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Property Rights and Natural Resource Conservation. A Bioeconomic Model with numerical illustrations from the Serengeti-Mara Ecosystem.

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Abstract

This study develops a model for wildlife species migrating seasonally between a conservation area and a neighbouring area. When being outside the conservation area, harvesting takes place by a group of peasants. The local people have two motives for harvesting; to get rid of ‘problem’ animals as roaming wildlife destroys crops and agricultural products, and hunting for meat and trophies. Depending on the specification of the property rights, the harvesting is legal or illegal. It is demonstrated that it is far from clear which of the two property rights regimes that gives the highest wildlife abundance. Hence, contrary to what is argued for in the literature, handing the property rights over to the local people means not automatically more wildlife and a more ‘sustainable’ resource utilisation. The reason lies in the nuisance motive for harvesting. The exploitation under the two different property rights regimes are illustrated by numerical calculations with data that fits reasonable well with the exploitation of the wildebeest population in the Serengeti-Mara ecosystem.

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1. Introduction

For a long time it has been recognised that institutions play an important role in natural resource management, and that the specification and function of property rights to a large extent determine whether resources can be utilised in a sustainable way. These dimensions will be at the focus in the present study when analysing the management and exploitation of wildlife in a sub-Saharan Africa context with example from the Serengeti-Mara ecosystem. When considering natural resources in the form of wildlife in this region, as in other regions in the sub-Saharan Africa, central issues are the behaviour of the rural people living close to the wildlife, and the interaction between the rural people, the wildlife and the agency managing and having the legal property rights of the wildlife. Often this interaction represents conflicting interests; both the legal owner of the wildlife (usually the State, or large private landowners) and the rural people claim their rights to reap the benefits of the huge amount of wildlife resources. In addition, the costs of having abundant wildlife populations differ between them. These conflicts, rooted in the prevailing property structure and its functioning, have serious implications for the resource exploitation and, thus, on the management of the wildlife in general (Marks 1984, Kiss 1990, Swanson and Barbier 1992, Naughton-Treves and Sanderson 1995, Sinclair and Arcese 1995, Skonhofs and Solstad 1998, Bulte and van Kooten 1999).

The common perception in the literature is that local communities will support wildlife conservation and reduce the wildlife offtake if they are ensured a sufficient share of the benefits from wildlife (see Kiss 1990, Swanson and Barbier 1992, Mangel *et al.* 1996). Under what conditions a community based management system results in a higher wildlife abundance and more conservation than the polar scheme where the local people have no legal rights to wildlife exploitation, is analysed in the present paper. The starting point is that we have a protected area, a national park or a conservation area of fixed size, with no harvesting of the wildlife. The protected area is the basic living area of the species, but they roam freely in and out of the park. When being outside, the game destroys the crops of the peasants living in the vicinity of the park and hence, the wildlife represents a nuisance for the local people. The park agency has the property rights of the wildlife within as well as outside the protected area, but illegal harvesting takes place outside as the property rights is not effectively

protected here. This is the first regime where the local people have no property rights¹. In the next step, the property rights are handed over to the local people and hence, the exploitation of the wildlife outside the protected area takes place in a legal manner². The different degree of wildlife utilisation is compared under these two regimes, and contrary to what is argued for in the literature the first property rights scheme may result in the highest wildlife abundance. Analysing the driving forces behind such an outcome is the main contribution of this paper.

The hunting activity and the utilisation of the wildlife under these two regimes are exemplified by the migration of wild herbivores in the Serengeti-Mara ecosystem. The migration pattern here, which is seasonal and related to rainfall and food supply, is also used as a motivation for the ecological part of the model. Parts of the year the wildlife is staying within the huge Serengeti national park and it is assumed that no human extraction takes place here³. However, when being outside there is hunting. The conceptual framework for analysing the exploitation of terrestrial species when there is migration, and hunting is a seasonal activity, is another contribution of the paper.

To simplify the following analysis we consider only two areas; the protected area and the whole boundary region collapsed into one boundary area. The protected area is the basic living area of the wildlife and is owned and managed by the State. No human extraction is allowed here, and hence, non-consumptive benefit (tourism, existence value, etc.) is the only benefit of this area. However, the park manager play no role in the subsequent model as the management of this area is supposed to

¹Economists frequently confuse what is meant with property rights. Daniel Bromley understands property as a benefit stream and a property right as ‘the capacity to call upon the collective to stand behind one’s claim to a benefit stream’ (Bromley 1991, p.15). Possession of inviolable property rights presupposes that the rights are authorised by law and that the law is effectively enforced by the state. Hence, a well-functioning property regime is characterised by, firstly, legally *welldefined* property rights and, secondly, effectively *protected* property rights. The existence or non-existence of these two factors defining the functioning of the property structure can be used to classify three different types of regimes. First of all we have the case where there is legally well-defined ownership and perfect state protection (exclusive rights, perfect law enforcement). This is in accordance with the classical Clark (1973) model. Secondly, we have the case where the ownership is legally well defined but not adequately enforced by the government. Under such a scheme, often due to lack of societal recognition of the property rights in place, the management is likely to be affected by conflicting property rights claims. Finally, we have the case where there is no legally defined ownership; that is, the open-access case regime (Gordon 1954). Hence, the second of these regimes is the starting point in the following analysis.

²These schemes are polar because the local people obviously can obtain other rights than full property rights of the wildlife when being outside the park, say, through a specific harvesting quota, the rights to harvest problem animals, and so forth (cf. the concluding section).

³In reality, poaching takes place within as well as outside the borders of Serengeti National Park (see Sinclair and Arcese 1995). However, to simplify the theoretical framework, without altering the qualitative results, we assume that hunting is only taking place in the outer area.

have no stock effects (no harvesting, no culling)⁴. The peasants are living in the boundary area, where the wildlife roams during the migration season. They are involved in two production activities; agricultural crop production and wildlife hunting (see, e.g., Barrett and Arcese 1998). First, we study today's situation where the local people have no property rights, and hence, they hunt illegal. This is in accordance with the 'open-access' model of Smith (1975). Under this scenario we disregard any enforcement use and consequently, the park manager is passively benefiting from non-consumptive utilisation of wildlife in the protected area. Because of small funds and large areas, this represents a good approximation to the present reality (see also footnote 1). In a next step, we study what happens when the property rights over the wildlife are given to the local people.

We start by formulating the ecological model and the migration pattern in section 2. In section 3 the benefit function of the local people is presented, and in section 4 it is studied how this translates into harvesting when they have no property rights. In this section, as later, it will for simplicity be assumed that only the current benefit, or current utility, is steering the harvesting activity. In section 5 the model is analysed when the local people have the property rights over the wildlife. In section 6 the two different property rights regimes are illustrated by numerical calculations with data that fits reasonable well to the exploitation of the wildebeest population in the Serengeti-Mara ecosystem.

2. The ecology

As already noted, we consider species migrating over the year in a well defined way, and where harvesting only takes place when the species are outside their core area, i.e., the conservation area or the national park. Because the seasonal migration is related to rainfall and food supply, there is no density dependent factors influencing the species flow. For simplicity it is assumed that the whole wildlife population is leaving the core area, and we separate sharply between the periods of the year when the wildlife is roaming outside and when the species are staying inside of the protected area. These two periods are synonymous with the 'fishery season' and 'between season' in Homans and Wilen (1997)⁵. We start by formulating the hunting season dynamics while the between season dynamics and the steady state follow next.

⁴ For a critical review of this assumption, see, e.g., Wright (1999).

⁵ There are few, if any, papers analysing time discrete models with migration for terrestrial animal species within a bio-economic concept. Fancy *et al.* (1994) formulate a time discrete migration model for the porcupine caribou herd. The migration here is also seasonal, but this paper has no economic content.

Migration and harvesting

As in Homans and Wilen (1997) (see also Getz and Haight 1989), we suppose that the hunting effort of the local people is utilised at a constant rate throughout the period when hunting takes place, i.e., when the wildlife is roaming outside the conservation area. The wildlife harvesting function is specified as a Schaefer function giving the harvest at time t as

$$(1) \quad h(t) = qaX(t),$$

where $X(t)$ is the wildlife stock (measured as biomass), while a is the fixed harvesting effort. q is a parameter, the so-called catchability coefficient.

When being within the conservation area the wildlife grows in a density dependent fashion (see below), and when being outside natural mortality is ignored⁶. Outside the conservation area the stock therefore shrinks according to $dX(t)/dt = -qaX(t)$. The stock level outside the conservation area at time t is accordingly

$$(2) \quad X(t) = X_0 e^{-qat}$$

where X_0 is the stock level when the species starts roaming outside the protected area.

The length of the migration period, i.e., the time when being outside the conservation area, is fixed as T . The stock level when the wildlife is returning back to the park is therefore

$$(3) \quad X_T = X_0 e^{-qaT} = X_0 s$$

where $0 < s = e^{-qaT} \leq 1$ is the survival rate of the species when being outside the conservation area.

Under these assumptions, total amount of wildlife harvested is

$$(4) \quad H = X_0 - X_T = X_0(1 - s).$$

Natural growth and ecological equilibrium

So far we have studied a single period migration, where the stock size X_0 at the beginning of the period of migration is predetermined. However, as the number of animals harvested depends on the initial stock, the harvest changes when X_0 changes. The factors affecting the initial stock size are found by analysing the dynamics between the seasons of migration. We assume that the conservation area is the basic living area where calving takes place, meaning that the stock size returning back determines the natural growth. When $X_{T,t}$ is the stock returning back to the conservation area at time t , the density dependent growth is accordingly determined as $F(X_{T,t})$. We represent natural growth by a logistic function with K as the carrying capacity and r as the maximum specific growth rate. Hence, if $X_{0,t+1}$ is the stock level at the beginning of the next period of migration, the dynamics between the migration seasons is given by

$$(5) \quad X_{0,t+1} = X_{T,t} + F(X_{T,t}) = X_{T,t} + rX_{T,t}(1 - X_{T,t}/K).$$

Equation (5) together with the balance equation (4) written as $H_{t+1} = X_{0,t+1} - X_{T,t+1}$, yields $H_{t+1} + X_{T,t+1} = X_{T,t} + F(X_{T,t})$. Ecological equilibrium is defined by a constant wildlife stock returning back to the park area, $X_{T,t} = X_{T,t+1} = X_T$ ⁷. In steady-state, natural growth, taking place within the conservation area, is therefore equal to harvesting, taking place in the surrounding area, $F(X_T) = H$. When substituting for the logistic growth function and equation (3) and using (4), we obtain $rX_0s(1 - X_0s/K) = X_0(1 - s)$. Consequently, the ecological equilibrium is given by $X_0 = 0$ and

⁶ Introducing natural mortality makes the model quite more complicated, but it will not change the qualitative content of the analysis.

⁷ Combining equation (5) and equation (3) written as $X_{T,t+1} = X_{=t+1}s$ yields

$X_{T,t+1} = s[X_{T,t} + F(X_{T,t})] = G(X_{T,t})$. It is well known (see, e.g., Clark 1990, Ch.7) that the condition for stability is that $-1 < G' < 1$ holds at the equilibrium. We have $G' = s(I + F') = s[I + (r - 2rX_{T,t}/K)]$ which at the equilibrium reads $s[2/s - (I + r)]$ when inserting for X_T (see the main text). Hence, stability demands $I/s - (I + r) < 0$ and $3/s - (I + r) > 0$. The first of these conditions is therefore the same as the condition for obtaining an interior steady-state, while the second condition is fulfilled if $\partial X_0 / \partial a < 0$ holds (see the main text).

$$(6) \quad X_0 = (K / rs)(I + r - I / s)$$

with $(1 + r - I / s) > 0$. From this equation it follows that the equilibrium stock returning back to the conservation area is equal to $X_T = (K / r)(I + r - I / s)$. Because more harvesting effort reduces the survival rate s , more effort also clearly reduces the stock size, $\partial X_T / \partial a < 0$. From the equilibrium condition $X_0 = X_T + F(X_T)$, we also have $\partial X_0 / \partial a < 0$ as long as $-I < F'$. When evaluating F' at the equilibrium, or differentiating equation (6) directly, this condition also writes $[2 / s - (I + r)] > 0$. In what follows, this is assumed to hold.

3. Agricultural production, crop damages, and harvesting

Above we established the ecological steady-state for a fixed amount of hunting effort. However, the hunting activity is determined by economic considerations and now these motives are outlined. Throughout we will think of the local people as a homogeneous group of peasants living in the boundary area of the park⁸. They are involved in two different production activities; agricultural production and wildlife hunting. In the model, as in reality, there are two basic motives behind hunting. In addition to the already mentioned nuisance motive where hunting takes place to get rid of 'problem' animals destroying crops and agricultural products, there is also hunting for meat or, occasionally, for trophies (Marks 1984, Barrett and Arcese 1998). As noted, because of small funds and large areas, the harvesting, when being illegal, takes place in an environment lacking any enforcement use.

We first look at the nuisance motive working through the agricultural yield function. The maximum agricultural yield, i.e., the yield without damages, depends on the amount of agricultural land, pesticides and fertiliser use, rainfall, etc., and effort use. Keeping all factors fixed except of effort, the yield function reads

⁸It is therefore assumed that there are no conflicting interests among them. Hence, prevalence of individual conformity to group norms is assumed to be present. In line with traditional reasoning, it is assumed that the elders are in charge of the group's activities (Marks 1984). This obviously means that the scenario of property rights to the local people in section 5 is implemented in its most optimistically way (cf. the concluding section).

$$(7) \quad A = A(N)$$

here N is the *total* (cumulative) labour input in agricultural production over the year (seeding, harvesting and maintaining the crop). A is an increasing function of N with $A(0) = 0$, but at a decreasing rate, so that $A'(N) > 0$ and $A''(N) \leq 0$ hold.

More wildlife means more nuisance, and following Zivin *et al.* (2000) we assume that the damages are proportional to the amount of wildlife. Wildlife consumes a fraction \mathbf{b} of its body weight in forage dry matter per day, and a fraction \mathbf{g} of this is eaten from crops. The nuisance stream at time t is therefore $\mathbf{gb}X(t)$ with $0 < \mathbf{b} < 1$ and $0 \leq \mathbf{g} \leq 1$ as fixed constants⁹. Consequently, the cumulative damage of the wildlife roaming outside the conservation area is

$$(8) \quad D = \mathbf{gb}X_0(1-s)/qa$$

when using equation (2) and integrating. If P_A is the crop price, assumed to be fixed, and leaving out fertiliser and pesticides costs, etc., the net crop benefit is

$$(9) \quad B = P_A [A(N) - D],$$

with D defined in the interval $[0, A(N)]$ as the damages can never exceed the yield. The nuisance motive for harvesting is therefore working through D . More harvesting reduces the damages for a given stock size as the total amount of harvested animals increases.

In addition to the nuisance motive for hunting, we have the direct benefit of hunting in the form of meat (and trophies). The hunting benefit is given as

$$(10) \quad V = P_H H = P_H X_0(1-s)$$

⁹ We assume that hunting is the only damage control performed by the local people (see below). In addition, and in reality, damage control is also performed through fencing and other measures more directly related to protecting the crop. In the model this would have worked through \mathbf{g} . As we are neglecting such measures, \mathbf{g} is assumed to be constant.

where P_H is the marginal valuation, or price, of the offtake, also assumed to be fixed.

There will be a constraint on effort used in hunting and crop growing. As a is hunting effort per unit of time and T is the length of the migration and hunting season, this resource constraint reads

$$(11) \quad L \geq aT + N.$$

L is the total available time per year, which also can be interpreted as the total human population living in the vicinity of the conservation area. Throughout the analysis L is fixed, meaning that we are ignoring any, if possible, Malthusian mechanism. Moreover, all the time the constraint is assumed to be binding, and hence, there is always a positive opportunity cost of labour use.

When using the resource constraint, we obtain the total current benefit of the local people as

$$(12) \quad U = P_H X_0(1-s) + P_A [A(L - aT) - gb X_0(1-s) / qa].$$

In the next section, we determine the optimal harvesting effort and wildlife population when the local people have no property rights to the wildlife roaming in the outer area. In section 5, we study the polar case when they are given full property rights.

4. No property rights of the local people

Assuming that the local people have no property rights corresponds, as already mentioned, to the stylised facts situation in sub-Saharan Africa today. When having no property rights and obtaining no legal benefit from the wildlife, they have few, if any, incentives to base their harvesting on long-term considerations. Technically, this means that they do not take the stock of wildlife into account when deciding their effort use, and hence, the number of animals entering the boundary area is treated as an exogenous variable. As already mentioned, this corresponds to what Smith (1975) calls an ‘open-access’ solution. See also Skonhøft and Solstad (1998).

Maximising the benefit function (12) with respect to the control a , when keeping X_0 exogenous, yields the first order condition

$$(13) \quad P_A A'(N)T - P_A \mathbf{g} b X_0 C / a = P_H X_0 q T s$$

when we have an interior solution so that effort is allocated to both production activities. C collects terms, and reads $C = (1/qa)[1 - (1 + qaT)s] > 0$ ¹⁰. The optimal hunting effort is therefore determined by the equality between the marginal benefit in agricultural production, corrected for damages of the roaming wildlife, and the marginal hunting benefit. This economic equilibrium is illustrated in Figure 1 where the marginal net benefit from agricultural production is measured along the left hand vertical axis, while the marginal hunting benefit is measured along the right hand axis. The second term on the left hand side of condition (13) reflects the nuisance motive caused by crop damages. Crop damages reduce the marginal profitability in agricultural production, and consequently, the local people channel more effort to hunting compared to a situation when crop damages are absent and $\mathbf{g} = 0$. This is illustrated by the downward shift in the marginal net agricultural benefit curve in the figure.

Figure 1 about here

The economic equilibrium condition (13) together with the ecological equilibrium condition (6) defines the equilibrium values of hunting effort a^∞ and the initial stock size X_0^∞ (superscript ‘ ∞ ’ denotes no property rights of the local people). In a next step, the equilibrium wildlife population returning back to the park area at the end of the season X_T^∞ is found by using equation (3). When taking the total differential of these equations, the economic and ecological forces at work can be demonstrated (for details, see Appendix 1). As already indicated, a higher nuisance effect reduces the marginal productivity of agricultural production and, hence, the local people increase their harvesting effort, $\partial a^\infty / \partial \mathbf{g} > 0$. Consequently, the initial stock X_0^∞ decreases, and this will also be so for X_T^∞ . A higher price of game meat will also motivate for more hunting effort, $\partial a^\infty / \partial P_H > 0$,

¹⁰ As noticed above, the survival rate is equal to $s = e^{-qaT}$. Hence, $C > 0$ for all $T > 0$.

and hence, X_T^∞ as well as X_0^∞ decrease. A higher crop price causes the local people to divert more effort towards agriculture and the stock will increase, $\partial X_0^\infty / \partial P_H > 0$. This result is in line with the reasoning of Brown *et al.* (1993) and the analysis in Skonhoft and Solstad (1998), who find that better economic conditions in the agricultural sector always works in the direction of more wildlife conservation when the land-use, as here, is fixed. The effect of a higher T is ambiguous. First, a longer migration period reduces the productivity of hunting as an instrument of damage control. This causes the local people to channel more effort towards agricultural production. Second, the marginal product of hunting for meat increases with the period of migration, leading the local people to increase their hunting effort. Consequently, if the costs of living with wildlife exceed the benefit, then more effort is directed to agriculture. However, the effect on total hunting effort (aT) and wildlife conservation is unclear. For other results, see Table 1.

Table 1 about here

5. Property rights to the local people

Above the harvesting activity and stock sizes were found when the local people had no property rights and hence, the harvesting activity was not based on long-term considerations. Now we proceed to study what happens in the polar case when the property rights are handed over to the local people and they do no longer harvest illegally. Giving the property rights to the local people means that they are investing in wildlife and hence, take the wildlife abundance into account when allocating effort among the two production activities. Technically, the problem now is therefore to maximise (12) with respect to a , subject to the ecological constraint (6)¹¹.

The first order condition for this problem is

$$(14) \quad \begin{aligned} & P_A A'(N)T - P_A \mathbf{g}bX_0 C / a \\ & - [P_A \mathbf{g}b - P_H qa]qT(1-s)K[2/s - (1+r)] / qars = P_H X_0 qTs \end{aligned}$$

when still assuming an interior solution. This equation together with equation (6) determines the equilibrium effort a^* and equilibrium stock size X_0^* while X_T^* again follows from equation (3) (superscript ‘*’ denotes the property rights case). The difference compared to the economic equilibrium condition (13) in the case of no local property rights, is the third term on the left hand side. $[2/s - (1+r)]$ is here positive because $\partial X_0 / \partial a < 0$ holds (see section 2). The new term, depending on the sign of $[P_A \mathbf{g}\mathbf{b} - P_H qa]$, can therefore be either positive or negative, and hence, a^* can be below as well as above that of a^∞ . Consequently, X_0^* can also be above or below that of X_0^∞ .

The general conclusion following the model is therefore that it is unclear whether handing the property rights over to the local people will result in a higher wildlife abundance and a more ‘sustainable’ resource utilisation. On the one hand, having the property rights and taking the size of the wildlife into account when determining hunting effort, work in the direction of a higher stock size. However, at the same time, crop damages related to the size of the wildlife will also be taken into account, and this works in the other direction. The actual outcome depends therefore critically on whether the marginal nuisance of the wildlife stock dominates the marginal harvesting benefit; that is, the sign of the term $[P_A \mathbf{g}\mathbf{b} - P_H qa]$ above. If $P_H qa > P_A \mathbf{g}\mathbf{b}$ holds we obtain the standard result from the literature saying that the presence of property rights means less harvesting effort and more wildlife conservation compared to the no property rights case; that is, $a^* < a^\infty$ and $X_0^* > X_0^\infty$. On the other hand, when the marginal nuisance dominates the marginal harvesting productivity, the nuisance case, we obtain $a^* > a^\infty$. Hence, taking the nuisance into account yields less wildlife than not doing so, $X_0^* < X_0^\infty$.

The comparative static results are now found by taking the total differential of equations (14) and (6) (for details, see again Appendix 1). An increased nuisance increases the agricultural damage and, hence, reduces the marginal benefit from crop production. This direct effect is the same as in the no property rights case, and increases the equilibrium hunting effort. In addition there is now an indirect

¹¹ A shadow price is therefore now imposed on the wildlife. It can easily be shown that it is positive only as long as $[P_A \mathbf{g}\mathbf{b} - P_H qa] > 0$ holds; that is, the non-nuisance case (see the main text below). Accordingly, the shadow price is zero in the no property rights case.

effect working through the stock size at the beginning of the next hunting season, which reinforces the direct effect. $\partial a^* / \partial g$ is therefore stronger when it is local property rights. The direct effect of a higher game meat price is more benefit from hunting. As for the no property rights regime, this motivates for increased hunting and stock depletion, which is the same result as in the standard harvesting model (see, e.g., Clark 1973). Now, however, this effect is weakened because a higher offtake reduces the crop damage and hence, motivates for more effort allocated to crop production because the crop damages are taken into account. Thus, the indirect effect is of the opposite and the sign of $\partial a^* / \partial P_H$ is ambiguous. In the same way, the effect of a higher agricultural price is ambiguous. These results contrast the no property case, but are in line with results from harvesting models with competing uses among different assets (see, e.g., Swallow 1990). Other results are again given in Table 1, and as seen, the outcomes are to a large extent ambiguous.

6. Numerical illustrations

The theoretical reasoning will now be illustrated by data that fits reasonable well with the exploitation of the wildebeest population in the Serengeti-Mara ecosystem. This ecosystem covers an area of some 25,000 km² on the border of Tanzania and Kenya (Sinclair 1995). The Serengeti National Park is a part of it, and is wide known for the migration of its large herbivore populations of which the wildebeest has been most in the focus. Each year about a million wildebeest migrate across the Serengeti-Mara ecosystem (Murray 1995). The overall migratory pattern is supposed to be related to food supply, which in turn is connected to rainfall. The Serengeti ecosystem can be divided into two main regions; the southern short grasslands with low annual rainfall and the wooded northern grassland with higher rainfall (Fryxell 1995). The migratory wildebeest use the short grasslands in the south during the wet season and the tall grassland in the north during the dry season (Sinclair 1984 and 1995, Fryxell 1995).

Figure 2 about here

The migratory herds know no boundaries, and make extensive use not only of the gazetted land, but also the open areas in the districts outside the borders of Serengeti National Park (cf. Figure 2). During the migration they spread beyond the park into the western frontier and enter land settled by humans. This side of Serengeti National Park is densely populated, and the population is increasing

(Barrett and Arcese 1998). As a consequence, there are threats to the species diversity and the ecosystem because the landscape is modified and habitat land is converted into agricultural use (Sinclair and Arcese 1995). The detrimental effects of the human-wildlife interaction are, however, not only one-way. The local people and the local communities also bear costs from the high wildlife abundance as the large herds of migrating herbivores induce crop damages (Emerton and Mfunda 1999). Villagers protect their crops and compensate themselves by hunting; in addition they also hunt for meat (Arcese *et. al.* 1995). Hunting in this region is basically illegal¹². Therefore, living with wildlife is twofold; it represents a cost to the local peasants due to crop damages while it represents a benefit due to illegal harvesting.

The ecological model, using the Serengeti-Mara ecosystem as an illustration, is specified for the numerical analysis at the scale of one km² and one year. The same scale is also used for the agricultural benefit as well as the hunting benefit. The closer definition of the protected area and the outer area is found in Appendix 2. The baseline values for prices and costs together with ecological data and data for crop production used in the simulations are also presented here. As demonstrated above, the conservation effect of giving the local people property rights depends critically on the costs and benefits of living with wildlife. Because of the unclear nuisance effects, the damage coefficient g will be varied throughout the simulations where a low value of g , as mentioned, may be interpreted as more extensive use of fencing or other measures taken to protect crop production from the roaming wildlife. In addition, because of the unclear price effect under the property rights scenario, we will also vary the crop price P_A . The conservation effect of shifting the hunting price P_H together with changing ecological and productivity conditions will be studied as well.

Figure 3 demonstrates how the wildlife abundance varies with the crop damage coefficient g under both property right schemes. More nuisance means less wildlife, and hence, the conservation-damage schedule slopes down under both scenarios. However, in accordance with the theoretical reasoning, the nuisance effect is quite more substantial under the local property rights regime. The figure also demonstrates that handing the property rights over to the local people gives more wildlife only when the crop damages are quite low. Hence, only when $g < 0.01$ and less than 1% of the

¹² Harvesting is not strictly illegal in parts of the western side of the Serengeti National Park, and persons having a car and firearms can obtain a hunting licence (Rugumayo 1996).

forage is eaten from the crop, local property rights results in higher wildlife abundance. Within this range the marginal benefit of living with wildlife exceeds the marginal cost, i.e., the term $[P_A g b - P_H q a]$ from section five is negative. The figure also demonstrates that the nuisance effect is so strong that the presence of local property rights means that the wildlife stock will be totally depleted when about 9 % and more of the forage is eaten from the crop. Hence, according to the model and the imposed parameter values where the baseline value of the damage coefficient is 12 %, $g = 0.12$ (see Appendix 2), the presence of local property rights would have represented an extinction of the wildlife.

Figure 3 about here

Hence, handing the property rights over to the local community is not recommendable as a strategy to promote wildebeest conservation in Serengeti. Instead, as seen from the figure, damage control, which reduces the costs of living with wildlife, increases the degree of wildlife conservation within the present property rights regime. For instance, a total eradication of crop damages from today's level (the damage coefficient reduces from $g=0.12$ to $g = 0.00$), increases wildlife abundance by somewhat above 20 %. This effect can be further strengthened when combined with, say, increased productivity and profitability in agricultural production (see below).

A positive shift in the game meat price P_H increases the offtake and lowers the species abundance under both property rights schemes, and hence, both conservation-damage schedules in Figure 3 shifts down. As expected from the theoretical model, however, the effect is stronger under the no property rights scheme, and when doubling P_H from today's level of species abundance the schemes coincide at $g = 0.07$. Accordingly, local property rights means now more conservation up to the point where 7 % of the forage is eaten from the crop. On the other hand, shifting up the crop price P_A from its baseline value works in the direction of higher wildlife abundance and more conservation under both regimes; i.e., both conservation-damage schedules in Figure 3 shift up. Contrary to what is expected from the theoretical reasoning, however, the effect is stronger under the property rights regime, and the same degree of conservation is present for $g = 0.03$ when P_A is doubled. Shifting up the crop productivity works in the same manner. A positive shift in the

catchability coefficient q due to, say, improved hunting technology, also means that the same degree of conservation is present for a higher value for the damage coefficient. As both conservation-damage schedules shift down, the wildlife abundance is reduced under both regimes. We have also studied the effects of shifting the migration period T , interpreted as taken place as a result of, say, drought. The conservation-damages schedules are only modestly influenced compared to the baseline case.

7. Discussion and concluding remarks

There is a growing recognition that a viable and sustainable wildlife utilisation in the future depends on the support of the local communities living close to the wildlife, and during the last years community based wildlife management projects have increasingly become one of the means for safeguarding wildlife in sub-Saharan Africa (Kiss, 1990, Swanson and Barbier 1992). The basic idea behind these schemes is to engender the co-operation of local communities in wildlife conservation and wildlife management by ensuring that parts of the benefits from wildlife utilisation go to the local communities. There are several ways in which benefit sharing can take place; through revenue sharing from tourism, safari hunting, or establishing user rights through hunting quotas (see Skonhofs 1998 for an analysis), or through local job creation in tourism, wildlife and park services. The experiences from two prototypes of community based projects are summarised by Kiss (1990), namely Windfall and Campfire (both in Zimbabwe).

What has been analysed in this paper is a more far-reaching community based management system as the local people have been given the full property rights of the wildlife; that is, they control and obtain the *whole* benefit stream from the wildlife when it is outside the protected area. This management scheme has been compared to the polar one of having no property rights. Under the scenario of no property rights, the local people have no incentive to take into account that their hunting today influences the wildlife stock and hence, reduces the potential for hunting next year. Technically, the stock size is then treated as an exogenous variable when allocating hunting effort, and is in line with the ‘open-access’ solution of Smith (1975). In the next step, when assuming that the property rights are handed over to the local people, they have incentives to invest in the wildlife stock, and take the stock size into account when hunting. Smith (1975) specifies this scenario within an inter-temporal framework where the present-value benefit is maximised. The present exposition

where the local people consider the wildlife stock as an endogenous variable under ecological equilibrium coincides with the steady-state of the Smith model when the discount rate is zero (see, e.g., Munro and Scott 1985). Throughout the analysis it is assumed that the local people is a homogenous group with no internal conflicting interests. That is, the property rights scenario is implemented in its most optimistically way.

The main point of the analysis is to find under what conditions community based management and property rights of the local people might result in a higher wildlife abundance compared to the no property rights scenario. It is shown that the actual outcome depends critically on the difference between the benefit and cost of living with wildlife; that is, the marginal harvesting value versus the marginal crop damage. Only when the nuisance from the roaming wildlife is small and the marginal benefit exceeds the cost, we find that community based management increases the wildlife stock compared to the scenario of no property rights. The main message from the theoretical analysis is therefore that relying on local property rights alone as a tool in wildlife conservation may not work. This conclusion contrasts Kiss (1990) and Swanson and Barbier (1992) who argue that local property rights generally will promote wildlife conservation.

The theoretical model is illustrated by data that represents the Serengeti-Mara ecosystem in a stylised manner. For the baseline parameter values we find that the nuisance from the roaming wildlife dominates the benefit, and hence, local property rights gives incentives to deplete the roaming stock of wildebeest. Consequently, from a conservation perspective, instead of giving the local people full control of the wildlife resources, management authorities should protect crop production by supporting fencing or take other types of control measures to reduce the damages caused by the roaming wildlife. The numerical simulations also demonstrate that an increased crop price or higher crop productivity through, say, improved fertiliser and pesticides use, may lead the local people to channel more effort towards agricultural production. Hence, such measures will also work in the direction of more wildlife conservation. This conclusion is in line with Barrett and Arcese (1998) who find that to succeed promoting wildebeest conservation in Serengeti, wildlife management schemes must attempt to increase local people's benefit from alternative activities.

The focus of the present analysis has been property rights and wildlife conservation. The basic idea behind giving the local people full property rights over the wildlife is, however, to promote wildlife conservation *together* with economic development and increased welfare (see, e.g., Kiss 1990). Obviously, everything else the same, the presence of property rights increases the welfare of the local people compared to the scenario of no property rights. When the damage of the roaming wildlife is small so that the benefit of living with wildlife exceeds the cost, we can therefore conclude that the presence of local property rights promotes wildlife conservation as well as local welfare. On the contrary, local welfare is promoted at the expense of wildlife conservation when the cost of living with wildlife exceeds the benefit.

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Appendix 1

Comparative static results

The comparative static of the no property rights scenario of the local people is found by taking the total differential of (6) and (13). The result is

(A1)

$$\begin{aligned} & \begin{bmatrix} 1 & -[1+r-2/s]KqT/rs \\ P_HqTs + P_A\mathbf{g}bC/a & Q \end{bmatrix} \begin{bmatrix} dX_0 \\ da \end{bmatrix} = \begin{bmatrix} 0 \\ -P_A\mathbf{b}X_0C/a \end{bmatrix} d\mathbf{g} + \begin{bmatrix} 0 \\ -qX_0Ts \end{bmatrix} dP_H \\ & + \begin{bmatrix} 0 \\ A'(N)T - \mathbf{g}bX_0C/a \end{bmatrix} dP_A + \begin{bmatrix} qaK[1+r-2/s]/rs \\ G \end{bmatrix} dT \end{aligned}$$

where $Q = -P_H X_0 (qT)^2 s + P_A A'' T^2 + P_A \mathbf{g}b X_0 qTs / a < 0$ from the second order maximum conditions while the sign of $G = P_A A'(N) - P_A A''(N) aT - s(1 - qaT) X_0 [P_H qa - P_A \mathbf{g}b] / a$ is ambiguous. The determinant of the system $Q + KqT [P_H qTs + P_A \mathbf{g}b C / a] [1 + r - 2 / s] / rs$ is also negative due to the second order conditions.

The comparative static results when having property rights are found when taking the total differentiation of (14) and (6). Equations (A2)-(A4) give the stock effects a where $Z > 0$ from the second order maximum condition.

$$(A2) \quad dX_0 / dg = P_A \mathbf{b} [X_0 C - (1-s)TK[(r+1)-2/s]/rs] qTK[(r+1)-2/s] / Zras < 0$$

$$(A3) \quad dX_0 / dP_H = qT[X_0 s + (1-s)K[(r+1)-2/s]/rs] qTK[(r+1)-2/s] / Zras$$

$$(A4) \quad dX_0 / dP_A = \left\{ \begin{array}{l} \mathbf{g} \mathbf{b} X_0 C - \mathbf{g} \mathbf{b} T(1-s)TK[(r+1)-2/s]/rs \\ - A'(N) a T \end{array} \right\} qTK[(r+1)-2/s] / Zras$$

Appendix 2

The numerical analysis

As mentioned in the main text, the ecological model is specified for the numerical analysis at the scale of one km² and one year. The same scale is also used for the agricultural benefit, as well as the hunting benefit, given in 1998/99 prices. The protected area, consisting of Serengeti National Park and its surrounding game reserves, covers an area of some 26 000 km² (TANAPA Planning Unit 1996). The outer area is considered as the catchment area surrounding the western edge of the protected area. Following Campbell and Hofer (1995), we assume that the catchment area, where the local people are originated, is located with a maximum distance of 45 km to the protected area. This area constitutes some 30 500 km² (Campbell and Hofer 1995). The human population in the outer area is estimated to be about 1.1 million with an average household size of about 7 persons (Campbell and Hofer 1995). Accordingly, there will be about 5 households per km² in the outer area. On average, 2 persons per household work in agricultural production and hunting. Hence, the effort constraint L yields $10 \cdot 365$ days a year, and we have $3650 = N + aT$.

The wildebeest migration is related to rainfall and the wildebeest population enters agricultural land while it grazes the northern grassland during the dry season. The length of the dry season is about four months (Barrett and Arcese 1998). Four months, or 122 days, is also assumed to be the length of the migration period, and hence, $T=122$ days. The wildebeest population is estimated to be about

1.3 million animals and the annual offtake to some 120 000 animals (Campbell and Hofer 1995). The wildlife density during the period of migration is therefore about 43 animals per km², while the total offtake is 4 animals per km². The number of hunters per household in the western Serengeti is assumed to be about 0.2 (Campbell and Hofer 1995). Consequently, it is one person involved in hunting at full time basis for every 5 households, and the baseline value of a is accordingly 1. By imposing these values for a and T into the survival rate together with H / X_0 as 4/43, the catchability coefficient is calculated from equation (4). The result is $q = 0.0008$. Following Barrett and Arcese (1998), the hunting value of a wildebeest is about US\$16, $P_H = 16$. The natural growth function is specified as logistic. To calibrate the wildlife stock to its base level the carrying capacity K is set to 59 animals per km² outer area, meaning that the ecosystem can carry a stock of wildebeest just below 1.8 million animals. The maximum specific growth rate is fixed as $r = 0.3$ (Caughley and Sinclair 1994).

The yield function is specified as $A(N) = mN^a$ with m as a productivity parameter and a as a scale parameter. The scale parameter is given as 0.8 (Barrett and Arcese 1998). According to a questionnaire among 300 households in Serengeti and Bunda Districts in June-September 2001 (Anne Borge Johannesen, work in progress), the average plot size per household is 7.4 acres, corresponding to a cultivated area of a fraction of 0.15 per km² for the average 5 households. For the same districts, the average value of the yield is estimated to be about US\$19 000 per km² cultivated land (Emerton and Mfunda 1999). At our scale of one km², this represents a value of $19.000 \cdot 0.15 = \text{US}\$2\ 850$ (or US\$570 per household). The main crops grown in the Western Serengeti are sorghum, cassava, maize, and cotton (SRCP 1998). Personal communication (SRCP 1999) indicates a per kg price of US\$ 0.18 for sorghum, US\$ 0.05 for cassava, US\$ 0.11 for maize, and US\$ 0.19 for cotton. By weighting the crop prices by the relative magnitude of these crops, we arrive at the price per kilo agricultural output equals US\$ 0.15. $P_A = 0.15$. Consequently, for the specified yield function we have $P_A mN^a = 0.15 \cdot m(3650 - 1 \cdot 122)^{0.8} = 2\ 850$, balancing with $m = 28$.

The wildebeest has a daily consumption of about 3% of their body weight in dry forage (Murray 1995), hence, $b = 0.03$. When assuming an average weight of 150 kg per animal, the daily

consumption of dry forage is therefore 4.5 kg per animal. To obtain a value for the fraction of the forage eaten from the crops g , we use the first order condition (13). For the above estimated parameter values together with the imposed values for a and X_0 , $g = 0.12$ balances this equation. Hence, through this calibrating we find that 12% of the wildebeests daily consumption of dry forage is from crops. Plugging into equation (8) this corresponds to a yearly damage on crop production of about 14%. This is the average of the estimates found in Emerton and Mfunda (1999) and Anne Borge Johannesen (work in progress). Table A1 summarises the baseline parameter values.

Table A1 about here

Figure3: The conservation-damage schedules.

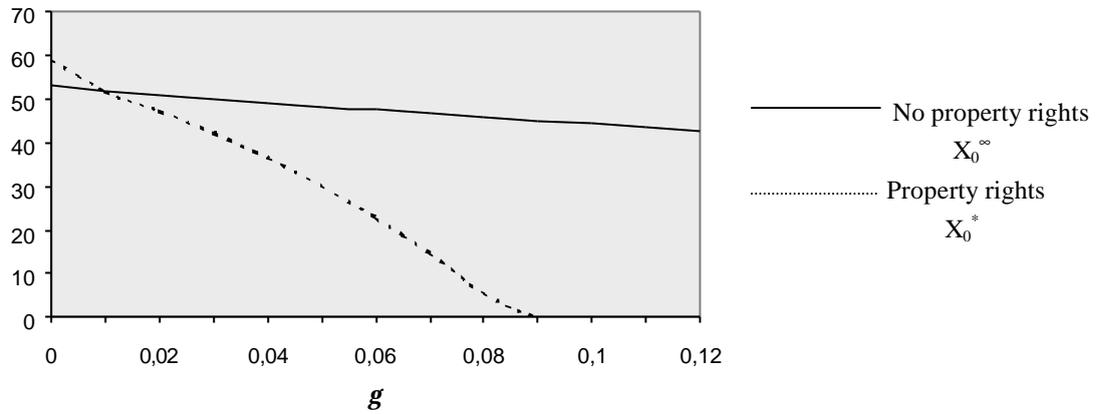


Table 1: Comparative static results.

Parameter	No property rights		Property rights	
	a^∞	X_0^∞	a^*	X_0^*
g	+	-	+	-
P_A	-	+	-/+	+/-
P_H	+	-	+/-	-/+
T	?	?	?	?
L	+	-	+	-
q	?	?	?	?
r	+	+	?	?
K	+	+	?	?

Note: When '+/-'; a reduction in a^* (-) is accompanied by a higher X_0^* (+), and vice versa. When '?'; ambiguous sign.

Table A1: Baseline values prices and costs (1998/99-prices), ecological parameters and other parameters

Parameter	Description	Value
P_H	Meat price	16 (\$/animal)
q	Catchability coefficient	0.0008 (1/day)
P_A	Crop price	0.15 (\$/kg)
a	Input elasticity labour crop production	0.8
m	Productivity crop production	28 (kg/day)
b	Fraction daily consumption of dry forage	0.03
g	Fraction dry forage consumption crop	0.12
L	Total available effort	3650 (day)
T	Length of migration period	122 (day)
K	Carrying capacity	59 (animal)
r	Intrinsic growth rate	0.3