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Optimization of wildlife management in a large game reserve through waterpoints manipulation: A bio-economic analysis

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A B S T R A C T

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 analyse how a landowner may realize his ecological and economic goals by manipulating waterpoints for the management of an elephant population, a water-dependent species in the presence of water-independent species. We develop a theoretical bio-economic framework to analyse 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 a bio-economic framework for analysing optimization problems where a control has direct effects on one herbivore species but indirect effects on the other. A landowner may be interested only in maximization of profits either from elephant offtake and/or tourism revenue, ignoring the negative effects that could be brought about by elephants to biodiversity. If the landowner does not take the indirect effects of waterpoints into consideration, then the game reserve management, as the authority entrusted with the sustainable management of the game reserve, might use economic instruments such as subsidies or taxes to the landowners to enforce sound waterpoint management.

1. Introduction

In South Africa private landowners play a crucial role in wildlife conservation, and often aim to conserve wildlife (Jordi and Peddie, 1988; APNR, 2005). Some private landowners have agreed to remove fences between their properties to allow wildlife to roam between their properties. A private nature reserve is one of the types of private land ownership for wildlife management in South Africa. A nature reserve also referred to as game reserve consists of several landowners who have often removed fences amongst their landholdings. They generally employ a management team to run the reserve and ensure that sustainable wildlife management actions are practiced by the landowners, although the individuals retain their individual property ownership rights (APNR, 2005). Management objectives of private nature reserves vary from preservation to the sustainable use of wildlife. Some nature reserves have formed associations whereby adjoining reserves pooled their resources and removed fences to create even larger units. One of such associations is the Associated Private Nature Reserves (APNR) which is the focus of this paper. The APNR is located to the west of the Kruger National Park and consists of the Timbavati, Klaserie, Umbabat and Balule Private Nature Reserves and has a combined total area of approximately 1850 km^2 (APNR, 2005). Furthermore, the APNR has entered into an agreement with the state-owned

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Kruger National Park which led to the removal of fences between the Kruger National Park and the APNR in 1994 to create a very large game reserve (APNR, 2005).

The APNR generates income from e.g., eco-tourism, fees paid by landowners, and hunting (APNR, 2005). Some commercial and trophy hunting is conducted in the APNR (2005). Hunting quotas are based on existing wildlife populations and are set by the province administration (APNR, 2005). Since the reserves within the APNR are "associations-not-for-gain", the proceeds generated by hunting activities are used exclusively for biodiversity conservation (APNR, 2005). However, the state has the ultimate say on hunting activities. For instance, to promote the growth of the elephant population, the South African government banned elephant culling and hunting in 1995, which resulted in a substantial increase in the elephant population (Grant et al., 2008). However, the ban was lifted in 2008 to improve management of the flourishing elephant populations (Nature, 2008).

Surface water provision, fire management, fencing and animal population manipulation by culling, translocation or introducing predators, are some of the most common interventions 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; Ripple and Beschta, 2011). 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 the impacts of herbivores more evenly over the area (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 construction would increase the number and visibility of such animals (Mabunda et al., 2003; Chamaillé-Jammes et al., 2007; Smit et al., 2007). This led to the assumption that a more extensive network of waterpoints would increase the revenues generated through tourism (Parker and Witkowski, 1999; Mabunda et al., 2003). For example, the number of waterpoints in the Klaserie Private Nature Reserve increased from only six waterpoints in 1965, to 144 by 1980 (Witkowski, 1983; Parker and Witkowski, 1999). Although waterpoints could be beneficial to wildlife viewing, they may compromise biodiversity (Harrington et al., 1999). Additional waterpoints could result in an increased number of herbivores. This could increase revenues in the short term, but in the long term it would adversely affect biodiversity (Thrash et al., 1991; Baxter, 2003).

Hence, the potential increase in animal impact with increasing numbers of artificial waterpoints is an issue of concern. A certain number of waterpoints per given area could be beneficial to the animals in a conservation area. Too few waterpoints could result in severe water shortages for animal populations, increasing animal mortality. On the other hand, too many waterpoints might result in increased environmental costs, such as reduced biodiversity. Too many waterpoints could result in widespread large impacts on the vegetation, an increase in predation, or an over-utilization of the vegetation, resulting in the homogenization of the vegetation composition and structure (Owen-Smith, 1996; Thrash, 1998; Harrington et al., 1999). Moreover, additional waterpoints might lead to an increase in the population of water-dependent species like elephant, zebra, buffalo, wildebeest and waterbuck (Collinson, 1983; Redfern et al., 2003, 2005), at the expense of water-independent (or less water-dependant) species, such as tsessebe, roan antelope, impala, kudu, giraffe, and warthog, which can tolerate limited water consumption and survive for long periods without access to surface water (Martin, 1983; Smithers, 1983; Estes, 1991).

Roan antelope, sable antelope 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 the Kruger National Park a severe drop in the roan antelope 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 antelope range, which attracted the large grazers such as zebra and wildebeest, particularly during drought conditions (Harrington et al., 1999; Grant et al., 2002).

Human society often pursues several goals, improving human welfare, increase sustainability of production methods, and conserving biodiversity. A positive feedback loop could emerge if sustainable human activities promote biodiversity, which in turn fosters successful and sustainable human activities. In this context, artificial waterpoints can be thought of as a proxy for human endeavours. However, the ecological impacts of waterpoints on biodiversity are at present not well understood, and there are no economic studies addressing the issue of waterpoint construction from a bio-economic perspective. The intention of this paper is therefore to address this knowledge gap. We aim to analyse how wildlife managers may achieve their objectives of generating returns from wildlife by manipulating the number of waterpoints simultaneously contributing to sustainable wildlife conservation. We develop a theoretical bio-economic framework to analyse 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 the distribution of wildlife populations (Redfern et al., 2005). We present a theoretical bio-economic model with various degrees of complexity based on a set of ecological assumptions presented in Table 1.

2. The model

We first consider a single species model without environmental costs. We then consider another single species model with environmental costs. Lastly we consider a two species model – with elephants representing a water-dependent species and roan antelope as proxy for 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. For simplicity, predation is not included as a controlling mechanism in the model. The model is based on a large closed (fenced) reserve in which immigration and emigration is absent.

2.1. The single species model without environmental costs

Two economic activities were considered *viz.* tourism and hunting. 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 own density (X) and is positively affected by the number of

Table 1
Major assumptions used in the bio-economic model and relevant references.

Assumption	Reference(s) to support the assumption	Brief explanation of the findings of the reference
Waterpoints increase the density of elephants	(a) Valeix et al., 2008 (b) Chamaillé-Jammes et al., 2007	(a) During dry years fewer waterholes retain water during the dry season with a consequent increase in elephant numbers at waterholes (b) During a long-term dry-season elephant densities across the park increased asymptotically with the density of artificial waterholes; during dry years elephant numbers at waterholes increased (c) Water availability is the best explanatory factor for elephant population fluctuations
Culling/hunting by humans decreases the density of herbivores	(a) Valeix et al., 2008 (b) Chamaillé-Jammes et al., 2007 (c) Whyte et al., 2003	(a) The elephant population has more than doubled since culling stopped (b) Since the culling stopped, dry-season elephant densities have increased in most areas of the park (c) Culling stabilized elephant population in Kruger park
Elephants have a negative influence other browser species and overall biodiversity when in high density	Valeix et al., 2007 Fritz et al., 2002	(a) Negative effect of elephants on other herbivore species when elephants are present at high densities (b) Elephants compete with mesobrowser species
Roan are water-independent	Thrash et al., 1995 Harrington et al., 1999	Roan antelope are water-independent species
Increasing waterpoints result in drop in number of roan	Harrington et al., 1999	The provision of numerous artificial waterpoints in the roan range, which attracted zebra and wildebeest resulted in a drop in roan population
Reducing waterpoints is beneficial for roan	Grant et al., 2002	When 12 boreholes were closed and one earth dam drained in Shingwedzi plains in KNP, the number of zebra declined and roan antelope population stabilized

waterpoints, W (Chamaillé-Jammes et al., 2007), q is the specific rate of elephant offtake, which has a negative effect on the elephant population size (Whyte et al., 2003). 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). The superscript denotes the partial derivative with respect to the variable shown in the superscript. We also assume that $h^{xx}(X, W) < 0$ and $h^{W}(X < W) > 0$.

In this case we assume that the index of profit (Π) is a measure of the landowner's net benefit. The landowner can influence his net benefit by manipulating the rate of elephant offtake through hunting, q, and the number of waterpoints W. The profit function for the landowner is given by:

$$\Pi = pqX + TX - ZW \tag{2}$$

Where pqX is the revenue generated from wildlife offtake. We assume that p, the price of a wildlife offtake, is fixed (de Boer et al., 2007). The price of offtake represents the profit (gain) that the landowner derives from the offtake. TX is the revenue realized from non-consumptive use of elephants. In the absence of more precise information it seems reasonable to assume a linear relationship between revenues from non-consumptive use of elephants and the magnitude of the population. Z is the cost of maintaining a waterpoint.

2.1.1. Optimal management without environmental costs

Where elephant does not impose costs on the environment, 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 the present value (PV) of profit (Π), given as:

$$\underset{q,W}{\mathrm{Max}}\, PV\Pi = \int_0^{\infty} [pqX + TX - ZW]e^{-rt}\mathrm{d}t \tag{3}$$

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

$$H = pqX + TX - ZW + \lambda[h(X, W) - qX] \tag{4}$$

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 density.

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

$$h^{W}(X, W) = \frac{Z}{p} \tag{5}$$

$$r = h^{X}(X, W) + \frac{T}{p} \tag{6}$$

The Equations (5) and (6) are the necessary conditions for a maximum when an interior solution is assumed to be present, that is, where the number of herbivores, the harvesting rate, and the waterpoints at the steady state are positive in the long term.

Equation (5) reflects the impact of the waterpoint control term in the objective function. The left hand side of Equation (5) 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 the relative cost of waterpoint maintenance (RCW). When the cost of waterpoint maintenance goes up or the price of herbivore offtake goes down, the relative cost of waterpoint maintenance shifts from RCW0 to RCW1 because of a lower demand of waterpoints (Fig. 1). Under these conditions the optimum herbivore biomass decreases from X_0^* to X_1^*. On the other hand, when the cost of waterpoint maintenance decreases or the price of herbivore offtake goes up, the relative cost of waterpoint maintenance shift from RCW0 to RCW2. This results in an increase in demand for waterpoints, so that the optimum herbivore biomass increases from X_0^* to X_2^* (Fig. 1).

The left hand side of Equation (6) is r, the economic discount rate determined by the market forces. The right hand side of Equation (6) can be regarded as the elephant's own interest rate (Clark and Munro, 1975) or returns to the elephant resource (Fleming and Alexander, 2003), which consists of two components: (i) the marginal productivity of the elephant; and (ii) the marginal revenue from non-consumptive use.

Since $h^{x}(X, W) > 0$ it is apparent that if the landowner increases the number of waterpoints, an increase in the elephant marginal

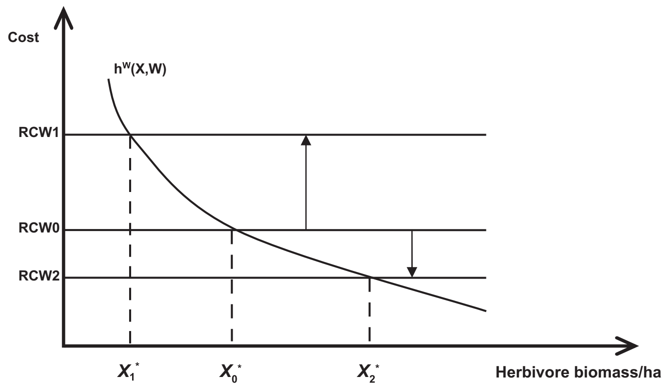

Fig. 1. Effects of change in cost of waterpoints maintenance (Z) and price of herbivore offtake (p) on the relative cost of waterpoint maintenance (RCW).

productivity would result. 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 2). It has a positive or negative effect on the marginal productivity, whilst a negative effect on the marginal benefits from the non-consumptive use term. If the increase in elephant 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 have a positive effect on the elephant's own interest rate. Although the effect of the cost of waterpoint maintenance is negative (Table 2), this is negligible. The effect of an increase in the marginal benefits from non-consumptive use is positive (Table 2). In this model, potential negative effects of elephants on biodiversity are not taken into consideration.

2.2. *The single species model with environmental costs*

The specification of the model as in the previous sub-section, 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 revenues 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 later in the two species model we let roan antelope, the water-independent species represent biodiversity.

Elephants are regarded as a charismatic megafauna species because of their size, intelligence, and social structure, and they are regarded as ecosystem engineers because of their direct and indirect impacts on the availability of resources to other species and their influence on structuring habitat heterogeneity (Kerley et al., 2008; Valeix et al., 2008). Elephants can have large effects on vegetation composition and influence the balance between trees, shrubs and herbaceous layers in plant communities (Holdo, 2007; Kerley et al., 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). This is suggested by the correlation between the increase in elephant population and the increase in population size for other species for which medium-term facilitation has been suggested. This facilitation may occur through an increase in browse from coppicing as a result of elephant activity (Baxter, 2003; Makhabu et al., 2006; de Knegt et al., 2008). However, extremely high densities of elephants may negatively affect biodiversity (Tchamba, 1995; Baxter, 2003; Kerley et al., 2008), as suggested by the

Table 2
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	$+$	$+$	$+$

negative correlation between the biomass of elephant and other browser species across ecosystems (Fritz et al., 2002). These findings suggest that there could be an optimum level above which elephant have a negative impact on biodiversity (Whyte et al., 2003).

Hence, an increase in the number of elephants can result in environmental costs such as a loss in biodiversity. 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:

$$M = M(X) \tag{7}$$

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

2.2.1. *Optimality with environmental costs*

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

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

The new current-value Hamiltonian (Clark, 2005) is given by the following:

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

After using the Pontryagin's maximum principle (Clark, 2005), we get the same Equation (5), however, Equation (6) is modified as:

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

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. 2; Equations (6) and (10)). However, this model does not explicitly model the effects of elephants on biodiversity. We address this in a separate model, the two species model.

2.3. *The two species model*

In the two species model we use the following notation: the subscript denotes the species whilst the superscript denotes the derivative with respect to the variable shown in the superscript. Now we consider two species: one species, X_1, is water-dependent, represented by elephant and the second species, X_2 is a water-independent species, represented here by the roan antelope, as a proxy for the negative effects of elephants on water-independent species, thereby symbolizing the effects of biodiversity. The dynamics of the elephant is given by:

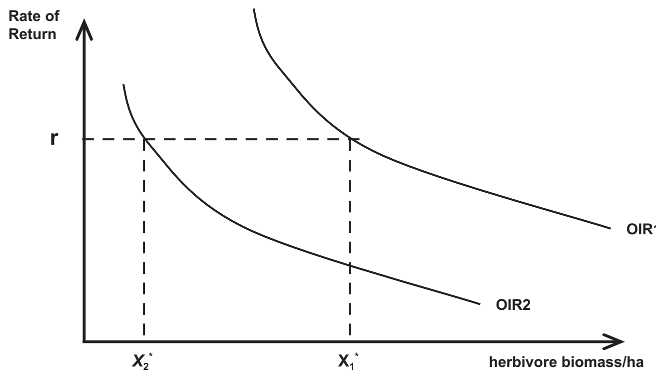

Fig. 2. The effect of considering environmental costs by elephants in a game reserve area. "r" is the economic interest rate; "OIR" is the Own Interest Rate; "OIR1" represents OIR where environmental costs have not been considered; "OIR2" is the OIR where environmental costs have been considered.

$$\frac{dX_1}{dt} = h_1(X_1, X_2, W) - q_1X_1 \tag{11}$$

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

$$\frac{dX_2}{dt} = h_2(X_1, X_2) - q_2X_2 \tag{12}$$

Where: the subscript $i = 1$ and 2 denotes elephant and roan antelope, respectively, $h_1(X_1, X_2, W)$ is the growth function of elephants, which depends on its own density, the roan antelope density, and the additional water provision through the number of waterpoints, W; $h_2(X_1, X_2)$ is the roan antelope growth function, which depends on its own density and the elephant density; q_i is the rate of herbivore offtake for species i. We assume the following: $h_1^1(X_1, X_2, W) > 0$ and $h_2^2(X_1, X_2) > 0$ implying that intra-specific competition is negligible $h_1^W(X_1, X_2, W) > 0$, 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 inter-specific competition for forage (Kerley et al., 2008). For model simplicity we assume that there is only one forage type available for which both species compete. Waterpoints have an indirect effect on the roan antelope population size through the growth of the elephant population.

For this case we also assume that the index of profit (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:

$$\Pi = p_1q_1X_1 + p_2q_2X_2 + T_1X_1 + T_2X_2 - ZW \tag{13}$$

Where: $p_iq_iX_i$ is the revenue generated from wildlife offtake of species, i. We assume that p_i, which is the price offtake for species i, is fixed, T_iX_i are the tourism revenues realized from species i.

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

$$\underset{q_i, W}{\text{Max}}\, PV\Pi = \int_0^\infty [p_1q_1X_1 + p_2q_2X_2 + T_1X_1 + T_2X_2 - ZW] \tag{14}$$

The current-value Hamiltonian of the problem is now:

$$\begin{aligned} H = {} & p_1q_1X_1 + p_2q_2X_2 + T_1X_1 + T_2X_2 - ZW \\ & + \lambda_1[h_1(X_1, X_2, W) - q_1X_1] + \lambda_2[h(X_1, X_2) - q_2X_2] \end{aligned} \tag{15}$$

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 species X_1 and X_2 respectively.

2.3.1. Optimality in the two species model

After using Pontryagin's maximum principle, we get:

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

$$r = h_1^1(X_1, X_2, W) + \frac{T_1}{p_1} + \frac{p_2h_2^1(X_1, X_2)}{p_1} \tag{17}$$

$$r = h_2^2(X_1, X_2) + \frac{T_2}{p_2} + \frac{p_1h_1^2(X_1, X_2, W)}{p_2} \tag{18}$$

Equations (16)–(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 (returns to the resource), for elephant and roan antelope, as represented by the right hand sides of Equations (17) and (18) respectively. Each herbivore's own rate comprises 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's term.

The landowner would open more waterpoints which would result in a higher density of elephants and a lower density of roan antelope with possibility of local extinction of roan antelope under any of these conditions: (a) the marginal effect of elephants on roan antelopes is very low (that is, elephant is a weak competitor), an ecological factor; (b) the marginal effect of roan antelope on elephant is very high (that is, roan antelope outcompete elephant), another ecological factor; (c) the price of elephant offtake is high, an economic factor; and (d) the price of roan antelope offtake is very low, also an economic factor. Since the first two conditions do not make ecological sense, it seems that additional waterpoints will only be opened for economic reasons. The landowner would close some of the waterpoints leading to lower elephant density and higher density of roan antelope under the opposite conditions.

The effect of change in the elephant offtake price on the elephant's own interest rate can be deduced from the above equations. If the price of the elephant's offtake increases, the landowner would establish more waterpoints, which would result in: an ambiguous effect on the elephant's marginal productivity; a negative effect on the marginal benefits from the elephant's non-consumptive use term; an ambiguous effect on the marginal effect of elephant on the roan antelope term (i.e. a positive effect if the increase in the price of the herbivore's offtake – the denominator, is more than the increase in the marginal effect of the elephant on roan antelope – the numerator). Overall, the effect of an increase in the elephant's offtake price on the elephant's own interest rate is ambiguous (Table 3). An increase in the price of the elephant's offtake would have positive effects on the elephant's own interest rate if the total positive effects from the elephant's marginal productivity and the marginal effect of elephant on the roan antelope's term are larger than the negative effects from the marginal benefits from the elephant's non-consumptive use term. The effects of an increase in the price of elephant offtake (p_1), price of roan antelope offtake (p_2), waterpoint maintenance cost (Z), elephant's marginal benefits from non-consumptive use (T_1) and roan antelope's marginal benefits from non-consumptive use (T_2) on the elephant's own interest rate and on the roan antelope's own interest rate are summarized in Table 3.

Table 3
Comparative statics for the elephant's and roan antelope's own interest rates – the effects of an increase in price of elephant offtake (p_1), price of roan antelope offtake (p_2), waterpoint maintenance cost (Z), elephant's marginal benefits from non-consumptive use (T_1) and roan antelope's marginal benefits from non-consumptive use (T_2) on.

(a) 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	+	−/+
(b) The roan antelope'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. Implications of the model

In the one species model without environmental costs, controls (elephant offtake and waterpoints) have direct effects on the state variable, elephant density and nothing else (Fig. 3a) which is far from reality. In the one species model with environmental costs, the controls have two types of effects: (i) direct effects on the state variable and (ii) indirect effects on the environmental costs/benefits through the state variable (Fig. 3b). However, the environmental costs have no control which directly affects them. In the two species model, there are three controls: elephant offtake, waterpoints and roan antelope offtake. The first two controls, elephant offtake and waterpoints, have two types of effects: (i) direct effects on elephant density, and (ii) indirect effects on roan antelope density through the influence of elephant density on roan antelope density (the state variables; Fig. 3c). The landowner could use waterpoints and elephant offtake to influence directly the elephant density and indirectly the roan antelope density. On the other hand, he could influence the roan antelope density directly by changing the roan antelope offtake, the third control, which would indirectly affect the elephant density. Although the two species model renders the modelling exercise complex, it brings some realism into this analytical model.

Just as in the case of the single species model, in the two species model, the effect of an increase in the price of elephant's offtake on the elephant's own interest rate, i.e. the returns to the elephant resource, 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 the roan antelope term are less than the total negative effects from the marginal benefits from the elephant's non-consumptive use term (Equation (17)). In such case, if the elephant offtake price increases, it would lead to a reduction in elephant population, and it would result in an increase in the roan antelope 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 relatively low. This would be in agreement with Clark's model (Clark, 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; Bulte and van Kooten, 1999). However, this would work if the consumer demand for elephant products is also suppressed or even eliminated coupled with strong anti-poaching management ('t Sas-Rolfes, 1997b). Given that the developing countries do not have the resources, the moratorium on ivory trade seems to be ineffective. This is because substantial ivory markets especially in Asia provide an incentive for continued illegal activity thereby increasing the price of ivory instead of lowering it ('t Sas-Rolfes, 1997a).

In the one species model without environmental costs, waterpoints are assumed to affect only the herbivore that uses them, and it ignores the indirect effects of waterpoints on other species

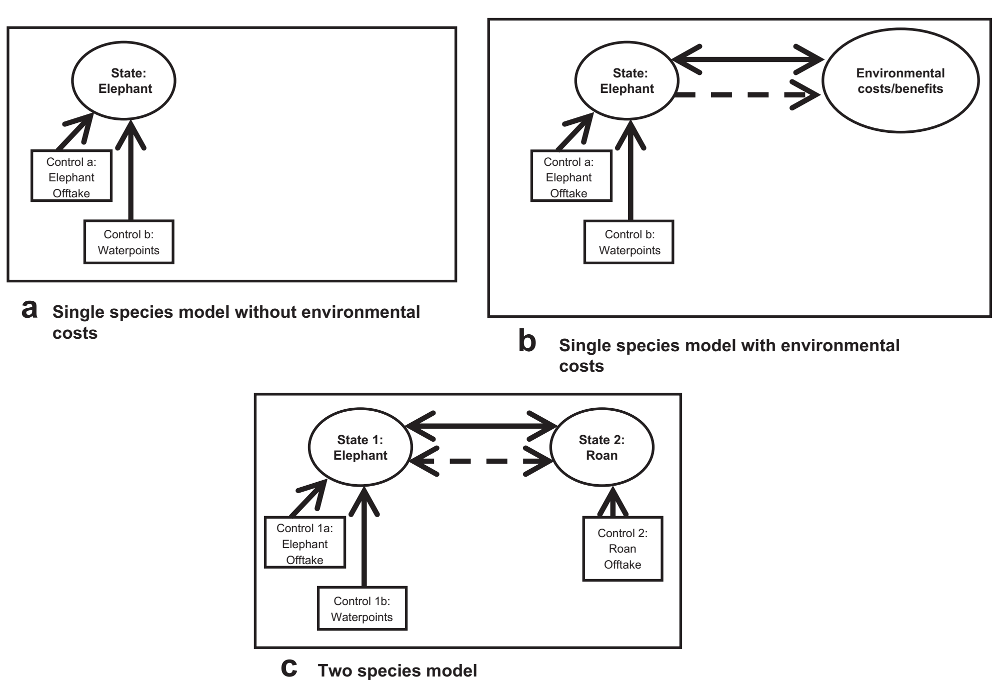

Fig. 3. Demonstrating the direct and indirect effects of controls on the models. Continuous arrows depict controls direct effects whilst broken arrow depicts controls indirect effects through the respective states. Predation is not included as a controlling mechanism in the model.

(Fig. 3). In the model where environmental costs are introduced, the indirect effects of waterpoints are captured in the elephant's own interest (Equation (10)) as externalities caused by an increasing number of elephants. Externalities are costs/benefits not captured in the price of a commodity but are incurred by a party who was not involved as either a buyer or seller of the goods or services causing those costs/benefits (Clark, 2005). However, this formulation (Equation (10)) 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 shows that there is an ecological–economic interaction of the species (Equations (17) and (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, over-riding 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 the elephant's marginal benefits from non-consumptive use will have a positive effect on the elephant's own interest rate plane, shifting it upwards (from OIR1 to OIR2 in Fig. 4), whilst the roan antelope's own interest rate plane will shift downwards as a result of more competition for forage as elephant density increases. On the other hand, if the roan antelope's non-consumptive use value is dominant, then an increase in the roan antelope's marginal benefit from non-consumptive use would encourage the landowner to shut down some waterpoints. This would result in an overall positive effect on the roan antelope's own interest rate, because the gain in marginal revenues from roan antelope's non-consumptive use effects will surpass the loss in other effects of the roan antelope's own interest rate. An increase in the marginal benefits from biodiversity could be from society's increase in perceived value of biodiversity (Fuller et al., 2007; Alho, 2008).

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 landowners to keep elephants at higher densities at the expense of biodiversity (Tisdell, 2004). However, it could be that the elephant density has already reached its optimum density such that a subsequent increase in elephant density leads to a decrease in biodiversity. In the one species model, this is not accommodated. However, in the one species model with environmental costs, it occurs when elephant density has reached its optimum density and the subsequent increase in elephant density leads to a decrease in biodiversity. In the two species model it is assumed that there is competition between the two species for forage. In such a 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. 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 a lower density of elephants. However, in cases, where both species have relatively moderate values, the landowner would sustain both species (Fleming and Alexander, 2003).

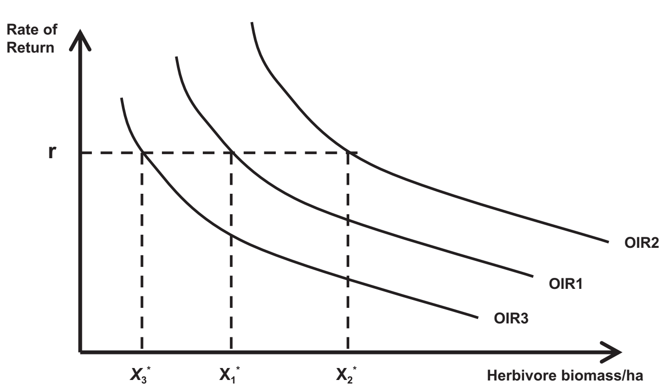

Fig. 4. The relationship between the rate of return and biomass for three cases where there is a shift in the "own interest rate" (OIR) curve due to a price change as discussed in the text.

The condition for obtaining an interior solution would be satisfied if the ecological competition between the two species is not intense. In addition, the conditions for an interior solution would hold if the marginal benefits from non-consumptive use between the two species does 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 antelope population. This places a very large value on the marginal benefits from non-consumptive use of elephants, T_1. At present, there is a strong campaign among preservationist organizations and animal rights movements for the preservation of elephants at all costs (Barnes, 1996; Tom, 2002; Moore, 2011). 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 the elephant's own interest rate (right hand side of Equation (17)) whilst reducing the biodiversity's own interest rate as represented by the roan antelope here (right hand side of Equation (18)).

4. Discussion

If the landowner only considers the species that is going to be affected directly by its control, then the indirect effects of such a control could have large consequences for 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 density level of the herbivore is reduced. By taking into consideration the externalities caused by 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 (Clark, 2005). Ignoring externalities may have severe long term consequences. For instance, the long term consequence of an excessive number of waterpoints is biodiversity loss. Thus, the landowner needs to consider both direct and indirect bio-economic effects of a control tool at his disposal. Ecological information of the direct and indirect effects of waterpoints on the different herbivore species should be provided to the landowners, so that they have advanced understanding of the ecological and economic impacts of their management decisions. Ecological studies should provide information to estimate the optimum waterpoint density for a given area. We encourage landowners to consider the complex influences that could be brought about by management decisions such as the construction of new waterpoints.

However, a landowner may be interested only in maximization of commercial profits from elephant offtake and tourism, and might ignore the negative effects on biodiversity that could be brought about by increasing elephant populations (Ntiamoa-Baidu, 1997; Tisdell, 2004). This is represented by the one species model that we have discussed in this paper. In this case the landowner would establish an excessive number of waterpoints, resulting in a more than optimum elephant population 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, 2004). This would compromise biodiversity in the long term. So the game reserve management as the authority entrusted with the sustainable management of the game reserve on behalf of all the landowners, should ensure that the landowner incorporates these externalities into his wildlife management activities. The game reserve management may do this by providing information, which would enable the landowner to take environmental costs into consideration while optimizing his management.

The game reserve management may use economic instruments such as subsidies or payments to the landowners who are complying with sound waterpoint management, and/or taxes or charges to landowners exceeding an optimal number of waterpoints per area (Tisdell, 2004). This is a command and control type method (The Royal Society, 2002). Theoretically, the number of waterpoints per area that is ecologically and economically optimal could be determined, so that a charge could be applied for additional waterpoints established and/or payment be paid for waterpoints closed below the optimum 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 (Brown, 2000).

By considering two species differentially affected by waterpoints, bio-economic effects could be determined and corrective action can be taken to avoid adverse effects of waterpoints on the species that is negatively affected. However, in an environment where there are various species differentially affected by waterpoints, the problem is more complex. Our model, although theoretical, provides a framework that could be used to analyse 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 analyse the bio-economic effects of these different species groups. Information would be required on how a control, such as waterpoints construction, affects the different species' groups directly and indirectly.

Estes et al. (2011) indicated the importance of trophic cascades as one of the major economic variables important in studying biodiversity. For example, Ripple and Beschta (2011) found that after the reintroduction of wolves in the Yellowstone National Park, after a 70-year absence, elk populations decreased, but that both beaver and bison numbers increased, possibly due to the increase in available woody plants and herbaceous forage resulting from less competition with elk. We acknowledge the importance of trophic cascades but for the tractability of the model, we did not include this component in the model. Moreover, large predators such as lions can regulate the densities of herbivore species, rather than or in addition to the availability of water. Although Wallach and O'Neill (2009) believe that research carried out so far does not demonstrate benefits of waterpoints closure, they do not deny however, that waterpoints could be an important factor in promoting or decreasing biodiversity. Inclusion of predation into the model would be subject of further work.

The results of our model are mainly applicable to large areas of wildlife management like large state owned national parks. The results are not valid for small fenced game reserves where all or some wildlife populations are deliberately fixed. In addition the model will only work at a game reserve management level, but not at individual landowner level. This is because if one landowner decides to close his waterpoints and the neighbours do not, then the elephants might not move. Furthermore, if there is a high density of permanent water sources, then closing a waterpoint might not generate substantial changes in the spatial distribution of elephants.

5. Conclusion

Our model provides a bio-economic optimization framework for analysing problems where a control has direct effects on one herbivore species but indirect effects on the other. Although the model includes several simplifications of reality, the findings are useful to wildlife management. Scientific theory is often developed from testing simple models in order to better understand complex systems (Noy-Meir, 1975; Starfield, 1997). It is clear that sound waterpoint strategy is required for the sustainable management of wildlife species.

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Appendix A. Derivations of Equations (5) and (6)

The landowner's objective is given by the following:

$$\underset{q,W}{\mathrm{Max}}\, PV\Pi = \int_0^\infty [pqX + TX - ZW]e^{-rt}\mathrm{d}t \tag{A1}$$

Subject to

$$\frac{\mathrm{d}X}{\mathrm{d}t} = h(X,W) - qX \tag{A2}$$

The current value Hamiltonian:

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

Pontryagin's necessary conditions for a maximum are:

$$\frac{\partial H}{\partial q} = pX - \lambda X = 0 \tag{A4}$$

$$\frac{\partial H}{\partial W} = -Z + \lambda h^W(X,W) = 0 \tag{A5}$$

$$\frac{\partial H}{\partial X} = pq + T + \lambda\left[-q + h^X(X,W)\right] \tag{A6}$$

$$-\frac{\partial H}{\partial X} = \dot{\lambda} - r\lambda \Rightarrow \dot{\lambda} = r\lambda - \frac{\partial H}{\partial X} \tag{A7}$$

$$\frac{\partial H}{\partial \lambda} = -qX + h(X,W) = 0 \Rightarrow q = \frac{h(X,W)}{X} \tag{A8}$$

Solve Equation (A4) for λ

$$\lambda = p \tag{A9}$$

Take d/dt of Equation (4)

$$\dot{\lambda} = \frac{d}{dt}[p] \tag{A10}$$

Substituting Equation (A9) into Equations (A5) and (A7)

$$-Z + ph^W(X,W) = 0 \tag{A11}$$

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

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)–(A12) equal to zero. Given these conditions and substituting for q as shown in Equation (A8) into Equation (A11), solve Equations (A11) and (A12).

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

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

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