q \quad (2)$$

Where: X denotes the wildlife stock at time t (for notational convenience, we suppress the time notation, but time should be understood to be implicit in all variables); $b(X, A, p)$ is the unit profit function of hunting wildlife which depends on the wildlife stock, the area under hunting activity, and p , the price of wildlife offtake. This means ε is the threat point for the hunting landowner when there is no clients' traversing rights agreement but only wildlife roaming between the properties is allowed.

On the other hand, the profit function for the tourism landowner, (α), is given by:

$$\alpha = Tf(X, L) \quad (3)$$

Where $f(X, L)$ is the unit net tourism profit function which depends on the wildlife stock and amount of land under tourism; T is the number of tourists. This implies that α is the threat point for the tourist landowner when there is no clients' traversing rights agreement.

Now when the landowners have given each other traversing rights, the potential agreement between the two can be characterized as follows:

$$Max_q PV = \int_0^{\infty} [\beta b(X, \Pi, p)q + \{1 - \beta\}Tf(X, \Pi)]e^{-rt} \quad (4)$$

Where β is the bargaining parameter and $0 < \beta < 1$; r is the economic discount rate; $b(X, \Pi, p)$ is the new unit profit function for the hunting landowner; and $f(X, \Pi)$ is the new unit tourism revenue function for the tourism landowner.

The more β is closer to 1, the more the bargaining power the landowner involved in hunting has. On the other hand, the more β is close to 0, the more the bargaining power the landowner involved in tourism has, thus dominating the bargaining process.

The Hamiltonian function in this case would be:

$$H = \beta b(X, \Pi, p)q + (1 - \beta)Tf(X, \Pi) + \mu[h(X) - q] \quad (5)$$

A routine application of the maximum principle yields a modified Golden rule from which compromise optimum biomass level, denoted by X_C^* is derived. The modified Golden rule is:

$$r = h_x(X) + \frac{b_x(X, \Pi, p)h(X, \Pi)}{b(X, \Pi, p)} + \frac{Tf_x(X, \Pi)[1 - \beta]/\beta}{b(X, \Pi, p)} \quad (6)$$

Comparison could be made between this modified Golden rule (equation (6)) with the one in chapter 5 (equation (17) when we set $S = 0$). An interesting observation on the modified Golden rule in this case is on the third term on the wildlife's own interest rate (right hand of equation (6)), the marginal wildlife effect on tourism's unit net revenue. In this case the bargaining process yields an additional element $[1 - \beta]/\beta$. Deductions can be made from this element. When $\beta = 1$, the marginal wildlife effect on tourism's unit net revenue would disappear from the wildlife's own interest rate. This means that hunting would become most important and hunting only would be taking place and not tourism. In that case, the hunting landowner would convince the tourism landowner to give up tourism. Let us assume hunting landowner pays ψ to the tourism landowner to forgo some tourism. On the other hand, when β approaches 0, the marginal wildlife effect on tourism net revenue approaches infinity. This implies that as the landowner involved in tourism gets more and more bargaining power, hunting would become less and less important, and more tourism would be practised. But for the hunting landowner to forgo some hunting, let also assume the tourism landowner would pay him κ .

So the income for the hunting landowner under the traversing agreement is given by: $b(X_C^*, \Pi, p)q - \psi + \kappa$. This would mean that the hunting landowner would allow a clients'

traversing rights agreement as long as the income from entering such an agreement exceeds the threat point (where there is no traversing agreement) given as follows:

$$b(X, A, p)q < b(X_C^*, \Pi, p)q - \psi + \kappa \quad (7)$$

On the other hand, tourism landowner's income under the traversing agreement is given by: $Tf(X_C^*, \Pi) + \psi - \kappa$. The tourism landowner would enter a clients' traversing agreements as long as:

$$Tf(X, L) < Tf(X_C^*, \Pi) + \psi - \kappa \quad (8)$$

From this analysis we can also deduce what happens to the optimum wildlife stock as bargaining power changes. Let X_B^* denote wildlife biomass per hectare under tourism only and X_A^* denote wildlife biomass per hectare under hunting only. From assumptions already made in chapter 5, it implies $X_B^* > X_A^*$. When the landowner who is involved in hunting has more bargaining power, then the compromise optimum wildlife biomass is closer to X_A^* but it shifts to X_B^* as the landowner involved in tourism gains more bargaining power (Fig. 6.1). What comes out of the analysis is that the compromise in wildlife biomass is nearer to the landowner or agent who has the highest bargaining power. The logic of the outcome is that greater weight is given to preferences of the landowner with the highest bargaining power.

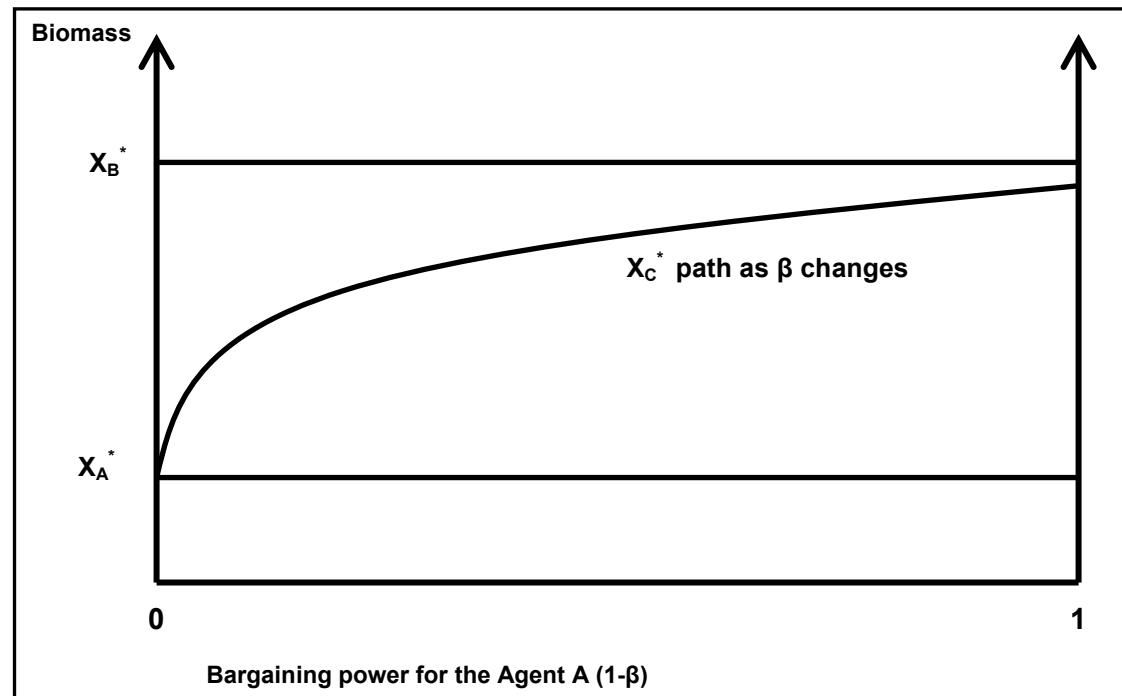


Fig. 6.1: Illustrating the effects of optimum wildlife biomass as landowner's bargaining power changes. X_A^* is wildlife biomass under Agent A; X_B^* is wildlife biomass under Agent B; X_C^* is compromised wildlife biomass due to cooperation

Furthermore the landowner who has the weaker bargaining power can even do more, if he has a lot to gain from the clients' traversing rights agreement (cooperation) with the landowner with the stronger bargaining power. For instance, if the landowner involved in hunting (Agent A) has the weakest bargaining power, he might actually persuade his neighbour to use a certain tool, like establishing or closing waterpoints so as to achieve his objectives (Nash, 1953; Munro, 1990). But he might need to pay his neighbour to carry out that particular action, as long as the benefits from such action exceed the costs (Nash, 1953; Clark, 1980; Munro, 1990). As for the neighbouring landowner who is asked to carry out a particular action, he would agree to that action as long as the payment he receives exceeds the forgone benefits (Nash, 1953; Munro, 1990).

This insight is essential for landowners given that there are potential benefits gained from cooperation for landowners who may even have conflicting objectives. For a single landowner with a single objective who is not interacting at all with the other landowners, the management of wildlife would be straight forward. However, in the real world, the landowner has multiple objectives and has to interact with the other landowners willingly or unwillingly. So the tools that the landowner uses, not only affects the attainment of his objectives but also those of others. So, it might be important for neighbouring landowners to consider further cooperation even if they were already cooperating through sharing of wildlife. By having

traverse rights agreement, the landowner might benefit from increased landsize for their clients and/or might benefit from a positive net transfer from his neighbour. However, the landowners might not agree to traverse rights agreement with their neighbour, if the net benefits from such agreement are negative maybe due to other landowner's clients causing more harm than good on his property or own clients.

The above framework could also be used to illustrate intervention by wildlife management authorities to discourage or encourage the use of a certain wildlife management tool. Biomass here could be for vegetation, herbivores, or combination. The authority could encourage the use of a certain tool in order to achieve increased biomass of a certain species or group of species. For instance, the authorities might encourage the use of certain fire frequency levels to encourage browse but avoid bush encroachment (Perrings and Walker, 1997; Kalwij et al., 2010). In such a case, Agent A would be the landowner, whilst Agent B would be the wildlife management authority, $(1-\beta)$ would represent the enforcement power of the authority. In this case the authority would be forcing the landowner to maintain higher levels of a given species. If the authority has stronger enforcement power, then the landowner would comply and biomass would be kept at higher levels near X_B^* (Fig. 6.1). However, if the authority has weaker enforcement power, then the biomass would be kept at lower levels near X_A^* (Fig. 6.1). To encourage the landowner to increase biomass by using a certain tool, the authority might use incentives such as payments and subsidies (chapter 3; Wallart, 1999; Bulte et al., 2003).

Alternatively, the authority could discourage the use of a certain tool so that biomass of certain species is kept at lower levels than an individual landowner would normally maintain without intervention. For example the authority might want to discourage the proliferation of water-dependent herbivore species by enforcing the closure of waterpoints (chapter 3; Pienaar et al., 1997; Grant et al., 2002; Mabunda et al., 2003). In such a case, Agent A would be the authority, whilst Agent B would be the landowner, β would represent the enforcement power of the authority. In this case, the authority would be forcing the landowner to keep lower biomass levels of certain species. If the authority has higher enforcement power, then the landowner would comply and biomass would be kept at lower levels near X_A^* (Fig. 6.1). However, if the authority has lower enforcement power, then the biomass would be kept at higher levels near X_B^* (Fig. 6.1). To discourage the landowner from maintaining higher

biomass, the authority could use taxes or charges to the landowner for exceeding the required levels of diversity (chapter 3; Wallart, 1999; Bulte et al., 2003).

Landowners need to know the ecological and economic implications associated with the tools that they are using (chapter 2, 3 and 4). In a game reserve management, some landowners might know and willingly comply with socially optimum application levels of a certain management tool whilst others would not. The game reserve management can use economic instruments to encourage compliance for those landowners who are not willing to comply (chapter 3).

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Summary

For the foreseeable future in Africa, the consumptive and non-consumptive uses of wildlife resources will all be essential in contributing to sustainable wildlife management. The challenge is therefore to use appropriate wildlife management tools in order to maximize community benefits without jeopardizing Africa's wildlife heritage.

Wildlife is a natural resource that has welfare effects on both the custodians and also the global community. However, wildlife conservation and its sustainability is determined mainly by the landowners on whose property wildlife roam on. Therefore, the survival of the wildlife is partly determined by how these landowners decide to use the land and the resources on it. They would stimulate sustainable wildlife management if their welfare requirements are met. Both public and private sector play important role in wildlife management in southern Africa. The study area for this thesis is Kruger National Park and Associated Private Nature Reserves, comprising Timbavati, Klaserie, Umbabat and Balule Private Nature Reserves.

Landowners use various tools to meet their objectives in wildlife management. Tools that we analyzed in this thesis are: improving land productivity; manipulating waterpoints; establishing and closing waterpoints; and full, partial use and non-use of physical barriers.

It is usually not easy to increase the land size in response to increased incentives. In that case the landowner might need to increase the land productivity instead. Bearing this in mind, the concept of investing in habitat quality improvement is introduced in chapter 2 by developing a mathematical model for wildlife utilization. The model has two state variables, vegetation biomass and herbivore biomass and two control variables, the investment in habitat quality improvement and the herbivore offtake rate. Empirical results from a large scale nutrient supplement experiment that was carried out in the Associated Private Nature Reserves, South Africa is used to calibrate the model and demonstrate the effects of an increase in land productivity on elephant biomass. It is shown that utilization of wildlife can contribute to wildlife conservation and enhancement of welfare as a result of investment by landowners into habitat quality improvement.

Surface water is one of the constraining resources for herbivore populations in semi-arid regions. Artificial waterpoints are constructed by wildlife managers to supplement natural water supplies, to support herbivore populations. Chapter 3 analyzes how a landowner may realize his ecological and economic goals by manipulating waterpoints. A theoretical bio-economic framework is developed to analyze the optimization of wildlife management objectives (in this case revenue generation from both consumptive and non-consumptive use and biodiversity conservation), using waterpoint construction as a control variable. The model provides an ecological economic framework for analyzing optimization problems where a control has direct effect on one herbivore but indirect effects on the other. A landowner may be interested only in maximization of profits or personal benefits either from elephant offtake and/or tourism revenue, thus might ignore the negative effects that could be brought about by elephants to biodiversity. If the landowner is not taking into consideration the indirect effects of tools such as waterpoints, then game reserve management as an authority entrusted with sustainable management of the game reserve on behalf of all the landowners might use economic instruments such as subsidies or payments to the landowners who are complying

with sound waterpoints management and/or taxes or charges to those who are not complying so as to enforce sound waterpoints management.

Game reserve managers often face multiple objectives, which are often conflicting. So there is a big challenge to develop wildlife management strategies that guarantee long term wildlife sustainable utilization. Tools used by managers to meet each of the objectives have impact on the plant and herbivores dynamics. One such tool is the construction or closing of artificial waterpoints. In chapter 4, the Savanna ecosystem model is used to analyze the differential impact of waterpoints on the Kruger National Park's regions under 26 waterpoints manipulation scenarios. The model is also used to analyze elephant impact on vegetation biomass diversity in four regions of Kruger National Park. The results showed that constructing (or closing) extra waterpoints in one region does not necessarily translate into higher (or lower) elephant densities in that region, but the effect depends on the vegetation and other conditions of the region in comparison to neighbouring regions. In one of the regions, the model showed that there is a trade-off between elephant density (representing an economic objective) and vegetation biomass diversity (an ecological objective). In another region, elephants' effect on vegetation biomass diversity follows the intermediate hypothesis, whilst in other regions the relationship is positive. The model thus suggests that different strategies should be adopted for different regions, e.g., an adaptive management strategy could be used for one of the regions where waterpoints are switched on and off depending on the elephant density.

Landowners or game reserve management are often faced with the decision whether to undertake consumptive and/or non-consumptive use on their properties. Hunting is a wildlife consumptive use whilst tourism is non-consumptive. In chapter 5, a theoretical model is constructed and is used to examine cases where the game reserve management decides whether to undertake hunting and/or tourism and in what combination. Three scenarios are postulated to analyze the effects of these decisions: (1) hunting area separate from tourism area but wildlife is shared; (2) hunting and tourism in the same area; and (3) hunting and tourism area separated by a fence. The key aspect to emerge from this work is that consumptive and non-consumptive use are not mutually exclusive, but that careful planning is needed to ensure that multiple reserve objectives can be met. It is also shown that the two uses can be undertaken in the same contiguous area. Whether they are spatially (or temporally) separated depends on the magnitude of the consumptive use. Where the consumptive use is not dominating, the two are compatible in the same shared area.

In the real world, the landowner has multiple objectives and has to interact with the other neighbouring landowners willingly or unwillingly. So the tools that the landowner uses, not only affects the attainment of his objectives but also those of others. Given these observations, what emerges from the synthesis (chapter 6) is that it is crucial for neighbouring landowners to consider further cooperation even if they were already cooperating through sharing of wildlife, as long as the benefits from cooperation are greater than the costs. In addition, landowners need to know the ecological and economic implications associated with the tools that they are using. In a game reserve management, some landowners might know and willingly comply with socially optimum application levels of a certain management tool whilst others would not. The game reserve management can use economic instruments to encourage compliance for those landowners who are not willing to comply.

Samenvatting

Voor de nabije toekomst in Afrika zal het consumptieve en niet-consumptieve gebruik van wilde fauna en flora een essentiële bijdrage leveren aan een duurzaam wildbeheer. De uitdaging is dan ook gebruik te maken van middelen in het wildbeheer die de gemeenschappelijke voordelen maximaliseren zonder Afrika's natuurlijk erfgoed in gevaar te brengen.

Wilde dieren zijn een natuurlijke hulpbron die effect heeft op het welzijn van zowel haar beheerders als ook de mondiale gemeenschap. Echter, het behoud van deze wilde dieren wordt vooral bepaald door de grondeigenaren op wiens land ze leven. Het voortbestaan van de dieren in het wild wordt daarom mede bepaald door de manier waarop de grondeigenaren gebruik maken van hun grond en de daarop aanwezige natuurlijke hulpbronnen. Zij zullen duurzaam wildbeheer stimuleren indien aan hun vereisten voor welzijn is voldaan. Zowel de publieke als private sectoren spelen een belangrijke rol in het wildbeheer in zuidelijk Afrika. Het studiegebied voor dit proefschrift is Kruger National Park en de bijbehorende Associatie van Private Natuur Reservaten, bestaande uit de Timbavati, Klaserie, Umbabat en Balule Private Natuurreservaten.

Grondeigenaren maken gebruik van diverse middelen om aan hun wildbeheer doelstellingen te voldoen. Middelen die we geanalyseerd hebben in dit proefschrift zijn: de verbetering van de productiviteit van land; de manipulatie van watervoorzieningen; de creatie en afsluiting van watervoorzieningen; en het niet, gedeeltelijk of volledig gebruik van fysieke barrières.

Het is meestal niet eenvoudig om het areaal land te vergroten als antwoord op een toegenomen economische drijfveer. In dat geval wordt de grondeigenaar gedwongen in plaats daarvan de productiviteit van zijn land te verhogen. Met dit in gedachten wordt het concept van investeren in kwaliteitsverbetering van de leefomgeving geïntroduceerd in hoofdstuk 2 door middel van het ontwikkelen van een wiskundig model voor het gebruik van wilde dieren. Het model heeft twee toestandsvariabelen, biomassa vegetatie en de biomassa herbivoren en twee controle variabelen, de investering in kwaliteitsverbetering van de leefomgeving en de snelheid waarmee herbivoren onttrokken worden. Empirische resultaten van een grootschalig voedingsstoftoevoeging-experiment dat werd uitgevoerd in de Associatie van Private Natuur Reservaten, Zuid Afrika, zijn gebruikt om het model te kalibreren en de effecten te demonstreren van het verhogen van de productiviteit van land op de biomassa olifanten. Hierbij werd aangetoond dat het gebruik van wilde dieren kan bijdragen aan duurzaam wildbeheer alsook een verhoging van het welzijn als gevolg van investeringen door grondeigenaren aangaande kwaliteitsverbetering van de habitat.

Oppervlaktewater is een van de beperkende factoren voor herbivoor populaties in droge gebieden. Kunstmatige watervoorzieningen zijn gebouwd door wildbeheerders om de natuurlijke watervoorraden aan te vullen voor de herbivoor populaties. Hoofdstuk 3 analyseert hoe een grondeigenaar zijn ecologische en economische doelen kan realiseren door manipulatie van deze watervoorzieningen. Een theoretisch bio-economisch kader is ontwikkeld om de doelstellingen voor optimalisatie van het wildbeheer te analyseren (in dit geval het genereren van inkomsten uit zowel consumptief als niet-consumptief gebruik en

behoud van biodiversiteit), met de aanleg van watervoorzieningen als controle variabele. Het model biedt een ecologisch-economisch kader voor de analyse van optimalisatieproblemen waar een controle variabele een direct effect heeft op een diersoort maar een indirect effect heeft op een andere diersoort. Een grondeigenaar is misschien alleen geïnteresseerd in de maximalisatie van de winst of van persoonlijk gewin door hetzij jacht op olifanten en/of toeristische inkomsten, en kan daarom wellicht voorbij gaan aan de negatieve effecten van olifanten op de biodiversiteit. Als een grondeigenaar geen rekening houdt met de indirecte effecten van beheersinstrumenten zoals watervoorzieningen, dan kan het management van het wildreservaat, als een autoriteit die belast is met het duurzaam beheer van het wildpark namens alle grondeigenaren, gebruik maken van economische instrumenten, zoals subsidies of betalingen aan de grondeigenaren die voldoen aan goed waterbeheer, en/of belastingen of heffingen aan diegenen die hier niet aan voldoen om zodoende goed beheer van watervoorzieningen af te dwingen.

Beheerders van wildreservaten worden vaak geconfronteerd met meerdere doelstellingen, die vaak onderling tegenstrijdig zijn. Daarom is een grote uitdaging om wildbeheer strategieën te ontwikkelen die op de lange termijn een duurzaam gebruik van de wilde dieren waarborgt. Instrumenten die worden gebruikt door beheerders om te voldoen aan elk van de doelstellingen hebben invloed op de dynamiek van zowel de planten en herbivore populaties. Een dergelijk instrument is de aanleg of de sluiting van kunstmatige watervoorzieningen. In hoofdstuk 4 wordt het Savanna ecosysteem model gebruikt om de invloed te analyseren van watervoorzieningen op de regio's van Kruger National Park met behulp van 26 scenario's waarin de watervoorzieningen worden gemanipuleerd. Het model wordt ook gebruikt om de invloed te analyseren van olifanten op de diversiteit van vegetatiebiomassa in vier regio's van het Kruger National Park. De resultaten toonden aan dat de aanleg (of het sluiten) van extra watervoorzieningen in de ene regio niet noodzakelijkerwijs een hogere (of lagere) olifantendichtheid in die regio tot gevolg heeft, maar dat het effect afhankelijk is van de vegetatie en andere kenmerken van de regio in vergelijking met naburige regio's. In een van de regio's toonde het model aan dat er een wisselwerking is tussen olifantendichtheid (wat een economische doelstelling weerspiegelt) en de vegetatiebiomassa diversiteit (een ecologische doelstelling). In een andere regio volgt het effect van olifanten op de diversiteit in vegetatiebiomassa de 'intermediate disturbance hypothesis', terwijl in andere regio's de relatie positief is. Het model geeft dus aan dat verschillende strategieën moeten worden gebruikt voor de verschillende regio's, een adaptieve strategie zou bijvoorbeeld gebruikt kunnen worden voor een van de regio's waarbij de watervoorzieningen afhankelijk van de olifantendichtheid in- en uitgeschakeld worden.

Grondeigenaren of het management van wildreservaten worden vaak geconfronteerd met de beslissing om consumptief en/of niet-consumptief gebruik te maken van hulpbronnen op hun grondgebied. Jagen is een consumptief gebruik van wild, terwijl toerisme een niet-consumptief gebruik is. In hoofdstuk 5 wordt een theoretisch model geconstrueerd en gebruikt om gevallen te onderzoeken waarin het management van een wildpark beslist over het uitvoeren van jacht en/of toerisme en in welke combinatie. Drie scenario's worden vooropgesteld om de effecten van deze beslissingen te analyseren: (1) het jachtgebied te scheiden van het toeristisch gebied, maar de wilde dieren worden gedeeld; (2) jacht en toerisme in hetzelfde gebied; en (3) het jachtgebied en toeristisch gebied worden gescheiden door een hek. Het belangrijkste aspect dat uit dit werk naar voren komt is dat consumptief en

niet-consumptief gebruik elkaar niet uitsluiten, maar dat een zorgvuldige planning nodig is om ervoor te zorgen dat aan meerdere doelstellingen van een reservaat kan worden voldaan. Het is ook gebleken dat de twee manieren van gebruik kunnen worden uitgevoerd in hetzelfde aaneengesloten gebied. Of ze ruimtelijk (of tijdelijk) gescheiden dienen te worden is afhankelijk van de mate van het consumptief gebruik. Wanneer het consumptief gebruik niet domineert zijn de twee verenigbaar in hetzelfde gebied.

In de echte wereld heeft een grondeigenaar meerdere doelstellingen en heeft hij willens of onwillens een interactie met andere naburige grondeigenaren. De beheersinstrumenten die de grondeigenaar gebruikt zijn dus niet alleen van invloed op de verwezenlijking van zijn eigen doelstellingen, maar ook die van anderen. Met deze waarnemingen in gedachte, blijkt uit de synthese (hoofdstuk 6) dat het van cruciaal belang is voor naburige grondeigenaars om verdere samenwerking te overwegen, zelfs als ze al samenwerken door het delen van wilde dieren, zolang de voordelen van samenwerking groter zijn dan de kosten. Bovendien moeten grondeigenaren kennis hebben van de ecologische en economische gevolgen die verband houden met de beheersmiddelen die ze gebruiken. Bij het beheer van een wildreservaat handelen sommige grondeigenaren gewillig in overeenstemming met een sociaal optimaal toepassingsniveau van een bepaald beheersinstrument, terwijl anderen dit niet doen. Het management van wildreservaten kan dan economische instrumenten hanteren om meegaandheid aan te moedigen bij grondeigenaren die zich niet uit zichzelf hierin schikken.

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Curriculum vitae

Emmanuel Mwakiwa was born on 23rd of October 1972 in Kadoma, Zimbabwe. He attended Patchway Primary School from 1980 to 1986, and Kwayedza Secondary School from 1987 to 1990, both schools in Kadoma. From 1991 to 1992, he went to attend Sandringham High School in Chivero District of Mhondoro, Norton. He went to University of Zimbabwe in 1992 to study at the Department of Agricultural Economics and Extension where he received his BSc Agriculture Honours (Agricultural Economics specialization) in 1995. He then worked for the Project Management Unit in the Ministry of Agriculture, Zimbabwe from 1997 to July 2003 as an Agricultural Economist. In 2002 to 2004, he studied for MSc Agricultural Economics at the University of Zimbabwe. For his research project he was awarded an AGROPOLIS scholarship in urban agriculture by the International Development Research Centre. His MSc research thesis is entitled “Evaluation of the Socio-economic Determinants and Benefits of Urban Agriculture”. In 2004, he studied for diploma equivalence in an exchange programme under the auspices of Regional Collaborative Masters Degree Program in Environmental Economics and Policy at the Centre for Environmental Economics and Policy in Africa at University of Pretoria, South Africa.



Between September 2003 and November 2005, he worked as an Operations Research Manager for Catholic Relief Services (CRS) an international non-governmental organization in Zimbabwe. Whilst working for the Government and international NGO (CRS) he interacted and worked with the following bilateral and multilateral organizations: USAID, IFAD, Dutch Embassy, DFiD, EC, FAO, UNDP, UNICEF, AusAID, SIDA, UNOPS, World Bank, African Development Bank, SADC, and WFP. He has coordinated at one time projects dealing with more than 16 national and international NGOs where he was involved in monitoring and evaluation, data collection tools development, project management and report writing for CRS head office and major donors.

From December 2005, he began studying for a PhD student degree at the Resource Ecology Group, Wageningen University, The Netherlands. In 2010, he worked as a Senior Regional Expert for a consortium comprising the Institute of Communications and Computer System of the National Technical University of Athens (ICCS-NTUA), Integration GmbH, ECORYS Netherlands and TMS International Limited. For this consortium, he was involved in results-oriented monitoring of Food and Agriculture Organization of the United Nations (FAO) and its implementing partners (13 national and international NGOs) EU food facility programme in Zimbabwe. As from January 2010 to present, he works as a lecturer in the Department of Natural Resources Management, Faculty of Agriculture at Africa University located in Mutare, Zimbabwe. In May 2011, he was appointed as a part time lecturer for the Institute of Development Studies, Faculty of Commerce, at the National University of Science and Technology, Bulawayo, Zimbabwe.

PE&RC PhD Education Certificate

With the educational activities listed below the PhD candidate has complied with the educational requirements set by the C.T. de Wit Graduate School for Production Ecology and Resource Conservation (PE&RC) which comprises of a minimum total of 32 ECTS (= 22 weeks of activities)



Review of literature (5 ECTS)

- Literature review on elephants economics and ecology (2006)

Writing of project proposal (4.5 ECTS)

- Optimising resource management actions by incorporating spatio-temporal distribution of elephants (2006)

Post-graduate courses (4.5 ECTS)

- Art of modelling (2006)
- Uncertainty analysis (2006)

Laboratory training and working visits (3 ECTS)

- Private players involvement in wildlife management in South Africa; University of Limpopo (2007)
- Optimization techniques; University of Pretoria (2007)

Invited review of (unpublished) journal manuscript (1 ECTS)

- Journal of Livestock Production: determinants of economic viability of a dairy goat enterprise in Kenya: implications for a breeding programme (2009)

Deficiency, refresh, brush-up courses (3 ECTS)

- Ecological methods I (2006)
- Advanced environmental economics and policy (2006)
- Economics and management of natural resources (2006)

Competence strengthening / skills courses (1 ECTS)

- Personal efficacy course (2008)
- Competence assessment (2006)

PE&RC Annual meetings, seminars and the PE&RC weekend (1.2 ECTS)

- PE&RC Introduction weekend (2006)
- PE&RC Day (2008)

Discussion groups / local seminars / other scientific meetings (7.4 ECTS)

- Statistics, maths and modelling in production ecology (2006, 2008, 2009)
- Forest ecology and nature conservation (2006, 2009)
- Centre for Environmental Economics and Policy Analysis in Africa PhD meetings at the University of Pretoria (2007)

International symposia, workshops and conferences (5.1 ECTS)

- Kruger National Park Network meeting (2007)
- Associated Private Nature Reserves meetings (2007)

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